NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT SECTION 7
BIOLOGICAL OPINION

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LIST OF ACRONYMS

AF – Anti-Fouling

AFC – Anti-Fouling Coating

AGRRA – Atlantic and Gulf Rapid Reef Assessment

AGSS – Research and Survey Type Submarine

ALC – Aquatic Life Criteria

ANS – Aquatic Nuisance Species

AoI – Area of Influence

ASMFC – Atlantic States Marine Fisheries Commission

ASSRT – Atlantic Sturgeon Species Recovery Team

ATSDR – Agency for Toxic Substances and Disease Registry

BE – Biological Evaluation

BRT – Biological Review Team

BTEX – Benzene, Toluene, Ethylbenzene, and Xylene

CDFW – California Department of Fish and Wildlife

C.F.R (or CFR) – Code of Federal Regulations

CITES – Convention on International Trade in Endangered Species of Wild Fauna and Flora

CPO – Chlorine-Produced Oxides

CPP – Controllable Pitch Propeller

CTET – Chronic Toxicity Effects Threshold

CV – Central Valley

CWA – Clean Water Act

DDT – Dichlorodiphenyltrichloroethane

DEET – N, N-diethyl-m-toluamide

DO – Dissolved Oxygen

DoD – Department of Defense

DPS – Distinct Population Segment

DFG – Department of Fish and Game

DQO – Data Quality Objective

ECOTOX – Ecotoxicology knowledgebase

EPA – U.S. Environmental Protection Agency

EPCRA – Emergency Planning and Community Right-to-Know Act

ERC – Environmental Research Consulting

ESA – Endangered Species Act

ESU – Evolutionarily Significant Unit

EEZ – Exclusive Economic Zone

FENA – Females Estimated to Nest Annually

FIFRA – Federal Insecticide, Fungicide, and Rodenticide Act

HCCC – Hood Canal Coordinating Council

HEM – Hexane Extractable Material

ICES – International Council for Exploration of the Seas

IMO – International Maritime Organization

INRMP – Integrated Natural Resources Management Plan

IPCC – Intergovernmental Panel on Climate Change

ISO – International Organization for Standardization

ITS – Incidental Take Statement

IUCN – International Union for Conservation of Nature

IWC – International Whaling Commission

LC50 – Concentration at which 50 Percent of Exposed Organisms Die

LOEC – Lowest Observed Effects Concentration

LSNFH – Livingston Stone National Fish Hatchery

MARPOL – Marine Pollution

MMC – Marine Mammal Commission

MMPA – Marine Mammal Protection Act

MOGAS – Motor Gasoline

MPCD – Marine Pollution Control Device

MSC – Military Sealift Command

MSD – Marine Sanitation Device

MSDS – Material Safety Data Sheet

MSPO – Maine State Planning Office

NAS – Non-indigenous Aquatic Species

NLAA – Not Likely to Adversely Affect

NMFS – National Marine Fisheries Service

NOAA – National Oceanic and Atmospheric Administration

NOEC – No Observed Effects Concentration

NRC – National Research Council

NUWC – Naval Undersea Warfare Center

NWFSC – Northwest Fishery Science Center

NWTT – Northwest Training and Testing

OWS – Oil and Water Separator

PAH – Polycyclic Aromatic Hydrocarbon

PBF – Physical and Biological Factors

PCB – Polychlorinated Biphenyl

PCE – Primary Constituent Element

PIC – Person in Charge

PIFSC – Pacific Islands Fishery Science Center

PNPTT – Point No Point Treaty Tribes

PSTT – Pacific Salmon Technical Team

RAA – Representative Action Area

RBDD – Red Bluff Diversion Dam

RC – Restoration Center (NOAA)

RCP – Representative Concentration Pathway

RPM – Reasonable and Prudent Measures

RQ – Risk Quotient

SE – Standard Error

SEFSC – Southeast Fishery Science Center

SGT-HEM – Silica Gel Hexane Extractable Material

SJRWMD – St. Johns River Water Management District

SPREP – Secretariat of the Pacific Regional Environment Programme

SSBN – Ballistic Missile Type Submarine

SSC Pacific – Space and Naval Warfare Systems Center Pacific

SSN – Attack Type Submarine

SSSRT – Shortnose Sturgeon Recovery Team

SURTASS – Surveillance Towed Array Sensor System

TBT – Tributyltin

TED – Turtle Excluder Device

TEWG – Technical Expert Working Group

TPH – Total Petroleum Hydrocarbon

TRT – Technical Recovery Team

TSCA – Toxic Substances Control Act

UME – Unusual Mortality Event

UNDS – Uniform National Discharge Standards

USASAC – United States Army Security Assistance Command

USCG – U.S. Coast Guard

USCOP – U.S. Commission on Ocean Policy

USFWS – U.S. Fish and Wildlife Service

USVI – U.S. Virgin Islands

WDFW – Washington Department of Fish and Game

WHO – World Health Organization

1. Introduction

The Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. 1531 et seq.), establishes a national program for conserving threatened and endangered species of fish, wildlife, plants, and the habitat on which they depend. Section 7(a)(2) of the ESA requires Federal agencies to insure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitat. Federal agencies must do so in consultation with National Marine Fisheries Service (NMFS) for threatened or endangered species (ESA-listed) or designated critical habitat that may be affected by the action that are under NMFS jurisdiction (50 C.F.R. §402.14(a)). If a Federal action agency determines that an action "may affect, but is not likely to adversely affect" endangered species, threatened species, or designated critical habitat and NMFS concurs with that determination for species under NMFS jurisdiction, consultation concludes informally (50 C.F.R. §402.14(b)).

Section 7(b)(3) of the ESA requires that at the conclusion of consultation, NMFS provides an opinion stating whether the Federal agency's action is likely to jeopardize ESA-listed species or destroy or adversely modify designated critical habitat. If NMFS determines that the action is likely to jeopardize listed species or destroy or adversely modify critical habitat, NMFS provides a reasonable and prudent alternative that allows the action to proceed in compliance with section 7(a)(2) of the ESA. If an incidental take is expected, section 7(b)(4) requires NMFS to provide an incidental take statement (ITS) that specifies the impact of any incidental taking and includes reasonable and prudent measures (RPMs) to minimize such impacts and terms and conditions to implement the RPMs.

Updates to the regulations governing interagency consultation (50 C.F.R. 402) are effective on October 28, 2019 (84 FR 44976). This consultation was pending at the time the regulations became effective and we are applying the updated regulations to the consultation. As the preamble to the final rule adopting the regulations noted, "[t]his final rule does not lower or raise the bar on section 7 consultations, and it does not alter what is required or analyzed during a consultation. Instead, it improves clarity and consistency, streamlines consultations, and codifies existing practice." We have reviewed the information and analyses relied upon to complete this biological opinion (Opinion) in light of the updated regulations and conclude the Opinion is fully consistent with the updated regulations.

The action agencies for this consultation are the U.S. Environmental Protection Agency (EPA) and the Department of Defense (DoD) represented by the U.S. Navy (Navy). The EPA and DoD propose to promulgate the Uniform National Discharge Standards (UNDS) for vessels of the armed forces – Phase II Batch Two rule under the authority of the Clean Water Act (CWA). The UNDS Phase II Batch Two rule will amend Title 40 of the Code of Federal Regulations (C.F.R.) Part 1700 to establish performance standards for 11 discharges that are incidental to the normal operation of vessels of the armed forces. The 11 discharges addressed by this rule are: catapult brake water tank and post-launch retraction exhaust, controllable pitch propeller (CPP) hydraulic fluid, deck runoff, firemain system discharge, graywater, hull coating leachate, motor gasoline (MOGAS) compensating discharge, sonar dome discharge, submarine bilgewater, surface vessel bilgewater/oil-water separator (OWS) effluent, and underwater ship husbandry.

This consultation, biological opinion, and incidental take statement, were completed in accordance with section 7 of the ESA (16 U.S.C. 1536), associated implementing regulations (50 C.F.R. §§402.01-402.16), and agency policy and guidance and was conducted by the NMFS Office of Protected Resources Endangered Species Act Interagency Cooperation Division (hereafter referred to as "we").

This document represents the opinion of the NMFS on the effects of these actions on blue, fin, humpback (Central America, Western North Pacific, Arabian Sea, Cape Verde Islands/Northwest Africa, and Mexico Distinct Population Segments [DPSs]), North Atlantic right, Southern right, North Pacific right, sei, bowhead, sperm, gray (Western North Pacific DPS), killer (Southern Resident DPS), Bryde's (Gulf of Mexico subspecies), false killer (Main Hawaiian Islands Insular DPS), and beluga (Cook Inlet DPS) whales; Maui's and South Island Hector's dolphins; ringed (Arctic DPS), Guadalupe fur, Hawaiian monk, bearded (Beringia DPS), and Mediterranean monk seals; Steller sea lion (Western DPS); green (North Atlantic, South Atlantic, East Pacific, Central North Pacific, Central West Pacific, Indian-West Pacific, Southwest Pacific, Central South Pacific, North Indian, Southwest Indian, and Mediterranean DPSs), hawksbill, Kemp's ridley, leatherback, loggerhead (Northwest Atlantic Ocean, South Atlantic Ocean, Southeast Indo-Pacific Ocean, Northeast Atlantic Ocean, North Pacific Ocean, South Pacific Ocean, North Indian Ocean, Southwest Indian Ocean, and Mediterranean Sea DPSs), and olive ridley (Mexico's Pacific coast breeding colonies and all other areas DPSs) sea turtles; dusky sea snake; shortnose, gulf, green (Southern DPS), Atlantic (Gulf of Maine, New York Bight, Chesapeake, Carolina, and South Atlantic DPSs), Sakhalin, Adriatic, European, and Chinese sturgeon; smalltooth (U.S. and non-U.S. portion of range DPSs), largetooth, narrow, dwarf, and green sawfish; scalloped hammerhead (Eastern Atlantic, Eastern Pacific, Southwest Atlantic, and Indo-West Pacific DPSs), oceanic whitetip, daggernose, striped smoothhound, narrownose smoothhound, and spiny, smoothback, sawback, Argentine, and common angel sharks; Brazilian, blackchin, and common guitarfish; Nassau, gulf, and island grouper; steelhead trout (Southern California, South-Central California Coast, Central California Coast, California Central Valley, Northern California, Lower Columbia River, Upper Willamette River, Middle Columbia River, Upper Columbia River, Snake River Basin, and Puget Sound DPSs); Atlantic (Gulf of Maine DPS); chinook salmon (Sacramento River Winter-Run, Central Valley Spring-Run, California Coastal, Upper Willamette River, Lower Columbia River, Upper Columbia River Spring-Run, Puget Sound, Snake River Fall-Run, and Snake River Spring/Summer-Run Evolutionarily Significant Units [ESUs]), coho (Central California, Lower Columbia River, Southern Oregon & Northern California Coasts, and Oregon Coast ESUs), chum (Columbia River and Hood Canal Summer-Run ESUs), and sockeye (Snake River and Ozette Lake ESUs) salmon; totoaba; bocaccio (Puget Sound/Georgia Basin DPS); yelloweye (Puget Sound/Georgia Basin DPS) rockfish; eulachon (Southern DPS); African coelacanth (Tanzanian DPS); elkhorn, staghorn, pillar, rough cactus, lobed star, mountainous star, boulder star, *Acropora globiceps*, *Acropora jacquelineae*, *Acropora lokani*, *Acropora pharaonis*, *Acropora retusa*, *Acropora rudis*, *Acropora speciosa*, *Acropora tenella*, *Anacropora spinosa*, *Eyphillia paradivisa*, *Isopora crateriformis*, *Montiplora australiensis*, *Pavona diffluens*, *Porites napopora*, *Seriatopora aculeata*, and *Cantharellus noumeae* corals; white and black abalone; and chambered nautilus.

A complete record of this consultation is on file at the NMFS Office of Protected Resources in Silver Spring, Maryland.

1.1 Background

As described in the Biological Evaluation (BE) prepared for this consultation (EPA and Navy 2018), section 325 of the National Defense Authorization Act of 1996, entitled “Discharges from Vessels of the Armed Forces”, amended the CWA part 312 to require the Administrator of the EPA (Administrator) and the Secretary of Defense of the DoD (Secretary) to develop uniform national standards to control certain discharges incidental to the normal operation of a vessel of the Armed Forces. UNDS is used to refer to the provisions in CWA §§ 312(a)(12)-(14) and (n) (33 U.S.C. 1322(a)(12)-(14) and (n)).

The UNDS rule is intended to enhance the operational flexibility of vessels of the armed forces domestically and internationally, stimulate the development of innovative vessel pollution control technology and practices, and advance the ability of the armed forces to better design and build environmentally sound vessels and manage discharges. Section 312(n)(3)(A) of the CWA requires the EPA and DoD to promulgate UNDS for certain discharges incidental to the normal operation of a vessel of the armed forces (CWA § 312(a)(12)), unless the Secretary finds that compliance with UNDS would not be in the national security interests of the United States (CWA § 312(n)(1)).

In the UNDS Phase I rule (64 FR 25126; 40 C.F.R Part 1700), EPA and DoD identified 14 discharges that would not require the use of a marine pollution control device (MPCD) and 25 discharges that would require the use of a MPCD. EPA and DoD decided to establish standards for the 25 discharges that will require a MPCD in three batches rather than two as originally proposed, as described in the UNDS Phase II rule (79 FR 6117). An informal ESA section 7 consultation with EPA and DoD was completed by NMFS for Phase II Batch One on December 9, 2016. The UNDS Phase II Batch One Rule, published January 11, 2017 at 82 FR 3173, addresses 11 routine discharges containing substances considered by NMFS to have relatively low toxicity, including aqueous film-forming foam, chain locker effluent, distillation and reverse osmosis brine, elevator pit effluent, gas turbine water wash, non-oily machinery wastewater, photographic laboratory drains, seawater cooling overboard discharge, seawater piping biofouling prevention, small boat engine wet exhaust, and welldeck discharges. The Phase II

Batch Two Rule, proposed October 7, 2016 at 81 FR 69753 is the subject of this consultation and addresses 11 other discharges containing substances considered by NMFS to have the potential to be more toxic. The Phase II Batch Three Rule will address the remaining three discharges related to ballast water: clean ballast, dirty ballast, and compensated fuel ballast. Rulemaking for the third batch will be subject to separate ESA consultation as necessary.

Discharges that do not fall within the scope of UNDS, and thus are not included in this consultation, include overboard discharges of rubbish, trash, garbage, or other such materials; sewage; air emissions from operation of a vessel propulsion system, motor-driven equipment, or incinerator; discharges that require a National Pollutant Discharge Elimination System permit; or discharges containing source, special nuclear, or byproduct materials regulated by the Atomic Energy Act (64 FR 25126; 40 C.F.R Part 1700). UNDS apply only to vessels of the armed forces while they are in U.S. navigable waters, including inland waters, the territorial sea, and the contiguous zone as defined in CWA Part 502. The seaward extent of the contiguous zone per CWA, and hence waters subject to UNDS, is 12 miles from the U.S. baseline, which is usually the low-water line along the U.S. coast (Convention on the Territorial Sea and the Contiguous Zone art. 24 1958). In an effort to better protect coastal waters, some UNDS performance standards direct discharge of pollutants seaward of waters subject to UNDS.

1.2 Consultation History

This opinion is based on information provided by the EPA and DoD, including the Biological Evaluation, correspondence and discussions with the action agencies, and other sources of information. Our communication with the EPA and DoD regarding this consultation is summarized as follows:

- **July 13, 2017:** Navy (as DoD representative) and EPA sponsored a pre-consultation kick-off meeting with NMFS and the U.S. Fish and Wildlife Service (USFWS) to present an overview of the action.
- **September 1, 2017:** Received draft UNDS Batch Two BE Chapters 1 to 4 for review and comment from the Navy via email.
- **September 30, 2017:** Sent NMFS comments on the draft BE chapters to Navy and EPA via email.
- **January 11, 2018:** Received responses to our comments via email and additional information for appendices of draft BE for review.
- **February 6, 2018:** Sent NMFS comments to Navy and EPA regarding the draft BE appendices and the response to our previous comments via email.
- **April 10, 2018:** Received additional draft BE sections via email from Navy for review and comment.

- **May 11, 2018:** Sent NMFS comments on the additional draft BE sections to Navy and EPA via email.
- **August 22, 2018:** Received responses to our comments on the sections of the draft BE via email from Navy.
- **August 28, 2018:** Participated in a meeting with EPA, Navy, their contractor (assisting with the preparation of the BE), and USFWS to discuss the Services' comments and the responses prepared by the Navy and EPA to address outstanding concerns, and the revised schedule for the ESA section 7 consultations with the Services and the final rulemaking for UNDS Phase II Batch Two.
- **October 16, 2018:** Received complete draft BE via email from Navy for review and comment.
- **October 24, 2018:** Sent NMFS comments on complete draft BE via email to Navy and EPA via email.
- **November 16, 2018:** Received consultation initiation request from Navy via email. The Navy is requesting an informal consultation.
- **November 26, 2018:** Received hard copy of consultation initiation request from Navy.
- **December 3, 2018:** Sent the Navy our consultation initiation letter with copy to the EPA notifying them that we are initiating formal consultation due to the nature and scope of the discharges regulated under the Phase II Batch Two UNDS rule. The consultation initiation date is November 16, 2018.
- **January 28, 2019:** Consultation resumed on this day after being held in abeyance for 38 days due to a lapse in appropriations that resulted in a partial government shutdown.
- **April 9, 2019:** The draft Opinion was sent to Navy and EPA on this day for review and comment.
- **May 3, 2019:** The Navy sent an Excel file with the comments from them, EPA, and U.S. Coast Guard (USCG) on the Opinion as well as a document detailing points they wanted to discuss with NMFS regarding the content of the Opinion and their comments.
- **May 8, 2019:** NMFS, EPA, Navy and USCG met to discuss their comments on the Opinion and next steps. During the meeting, we discussed the possibility of their writing a 7(d) letter in order to proceed with publishing the rule while we are concluding the consultation. NMFS sent example 7(d) letters to EPA and Navy for their information the same day via email. During the meeting NMFS also recommended that we move to a programmatic consultation given the nature of the proposed action (rule-making).

- **May 17, 2019:** The Navy sent a summary of the discussion points from the in-person meeting via email for review and comment.
- **May 22, 2019:** NMFS edited the discussion point summary and returned the document to the Navy via email.
- **May 30, 2019:** The Navy sent a final version of the meeting summary with our changes accepted.
- **June 20, 2019:** The Navy informed NMFS via email that they and the EPA do not want the consultation to be programmatic and requested that we proceed in revising our Opinion in order to provide them with a revised draft during the summer.
- **June 21, 2019:** NMFS sent an email to the Navy noting that we will continue working on revisions to the draft but that we need to have further discussions in order to address their concerns regarding their ability to achieve the monitoring objectives in the ITS as written.
- **July 12, 2019:** NMFS, Navy, and EPA had a conference call to discuss the consultation, including NMFS rolling back changes to the Opinion to reflect that the Navy and EPA decided they don't want to do a programmatic consultation and changes to the ITS. NMFS emailed more information to the Navy and EPA regarding programmatic consultations.
- **July 19, 2019:** NMFS, after getting clearance from the attorney working on the consultation, sent the comment matrix the Navy, EPA and USCG had shared in May 2019 with our responses to their comments and the revised ITS language NMFS developed based on conversations with the Navy and EPA.
- **July 25, 2019:** The Navy sent an email acknowledging receipt of the comment matrix and ITS and clarifying information regarding the Representative Action Areas (RAAs) and how contaminant concentrations were modeled in these areas in response to some of the comments from NMFS in the comment matrix.
- **August 12, 2019:** The Navy sent an email with their responses to our comments on the comment matrix as well as an additional justification of why they believe the effects of underwater ship husbandry and hull coating leachate (the two discharges we found to be likely to adversely affect ESA-listed species and their designated critical habitat in our Opinion) are not likely to adversely affect ESA resources.
- **August 16, 2019:** NMFS sent an email response to the Navy regarding species and adverse effects indicating that we continue to believe there are likely to be adverse affects to some species from two of the Batch Two discharges.

- **August 30, 2019:** The Navy sent an email with two versions of the revised ITS language, a marked up version showing the changes they made to the document and another clean version that incorporated all their changes.
- **September 25, 2019:** NMFS sent an email reply to the Navy and EPA detailing our concerns related to the ITS revisions suggested by the Navy and EPA.
- **October 8, 2019:** NMFS sent the revised draft of the Opinion with the Navy and EPA.
- **October 24, 2019:** The Navy sent a NMFS document with comments still requiring discussion related mainly to the ITS.
- **November 1, 2019:** NMFS, Navy, and EPA held a conference call to discuss the remaining concerns regarding the ITS in order to conclude the drafting of the Opinion.
- **November 7, 2019:** The Navy and EPA sent NMFS the calculations for determining the area of influence (AoI) and calculating mass loading for the ITS.
- **November 12 and 13, 2019:** NMFS sent the Navy and EPA the revised version of the ITS that incorporated the language provided by the Navy and EPA. The Navy sent an email to NMFS approving the revised language with minor edits.

2. The Assessment Framework

Section 7(a)(2) of the ESA requires Federal agencies, in consultation with NMFS, to ensure that their actions are not likely to jeopardize the continued existence of endangered or threatened species; or adversely modify or destroy their designated critical habitat.

"Jeopardize the continued existence of" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of an ESA-listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 C.F.R. §402.02).

"Destruction or adverse modification" means a direct or indirect alteration that appreciably diminishes the value of designated critical habitat for the conservation of an ESA-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 (50 C.F.R. §402.02).

An ESA section 7 assessment involves the following steps:

Description of the Action (Section 3): In the case of this consultation, we provide a general description of the 11 UNDS discharges that are the subject of the Batch Two rule and actions expected to be implemented in the future as part of the implementation of the requirements under the rule.

Action Area (Section 4): We define the action area based on the spatial extent of potential effects or stressors from the action.

Stressors Associated with the Action (Section 5): We discuss the potential stressors we expect to result from the action.

Status of Endangered Species Act Protected Resources (Section 6): We identify the ESA-listed species and designated critical habitat that are likely to co-occur with those stressors in space and time and evaluate the status of those species and habitat. In this section, we also identify those *Species and Designated Critical Habitat Not Likely to be Adversely Affected* (Section 6.1), and those *Species and Designated Critical Habitat Likely to be Adversely Affected* (Section 6.2).

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

Effects of the Action (Section 8): In Section 8.1, we discuss the uncertainties associated with the exposure and responses analyses and our concerns related to the methods used in the BE to calculate exposure and response. Section 8.2 contains a description of mitigation measures to reduce exposure of ESA-listed species and designated critical habitat to stressors. In Section 8.3 (*Exposure and Response Analysis*), we identify the number, age (or life stage), and gender of ESA-listed individuals that are likely to be exposed to the stressors and the populations or subpopulations to which those individuals belong. We also consider whether the action "may affect" designated critical habitat. We evaluate the available evidence to determine how individuals of those ESA-listed species are likely to respond given their probable exposure to stressors and consider how the action may affect designated critical habitat, which is our exposure analysis. In Section 8.4 (*Risk Analysis*), we assess the consequences of these responses of individuals that are likely to be exposed to the populations those individuals represent, and the species those populations comprise. The risk analysis also considers the impacts of the action on the essential habitat features and conservation value of designated critical habitat.

Cumulative Effects (Section 9): Cumulative effects are the effects to ESA-listed species and designated critical habitat of future state or private activities that are reasonably certain to occur within the action area (50 C.F.R. §402.02). Effects from future Federal actions that are unrelated to the action are not considered because they require separate ESA section 7 compliance.

Integration and Synthesis (Section 10): In this section, we integrate the analyses of *Effects of the Action* (Section 8), the *Environmental Baseline* (Section 7), and the *Cumulative Effects* (Section 9) to formulate the agency's opinion as to whether the action is likely to appreciably reduce the likelihood of survival and recovery of an ESA-listed species in the wild or reduce the conservation value of designated critical habitat.

Conclusion (Section 11); With full consideration of the status of the species and the designated critical habitat, we consider the effects of the action within the action area on populations or subpopulations and on essential habitat features when added to the environmental baseline and the cumulative effects to determine whether the action could reasonably be expected to:

- Reduce appreciably the likelihood of survival and recovery of ESA-listed species in the wild by reducing its numbers, reproduction, or distribution, and state our conclusion as to whether the action is likely to jeopardize the continued existence of such species; or
- Appreciably diminish the value of designated critical habitat for the conservation of an ESA-listed species, and state our conclusion as to whether the action is likely to destroy or adversely modify designated critical habitat.

If, in completing the last step in the analysis, we determine that the action under consultation is likely to jeopardize the continued existence of ESA-listed species or destroy or adversely modify designated critical habitat, then we must identify reasonable and prudent alternative(s) to the action, if any, or indicate that to the best of our knowledge there are no reasonable and prudent alternatives. See 50 C.F.R. §402.14(h)(3).

In addition, if take is reasonably certain to occur, we include an *Incidental Take Statement* (Section 12) that specifies the impact of the take, reasonable and prudent measures to minimize the impact of the take, and terms and conditions to implement the reasonable and prudent measures. ESA section 7(b)(4); 50 C.F.R. §402.14(i). We also provide discretionary *Conservation Recommendations* (Section 13) that may be implemented by the action agency; 50 C.F.R. §402.14(j). Finally, we identify the circumstances in which *Reinitiation of Consultation* (Section 14) is required (50 C.F.R. §402.16).

To comply with our obligation to use the best scientific and commercial data available, we collected information identified through searches of Google Scholar, Web of Science, peer reviewed articles and their literature cited sections, species listing documentation, and reports published by government and private entities. Searches were used to identify information relevant to the potential stressors associated with the 11 discharges considered in UNDS Phase II Batch Two and responses of ESA-listed species and designated critical habitat. This opinion is based on our review and analysis of various information sources, including:

- Information submitted by EPA and DoD
- Government reports (including previous NMFS consultation documents, status reviews, and recovery plans)

- Peer-reviewed scientific literature
- The EPA Ecotoxicology knowledgebase (ECOTOX)

These resources were used to identify information relevant to the potential stressors and responses of ESA-listed species and designated critical habitat under NMFS jurisdiction that may be affected by the action to draw conclusions on risks the action may pose to the continued existence of these species and the value of designated critical habitat for the conservation of ESA-listed species.

3. Description of the Action

"Action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by federal agencies in the United States or upon the high seas. Examples include, but are not limited to:

(a) actions intended to conserve listed species or their habitat;

(b) the promulgation of regulations;

(c) the granting of licenses, contracts, leases, easements, rights-of-way, permits, or grants-in-aid; or

(d) actions directly or indirectly causing modifications to the land, water, or air.

The EPA and DoD propose the promulgation of the UNDS Phase II Batch Two Rule for 11 discharges from military vessels operating in waters of the U.S. that will require a MPCD. In promulgating Phase II performance standards, CWA §312(n)(2)(B) directs the EPA and DoD to consider seven factors: the nature of the discharge; the environmental effects of the discharge; the practicability of using the MPCD; the effect that installation or use of the MPCD would have on the operation or the operational capability of the vessel; applicable U.S. law; applicable international standards; and the economic costs of installation and use of the MPCD. Section 312(n)(3)(C) of the CWA further provides that the EPA and DoD may establish performance standards that (1) distinguish among classes, types, and sizes of vessels; (2) distinguish between new and existing vessels; and (3) provide for a waiver of applicability of standards as necessary or appropriate to a particular class, type, age, or size of vessel.

3.1 Vessel Types

Vessels of the armed forces affected by the rule total nearly 6,400, distributed among the Navy, Military Sealift Command (MSC), USCG, Army, Marine Corps, and Air Force. Armed forces vessel types include aircraft carriers, amphibious support ships, auxiliary ships, boats, patrol ships, service craft, submarines, and surface combatants. Most vessels have the capacity to hold or collect containable discharges such as graywater and bilgewater and hold them for onshore disposal.

Aircraft carriers are the largest vessels of the armed forces designed primarily for conducting combat operations by fixed wing aircraft launched with catapults. All vessels in this group are nuclear powered. Aircraft carriers exceed 1,000 feet (ft) in length and have crews of 4,000 to 6,000. The armed forces have 11 aircraft carriers.

Amphibious support ships are large vessels, ranging in length from 569 to 847 ft with a crew size of 300 to 500. Many of these vessels have large clean ballast tanks used to lower and raise the hull during amphibious operations, and welldecks to support recovery of landing craft and amphibious vehicles. The armed forces have 37 amphibious support ships meaning these vessels comprise less than one percent of the fleet.

Auxiliary ships are a large and diverse group of self-propelled vessels with lengths greater than or equal to 79 ft. These ships are designed to provide general support to combatant forces or shore-based establishments, including transporting supplies and troops to and from the theater of operations, executing mine countermeasure operations, conducting research, maintaining navigation systems, and recovering targets and drones. Crew sizes range from ten to 200 people for this vessel class. The armed forces have 368 auxiliary ships comprising six percent of the fleet.

Boats include all self-propelled vessels less than 79 ft in length used for things like security, combat operations, rescue, and training. These vessels have small crews of one to 19 and produce limited liquid discharges. The armed forces have 5,132 boats meaning these vessels comprise 81 percent of the fleet.

Patrol ships are self-propelled vessels with lengths greater than or equal to 79 ft designed to conduct patrol duties (i.e., maritime homeland security, law enforcement, and national defense missions). Vessels in this group have crew sizes ranging from ten to 200. The armed forces have 203 patrol ships comprising three percent of the fleet.

Service craft are a diverse group of non-self-propelled vessel classes designed to provide general support to other vessels in the armed forces fleet or shore-based establishments. Vessel classes in this group have an average length of 155 ft with more than 95 percent being between 40 and 310 ft. While most have a limited crew or no crew, barracks craft can provide sleeping accommodations for 100 to 1,200 crew members. These vessels include multiple barges, dredges, floating dry docks, floating cranes, floating causeway ferries, floating roll-on-off discharge facilities, dry deck shelters, floating workshops, and floating barracks. The armed forces have 355 service craft with a length of 79 ft or more, comprising six percent of the fleet, and 12 service craft less than 79 ft in length, which represent less than one percent of the fleet.

Submarines are submersible, nuclear-powered combat vessels that can fulfill combatant, auxiliary, or research and development roles. Submarines provide strategic missile, battlefield support, stealth strike, special forces, littoral warfare, and other miscellaneous capabilities. Submarines are categorized as attack (SSN), ballistic missile (SSBN), and research and survey

(AGSS) types. Navy submarines range from the 165-ft-long AGSS to the 560-ft-long SSBN. The armed forces have 72 submarines, comprising one percent of the fleet.

Surface combatants are surface ships designed primarily to engage in attacks against airborne, surface, sub-surface, and shore targets. Vessel classes in this group range from 378 to 567 ft in length and have crew sizes ranging from 40 for the Littoral Combat Ship to almost 400 for a Guided Missile Destroyer or Cruiser. The armed forces have 115 surface combatants, comprising two percent of the fleet.

The geographic distribution and sizes of all vessels of the armed forces in U.S. waters are summarized in Table 1. There are also a limited number of armed forces vessels overseas. The EPA and DoD considered vessel class, type, and size when developing the discharge performance standards because not all vessels generate the same discharges.

Table 1. Total Number of Vessels of the Armed Forces at Locations with Ten or More Vessels of Different Sizes and Total Number of All Vessels in States and Territories with Homeports and Totals in Foreign Waters (from USEPA and Navy 2018)

Homeport Areas	Less than 79 ft	Greater than or Equal to 79 ft	Grand Total
Pacific Islands			
Guam	**45**	**11**	**57**
Apra Harbor	45	7	53
Hawai'i	**175**	**114**	**289**
Pearl Harbor	116	101	217
Honolulu	49	9	58
Kauai	10	4	14
West Coast			
Alaska	**67**	**18**	**85**
Auke Bay/Juneau	36	8	44
Cook Inlet	21	6	27
Prince William Sound	10	4	14
Washington	**264**	**119**	**363**
Puget Sound	223	114	337
Port Angeles	11	5	16
Long Beach	10		10

Oregon	**44**	**5**	**49**
Coos bay	11	1	12
Astoria	8	3	11
California	**931**	**166**	**1,097**
San Diego	651	140	791
Oceanside	107		107
San Francisco	81	17	98
Ventura	50	6	56
Long Beach	33	1	34
Newport Beach	9	2	11
East Coast			
Maine	**49**	**9**	**58**
Portland	17	3	20
Rockland	18	2	20
New Hampshire	**23**	**15**	**28**
Portsmouth	23	15	28
Massachusetts	**76**	**14**	**90**
Boston	42	5	47
Cape Cod	23	7	30
Rhode Island	**27**	**7**	**34**
Newport	23	7	30
Connecticut	**161**	**22**	**183**
New London/Groton	149	22	171
New Haven	12		12
New York	**89**	**2**	**91**
Long Island	27	1	28
Staten Island	19		19
St. Lawrence River/Lake Ontario	13		13
Lake Erie/Buffalo	10		10
New Jersey	**76**	**13**	**89**

Long Beach Island	20	4	24
Cape May	18	3	21
Bayonne	14	4	18
Sandy Hook	9	2	11
Washington, DC	**39**	**2**	**41**
Maryland	**176**	**40**	**216**
Baltimore/Annapolis	130	38	168
Lexington Park	37	2	39
Virginia	**850**	**267**	**1,117**
Norfolk	707	267	974
Yorktown	73		73
Dahlgren	26		26
Alexandria	25		25
Cape Charles	12		12
North Carolina	**271**	**11**	**282**
Jacksonville	200		200
Atlantic Beach	36	9	45
Wilmington	14	2	16
Outer Banks	13		13
South Carolina	**42**	**7**	**49**
Charleston	28	7	35
Georgia	**167**	**11**	**178**
Brunswick	90	10	100
Albany	63		65
Atlanta	14	1	15
Florida, Atlantic Coast	**211**	**58**	**269**
Jacksonville	91	29	120
Florida Keys	44	14	58
Miami/Miami Beach	27	9	36
Cape Canaveral	20	3	23
Ft Lauderdale	11	1	12

Port St. Lucie	8	2	10
U.S. Caribbean			
San Juan, Puerto Rico	**21**	**10**	**31**
Florida, Gulf Coast			
Florida, Gulf Coast	**169**	**20**	**189**
Panama City	78	9	87
Tampa	26	8	34
Pensacola	26	1	27
Destin	14		14
Clearwater	11		11
Alabama	**35**	**4**	**39**
Mobile Bay	30	4	34
Mississippi	**58**	**6**	**64**
Gulfport	49	3	52
Louisiana	**63**	**3**	**66**
New Orleans	35	1	36
Morgan City	11		11
Texas	**111**	**11**	**122**
Galveston	28	5	33
Corpus Christi	26	3	29
Beaumont	25	2	27
San Antonio	12		12
Houston	10		10
South Padre Island	9	1	10
Inland Waterways			
Bayview, Idaho	**22**		**22**
Illinois	**30**	**1**	**31**
Chicago	23		23
Kentucky	**21**	**2**	**23**
Louisville	10		10
Michigan	**96**	**6**	**102**

Lake Huron	31	3	34
Lake Michigan	24		24
Upper Peninsula/Lake Superior	19	2	21
Detroit	18	1	19
Minnesota	**15**	**1**	**16**
Duluth	11	1	12
St. Louis, Missouri	**20**	**1**	**21**
Ohio	**43**	**1**	**44**
Marblehead	18		18
Cleveland	16	1	17
Pennsylvania	**40**	**14**	**54**
Philadelphia	29	13	43
Tennessee	**22**	**4**	**26**
Wisconsin	**31**	**1**	**32**
Milwaukee	18		18
Grand Totals, U.S. Waters	**4,551**	**996**	**5,547**

3.2 Discharges Covered Under UNDS Phase II Batch Two and Their Performance Standards

The following subsections provide details of the 11 discharges that are the subject of the UNDS Phase II Batch Two Rule. If two or more regulated discharge streams are combined prior to discharge, then the resulting discharge would need to meet the discharge performance standards applicable to each of the discharges that are being combined (40 CFR 1700.40).

Notwithstanding each of the MPCD performance standards, a vessel of the armed forces is authorized to discharge into waters subject to UNDS, when the person in charge (PIC) or their designated representative determines such discharge is necessary to: prevent loss of life, personal injury, vessel endangerment, or severe damage to the vessel (e.g., discharge to avoid flooding of compartments, sinking, or to extinguish onboard fires).

Some UNDS Phase II Batch Two discharges would occur seaward of waters subject to UNDS. Specifically, the frequency of surface vessel bilgewater/OWS effluent, submarine bilgewater, and graywater discharges may increase farther from shore according to information from the Navy. Because UNDS performance standards would apply only to vessels of the armed forces within waters subject to UNDS and because some of the UNDS Batch Two discharges are limited or prohibited in certain nearshore waters, an increase in the frequency of some Batch Two discharges beyond 12 miles from shore could result from the implementation of performance standards in waters subject to UNDS because vessels would relocate seaward of these waters in order to discharge prohibited pollutants. However, this increase in volume discharged to surface waters more than 12 miles from shore would occur only in cases where onshore disposal in port is not possible because the port does not contain shoreline disposal facilities.

3.2.1 Catapult Brake Water Tank and Post-Launch Retraction Exhaust

Catapult water brake tank and post-launch retraction exhaust is the oily water skimmed from the tank for the water brake used to stop the forward motion of an aircraft carrier catapult piston when launching aircraft, and the condensed steam discharged when the catapult is retracted. While testing a catapult water brake does not generate a discharge, the oily water from the catapult water brake tank is discharged above the waterline after flight operations. Testing and catapult flight operations both generate the post-launch retraction exhaust discharge.

Most flight operations occur outside of waters subject to UNDS. In waters subject to UNDS, the catapult water brake is primarily used for testing catapults on recently constructed aircraft carriers, following major drydock overhauls, or after major catapult modifications.

Only Navy aircraft carriers, which total 11 vessels and represent less than one percent of vessels of the armed forces, are likely to produce catapult water brake tank and post-launch retraction exhaust discharge.

3.2.1.1 Discharge Constituents

The catapult water brake tank and post-launch retraction exhaust discharges contain lubricating oil and small amounts of metals generated within the catapult system itself. Additionally, the post-launch retraction exhaust discharge contains oil and water (in the condensed steam), nitrogen (in the form of ammonia, nitrates and nitrites, and total nitrogen), and metals such as copper and nickel from the piping systems. Among the constituents, oil, copper, lead, nickel, nitrogen, ammonia, bis(2-ethylhexyl) phthalate, phosphorus, and benzidine could be present in concentrations that exceed the EPA recommended water quality criteria.

3.2.1.2 Performance Standard

The EPA and DoD propose to prohibit the discharge of catapult water brake tank effluent and to minimize post launch retraction exhaust discharges by limiting the number of launches required to test and validate the system and conduct qualification and operational training.

3.2.2 Controllable Pitch Propeller (CPP) Hydraulic Fluid

The CPP hydraulic fluid is a high-pressure hydraulic oil that is used throughout the CPP system of pumps, pistons, crossheads, and crank rings. The discharge originates from propeller seals during normal operation to control a vessel's speed or direction or during routine maintenance or replacement of the propellers.

Leakage through CPP seals is most likely to occur while the vessel is underway because the CPP system operates under higher pressure when underway than at pierside or at anchor. CPP seals are designed to last five to seven years, which is the longest period between scheduled dry-dock cycles, and are inspected quarterly for damage or excessive wear. Because of the hub design and frequent CPP seal inspections, leaks of hydraulic fluid from CPP hubs are expected to be negligible. Discharge of hydraulic fluid into surrounding waters may result from CPP blade maintenance or replacement when dry-docking is unavailable or impractical.

USCG patrol ships, Navy surface combatants, some amphibious support ships, and some MSC auxiliary ships might produce this discharge. Those ships that are greater than or equal to 79 ft total 203, 115, 37, and 368, respectively and represent approximately 11 percent of the vessels of the armed forces (based on information in Table 3-1 of the BE, EPA and Navy 2018 (USEPA and Navy 2018)).

3.2.2.1 Discharge Constituents

The CPP discharge includes paraffins, olefins, and metals such as copper, aluminum, tin, nickel, and lead. The hydraulic fluid released during underwater CPP maintenance could cause a sheen in the receiving waters. Metal concentrations are expected to be insignificant because hydraulic fluid is not corrosive to metal piping, and the hydraulic fluid is continually filtered to protect against system failures.

3.2.2.2 Performance Standard

The EPA and DoD propose to require that the protective seals on CPPs be maintained in good operating order to minimize the leakage of hydraulic fluid. To the greatest extent practicable, maintenance activities on CPPs should be conducted when a vessel is in drydock. If maintenance and repair activities must occur when the vessel is not in drydock, appropriate spill response equipment (e.g., oil booms) must be used to contain and clean any oil leakage. The discharge of CPP hydraulic fluid must not contain oil in quantities that: cause a film or sheen upon or discoloration of the surface of the water or adjoining shorelines; cause a sludge or emulsion to be deposited beneath the surface of the water or upon adjoining shorelines; or contain an oil content above 15 parts per million (ppm)[1] as measured by EPA Method 1664a or other appropriate method for determination of oil content as accepted by the International Maritime Organization

[1] This standard has been promulgated in regulations and permits since it was established by the IMO in Marine Pollution (MARPOL) 73/78, Annex I, which contains oil discharge criteria established in 1969 by amendments to the Oil Pollution Convention. The 15 ppm standard is based on the performance of bilge oil-water separators, what bilge alarms are able to detect, and the approximate concentration at which there is a visible sheen.

(IMO) (e.g., ISO Method 9377) or USCG; or otherwise are harmful to the public health or welfare of the U.S..

3.2.3 Deck Runoff

Deck runoff includes the precipitation, washdowns, and seawater spray falling on the weather or flight deck of a vessel and discharged overboard through deck openings. Deck runoff contains any residues that may be present on the deck surface originating from topside equipment components, shipboard activities, storage of material, maintenance activities, and the decking material itself.

All vessels of the armed forces generate deck runoff and the discharge occurs whenever the deck surface is exposed to water. Only vessels of the armed forces that support flight operations have flight decks. The standards distinguish between flight decks and other vessel decks.

3.2.3.1 Discharge Constituents

Constituents and volumes of deck runoff vary widely depending on the purpose, service, and practices of the vessel. This includes oil and grease, petroleum hydrocarbons, surfactants, soaps and detergents, glycols, solvents, and metals. These constituents may be present in concentrations that could potentially contribute to an exceedance of the EPA recommended water quality criteria.

3.2.3.2 Performance Standard

The EPA and DoD propose to require that vessels prohibit flight deck washdowns and minimize deck washdowns while in port and in federally-protected-waters. Additionally, before deck washdowns occur, exposed decks must be broom cleaned and on-deck debris, garbage, paint chips, residues, and spills must be removed, collected, and disposed of onshore in accordance with any applicable solid waste or hazardous waste management and disposal requirements. If a deck washdown or above water line hull cleaning would create a discharge, the washdown or above water line cleaning must be conducted with minimally-toxic and phosphate free soaps, cleaners, and detergents. The use of soaps that are labeled as toxic is prohibited. Specifically, according to 40 CFR 1700.s, “Minimally-toxic soaps, cleaners, and detergents typically contain little to no nonylphenols.”[2] All soaps and cleaners must be used as directed by the label. Furthermore, soaps, cleaners, and detergents should not be caustic and must be biodegradable. Where feasible, machinery on deck must have coamings or drip pans where necessary to collect

[2] Products containing nonylphenol and precursor nonylphenol ethoxylates are expected to be labeled as toxic because nonylphenol and precursor nonylphenol ethoxylates are on the list of toxic substances subject to reporting under Toxic Substances Control Act (TSCA) and Emergency Planning and Community Right-to-Know Act (EPCRA), and Material Safety Data Sheets (MSDS) sheets for detergents containing nonylphenol and nonylphenol are expected to identify the product as toxic to aquatic life.

any oily discharge that may leak from machinery and prevent spills. The drip pans must be drained to a waste container for proper disposal onshore in accordance with any applicable oil and hazardous substance management and disposal requirements. The presence of floating solids, visible foam, halogenated phenol compounds, and dispersants and surfactants in deck washdowns must be minimized. Topside surfaces and other above-water-line portions of the vessel must be well-maintained to minimize the discharge of rust and other corrosion by-products, cleaning compounds, paint chips, non-skid material fragments, and other materials associated with exterior topside surface preservation. Residual paint droplets entering the water must be minimized when conducting maintenance painting. The discharge of unused paint is prohibited. Paint chips and unused paint residues must be collected and disposed of onshore in accordance with applicable solid waste and hazardous substance management and disposal requirements. When vessels conduct underway fuel replenishment, scuppers must be plugged to prevent the discharge of oil. Any oil spilled must be cleaned, managed, and disposed of onshore in accordance with any applicable onshore oil and hazardous substance management and disposal requirements.

3.2.4 Firemain System Discharge

The firemain system discharge is the seawater pumped through the firemain system for firemain testing, maintenance, and training, and to supply water for the operation of certain vessel systems (i.e., secondary uses). The water passed through the firemain system is drawn from the sea and returned to the sea by either discharge over the side from fire hoses or through submerged pipe outlets.

Most vessels of the armed forces greater than or equal to 79 ft in length are expected to discharge from firemain systems. Most boats and service craft that are less than 79 ft in length do not generate firemain systems discharge because smaller boats and craft typically use portable fire pumps or fire extinguishers. Approximately 20 percent of vessels of the armed forces produce firemain systems discharge.

Firemain systems are essential to the safety of a vessel and crew and therefore, require testing and maintenance. The firemain system includes all components between the fire pump suction sea chest and the cutout valves to the various services including sea chests, fire pumps, valves, piping, fire hoses, seawater sprinkling systems, foam proportioning stations, and heat exchangers. Any foam discharges associated with firemain systems are not covered under this performance standard but would need to meet the requirements of 40 CFR 1700.14 (aqueous film-forming foam). The secondary uses of wet firemain systems may include deck washdowns, cooling water for auxiliary machinery, eductors, ship stabilization and ballast tank filling, and flushing for urinals, commodes, firemain loop recirculation, and pulpers.

3.2.4.1 Discharge Constituents

The seawater discharged overboard from the firemain system can contain entrained or dissolved materials, principally metals, from natural degradation of the internal components of the firemain system itself. Some traces of oil or other lubricants may also enter the seawater from valves or pumps. The presence of copper, zinc, nickel, aluminum, tin, silver, iron, titanium, and chromium. in firemain discharges can be traced to the corrosion and erosion of the firemain piping system, valves, or pumps.

3.2.4.2 Performance Standard

The EPA and DoD propose to require that to the greatest extent practicable, firemain system maintenance and training be conducted outside of port and as far away from shore as possible. In addition, firemain systems must not be discharged in federally protected waters except when needed to comply with anchor washdown requirements in Subpart 1700.16 (Chain locker effluent). Firemain systems may be used for secondary uses if the intake comes directly from the surrounding waters or potable water supplies. If the firemain system is used for a secondary use and a performance standard does not exist for that secondary use, then the performance standard for the firemain system applies.

3.2.5 Graywater

Graywater includes galley, bath and shower water, as well as wastewater from lavatory sinks, laundry, interior deck drains, water fountains, and shop sinks. Approximately 20 percent of the vessels of the armed forces (i.e., aircraft carriers, surface combatants, amphibious support ships, submarines, patrol ships, and some auxiliary ships, boats, and service craft) generate graywater.

Vessels of the armed forces have different methods for collecting and discharging graywater. Most vessels are designed to direct graywater to the vessel's sewage tanks while pierside for transfer to a shore-based treatment facility. These vessels are not generally designed to hold graywater for extended periods and must drain or pump their graywater overboard while operating away from the pier in order to preserve holding capacity for sewage tanks. Some vessels with either larger graywater holding capacity or USCG-certified marine sanitation devices (MSDs) have the capacity to hold or treat graywater for longer periods.

3.2.5.1 Discharge Constituents

Graywater discharges may contain soaps and detergents; oil and grease from foods; food residue; nutrients and oxygen demand from food residues and detergents; hair; bleach and other cleaners and disinfectants; pathogens; and a variety of additional personal care products such as moisturizer, deodorant, perfume, and cosmetics. Graywater discharge could negatively impact receiving waters due to the possible presence of bacteria, pathogens, oil and grease, detergent and soap residue, metals (e.g., cadmium, chromium, lead, copper, zinc, silver, nickel, and mercury), solids, and nutrients (e.g., phosphates from the detergents). Of these constituents, the EPA and DoD have found graywater ammonia, copper, lead, mercury, nickel, silver, and zinc at concentrations that may exceed the EPA recommended water quality criteria.

3.2.5.2 Performance Standard

The EPA and DoD propose to require that large quantities of cooking oils (e.g., from deep fat fryers), including animal fats and vegetable oils, must not be added to graywater systems. The EPA and DoD further propose to require that the addition of smaller quantities of cooking oils (e.g., from pot and dish rinsing) to the graywater system must be minimized when the vessel is within three miles of shore. The EPA and DoD propose to require that graywater discharges must not contain oil in quantities that cause a film or sheen upon or discoloration of the surface of the water or adjoining shorelines; cause a sludge or emulsion to be deposited beneath the surface of the water or upon adjoining shorelines; or contain an oil content above 15 ppm as measured by EPA Method 1664a or other appropriate method for determination of oil content as accepted by the IMO (e.g., ISO Method 9377) or USCG; or otherwise are harmful to the public health or welfare of the U.S..

In addition, minimally toxic soaps, cleaners and detergents and phosphate free soaps, cleaners, and detergents must be used in the galley, scullery, and laundry. These soaps, cleaners, and detergents should also be free from bioaccumulative compounds and not lead to extreme shifts in the receiving water pH (i.e., pH to fall below 6.0 or rise above 9.0). For vessels designed with the capacity to hold graywater, the EPA and DoD propose to require that graywater must not be discharged in federally-protected waters or the Great Lakes. In addition, such vessels would be prohibited from discharging graywater within one mile of shore if an onshore facility is available and use of such a facility is reasonable and practicable. When an onshore facility is either not available or when use of such a facility is not reasonable and practicable, production and discharge of graywater must be minimized within one mile of shore. For vessels that do not have the capacity to hold graywater, the EPA and DoD propose to require that graywater production must be minimized in federally-protected waters or the Great Lakes (e.g., no laundry, reduced cooking and dishwashing, reduced showers). In addition, such vessels would be prohibited from discharging graywater within one mile of shore if an onshore facility is available and use of such a facility is reasonable and practicable. When an onshore facility is either not available or use of such a facility is not reasonable and practicable, production and discharge of graywater must be minimized within one mile of shore.

3.2.6 Hull Coating Leachate

Hull coating leachate is defined as the constituents that leach, dissolve, ablate, or erode from the paint on the vessel hull into the surrounding seawater. Antifouling hull coatings continuously leach biocides into the surrounding water to prevent or inhibit the attachment and growth of aquatic life or biofouling to minimize the attachment and transport of non-indigenous species, decrease fuel usage, and reduce gaseous emissions.

The primary biocide in most antifouling hull coatings is copper, although zinc is also used. Copper ablative coatings, which are designed to wear or ablate away as a result of water flow over a hull, and vinyl antifouling hull coatings, which release copper as a result of copper leaching and hydrolysis of rosin particles, are the most predominantly used copper-containing coatings. Tributyltin (TBT)-based coatings were historically used on vessel hulls; however, antifouling coatings with organotin (e.g., TBT) compounds used as active ingredients are no longer authorized for use in the U.S. and as such are no longer applied to vessels of the armed forces.

Approximately 50 percent of the vessels of the armed forces use antifouling hull coatings and contribute to the hull coating leachate discharge when they are waterborne.

3.2.6.1 Discharge Constituents

Hull coating leachate includes the copper and zinc that are used as biocides. While the rate at which the metals leach from coatings is relatively slow (4-17 micrograms per square centimeter-day [$\mu g/cm^2/day$]), metal-leaching coatings can account for significant accumulations of metals in receiving waters of ports where numerous vessels are present.

3.2.6.2 Performance Standard

The EPA and DoD propose to require that anti-fouling (AF) hull coatings subject to Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA;7 U.S.C 136 et seq.) must be applied, maintained, and removed in a manner consistent with requirements on the coatings' FIFRA labels. The EPA and DoD also propose to prohibit the use of biocides or toxic materials banned for use in the U.S. (including those on EPA's List of Banned or Severely Restricted Pesticides). This requirement would apply to all vessels, including vessels with a hull coating applied outside of the U.S. AF hull coatings must not contain TBT or other organotin compounds as a hull coating biocide. AF hull coatings may contain small quantities of organotin compounds when the organotin is used as a chemical catalyst and is not present above 2,500 milligrams of total tin per kilogram of dry paint film. Any such AF hull coatings used must be designed to not slough or peel from the vessel hull. In addition, the performance standard would encourage the use of non-biocidal alternatives to copper coatings to the greatest extent practicable. The EPA and DoD also recommend to the greatest extent practicable, the use of AF hull coatings with the lowest effective biocide release rates, rapidly biodegradable components (once separated from the hull surface), or use of non-biocidal alternatives, such as silicone coatings. Not all of these alternatives are currently practical available alternatives; but it is possible that they may become practical alternatives in the future. Finally, to the greatest extent practicable, avoid the use of AF hull coatings on vessels that are regularly removed from the water and unlikely to accumulate hull growth.

3.2.7 Motor Gasoline Compensating Discharge (MOGAS)

The MOGAS compensating discharge is the seawater taken into, and discharged from, MOGAS tanks to eliminate free space where vapors could accumulate. Seawater, which is less buoyant than gasoline, occupies the free space to prevent potentially explosive gasoline vapors from forming. The retained seawater is then discharged when the vessel refills the tanks with gasoline in port or when performing maintenance. Only Navy amphibious support ships, totaling 37 vessels and representing less than one percent of the vessels of the armed forces, produce motor gasoline and compensating discharge.

3.2.7.1 Discharge Constituents

MOGAS effluent is likely to contain residual oils and soluble traces of gasoline components and additives, as well as metals. Gasoline components include alkanes, alkenes, aromatics (e.g., benzene, toluene, ethylbenzene, phenol, and naphthalene), metals, and additives. Analyses of compensating discharge have shown that benzene, toluene, ethylbenzene, phenol, and naphthalene may exceed the EPA recommended water quality criteria.

3.2.7.2 Performance Standard

The EPA and DoD propose to require that the discharge of MOGAS compensating effluent must not contain oil in quantities that: cause a film or sheen upon or discoloration of the surface of the water or adjoining shorelines; cause a sludge or emulsion to be deposited beneath the surface of the water or upon adjoining shorelines; or contain an oil content above 15 ppm as measured by the EPA Method 1664a or other appropriate method for determination of oil content as accepted by the IMO (e.g., ISO Method 9377) or USCG; or otherwise are harmful to the public health or welfare of the U.S.. In addition, if an oily sheen is observed, the EPA and DoD propose to require that any spill or overflow of oil must be cleaned up, recorded, and reported to the National Response Center immediately. The release of MOGAS compensating discharge must be minimized in port and is prohibited in federally protected waters.

3.2.8 Sonar Dome Discharge

The sonar dome discharge consists of the antifoulant materials leaching into the surrounding seawater and the release of seawater or freshwater retained within the sonar dome. Sonar domes are structures located on the hull of ships and submarines, used for the housing of electronic equipment for detection, navigation, and ranging. The shape and design pressure in sonar domes are maintained by filling them with water. Antifouling materials are used on the exterior of the sonar dome to prevent fouling which degrades sonar performance. Navy surface ship domes are made of rubber with an exterior layer that is impregnated with TBT. On submarines and MSC surface ships, the sonar domes are made of steel or glass reinforced plastic and do not contain TBT but are covered with an antifouling coating.

The discharge of the water from the interior of the sonar domes primarily occurs when the vessel is pierside and is intermittent depending on when the dome is emptied for maintenance. On average, sonar domes on surface vessels are emptied twice a year and sonar domes on submarines are emptied once a year. The discharge of sonar dome water can range between 300 gallons to 74,000 gallons depending on the size of the sonar dome and the type of maintenance event.

Approximately ten percent of vessels of the armed forces generate sonar dome discharge. These vessel types include auxiliary ships (368), submarines (72), and surface combatants (115), all of which are greater than or equal to 79 ft in length.

3.2.8.1 Discharge Constituents

Sonar dome discharges include the antifouling agents on the exterior rubber boots of the sonar dome, as well as tin, zinc, copper, nickel, and epoxy paint from a sonar dome interior. The concentrations of some of these components are estimated to exceed the EPA recommended water quality criteria.

3.2.8.2 Performance Standards

The EPA and DoD propose to require that the water inside the sonar dome not be discharged for maintenance activities unless the use of a drydock for the maintenance activity is not feasible. The water inside the sonar dome may be discharged for equalization of pressure between the interior and exterior of the dome. This would include the discharge of water required to protect the shape, integrity, and structure of the sonar dome due to internal and external pressures and forces. The EPA and DoD also propose to require that a biofouling chemical that is bioaccumulative should not be applied to the exterior of a sonar dome when a non-bioaccumulative alternative is available.

3.2.9 Submarine Bilgewater

Submarine bilgewater consists of a mixture of discharges and leakage from a wide variety of sources (e.g., seawater accumulation, normal water leakage from machinery, and fresh water washdowns), and includes all the wastewater collected in the bilge compartment, oily waste holding tank, or any other oily water or holding tank. Submarines have a drain system consisting of a series of oily bilge collecting tanks and a waste oil collection tank or tank complex to collect oily wastewater. Discharges from these tanks occur from the bottom of the tank after gravity separation. Some submarines have baffles to enhance the separation of oil and water.

Approximately one percent of the vessels of the armed forces are submarines and generate submarine bilgewater. Most submarines do not discharge bilgewater while in transit within waters subject to UNDS, and instead hold and transfer submarine bilgewater to a shore-based facility. However, one class of submarines (SSN 688) discharges some of the water phase of the separated bilgewater collecting tank, as necessary.

3.2.9.1 Discharge Constituents

Submarine bilgewater discharges can contain a variety of constituents including cleaning agents, solvents, fuel, lubricating oils, and hydraulic oils, oil and grease, volatile and semivolatile organic compounds, and metals. These constituents may be present in concentrations that could contribute to an exceedance of the EPA recommended water quality criteria.

3.2.9.2 Performance Standards

The EPA and DoD propose to require that the discharge of submarine bilgewater must not contain oil in quantities that cause a film or sheen upon or discoloration of the surface of the water or adjoining shorelines; cause a sludge or emulsion to be deposited beneath the surface of the water or upon adjoining shorelines; or contain an oil content above 15 ppm as measured by the EPA Method 1664a or other appropriate method for determination of oil content as accepted by the IMO (e.g., ISO Method 9377) or USCG; or otherwise are harmful to the public health or welfare of the U.S. In addition, the discharge of submarine bilgewater must not contain dispersants, detergents, emulsifiers, chemicals, or other substances to remove the appearance of a visible sheen. The performance standard would not prohibit the use of these materials in machinery spaces for the purposes of cleaning and maintenance activities associated with vessel equipment and structures. The discharge of submarine bilgewater also must only contain substances that are produced in the normal operation of a vessel. Oil solidifiers, flocculants, or other additives (excluding any dispersants or surfactants) may be used to enhance oil/water separation during processing in an OWS only if such solidifiers, flocculants, or other additives are minimized in the discharge and do not alter the chemical composition of the oils in the discharge. Solidifiers, flocculants, or other additives must not be directly added to, or otherwise combined with the water in the bilge. The EPA and DoD propose to require that submarine bilgewater discharges to surface water must not occur while the submarine is in port when the port has the capability to collect and transfer the bilgewater to an onshore facility.[3] If the submarine is not in port, then any such discharge must be minimized and discharged as far from shore as technologically feasible. The EPA and DoD also propose to require that submarine bilgewater discharges be minimized in federally-protected waters. Finally, submarines would need to employ management practices to minimize leakage of oil and other harmful pollutants, such as cleaning agents, solvents, fuel, lubricating oils, and hydraulic oils that are incidentally leaked or generated from machinery and during freshwater washdowns and drain to the lowest vessel compartment into the bilge.

3.2.10 Surface Vessel Bilgewater/Oil-Water Separator (OWS) Effluent

Surface vessel bilgewater is the wastewater from a variety of sources that accumulates in the lowest part of the vessel (the bilge) and the OWS effluent is produced when the wastewater is processed by an oil-water separator. Bilgewater consists of water and other residue that accumulates in a compartment of the vessel's hull or is collected in the oily waste holding tank or any other oily water holding tank. The primary sources of drainage into the bilge are the main

[3] If the submarine is in port and there is an onshore receiving facility, there will not be any discharge of bilgewater to receiving water because it will be transferred to the onshore facility.

engine room(s) and auxiliary machinery room(s), which house the vessel's propulsion system and auxiliary systems (i.e., steam boilers and water purification systems), respectively.

The composition of bilgewater varies from vessel-to-vessel and from day-to-day on the same vessel. Bilgewater generation rates vary by vessel and by vessel class because of the differences in vessel age, shipboard equipment (e.g., type of propulsion system), operations, whether the vessel segregates its non-oily wastewater from the bilge, and other procedures.

Approximately 75 percent of vessels of the armed forces generate surface vessel bilgewater/oil-water separator effluent; submarines and some of the smaller boats and service craft do not generate surface vessel bilgewater discharge/oil-water separator effluent. Oil-water separator systems are installed on most vessels of the armed forces to collect the waste oil for onshore disposal. Some smaller vessels are not outfitted with oil-water separator systems; thus, bilgewater is stored for onshore disposal.

3.2.10.1 Discharge Constituents

The propulsion and auxiliary systems use fuels, lubricants, hydraulic fluid, antifreeze, solvents, and cleaning chemicals as part of routine operation and maintenance. Small quantities of these materials enter the bilge as leaks and spills in the engineering spaces. Consequently, discharges contain oil and grease, volatile and semivolatile organic compounds, and metals which may be present in concentrations that could potentially contribute to an exceedance of the EPA recommended water quality criteria.

3.2.10.2 Performance Standards

The EPA and DoD propose to require that surface vessels equipped with an OWS must not discharge bilgewater and must only discharge OWS effluent through an oil-content monitor. All surface vessels greater than 400 gross tons must be equipped with an OWS. If measurements for gross tonnage are not available for a particular vessel, full displacement measurements may be used instead. The EPA and DoD also propose to require that the discharge of OWS effluent not occur in port if the port has the capability to collect and transfer OWS effluent to an onshore facility. In addition, the discharge of OWS effluent must be minimized within one mile of shore, must occur at speeds greater than six knots if the vessel is underway, and must be minimized in federally protected waters.

For surface vessels not equipped with an OWS, the EPA and DoD propose to require that bilgewater must not be discharged if the vessel has the capability to collect, hold, and transfer to an onshore facility. In addition, the discharge of bilgewater/OWS effluent must not contain dispersants, detergents, emulsifiers, chemicals, or other substances to remove the appearance of a visible sheen. The performance standard would not prohibit the use of these materials in machinery spaces for the purposes of cleaning and maintenance activities associated with vessel equipment and structures. The discharge of bilgewater/OWS effluent must contain substances that are produced in the normal operation of a vessel. For the discharge of OWS effluent, oil solidifiers, flocculants or other additives (excluding any dispersants or surfactants) may be used to enhance oil/water separation during processing only if such solidifiers, flocculants, or other additives are minimized and do not alter the chemical composition of the oils in the discharge. Solidifiers, flocculants, or other additives must not be directly added to, or otherwise combined with the water in the bilge.

The discharge of surface vessel bilgewater/OWS effluent must not contain oil in quantities that: cause a film or sheen upon or discoloration of the surface of the water or adjoining shorelines; cause a sludge or emulsion to be deposited beneath the surface of the water or upon adjoining shorelines; contain an oil content above 15 ppm as measured by the EPA Method 1664a or other appropriate method for determination of oil content as accepted by the IMO (e.g., ISO Method 9377) or USCG; or otherwise are harmful to the public health or welfare of the U.S.. When a visible sheen is observed as a result of a surface vessel bilgewater/OWS effluent discharge, the discharge must be suspended immediately until the problem is corrected. Any spill or overflow of oil or other engine fluids must be cleaned up, recorded, and reported immediately to the National Response Center. The surface vessel must also employ management practices to minimize leakage of oil and other harmful pollutants into the bilge. Such practices may include regular inspection and maintenance of equipment and remediation of oil spills or overflows into the bilge using oil-absorbent or other spill clean-up materials.

3.2.11 Underwater Ship Husbandry

Underwater ship husbandry discharges occur during the inspection, maintenance, cleaning, and repair of hulls and hull appendages while a vessel is waterborne. Underwater ship husbandry operations are normally conducted pierside. All vessels of the armed forces greater than or equal to 79 ft in length and some boats and service craft less than 79 ft in length, comprising 60 percent of the vessels, are expected to generate underwater ship husbandry discharge. While underwater ship husbandry discharges occur during the maintenance of all classes of vessels, many vessels less than 79 ft in length are regularly pulled from the water for hull maintenance or stored on land.

3.2.11.1 Discharge Constituents

Hull maintenance activities could result in the release of metals, primarily copper or zinc, or the introduction of non-indigenous aquatic species (NAS). Metals could be released in concentrations that have the potential to cause an adverse environmental effect and could contribute to an exceedance of the EPA recommended water quality criteria.

3.2.11.2 Performance Standards

The EPA and DoD propose to require that to the greatest extent practicable, vessel hulls with AF hull coatings must not be cleaned within 90 days after the anti-fouling coating (AFC) application. Vessel hulls must be inspected, maintained, and cleaned to minimize the removal and discharge of AF hull coatings and transport of fouling organisms. To the greatest extent practicable, rigorous vessel hull cleanings must take place in drydock or at a land-based facility where the removed fouling organisms or spent AF hull coatings can be disposed of onshore in accordance with any applicable solid waste or hazardous substance management and disposal requirements. The performance standard would also require that vessel hull cleanings be conducted in a manner that minimizes the release of AF hull coatings and viable fouling organisms (e.g., less abrasive techniques and softer brushes to the greatest extent practicable).[4] Vessel hull cleanings must also adhere to any applicable cleaning requirements found on the coatings' FIFRA label. For vessels less than 79 ft in length, the performance standard would require inspection of vessels before overland transport to a different body of water to control invasive species. For vessels greater than 79 ft in length, the performance standard would require that to the greatest extent practicable, vessel hulls with a copper-based AFC must not be cleaned within 365 days after the AFC application.

3.3 Inspections, Monitoring, Reporting, and Recordkeeping

As referenced in the rule, recordkeeping (40 CFR 1700.41) and non-compliance reporting (40 CFR 1700.42) for UNDS apply generally to each discharge performance standard unless expressly provided in a particular discharge performance standard. A vessel of the armed forces must maintain the following records for discharges:

- Name and title of the PIC who determined the necessity of the discharge;
- Date, location, and estimated volume of the discharge;
- Explanation of the reason the discharge occurred; and
- Actions taken to avoid, minimize, or otherwise mitigate the discharge (for the documented exception and to avoid future exceptions, as applicable).

All records prepared as exceptions must be maintained in accordance with 40 CFR 1700.41. All records shall be generated and maintained in the ship's logs (main, engineering, and/or damage control) or a UNDS Record Book and shall include the vessel owner information (e.g., Navy, USCG); vessel name and class; and name of the PIC. The PIC shall maintain complete records of the following information:

- Any inspection or recordkeeping requirement as specified in §§1700.14-1700.38;
- Any instance of an exception and the associated recordkeeping requirements as specified in §1700.39; and
- Any instance of non-compliance with any of the performance standards as specified in §§1700.14-1700.38.

[4]Although hull cleaning is intended to remove hull fouling organisms, both AFC and hull fouling organisms are removed during the cleaning process. The use of less abrasive cleaning techniques is intended to reduce the amount of AFC, removing more fouling organisms than AFC during cleaning. This in turn exposes the AFC rather than removing it, improving its performance and reducing the amount of fouling that occurs.

The information recorded for any instance of non-compliance shall include the following:

- Description of any non-compliance and its cause;
- Date of non-compliance;
- Period of non-compliance (time and duration);
- Location of the vessel during non-compliance;
- Corrective action taken;
- Steps taken or planned to reduce, eliminate, and prevent non-compliance in the future; and
- If the non-compliance has not been corrected, an estimate of the time the non-compliance is expected to continue.

The PIC must report any non-compliance, including the information as required under §1700.41, to the Navy Regional Environmental Coordinator office, Type Commander, and fleet commander in writing and/or electronically within five days of the time the PIC becomes aware of the circumstances. All records prepared under this section must be maintained for a period of five years from the date they are created. The information will be available to the EPA, states, or the USCG upon request. Any information made available upon request shall be appropriately classified, as applicable, and handled in accordance with applicable legal requirements regarding national security.

4. ACTION AREA

Action area means all areas affected directly, or indirectly, by the Federal action, and not just the immediate area involved in the action (50 C.F.R. §402.02).

Although UNDS are applicable only to vessels of the armed forces within waters subject to UNDS (i.e., 12 miles seaward of the U.S. baseline), effects of UNDS could occur in any waters where vessels of the armed forces operate. For this opinion, the action area includes all U.S. inland waters, estuaries and harbors, U.S. coastal waters, coastal waters of territories under U.S. jurisdiction (e.g., Guam and the U.S. Virgin Islands [USVI]), the open ocean, and coastal waters up to 12 miles from foreign shores. Although UNDS can also affect aquatic habitats within 12 miles of foreign shores where vessels of the armed forces operate, coastal and inland waters that fall within 12 nautical miles (nm) from foreign shores are not included in the action area because they fall under foreign jurisdiction.

For the purposes of analysis, NMFS adopted EPA's and DoD's use of representative action areas (RAAs), which are ecologically and geographically diverse harbors where vessels congregate and where water volumes and mixing are less than those in the open ocean. Using these RAAs, the analytical approach is intended to capture the reasonable maximum exposure scenarios for effects of pollutant discharges by conducting a detailed evaluation of potential effects on ESA-listed species and designated critical habitats in the area within the RAAs where vessels of the armed forces are concentrated (estimated as three miles around the facilities based on information provided by the Navy in their response to the draft biological opinion, July 25, 2019). EPA and DoD selected seven RAAs based on the following criteria:

- High density of regulated vessels of the armed forces present;
- Varying distribution of vessel types to capture variability in pollutant loading rates;
- Diverse ecosystem types and broad geographic representation of ESA-listed species;
- Known or expected sensitivities for appropriate taxa and their critical habitats;
- Different geography and environmental conditions; and
- Variety of ESA-listed aquatic and aquatic-dependent species representative of the taxonomic groups of all potentially affected ESA-listed aquatic and aquatic-dependent species.

The RAAs selected by the EPA and DoD are Miami, Florida (Figure 1); Norfolk, Virginia (Figure 2); Pearl Harbor, Hawaii (Figure 3); San Diego, California (Figure 4); San Francisco, California (Figure 5); Puget Sound/Seattle, Washington (Figure 6); and St. Louis, Missouri, which are all major harbors (Table 2). St. Louis is a freshwater riverine environment while the rest are marine/estuarine systems. Because St. Louis is outside the range of ESA-listed species under NMFS jurisdiction, it will not be discussed in depth in this opinion. RAA boundaries were established using the natural boundaries of the waterbody and modified in some cases to incorporate closely connected waterbodies and ESA-listed species that may occur in the vicinity of an RAA. EPA and DoD defined the natural boundaries of the waterbody to include shorelines and, for estuaries, where the waterbody meets the ocean and the estuary portion of major tributary rivers.

Table 2. RAA's Selected by the EPA and Navy for Detailed Analysis and Justification for Selection (from USEPA and Navy 2018)

Reference Port/Harbor	Armed Forces Homeports	Waterbody Represented	Ecosystem Represented	Justification
Miami, Florida (FL)	U.S. Coast Guard Station Miami U.S. Coast Guard Station Miami Beach	Biscayne Bay	Estuarine/Marine	RAA with lower large and small vessel presence. The area encompasses extensive listed Johnsons seagrass beds, contains a portion of FL's coral reef tract, and supports endangered species and associated critical habitats with unique characteristics not found in other RAAs.
Norfolk, Virginia	U.S. Coast Guard Station Joint Base Langley-Eustis JEB Little Creek-Fort Story Naval Station Norfolk, VA Newport News Navy Shipyard Norfolk Naval Shipyard Yorktown Naval Weapons Station	Chesapeake Bay	Estuarine/Marine	RAA with significant large and small vessel presence.
Pearl Harbor, Hawaii	Ford Island Inactive Ships Pearl Harbor Naval Shipyard	Pearl Harbor	Estuarine/Marine	RAA with significant large and small vessel presence. RAA includes multiple unique species and critical habitat areas.

Reference Port/Harbor	Armed Forces Homeports	Waterbody Represented	Ecosystem Represented	Justification
Puget Sound/ Seattle, Washington	Joint Base Lew-McChord Naval Air Station Whidbey Island Naval Base Everett Naval Undersea Warfare Center Division Keyport Navy Inactive Ship Maintenance Facility-Bremerton Puget Sound Navy Shipyard Sub base Bangor U.S. Coast Guard Base Seattle U.S. Coast Guard Station Port Townsend	Puget Sound	Estuarine/Marine	RAA with significant large and small vessel presence. Puget Sound near Seattle is home to several Pacific Northwest salmonid populations. RAA includes multiple critical habitat areas.
San Diego, California (CA)	Coastal River Group 1 Imperial Beach Naval Amphibious Base Coronado Naval Base San Diego North Island Naval Air Station NSC San Diego Point Loma Annex SPAWAR Systems Center U.S. Coast Guard Station San Diego	San Diego Bay	Estuarine/Marine	RAA with significant large and small vessel presence. RAA includes multiple critical habitat areas.

Reference Port/Harbor	Armed Forces Homeports	Waterbody Represented	Ecosystem Represented	Justification
San Francisco, CA	U.S. Army Reserve Watercraft Unit U.S. Coast Guard Base Alameda U.S. Coast Guard Sector San Francisco U.S. Coast Guard-Strike Team Novato U.S. Coast Guard Station Golden Gate U.S. Coast Guard Station Vallejo	San Francisco Bay	Estuarine/Marine	RAA with significant populations of small vessels. Although the estuary is emblematic of one subjected to heavy NAS invasion, the RAA includes a significant number of endangered and threatened species from a broad range of taxa and critical habitat areas.
St. Louis, Missouri	U.S. Coast Guard St. Louis	Mississippi River (upper)	Large Freshwater River	RAA with most significant population of vessels in a freshwater homeport. RAA includes a large freshwater ecosystem home to a sensitive listed freshwater unionid mussel species.

Figure 1. Representative Action Area Water Boundary Map for Miami, Florida (from USEPA and Navy 2018)

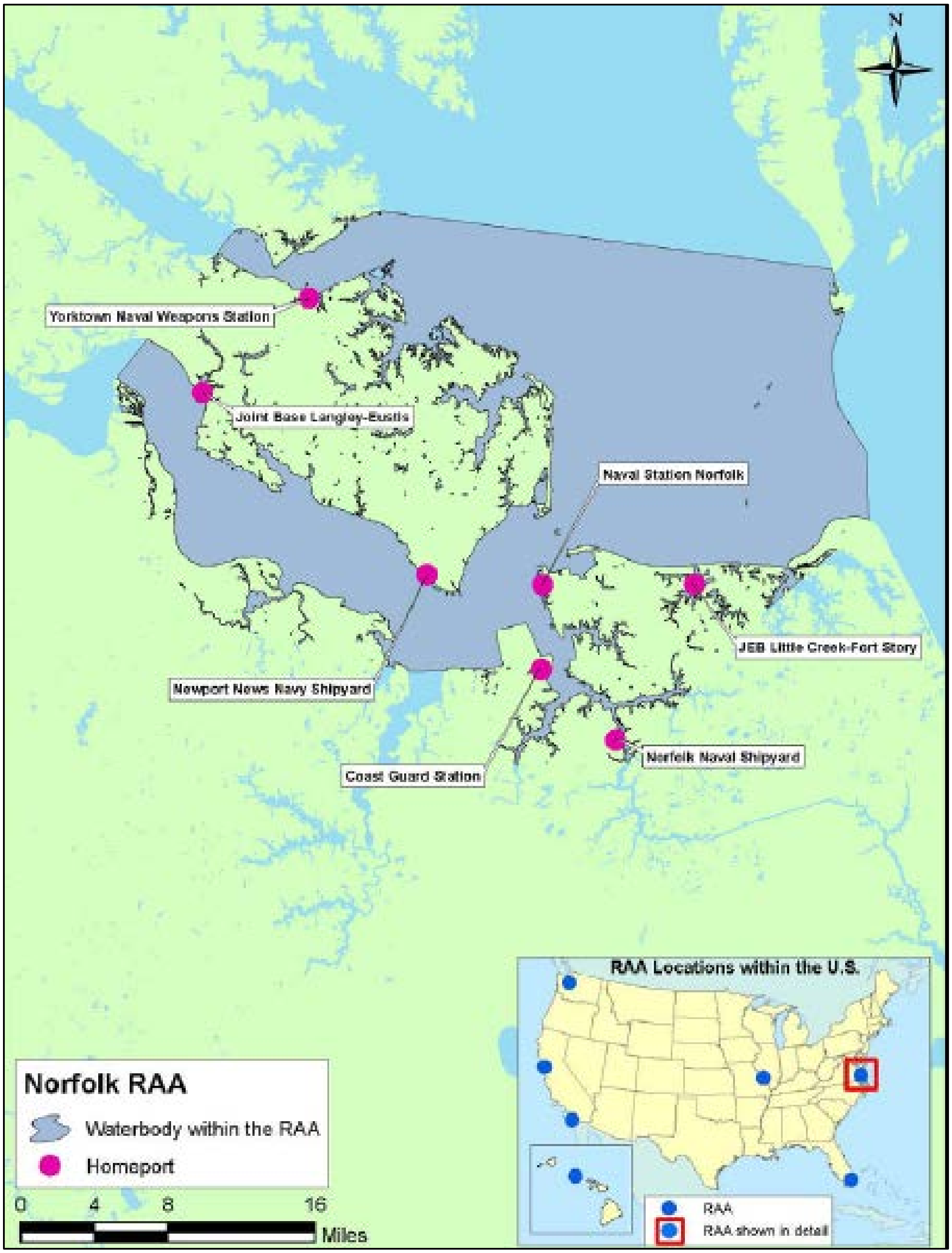

Figure 2. Representative Action Area Water Boundary for Norfolk, Virginia (from USEPA and Navy 2018)

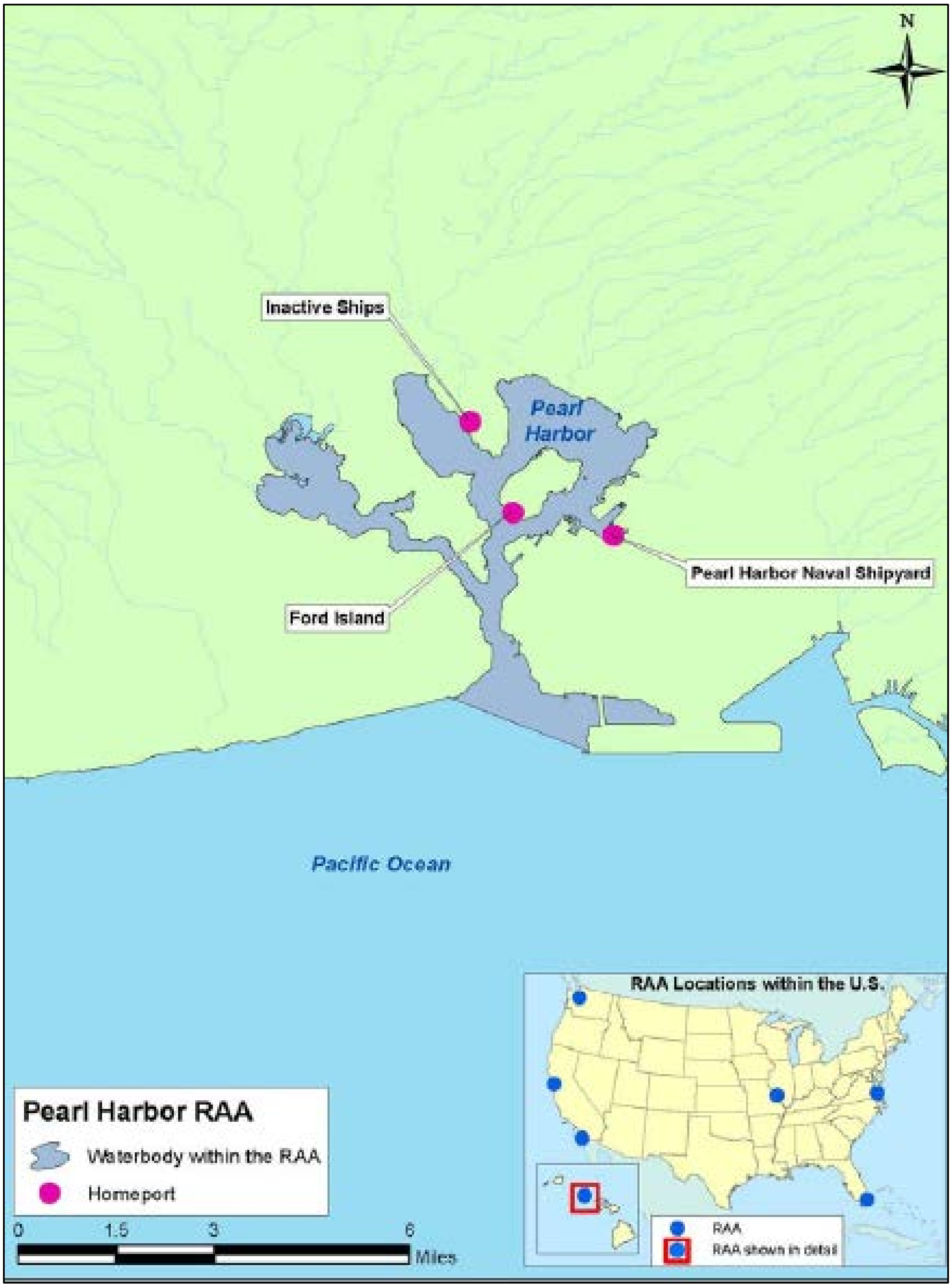

Figure 3. Representative Action Area Water Boundary for Pearl Harbor, Hawaii (from USEPA and Navy 2018)

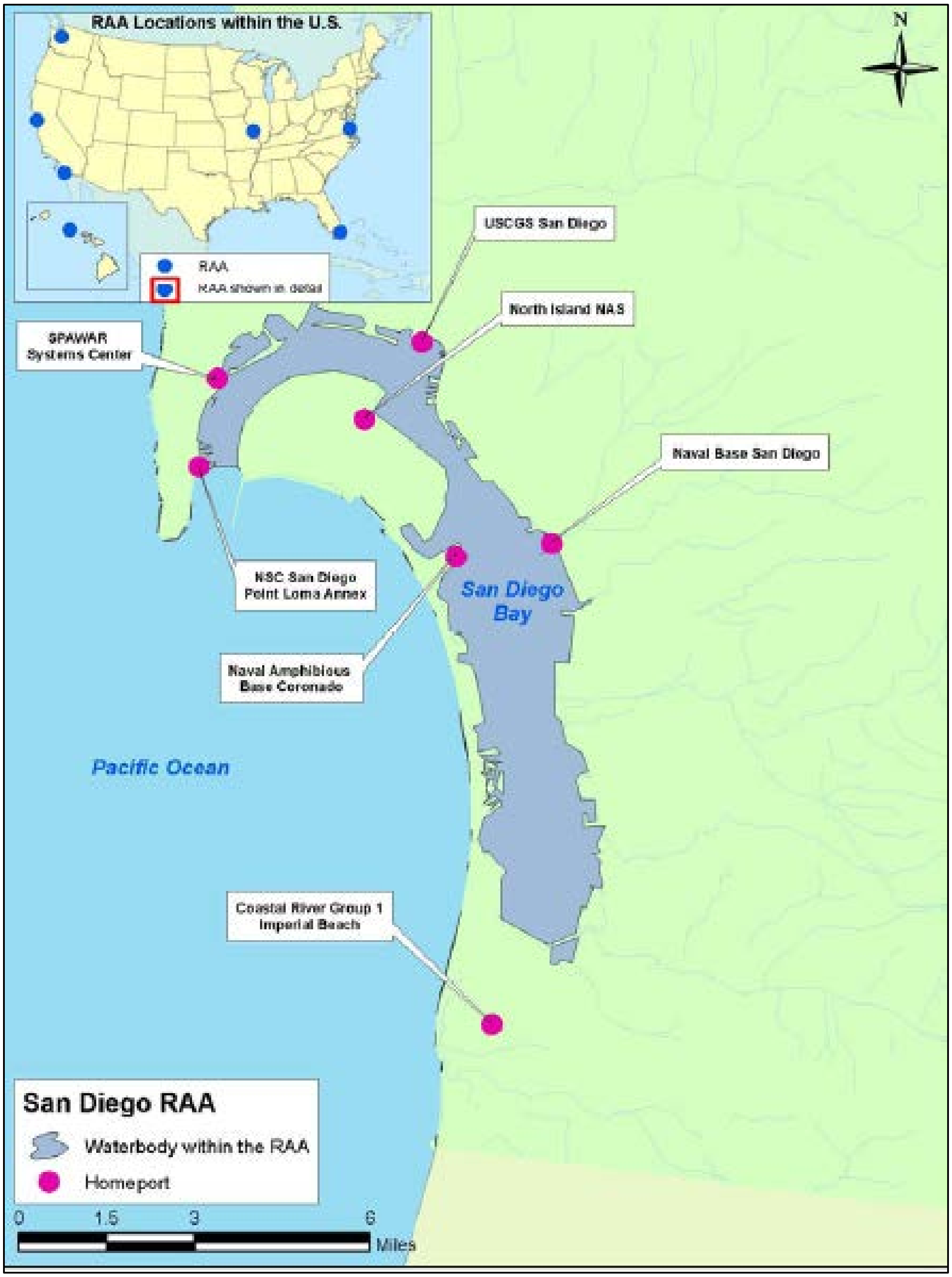

Figure 4. Representative Action Area Water Boundary for San Diego, California (from USEPA and Navy 2018)

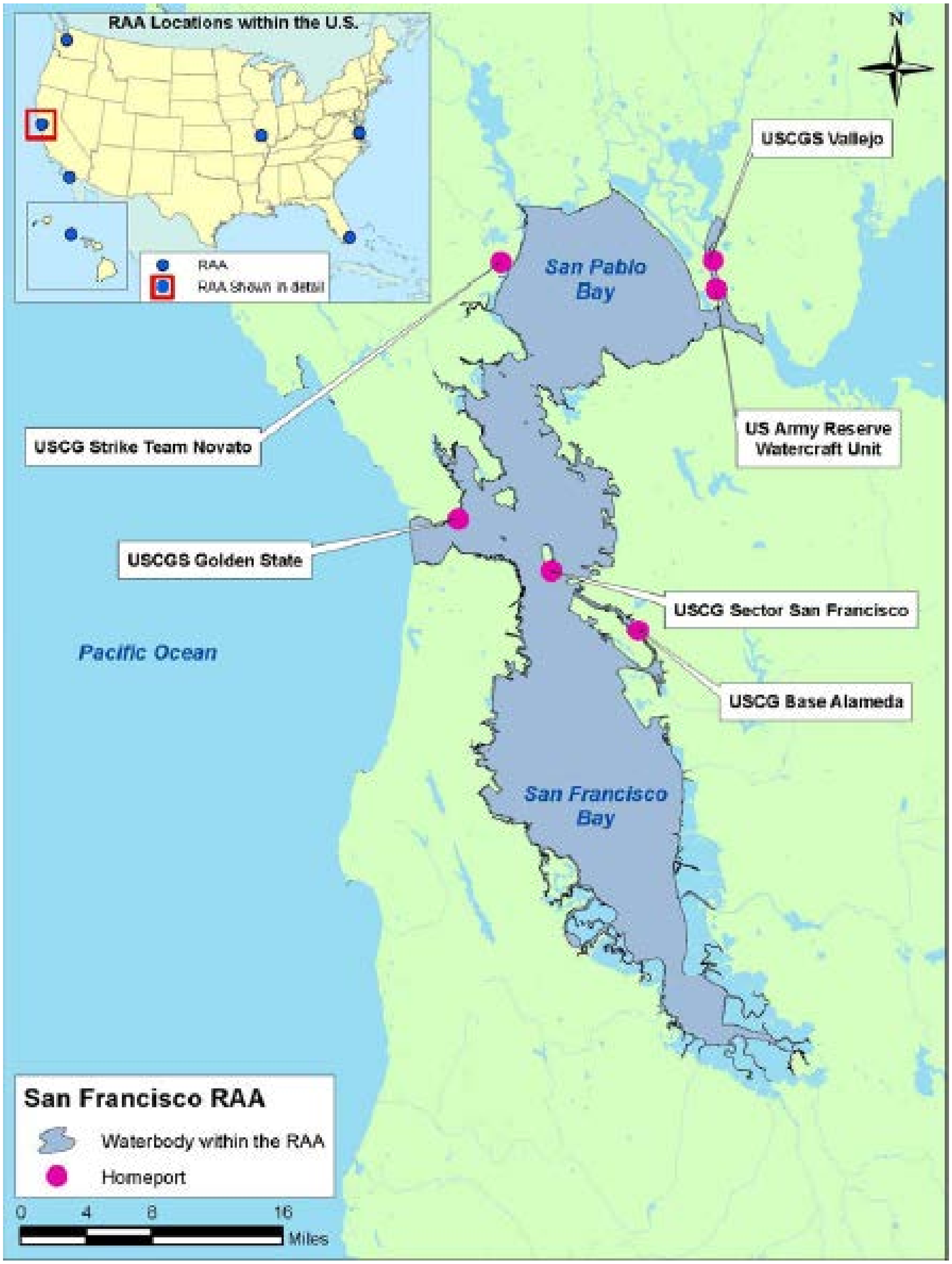

Figure 5. Representative Action Area Water Boundary for San Francisco, California (from USEPA and Navy 2018)

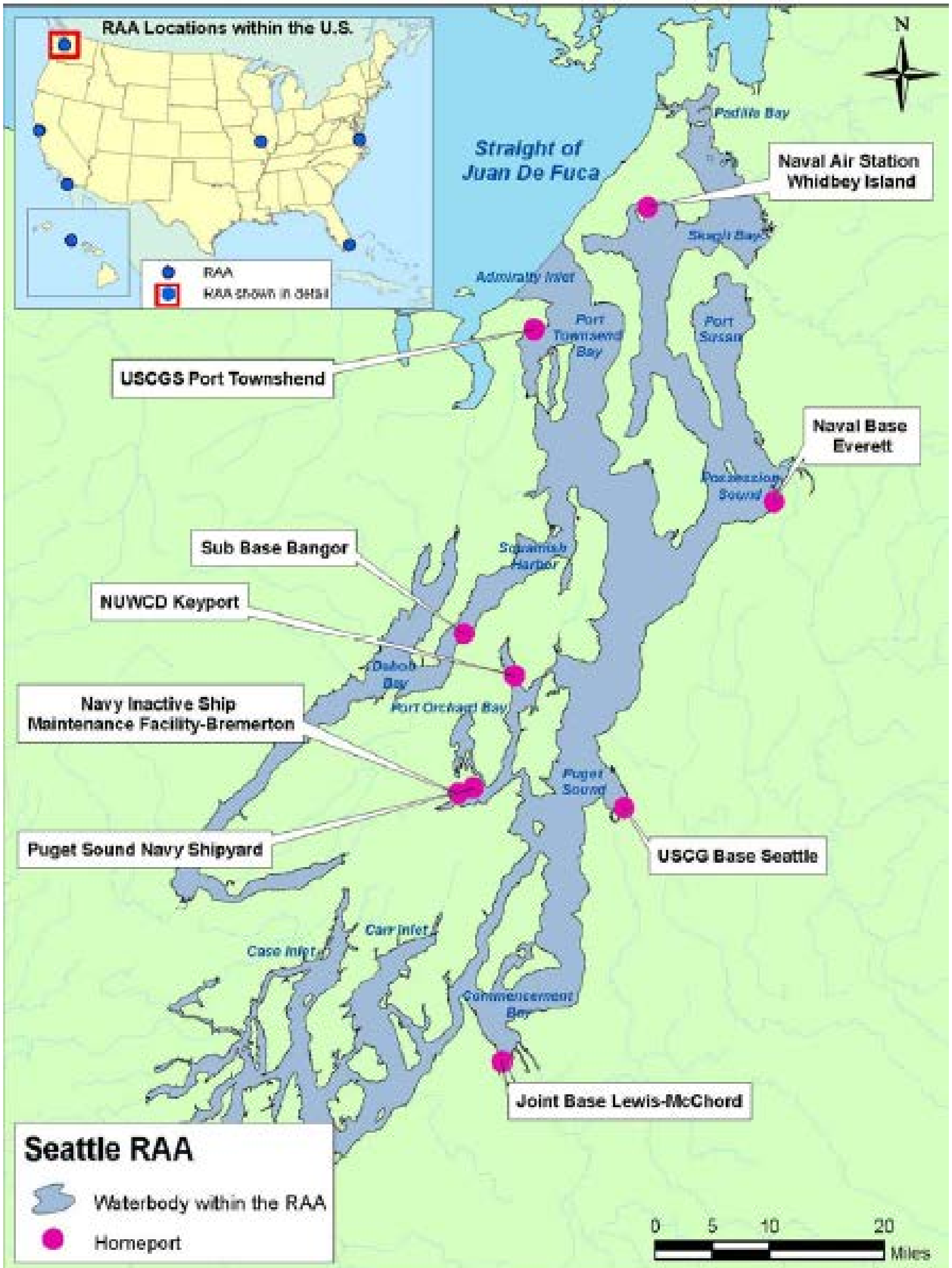

Figure 6. Representative Action Area Water Boundary for Seattle, Washington (from USEPA and Navy 2018)

5. POTENTIAL STRESSORS

Stressors are any physical, chemical or biological agent, environmental condition, external stimulus or event that may induce an adverse response in an ESA-listed species or its designated critical habitat. The action is the promulgation of a rule to establish performance standards for 11 discharges that are incidental to the normal operation of vessels of the armed forces. The vessels of the armed forces affected by the rule are of different classes, types, and sizes, which affects the discharges generated by the vessels and, therefore, the potential stressors to ESA-listed species and their designated critical habitats resulting from the Batch Two discharges.

As discussed previously, the 11 discharges that may result in stressors to ESA-listed species and their designated critical habitat are:

- catapult brake water tank and post-launch retraction exhaust
- CPP hydraulic fluid
- deck runoff
- firemain system discharge
- graywater
- hull coating leachate
- MOGAS compensating discharge
- sonar dome discharge
- submarine bilgewater
- surface vessel bilgewater/OWS effluent
- underwater ship husbandry

Table 3 identifies which of the 11 discharges included in the UNDS Batch Two rule are applicable to each vessel type. Deck runoff, hull coating leachate, and underwater ship husbandry are common to all vessel types and surface water bilgewater/OWS effluent common to surface vessels (submarines have a similar bilgewater discharge specific to this vessel type). Graywater discharge is common to all vessels 79 ft or longer and firemain system discharge is common to most large vessels.

Table 3. Discharges from Each Vessel Type Under UNDS Batch Two Rule (USEPA and Navy 2018)

	Vessel Type							
Discharge	Aircraft Carriers	Amphibious Support Ships	Auxiliary Ships	Boats	Patrol Ships	Service Craft	Submarines	Surface Combatants
Catapult Brake and Retraction Exhaust	X							
CPP Hydraulic Fluid			X					X
Deck Runoff	X	X	X	X	X	X	X	X
Firemain System Discharge	X	X	X		X		X	X
Graywater	X	X	X		X	X	X	X
Hull Coating Leachate	X	X	X	X	X	X	X	X
MOGAS Compensating Discharge		X						
Sonar Dome Discharge			X				X	X
Submarine Bilgewater							X	
Surface Water Bilgewater/OWS Effluent	X	X	X	X	X	X		X
Underwater Ship Husbandry	X	X	X	X	X	X	X	X

The types of stressors resulting from the action are the pollutants identified for each of the discharges described in section 3.2. The BE used most stringent federal and state water quality criteria for the protection of aquatic life (ALC) to identify pollutants for further evaluation because they are present in the discharges at concentrations with a higher potential to have adverse effects on aquatic and aquatic-dependent species, including those listed under ESA. The BE acknowledges the uncertainty in relying on water quality criteria due to the absence of criteria for many constituents (see the BE, Section 5.6.1, Appendix A). In addition to this uncertainty, NMFS notes that the EPA water quality criteria are intended to provide for the protection and propagation of fishes, shellfish, and wildlife and provide for recreation in and on the water. The EPA guidelines state:

> *"Because aquatic ecosystems can tolerate some stress and occasional adverse effects, protection of all species at all times and places it is not deemed necessary for the derivation of a standard. ...[given adequate data]... a reasonable level of protection will probably be provided if all except a small fraction of the taxa are protected, unless a commercially or recreationally important species is very sensitive* (Stephen et al. 1985)*."*

Because the criteria developed using the 1985 water quality guidelines are not expected to protect all species under all circumstances, exposures at the criteria may result in adverse effects in threatened and endangered species. Thus, using ALCs to identify the pollutants most likely to cause adverse effects in the analysis in the BE may not have addressed all discharges and pollutants that may affect ESA-listed species under NMFS jurisdiction.

Many pollutants occur in more than one discharge (Table 4). The descriptions of each discharge in Section 3.2 contain details on the pollutants in each discharge. The net pollutant loading under the rule was evaluated in the BE (see the BE, Section 5.1.2, Appendix A) qualitatively for those discharges having only indirect effects or lacking quantitative thresholds, or as estimated discharge loadings to the discharge zones in areas within the RAAs where vessels of the armed forces are concentrated (Table 5). In this opinion, the implications of exposures of ESA-listed species and designated critical habitat to these pollutants because of the action are evaluated by placing the analysis in the BE in the context of NMFS perspective. In other words, NMFS evaluates whether the reduction in exposure through the implementation of required performance standards under UNDS Batch Two is sufficiently protective of ESA-listed species and designated critical habitat to avoid all take and destruction and adverse modification of critical habitat.

Table 4. Stressors of Concern Present in Discharges Affected by the Rule

	Catapult Brake Water Tank and Post-Launch Retraction Exhaust	CPP Hydraulic Fluid	Deck Runoff	Firemain Systems	Graywater	Hull Coating Leachate	MOGAS	OWS	Sonar Dome Discharge	Submarine Bilgewater	Underwater Ship Husbandry
Non-indigenous Aquatic Species											✓
Hydrocarbons/Petroleum Residue (oil, grease, organics)	✓	✓	✓	✓	✓		✓	✓		✓	
Oxygen Demanding Substances					✓						
Pharmaceutical and Personal Care Products					✓						
Detergents, Surfactants, Disinfectants, Solvents			✓		✓					✓	
Manufactured Chemicals (plasticizers, additives)	✓						✓			✓	
Solids (e.g., plastic, metal, or paint chips)			✓		✓				✓		
Nutrients	✓				✓						
Pathogens and Bacteria					✓						✓
Metals	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓

Table 5. Range in Estimated Discharge Concentrations of Stressors of Concern in Modeled RAAs in Estuaries (from USEPA and Navy 2018)

Class	Pollutant	Range of Pollutant Concentrations for Estuarine and Harbor RAAs (µg/L)
Metals	Cadmium	7.40E-08 - 7.80E-06
	Chromium	3.60E-06 - 0.00039
	Total Copper[1]	0.0016 - 0.3
	Iron	0.000023 - 0.038
	Lead	8.30E-06 - 0.00082
	Mercury	5.7E-10 - 4.6E-07
	Nickel	5.70E-06 - 0.0097
	Silver	1.4E-09 - 2.80E-06
	Total Zinc[1]	0.004 - 0.41
Petroleum Hydrocarbons	Oil and Grease	0.00073 - 0.074
	Total Petroleum Hydrocarbons (TPH)	3.70E-08 - 1.90E-06
Toxics and Non-Conventional Pollutants with Toxic Effects	Bis (2-ethylhexyl) phthalate	1.8E-07 - 0.014
	Tributyltin	0 - 0.00021
	Chlorine Produced Oxidants	0 - 0.0037
Nutrients/ Water Quality	Nitrate/Nitrite	2.4E-06 - 0.0011
	Total Kjeldahl Nitrogen	0.000035 - 0.049
	Total Nitrogen	0.000037 - 0.05
	Ammonia as Nitrogen	0.000019 - 0.036
	Total Phosphorus	1.3E-05 - 0.0025
	Total Organic Carbon	0 - 0.0039
	Total Suspended Solids	0.00014 - 0.28
	Biological Oxygen Demand (BOD)	0.000096 - 0.19
	Chemical Oxygen Demand (COD)	0.00026 - 0.28

[1] Measures of total copper and zinc concentrations assumed to be dissolved in EPA's analysis because dissolved copper and zinc were not measured in all discharges

6. Status of Endangered Species Act Protected Resources

This section identifies the ESA-listed species and designated critical habitat that potentially occur within the action area (Table 6) that may be affected by the UNDS Phase II Batch Two rulemaking. This section first identifies the species and designated critical habitats in the action area that may be affected, but are not likely to be adversely affected by the 11 types of discharges covered under the UNDS Phase II Batch Two rule. The remaining species and designated critical habitats deemed likely to be adversely affected by discharges from vessels of the armed forces and associated water quality conditions from complying with the UNDS Phase II Batch Two rule are carried forward through the remainder of this opinion.

Table 6. Threatened and Endangered Species that May be Affected by EPA and DoD's UNDS Phase II Batch Two Rule.

Species	ESA Status	Critical Habitat	Recovery Plan
Marine Mammals – Cetaceans			
Blue Whale (*Balaenoptera musculus)*	E – 35 FR 18319	-- --	07/1998
Fin Whale (*Balaenoptera physalus)*	E – 35 FR 18319	-- --	75 FR 47538
Bowhead Whale (*Balaena mysticetus*)	E – 35 FR 18319	-- --	-- --
Gray Whale (*Eschrichtius robustus*) - Western North Pacific DPS*	E – 35 FR 18319	-- --	-- --
Humpback Whale (*Megaptera novaeangliae*) - Western North Pacific, Central America, Arabian Sea*, Cape Verde Islands/Northwest Africa*, and Mexico DPSs	E – Western North Pacific and Central America DPSs T – Mexico DPS 81 FR 62259	-- --	11/1991
North Pacific Right Whale (*Eubalaena jabonica*)	E – 73 FR 12024	73 FR 19000	78 FR 34347
North Atlantic Right Whale (*Eubalaena glacialis*)	E – 73 FR 12024	81 FR 4837	70 FR 32293
Southern Right Whale (*Eubalaena australis*)	E – 35 FR 8491	-- --	-- --

Species	ESA Status	Critical Habitat	Recovery Plan
False Killer Whale (*Pseudorca crassidens*) - Main Hawaiian Islands Insular DPS	E – 77 FR 70915	83 FR 35062	-- --
Sei Whale (*Balaenoptera borealis*)	E – 35 FR 18319	-- --	12/2011
Killer Whale (*Orcinus orca*) - Southern Resident DPS	E – 70 FR 69903 Amendment 80 FR 7380	71 FR 69054	73 FR 4176
Beluga Whale (*Delphinapterus leucas*)- Cook Inlet DPS	E – 73 FR 62919	76 FR 20179	82 FR 1325
Sperm Whale (*Physeter macrocephalus*)	E – 35 FR 18319	-- --	75 FR 81584
Bryde's Whale – Gulf of Mexico subspecies	E – 84 FR 15446	-- --	-- --
Hector's Dolphin* (Cephalorhynchus hectori) – Maui's Dolphin and South Island Hector's Dolphin Subspecies	E – Maui's Subspecies T – South Island Subspecies 82 FR 43701	-- --	-- --
Marine Mammals – Pinnipeds			
Ringed Seal (*Phoca hispida hispida*) –Arctic DPS	T – Arctic DPS 77 FR 76706	79 FR 73010 (Proposed – Arctic DPS)	-- --
Guadalupe Fur Seal (*Arctocephalus townsendi*)	T – 50 FR 51252	-- --	-- --
Hawaiian Monk Seal (*Neomonachus schauinslandi*)	E – 41 FR 51611	80 FR 50925	72 FR 46966 2007
Bearded Seal (*Erignathus barbatus*) – Beringia DPS	T – 77 FR 76739	-- --	-- --
Mediterranean Monk Seal* (*Monachus monachus*)	E – 35 FR 8491	-- --	-- --
Steller Sea Lion (*Eumetopias jubatus*) – Western DPS	E – 55 FR 49204	58 FR 45269	73 FR 11872 2008

Species	ESA Status	Critical Habitat	Recovery Plan
Sea Turtles			
Green Turtle (*Chelonia mydas*) – North Atlantic, South Atlantic, East Pacific, Central North Pacific, Central West Pacific, East Indian-West Pacific*, Southwest Pacific*, Central South Pacific, North Indian Ocean*, Southwest Indian Ocean*, and Mediterranean* DPSs	E – Central South Pacific Ocean DPS T - rest of DPSs in action area 81 FR 20057	63 FR 46693 (North Atlantic DPS only)	U.S. Atlantic – 10/1991 U.S. Pacific – 63 FR 28359
Hawksbill Turtle (*Eretmochelys imbricata)*	E – 35 FR 8491	63 FR 46693	U.S. Caribbean, Atlantic, and Gulf of Mexico - 57 FR 38818 U.S. Pacific - 63 FR 28359
Kemp's Ridley (*Lepidochelys kempii*)	E – 35 FR 18319	-- --	U.S. Caribbean, Atlantic, and Gulf of Mexico - 09/2011 (2nd revision)
Loggerhead Turtle (*Caretta caretta*) – Northwest Atlantic Ocean, South Atlantic Ocean, Southeast Indo-Pacific Ocean, Northeast Atlantic Ocean, North Pacific Ocean, South Pacific Ocean, North Indian Ocean, Southwest Indian Ocean, and Mediterranean Sea DPSs	E – North Pacific, South Pacific, Mediterranean Sea, North Indian Ocean, and Northeast Atlantic DPSs T - rest of DPSs in action area 76 FR 58868	79 FR 39855 (Northwest Atlantic Ocean DPS only)	U.S. Pacific – 63 FR 28359 Northwest Atlantic - 74 FR 2995 U.S. Caribbean, Atlantic, and Gulf of Mexico - 10/1991 U.S. Pacific - 05/1998 Northwest Atlantic - 01/2009

Species	ESA Status	Critical Habitat	Recovery Plan
Olive Ridley (*Lepidochelys olivacea*) – Mexico's Pacific Coast Breeding Populations, All Other Populations	E – Mexico's Pacific Coast Breeding Populations T – All Other 43 FR 32800	-- --	Mexico's Pacific Coast - 63 FR 28359
Leatherback (*Dermochelys coriacea*)	E – 35 FR 8491	44 FR 17710 and 77 FR 4170	U.S. Caribbean, Atlantic, and Gulf of Mexico - 63 FR 28359 U.S. Pacific - 05/1998
Fishes			
Shortnose sturgeon (*Acipenser brevirostrum*)	E – 32 FR4001	-- --	63 FR 69613
Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) – Gulf of Maine, New York Bight, Chesapeake, Carolina, and South Atlantic DPSs	E – South Atlantic, Carolina, Chesapeake Bay, and New York Bight DPSs T – Gulf of Maine DPS 77 FR 5879	82 FR 39160	-- --
Green Sturgeon (*Acipenser medirostris*) – Southern DPS	T – 71 FR 17757	74 FR 52300	2010 (Outline)
Gulf Sturgeon (*Acipenser oxyrinchus desotoi*)	T – 56 FR 49653	68 FR 13370	09/1995
Sakhalin Sturgeon* (Acipenser mikadoi)	E – 79 FR 31222	-- --	-- --
Adriatic Sturgeon* (Acipenser naccarii)	E – 79 FR 31222	-- --	-- --
European Sturgeon* (Acipenser sturio)	E – 79 FR 31222	-- --	-- --
Chinese Sturgeon* (Acipenser sinensis)	E – 79 FR 31222	-- --	-- --
Dusky Sea Snake* (*Aipysurus fuscus*)	E – 80 FR 60560	-- --	-- --

Species	ESA Status	Critical Habitat	Recovery Plan
Smalltooth Sawfish (*Pristis pectinata*) – U.S. and Non-U.S. Portion of Range* DPSs	E – 68 FR 15674	74 FR 45353 (U.S. DPS)	74 FR 3566 - (U.S. DPS)
Largetooth Sawfish (*Pristis pristis*)	E – 79 FR 73978	-- --	-- --
Narrow Sawfish* (*Anoxypristis cuspidata*)	E – 79 FR 73977	-- --	-- --
Dwarf Sawfish* (*Pristis clavata*)	E – 79 FR 73977	-- --	-- --
Green Sawfish* (*Pristis zijsron*)	E – 79 FR 73977	-- --	-- --
Giant Manta Ray (*Manta birostris*)	T – 83 FR 2916	-- --	-- --
Oceanic Whitetip Shark (*Carcharhinus longimanus*)	T – 83 FR 4153	-- --	-- --
Scalloped Hammerhead Shark (*Sphyrna lewini*) – Central and Southwest Atlantic, Eastern Atlantic*, Eastern Pacific, Indo-West Pacific DPSs	T - Central and Southwest Atlantic, Indo-West Pacific E - Eastern Atlantic, Eastern Pacific 79 FR 38213	-- --	-- --
Daggernose Shark* (*Isogomphodon oxyrhynchus*)	E – 82 FR 21722	-- --	-- --
Striped Smoothhound Shark* (*Mustelus fasciatus*)	E – 82 FR 21722	-- --	-- --
Narrownose Smoothhound Shark* (*Mustelus schmitti*)	E – 82 FR 21722	-- --	-- --
Spiny Angelshark* (*Squatina guggenheim*)	E – 82 FR 21722	-- --	-- --
Smoothback Angelshark* (*Squatina oculata*)	E – 81 FR 50394	-- --	-- --
Sawback Angelshark* (*Squatina aculeata*)	E – 81 FR 50394	-- --	-- --
Argentine Angelshark* (*Squatina argentina*)	E – 82 FR 21722	-- --	-- --
Common Angelshark* (*Squatina squatina*)	E – 81 FR 50394	-- --	-- --
Brazilian Guitarfish* (*Rhinobatos horkelii*)	E – 82 FR 21722	-- --	-- --

Species	ESA Status	Critical Habitat	Recovery Plan
Blackchin Guitarfish* (*Rhinobatos cemiculus*)	T – 82 FR 6309	-- --	-- --
Common Guitarfish* (*Rhinobatos rhinobatos*)	T – 82 FR 6309	-- --	-- --
Gulf Grouper* (*Mycteroperca jordani*)	E – 81 FR 72545	-- --	-- --
Nassau Grouper (*Epinephelus striatus*)	T – 81 FR 42268	-- --	-- --
Island Grouper* (*Mycteroperca fusca*)	T – 81 FR 72545	-- --	-- --

Species	ESA Status	Critical Habitat	Recovery Plan
Steelhead Trout (*Oncorhynchus mykiss*) – Southern California, Upper Columbia River, Snake River Basin, Middle Columbia River, Lower Columbia River, Upper Willamette River, South-Central California Coast, Central California Coast, Northern California, California Central Valley, and Puget Sound DPSs	E - Southern California T - All other DPSs in action area 72 FR 26722	Southern California, South-Central California Coast, Central California Coast, Northern California, California Central Valley - 70 FR 52487 Upper Columbia River, Snake River Basin, Middle Columbia River, Lower Columbia River, Upper Willamette River - 70 FR 52629 Puget Sound - 81 FR 9251	Southern California - 77 FR 1669 Upper Columbia River - 72 FR 57303 Snake River Basin - 81 FR 74770 (Draft) Middle Columbia River - 74 FR 50165 Lower Columbia River - 78 FR 41911 Upper Willamette River - 76 FR 52317 South-Central California Coast - 78 FR 77430 Central California Coast, Northern California - 81 FR 70666 California Central Valley - 79 FR 42504
Altantic Salmon (*Salmo salar*)– Gulf of Maine DPS	E – 74 FR 29344	74 FR 39903	70 FR 75473 and 81 FR 18639 (Drafts)

Species	ESA Status	Critical Habitat	Recovery Plan
Chinook Salmon (*Oncorhynchus tshawytscha*) – Sacramento River Winter-Run, Upper Columbia River Spring-Run, Snake River Spring/Summer-Run, Snake River Fall-Run, Central Valley Spring-Run, California Coast, Puget Sound, Lower Columbia River, and Upper Willamette River Evolutionary Significant Units (ESUs)	70 FR 37160	Sacramento River Winter-Run - 58 FR 33212 Upper Columbia River Spring-Run and Upper Willamette River - 70 FR 52629 Snake River Spring/Summer-Run - 64 FR 57399 Snake River Fall-Run - 58 FR 68543 Central Valley Spring-Run and California Coast - 70 FR 52488 Puget Sound and Lower Columbia River - 70 FR 52629	Sacramento River Winter-Run and Central Valley Spring-Run - 79 FR 42504 Upper Columbia River Spring-Run - 72 FR 57303 Snake River Spring/Summer-Run - 81 FR 74770 (Draft) Snake River Fall-Run - 80 FR 67386 (Draft) California Coast - 81 FR 70666 Puget Sound - 72 FR 2493 Lower Columbia River - 78 FR 41911 Upper Willamette River - 76 FR 52317
Chum Salmon (*Oncorhynchus keta*) – Hood Summer-Run and Columbia River ESUs	T – 70 FR 37160	70 FR 52629	Hood Summer-Run - 72 FR 29121 Columbia River - 78 FR 41911
Sockeye Salmon (*Oncorhynchus nerka*) – Snake River and Ozette Lake ESUs	E - Snake River T - Ozette Lake 70 FR 37160	Snake River - 58 FR 68543 Ozette Lake - 70 FR 52630	Snake River - 80 FR 32365 Ozette Lake - 74 FR 25706

Species	ESA Status	Critical Habitat	Recovery Plan
Coho Salmon (*Oncorhynchus kisutch*) – Central California Coast, Southern Oregon/Northern California Coasts, Lower Columbia River, and Oregon Coast ESUs	E - Central California Coast T - rest of ESUs in action area (Southern Oregon/Northern California Coasts, Lower Columbia River) 70 FR 37160 Oregon Coast - 73 FR 7816	Central California Coast, Southern Oregon/Northern California Coasts - 64 FR 24049 Lower Columbia River - 81 FR 9251 Oregon Coast - 73 FR 7816	Central California Coast - 77 FR 54565 Southern Oregon/ Northern California Coasts - 79 FR 58750 Lower Columbia River - 78 FR 41911 Oregon Coast - 81 FR 90780
Totoaba* (*Totoaba macdonaldi*)	E – 44 FR 29478	-- --	-- --
Bocaccio (*Sebastes paucispinis*) – Puget Sound/Georgia Basin DPS	E – 75 FR 22276 and amendment 82 FR 7711	79 FR 68041	81 FR 54556 (Draft)
Yelloweye Rockfish (*Sebastes ruberrimus*) – Puget Sound/Georgia Basin DPS	T – 82 FR 7711	79 FR 68041	81 FR 54556 (Draft)
Pacific Eulachon (*Thaleichthys pacificus*) – Southern DPS	T – 75 FR 13012	76 FR 65323	9/2017
African Coelanth* (*Latimeria chalumnae*) – Tanzanian DPS	T – 81 FR 17398	-- --	-- --
Corals			
Elkhorn Coral (*Acropora palmata*)	T – 79 FR 53851	73 FR 72210	80 FR 12146
Staghorn Coral (*Acropora cervicornis*)	T – 79 FR 53851	73 FR 72210	80 FR 12146
Lobed Star Coral (*Orbicella annularis*)	T – 79 FR 53851	-- --	-- --
Boulder Star Coral (*Orbicella franksi*)	T – 79 FR 53851	-- --	-- --
Mountainous Star Coral (*Orbicella faveolata*)	T – 79 FR 53851	-- --	-- --

Species	ESA Status	Critical Habitat	Recovery Plan
Pillar Coral (*Dendrogyra cylindrus*)	T – 79 FR 53851	-- --	-- --
Rough Cactus Coral (*Mycetophyllia ferox*)	T – 79 FR 53851	-- --	-- --
Acropora globiceps	T – 79 FR 53851	-- --	-- --
Acropora jacquelineae	T – 79 FR 53851	-- --	-- --
*Acropora lokani**	T – 79 FR 53851	-- --	-- --
*Acropora pharaonis**	T – 79 FR 53851	-- --	-- --
Acropora retusa	T – 79 FR 53851	-- --	-- --
*Acropora rudis**	T – 79 FR 53851	-- --	-- --
Acropora speciosa	T – 79 FR 53851	-- --	-- --
*Acropora tenella**	T – 79 FR 53851	-- --	-- --
*Anacropora spinosa**	T – 79 FR 53851	-- --	-- --
Euphyllia paradivisa	T – 79 FR 53851	-- --	-- --
Isopora crateriformis	T – 79 FR 53851	-- --	-- --
*Montiplora australiensis**	T – 79 FR 53851	-- --	-- --
*Pavona diffluens**	T – 79 FR 53851	-- --	-- --
*Porites napopora**	T – 79 FR 53851	-- --	-- --
Seriatopora aculeata	T – 79 FR 53851	-- --	-- --
*Cantharellus noumeae**	E – 80 FR 60560	-- --	-- --
Black Abalone (*Haliotis cracherodii*)	E – 74 FR 1937	76 FR 66805	-- --
White Abalone (*Haliotis sorenseni*)	E – 66 FR 29046	-- --	73 FR 62257
Chambered Nautilus (*Nautilus pompilius*)	T – 83 FR 48976	-- --	-- --
* Foreign Species			

6.1 Species and Designated Critical Habitat Not Likely to be Adversely Affected

NMFS uses two criteria to identify the ESA-listed or designated critical habitat that are not likely to be adversely affected by the action, as well as the effects of activities that are interrelated to or interdependent with the Federal agency's action. The first criterion is exposure, or some reasonable expectation of a co-occurrence, between one or more potential stressors associated with the action and ESA-listed species or designated critical habitat. If we conclude that an ESA-listed species or designated critical habitat is not likely to be exposed to the activities associated with the action, we must also conclude that the species or critical habitat is not likely to be adversely affected by those activities.

The second criterion is the probability of a response given exposure. ESA-listed species or designated critical habitat that co-occurs with a stressor of the action but is not likely to respond to the stressor is also not likely to be adversely affected by the action. We applied these criteria to the ESA-listed species in Table 6 and we summarize our results below.

In the case of the UNDS Phase II Batch Two rule, ESA-listed species and designated critical habitat occur in waters affected by performance standards proposed by EPA and DoD and will co-occur with water quality conditions that are subject to the discharge requirements for catapult water brake tank and post-launch retraction exhaust, CPP hydraulic fluid, deck runoff, firemain systems, graywater, hull coating leachate, MOGAS compensating discharge, sonar dome discharge, submarine bilgewater, surface vessel bilgewater/OWS effluent, and underwater ship husbandry.

The probability of an effect in a species or designated critical habitat is a function of exposure intensity and susceptibility of a species to the stressor's effects (i.e., probability of response). An action warrants a "may affect, not likely to be adversely affected" finding when its effects are wholly *beneficial, insignificant* or *discountable*. *Beneficial* effects have an immediate positive effect without any adverse effects to the species or habitat.

Insignificant effects relate to the size or severity of the impact and include those effects that are undetectable, not measurable, or so minor that they cannot be meaningfully evaluated. Insignificant is the appropriate effect conclusion when plausible effects are going to happen, but will not rise to the level of constituting an adverse effect.

Discountable effects are those that are extremely unlikely to occur. For an effect to be discountable, there must be a plausible adverse effect (i.e., a credible effect that could result from the action and that would be an adverse effect if it did affect a listed species), but it is very unlikely to occur.

Because UNDS applies only to vessels of the armed forces within waters subject to UNDS, the discharges prohibited or restricted in certain UNDS-regulated waters may occur in waters at or beyond 12 mi from shore. These discharges already occur in waters seaward of 12 mi from shore incidental to normal vessel operations and in compliance with international standards. While dissipation of discharges would be expected to be more rapid at sea, discharges in areas where *Sargassum* occurs could be problematic (e.g., sorption, probability of marine organisms being exposed to discharge plumes). Given the amount of vessel traffic in waters outside 12 mi from shore, the potential for increased frequency of some discharges resulting from vessels of the armed forces operating in these waters would be negligible, particularly when compared to the global maritime traffic and the total number of vessels likely to be discharging to offshore waters. Additionally, armed forces vessels would still operate with environmental controls, regardless of where the vessels transit. Because of the potentially small increase in frequency of some Batch Two discharges that would occur in a very large area (all waters between 12 mi from the U.S. baseline seaward to the seaward boundaries of foreign territorial seas), potential effects from these discharges to ESA-listed species would be insignificant. Therefore, these activities are not likely to adversely affect listed species and will not be considered further in this opinion.

Factors Influencing Probability of Exposure and Response

The primary exposure pathway for toxic aquatic pollutants in gilled species (fishes and some invertebrates) is uptake via the gills as water continuously passes over the gill filaments to oxygenate blood and regulate ion balance. For saltwater fishes, exposure to toxicants in water also occurs through ingestion because most marine fishes "osmoregulate" by drinking water and excreting solutes in order to maintain a lower concentration in their body fluids than saltwater. Most marine invertebrates have the same internal concentration of solutes as the water they live in and do not osmoregulate (Larsen et al. 2014). The exception is filter-feeding invertebrates, which ingest small quantities of seawater when feeding. For saltwater fishes and filter-feeding invertebrates, exposure to toxic pollutants may also occur by ingestion of contaminated food resources.

While sea turtles breathe air, they also occasionally drink water and excrete solute to regulate their internal ion balance. Whales do not drink seawater. Whale osmoregulation employs genetic and physiologic adaptations such as increased filtration rates, urine volume, and kidney size along with high solute levels in urine and plasma (Kjeld 2003; Birukawa et al. 2005). Baleen whales, similar to filter-feeding invertebrates, do ingest some seawater when feeding. The pathway for direct exposure, and subsequent response of whales to pollutants in saltwater is therefore limited relative to marine fishes and invertebrates. Sea turtle exposures are less than those of marine fishes because turtles do not drink continuously, whereas saltwater fishes both drink and continuously pass water over their gills. Pinnipeds also appear to restrict their intake of seawater and derive fluids from their food resources. Therefore, sea turtles and marine mammals are more likely to be exposed to pollutants in the food they consume.

Aquatic pollutants may cause effects due to dietary exposures or through altering the quantity or quality of prey. The impacts of bioaccumulating and biomagnifying toxicants like mercury and persistent organic pollutants are more complicated to evaluate for sea turtles and marine mammals than for many fish species. There are no reliable exposure-response data for exposures in water to indicate when pollutants reach tissue concentrations resulting in adverse effects to individuals or their offspring. The age of exposed organisms, their position in the food web, and their home range and extent of migrations are important considerations in such analyses. While there are data for toxic effects that may influence prey populations and tissue accumulation in prey or prey-like species under controlled laboratory conditions, information on dietary toxicity and food web accumulation is extremely limited for marine environments.

Information on prey items and foraging areas can suggest the potential for toxic exposures, but uncertainty in whether actual adverse effects will occur can be substantial. This uncertainty is because the presence of a contaminant in the tissues of an organism only confirms exposure and does not provide useful information about adverse effects. The impact of this uncertainty is less when the action area in which pollutant exposures may occur comprises a very small portion of a species foraging area and exposure to pollutants in seawater is expected to be minimal. For example, whales do not drink seawater and many forage over a very wide geographic area, so a determination of not likely to adversely affect (NLAA) for areas affected by UNDS Batch Two for these species may be reasonable. Similarly, as a pelagic species, only a fraction of the diet of leatherback sea turtles will be from waters affected by the UNDS Batch Two rule. While green and hawksbill sea turtles are likely to forage in waters affected by UNDS, if adverse effects to forage species under the chronic criteria are not expected, adverse effects to these sea turtle species due to dietary exposures would not be expected.

Allometric differences (e.g., body size, membrane area, organ size) between species and between different life stages are also a consideration when evaluating the probability of a response to a toxicant. For example, smaller aquatic organisms generally succumb to toxic effects more rapidly than larger organisms because it takes a longer time for exposures to reach critical concentrations within the target organs of larger organisms.

Species and Critical Habitats Not Likely to be Adversely Affected

The contaminants in the 11 Batch Two discharges (Table 4) could result in habitat loss or degradation, eutrophication associated with increased nutrients, reductions in dissolved oxygen (DO), harmful algal blooms from nutrient inputs, and physical and toxic effects from exposure to contaminants on listed species and their food resources. These effects are expected to be concentrated in RAAs and other ports and harbors regularly used by vessels of the armed forces where ESA-listed species and designated critical habitat co-occur.

Several ESA-listed species and designated critical habitat that may be present in the action area occur outside of any ports and harbors, including RAAs selected by the EPA and Navy for analysis of the effects of the action, where vessels of the armed forces could be concentrated. Exposure of these species and designated critical habitats to Batch Two discharges would only occur during transit of vessels through areas occupied by these species and designated critical habitat. It is unlikely that these species will be exposed to stressors associated with the introduction of NAS from ship husbandry activities, as these will occur in ports and preferentially while vessels are in drydock. NAS introduced into waters of these ports during ship husbandry activities are not expected to spread to offshore habitats where they could interact with ESA-listed species and designated critical habitats. It is also unlikely that hull-fouling species would release from the hull during transit of vessels from one port to another or that they would be able to survive and propagate if they do detach. Therefore, we believe the potential effects of exposure to NAS to ESA-listed species and designated critical habitat that occur outside of any areas where vessels of the armed forces concentrate will be discountable.

Similarly, the performance standards for the majority of the discharges require that discharges not occur or are minimized in federally-protected waters, which include designated critical habitats based on the information in the BE. Performance standards for some discharges also require that they take place at least one mile from shore. This means these discharges will occur in deeper waters where mixing will take place rapidly thus diluting the discharge and limiting potential exposure of ESA-listed species and designated critical habitat to any contaminants contained in the discharge stream. While other discharges may occur in offshore and nearshore waters as vessels are in transit, these discharges will dilute rapidly as vessels are in motion and as discharges mix with ocean water. Therefore, we believe the potential effects of exposure to Batch Two UNDS discharges to ESA-listed species and designated critical habitat that occur outside of areas where vessels of the armed forces concentrate will be discountable.

Several other ESA-listed species are highly mobile and have large geographic ranges while others have more restricted ranges and, in some cases, occur in areas with a high volume of non-military vessel traffic. However, these species and their designated critical habitats are not residents of any of the RAAs or in other areas containing bases and other facilities regularly used by U.S. military vessels. Exposure of these species and designated critical habitat to Batch Two discharges would only occur during transit of either the vessels through areas occupied by these species and designated critical habitats or during movement of species through areas where military vessels are present. ESA-listed species that transit through areas where military vessels are concentrated could be exposed to NAS if the transit of these species coincides with ship husbandry activities resulting in the introduction of NAS to coastal waters. As stated above, NAS are not expected to spread outside the port and harbor areas where they are released. ESA-listed species that transit through areas when and if ship husbandry activities produce discharges of NAS to the water column could be briefly exposed to NAS but this transitory exposure is not expected to result in any changes in behavior or health of these ESA-listed species. Therefore, we believe the potential effects of exposure to NAS to ESA-listed species that transit through areas where vessels of the armed forces concentrate will be discountable. Designated critical habitats located in areas with only a transient presence of military vessels could also be exposed to NAS although, as stated above, we do not expect NAS to establish outside of RAAs and other areas where military vessels concentrate. If NAS were to spread outside an RAA, for example, we do not expect it would be to a degree that could alter the structure and function of designated critical habitat. Coles and Eldredge (2002) reviewed scientific literature for information regarding the occurrence and impacts of nonindigenous species from harbors, embayments, and coral reef surveys in the tropical Pacific. Coles and Eldredge (2002) found, for U.S. waters of

Apra Harbor, Guam, and Pearl Harbor and other harbors in O’ahu, Hawaii, that low percentages of nonnative species or species that could not be confirmed to be native were present with larger numbers in the most-used harbor areas. They found Inner Apra Harbor, which is dedicated to military use, had 27 nonindigenous species and 29 cryptogenic species (i.e., of unknown origin these species are not demonstrably native but information does not exist to confirm they were introduced), making up 6.7 percent of the total species in the harbor. In Outer Apra Harbor and island-wide, nonindigenous and cryptogenic species made up only 1.7 percent of the total species. In Hawaii, the nonindigenous and cryptogenic species in Pearl Harbor comprised 23 percent of the total number of species and 17 percent in harbors on the south and west shores of O’ahu, while Midway and Kaho’olawe had only 1.5 and 1 percent, respectively. However, with the exception of some invasive algae in Hawaii, results of studies indicate that the nonindigenous and cryptogenic species in tropical areas are relatively rare on coral reefs and do not appear to cause substantial negative effects (Coles and Eldredge 2002). Therefore, we believe the potential effects of exposure of designated critical habitat to NAS from the transient presence of military vessels is discountable.

In terms of other discharges that contain contaminants that could affect ESA-listed species if they are exposed, very few of the Batch Two discharges will occur while military vessels are in port and, when they do, these discharges are expected to meet performance standards designed to minimize both the discharges and their possible effects on environmental resources. Because these species are transient in areas where military vessels may be present or are present in areas without military bases and facilities, meaning military vessels are not regularly present, exposure to discharges, if it occurs, is infrequent and not expected to result in long-term consequences to animals. Similarly, while designated critical habitat in areas through which military vessels may transit may be exposed to Batch Two discharges, this exposure would be temporary and would be limited by the performance standards required for the various discharges. Because these critical habitats are outside any of the RAAs and other ports and harbors where vessels of the armed forces concentrate, even if discharges result in contaminant accumulation in sediments, these habitats would not be exposed to contaminants released into the water column during sediment resuspension. Therefore, we believe the potential effects of infrequent, temporary exposure to contaminants in Batch Two discharges from military vessels to ESA-listed species and designated critical habitat will be discountable.

There are some Pacific invertebrate species listed under the ESA, some of which are sessile benthic organisms during most of their life cycle. These include ESA-listed Pacific corals and black and white abalone. Only three of the ESA-listed Pacific coral species are reported in waters of Guam where military vessels may be present. None of the listed corals are reported in waters of Hawaii. After reviewing a number of benthic survey reports for Apra Harbor (Smith et al. 2009; Shafer-Nelson et al. 2016; Foster et al. 2007) , none of the ESA-listed species were documented during surveys, many of which were conducted where military vessels moor and transit.

Black abalone is found off the Western Coast of the U.S. from Point Arena (Mendocino County, California) to Northern Baja California. Inside this broad geographic range, black abalone mostly inhabits coastal and offshore island intertidal habitats on uncovered rough shores where bedrock offers profound, protective crevice shelter (Leighton 2005 as cited in Butler et al. 2009). Compared to other native species of abalone found along California and its coastal islands, black abalone bathymetrically inhabits shallower locations situated predominantly in rocky intertidal environments (Morris et al. 1980). Critical habitat has been designated for this species and includes rocky areas from mean high water to six m water depth in the Farallon, Channel, and Año Nuevo islands, as well as the California coastline from Del Mar Ecological Reserve south to Government Point (excluding some stretches, such as in Monterey Bay and between Cayucos and Montaña de Oros State Park) in northern and central California and between the Palos Verdes and Torrance border south to Los Angeles Harbor.

Major concentrations of white abalone historically occurred in water depths of 25 to 30 m and depth distribution since the collapse of the fishery has shifted toward deeper waters with white abalone most abundant and dense at depths of 40 to 50 m (Stierhoff et al. 2012). In addition, while the San Diego area is part of the species' range, the species is not reported within the harbor, appearing to prefer deeper waters along the mainland coast and offshore islands. Based on the review by Coles and Eldredge (2002), invasive species do not appear to be having an adverse effect on native species of Apra Harbor, which include corals. Additionally, the apparent lack of ESA-listed Pacific corals and black and white abalone in harbors where military vessels are housed means that exposure to Batch Two discharges would be limited to those that may occur as vessels transit through areas containing these species. Therefore, as for the other species in this section, we believe the potential effects of exposure to Batch Two discharges on ESA-listed Pacific corals, black and white abalone, and black abalone designated critical habitat will be discountable.

Other designated critical habitat areas for ESA-listed fish are in rivers that flow to the ocean along the coasts of Washington, Oregon, and/or California. These critical habitats are: chinook salmon California Coastal, Lower Columbia River, and Snake River Fall-Run ESUs; chum salmon Columbia River ESU; coho salmon Lower Columbia River, Oregon Coast, and Southern Oregon/Northern California ESUs; sockeye salmon Ozette Lake and Snake River ESUs; and steelhead trout Lower Columbia River, Northern California, South-Central California Coast, and Southern California DPSs. Military vessels are not expected to transit in these rivers and critical habitats but tidal exchanges could transport contaminants from Batch Two discharges into the coastal portions of these critical habitats if military vessels discharge while in transit along the coast. However, the performance standards for the majority of the discharges require that discharges not occur or are minimized in federally-protected waters, which include designated critical habitats, and some discharges are only allowed one mile or more from shore. Therefore, the likelihood that contaminants from Batch Two discharges be transported into these coastal critical habitats is extremely low and likely to be unmeasurable and thus insignificant.

In summary, for the following species and designated critical habitats, we believe the Phase II Batch Two UNDS rule is not reasonably likely to result in adverse effects:

- Cetaceans:
  - Species: blue whale, fin whale, bowhead whale, gray whale (Western North Pacific DPS), humpback whale (Western North Pacific, Central America, Arabian Sea, Cape Verde Islands/Northwest Africa, and Mexico DPSs), North Pacific right whale, North Atlantic right whale, Southern right whale, false killer whale (Main Hawaiian Islands insular DPS), sei whale, Beluga whale (Cook Inlet DPS), sperm whale, Bryde's whale (Gulf of Mexico subspecies), and Hector's dolphin (Maui's dolphin and South Island Hector's dolphin subspecies)
  - Critical habitat: North Pacific right whale, North Atlantic right whale, false killer whale (Main Hawaiian Islands insular DPS), and Cook Inlet beluga whale
- Pinnipeds:
  - Species: ringed seal (Arctic DPS), Guadalupe fur seal, Hawaiian monk seal, bearded seal (Beringia DPS), Mediterranean monk seal, and Steller sea lion (Western DPS)
  - Critical habitat: Arctic ringed seal (proposed), Hawaiian monk seal, and Steller sea lion
- Sea Turtles:
  - Species: green sea turtle (North Atlantic, South Atlantic, East Pacific, Central North Pacific, Central West Pacific, East Indian-West Pacific, Southwest Pacific, Central South Pacific, North Indian Ocean, Southwest Indian Ocean, and Mediterranean DPSs), hawksbill sea turtle, olive ridley sea turtle (Mexico's Pacific coast breeding populations and all other populations), loggerhead (South Atlantic Ocean, Southeast Indo-Pacific Ocean, Northeast Atlantic Ocean, North Pacific Ocean, South Pacific Ocean, North Indian Ocean, Southwest Indian Ocean, and Mediterranean Sea DPSs), Kemp's ridley sea turtle, and leatherback sea turtle
  - Critical habitat: North Atlantic DPS green sea turtle, hawksbill sea turtle, loggerhead sea turtle (Northwest Atlantic Ocean DPS), and leatherback sea turtle (Sandy Point, St. Croix; and West Coast)
- Fishes:
  - Species: Sakhalin sturgeon, Adriatic sturgeon, European sturgeon, Chinese sturgeon, dusky sea snake, smalltooth sawfish (U.S. and non-U.S. portions of range DPSs), largetooth sawfish, narrow sawfish, dwarf sawfish, green sawfish,

giant manta ray, oceanic whitetip shark, scalloped hammerhead shark (Central and Southwest Atlantic, Eastern Atlantic, Eastern Pacific, and Indo-West Pacific DPSs), daggernose shark, striped smoothhound shark, narrownose smoothhound shark, spiny angelshark, smoothback angelshark, sawback angelshark, Argentine angelshark, common angelshark, Brazilian guitarfish, blackchin guitarfish, common guitarfish, gulf grouper, island grouper, Nassau grouper, totoaba, sockeye salmon (Snake River and Ozette Lake ESUs), steelhead trout (Southern California, Upper Columbia River, Snake River Basin, Middle Columbia River, Lower Columbia River, Upper Willamette River, South-Central California Coast, and Northern California DPSs), chinook salmon (Upper Columbia River Spring-Run, Snake River Spring/Summer Run, Snake River Fall-Run, California Coast, Lower Columbia River, and Upper Willamette River ESUs), coho salmon (Southern Oregon/Northern California Coasts, Lower Columbia River, and Oregon Coast ESUs), chum salmon (Columbia River ESU), African coelacanth, and Pacific eulachon (Southern DPS)
    - Critical habitat: smalltooth sawfish (U.S. portion of range DPS), Southern DPS Pacific eulachon, chinook salmon (California Coastal, Lower Columbia River, and Snake River Fall-Run ESUs), chum salmon (Columbia River ESU), coho salmon (Central California Coast, Lower Columbia River, Oregon Coast, and Southern Oregon/Northern California ESUs), sockeye salmon (Ozette Lake and Snake River ESUs), and steelhead trout (Lower Columbia River, Northern California, South-Central California Coast, and Southern California DPSs)
- Invertebrates
    - Species: *Acropora globiceps*, *Acropora jacquelineae*, *Acropora lokani*, *Acropora pharaonis*, *Acropora retusa*, *Acropora rudis*, *Acropora speciosa*, *Acropora tenella*, *Anacropora spinosa*, *Euphyllia paradivisa*, *Isopora crateriformis*, *Montiplora australiensis*, *Pavona diffluens*, *Porites napopora*, *Seriatopora aculeata*, *Cantharellus noumeae*, black abalone white abalone, and chambered nautilus
    - Critical habitat: black abalone

Some of the designated critical habitat areas for ESA-listed fishes are in inland freshwater rivers. These designated critical habitats are: chinook salmon Upper Columbia River Spring-Run, Snake River Spring/Summer-Run, and Upper Willamette River ESUs; and steelhead trout Middle Columbia River, Snake River Basin, Upper Columbia River, and Upper Willamette River ESUs. Because of their location, there will be no effect to these habitats as a result of Batch Two discharges.

6.2 Status of Species and Critical Habitats Analyzed Further

This opinion examines the status of each species and critical habitat that may be adversely affected by the action. We determined that the following ESA-listed species and designated critical habitats warrant further analysis in this opinion because of their residency in RAAs with concentrations of vessels of the armed forces, meaning they have a high likelihood of exposure to UNDS Batch Two discharges: Southern Resident killer whale and designated critical habitat; loggerhead sea turtle Northwest Atlantic Ocean DPS; elkhorn and staghorn corals and designated critical habitat; lobed star, mountainous star, and boulder star corals; rough cactus coral; pillar coral; Johnson's seagrass and designated critical habitat; bocaccio Puget Sound/Georgia Basin DPS and designated critical habitat; yelloweye rockfish Puget Sound/Georgia Basin DPS and designated critical habitat; Atlantic sturgeon Gulf of Maine, New York Bight, Chesapeake, Carolina, and South Atlantic DPSs and designated critical habitat; gulf sturgeon and designated critical habitat; green sturgeon Southern DPS and designated critical habitat; shortnose sturgeon; Atlantic salmon Gulf of Maine DPS and designated critical habitat; Chinook salmon Central Valley Spring-Run, Puget Sound, and Sacramento Winter-Run ESUs and designated critical habitat; chum salmon Hood Canal Summer-Run ESU and designated critical habitat; coho salmon Central California Coast ESU and designated critical habitat; and steelhead trout California Central Valley, Central California Coast, and Puget Sound DPSs and designated critical habitat.

The evaluation of adverse effects in this opinion begins by summarizing the biology and ecology of those species that are reasonably likely to be adversely affected and what is known about their life histories in the action area and the condition of designated critical habitat within the applicable critical habitat unit and in the action area. The status is determined by the level of risk that the ESA-listed species and designated critical habitat face, based on parameters considered in documents such as recovery plans, status reviews, and listing decisions. This helps to inform the description of the species' current "reproduction, numbers or distribution" that is part of the jeopardy determination as described in 50 CFR §402.02. This section also examines the condition of critical habitat throughout the designated area (such as various watersheds and coastal and marine environments that make up the designated area) and discusses the condition and current function of designated critical habitat, including the essential physical and biological features that contribute to that conservation value of the critical habitat. More detailed information on the status and trends of these ESA-listed species, and their biology and ecology can be found in the listing regulations and critical habitat designations published in the Federal Register, status reviews, recovery plans, and on the NMFS Web sites: [http://www.nmfs.noaa.gov/pr/species/esa/listed.htm].

6.2.1 Southern Resident Killer Whale and Designated Critical Habitat

Killer whales are distributed worldwide, but populations are isolated by region and ecotype. Killer whales have been divided into DPSs on the basis of differences in genetics, ecology, morphology and behavior. The Southern Resident DPS of killer whale can be found along the Pacific Coast of the U.S. and Canada, and in the Salish Sea, Strait of Juan de Fuca, and Puget Sound (Figure 7).

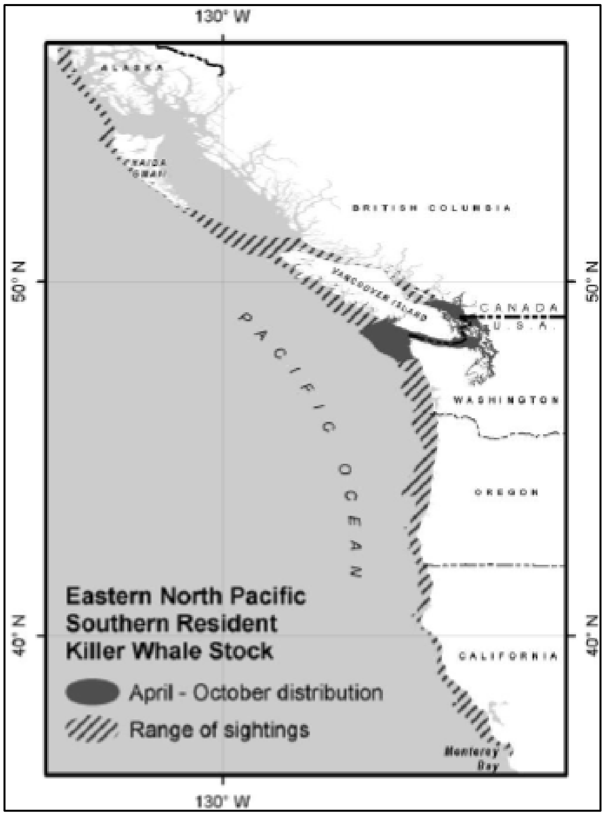

Figure 7. Map identifying the Range of the Endangered Southern Resident DPS of Killer Whale. Approximate April to October Distribution of the Southern Resident DPS of Killer Whale (Shaded Area) and Range of Sightings (Diagonal Lines; Carretta et al. 2017)

Killer whales are odontocetes and the largest delphinid species with black coloration on their dorsal side and white undersides and patches near the eyes. They also have a highly variable gray or white saddle behind the dorsal fin.

We used information available in the final rule, the Recovery Plan (NMFS 2008b), the 2016 5-year review (NMFS 2016f) and the recent stock Assessment report (Carretta et al. 2017) to summarize the life history, population dynamics and status of this species, as follows.

Life History

Southern Resident DPS of killer whales are geographically, matrilineally, and behaviorally distinct from other killer whale populations. The Southern Resident DPS includes three large, stable pods (J, K, and L), which occasionally interact (Parsons et al. 2009). Most mating occurs outside natal pods, during temporary associations of pods, or as a result of the temporary dispersal of males (Pilot et al. 2010). Males become sexually mature at ten to 17 years of age. Females reach maturity at 12 to 16 years of age and produce an average of 5.4 surviving calves during a reproductive life span of approximately 25 years. Mothers and offspring maintain highly stable, life-long social bonds, and this natal relationship is the basis for a matrilineal social structure. They prey upon salmonids, especially chinook salmon (Hanson et al. 2010).

Population Dynamics

The most recent abundance estimate for the Southern Resident DPS is 81 whales in 2015 (Carretta et al. 2017; 80 whales in 2016)[5]. This represents a decline from just a few years ago, when in 2012, there were 85 whales. Population abundance has fluctuated over time with a maximum of approximately 100 whales in 1995 (Carretta et al. 2017), with an increase between 1974 and 1993, from 76 to 93 individuals. As compared to stable or growing populations, the DPS reflects lower fecundity and has demonstrated little to no growth in recent decades (NMFS 2016f).

For the period between 1974 and the mid-1990s, when the population increased from 76 to 93 animals, the population growth rate was 1.8 percent. More recent data indicate the population is now in decline (Carretta et al. 2017).

After thorough genetic study, the Biological Review Team concluded that Southern Resident DPS of killer whales were discrete from other killer whale groups (NMFS 2008b) . Despite the fact that their ranges overlap, Southern Resident DPS of killer whales do not intermix with Northern Resident killer whales. Southern Resident DPS of killer whales consist of three pods, called J, K, and L. Low genetic diversity within a population is believed to be in part due to the matrilineal social structure (NMFS 2008b) .

Distribution

Southern Resident DPS of killer whales occur in the inland waterways of Puget Sound, Strait of Juan de Fuca, and Southern Georgia Strait during the spring, summer and fall. During the winter, they move to coastal waters primarily off Oregon, Washington, California, and British Columbia (Figure 7).

[5] http://www.orcanetwork.org/Main/index.php?categories_file=Births%20and%20Deaths; accessed 11/15/2016

In spring and summer months, Southern Resident killer whales are frequently seen in the San Juan Islands region with intermittent sightings in Puget Sound (Whale Museum 2012). In the fall and early winter months, Southern Resident killer whales are seen more frequently in Puget Sound, where returning chum and chinook salmon are concentrated (Osborne et al. 1988). In winter, they spend progressively less time in the inland marine waters and more time off the coast of Washington, Oregon, and California (Black 2011). Southern Resident killer whales have not been reported in Hood Canal or Dabob Bay since 1995 according to NMFS (2008b). Southern Resident killer whales (J pod) were historically documented in Hood Canal by sound recordings in 1958 (Ford 1991), a photograph from 1973, and anecdotal accounts of historical use, but the latter sightings may be transient whales (NMFS 2008b). Transients and Southern Resident killer whales have been observed in southern Puget Sound in the Carr Inlet area (NMFS 2015a). Southern Resident killer whales are not observed frequently near existing Naval bases. In terms of critical habitat for this species, while military facilities are not included in the designation of critical habitat, prey species are included as an essential feature.

Vocalization and Hearing

Killer whales have advanced vocal communication and also use vocalizations to aid in navigation and foraging (NMFS 2008b). Their vocalizations typically have both a low frequency component (250 Hertz [Hz] to 1.5 kilohertz [kHz]) and a high frequency component (five to 12 kHz; Holt 2008). Killer whale vocalizations consist of three main types, echolocation clicks, which are primarily used for navigation and foraging, and tonal whistles and pulse calls, which are thought to be used for communication (NMFS 2008b). Individual Southern Resident DPS of killer whale pods have distinct call repertoires, with each pod being recognizable by its acoustic dialect (NMFS 2008b). Killer whale hearing is one of the most sensitive of any odontocete, with a hearing range of one to 120 kHz, with the most sensitive range being between 18 and 42 kHz range (Szymanski et al. 1999).

Status

The Southern Resident DPS of killer whale was listed as endangered in 2005 in response to the population decline from 1996 through 2001, small population size, and reproductive limitations (i.e., few reproductive males and delayed calving). Current threats to its survival and recovery include contaminants, ship traffic, and reduction in prey availability. Chinook salmon populations have declined due to degradation of habitat, hydrology issues, harvest, and hatchery introgression; such reductions may require an increase in foraging effort. In addition, these prey contain environmental pollutants. These contaminants become concentrated at higher trophic levels and may lead to immune suppression or reproductive impairment. The inland waters of Washington and British Columbia support a large whale watch industry, commercial shipping, and recreational boating; these activities generate underwater noise, which may mask whales' communication or interrupt foraging. The factors that originally endangered the species persist throughout its habitat: contaminants, ship traffic, and reduced prey. The DPS's resilience to future perturbation is reduced as a result of its small population size. The recent decline, unstable population status, and population structure (i.e., few reproductive age males and non-calving adult females) continue to be causes for concern. The relatively low number of individuals in this population makes it difficult to resist or recover from natural spikes in mortality, including disease and fluctuations in prey availability.

Designated Critical Habitat

On November 29, 2006, NMFS designated critical habitat for the Southern Resident DPS of killer whale. The critical habitat consists of approximately 6,630 km^2 (1,933 nmi^2) in three areas: the Summer Core Area in Haro Strait and waters around the San Juan Islands; Puget Sound; and the Strait of Juan de Fuca (Figure 8). It provides the following physical and biological features (PBFs; formerly referred to as primary constituent elements or PCEs) essential to the conservation of Southern Resident DPS of killer whales: water quality to support growth and development; prey species of sufficient quantity, quality and availability to support individual growth, reproduction and development, as well as overall population growth; and inter-area passage conditions to allow for migration, resting, and foraging. The Puget Sound Naval Shipyard at Naval Base Kitsap was excluded from critical habitat designation for Southern Resident DPS killer whale.

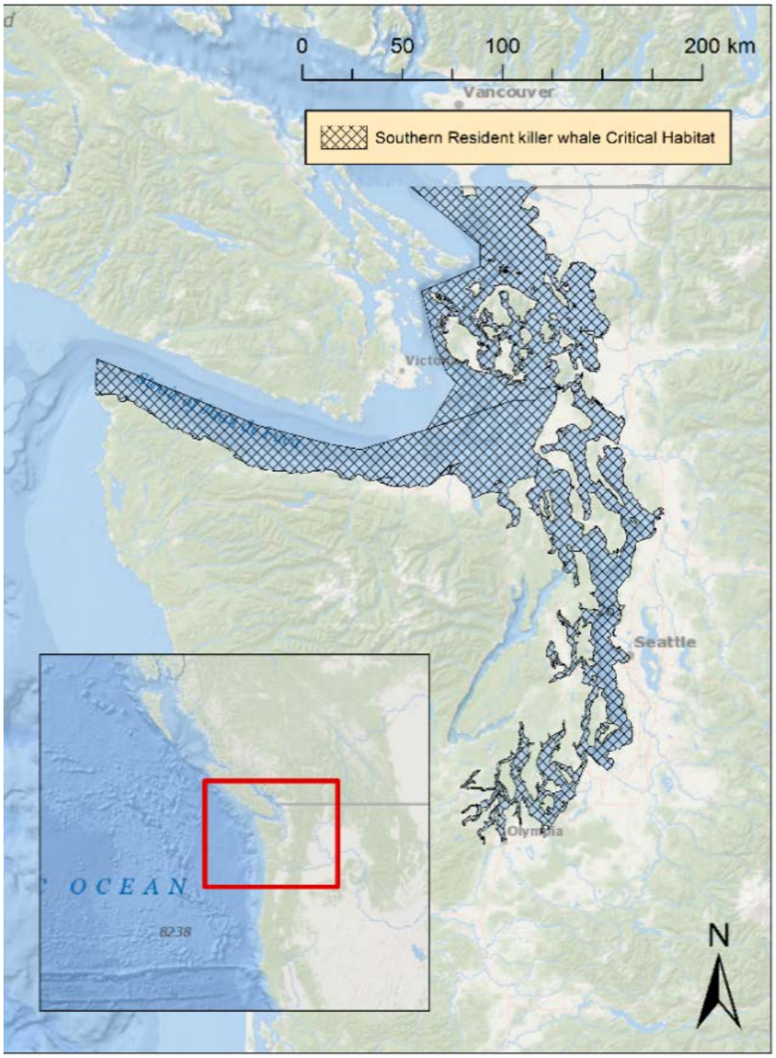

Figure 8. Map Depicting Designated Critical Habitat for the Endangered Southern Resident DPS of Killer Whale

Recovery Goals

See the 2008 Final Recovery Plan for the Southern Resident DPS of killer whale for complete downlisting/delisting criteria for each of the following recovery goals.

1. Prey Availability: Support salmon restoration efforts in the region including habitat, harvest and hatchery management considerations and continued use of existing NMFS authorities under the ESA and Magnuson-Stevens Fishery Conservation and Management Act to ensure an adequate prey base
2. Pollution/Contamination: Clean up existing contaminated sites, minimize continuing inputs of contaminants harmful to killer whales, and monitor emerging contaminants.

3. Ship Effects: Continue with evaluation and improvement of guidelines for ship activity near Southern Resident DPS of killer whales and evaluate the need for regulations or protected areas.
4. Oil Spills: Prevent oil spills and improve response preparation to minimize effects on Southern Resident DPS and their habitat in the event of a spill.
5. Acoustic Effects: Continue agency coordination and use of existing ESA and Marine Mammal Protection Act mechanisms to minimize potential impacts from anthropogenic sound.
6. Education and Outreach: Enhance public awareness, educate the public on actions they can participate in to conserve killer whales and improve reporting of Southern Resident DPS killer whale sightings and strandings.
7. Response to Sick, Stranded, Injured Killer Whales: Improve responses to live and dead killer whales to implement rescues, conduct health assessments, and determine causes of death to learn more about threats and guide overall conservation efforts.
8. Transboundary and Interagency Coordination: Coordinate monitoring, research, enforcement, and complementary recovery planning with Canadian agencies, and Federal and State partners.
9. Research and Monitoring: Conduct research to facilitate and enhance conservation efforts. Continue the annual census to monitor trends in the population, identify individual animals, and track demographic parameters.

6.2.2 Loggerhead Sea Turtle Northwest Atlantic Ocean Distinct Population Segment

Loggerhead sea turtles are circumglobal and are found in the temperate and tropical regions of the Indian, Pacific and Atlantic Oceans (Figure 9).

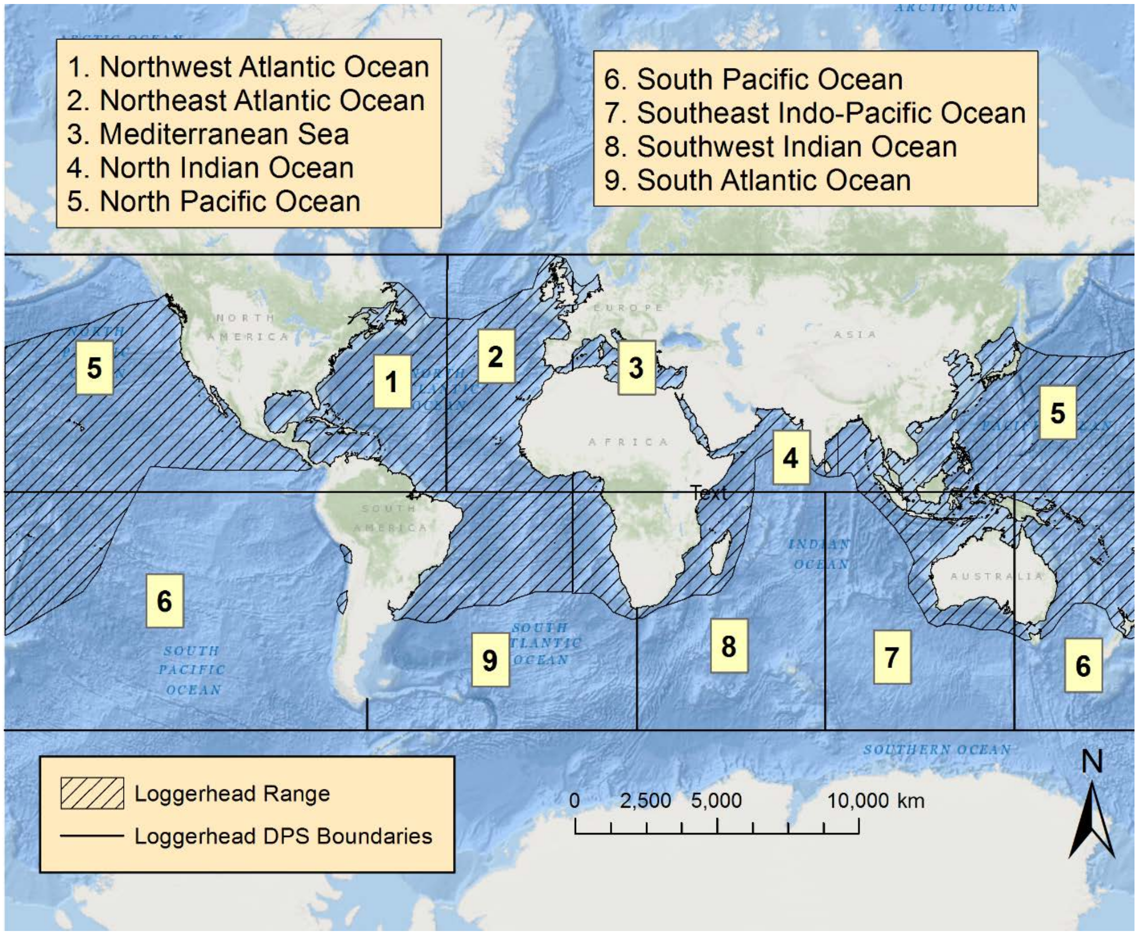

Figure 9. Map Identifying the Range and DPS Boundaries of the Loggerhead Sea Turtle

The loggerhead sea turtle is distinguished from other turtles by its large head and powerful jaws. The species was first listed as threatened under the ESA in 1978. On September 22, 2011, the NMFS designated nine DPSs of loggerhead sea turtles: South Atlantic Ocean and Southwest Indian Ocean as threatened as well as Mediterranean Sea, North Indian Ocean, North Pacific Ocean, Northeast Atlantic Ocean, Northwest Atlantic Ocean, South Pacific Ocean, and Southeast Indo-Pacific Ocean as endangered. Recent ocean-basin scale genetic analysis supports this conclusion, with additional differentiation apparent based upon nesting beaches (Shamblin et al. 2014). The DPSs considered in this opinion are the Northwest Atlantic Ocean and the North Pacific Ocean.

We used information available in the 2009 status review (Conant et al. 2009) and the final listing rule to summarize the life history, population dynamics and status of the species, as follows.

Life History

Mean age at first reproduction for female loggerhead sea turtles is thirty years. Females lay an average of three clutches per season. The annual average clutch size is 112 eggs per nest. The average remigration interval is 2.7 years. Nesting occurs on beaches, where warm, humid sand temperatures incubate the eggs. Temperature determines the sex of the turtle during the middle of the incubation period. Turtles spend the post-hatchling stage in pelagic waters. The juvenile stage is spent first in the oceanic zone and later in the neritic zone (i.e., coastal waters). Coastal waters provide important foraging habitat, inter-nesting habitat, and migratory habitat for adult loggerheads.

Population Dynamics

There is general agreement that the number of nesting females provides a useful index of the species' population size and stability at this life stage, even though there are doubts about the ability to estimate the overall population size (Bjorndal et al. 2005). Adult nesting females often account for less than one percent of total population numbers. The global abundance of nesting female loggerhead turtles is estimated at 43,320 to 44,560 (Spotila 2004).

Northwest Atlantic Ocean DPS

Using a stage/age demographic model, the adult female population size of the DPS is estimated at 20,000 to 40,000 females and 53,000 to 92,000 nests annually (NMFS-SEFSC 2009). Based on genetic information, the Northwest Atlantic Ocean DPS is further categorized into five recovery units corresponding to nesting beaches. These are: Northern Recovery Unit, Peninsular Florida Recovery Unit, Dry Tortugas Recovery Unit, Northern Gulf of Mexico Recovery Unit, and the Greater Caribbean Recovery Unit.

The Northern Recovery Unit, from North Carolina to northeastern Florida, is the second largest nesting aggregation in the DPS with an average of 5,215 nests from 1989 to 2009 and approximately 1,272 nesting females (NMFS and USFWS 2008). The Peninsular Florida Recovery Units hosts more than 10,000 females nesting annually, which constitutes 87 percent of all nesting effort in the DPS (Ehrhart et al. 2003). The Greater Caribbean Recovery Unit encompasses nesting subpopulations in Mexico to French Guiana, the Bahamas, and the Lesser and Greater Antilles. The majority of nesting for this recovery unit occurs on the Yucatán Peninsula in Quintana Roo, Mexico with 903 to 2,331 nests annually (Zurita et al. 2003). Other significant nesting sites are found throughout the Caribbean such as in Cuba with approximately 250 to 300 nests reported annually (Ehrhart et al. 2003) and in the Bahamas on Cay Sal with over 100 nests annually (NMFS and USFWS 2008). The Dry Tortugas Recovery Unit includes Key West, Florida, and all islands west of it. The only available data for the nesting subpopulation on Key West comes from a census conducted from 1995 to 2004 (excluding 2002), which provided a mean of 246 nests per year, or about 60 nesting females (NMFS and USFWS 2008). The Gulf of Mexico Recovery Unit has between 100 to 999 nesting females annually and a mean of 910 nests per year.

Population Growth Rate

Available information on the population growth rates and trends for the Northwest Atlantic DPS is presented below.

Northwest Atlantic Ocean DPS

Using a stage/age demographic model, the adult female population size of the DPS is estimated at 20,000 to 40,000 females, and 53,000 to 92,000 nests annually (NMFS-SEFSC 2009).

The Peninsular Florida Recovery Unit hosts more than 10,000 females nesting annually, which constitutes eighty-seven percent of all nesting effort in the DPS (Ehrhart et al. 2003). Nest counts taken at index beaches in Peninsular Florida show a significant decline in loggerhead nesting from 1989 to 2006, most likely attributed to mortality of oceanic-stage loggerheads caused by fisheries bycatch (Witherington et al. 2009). Loggerhead nesting on the Archie Carr National Wildlife Refuge (representing individuals of the Peninsular Florida subpopulation) has fluctuated over the past few decades. There was an average of 9,300 nests throughout the 1980s, with the number of nests increasing into the 1990s until it reached an all-time high in 1998, with 17,629 nests. From that point, the number of loggerhead nests at the Refuge have declined steeply to a low of 6,405 in 2007, increasing again to 15,539, still a lower number of nests than in 1998 (Bagley et al. 2013).

The Northern Recovery Unit, from North Carolina to northeastern Florida, and is the second largest nesting aggregation in the DPS, with an average of 5,215 nests from 1989 to 2008, and approximately 1,272 nesting females(NMFS and USFWS 2008). For the Northern recovery unit, nest counts at loggerhead nesting beaches in North Carolina, South Carolina and Georgia declined at 1.9 percent annually from 1983 to 2005 (NMFS and USFWS 2007).

The Gulf of Mexico Recovery Unit has between one hundred to 999 nesting females annually, and a mean of 910 nests per year. The nesting subpopulation in the Florida panhandle has exhibited a significant declining trend from 1995 to 2005 (Conant et al. 2009; NMFS and USFWS 2007). Recent model estimates predict an overall population decline of 17 percent for the St. Joseph Peninsula, Florida subpopulation of the Northern Gulf of Mexico recovery unit (Lamont et al. 2014).

The Greater Caribbean Recovery Unit encompasses nesting subpopulations in Mexico to French Guiana, the Bahamas, and the Lesser and Greater Antilles. The majority of nesting for this recovery unit occurs on the Yucatán peninsula, in Quintana Roo, Mexico, with 903 to 2,331 nests annually (Zurita et al. 2003). Other significant nesting sites are found throughout the

Caribbean, and including Cuba, with approximately 250 to 300 nests annually (Ehrhart et al. 2003) and over one hundred nests annually in Cay Sal in The Bahamas (NMFS and USFWS 2008).

The population growth rate for each of the four of the recovery units for the Northwest Atlantic Ocean DPS (Peninsular Florida, Northern, Northern Gulf of Mexico, and Greater Caribbean) all exhibit negative growth rates (Conant et al. 2009).

Genetic Diversity

There are nine loggerhead DPSs, which are geographically separated and genetically isolated, as indicated by genetic, tagging, and telemetry data. Our understanding of the genetic diversity and population structure of the different loggerhead DPSs is being refined as more studies examine samples from a broader range of specimens using longer mitochondrial DNA sequences.

Northwest Atlantic Ocean DPS

Based on genetic analysis of nesting subpopulations, the Northwest Atlantic Ocean DPS is further divided into five recovery units: Northern, Peninsular Florida, Dry Tortugas, Northern Gulf of Mexico, and Greater Caribbean (Conant et al. 2009). A more recent analysis using expanded mitochondrial DNA sequences revealed that rookeries from the Gulf and Atlantic coasts of Florida are genetically distinct, and that rookeries from Mexico’s Caribbean coast express high haplotype diversity (Shamblin et al. 2014). Furthermore, the results suggest that the Northwest Atlantic Ocean DPS should be considered as ten management units: (1) South Carolina and Georgia, (2) central eastern Florida, (3) southeastern Florida, (4) Cay Sal, Bahamas, (5) Dry Tortugas, Florida, (6) southwestern Cuba, (7) Quintana Roo, Mexico, (8) southwestern Florida, (9) central western Florida, and (10) northwestern Florida (Shamblin et al. 2012).

Distribution

Loggerheads are circumglobal, occurring throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian oceans, returning to their natal region for mating and nesting. Adults and sub-adults occupy nearshore habitat. While in their oceanic phase, loggerheads undergo long migrations using ocean currents. Individuals from multiple nesting colonies can be found on a single feeding ground.

Northwest Atlantic Ocean DPS

Loggerhead hatchlings from the western Atlantic disperse widely, most likely using the Gulf Stream to drift throughout the Atlantic Ocean. Mitochondrial DNA evidence demonstrates that juvenile loggerheads from southern Florida nesting beaches comprise the vast majority (71 to 88 percent) of individuals found in foraging grounds throughout the western and eastern Atlantic: Nicaragua, Panama, Azores and Madeira, Canary Islands, Gulf of Mexico and Brazil (Masuda 2010).

The loggerhead is the most common sea turtle in the southeastern U.S. Loggerhead sea turtles are not common in the U.S. Caribbean where there are only infrequent reports of these animals in stranding data and reports of nesting by two females on Buck Island, St. Croix in the early 2000's and the east coast of Puerto Rico in the late 1990's. After departing the oceanic zone, neritic (neritic refers to the inshore marine environment from the surface to the sea floor where water depths do not exceed 200 m) juvenile loggerheads in the Northwest Atlantic inhabit continental shelf waters from Cape Cod Bay, Massachusetts, south through Florida, The Bahamas, Cuba, and the Gulf of Mexico. In the U.S., estuarine waters, including areas such as Long Island Sound, Chesapeake Bay, Pamlico and Core Sounds, Mosquito and Indian River Lagoons, Biscayne Bay, Florida Bay, and numerous embayments fringing the Gulf of Mexico, comprise important inshore habitat. Along the Atlantic and Gulf of Mexico shoreline, essentially all shelf waters are inhabited by loggerheads. Habitat preferences of Northwest Atlantic non-nesting adult loggerheads in the neritic zone differ from the juvenile stage in that relatively enclosed, shallow water estuarine habitats with limited ocean access are less frequently used. Areas such as Pamlico Sound and the Indian River Lagoon in the U.S., regularly used by juveniles, are only rarely frequented by adult loggerheads. In comparison, estuarine areas with more open ocean access, such as Chesapeake Bay in the U.S. mid-Atlantic, are also regularly used by juveniles, as well as by adults primarily during warmer seasons. Shallow water habitats with large expanses of open ocean access, such as Florida Bay, provide year-round resident foraging areas for significant numbers of male and female adult loggerheads. Offshore, adults primarily inhabit continental shelf waters, from New York south through Florida, The Bahamas, Cuba, and the Gulf of Mexico. Seasonal use of mid-Atlantic shelf waters, especially offshore New Jersey, Delaware, and Virginia during summer months, and offshore shelf waters, such as off the North Carolina coast. Shelf waters along the west Florida coast, The Bahamas, Cuba, and the Yucatán Peninsula have been identified as resident areas for adult female loggerheads that nest in Florida.

Status

Once abundant in tropical and subtropical waters, loggerhead sea turtles worldwide exist at a fraction of their historical abundance, as a result of over-exploitation. Globally, egg harvest, the harvest of females on nesting beaches and directed hunting of turtles in foraging areas remain the greatest threats to their recovery. In addition, bycatch in drift-net, long-line, set-net, pound-net and trawl fisheries kill thousands of loggerhead sea turtles annually. Increasing coastal development (including beach erosion and re-nourishment, construction and artificial lighting) threatens nesting success and hatchling survival. On a regional scale, the different DPSs experience these threats as well, to varying degrees. Differing levels of abundance combined with different intensities of threats and effectiveness of regional regulatory mechanisms make each DPS uniquely susceptible to future perturbations.

Northwest Atlantic Ocean DPS

Due to declines in nest counts at index beaches in the U.S. and Mexico, and continued mortality of juveniles and adults from fishery bycatch, the Northwest Atlantic Ocean DPS is at risk and likely to decline in the foreseeable future (Conant et al. 2009).

Designated Critical Habitat

Northwest Atlantic Ocean DPS

NMFS has designated critical habitat for the Northwest Atlantic Ocean DPS loggerhead sea turtles. On July 10, 2014, NMFS and FWS designated critical habitat for the Northwest Atlantic Ocean DPS loggerhead sea turtles along the U.S. Atlantic and Gulf of Mexico coasts from North Carolina to Mississippi. These areas contain one or a combination of nearshore reproductive habitat, winter area, breeding areas, and migratory corridors. The critical habitat is categorized into 38 occupied marine areas and 685 miles of nesting beaches (Figure 10).

The PBFs (formerly *primary constituent elements*) identified for the different habitat types include waters adjacent to high density nesting beaches, waters with minimal obstructions and manmade structures, high densities of reproductive males and females, appropriate passage conditions for migration, conditions that support *Sargassum* habitat, available prey, and sufficient water depth and proximity to currents to ensure offshore transport of post-hatchlings.

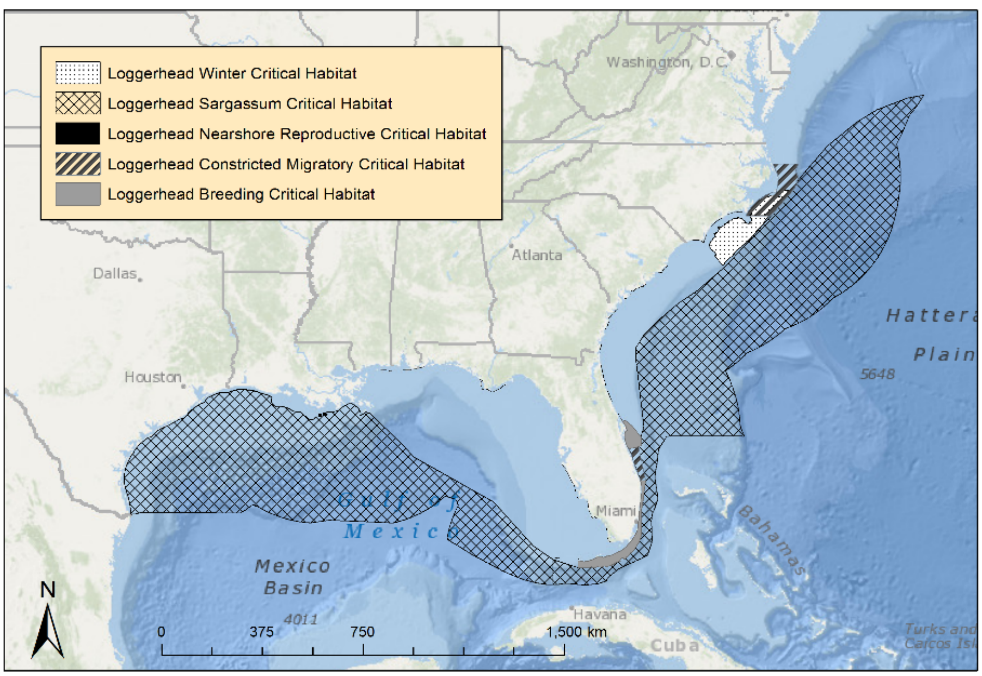

Figure 10. Map Depicting Loggerhead Turtle Designated Critical Habitat

Recovery Goals

See the 2008 Final Recovery Plan (NMFS and USFWS 2008) for the Northwest Atlantic Population of Loggerheads for complete down listing/delisting criteria for each of the following recovery objectives:

1. Ensure that the number of nests in each recovery unit is increasing and that this increase corresponds to an increase in the number of nesting females.
2. Ensure the in-water abundance of juveniles in both neritic and oceanic habitats is increasing and is increasing at a greater rate than strandings of similar age classes.
3. Manage sufficient nesting beach habitat to ensure successful nesting.
4. Manage sufficient feeding, migratory and inter-nesting marine habitats to ensure successful growth and reproduction.
5. Eliminate legal harvest.
6. Implement scientifically based nest management plans.
7. Minimize nest predation.
8. Recognize and respond to mass/unusual mortality or disease events appropriately.
9. Develop and implement local, state, Federal and international legislation to ensure long-term protection of loggerheads and their terrestrial and marine habitats.
10. Minimize bycatch in domestic and international commercial and artisanal fisheries.
11. Minimize trophic changes from fishery harvest and habitat alteration.
12. Minimize marine debris ingestion and entanglement.
13. Minimize ship strike mortality.

6.2.3 Elkhorn and Staghorn Corals and Designated Critical Habitat

Elkhorn and staghorn corals occur throughout the Caribbean, Gulf of Mexico, and southwestern Atlantic (Figure 11).

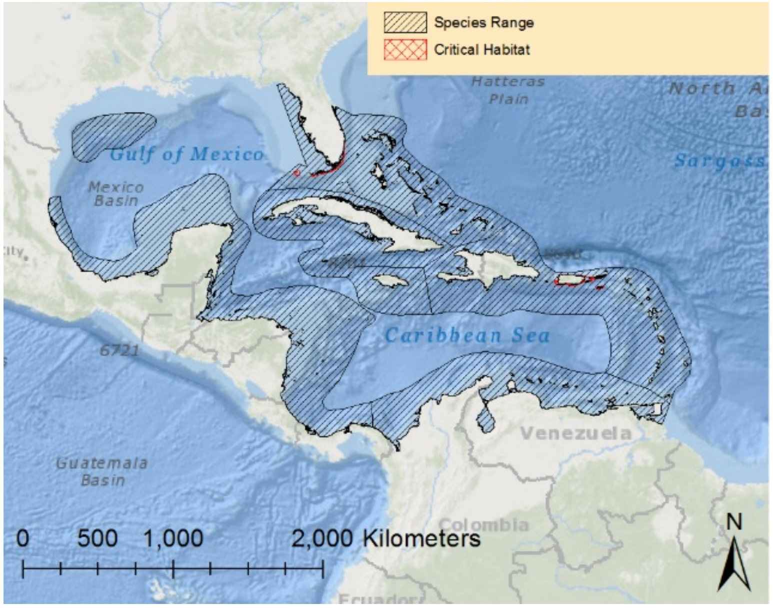

Figure 11. Range Map for Elkhorn and Staghorn Corals

Elkhorn and staghorn coral were listed as threatened under the ESA in May 2006 (71 FR 26852). In December 2012, NMFS proposed changing their status from threatened to endangered (77 FR 73219). On September 10, 2014, NMFS determined that elkhorn and staghorn coral should remain listed as threatened (79 FR 53851).

Elkhorn coral colonies have frond-like branches, which appear flattened to near round, and typically radiate out from a central trunk and angle upward. Branches are up to approximately 20 in (50 cm) wide and range in thickness from about 1.5-2 in (four to five cm). Individual colonies can grow to at least 6.5 ft (two m) in height and 13 ft (four m) in diameter (*Acropora* Biological Review Team 2005). Colonies of elkhorn coral can grow in nearly single-species, dense stands and form an interlocking framework known as thickets.

Staghorn coral is characterized by antler-like colonies with straight or slightly curved, cylindrical branches. The diameter of branches ranges from 0.1-2 in (0.25-5 cm; Lirman et al. 2010), and linear branch growth rates have been reported to range between 1.2-4.5 in (3-11.5 cm) per year (*Acropora* Biological Review Team 2005). The species can exist as isolated branches, individual colonies up to about five ft (1.5 m) diameter, and thickets comprised of multiple colonies that are difficult to distinguish from one another (*Acropora* Biological Review Team 2005).

Information on elkhorn coral status and populations dynamics is spotty throughout its range. Information on staghorn coral status and populations dynamics is infrequently documented throughout its range. Comprehensive and systematic census and monitoring have not been conducted for these species. Thus, the status and populations dynamics of elkhorn and staghorn corals must be inferred from the few locations were data exist.

Life History

Relative to other corals, elkhorn and staghorn corals have a high growth rate that have allowed acroporid reef growth to keep pace with past changes in sea level (Fairbanks 1989). Growth rates of staghorn coral, measured as skeletal extension of the end of branches, range from approximately 2-4 in (4-11 cm) per year (*Acropora* Biological Review Team 2005). However, growth rates in Curaçao have been reported to be slower today than they were several decades ago (Brainard et al. 2011). Annual linear extension has been found to be dependent on the size of the colony, and new recruits and juveniles typically grow at slower rates. Additionally, stressed colonies and fragments may also exhibit slower growth.

Elkhorn and staghorn corals are hermaphroditic broadcast spawning[6] species that reproduce sexually after the full moon of July, August, and/or September, depending on location and timing of the full moon (*Acropora* Biological Review Team 2005). Split spawning of elkhorn coral (spawning over a two month period) has been reported from the Florida Keys (Fogarty et al. 2012). Staghorn coral may also split spawning over the course of more than one lunar cycle (Vargas-Angel et al. 2006; Szmant 1986). The estimated size at sexual maturity for elkhorn coral is approximately 250 in^2 (1,600 cm^2), and growing edges and encrusting base areas are not fertile (Soong and Lang 1992). Larger colonies have higher fecundity per unit area, as do the upper branch surfaces (Soong and Lang 1992). The estimated size at sexual maturity for staghorn coral is approximately six in (17 cm) branch length, and large colonies produce proportionally more gametes than small colonies (Soong and Lang 1992). Basal and branch tip tissue is not fertile (Soong and Lang 1992).

Sexual recruitment rates are low, and these species are generally not observed in coral settlement studies in the field. Rates of post-settlement mortality of elkhorn coral after nine months are high based on settlement experiments (Szmant and Miller 2005). Laboratory studies have found that certain species of crustose-coralline algae facilitate larval settlement and post-settlement survival (Ritson-Williams et al. 2010) of both species. Laboratory experiments with elkhorn coral have shown that some individuals (i.e., genotypes) are sexually incompatible (Baums 2013) and that the proportion of eggs fertilized increases with higher sperm concentration (Fogarty et al. 2012). Experiments using elkhorn gametes collected in Florida and Belize showed that Florida corals had lower fertilization rates than those from Belize, possibly due to genotype incompatibilities (Fogarty et al. 2012).

Reproduction of elkhorn and staghorn coral occurs primarily through asexual fragmentation that produces multiple colonies that are genetically identical (Bak and Criens 1982; Highsmith 1982; Lirman 2000; Miller et al. 2007; Wallace 1985). Storms can be a method of producing fragments

[6] Simultaneously containing both sperm and eggs, which are released into the water column for fertilization.

to establish new colonies (Fong and Lirman 1995). Fragmentation is an important mode of reproduction in many reef-building corals, especially for branching species such as elkhorn and staghorn coral (Highsmith 1982; Lirman 2000; Wallace 1985). However, in the Florida Keys where populations have declined, there have been reports of failure of asexual recruitment due to high fragment mortality after storms (Porter et al. 2012; Williams and Miller 2010; Williams et al. 2008).

The combination of relatively rapid skeletal growth rates and frequent asexual reproduction by fragmentation can enable effective competition within, and domination of, elkhorn coral in reef-high-energy environments such as reef crests and staghorn coral in portions of the reef flat and reef front. Rapid skeletal growth rates and frequent asexual reproduction by fragmentation facilitate potential recovery from disturbances when environmental conditions permit (Highsmith 1982; Lirman 2000). However, low sexual reproduction can lead to reduced genetic diversity and limits the capacity to repopulate sites distant from the parent for both species.

Population Dynamics

There appears to be two distinct populations of elkhorn coral. Genetic samples from 11 locations throughout the Caribbean indicate that elkhorn coral populations in the eastern Caribbean (St. Vincent and the Grenadines, USVI, Curaçao, and Bonaire) have had little or no genetic exchange with populations in the western Atlantic and western Caribbean (Bahamas, Florida, Mexico, Panama, Navassa, and Puerto Rico; Baums et al. 2005a). While Puerto Rico is more closely connected with the western Caribbean, it is an area of mixing with contributions from both regions (Baums et al. 2005b). Models suggest that the Mona Passage between the Dominican Republic and Puerto Rico acts as a filter for larval dispersal and gene flow between the eastern Caribbean and western Caribbean (Baums et al. 2006b).

The western Caribbean is characterized by genetically depauperate populations with lower densities (0.13 ± 0.08 colonies per m^2), while denser (0.30 ± 0.21 colonies per m2), genotypically rich stands characterize the eastern Caribbean (Baums et al. 2006a). Baums et al. (2006b) concluded that the western Caribbean had higher rates of asexual recruitment and that the eastern Caribbean had higher rates of sexual recruitment. They postulated these geographic differences in the contribution of reproductive modes to population structure may be related to habitat characteristics, possibly the amount of shelf area available.

Genotypic diversity is highly variable. At two sites in the Florida Keys, only one genotype per site was detected out of 20 colonies sampled at each site (Baums et al. 2005a). In contrast, all 15 colonies sampled in Navassa had unique genotypes (Baums et al. 2006b). Some sites have relatively high genotypic diversity such as in Los Roques, Venezuela (118 unique genotypes out of 120 samples; (Zubillaga et al. 2008) and in Bonaire and Curaçao (18 genotypes of 22 samples and 19 genotypes of 20 samples, respectively; Baums et al. 2006a). In the Bahamas, about one third of the sampled colonies were unique genotypes, and in Panama between 24 and 65 percent of the sampled colonies had unique genotypes, depending on the site (Baums et al. 2006a).

A genetic study found significant population structure in Puerto Rico locations (Mona Island, Desecheo Island, La Parguera) both between reefs and between locations; population structure in La Parguera suggests restriction of gene flow between some reefs in close proximity (Garcia Reyes and Schizas 2010). A more-recent study provided additional detail on the genetic structure of elkhorn coral in Puerto Rico, as compared to Curaçao, the Bahamas, and Guadeloupe that found unique genotypes in 75 percent percent of the samples with high genetic diversity (Mège et al. 2014). The recent results support two separate populations of elkhorn coral in the eastern Caribbean and western Caribbean; however, there is less evidence for separation at Mona Passage, as found by Baums et al. (2006a).

Elkhorn coral was historically one of the dominant species on Caribbean reefs, forming large, monotypic thickets and giving rise to the nominal distinct zone in classical descriptions of Caribbean reef morphology (Goreau 1959). Mass mortality, apparently from white-band disease (Aronson and Precht 2001), spread throughout the Caribbean in the mid-1970s to mid-1980s and precipitated widespread and radical changes in reef community structure (Brainard et al. 2011). This mass mortality occurred throughout the range of the species within all Caribbean countries and archipelagos, even on reefs and banks far from localized human influence (Aronson and Precht 2001; Wilkinson 2008). In addition, continuing coral mortality from periodic acute events such as hurricanes, disease outbreaks, and mass bleaching events added to the decline of elkhorn coral (Brainard et al. 2011). In locations where historic quantitative data are available (Florida, Jamaica, USVI), there was a reduction of greater than 97 percent between the 1970s and early 2000s in elkhorn coral populations (*Acropora* Biological Review Team 2005).

Since the 2006 listing of elkhorn coral, continued population declines have occurred in some locations with certain populations of elkhorn coral decreasing up to an additional 50 percent or more (Colella et al. 2012; Lundgren and Hillis-Starr 2008; Muller et al. 2008; Rogers and Muller 2012; Williams et al. 2008). In addition, Williams et al. (2008) reported asexual recruitment failure between 2004 and 2007 in the upper Florida Keys after a major hurricane season in 2005; less than 5 percent of the fragments produced recruited into the population. In contrast, several studies describe elkhorn coral populations that are showing some signs of recovery or are stable including in the Turks and Caicos Islands (Schelten et al. 2006), USVI (Grober-Dunsmore et al. 2006; Mayor et al. 2006; Rogers and Muller 2012), Venezuela (Zubillaga et al. 2008), and Belize (Macintyre and Toscano 2007).

Extrapolated population estimates of elkhorn coral from stratified random samples across habitat types in the Florida Keys were 0.6 ± 0.5 million (standard error [SE]) colonies in 2005, 1.0 ± 0.3 million (SE) colonies in 2007, and 0.5 ± 0.3 million (SE) colonies in 2012. Because these population estimates are based on random sampling, differences between years may be a function of sampling effort rather than an indication of population trends. Relative to the abundance of other corals in the Florida Keys region, elkhorn coral was among the least abundant, ranking among corals that are naturally rare in abundance; historically elkhorn coral was a dominant species on Florida reef. Further, no colonies of elkhorn coral were observed in surveys of the Dry Tortugas in 2006 and 2008. The size class distribution of the Florida Keys population included both small and large individuals (> approximately 103 in [260 cm]), but after 2005 the majority of the colonies were smaller in size. These smallest corals (0-8 in [0-20 cm]) had approximately 0-2 percent partial mortality during all three survey years. Partial mortality across all other size classes was approximately 20-70 percent in 2005, 5-50 percent in 2007, and 15-90 percent in 2012 (Miller et al. 2013).

Colonies monitored in the upper Florida Keys showed a greater than 50 percent loss of tissue as well as a decline in the number of colonies, and a decline in the dominance by large colonies between 2004 and 2010 (Vardi et al. 2012; Williams and Miller 2012). Elasticity analysis from a population model based on data from the Florida Keys has shown that the largest individuals have the greatest contribution to the rate of change in population size (Vardi et al. 2012). Between 2010 and 2013, elkhorn coral in the middle and lower Florida Keys had mixed trends. Population densities remained relatively stable at two sites and decreased at two sites by 21 percent and 28 percent (Lunz 2013).

Relatively abundant elkhorn coral communities have been documented from various locations, including Cuba (Alcolado et al. 2010; González-Díaz et al. 2010), Colombia (Sanchez and Pizarro 2005), Venezuela (Martínez and Rodríguez Quintal 2012), Navassa (Bruckner 2012b), Jamaica (Jackson et al. 2014), and the USVI (Muller et al. 2014). Density estimates from sites in Cuba range from 0.14 colonies per m^2 (Alcolado et al. 2010) to 0.18 colonies per m^2 (González-Díaz et al. 2010). Maximum elkhorn coral density at ten sites in St. John, USVI was 0.18 colonies per m^2 (Muller et al. 2014).

Mayor et al. (2006) reported the abundance of elkhorn coral in Buck Island Reef National Monument, St. Croix, USVI. They surveyed 617 sites from May to June 2004 and extrapolated density observed per habitat type to total available habitat. Within an area of 795 ha, they estimated 97,232–134,371 (95 percent confidence limits) elkhorn coral colonies with any dimension of connected live tissue greater than one meter. Mean densities (colonies ≥ one m) were 0.019 colonies per m^2 in branching coral-dominated habitats and 0.013 colonies per m^2 in other hard bottom habitats.

Puerto Rico contains the greatest known extent of elkhorn coral in the U.S. Caribbean, however, the species is still rarely encountered. Between 2006 and 2007, a survey of 431 random points in habitat suitable for elkhorn coral in six marine protected areas in Puerto Rico revealed a variable density of 0-52 elkhorn coral colonies per 100 m^2 (0.52 colonies per m^2), with average density of 3.3 colonies per 100 m^2 (0.03 colonies per m^2). Overall 30.7 percent of all points sampled had live elkhorn coral colonies and total loss of elkhorn coral was evidenced in 13.6 percent of the random survey areas where only dead standing colonies were present (Schärer et al. 2009).

In stratified random surveys along the south, southeast, southwest, and west coasts of Puerto Rico designed to locate *Acropora* colonies, elkhorn coral was observed at five out of 301 stations with sightings outside of the survey area at an additional two stations (García Sais et al. 2013). Elkhorn coral colonies were absent from survey sites along the southeast coast. Maximum density was 18 colonies per 15 m^2 (1.2 colonies per m^2), and maximum colony size was approximately 7.5 ft (2.3 m) in diameter (García Sais et al. 2013).

Zubillaga et al. (2005) report densities of 3.2 colonies of elkhorn coral per ten m^2 (0.32 colonies per m^2) in Los Roques National Park, Venezuela. At ten sites surveyed in the national park in 2003 to 2004, density ranged from 0 to 3.4 colonies per ten m^2 (0 to 0.34 colonies per m^2) with four of the sites showing only standing dead colonies (Zubillaga et al. 2008). In the six sites with live colonies, small (0.1 to 50 cm^2), and medium-sized (50 to 4,550 cm^2) colonies predominated over larger-sized (4,550 to16,500 cm^2) colonies.

At Los Colorados reef in northwestern Cuba, a 2006 study at 12 reef crest sampling stations reported average elkhorn coral densities of 0.18 colonies per m^2, and that elkhorn coral made up 8.7 percent of the total live coral colonies at the study sites. The study also reported that the nearby Baracoa and Rincon de Guanabo reefs had similar elkhorn coral densities (González-Díaz et al. 2010). The size of elkhorn coral colonies indicates some recruitment in Cuba, but not the proportions of sexual versus asexual recruits. In a 2005 study of 280 elkhorn coral colonies at four sites on the north coast of Cuba, 30.4 percent were less than ten cm in diameter (González-Díaz et al. 2008). In a 2006 study of approximately 1,100 elkhorn coral colonies at three sites on the north coast of Cuba, diameter and height size-classes were measured (<2, 3-5, 6-7, 8-10, 11-80, and >80 cm). For the three sites combined, there were approximately 25 to100 colonies in each of the four smaller size classes (Perera-Pérez et al. 2012).

Supplemental information we found on elkhorn coral’s population trends includes the following. At eight of 11 sites in St. John, USVI, colonies of elkhorn coral increased in abundance, between 2001 and 2003, particularly in the smallest size class, with the number of colonies in the largest size class decreasing (Grober-Dunsmore et al. 2007). Colonies of elkhorn coral monitored monthly between 2003 and 2009 in Haulover Bay on St. John, USVI suffered bleaching and mortality from disease but showed an increase in abundance and size at the end of the monitoring period (Rogers and Muller 2012). The overall density of elkhorn coral colonies around St. John did not significantly differ between 2004 and 2010 with six out of the ten sites showing an increase in colony density. Size frequency distribution did not significantly change at seven of the ten sites, with two sites showing an increased abundance of large-sized (> 51 cm) colonies (Muller et al. 2014).

In Colombia, elkhorn coral was present at four of the 32 plots (three of the six reefs) monitored annually from 1998 to 2004. Coverage of elkhorn coral ranged from 0.8-2.4 percent. Over the eight-year period, the species was stable at two reefs and declined at the other reef, likely in response to a hurricane in 1999 (Rodriguez-Ramirez et al. 2010). Macintyre and Toscano (2007) report the return of “numerous large colonies” of elkhorn coral on the shallow fore-reef at the southern limit of Carrie Bow Cay, Belize though no quantitative data were presented.

Elkhorn coral monitored in Curaçao between 2009 and 2011 decreased in abundance and increased in colony size, with stable tissue abundance following hurricane damage (Bright et al. 2013). The authors explained that the apparently conflicting trends of increasing colony size but similar tissue abundance likely resulted from the loss of small-sized colonies that skewed the distribution to larger size classes, rather than colony growth.

Simulation models using data from matrix models of elkhorn coral colonies from specific sites in Curaçao (2006-2011), the Florida Keys (2004-2011), Jamaica (2007-2010), Navassa (2006 and 2009), Puerto Rico (2007 and 2010), and the British Virgin Islands (2006 and 2007) indicate that most of these studied populations will continue to decline in size and extent by 2100 if environmental conditions remain unchanged (i.e., disturbance events such as hurricanes do not increase; (Vardi 2011). In contrast, the studied populations in Jamaica were projected to increase in abundance, and studied populations in Navassa were projected to remain stable. Studied populations in the British Virgin Islands were predicted to decrease slightly from their initial very low levels. Studied populations in Florida, Curaçao, and Puerto Rico were predicted to decline to zero by 2100. Because the study period did not include physical damage (storms), the population simulations in Jamaica, Navassa, and the British Virgin Islands may have contributed to the differing projected trends at sites in these locations.

Vollmer and Palumbi (2007) examined 22 populations of staghorn coral from nine regions in the Caribbean (Panama, Belize, Mexico, Florida, Bahamas, Turks and Caicos, Jamaica, Puerto Rico, and Curaçao) and concluded that populations greater than approximately 310 mi (500 km) apart are genetically different from each other with low gene flow across the greater Caribbean. Fine-scale genetic differences have been detected at reefs separated by as little as 1.25 mi (two km), suggesting that gene flow in staghorn coral may not occur at much smaller spatial scales (Vollmer and Palumbi 2007; Garcia Reyes and Schizas 2010). This fine-scale population structure was greater when considering genes of elkhorn coral were found in staghorn coral due to back-crossing of the hybrid *A. prolifera* with staghorn coral (Garcia Reyes and Schizas 2010; Vollmer and Palumbi 2007). Populations in Florida and Honduras are genetically distinct from each other and other populations in the USVI, Puerto Rico, Bahamas, and Navassa (Baums et al. 2010), indicating little to no larval connectivity overall. However, some potential connectivity between the USVI and Puerto Rico was detected and also between Navassa and the Bahamas (Baums et al. 2010).

Staghorn coral historically was one of the dominant species on most Caribbean reefs, forming large, single-species thickets and giving rise to the nominal distinct zone in classical descriptions of Caribbean reef morphology (Goreau 1959). Massive, Caribbean-wide mortality, apparently primarily from white band disease (Aronson and Precht 2001), spread throughout the Caribbean in the mid-1970s to mid-1980s and precipitated widespread and radical changes in reef community structure (Brainard et al. 2011). In addition, continuing coral mortality from periodic acute events such as hurricanes, disease outbreaks, and mass bleaching events has added to the decline of staghorn coral (Brainard et al. 2011). In locations where quantitative data are available (Florida, Jamaica, USVI, Belize), there was a reduction of approximately 92 to greater than 97 percent between the 1970s and early 2000s (*Acropora* Biological Review Team 2005).

Since the 2006 listing of staghorn coral as threatened, continued population declines have occurred in some locations with certain populations of both listed *Acropora* species decreasing up to an additional 50 percent or more (Lundgren and Hillis-Starr 2008; Muller et al. 2008; Williams et al. 2008; Colella et al. 2012; Rogers and Muller 2012). There are some small pockets of remnant robust populations such as in southeast Florida (Vargas-Angel et al. 2003; Riegl et al. 2009; Keck et al. 2005), and Dominican Republic (Lirman et al. 2010). Additionally, Lidz and Zawada (2013) observed 400 colonies of staghorn coral along 44 mi (70.2 km) of transects near Pulaski Shoal in the Dry Tortugas where the species had not been seen since the cold water die-off of the 1970s. Cover of staghorn coral increased on a Jamaican reef from 0.6 percent in 1995 to 10.5 percent in 2004 (Idjadi et al. 2006).

Riegl et al. (2009) monitored staghorn coral in photo plots on the fringing reef near Roatan, Honduras from 1996 to 2005. Staghorn coral cover declined from 0.42 percent in 1996 to 0.14 percent in 1999 after the Caribbean bleaching event in 1998 and mortality from run-off associated with a Category five hurricane. Staghorn coral cover further declined to 0.09 percent in 2005. Staghorn coral colony frequency decreased 71 percent between 1997 and 1999. In sharp contrast, offshore bank reefs near Roatan had dense thickets of staghorn coral with 31 percent cover in photo-quadrats in 2005 and appeared to survive the 1998 bleaching event and hurricane, most likely due to bathymetric separation from land and greater flushing. Modeling showed that under undisturbed conditions, retention of the dense staghorn coral stands on the banks off Roatan is likely with a possible increased shift towards dominance by other coral species. However, the authors note that because their data and the literature seem to point to extrinsic factors as driving the decline of staghorn coral, it is unclear what the future may hold for this dense population (Riegl et al. 2009).

Miller et al. (2013) extrapolated population abundance of staghorn coral in the Florida Keys and Dry Tortugas from stratified random samples across habitat types. Population estimates of staghorn coral in the Florida Keys were 10.2 ± 4.6 (standard error [SE]) million colonies in 2005, 6.9 ± 2.4 (SE) million colonies in 2007, and 10.0 ± 3.1 (SE) million colonies in 2012. Population estimates in the Dry Tortugas were 0.4 ± 0.4 (SE) million colonies in 2006 and 3.5 ±

2.9 (SE) million colonies in 2008, though the authors note their sampling scheme in the Dry Tortugas was not optimized for staghorn coral. Because these population estimates were based on random sampling, differences in abundance estimates between years is more likely to be a function of sample design rather than population trends. In both the Florida Keys and Dry Tortugas, most of the population was dominated by small colonies less than 12 in (30 cm) diameter. Further, partial mortality was reported as highest in 2005 with up to 80 percent mortality observed and lowest in 2007 with a maximum of 30 percent. In 2012, partial mortality ranged from 20-50 percent across most size classes.

Staghorn coral was observed in 21 out of 301 stations between 2011 and 2013 in stratified random surveys designed to detect *Acropora* colonies along the south, southeast, southwest, and west coasts of Puerto Rico (García Sais et al. 2013). Staghorn coral was also observed at 16 sites outside of the surveyed area. The largest colony was 24 in (60 cm) and density ranged from 1-10 colonies per 162 ft^2 (15 m2; García Sais et al. 2013).

While cover of staghorn coral increased from 0.6 percent in 1995 to 10.5 percent in 2004 (Idjadi et al. 2006) and 44 percent in 2005 on a Jamaican reef, it collapsed after the 2005 bleaching event and subsequent disease to less than 0.5 percent in 2006 (Quinn and Kojis 2008). A cold water die-off across the lower to upper Florida Keys in January 2010 resulted in the complete mortality of all staghorn coral colonies at 45 of the 74 reefs surveyed (61 percent) (Schopmeyer et al. 2012). Walker et al. (2012) report increasing size of two thickets (expansion of up to 7.5 times the original size of one of the thickets) monitored off southeast Florida, but also noted that cover within monitored plots concurrently decreased by about 50 percent highlighting the dynamic nature of staghorn coral distribution via fragmentation and re-attachment.

Distribution

Elkhorn coral is distributed throughout the western Atlantic, Caribbean, and Gulf of Mexico. The northern extent of the range in the Atlantic is Broward County, Florida, where it is relatively rare (only a few known colonies), but fossil elkhorn coral reef framework extends into Palm Beach County, Florida. There are two known colonies of elkhorn coral, which were discovered in 2003 and 2005, at the Flower Garden Banks, which is located 100 mi (161 km) off the coast of Texas in the Gulf of Mexico (Zimmer et al. 2006). The species has been affected by extirpation from many localized areas throughout its range (Jackson et al. 2014).

Goreau (1959) described ten habitat zones on a Jamaican fringing reef from inshore to the deep slope, finding elkhorn coral in eight of the ten zones. Elkhorn coral commonly grows in turbulent water on the fore-reef, reef crest, and shallow spur-and-groove zone (Cairns 1982; Miller et al. 2008; Rogers et al. 1982; Shinn 1963) in water ranging from approximately 3-15 ft (1-5 m) depth. Elkhorn coral often grows in thickets in fringing and barrier reefs (Jaap 1984; Tomascik and Sander 1987; Wheaton and Jaap 1988) and formed extensive barrier-reef structures in Belize (Cairns 1982), the greater and lesser Corn Islands, Nicaragua (Lighty et al. 1982), and Roatan, Honduras, and built extensive fringing reef structures throughout much of the Caribbean (Adey 1978). Early studies termed the reef crest and adjacent seaward areas from the surface down to approximately 20 ft (5-6 m) depth the "palmata zone" because of the domination by the species (Goreau 1959; Shinn 1963). Although elkhorn coral's predominant habitat is reef crests and shallow fore-reefs less than 40 ft (12 m) in depth, it also occasionally occurs in back-reef environments and in depths up to 98 ft (30 m).

Staghorn coral commonly occurs in water ranging from five to 20 m in depth, though it occurs in depths of 16-30 m at the northern extent of its range, and has been rarely found to 60 m in depth. Staghorn coral naturally occurs on spur and groove, bank reef, patch reef, and transitional reef habitats, as well as on limestone ridges, terraces, and hard bottom habitats (Goldberg 1973; Gilmore and Hall 1976; Cairns 1982; Davis 1982; Jaap 1984; Wheaton and Jaap 1988; Miller et al. 2008). Historically it grew in thickets in water ranging from approximately 16-65 ft (5-20 m) in depth; though it has rarely been found to approximately 195 ft (60 m; Schuhmacher and Zibrowius 1985; Davis 1982; Jaap 1984; Wheaton and Jaap 1988; Jaap et al. 1989). At the northern extent of its range, it grows in deeper water (~53-99 ft [16-30 m]; Goldberg 1973). Historically, staghorn coral was one of the primary constructors of mid-depth (approximately 33-50 ft [10-15 m]) reef terraces in the western Caribbean, including Jamaica, the Cayman Islands, Belize, and some reefs along the eastern Yucatan peninsula (Adey 1978). In the Florida Keys, staghorn coral occurs in various habitats but is most prevalent on patch reefs as opposed to their former abundance in deeper fore-reef habitats (Miller et al. 2008). There is no evidence of range constriction, though loss of staghorn coral at the reef level has occurred (*Acropora* Biological Review Team 2005).

Precht (2004) suggest that coincident with climate warming, staghorn coral only recently re-occupied its historic range after contracting to south of Miami, Florida, during the late Holocene. They based this idea on the presence of large thickets off Ft. Lauderdale, Florida, which were discovered in 1998 and had not been reported in the 1970s or 1980s (Precht 2004). However, because the presence of sparse staghorn coral colonies in Palm Beach County, north of Ft. Lauderdale was reported in the early 1970s (though no thicket formation was reported; Goldberg 1973), there is uncertainty associated with whether these thickets were present prior to their discovery or if they recently appeared coincident with warming. The proportion of reefs with staghorn coral present decreased dramatically after the Caribbean-wide mass mortality in the 1970s and 1980s, indicating the spatial structure of the species has been affected by extirpation from many localized areas throughout its range (Jackson et al. 2014).

Status

A report on the status and trends of Caribbean corals over the last century indicates that cover of elkhorn and staghorn coral has remained relatively stable (though much reduced) throughout the region since the large mortality events of the 1970s and 1980s. The report indicates that the number of reefs with elkhorn coral present steadily declined from the 1980s to 2000-2004, then remained stable between 2000-2004 and 2005-2011. Elkhorn coral was present at about 20 percent of reefs surveyed in both the 5-year period of 2000-2004 and the 7-year period of 2005-2011. Elkhorn coral was dominant on approximately five to ten percent of hundreds of reef sites surveyed throughout the Caribbean during the four periods of 1990-1994, 1995-1999, 2000-2004, and 2005-2011 (Jackson et al. 2014). The frequency of reefs at which staghorn coral was described as the dominant coral remained stable. The number of reefs with staghorn coral present declined during the 1980s (from approximately 50 to 30 percent of reefs), remained relatively stable at 30 percent through the 1990s, and decreased to approximately 20 percent of the reefs in 2000-2004 and approximately ten percent in 2005-2011 (Jackson et al. 2014).

Based on population estimates from both the Florida Keys and St. Croix, USVI, there are at least hundreds of thousands of elkhorn coral colonies. Absolute abundance is higher than estimates from these two locations given the presence of this species in many other locations throughout its range. The effective population size is smaller than indicated by abundance estimates due to the tendency for asexual reproduction. Across the Caribbean, percent cover appears to have remained relatively stable, albeit it at extremely low levels, since the population crash in the 1980s. Frequency of occurrence has decreased since the 1980s, indicating potential decreases in the extent of occurrence and effects on the species' range. However, the proportions of Caribbean sites where elkhorn coral is present and dominant have recently stabilized since the mid-2000s. There are locations such as the USVI where populations of elkhorn coral appear stable or possibly increasing in abundance and some such as the Florida Keys where population number appears to be decreasing. In some cases when size class distribution is not reported, there is uncertainty of whether increases in abundance indicate growing populations or fragmentation of larger size classes into more small-sized colonies. From locations where size class distribution is reported, there is evidence of recruitment, but not the proportions of sexual versus asexual recruits. The best evidence of recovery would come from multi-year studies showing an increase in the overall amount of living tissue of this species, growth of existing colonies, and an increase in the number of small corals arising from sexual recruitment (Rogers and Muller 2012). Simulation models predict by 2100 that elkhorn coral will become absent at specific sites in several locations (Florida, Curaçao, and Puerto Rico), decrease at specific sites in the British Virgin Islands, remain stable at specific sites in Navassa, and increase at specific sites in Jamaica. These simulations are based on the assumption that conditions experienced during the monitoring period, ranging from one to seven years depending on location, would remain unchanged in the future. We conclude there has been a significant decline of elkhorn coral throughout its range, with recent population stability at low percent coverage. We also conclude that absolute abundance is at least hundreds of thousands of colonies, but likely to decrease in the future with increasing threats.

Based on population estimates there are at least tens of millions of staghorn coral colonies present in the Florida Keys and Dry Tortugas combined. Absolute abundance is higher than the estimate from these two locations given the presence of this species in many other locations throughout its range. The effective population size is smaller than indicated by abundance estimates due to the tendency for asexual reproduction. There is no evidence of range constriction or extirpation at the island level. However the species is absent at the reef level. Populations appear to consist mostly of isolated colonies or small groups of colonies compared to the vast thickets once prominent throughout its range. Thickets are a prominent feature at only a few known locations. Across the Caribbean, percent cover appears to have remained relatively stable since the population crash in the 1980s. Frequency of occurrence has decreased since the 1980s. There are examples of increasing trends in some locations (Dry Tortugas and southeast Florida), but not over larger spatial scales or longer times. Population model projections from Honduras at one of the only known remaining thickets indicate the retention of this dense stand under undisturbed conditions. If refuge populations are able to persist, it is unclear whether they would be able to repopulate nearby reefs as observed sexual recruitment is low. Thus, we conclude that the species has undergone substantial population decline and decreases in the extent of occurrence throughout its range. Percent benthic cover and proportion of reefs where staghorn coral is dominant have remained stable since the mid-1980s and since the listing of the species as threatened in 2006. We also conclude that population abundance is at least tens of millions of colonies, but likely to decrease in the future with increasing threats.

Designated Critical Habitat

On November 26, 2008, a Final Rule designating *Acropora* critical habitat was published in the Federal Register (73 FR 72210). Within the geographical area occupied by a listed species, critical habitat consists of specific areas on which are found those physical or biological features essential to the conservation of the species. The feature essential to the conservation of *Acropora* species (also known as the essential feature) is substrate of suitable quality and availability in water depths from the mean high water line to 30 m in order to support successful larval settlement, recruitment, and reattachment of fragments. “Substrate of suitable quality and availability” means consolidated hard bottom or dead coral skeletons free from fleshy macroalgae or turf algae and sediment cover. Areas containing this feature have been identified in four locations within the jurisdiction of the U.S.: the Florida area, which comprises approximately 1,329 square miles (3,442 sq km) of marine habitat; the Puerto Rico area, which comprises approximately 1,383 square miles (3,582 sq km) of marine habitat; the St. John/St. Thomas area, which comprises approximately 121 square miles (313 sq km) of marine habitat; and the St. Croix area, which comprises approximately 126 square miles (326 sq km) of marine habitat. The total area covered by the designation is thus approximately 2,959 square miles (7,664 sq km).

The essential feature can be found unevenly dispersed throughout the critical habitat units, interspersed with natural areas of loose sediment, fleshy or turf macroalgae covered hard substrate. Existing federally authorized or permitted artificial structures such as artificial reefs, boat ramps, docks, pilings, channels or marinas do not provide the essential feature. The proximity of this habitat to coastal areas subjects this feature to impacts from multiple activities including dredging and disposal activities, stormwater run-off, coastal and maritime construction, land development, wastewater and sewage outflow discharges, point and non-point source pollutant discharges, fishing, placement of large vessel anchorages, and installation of submerged pipelines or cables. The impacts from these activities, combined with those from natural factors (i.e., major storm events), significantly affect the quality and quantity of available substrate for these threatened species to successfully sexually and asexually reproduce.

For this consultation, only the Florida unit of elkhorn and staghorn coral critical habitat (Figure 12) is reasonably likely to be adversely affected by the action.

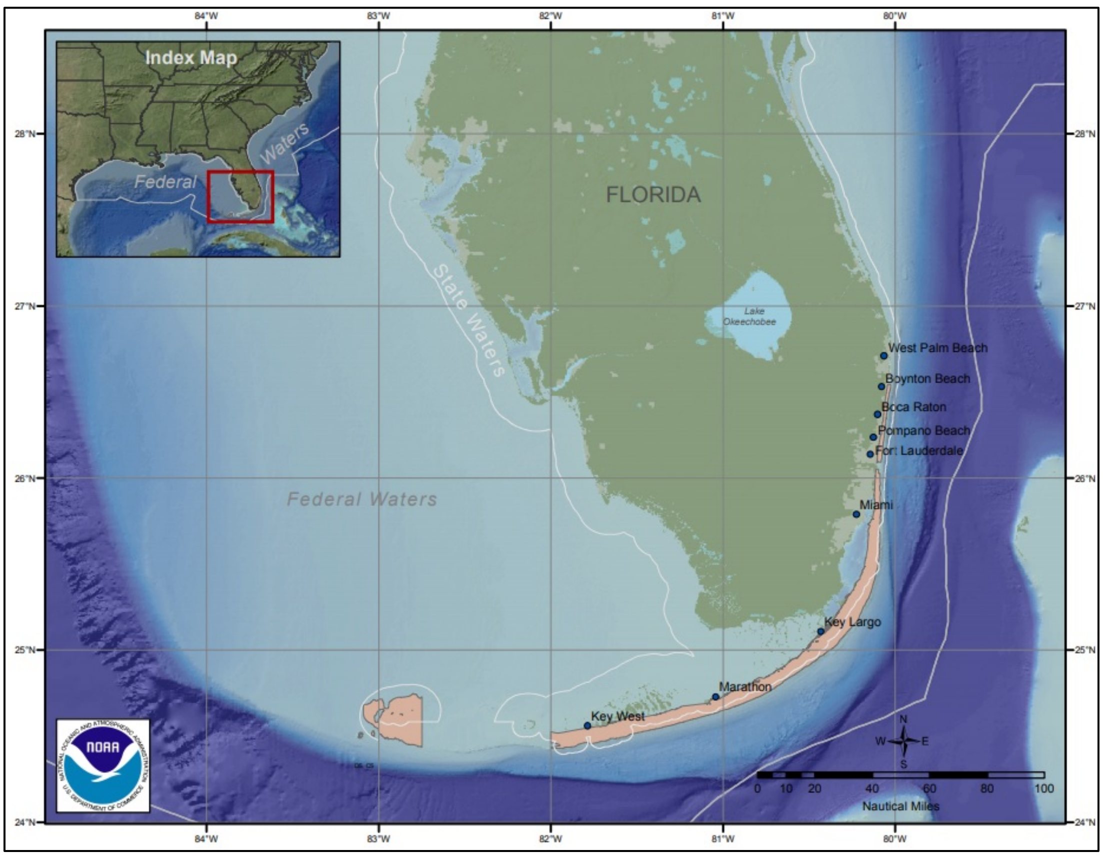

Figure 12. Elkhorn and staghorn coral critical habitat Florida unit

Recovery Goals

A recovery plan for elkhorn and staghorn corals was published March 5, 2015 (80 FR 12146). The recovery plan notes that elkhorn and staghorn corals continue to decline and are at only a small percentage of their abundance throughout their ranges. The recovery plan outlines a recovery strategy for the species:

"Elkhorn and staghorn coral populations should be large enough so that successfully reproducing individuals comprise numerous populations across the historical ranges of these species and are large enough to protect their genetic diversity and maintain their ecosystem functions. Threats to these species and their habitat must be sufficiently abated to ensure a high probability of survival into the future" (NMFS 2015c).

The recovery plan established three recovery criteria associated with the objective of ensuring population viability and seven recovery criteria associated with the objective of eliminating or sufficiently abating global, regional, and local threats that contribute to species' status. The best available information indicates that all recovery objectives must be met for elkhorn and staghorn corals to achieve recovery. The recovery objectives and criteria are:

1. Ensure population viability based on Criterion 1: abundance, Criterion 2: genotypic diversity, and Criterion 3: recruitment
2. Eliminate or sufficiently abate global, regional, and local threats based on Interim Criterion 4: disease, Criterion 5: local and global impacts of rising ocean temperature and acidification, Criterion 6: loss of recruitment habitat, Interim Criterion 7: nutrients, sediments, and contaminants (land-based sources of pollution), Criterion 8: regulatory mechanisms, Criterion 9: natural and anthropogenic abrasion and breakage, and Interim Criterion 10: predation.

6.2.4 Lobed Star, Mountainous Star, and Boulder Star Corals

Lobed star coral (*Orbicella annularis*), mountainous star coral (*Orbicella faveolata*), and boulder star coral (*Orbicella franksi*) are the three species in the *Orbicella annularis* star coral complex. These three species were formerly in the genus *Montastraea*; however, recent work has reclassified the three species in the *annularis* complex to the genus *Orbicella* (Budd et al. 2012). The star coral species complex was historically one of the primary reef framework builders throughout the wider Caribbean. The complex was considered a single species – *Montastraea annularis* – with varying growth forms ranging from columns, to massive boulders, to plates. In the early 1990s, Weil and Knowton (1994) suggested the partitioning of these growth forms into separate species, resurrecting the previously described taxa, *Montastraea* (now *Orbicella*) *faveolata* and *Montastraea* (now *Orbicella*) *franksi*. The three species were differentiated on the basis of morphology, depth range, ecology, and behavior (Weil and Knowton 1994). Subsequent reproductive and genetic studies have supported the partitioning of the *annularis* complex into three species. Lobed and boulder star coral occur in the western Atlantic and greater Caribbean, as well as the Flower Garden Banks, although lobed star may not be present in Bermuda (Figure 13). Mountainous star coral shares the same range but is not present in Bermuda (Figure 14).

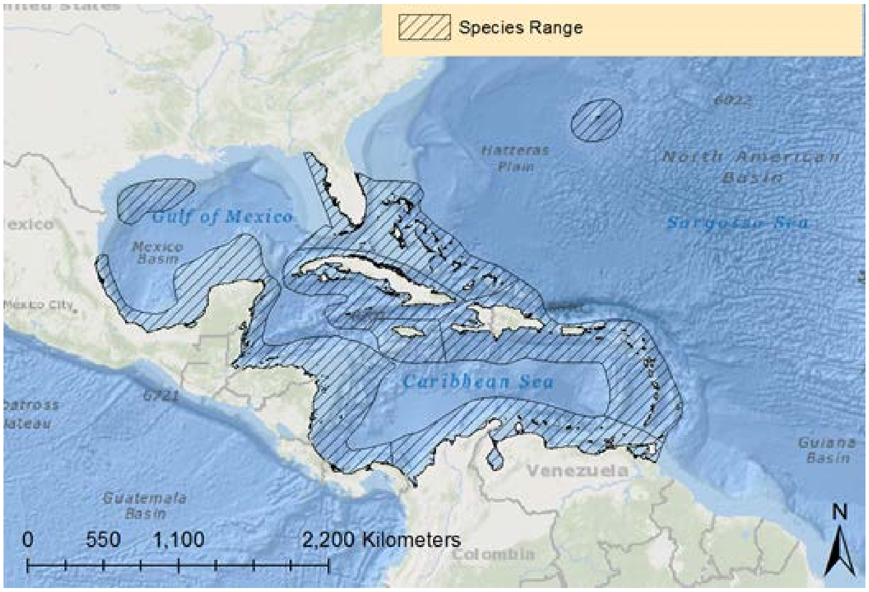

Figure 13. Range map for lobed and boulder star corals

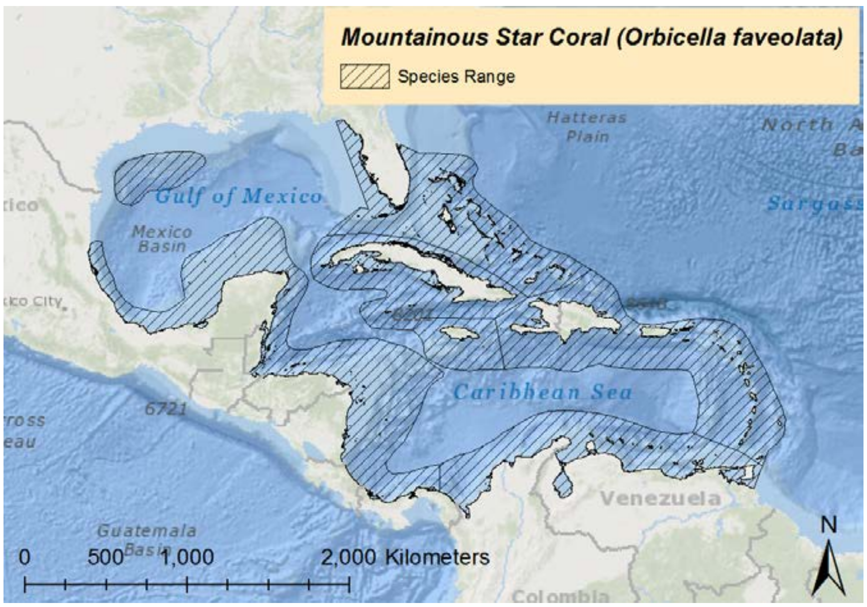

Figure 14. Range map of mountainous star coral

Some studies report on the star coral species complex rather than individual species since visual distinction can be difficult where colony morphology cannot be discerned (e.g. small colonies or photographic methods). Information from these studies is reported for the species complex. Where species-specific information is available, it is reported. However, information about *Orbicella annularis* published prior to 1994 will be attributed to the species complex since it is dated prior to the split of *Orbicella annularis* into three separate species.

Information on lobed star, mountainous star, and boulder star coral status and population dynamics is infrequently documented throughout their range. Comprehensive and systematic census and monitoring for these species has not been conducted. Thus, the status and population dynamics must be inferred from the few locations were data exist.

Life History

The star coral species complex has growth rates ranging from 0.02-0.5 in (0.06-1.2 cm) per year and averaging approximately 0.3 in (one cm) linear growth per year. The reported growth rate of lobed star coral is 0.4 to 1.2 cm per year (Cruz-Piñón et al. 2003; Tomascik 1990). They grow more slowly in deeper water and in less clear water.

All three species of the star coral complex are hermaphroditic broadcast spawners, with spawning concentrated on 6-8 nights following the full moon in late August, September, or early October depending on location and timing of the full moon. All three species are largely self-incompatible (Knowlton et al. 1997; Szmant et al. 1997). Further, mountainous star coral is largely reproductively incompatible with boulder star coral and lobed star coral, and it spawns about 1-2 hours earlier. Fertilization success measured in the field was generally below 15 percent for all three species, as it is closely linked to the number of colonies concurrently spawning. Lobed star coral is reported to have slightly smaller egg size and potentially smaller size/age at first reproduction that the other two species of the *Orbicella* genus. In Puerto Rico, minimum size at reproduction for the star coral species complex was 12 in^2 (83 cm^2).

Successful recruitment by the star coral complex species has seemingly always been rare. Only a single recruit of *Orbicella* was observed over 18 years of intensive observation of 130 ft^2 (12 m^2) of reef in Discovery Bay, Jamaica. Many other studies throughout the Caribbean also report negligible to absent recruitment of the species complex.

<u>Lobed Star Coral</u>: In addition to low recruitment rates, lobed star corals have late reproductive maturity. Colonies can grow very large and live for centuries. Large colonies have lower total mortality than small colonies, and partial mortality of large colonies can result in the production of clones. The historical absence of small colonies and few observed recruits, even though large numbers of gametes are produced on an annual basis, suggests that recruitment events are rare and were less important for the survival of the lobed star coral species complex in the past (Bruckner 2012a). Large colonies in the species complex maintain the population until conditions favorable for recruitment occur; however, poor conditions can influence the frequency of recruitment events. While the life history strategy of the star coral species complex has allowed the taxa to remain abundant, the buffering capacity of this life history strategy has likely been reduced by recent population declines and partial mortality, particularly in large colonies.

<u>Mountainous Star Coral</u>: Life history characteristics of mountainous star coral is considered intermediate between lobed star coral and boulder star coral especially regarding growth rates, tissue regeneration, and egg size. Spatial distribution may affect fecundity on the reef, with

deeper colonies of mountainous star coral being less fecund due to greater polyp spacing. Reported growth rates of mountainous star coral range between 0.12 and 0.64 in (0.3 and 1.6 cm) per year (Cruz-Piñón et al. 2003; Tomascik 1990; Villinski 2003; Waddell 2005). Graham and van Woesik (2013) report that 44 percent of small colonies of mountainous star coral in Puerto Morelos, Mexico that resulted from partial colony mortality produced eggs at sizes smaller than those typically characterized as being mature. The number of eggs produced per unit area of smaller fragments was significantly less than in larger size classes. (Szmant and Miller 2005) reported low post-settlement survivorship for mountainous star coral transplanted to the field with only 3-15 percent remaining alive after 30 days. Post-settlement survivorship was much lower than the 29 percent observed for elkhorn coral after seven months (Szmant and Miller 2005).

Mountainous star coral has slow growth rates, late reproductive maturity, and low recruitment rates. Colonies can grow very large and live for centuries. Large colonies have lower total mortality than small colonies, and partial mortality of large colonies can result in the production of clones. The historical absence of small colonies and few observed recruits, even though large numbers of gametes are produced on an annual basis, suggests that recruitment events are rare and were less important for the survival of the star coral species complex in the past (Bruckner 2012a). Large colonies in the species complex maintain the population until conditions favorable for recruitment occur; however, poor conditions can influence the frequency of recruitment events. While the life history strategy of the star coral species complex has allowed the taxa to remain abundant, we conclude that the buffering capacity of this life history strategy has been reduced by recent population declines and partial mortality, particularly in large colonies.

<u>Boulder Star Coral</u>: Of 351 boulder star coral colonies observed to spawn at a site off Bocas del Toro, Panama, 324 were unique genotypes. Over 90 percent of boulder star coral colonies on this reef were the product of sexual reproduction, and 19 genetic individuals had asexually propagated colonies made up of two to four spatially adjacent clones of each. Individuals within a genotype spawned more synchronously than individuals of different genotypes. Additionally, within 16 ft (five m), colonies nearby spawned more synchronously than farther spaced colonies, regardless of genotype. At distances greater than 16 ft (five m), spawning was random between colonies (Levitan et al. 2011).

In addition to low recruitment rates, lobed star corals have late reproductive maturity. Colonies can grow very large and live for centuries. Large colonies have lower total mortality than small colonies, and partial mortality of large colonies can result in the production of clones. The historical absence of small colonies and few observed recruits, even though large numbers of gametes are produced on an annual basis, suggests that recruitment events are rare and were less important for the survival of the lobed star coral species complex in the past (Bruckner 2012a). Large colonies in the species complex maintain the population until conditions favorable for recruitment occur; however, poor conditions can influence the frequency of recruitment events.

While the life history strategy of the star coral species complex has allowed the taxa to remain abundant, the buffering capacity of this life history strategy has likely been reduced by recent population declines and partial mortality, particularly in large colonies.

Population Dynamics

Lobed Star Coral: Lobed star coral has been described as common overall. Demographic data collected in Puerto Rico over nine years before and after the 2005 bleaching event showed that population growth rates were stable in the pre-bleaching period (2001–2005) but declined one year after the bleaching event. Population growth rates declined even further two years after the bleaching event, but they returned and then stabilized at the lower rate the following year.

In the Florida Keys, abundance of lobed star coral ranked 30 out of 47 coral species in 2005, 13 out of 43 in 2009, and 12 out of 40 in 2012. Extrapolated population estimates from stratified random samples were 5.6 million ± 2.7 million (SE) in 2005, 11.5 million ± 4.5 million (SE) in 2009, and 24.3 million ± 12.4 million (SE) in 2012. Size class distribution was somewhat variable between survey years, with a larger proportion of colonies in the smaller size classes in 2005 compared to 2009 and 2012 and a greater proportion of colonies in the greater than 36-in (90 cm) size class in 2012 compared to 2005 and 2009. Partial colony mortality was lowest at less than four in (ten cm; as low as approximately 5 percent) and up to approximately 70 percent in the larger size classes. In the Dry Tortugas, Florida, abundance of lobed star coral ranked 41 out of 43 in 2006 and 31 out of 40 in 2008. The extrapolated population estimate was 0.5 million ± 0.3 million (SE) colonies in 2008. Differences in population estimates between years may be attributed to sampling effort rather than population trends (Miller et al. 2013).

Colony density varies by habitat and location, and ranges from less than 0.1 to greater than one colony per approximately 100 ft^2 (ten m^2). In surveys of 1,176 sites in southeast Florida, the Dry Tortugas, and the Florida Keys between 2005 and 2010, density of lobed star coral ranged between 0.09 and 0.84 colonies per approximately 100 ft^2 (ten m^2) and was highest on mid-channel reefs followed by inshore reefs, offshore patch reefs, and fore-reefs (Burman et al. 2012). Along the east coast of Florida, density was highest in areas south of Miami (0.34 colonies per approximately 100 ft^2 [10 m^2]) compared to Palm Beach and Broward Counties (0.04 colonies per ~100 ft^2 [10m^2]; Burman et al. 2012). In surveys between 2005 and 2007 along the Florida reef tract from Martin County to the lower Florida Keys, density of lobed star coral was approximately 1.3 colonies per approximately 100 ft^2 (ten m^2; Wagner et al. 2010). Off southwest Cuba on remote reefs, lobed star coral density was 0.31 ± 0.46 (SD) per approximately 30 ft (ten m) transect on 38 reef-crest sites and 1.58 ± 1.29 colonies per approximately 30 ft (ten m) transect on 30 reef-front sites. Colonies with partial mortality were far more frequent than those with no partial mortality which only occurred in the size class less than 40 in (100 cm; Alcolado et al. 2010).

Population trends are available from a number of studies. In a study of sites inside and outside a marine protected area in Belize, lobed star coral cover declined significantly over a 10-year period (1998/99 to 2008/09; Huntington et al. 2011). In a study of ten sites inside and outside of a marine reserve in the Exuma Cays, Bahamas, cover of lobed star coral increased between 2004 and 2007 inside the protected area and decreased outside the protected area (Mumby and Harborne 2010). Between 1996 and 2006, lobed star coral declined in cover by 37 percent in permanent monitoring stations in the Florida Keys (Waddell and Clarke 2008). Cover of lobed star coral declined 71 percent in permanent monitoring stations between 1996 and 1998 on a reef in the upper Florida Keys (Porter et al. 2001).

Star corals are the 3rd most abundant coral by percent cover in permanent monitoring stations in the USVI. A decline of 60 percent was observed between 2001 and 2012 primarily due to bleaching in 2005. However, most of the mortality was partial mortality and colony density in monitoring stations did not change (Smith 2013).

Bruckner and Hill (2009) did not note any extirpation of mountainous star coral at nine sites off Mona and Desecheo Islands, Puerto Rico, monitored between 1995 and 2008. However, mountainous star coral and lobed star coral sustained the largest losses with the number of colonies of lobed star coral decreasing by 19 percent and 20 percent at Mona and Desecheo Islands, respectively. In 1998, 8 percent of all corals at six sites surveyed off Mona Island were lobed star coral colonies, dipping to approximately 6 percent in 2008. At Desecheo Island, 14 percent of all coral colonies were lobed star coral in 2000 while 13 percent were in 2008 (Bruckner and Hill 2009).

In a survey of 185 sites in five countries (Bahamas, Bonaire, Cayman Islands, Puerto Rico, and St. Kitts and Nevis) in 2010 and 2011, size of lobed star coral and boulder star coral colonies was significantly smaller than mountainous star coral. Total mean partial mortality of lobed star coral colonies at all sites was 40 percent. Overall, the total area occupied by live lobed star coral declined by a mean of 51 percent, and mean colony size declined from 299 in^2 to 146 in^2 (1927 cm^2 to 939 cm^2). There was a 211 percent increase in small tissue remnants less than 78 in^2 (500 cm^2), while the proportion of completely live large (1.6-32 ft^2 [1,500- 30,000 cm^2]) colonies declined. Star coral colonies in Puerto Rico were much larger with large amounts of dead sections. In contrast, colonies in Bonaire were also large with greater amounts of live tissue. The presence of dead sections was attributed primarily to outbreaks of white plague and yellow band disease, which emerged as corals began recovering from mass bleaching events. This was followed by increased predation and removal of live tissue by damselfish algal lawns (Bruckner 2012a).

Cover of lobed star coral at Yawzi Point, St. John, USVI declined from 41 percent in 1988 to approximately 12 percent by 2003 as a rapid decline began with the aftermath of Hurricane Hugo in 1989. This decline continued between 1994 and 1999 during a time of two hurricanes (1995) and a year of unusually high sea temperature (1998) but percent cover remained statistically unchanged between 1999 and 2003. Colony abundances declined from 47 to 20 colonies per approximately ten ft^2 (one m^2) between 1988 and 2003, due mostly to the death and fission of medium-to-large colonies (≥ 24 in^2 [151 cm^2]). Meanwhile, the population size class structure shifted between 1988 and 2003 to a higher proportion of smaller colonies in 2003 (60 percent less than seven in^2 [50 cm^2] in 1988 versus 70 percent in 2003) and lower proportion of large colonies (6 percent greater than 39 in^2 [250 cm^2] in 1988 versus 3 percent in 2003). The changes in population size structure indicated a population decline coincident with the period of apparent stable coral cover. Population modeling forecasted the 1988 size structure would not be reestablished by recruitment and a strong likelihood of extirpation of lobed star coral at this site within 50 years (Edmunds and Elahi 2007).

Lobed star coral colonies were monitored between 2001 and 2009 at Culebra Island, Puerto Rico. The population was in demographic equilibrium (high rates of survival and stasis) before the 2005 bleaching event, but it suffered a significant decline in growth rate (mortality and shrinkage) for two consecutive years after the bleaching event. Partial tissue mortality due to bleaching caused dramatic colony fragmentation that resulted in a population made up almost entirely of small colonies by 2007 (97 percent were less than seven in^2 [50 cm^2]). Three years after the bleaching event, the population stabilized at about half of the previous level, with fewer medium-to-large size colonies and more smaller colonies (Hernandez-Delgado et al. 2011).

<u>Mountainous Star Coral</u>: Information regarding population structure is limited. Observations of mountainous star coral from 182 sample sites in the upper and lower Florida Keys and Mexico showed three well-defined populations based on five genetic markers, but the populations were not stratified by geography, indicating they were shared among the three regions (Baums et al. 2010). Of ten mountainous star coral colonies observed to spawn at a site off Bocas del Toro, Panama, there were only three genotypes (Levitan et al. 2011) potentially indicating 30 percent clonality.

Extrapolated population estimates from stratified random samples in the Florida Keys were 39.7 ± eight million (SE) colonies in 2005, 21.9 ± seven million (SE) colonies in 2009, and 47.3 ± 14.5 million (SE) colonies in 2012. The greatest proportion of colonies tended to fall in the 4-8 in (10-20 cm) and 8-12 in (20-30 cm) size classes in all survey years, but there was a fairly large proportion of colonies in the greater than 36-in (90 cm)-size class. Partial mortality of the colonies was between 10 percent and 60 percent of the surface across all size classes. In the Dry Tortugas, Florida, mountainous star coral ranked seventh most abundant out of 43 coral species in 2006 and fifth most abundant out of 40 in 2008. Extrapolated population estimates were 36.1 ± 4.8 million (SE) colonies in 2006 and 30 ± 3.3 million (SE) colonies in 2008. The size classes with the largest proportion of colonies were 4-8 in (10-20 cm) and 8-12 in (20-30 cm), but there was a fairly large proportion of colonies in the greater-than-36-in (90 cm) size class. Partial mortality of the colonies ranged between approximately 2 percent and 50 percent. Because these population abundance estimates are based on random surveys, differences between years may be attributed to sampling effort rather than population trends (Miller et al. 2013).

In a survey of 31 sites in Dominica between 1999 and 2002, mountainous star coral was present at 80 percent of the sites at one to ten percent cover (Steiner 2003). In a 1995 survey of 16 reefs in the Florida Keys, mountainous star coral ranked as the coral species with the second highest percent cover (Murdoch and Aronson 1999). On 84 patch reefs (ten ft [3 m] to 16.5 ft [5 m] depth) spanning 149 mi (240 kilometers [km]) in the Florida Keys, mountainous star coral was the third most abundant coral species comprising 7 percent of the 17,568 colonies encountered. It was present at 95 percent of surveyed reefs between 2001 and 2003 (Lirman and Fong 2007). In surveys of 280 sites in the upper Florida Keys in 2011, mountainous star coral was present at 87 percent of sites visited (Miller et al. 2011). In 2003 on the East Flower Garden Bank, mountainous star coral comprised 10 percent of the 76.5 percent coral cover on reefs 105-132 ft (32-40 m), and partial mortality due to bleaching, disease, and predation were rare at monitoring stations (Precht et al. 2005).

Colony density ranges from approximately 0.1-1.8 colonies per 108 ft^2 (ten m^2) and varies by habitat and location. In surveys along the Florida reef tract from Martin County to the lower Florida Keys, density of mountainous star coral was approximately 1.6 colonies per 108 ft^2 (ten m^2; Wagner et al. 2010). On remote reefs off southwest Cuba, density of mountainous star coral was 0.12 ± 0.20 (SE) colonies per 33 ft (ten m) transect on 38 reef-crest sites and 1.26 ± 1.06 (SE) colonies per 33 ft (ten m) transect on 30 reef-front sites (Alcolado et al. 2010). In surveys of 1,176 sites in southeast Florida, the Dry Tortugas, and the Florida Keys between 2005 and 2010, density of mountainous star coral ranged between 0.17 and 1.75 colonies per 108 ft^2 (ten m^2) and was highest on mid-channel reefs followed by offshore patch reefs and fore-reefs (Burman et al. 2012). Along the east coast of Florida, density was highest in areas south of Miami at 0.94 colonies per 108 ft^2 (ten m^2) compared to 0.11 colonies per 108 ft^2 (ten m^2) in Palm Beach and Broward Counties (Burman et al. 2012).

Mountainous star coral is the sixth most abundant species by percent cover in permanent monitoring stations in the USVI. The star coral species complex had the highest abundance at these stations and included all colonies where species identification was uncertain. Population estimates in the 19 mi^2 (49 km^2) of the Red Hind Marine Conservation District are at least 16 million colonies of mountainous star corals (Smith 2013).

Population trend data exists for several locations. At nine sites off Mona and Desecheo Islands, Puerto Rico, no species extirpations were noted at any site over ten years of monitoring between 1998 and 2008 (Bruckner and Hill 2009). Both mountainous star coral and lobed star coral sustained large losses during the period. The number of colonies of mountainous star coral decreased by 36 percent and 48 percent at Mona and Desecheo Islands, respectively (Bruckner and Hill 2009). In 1998, 27 percent of all corals at six sites surveyed off Mona Island were mountainous star coral colonies, but this statistic decreased to approximately 11 percent in 2008 (Bruckner and Hill 2009). At Desecheo Island, 12 percent of all coral colonies were mountainous star coral in 2000, compared to 7 percent in 2008.

In a survey of 185 sites in five countries (Bahamas, Bonaire, Cayman Islands, Puerto Rico, and St. Kitts and Nevis) between 2010 and 2011, size of mountainous star coral colonies was significantly greater than boulder star coral and lobed star coral. The total mean partial mortality of mountainous star coral at all sites was 38 percent. The total live area occupied by mountainous star coral declined by a mean of 65 percent, and mean colony size declined from 43 ft^2 to 15 ft^2 (4005 cm^2 to 1413 cm^2). At the same time, there was a 168 percent increase in small tissue remnants less than five ft^2 (500 cm^2), while the proportion of completely live large (1.6 ft^2 to 32 ft^2 [1,500- 30,000 cm^2]) colonies decreased. Mountainous star coral colonies in Puerto Rico were much larger and sustained higher levels of mortality compared to the other four countries. Colonies in Bonaire were also large, but they experienced much lower levels of mortality. Mortality was attributed primarily to outbreaks of white plague and yellow band disease, which emerged as corals began recovering from mass bleaching events. This was followed by increased predation and removal of live tissue by damselfish to cultivate algal lawns (Bruckner 2012a).

<u>Boulder Star Coral</u>: Boulder star coral is reported as common. In a 1995 survey of 16 reefs in the Florida Keys, boulder star coral had the highest percent cover of all species (Murdoch and Aronson 1999). In surveys throughout the Florida Keys, boulder star coral in 2005 ranked 26th most abundant out of 47 coral species, 32nd out of 43 in 2009, and 33rd out of 40 in 2012. Extrapolated population estimates from stratified random surveys were 8.0 ± 3.5 million (SE) colonies in 2005, 0.3 ± 0.2 million (SE) colonies in 2009, and 0.4 ± 0.4 million (SE) colonies in 2012. The authors note that differences in extrapolated abundance between years were more likely a function of sampling design rather than an indication of population trends. In 2005, the greatest proportions of colonies were in the smaller size classes of approximately 4-8 in (10-20 cm) and approximately 8-12 in (20-30 cm). Partial colony mortality ranged from 0 percent to approximately 73 percent and was generally higher in larger colonies (Miller et al. 2013).

In the Dry Tortugas, Florida, boulder star coral ranked 4th highest in abundance out of 43 coral species in 2006 and 8th out of 40 in 2008. Extrapolated population estimates were 79 ± 19 million (SE) colonies in 2006 and 18.2 ± 4.1 million (SE) colonies in 2008. The authors note the difference in estimates between years was more likely a function of sampling design rather than population decline. In the first year of the study (2006), the greatest proportion of colonies were in the size class approximately eight-12 in (20-30 cm) with twice as many colonies as the next most numerous size class and a fair number of colonies in the largest size class of greater than three ft (90 cm). Partial colony mortality ranged from approximately ten-55 percent. Two years later (2008), no size class was found to dominate, and proportion of colonies in the medium-to-large size classes (approximately 24-36 in) appeared to be less than in 2006. The number of colonies in the largest size class of greater than three ft (90 cm) remained consistent. Partial colony mortality ranged from approximately 15-75 percent (Miller et al. 2013).

In 2003, on the east Flower Garden Bank, boulder star coral comprised 46 percent of the 76.5 percent coral cover on reefs approximately 105-131 ft (32-40 m) in depth. Partial coral mortality due to bleaching, disease and predation was rare in survey stations (Precht et al. 2005). In a survey of 31 sites in Dominica between 1999 and 2002, boulder star coral was present in 7 percent of the sites at less than 1 percent cover (Steiner 2003).

Reported density is variable by location and habitat and is reported to range from 0.02 to 1.05 colonies per approximately (~) 100 ft^2 (ten m^2). In surveys of 1,176 sites in southeast Florida, the Dry Tortugas, and the Florida Keys between 2005 and 2010, density of boulder star coral ranged between 0.04 and 0.47 colonies per ~100 ft^2 (ten m^2) and was highest on the offshore patch reef and fore-reef habitats (Burman et al. 2012). In south Florida, density was highest in areas south of Miami at 0.44 colonies per ~100 ft^2 (ten m^2) compared to 0.02 colonies per ~100 ft^2 (ten m^2) in Palm Beach and Broward Counties (Burman et al. 2012). Along the Florida reef tract from Martin County to the lower Florida Keys, density of boulder star coral was ~0.9 colonies per ~100 ft^2 (ten m^2; (Wagner et al. 2010). On remote reefs off southwest Cuba, colony density was 0.083 ± 0.17 (SD) per ~100 ft^2 (ten m^2) transect on 38 reef-crest sites and 1.05 ± 1.02 colonies per ~100 ft^2 (ten m^2) transect on 30 reef-front sites (Alcolado et al. 2010). The number of boulder star coral colonies in Cuba with partial colony mortality were far more frequent than those with no mortality across all size classes, except for one (i.e., less than ~20 in [50 cm]) that had similar frequency of colonies with and without partial mortality (Alcolado et al. 2010).

In the USVI, boulder star coral is the second most abundant species by percent cover at permanent monitoring stations. However, because the species complex, which is the most abundant by cover, was included as a category prior to separating the three sibling species, it is likely that boulder star coral is the most abundant, when including mesophotic reefs. Population estimates of boulder star coral in the approximately 19-mi^2 (49 km^2) area of the Red Hind Marine Conservation District are at least 34 million colonies (Smith 2013).

Abundance in Curaçao and Puerto Rico appears to be stable over an eight to ten year period. In Curaçao, abundance was stable between 1997 and 2005, with partial mortality similar or less in 2005 compared to 1998 (Bruckner and Bruckner 2006). Abundance was also stable between 1998-2008 at nine sites off Mona and Desecheo Islands, Puerto Rico. In 1998, 4 percent of all corals at six sites surveyed off Mona Island were boulder star coral colonies and approximately 5 percent in 2008; at Desecheo Island, about 2 percent of all coral colonies were boulder star coral in both 2000 and 2008 (Bruckner and Hill 2009).

On the other hand, colony size has decreased over the past several decades. Bruckner conducted a survey of 185 sites (2010 and 2011) in five countries (The Bahamas, Bonaire, Cayman Islands, Puerto Rico, and St. Kitts and Nevis) and reported the size of boulder star coral and lobed star coral colonies as significantly smaller than mountainous star coral. The total mean partial mortality of boulder star coral was 25 percent. Overall, the total live area occupied by boulder star coral declined by a mean of 38 percent, and mean colony size declined from 210 in^2 to 131 in^2 (1356 cm^2 to 845 cm^2). At the same time, there was a 137 percent increase in small tissue remnants, along with a decline in the proportion of large (1,500 to 30,000 cm^2), completely alive colonies. Mortality was attributed primarily to outbreaks of white plague and yellow band disease, which emerged as corals began recovering from mass bleaching events. This was followed by increased predation and removal of live tissue by damselfish to cultivate algal lawns (Bruckner 2012a).

Distribution

Lobed Star Coral: Lobed star coral is common throughout the western Atlantic Ocean and greater Caribbean Sea including the Flower Garden Banks, but may be absent from Bermuda. Lobed star coral is reported from most reef environments in depths of approximately 1.5-66 ft (0.5-20 m). The star coral species complex is a common, often dominant component of Caribbean mesophotic (e.g., >100 ft [30 m]) reefs, suggesting the potential for deep refuge across a broader depth range, but lobed star coral is generally described with a shallower distribution.

Asexual fission and partial mortality can lead to multiple clones of the same colony. The percentage of unique individuals is variable by location and is reported to range between 18 percent and 86 percent (thus, 14-82 percent are clones). Colonies in areas with higher disturbance from hurricanes tend to have more clonality. Genetic data indicate that there is some population structure in the eastern, central, and western Caribbean with population connectivity within but not across areas. Although lobed star coral is still abundant, it may exhibit high clonality in some locations, meaning that there may be low genetic diversity.

Mountainous Star Coral: Mountainous star coral occurs in the western Atlantic and throughout the Caribbean, including the Bahamas, Flower Garden Banks, and the entire Caribbean coastline. There is conflicting information on whether it occurs in Bermuda. Mountainous star coral has been reported in most reef habitats and is often the most abundant coral at 33-66 ft (10-20 m) in fore-reef environments. The depth range of mountainous star coral has been reported as approximately 1.5-132 ft (0.5-40 m), though the species complex has been reported to depths of 295 ft (90 m), indicating mountainous star coral's depth distribution is likely deeper than 132 ft (40 m). Star coral species are a common, often dominant component of Caribbean mesophotic reefs (e.g., > 100 ft [30 m]), suggesting the potential for deep refugia for mountainous star coral.

Boulder Star Coral: Boulder star coral is distributed in the western Atlantic Ocean and throughout the Caribbean Sea including in the Bahamas, Bermuda, and the Flower Garden Banks. Boulder star coral tends to have a deeper distribution than the other two species in the *Orbicella* species complex. It occupies most reef environments and has been reported from water depths ranging from approximately 16-165 ft (5-50 m), with the species complex reported to 250 ft (90 m). *Orbicella* species are a common, often dominant, component of Caribbean mesophotic reefs (e.g., >100 ft [30 m]), suggesting the potential for deep refugia for boulder star coral.

Status

Lobed Star Coral: Lobed star coral was historically considered to be one of the most abundant species in the Caribbean (Weil and Knowton 1994). Percent cover has declined to between 37 percent and 90 percent over the past several decades at reefs at Jamaica, Belize, Florida Keys, The Bahamas, Bonaire, Cayman Islands, Curaçao, Puerto Rico, USVI, and St. Kitts and Nevis. Based on population estimates, there are at least tens of millions of lobed star coral colonies present in the Florida Keys and Dry Tortugas combined. Absolute abundance is higher than the estimate from these two locations given the presence of this species in many other locations throughout its range. Star coral remains common in occurrence. Abundance has decreased in some areas to between 19 percent and 57 percent, and shifts to smaller size classes have occurred in locations such as Jamaica, Colombia, The Bahamas, Bonaire, Cayman Islands, Puerto Rico, USVI, and St. Kitts and Nevis. At some reefs, a large proportion of the population is comprised of non-fertile or less-reproductive size classes. Several population projections indicate population decline in the future is likely at specific sites, and local extirpation is possible within 25-50 years at conditions of high mortality, low recruitment, and slow growth rates. We conclude that while substantial population decline has occurred in lobed star coral, it is still common throughout the Caribbean and remains one of the dominant species numbering at least in the tens of millions of colonies. We conclude that the buffering capacity of lobed star coral's life history strategy that has allowed it to remain abundant has been reduced by the recent population declines and amounts of partial mortality, particularly in large colonies. We also conclude that the population abundance is likely to decrease in the future with increasing threats.

Mountainous Star Coral: Based on population estimates, there are at least tens of millions of colonies present in each of several locations including the Florida Keys, Dry Tortugas, and the USVI. Absolute abundance is higher than the estimate from these three locations given the presence of this species in many other locations throughout its range. Population decline has occurred over the past few decades with a 65 percent loss in mountainous star coral cover across five countries. Losses of mountainous star coral from Mona and Desecheo Islands, Puerto Rico include a 36-48 percent reduction in abundance and a decrease of 42-59 percent in its relative abundance (i.e., proportion relative to all coral colonies). High partial mortality of colonies has led to smaller colony sizes and a decrease of larger colonies in some locations such as The Bahamas, Bonaire, Puerto Rico, Cayman Islands, and St. Kitts and Nevis. Partial colony mortality is lower in some areas such as the Flower Garden Banks. We conclude that mountainous star coral has declined but remains common and likely has at least tens of millions of colonies throughout its range. Additionally, as discussed in the genus section, we conclude that the buffering capacity of mountainous star coral's life history strategy which has allowed it to remain abundant has been reduced by the recent population declines and amounts of partial mortality, particularly in large colonies. We also conclude that the population abundance is likely to decrease in the future with increasing threats.

Boulder Star Coral: Based on population estimates, there are at least tens of millions of colonies present in both the Dry Tortugas and USVI. Absolute abundance is higher than the estimate from these two locations given the presence of this species in many other locations throughout its range. The frequency and extent of partial mortality, especially in larger colonies of boulder star coral, appear to be high in some locations such as Florida and Cuba, though other locations like the Flower Garden Banks appear to have lower amounts of partial mortality. A decrease in boulder star coral percent cover by 38 percent and a shift to smaller colony size across five countries suggest that population decline has occurred in some areas; colony abundance appears to be stable in other areas. We anticipate that while population decline has occurred, boulder star coral is still common with the number of colonies at least in the tens of millions. Additionally, we conclude that the buffering capacity of boulder star coral's life history strategy that has allowed it to remain abundant has been reduced by the recent population declines and amounts of partial mortality, particularly in large colonies. We also anticipate that the population abundance is likely to decrease in the future with increasing threats.

Designated Critical Habitat

No critical habitat has been designated for corals in the star coral complex.

Recovery Goals

No final recovery plan currently exists for boulder star coral; however, a recovery outline was developed in 2014 to serve as interim guidance to direct recovery efforts, including recovery planning, until a final recovery plan is developed and approved. The following contains the recovery goals listed in the document:

Short Term Goals

- Increase understanding of population dynamics, population distribution, abundance, trends, and structure through research, monitoring, and modeling
- Through research, increase understanding of genetic and environmental factors that lead to variability of bleaching and disease susceptibility
- Decrease locally-manageable stress and mortality sources (e. g. , acute sedimentation, nutrients, contaminants, over-fishing).
- Prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit *D. cylindrus*, *M. ferox*, and *Orbicella* spp.

Long Term Goals

- Cultivate and implement U. S. and international measures to reduce atmospheric carbon dioxide concentrations to curb warming and acidification impacts and possibly disease threats.
- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment.

6.2.5 Rough Cactus Coral

The rough cactus coral is a cnidarian belonging to the taxonomic genus of Mycetophyllia, a group of ridged corals that form colonies with a flat disc shape. Rough cactus coral occurs in the western Atlantic Ocean and throughout the wider Caribbean Sea (Figure 15). While rough cactus coral occurs in the western Atlantic Ocean and throughout the wider Caribbean Sea, it has not been reported in the Flower Garden Banks (Gulf of Mexico) or in Bermuda. It inhabits reef environments in water depths of five to ninety m, including shallow and mesophotic habitats (e.g., > 30 m).

Rough cactus coral forms a thin, encrusting plate that is weakly attached to substrate. Rough cactus coral is taxonomically distinct (i.e., separate species), though difficult to distinguish in the field from other *Mycetophyllia* species. The maximum colony size of the species is 50 centimeters (cm) in diameter.

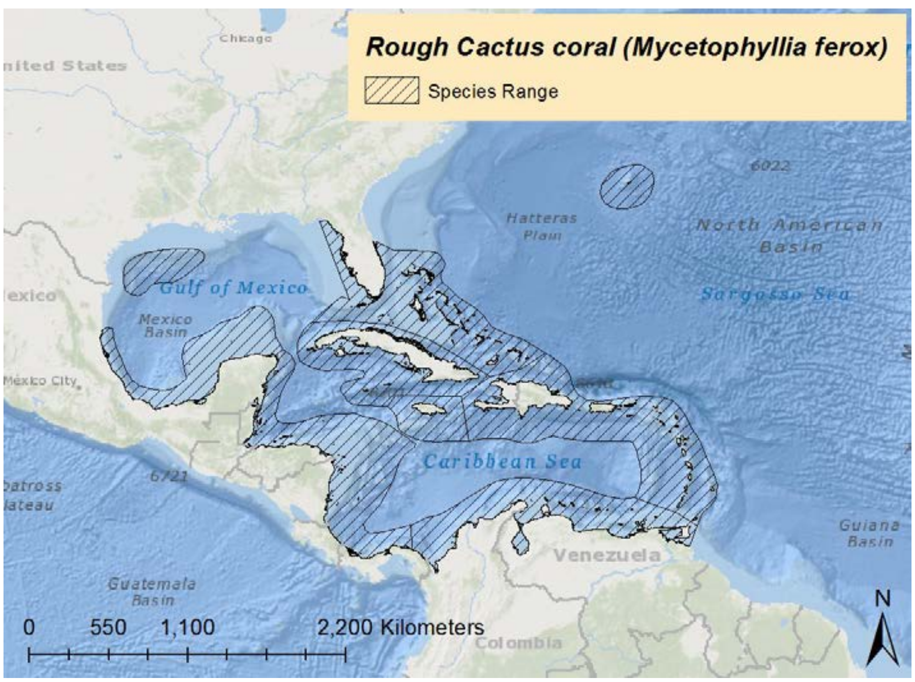

Figure 15. Range map for rough cactus coral

Life history

Rough cactus coral is a hermaphroditic brooding[7] species. Colony size at first reproduction is greater than 100 square centimeters. Recruitment of rough cactus coral appears to be very low, even in studies from the 1970s. Rough cactus coral has a lower fecundity compared to other species in its genus (Morales Tirado 2006). Over a ten year period, no colonies of rough cactus coral were observed to recruit to an anchor-damaged site in the USVI, although adults were observed on the adjacent reef (Rogers and Garrison 2001). No other life history information appears to exist for rough cactus coral.

[7] Simultaneously containing both sperm and eggs, which are fertilized within the parent colony and grow for a time before release.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section consists of abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the rough cactus coral.

Rough cactus coral is usually uncommon or rare according to published and unpublished records, indicating that it constitutes < 0.1 percent species contribution (percent of all colonies counted) and occurs at densities < 0.8 colonies per ten square meters (m^2) in Florida and at 0.8 colonies per 100 m transect in Puerto Rico sites sampled by the Atlantic and Gulf Rapid Reef Assessment (Veron 2002, Wagner el al., 2010, and AGRRA database as cited in Brainard et al. 2011). Recent monitoring data (e.g., since 2000) from Florida (National Park Service permanent monitoring stations), La Parguera Puerto Rico, and St. Croix (USVI/ NOAA Center for Coastal Monitoring and Assessment randomized monitoring stations) show rough cactus coral cover to be consistently less occasional observations up to two percent and no apparent temporal trend (Brainard et al. 2011).

Dustan (1977) proposes that rough cactus coral was much more abundant in the upper Florida Keys in the early mid- 1970s (the methods are not well described for that study) than current observations, but that it was highly affected by disease. This could be interpreted as a substantial decline. Long-term Coral Reef Evaluation and Monitoring Program monitoring data in Florida on species presence/absence from fixed sites (stations) show a dramatic decline; for 97 stations in the main Florida Keys, occurrence had declined from 20 stations in 1996 to four stations in 2009; in Dry Tortugas occurrence had declined from eight out of twenty-one stations in 2004 to three stations in 2009 (R. Ruzicka and M. Colella, Florida Marine Research Institute, St. Petersburg, FL. pers. comm., Oct 2010 cited in Brainard et al. 2011).

Distribution

According to the International Union for Conservation of Nature (IUCN) Species Account and the Convention on International Trade in Endangered Species (CITES) species database, rough cactus coral occurs throughout the U.S. waters of the western Atlantic but has not been reported from Flower Garden Banks (Hickerson et al. 2008). The following areas include locations within federally protected waters where rough cactus coral has been observed and recorded (cited in Brainard et al. 2011): Dry Tortugas National Park; Virgin Island National Park/Monument; Florida Keys; National Marine Sanctuary; Navassa Island National Wildlife Refuge; Biscayne National Park; Buck Island Reef National Monument.

On reefs where rough cactus coral is found, it generally occurs at abundances of less than one colony per approximately ten m^2 and percent cover of less than 0.1 (Burman et al. 2012). Based on population estimates, there are at least hundreds of thousands of rough cactus coral colonies present in the Florida Keys and Dry Tortugas combined. Absolute abundance is higher than the estimate from these two locations given the presence of this species in many other locations throughout its range. Low encounter rate and percent cover coupled with the tendency to include *Mycetophyllia* spp. at the genus level make it difficult to discern population trends of rough cactus coral from monitoring data. However, reported losses of rough cactus coral from monitoring stations in the Florida Keys and Dry Tortugas (63-80 percent loss) indicate population decline in these locations. Based on declines in Florida, we conclude rough cactus coral has likely declined throughout its range, and will continue to decline based on increasing threats. As a result, it is presumed that genetic diversity for the species is low.

Status

Rough cactus coral has declined due to disease in at least a portion of its range and has low recruitment, which limits its capacity for recovery from mortality events and exacerbates vulnerability to extinction. Its depth range of five to 90 m moderates vulnerability to extinction over the foreseeable future because deeper areas of its range will usually have lower temperatures than surface waters. Acidification is predicted to accelerate most in deeper and cooler waters than those in which the species occurs. Its habitat includes shallow and mesophotic reefs which moderates vulnerability to extinction over the foreseeable future because the species occurs in numerous types of reef environments that are predicted, on local and regional scales, to experience highly variable thermal regimes and ocean chemistry at any given point in time. Rough cactus coral is usually uncommon to rare throughout its range. Its abundance, combined with spatial variability in ocean warming and acidification across the species' range, moderate vulnerability to extinction because the threats are non-uniform. Subsequently, there will likely be a large number of colonies that are either not exposed or do not negatively respond to a threat at any given point in time.

Designated Critical Habitat

No critical habitat has been designated for rough cactus coral.

Recovery Goals

No final recovery plan currently exists for rough cactus coral, however a recovery outline was developed in 2014 to serve as interim guidance to direct recovery efforts, including recovery planning, until a final recovery plan is developed and approved. The following contains the recovery goals listed in the document:

Short Term Goals

- Increase understanding of population dynamics, population distribution, abundance, trends, and structure through research, monitoring, and modeling
- Through research, increase understanding of genetic and environmental factors that lead to variability of bleaching and disease susceptibility

- Decrease locally-manageable stress and mortality sources (e.g., acute sedimentation, nutrients, contaminants, over-fishing).
- Prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit *D. cylindrus*, *M. ferox*, and *Orbicella* spp.

Long Term Goals

- Cultivate and implement U.S. and international measures to reduce atmospheric carbon dioxide concentrations to curb warming and acidification impacts and possibly disease threats.
- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment.

6.2.6 Pillar Coral

Pillar coral is present in the western Atlantic Ocean and throughout the greater Caribbean Sea, though absent from the southwest Gulf of Mexico (Figure 16; Tunnell Jr. 1988).

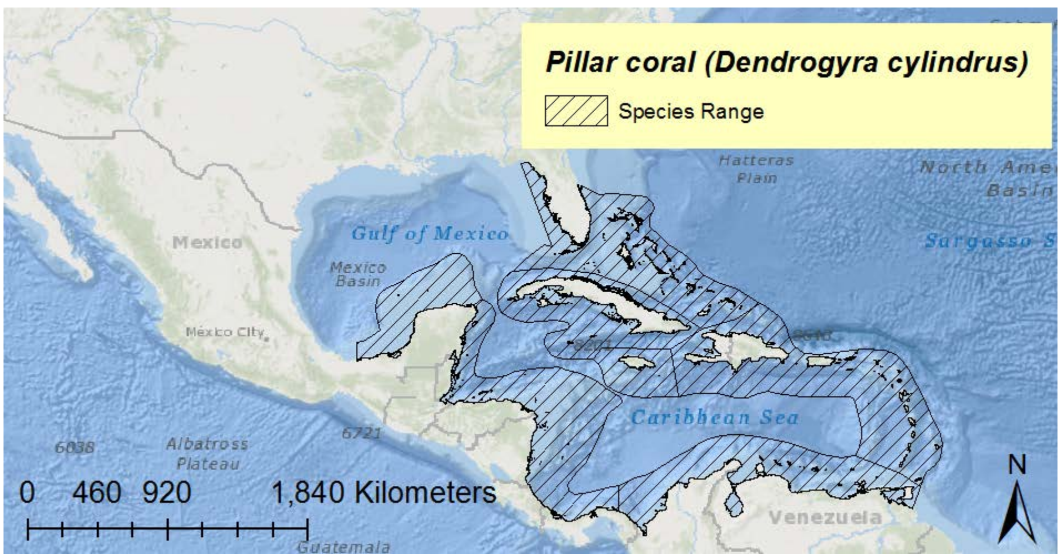

Figure 16. Range map for pillar coral

Pillar corals form tubular columns on top of encrusted foundations. Colonies are generally grey-brown in color and may reach approximately three m in height. Polyps' tentacles remain extended during the day, giving columns a furry appearance.

Brainard et al. (2011) identified a single known colony in Bermuda that is in poor condition. There is fossil evidence of the presence of the species off Panama less than 1,000 years ago, but it has been reported as absent today (Florida Fish and Wildlife Conservation Commission 2013). Pillar coral inhabits most reef environments in water depths ranging from approximately one to twentyfive m, but it is most common in water between approximately five to fifteen m deep (Cairns 1982; Acosta and Acevedo 2006; Goreau and Wells 1967).

Life history

Reported average growth rates for pillar coral have been documented to be approximately 1.8-2.0 cm per year in linear extension within the Florida Keys, compared to 0.8 cm per year as reported in Colombia and Curaçao. Partial mortality rates are size-specific with larger colonies having greater rates. Frequency of partial mortality can be high (e.g., 65 percent of 185 colonies surveyed in Colombia), while the amount of partial mortality per colony is generally low (average of three percent of tissue area affected per colony).

Pillar coral is a gonochoric broadcast spawning species with relatively low annual egg production for its size. The combination of gonochoric spawning with persistently low population densities is expected to yield low rates of successful fertilization and low larval supply. Sexual recruitment of this species is low, and reports indicate juvenile colonies are lacking in the Caribbean. Spawning has been observed to occur several nights after the full moon of August in the Florida Keys (Waddell and Clarke 2008; Neely et al. 2013) and in La Parguera, Puerto Rico (Szmant 1986). Pillar coral can also reproduce asexually by fragmentation following storms or other physical disturbance, but it is uncertain how much storm-generated fragmentation contributes to asexually produced offspring.

Population Dynamics

Pillar coral is uncommon but conspicuous with scattered, isolated colonies and is rarely found in aggregations. Benthic cover is generally less than one percent in monitoring studies. Mean density of pillar coral was approximately 0.5 colonies per ten m^2 in the Florida Keys between 2005 and 2007. In a study of pillar coral demographics at Providencia Island, Colombia, 283 pillar coral colonies were detected in a survey of 1.66 square kilometers for an overall density of approximately 450 colonies per square mile.

Pillar coral ranked least common out of 47 coral species in 2005 and 41 out of 43 species in 2009 in stratified random samples of the Florida Keys. Based on random surveys stratified by habitat type, extrapolated abundance for the Florida Keys was 23,000 ± 23,000 (SE) colonies in 2005 and 25,000 ± 25,000 (SE) colonies in 2009 (Miller et al. 2013). Because these population estimates were based on random sampling, differences between years is more likely a function of sampling effort rather than an indication of population trends. All pillar coral colonies reported in 2005 were in the 70-80 centimeter size class with less than two percent partial mortality. Four years later in 2009, all reported colonies were greater than 90 cm. In 2012, no pillar coral colonies were encountered in 600 surveys from Key Biscayne to Key West, Florida, although the sampling design was not optimized for this species. This species was not reported in the Dry Tortugas in 2006 and 2008, rarely encountered during pilot studies conducted over several years (1999 to 2002), and was the least abundant of 49 coral species encountered (Miller et al. 2013).

Out of 283 pillar coral colonies at Providencia Island, Colombia, 70 colonies resulted from asexual fragmentation and no sexual recruits were observed. Size class distribution was skewed to smaller size classes less than 60 cm in height, and average colony height was approximately 0.73 m (Acosta and Acevedo 2006). During surveys of Utila, Honduras, between 1999 and 2000, pillar coral was sighted in 19.6 percent of 784 surveys and ranked 26th most common in abundance out of 48 coral species (Afzal et al. 2001).

Pillar coral's average percent cover was 0.002 on patch reefs and 0.303 in shallow offshore reefs in annual surveys of 37 sites in the Florida Keys between 1996 and 2003 (Somerfield et al. 2008). At permanent monitoring stations in the USVI, pillar coral has been observed in low abundance at ten of 33 sites and ranged in cover from less than 0.05-0.22 percent where present (Smith 2013). In Dominica, pillar coral comprised less than 0.9 percent cover and was present at 13.3 percent of 31 surveyed sites (Steiner 2003). Of seven fringing reefs surveyed off Barbados, pillar coral was observed on one of them, and cover was 2.7 ± 1.4 percent (Tomascik and Sander 1987). In monitored photo-stations in Roatan, Honduras, cover of pillar coral increased slightly from 1.35 percent in 1996 to 1.67 percent in 1999 and then declined to 0.44 percent in 2003 and to 0.43 percent in 2005 (Riegl et al. 2009). In the USVI, seven percent of 26 monitored colonies experienced total colony mortality between 2005 and 2007, though the very low cover of pillar coral (0.04 percent) remained relatively stable during this time period (Smith et al. 2013).

In stratified random surveys from Palm Beach County to the Dry Tortugas, Florida, between 2005 and 2010, pillar coral was seen only on the ridge complex and mid-channel reefs at densities of approximately one and 0.1 colonies per approximately ten m^2, respectively (Burman et al. 2012). Average number of pillar coral colonies in remote reefs off southwest Cuba was 0.013 ± 0.045 colonies per approximately ten m transects, and the species ranked sixth rarest out of 38 coral species (Alcolado et al. 2010). In surveys of the upper Florida Keys in 2011, pillar coral was the second rarest out of 37 coral species and encountered at one percent of sites (Miller et al. 2011).

Information on pillar coral is most extensive for Florida. Pillar coral ranked as the least abundant to third least abundant coral species in stratified random surveys of the Florida Keys between 2005 and 2009 and was not encountered in surveys in 2012 (Miller et al. 2013). Pillar coral was seen only on the ridge complex and mid-channel reefs at densities of approximately one and 0.1 colonies per ten m^2 (approximately 100 ft^2), respectively, between 2005 and 2010 in surveys from West Palm Beach to the Dry Tortugas (Burman et al. 2012). In surveys conducted between 1999 and 2016 from Palm Beach to the Dry Tortugas, pillar coral was present at 2 percent of sites surveyed and ranged in density from 0 to 0.4 colonies per m^2 with an average density of 0.004 colonies per ten m^2 (approximately 100 ft^2; NOAA, National Coral Reef Monitoring Program). In 2014, there were 714 known colonies of pillar coral along the Florida reef tract from southeast Florida to the Dry Tortugas. By 2017, many of these colonies had suffered tissue loss, and over half (57 percent) suffered complete mortality due to disease, most likely associated with multiple years of warmer than normal temperatures (K. Neely and C. Lewis, unpublished data). The majority of these colonies were lost from the northern portion of the reef tract (Figure 17).

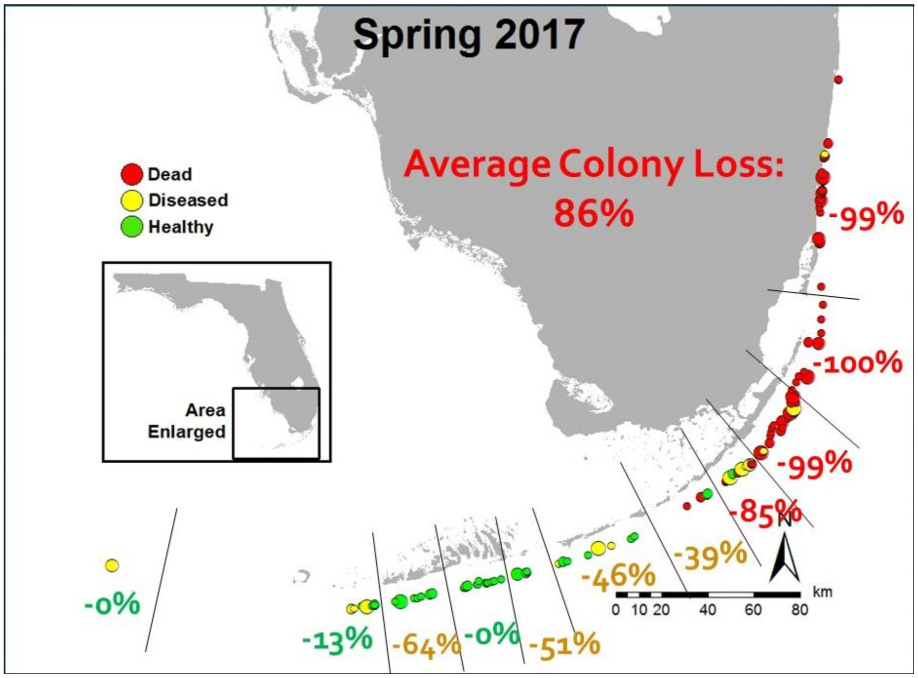

Figure 17. Condition of known pillar coral colonies in Florida between 2014 and 2017 (Figure courtesy of K. Neely and C. Lewis)

Distribution

There are at least tens of thousands of pillar coral colonies present in the Florida Keys based on population estimates. Absolute population abundance is higher than the estimate from this location given the presence of this species in many other locations throughout its range. Although there is evidence of potentially higher population levels in some areas of the Caribbean during the Pleistocene, pillar coral is currently uncommon to rare. Few studies report pillar coral population trends, and the low abundance and infrequent encounter rate in monitoring programs result in small samples sizes. The low coral cover of this species renders monitoring data difficult to extrapolate to realize trends. Therefore, we conclude that pillar coral is naturally uncommon to rare and that trends are unknown.

Status

Pillar coral survival is susceptible to a number of threats, but there is little evidence of population declines thus far. Despite the large number of islands and environments that are included in the species' range, geographic distribution in the highly disturbed Caribbean exacerbates vulnerability to extinction over the foreseeable future because pillar coral is limited to an area with high, localized human impacts and predicted increasing threats. Pillar coral inhabits most reef environments in water depths ranging from one to 25 m, but is naturally rare. Estimates of absolute abundance are at least tens of thousands of colonies in the Florida Keys, and absolute abundance is higher than estimates from this location due to the occurrence of the species in many other areas throughout its range. It is a gonochoric broadcast spawner with observed low sexual recruitment. Its low abundance, combined with its geographic location, exacerbates vulnerability to extinction. This is because increasingly severe conditions within the species' range are likely to affect a high proportion of its population at any given point in time. Also, low sexual recruitment is likely to inhibit recovery potential from mortality events, further exacerbating its vulnerability to extinction. We anticipate that pillar coral is likely to decrease in abundance in the future with increasing threats.

Designated Critical Habitat

No critical habitat has been designated for pillar coral.

Recovery Goals

No final recovery plans currently exists for pillar coral, however a recovery outline was published in 2015. The following contains the recovery goals listed in the document:

Short Term Goals

- Increase understanding of population dynamics, population distribution, abundance, trends, and structure through research, monitoring, and modeling
- Through research, increase understanding of genetic and environmental factors that lead to variability of bleaching and disease susceptibility
- Decrease locally manageable stress and mortality sources (e.g., acute sedimentation, nutrients, contaminants, over-fishing).
- Prioritize implementation of actions in the recovery plan for elkhorn and staghorn corals that will benefit *D. cylindrus*, *M. ferox*, and *Orbicella* spp.

Long Term Goals

- Cultivate and implement U.S. and international measures to reduce atmospheric carbon dioxide concentrations to curb warming and acidification impacts and possibly disease threats.
- Implement ecosystem-level actions to improve habitat quality and restore keystone species and functional processes to maintain adult colonies and promote successful natural recruitment.

6.2.7 Johnson’s Seagrass and Designated Critical Habitat

Johnson’s seagrass (*Halophila johnsonii*) was identified as a species by Eiseman and McMillan in 1980. Prior, *H. johnsonii* was referred to either as *H. decipiens* or *H. baillonis*, but based on morphological, anatomical, and phylogenetic information, it is most closely related to, and most closely resembles, *H. ovalis*, an Indo-Pacific, dioecious species (McMillan 1980:Posluszny, 1990 #958; Freshwater 1999; Waycott et al. 2002). NMFS listed Johnson’s seagrass as threatened under the ESA on September 14, 1998 (63 FR 49035).

Johnson’s seagrass is a short-statured, shallow-rooted, seagrass species found in the intracoastal waters of southeastern Florida. It is characterized by pairs of linearly shaped leaves, each with a petiole (stalk) formed on the node (portion of the stem from which leaves grow) of a horizontally creeping rhizome (stem). Unbranched roots anchor the rhizome to the substrate at or just below the sediment surface. The leaves have smooth margins and are generally 2-5 cm in length (including the petioles). The distance between leaf pairs (internodes) rarely exceed 3-5 cm.

Life History

Johnson’s seagrass is a perennial species (meaning it lasts for greater than two growing seasons), showing no consistent seasonal or year-to-year pattern based on transect surveys. Johnson’s seagrass occurs in a variety of habitat types, including intertidal wave-washed sandy shoals, flood deltas near inlets, in deep water, in soft mud, and near the mouths of canals and rivers, where presumably water quality is sometimes poor and where salinity fluctuates widely. It is an opportunistic plant that occurs in a patchy, disjunctive distribution from the intertidal zone to depths of approximately two to three m. Johnson’s seagrass is found in a wide range of sediment types, salinities, and water quality conditions (NMFS 2007a).

Information on the species’ distribution and results of limited experimental work suggest that Johnson’s seagrass has a wider tolerance range for salinity, temperature, and optical water quality conditions than other species such as paddle grass, *Halophila decipiens* (Dawes et al. 1989; Kenworthy and Haunert 1991; Gallegos and Kenworthy 1996; Kenworthy and Fonseca 1996; Durako et al. 2003; Torquemada et al. 2005). Johnson’s seagrass has been observed near the mouths of freshwater discharge canals (Gallegos and Kenworthy 1996), in deeper turbid waters of the interior portion of the Indian River Lagoon (Kenworthy 2000; Virnstein and Morris 2007), and in clear water associated with the high energy environments and flood deltas inside ocean inlets (Kenworthy 1993;1997; Virnstein et al. 1997; Heidelbaugh et al. 2000; Virnstein and Morris 2007). It can colonize and persist in high-tidal energy environments and has been observed where tidal velocities approach the threshold of motion for unconsolidated sediments (35-40 cm s^{-1}). The persistent presence of high-density, elevated patches of Johnson’s seagrass on flood tidal deltas near inlets suggests that it is capable of sediment stabilization. Intertidal populations of Johnson’s seagrass may be completely exposed at low tides, suggesting high tolerance to desiccation and wide temperature tolerance.

Johnson's seagrass reproduction is believed to be entirely asexual and dispersal is by vegetative fragmentation. Female flowers have been found; however, dedicated surveys in the Indian River Lagoon have not discovered male flowers, fertilized ovaries, fruits, or seeds (Jewett-Smith et al. 1997; NMFS 2007a; Hammerstrom and Kenworthy 2002). Searches throughout the entire range of Johnson's seagrass have produced the same results, suggesting either that the species does not reproduce sexually or that the male flowers are difficult to observe or describe, as noted for other *Halophila* species (Kenworthy 1997). Surveys to date indicate that the incidence of female flowers appears to be much higher near the inlets leading to the Atlantic Ocean.

Population Dynamics

Throughout its range, Johnson's seagrass occurs in dynamic and disjunctive patches. It spreads rapidly, growing horizontally from dense apical meristems with leaf pairs having short life spans (Kenworthy 1997). Kenworthy suggested that the observed horizontal spreading, rapid growth patterns, and high biomass turnover could explain the dynamic patches observed in distribution studies of this species. While patches may colonize quickly, they may also disappear rapidly. Sometimes they will disappear for several years and then re-establish, a process referred to as "pulsating patches" (Heidelbaugh et al. 2000; Virnstein and Morris 2007; Virnstein and Hall 2009). Mortality, or the disappearance of patches, can be caused by a number of processes, including burial from bioturbation and sediment deposition (Heidelbaugh et al. 2000), erosion, herbivory, desiccation, and turbidity. In the absence of sexual reproduction, one possible explanation for the pulsating patches is dispersal and re-establishment of vegetative fragments, a process that commonly occurs in aquatic plants and has been demonstrated in other seagrasses (Philbrick and Les 1996; Di Carlo et al. 2005), and was also confirmed by experimental mesocosm[8] studies with Johnson's seagrass (Hall et al. 2006).

Two survey programs have monitored the presence and abundance of Johnson's seagrass within its range. One program, conducted by the St. Johns River Water Management District (SJRWMD) since 1994, continues to survey the northern section of the species' geographic range between Sebastian Inlet and Jupiter Inlet (Virnstein and Morris 2007; Virnstein and Hall 2009). The second survey, initiated in 2006, monitored the southern range of the species between Jupiter Inlet and Virginia Key in Biscayne Bay (Kunzelman 2007) annually through 2012. Since the last status review (NMFS 2007a), there has not been any reported reduction in the geographic range of the species but rather a slight increase in the known northern range has been observed (Virnstein and Hall 2009).

[8] A mesocosm is an experimental tool that brings a small part of the natural environment under controlled conditions.

Based on the results of the southern transect sampling, it appears there is a relatively continuous, although patchy, distribution of the species from Jupiter Inlet to Virginia Key (NMFS 2007a). The largest reported contiguous patch of Johnson's seagrass in the southern range was observed in Lake Worth Lagoon and was estimated to be 30 acres (Kenworthy 1997). The presence of Johnson's seagrass in northern Biscayne Bay (north of Virginia Key) is well documented. There have been no reports of this species further south of the currently known southern distribution. Findings from the southern transect sampling (summer 2006 and winter 2007) show little difference in the species' frequency or abundance between the summer and winter sampling period. The lower frequencies of Johnson's seagrass occurred at those sites where larger-bodied seagrasses (e.g., turtle grass, *Thalassia testudinum,* and manatee grass, *Syringodium filiforme*) were more abundant (NMFS 2007a).

This is a rare species; however, it can be abundant where it does occur. Based on the results of the southern transect sampling by the SJWMD, it appears that, although Johnson's seagrass is disjunctively distributed and patchy, there is some continuity in the southern distribution, at least during periods of relatively good environmental conditions and no significant large-scale disturbances (NMFS 2007a).

Within the Indian River Lagoon in the northern section of the species' range, Johnson's seagrass was found associated with other seagrass species or growing alone in the intertidal, and more commonly, at the deep edge of some transects in water depths down to 180 cm. Fixed-site transect surveys, conducted between 1995 and 2017, indicate variable occurrence of Johnson's seagrass through time but a clear decline since 2010, apparently associated with several years of poor water quality. The decrease in occurrence also corresponded with a decline in percent coverage based on monitoring data from the SJRWMD.

Distribution

Johnson's seagrass has a narrow geographical range and has only been documented along approximately 200 km of coastline in southeastern Florida. *H. johnsonii* occurs just north of Sebastian Inlet (28°01'58.1"N, -80°33'02.7"W) south to Virginia Key (25°44'52.18"N, -80°08'38.45"W; Figure 18). This apparent endemism (uniqueness to a particular area) suggests that Johnson's seagrass has the most limited geographic distribution of any seagrass species in the world. Since the listing in 1998, there have been no observed reductions in the species' geographic extent. However, the SJRWMD recently observed *H. johnsonii* 21 km north of Sebastian Inlet on the western shore of the Indian River Lagoon; a discovery that slightly extends the species' previously known range limit (Virnstein and Hall 2009).

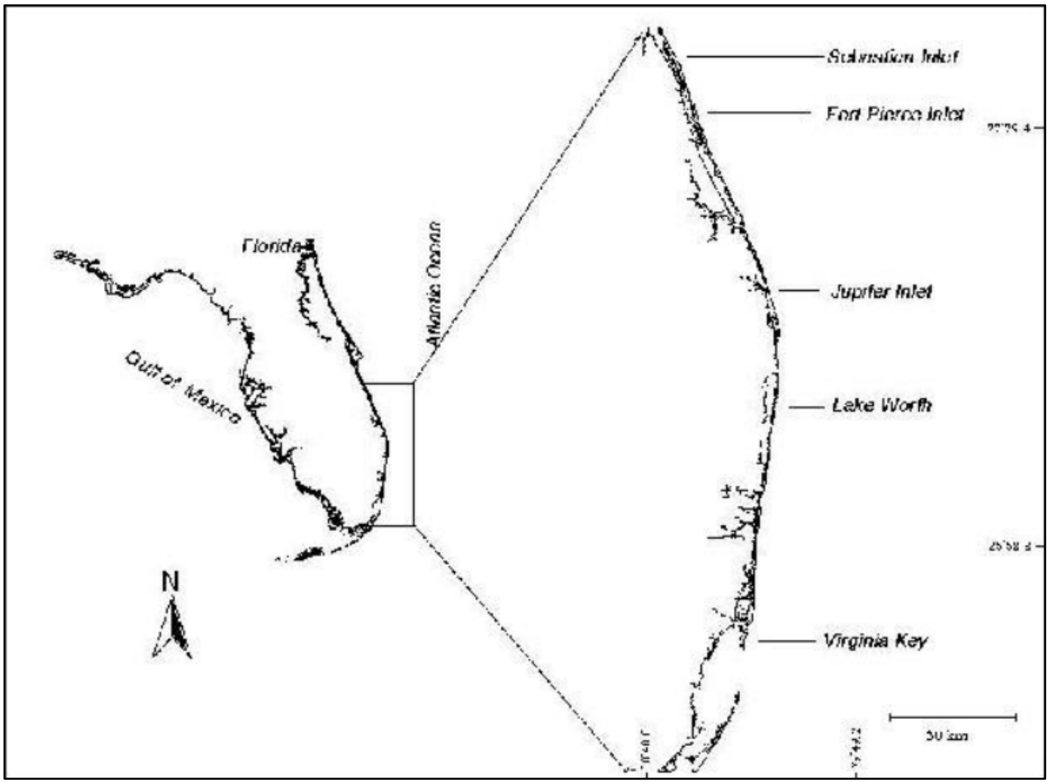

Figure 18. Geographic range of Johnson's seagrass: Sebastian Inlet to northern Virginia Key (Kenworthy 1997)

Status

Observations by researchers have suggested that Johnson's seagrass exploits unstable environments or newly-created unvegetated patches by exhibiting fast growth and support for all local ramets in order to exploit areas in which it could not otherwise compete. It may quickly recruit to locally uninhabited patches through prolific lateral branching and fast horizontal growth. While these attributes may allow it to compete effectively in periodically disturbed areas, if the distribution of this species becomes limited to stable areas it may eventually be outcompeted by more stable-selected plants represented by the larger-bodied seagrasses (Durako et al. 2003). In addition, the physiological attributes of Johnson's seagrass may limit growth (i.e., spreading) over large areas of substrate if the substrate is somehow altered (e.g., dredged to a depth that would preclude future recruitment of Johnson's seagrass); therefore, its ability to recover from widespread habitat loss may be limited. The clonal and reproductive growth characteristics of Johnson's sea grass result in its distribution being patchy, non-contiguous, and temporally fluctuating. These attributes suggest that colonization between broadly disjunctive areas is likely difficult and that the species is vulnerable to becoming endangered if it is removed from large areas within its range by natural or anthropogenic means.

The decline in Johnson's seagrass in the northern portion of its range began just prior to several years of poor water quality involving a persistent drought 2009-2011, a phytoplankton "superbloom" in 2011, and a brown tide event in 2014. The "superbloom" exceeded any past bloom events in both geographic scale, bloom intensity, and duration, creating a decline in water clarity and a significant seagrass die-off. The persistent poor water quality has affected all seagrass species in the Indian River Lagoon and recovery of seagrasses will depend on improved water quality. A consortium of environmental agencies developed a plan to investigate the "superbloom" in an effort resolve lingering water quality issues in the Indian River Lagoon (SJRWMD, 2012).

Designated Critical Habitat

NMFS designated Johnson's seagrass critical habitat on April 5, 2000 (65 FR 17786; see also 50 CFR 226.213; Figure 19). The specific areas occupied by Johnson's seagrass and designated by NMFS as critical habitat are those with one or more of the following criteria:

1. Locations with populations that have persisted for ten years
2. Locations with persistent flowering populations
3. Locations at the northern and southern range limits of the species
4. Locations with unique genetic diversity
5. Locations with a documented high abundance of Johnson's seagrass compared to other areas in the species' range

Ten areas (Units) within the range of Johnson's seagrass (approximately 200 km of coastline from Sebastian Inlet to northern Biscayne Bay, Florida) are designated as Johnson's seagrass critical habitat (Table 7). The total range-wide acreage of critical habitat for Johnson's seagrass is roughly 22,574 ac (NMFS 2002).

Table 7. Designated Critical Habitat Units for Johnson's Seagrass

Unit A	A portion of the Indian River, Florida, north of the Sebastian Inlet Channel
Unit B	A portion of the Indian River, Florida, south of the Sebastian Inlet Channel
Unit C	A portion of the Indian River Lagoon, Florida, in the vicinity of the Fort Pierce Inlet
Unit D	A portion of the Indian River Lagoon, Florida, north of the St. Lucie Inlet
Unit E	A portion of Hobe Sound, Florida, excluding the federally marked navigation channel of the Intracoastal Waterway
Unit F	A portion of the south side of Jupiter Inlet, Florida
Unit G	A portion of Lake Worth, Florida, north of Bingham Island
Unit H	A portion of Lake Worth Lagoon, Florida, located just north of the Boynton Inlet
Unit I	A portion of northeast Lake Wyman, Boca Raton, Florida, excluding the federally marked navigation channel of the Intracoastal Waterway
Unit J	A portion of northern Biscayne Bay, Florida, including all parts of the Biscayne Bay Aquatic Preserve excluding the Oleta River, Miami River, and Little River beyond their mouths, the federally marked navigation channel of the Intracoastal Waterway, and all existing federally authorized navigation channels, basins, and berths at the Port of Miami to the currently documented southernmost range of Johnson's seagrass, Central Key Biscayne

NMFS identified four habitat features essential for the conservation of Johnson's seagrass: (1) adequate water quality, defined as being free from nutrient over-enrichment by inorganic and organic nitrogen and phosphorous or other inputs that create low oxygen conditions; (2) adequate salinity levels, indicating a lack of very frequent or constant discharges of fresh or low-salinity waters; (3) adequate water transparency, which would allow sunlight necessary for photosynthesis; and (4) stable, unconsolidated sediments that are free from physical disturbance. All four essential features must be present in an area for it to function as critical habitat for Johnson's seagrass.

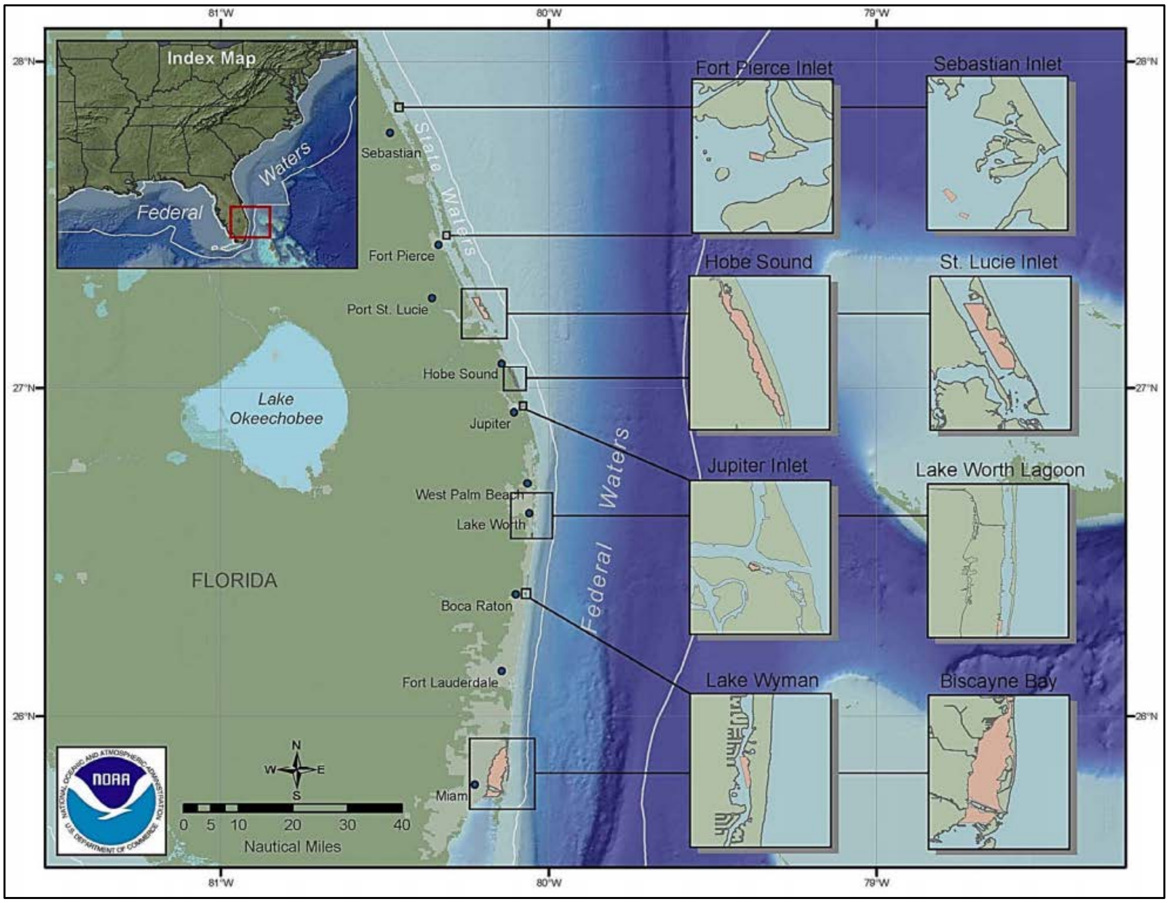

Figure 19. Johnson's Seagrass Designated Critical Habitat Units

Recovery Goals

See the 2002 Final Recovery Plan for Johnson's Seagrass (*Halophila johnsonii* Eiseman) for complete down listing/delisting criteria for each of the respective recovery goals (NMFS 2002).The following recovery objectives were identified for Johnson's seagrass in the Recovery Plan:

1. The species' present geographic range remains stable for at least ten years or increases
2. Self-sustaining populations are present throughout the range at distances less than or equal to the maximum dispersal distance to allow for stable vegetative recruitment and genetic diversity
3. Populations and supporting habitat in its geographic range have long-term protection (through regulatory action or purchase acquisition).

6.2.8 Bocaccio Puget Sound/Georgia Basin Distinct Population Segment and Designated Critical Habitat

The bocaccio is a long-lived, large species of rockfish, occupying the eastern Pacific Ocean in waters from California to Alaska. Puget Sound/Georgia Basin DPS bocaccio are those that reside in the Puget Sound/Georgia Basin (Figure 20).

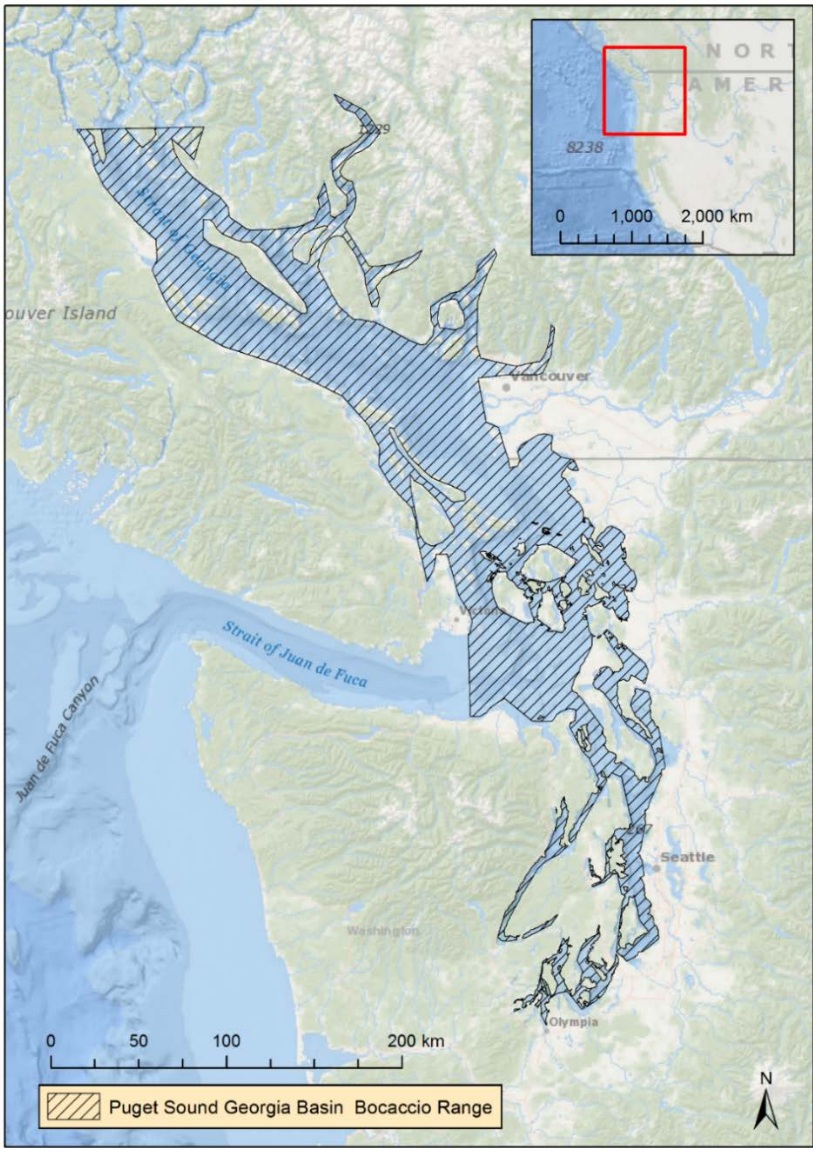

Figure 20. Map Identifying the Range of the Puget Sound/Georgia Basin DPS of Bocaccio

Bocaccio are a large (three ft/one m) Pacific rockfish, olive to burnt orange-brown, with a distinctively long jaw. The Puget Sound/Georgia Basin DPS was first listed as endangered by NMFS on April 28, 2010. The listing was updated on January 23, 2017, when NMFS amended the listing description to include fishes residing within the Puget Sound/Georgia Basin rather than fishes originating from the Puget Sound/Georgia Basin.

Life History

Bocaccio larvae and young juveniles tend to be found in offshore regions (one to 148 km offshore), associated with the surface and occasionally with floating kelp mats (NMFS 2016h). As adults, fishes move into waters 18 to 30 m deep and occupy rocky reefs (Feder et al. 1974; Carr 1983; Eschmeyer et al. 1983; Johnson 2006; Love and Yoklavich 2008). As adults, bocaccio may be found in depths of 12 to 478 m, but tend to remain in shallow waters on the continental shelf (20 to 250 m), still associating mostly with reefs or other hard substrate, but may move over mud flats.

Bocaccio are live-bearers with internal fertilization. Once females become mature (at 54 to 61 cm total length), they produce 20,000 to 2.3 million eggs annually, with the number increasing as females age and grow larger (Hart 1973; Echeverria 1987; Love et al. 2002). However, either sex has been known to attain sexual maturity as small as 35 cm or three years of age. In recent years as populations have declined, average age at sexual maturity may have declined as well (Echeverria 1987; Hart 1973; Love et al. 2002; MacCall and He 2002). Mating occurs between August and November, with larvae born between January and April (NMFS 2016h).

Upon birth, bocaccio larvae measure four to five mm in length. These larvae move into pelagic waters as juveniles when they are 1.5 to three cm and remain in oceanic waters from 3.5 to 5.5 months after birth (usually until early June), where they grow at approximately 0.5 to one millimeter per day (NMFS 2016h). However, growth can vary from year-to-year (Woodbury and Ralston 1991). Once individuals are three to four cm in length, they return to nearshore waters, where they settle into bottom habitats. Females tend to grow faster than males, but fishes may take five years to reach sexual maturity (MacCall 2003). Individuals continue to grow until they reach maximum sizes of 91 cm, or 9.6 kilograms, at an estimated maximum age of 50 years (Piner et al. 2006; Eschmeyer et al. 1983; Halstead et al. 1990; Andrews et al. 2005; Ralston and Ianelli 1998; Love et al. 2002). Prey of bocaccio vary with fish age, with bocaccio larvae starting with larval krill, diatoms, and dinoflagellates. Pelagic juveniles consume fish larvae, copepods, and krill, while older, nearshore juveniles and adults prey upon rockfishes, hake, sablefish, anchovies, lanternfish, and squid (Reilly et al. 1992; Love et al. 2002).

Population Dynamics

There is no current population abundance estimate for the Puget Sound/Georgia Basin DPS bocaccio. There is a lack of long-term information on this DPS for bocaccio abundance, although among rockfish of the Puget Sound, bocaccio appear to have undergone a particular decline. This was likely because of the removal of the largest, most fecund individuals of the population due to overfishing and the frequent failure of recruitment classes, possibly because of unfavorable climactic/oceanographic conditions (MacCall and He 2002). The rate of decline for rockfish in

Puget Sound has been estimated at 3.1 to 3.8 percent annually for the period 1977 to 2014 (NMFS 2016h).

Genetic Diversity

Puget Sound/Georgia Basin DPS bocaccio are distinct from bocaccio elsewhere in its range, likely due to its inhabitance of a geographically isolated area. There is no genetic information available for bocaccio in Puget Sound/Georgia Basin (NMFS 2016h).

Distribution

Puget Sound/Georgia Basin bocaccio occupy the inland marine waters east of the Strait of Juan de Fuca and south of the northern Strait of Georgia.

Status

Bocaccio resistance to depletion and recovery is also hindered by demographic features (Love et al. 1998). Bocaccio are long-lived fishes, taking several years to reach sexual maturity and becoming more fecund with age (Dorn 2002). As harvesting targeted the largest individuals available, bocaccio have become less capable of recovering population numbers (Love et al. 1998). At present, in the complete absence of directed or bycatch fishing pressure, it is estimated that bocaccio populations would have to have frequent good recruitment to restrain their present decline (Tolimieri and Levin 2005). In addition, bocaccio reproduction appears to be characterized by frequent recruitment failures, punctuated by occasional high success years (Love et al. 1998; MacCall and He 2002). Over the past 30 years, 1977, 1984, and 1988 are the only years in which recruitment appears to have been significant successes. Recruitment success appears to be linked to oceanographic/climatic patterns and may be related to cyclic warm/cool ocean periods, with cool periods having greater success (Love et al. 1998; MacCall 1996; Moser et al. 2000; Sakuma and Ralson 1995). Harvey et al. (2006) suggested that bocaccio may have recently diverted resources from reproduction, potentially resulting in additional impairment to recovery.

Larval rockfish (that include bocaccio and yelloweye rockfish) have been documented in Puget Sound (Greene and Godersky 2012). Oceanographic conditions within many areas of Puget Sound likely result in the larvae staying within the basin where they are born rather than being more broadly dispersed by tides and currents (Drake et al. 2010). Areas with floating and submerged kelp species support the highest densities of juvenile bocaccio (Carr 1983; Haldorson and Richards 1987; Matthews 1990; Love et al. 2002).

Designated Critical Habitat

Critical habitat for the Puget Sound/Georgia Basin DPS for bocaccio, canary rockfish, and yelloweye rockfish was finalized in 2014 (79 FR 68041). The critical habitat designation was updated in 2017 when canary rockfish were delisted. The specific areas designated for bocaccio include approximately 1,184.75 square miles (3,068.5 square km) of marine habitat in Puget Sound, Washington (Figure 21). Designated habitat was divided into two units: nearshore, to support juveniles, and deeper, rocky habitat for adults. Features essential for adult bocaccio (greater than 30 m deep) include sufficient prey resources, water quality, and rocks or highly rugose habitat. For juvenile bocaccio, features essential for their conservation include sufficient prey resources and water quality. The Puget Sound Naval Shipyard at Naval Base Kitsap was excluded from critical habitat designation for Puget Sound/Georgia Basin bocaccio.

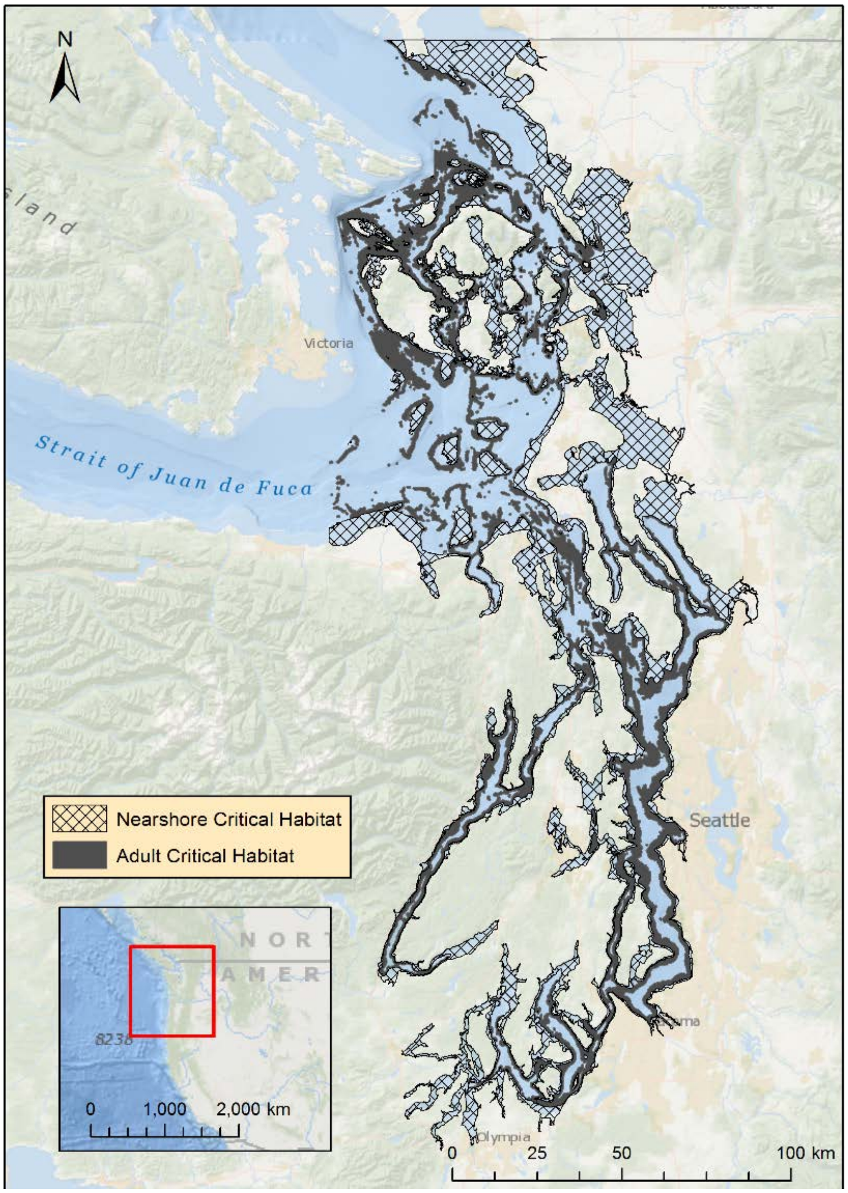

Figure 21. Map of Designated Critical Habitat for the Bocaccio and Yelloweye Rockfish Puget Sound/Georgia Basin DPSs

Recovery Goals

See the 2016 Draft Rockfish Recovery Plan: Puget Sound/Georgia Basin yelloweye rockfish (*Sebastes ruberrimus*) and bocaccio (*Sebastes paucispinis*), for complete down listing/delisting criteria for each of their respective recovery goals (NMFS 2016h) The following items were the top recovery objectives identified to support in the Draft Recovery Plan:

1. Improve our knowledge of the current and historical status of the yelloweye rockfish and bocaccio and their habitats.
2. Reduce or eliminate existing threats to listed rockfish from fisheries/anthropogenic mortality.
3. Reduce or eliminate existing threats to listed rockfish habitats and restore important rockfish habitat.

6.2.9 Yelloweye Rockfish Puget Sound/Georgia Basin Distinct Population Segment and Designated Critical Habitat

Yelloweye rockfish occur throughout most of the eastern Pacific Ocean ranging from northern Baja California to the Aleutian Islands, Alaska. The Puget Sound/Georgia Basin DPS is located along the coastal/inlet waters off the state of Washington and province of British Columbia and is the only population listed under the ESA (Figure 22).

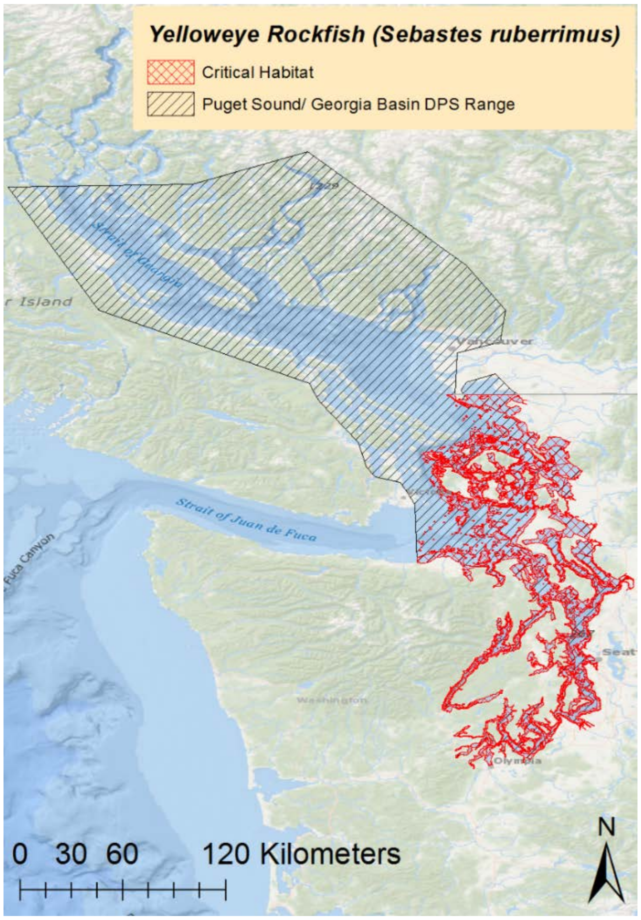

Figure 22. Geographic Range and Designated Critical Habitat for Yelloweye Rockfish, Puget Sound/Georgia Basin DPS

Yelloweye rockfish is one of the largest species belonging to the genus *Sebastes*. They are orange-red to orange-yellow in color and may have black fin tips with bright yellow eyes. Adults usually have a light to white stripe on the lateral line; juveniles have two light stripes, one on the lateral line and a shorter one below the lateral line (Yamanaka et al. 2006).

Puget Sound/Georgia Basin yelloweye rockfish were listed on the ESA as threatened on April 28, 2010. We used information available in the status review (NMFS 2010b) and recent scientific publications to summarize the life history, population dynamics, and status of the species, as follows.

Population Dynamics

The apparent steep reduction of ESA-listed rockfish in Puget Sound proper (and their consequent fragmentation) has led to concerns about the viability of these populations (Drake et al. 2010). Recreationally caught yelloweye rockfish in the 1970s spanned a broad size range. By the 2000s, fewer older fish in the population were observed (Drake et al. 2010). However, overall fish numbers in the database were also much lower, making it difficult to determine if clear size truncation occurred. With age truncation, the reproductive burden may have shifted to younger and smaller fish. This could alter larval release timing and condition, which may create a mismatch with habitat conditions and potentially reduce offspring viability (Drake et al. 2010).

Yelloweye rockfish were 2.4 percent of the rockfish harvest in the North Sound during the 1960s, 2.1 percent of the harvest during the 1980s, and further decreased to an average of one percent from 1996 to 2002 (Palsson et al. 2009). In Puget Sound proper, yelloweye rockfish were 4.4 percent of the rockfish harvest during the 1960s, 0.4 percent during the 1980s, and 1.4 percent from 1996 to 2002 (Palsson et al. 2009). By the 2000s, evidence of fewer older fish in the population prevailed. Since overall fish numbers in the database were also much lower, it is difficult to determine if size truncation occurred.

In 2008, fishery-independent estimate surveys conducted by WDFW estimated that 47,407 yelloweye rockfish are present in the in the San Juan Islands basin. Since this estimate only includes the San Juan Island basin, this estimate is considered a conservative estimate of Puget Sound/Georgia Basin yelloweye rockfish abundance. Though yelloweye rockfish were detected via bottom trawl surveys in Puget Sound proper, we do not consider the WDFW estimate of 600 fish to be a complete estimate and were not included. Since juvenile yelloweye rockfish are less dependent on rearing in shallow nearshore environments than canary rockfish and bocaccio, the drop camera surveys were not expected to result in any detections.

Productivity measures a population’s growth rate through all or a portion of its life cycle. Yelloweye rockfish life-history traits suggest generally low inherent productivity levels because they are long-lived, mature slowly, and have sporadic episodes of successful reproduction (Drake et al. 2010; Tolimieri and Levin 2005). Adult yelloweye rockfish typically occupy relatively small ranges (Love et al. 2002) and may not move to find suitable mates. So, as the density of mature fish has decreased, productivity may have also been impacted by Allee effects. Further, past commercial and recreational fishing may have depressed the DPS to a threshold beyond which optimal productivity is unattainable (Drake et al. 2010). In addition, historic over-fishing may have had dramatic impacts on population size or age structure.

Genetic Diversity

Results from a recent genetic study comparing yelloweye rockfish individuals from within the Puget Sound/Georgia Basin DPS (n=52) to those outside the DPS (n=52) provided multiple results (Tonnes et al. 2016). First, yelloweye rockfish in inland Canadian waters as far north as Johnstone Strait were genetically similar to those within the Puget Sound/Georgia Basin DPS. Currently, these areas are not included within the boundaries of the DPS. Second, a significant genetic difference exists between individuals (1) outside the DPS and (2) within the DPS and north of the DPS in inland Canadian waters to as far north as Johnstone Strait. Lastly, individuals within Hood Canal are genetically differentiated from the rest of the DPS; thereby indicating a previous unknown degree of population differentiation within the DPS (Tonnes et al. 2016).

Distribution

The leading factors affecting diversity are the relatively small home ranges of juveniles and subadults (Love et al. 2002) and low population size of all life stages. Yelloweye rockfish spatial structure and connectivity are likely threatened by the apparently severe reduction of fish numbers throughout Hood Canal and South Puget Sound. At 2,330 square km, Puget Sound is a small geographic area compared with the entire yelloweye rockfish range in the northeastern Pacific.

Status

Puget Sound/Georgia Basin yelloweye rockfish abundance is much less than it was historically. The fish face several threats including bycatch in commercial and recreational harvest, non-native species introductions, and habitat degradation. Results from a recent genetic study comparing yelloweye rockfish individuals from within the Puget Sound/Georgia Basin DPS to those outside the DPS concluded that a significant genetic difference exists between individuals (1) outside the DPS and (2) within the DPS and north of the DPS in inland Canadian waters to as far north as Johnstone Strait (Tonnes et al. 2016). Further, individuals within Hood Canal are genetically differentiated from the rest of the Puget Sound/Georgia Basin DPS; thereby indicating a previous unknown degree of population differentiation within the DPS (Tonnes et al. 2016). NMFS has determined that this DPS is likely to be in danger of extinction in the foreseeable future throughout all of its range; and in its 2016 status review (Tonnes et al. 2016), NMFS has recommended no change in the Puget Sound/Georgia Basin yelloweye rockfish's threatened classification.

Status of the Species in the Action Area

Larval rockfish (including yelloweye rockfish) have been documented in Puget Sound (Greene and Godersky 2012). Oceanographic conditions within many areas of Puget Sound likely result in the larvae staying within the basin where they are born rather than being more broadly dispersed by tides and currents (Drake et al. 2010). Juvenile yelloweye rockfish are most frequently observed in waters deeper than 30 m (98 ft), which is also near the upper depth range of adults (Yamanaka et al. 2006), although adults generally occupy habitats off the coast (Love et al. 2002).

Designated Critical Habitat

Critical habitat was designated for Puget Sound/Georgia Basin yelloweye rockfish on November 13, 2014, when NMFS published a final rule in the Federal Register (79 FR 68041). The critical habitat in the U.S. is spread amongst five interconnected, biogeographic basins (San Juan/Strait of Juan de Fuca basin, Main basin, Whidbey basin, South Puget Sound, and Hood Canal) based upon presence and distribution of adult and juvenile yelloweye rockfish, geographic conditions, and habitat features (Figure 22). The Puget Sound Naval Shipyard at Naval Base Kitsap was excluded from critical habitat designation for Puget Sound/Georgia Basin yelloweye rockfish.

Recovery Goals

There is no federal recovery plan for the Puget Sound/Georgia Basin yelloweye rockfish at this time.

6.2.10 Atlantic Sturgeon and Designated Critical Habitat

Sturgeon are among the most primitive of the bony fishes. The Atlantic sturgeon is a long-lived (approximately 60 years), late maturing, iteroparous, estuarine dependent species (ASSRT 2007; Dadswell 2006). Atlantic sturgeon are anadromous, spawning in freshwater but spending most of their subadult and adult life in the marine environment. They can grow to approximately 14 ft long and can weigh up to 800 pounds. Atlantic sturgeon are bluish-black or olive brown dorsally (on their back) with paler sides, a white belly, and have five major rows of dermal "scutes.”

Five DPSs of Atlantic sturgeon were listed under the ESA in 2012. The Gulf of Maine DPS is listed as threatened while the New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs are listed as endangered (Figure 23).

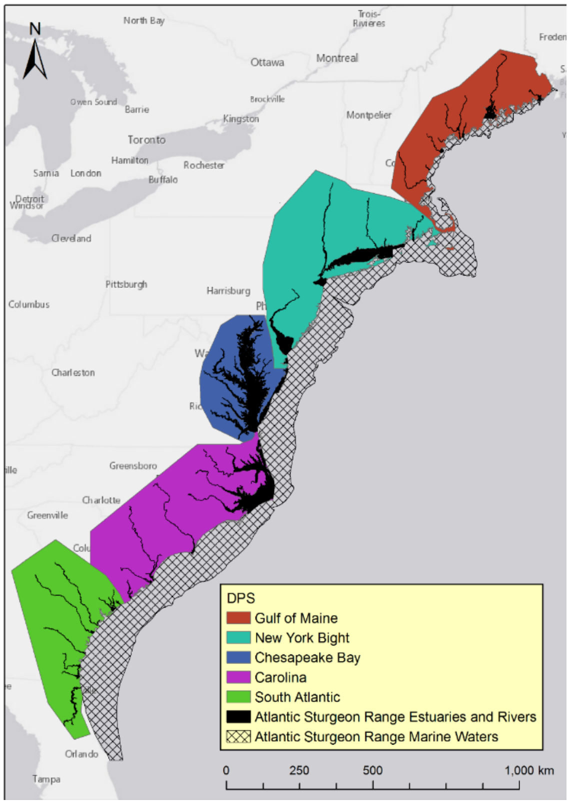

Figure 23. Range and Boundaries of the Five Atlantic Sturgeon DPSs

This section provides general information on the Atlantic sturgeon coast-wide population, including information about the species life history, population dynamics, and status. The subsections that follow provide information and characteristics particular to each of the five listed DPSs of Atlantic sturgeon.

Life History

The general life history pattern of Atlantic sturgeon is that of a long lived, late maturing, iteroparous, anadromous species. Atlantic sturgeon spawn in freshwater, but spend most of their subadult and adult life in the marine environment.

Traditionally, it was believed that spawning within all populations occurred during the spring and early summer months. More recent studies, however, suggest that spawning occurs from late summer to early autumn in two tributaries of the Chesapeake Bay (James River and York River, Virginia) and in the Altamaha River, Georgia (Balazik et al. 2012; Hager et al. 2014).

Sturgeon eggs are highly adhesive and are deposited on the bottom substrate, usually on hard surfaces (e.g., cobble; Smith and Clugston 1997). Hatching occurs approximately 94 to 140 hours after egg deposition, and larvae assume a demersal existence (Smith et al. 1980). The yolk sac larval stage is completed in about eight to 12 days, during which time the larvae move downstream to rearing grounds over a six to 12-day period (Kynard and Horgan 2002). During the first half of their migration downstream, movement is limited to nighttime. During the day, larvae use benthic structure (e.g., gravel matrix) as refugia (Kynard and Horgan 2002). During the latter half of migration when larvae are more fully developed, movement to rearing grounds occurs both day and night. The larvae grow rapidly and are four to five and a half inches long at a month old (MSPO 1993). At this size, the young sturgeon bear teeth and have sharp, closely spaced spine-tipped scutes. As growth continues, they lose their teeth, the scutes separate and lose their sharpness.

Juvenile Atlantic sturgeon continue to move downstream into brackish waters, and eventually become residents in estuarine waters. Juvenile Atlantic sturgeon are resident within their natal estuaries for two to six years, depending on their natal river of origin, after which they emigrate as subadults to coastal waters (Dovel 1983) or to other estuaries seasonally (Waldman et al. 2013). Atlantic sturgeon undertake long marine migrations and utilize habitats up and down the East Coast for rearing, feeding, and migrating (Dovel 1983; Bain 1997; Stevenson 1997). Migratory subadults and adults are normally located in shallow (10-50m) nearshore areas dominated by gravel and sand substrate (Stein et al. 2004). Tagging and genetic data indicate that subadult and adult Atlantic sturgeon may travel widely once they emigrate from rivers (Bartron 2007; Wirgin et al. 2015). Once in marine waters, subadults undergo rapid growth (Dovel 1983; Stevenson 1997). Despite extensive mixing in coastal waters, Atlantic sturgeon display high site fidelity to their natal streams.

Atlantic sturgeon have been aged to 60 years (Mangin 1964), but this should be taken as an approximation because the age validation studies conducted to date show ages cannot be reliably estimated after 15 to 20 years (Stevenson and Secor 2000). Vital parameters of sturgeon populations generally show clinal variation with faster growth, earlier age at maturation, and shorter life span in more southern systems. Spawning intervals range from one to five years for male Atlantic sturgeon (Smith 1985; Collins et al. 2000) and three to five years for females (Stevenson and Secor 2000; Schueller and Peterson 2010). Fecundity of Atlantic sturgeon is correlated with age and body size, ranging from approximately 400,000 to eight million eggs (Smith et al. 1982; Van Eenennaam and Doroshov 1998; Dadswell 2006). The average age at which 50 percent of Atlantic sturgeon maximum lifetime egg production is achieved is estimated to be 29 years, approximately three to ten times longer than for most other bony fish species (Boreman 1997).

Atlantic sturgeon feed on mollusks, polychaeta worms, gastropods, shrimps, pea crabs, decapods, amphipods, isopods, and small fishes in the marine environment (Guilbard et al. 2007; Savoy 2007; Collins et al. 2008). The sturgeon "roots" in the sand or mud with its snout, like a pig, to dislodge worms and mollusks that it sucks into its protrusible mouth, along with considerable amounts of mud. The Atlantic sturgeon has a stomach with very thick, muscular walls that resemble the gizzard of a bird. This gizzard enables it to grind such food items as mollusks and gastropods (MSPO 1993).

Population Dynamics

The following is a discussion of the species' population and its variance over time for each DPS.

Gulf of Maine DPS

There are some positive signs for the Gulf of Maine DPS, which include observations of Atlantic sturgeon in rivers from which sturgeon observations have not been reported for many years (i.e., Saco, Presumpscot, and Charles rivers) and potentially higher catch-per-unit-effort levels than in the past (i.e., Kennebec; ASSRT 2007). Precise estimates of population growth rate (intrinsic rates) are unknown due to lack of long-term abundance data.

New York Bight DPS

Precise estimates of population growth rate (intrinsic rates) for the New York Bight DPS are unknown due to lack of long-term abundance data. Long-term juvenile surveys indicate that the Hudson River population supports successful annual year classes since 2000 and the annual production has been stable and/or slightly increasing in abundance (ASSRT 2007). Recently, juvenile Atlantic sturgeon collected in the Connecticut River suggest at least one successful colonizing spawning event may have occurred (Savoy et al. 2017). Around the same time, a dead 213-centimeter Atlantic sturgeon was recovered on the banks of the Connecticut River.

Chesapeake Bay DPS

The Chesapeake Bay once supported at least six historical spawning populations; however, today the bay is believed to support at the most, four to five spawning populations. Precise estimates of population growth rate (intrinsic rates) for the Chesapeake Bay DPS are unknown due to lack of long-term abundance data. The status review team (ASSRT 2007) concluded that the populations in the James and York Rivers are at a moderate and moderately high risk of extinction.

Carolina DPS

Precise estimates of population growth rate (intrinsic rates) for the Carolina DPS are unknown due to lack of long-term abundance data. The status review team (ASSRT 2007) concluded that the populations in the Roanoke, Tar/Pamlico, Neuse, Waccamaw, and Pee Dee river systems are at a moderate extinction risk and the populations in the Cape Fear and Santee-Cooper river systems are at a moderately high risk of extinction.

South Atlantic DPS

Precise estimates of population growth rate (intrinsic rates) for the South Atlantic DPS are unknown due to lack of long-term abundance data. During the last two decades, Atlantic sturgeon have been observed in most South Carolina coastal rivers, although it is not known if all rivers support a spawning population (Collins 1997). The Altamaha River supports the healthiest Atlantic sturgeon populations in the South Atlantic DPS. In a telemetry study by Peterson et al.

(2008), most tagged adult Atlantic sturgeon were found between river kilometer 215 and 420 in October and November when water temperatures were appropriate for spawning. The status review team (ASSRT 2007) found that, overall, the South Atlantic DPS had a moderate risk (less than 50 percent chance) of becoming endangered over the next 20 years.

Genetic Diversity

Atlantic sturgeon throughout their range exhibit ecological separation during spawning that has resulted in multiple, genetically distinct, interbreeding population segments. Studies have consistently found populations to be genetically diverse and indicate that there are between seven and ten populations that can be statistically differentiated (King et al. 2001; Waldman et al. 2002; Wirgin et al. 2007; Grunwald et al. 2008). However, there is some disagreement among studies, and results do not include samples from all rivers inhabited by Atlantic sturgeon. Recent studies conducted indicate that genetically distinct populations of spring and fall-run Atlantic sturgeon can exist within a given river system (Balazik and Musick 2015; Farrae et al. 2017; Balazik et al. 2017).

Distribution

The Atlantic sturgeon's historic range included major estuarine and riverine systems that spanned from Hamilton Inlet on the coast of Labrador, Canada, to the Saint Johns River in Florida (ASSRT 2007; Smith and Clugston 1997). Atlantic sturgeon have been documented as far south as Bermuda and Venezuela (Lee et al. 1980). Historically, Atlantic sturgeon were present in approximately 38 rivers in the U.S. from St. Croix, Maine, to the Saint Johns River, Florida, of which 35 rivers have been confirmed to have had historic spawning populations. Atlantic sturgeon are currently present in 36 rivers, and spawning occurs in at least 21 of these (ASSRT 2007). Other estuaries along the U.S. Atlantic coast formed by rivers that do not support Atlantic sturgeon spawning populations may still be important as rearing habitats.

Gulf of Maine DPS

The Gulf of Maine DPS includes all anadromous Atlantic sturgeons that are spawned in the watersheds from the Maine/Canadian border and, extending southward, all watersheds draining into the Gulf of Maine as far south as Chatham, Massachusetts (Figure 23). The geomorphology of most small coastal rivers in Maine is not sufficient to support Atlantic sturgeon spawning populations, except for the Penobscot and the estuarial complex of the Kennebec, Androscoggin, and Sheepscot rivers. Spawning still occurs in the Kennebec and Androscoggin Rivers, and may occur in the Penobscot River. Atlantic sturgeon have more recently been observed in the Saco, Presumpscot, and Charles rivers.

New York Bight DPS

The natal river systems of the New York Bight DPS span from the Connecticut River south to the Delaware River (Figure 23). The Connecticut River has long been known as a seasonal aggregation area for subadult Atlantic sturgeon, and both historical and contemporary records document presence of Atlantic sturgeon in the river as far upstream as Hadley, Massachusetts (Savoy 1992; Savoy and Pacileo 2003). The upstream limit for Atlantic sturgeon on the Hudson River is the Federal Dam at the fall line, approximately river kilometer 246 (Dovel 1983; Kahnle et al. 2007). In the Delaware River, there is evidence of Atlantic sturgeon presence from the mouth of the Delaware Bay to the head-of-tide at the fall line near Trenton on the New Jersey side and Morrisville on the Pennsylvania side of the River, a distance of 220 river km (Breece et al. 2013).

Of the Navy's origination and destination ports for the action, New York Bight DPS of Atlantic sturgeon are found in the port of Philadelphia on the Delaware River. This port is also part of the critical habitat designation (Unit 4) for the New York Bight DPS (Figure 24).

Chesapeake Bay DPS

The natal river systems of the Chesapeake Bay DPS span from the Susquehanna River south to the James River (Figure 23).

Carolina DPS

The natal river systems of the Carolina DPS span from the Roanoke River, North Carolina south to the Santee-Cooper system in South Carolina (Figure 23). The Carolina DPS ranges from the Santee-Cooper River to the Albemarle Sound and consists of seven extant populations; one population (the Sampit River) is believed to be extirpated.

South Atlantic DPS

The natal river systems of the South Atlantic DPS span from Edisto south to the St. Mary's River (Figure 23). Seventy-six Atlantic sturgeon were tagged in the Edisto River during a 2011 to 2014 telemetry study (Post et al. 2014). Fish entered the river between April and June and were detected in the saltwater tidal zone until water temperature decreased below 25 degrees Celsius. They then moved into the freshwater tidal area, and some fish made presumed spawning migrations in the fall around September to October. Atlantic sturgeon in the Savannah River were documented displaying similar behavior three years in a row—migrating upstream during the fall and then being absent from the system during spring and summer. Forty three Atlantic sturgeon larvae were collected in upstream locations (river kilometer 113 to 283) near presumed spawning locations (Collins 1997).

Status

Atlantic sturgeon were once present in 38 river systems and, of these, spawned in 35 of them. Individuals are currently present in 36 rivers, and spawning occurs in at least 20 of these (ASSRT 2007). The decline in abundance of Atlantic sturgeon has been attributed primarily to the large U.S. commercial fishery that existed for the Atlantic sturgeon from the 1870s through the mid-1990s. The fishery collapsed in 1901 and landings remained at between one to five percent of the pre-collapse peak until the Atlantic States Marine Fisheries Commission placed a two generation moratorium on the fishery in 1998 (ASMFC 1998). The majority of the populations show no signs of recovery, and new information suggests that stressors such as bycatch, ship strikes, and low DO can and do have substantial impacts on populations (ASSRT 2007). Additional threats to Atlantic sturgeon include habitat degradation from dredging, damming, and poor water quality (ASSRT 2007). Climate change related impacts on water quality (e.g., temperature, salinity, DO, contaminants) have the potential to impact Atlantic sturgeon populations using impacted river systems. These effects are expected to be more severe for southern portions of the U.S. range of Atlantic sturgeon (Carolina and South Atlantic DPSs). None of the spawning populations are currently large or stable enough to provide any level of certainty for continued existence of any of the DPSs.

In 2012, NMFS listed the New York Bight and Chesapeake Bay DPSs as endangered and the Gulf of Maine DPS as threatened based on low population sizes and the level of continuing threats such as degraded water quality, habitat impacts from dredging, bycatch in state and federally managed fisheries, and ship strikes. Historically, each of these DPSs likely supported more than 10,000 spawning adults (Secor and Niklitschek 2002; MSPO 1993). The best available data indicate that current numbers of spawning adults for each DPS are one to two orders of magnitude smaller than historical levels (ASSRT 2007; Kahnle et al. 2007). The Carolina and South Atlantic DPSs were estimated to have declined to less than three and six percent of their historical population sizes, respectively (ASSRT 2007). Both of these DPSs were listed as endangered in 2012 due to a combination of habitat curtailment and alteration, bycatch in commercial fisheries, and inadequacy of regulatory mechanisms in ameliorating these impacts and threats. The largest estimated adult Atlantic sturgeon populations are currently found in the Hudson (3,000), Altamaha (1,325), Delaware (1,305), Kennebec (865), Savannah (745), and James (705) Rivers. Published estimates of Atlantic sturgeon juvenile abundance are available in the following river systems: 4,314 age one fish in the Hudson in 1995 (Peterson et al. 2000); 3,656 age 0-1 fish in the Delaware in 2014 (Hale et al. 2016); between 1,072 to 2,033 age 1-2 fish on average from 2004-2007 in the Altamaha (Schueller and Peterson 2010); and 154 age one fish in 2010 in the Satilla (Fritts et al. 2016).

Designated Critical Habitat

NMFS designated critical habitat for each ESA-listed DPS of Atlantic sturgeon in August of 2017 (Figure 24; 82 FR 39160). PBFs determined to be essential for Atlantic sturgeon reproduction and recruitment include (1) suitable hard bottom substrate in low salinity waters for settlement of fertilized eggs, refuge, growth, and development of early life stages, (2) transitional salinity zones for juvenile foraging and physiological development, (3) water of appropriate depth and absent physical barriers to passage, (4) unimpeded movement of adults to and from spawning sites, and (5) water quality conditions that support spawning, survival, growth, development, and recruitment.

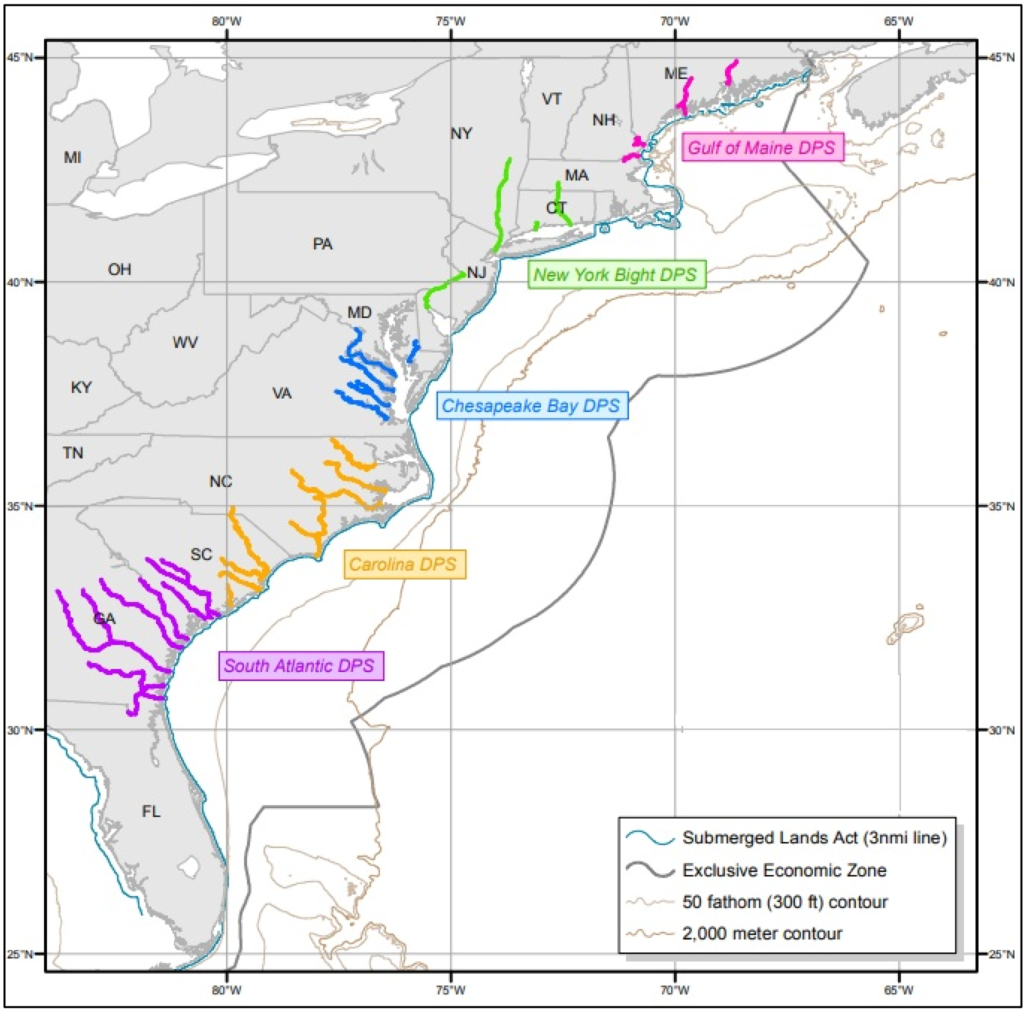

Figure 24. General Map of Critical Habitat for Each DPS of Atlantic Sturgeon

Recovery Goals

Recovery Plans have not yet been drafted for any of the Atlantic sturgeon DPSs.

6.2.11 Gulf Sturgeon and Designated Critical Habitat

The current range of the gulf sturgeon (also known as Gulf of Mexico sturgeon) extends from Lake Pontchartrain in Louisiana east to the Suwannee river system in Florida (Figure 25). Within that range, seven major rivers are known to support reproducing populations: Pearl, Pascagoula, Escambia, Yellow, Choctawhatchee, Apalachicola, and Suwannee (NMFS and USFWS 2009).

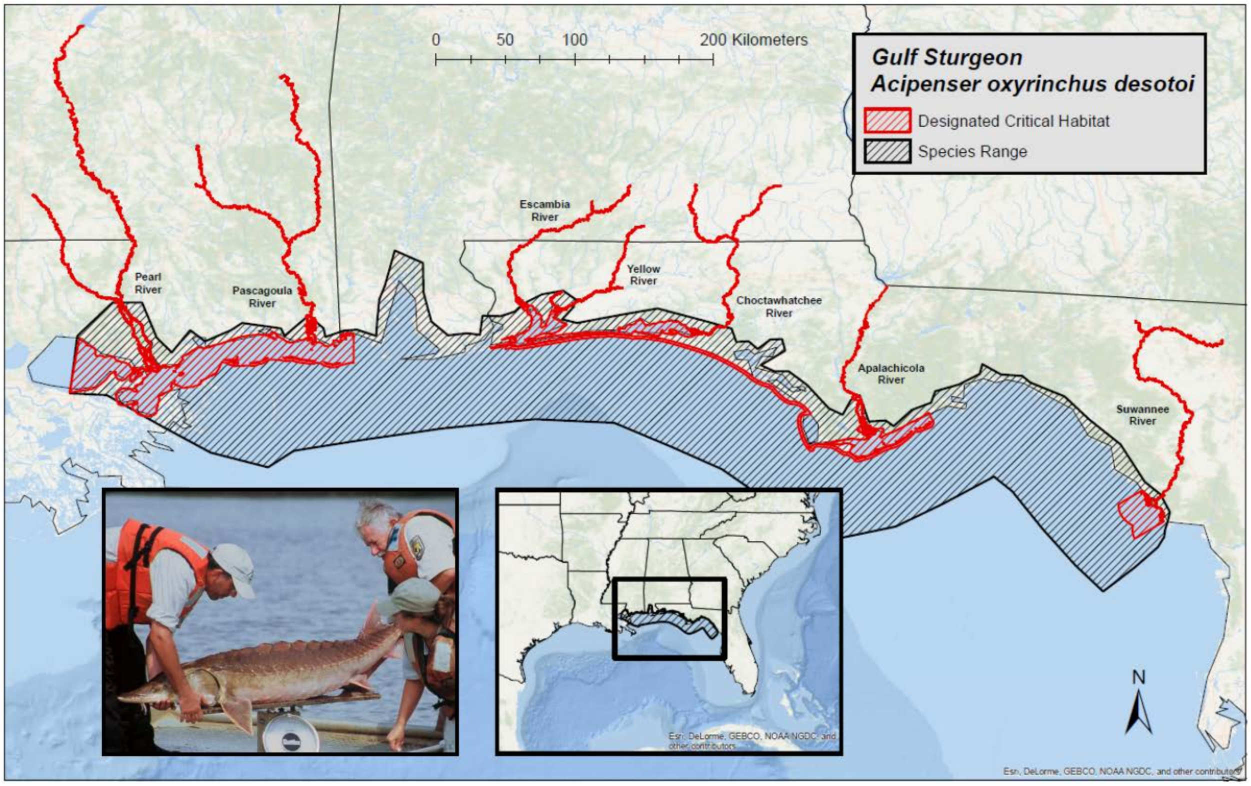

Figure 25. Geographic Range and Designated Critical habitat of the Gulf Sturgeon

Gulf sturgeon are nearly cylindrical fish with an extended snout, vertical mouth, five rows of scutes (bony plates surrounding the body), four barbels (slender, whisker-like feelers anterior to the mouth used for touch and taste), and a heterocercal (upper lobe is longer than lower) caudal fin. Adults range from six to eight ft in length and weigh up to 200 pounds; females grow larger than males (USFWS 2009).

Life History

Gulf sturgeon are long-lived, with some individuals reaching at least 42 years in age. Surveys in the Suwannee River suggest that a more common maximum age may be around 25 years (Sulak and Clugston 1999). Age at sexual maturity for females ranges from eight to 17 years, and for males from seven to 21 years (Huff 1975). In general, gulf sturgeon spawn up-river in spring, spend winter months in near-shore marine environments, and utilize pre- and post-spawn staging and nursery areas in the lower rivers and estuaries (Heise et al. 2004; Heise et al. 2005). There is some evidence of autumn spawning in the Suwannee River, however there is uncertainty as to whether this spawning is due to environmental conditions or represents a genetically distinct population (Randall and Sulak 2012). Gulf sturgeon spawn at intervals ranging from three to five years for females and one to five years for males (Smith 1985; Fox et al. 2000). The spring migration to up-river spawning sites begins in mid-February and continues through May.

Fertilization is external; females deposit their eggs in the upper reaches of and show preference for hard, clean substrate (e.g. bedrock covered in gravel and small cobble).

Upon hatching from their eggs, gulf sturgeon larvae spend the first few days of life sheltered in interstitial spaces at the spawning site (Kynard and Parker 2004). At the onset of feeding, age-0 gulf sturgeon disperse and are often found on shallow sandbars and rippled sand shoals (less than four m depth; Sulak and Clugston 1998). Young-of-the-year spend six to ten months slowly working their way downstream feeding on aquatic insects (e.g., mayflies and caddisflies), worms (oligochaetes), and bivalve mollusks, and arrive in estuaries and river mouths by mid-winter (Sulak and Clugston 1999) where they will spend their next six years developing. After spawning, adult gulf sturgeon migrate downstream to summer resting and holding areas in the mid to lower reaches of the rivers where they may hold until November (Wooley and Crateau 1985). While in freshwater adults lose a substantial amount of their weight, but regain it upon entering the estuaries. Sub adult and non-spawning adults also spend late spring through fall in these holding areas (Foster and Clugston 1997). By early December all adult and sub-adult gulf sturgeon return to the marine environment to forage on benthic (bottom dwelling) invertebrates along the shallow nearshore (two to four m depth), barrier island passes, and in unknown off-shore locations in the gulf (Ross et al. 2009; Carr et al. 1996; Fox et al. 2002; Huff 1975). Juvenile gulf sturgeon overwinter in estuaries, river mouths, and bays; juveniles do not enter the nearshore/offshore marine environments until around age six (Sulak and Clugston 1999). Gulf sturgeon show a high degree of river-specific fidelity (Rudd et al. 2014). Adult and sub-adult gulf sturgeon fast while in freshwater environments and are almost entirely dependent on the estuarine/marine environment for food (Wooley and Crateau 1985; Gu et al. 2001). Some juveniles (ages one to six) will also fast in the freshwater summer holding areas, but the majority feed year round in the estuaries, river mouths, and bays (Sulak et al. 2009).

Population Dynamics

Currently, seven rivers are known to support reproducing populations of gulf sturgeon. The most recent abundance estimates reported in the 5-Year Review (NMFS and USFWS 2009) (Table 8).

Table 8. Gulf Sturgeon Abundance Estimates by River and Year with Confidence Intervals for the Seven Major Rivers with Reproducing Populations (Modified from USFWS 2009)

River	Year of Data Collection	Abundance Estimate[a]	Lower/Upper 95 percent Confidence Interval[b]	Source
Pearl	2001	430	323/605	(Rogillio et al. 2001)
Pascagoula	2000	216	124/429	(Ross et al. 2001)
Escambia	2006	451	338/656	(USFWS 2007)
Yellow	2003 fall	911	550/1550	(Berg et al. 2007)
Choctawhatchee	2008	3314	not reported	(USFWS 2009)
Apalachicola	2004	350	221/648	(USFWS 2004)
Suwannee	2007	14,000	not reported	(USFWS 2009)

[a] Estimates refer to numbers of individuals greater than a certain size, which varies between studies depending on sampling gear, and in some cases, numbers of individuals that use a particular portion of the river (refer to original publication for details).

[b] Large confidence intervals around the mean estimates reflect the low capture probability in mark-recapture studies.

Gulf sturgeon abundance trends are typically assessed on a riverine basis. In general, gulf sturgeon populations in the eastern portion of the range appear to be stable or slightly increasing, while populations in the western portion are associated with lower abundances and higher uncertainty (NMFS and USFWS 2009). Pine and Martell (2009) reported that, due to low recapture rates and sparse data, the population viability of gulf sturgeon is currently uncertain.

Genetic Diversity

When grouped by genetic relatedness, five regional or river-specific stocks emerge: (1) Lake Pontchartrain and Pearl River; (2) Pascagoula River; (3) Escambia, Blackwater and Yellow Rivers; (4) Choctawhatchee River; and (5) Apalachicola, Ochlocknee and Suwanee Rivers (Stabile et al. 1996; Rudd et al. 2014). Gene flow is low in gulf sturgeon stocks, with each stock exchanging less than one mature female per generation (Waldman and Wirgin 1998).

Distribution

Gulf sturgeon inhabit coastal rivers from Louisiana to Florida during the warmer months, and the Gulf of Mexico and its estuaries and bays in the cooler months. Gulf sturgeon are anadromous: adults spawn in freshwater and migrate into marine waters in the fall to forage and overwinter.

Juvenile gulf sturgeon stay in the river for about the first two to three years. Gulf sturgeon return to their natal stream to spawn. Gulf sturgeon initiate movement up to the rivers between February and April and migrate back out to the Gulf of Mexico between September and November.

Status

The decline in the abundance of gulf sturgeon has been attributed to targeted fisheries in the late 19th and early 20th centuries, habitat loss associated with dams and sills, habitat degradation associated with dredging, de-snagging, and contamination by pesticides, heavy metals, and other industrial contaminants, and certain life history characteristics (e.g. slow growth and late maturation). Effects of climate change (warmer water, sea level rise and higher salinity levels) could lead to accelerated changes in habitats utilized by gulf sturgeon. The rate that climate change and corollary impacts are occurring may outpace the ability of the gulf sturgeon to adapt given its limited geographic distribution and low dispersal rate. In general, gulf sturgeon populations in the eastern portion of the range appear to be stable or slightly increasing, while populations in the western portion are associated with lower abundances and higher uncertainty (NMFS and USFWS 2009).

Designated Critical Habitat

NMFS and the USFWS jointly designated gulf sturgeon critical habitat on April 18, 2003 (50 CFR §226.214). The agencies designated seven riverine areas (Units one to seven) encompassing 2,783 river kilometers and seven estuarine/marine areas (Units eight to 14) encompassing 6,042 square kilometers as critical habitat based on the PBFs that support the species (Figure 25). PBFs considered essential for the conservation of gulf sturgeon are abundant food items, riverine spawning sites with substrates suitable for egg deposition and development, riverine aggregation areas, a flow regime necessary for normal behavior, growth, and survival, water and sediment quality necessary for normal behavior, growth, and viability of all life stages, and safe and unobstructed migratory pathways.

Recovery Goals

The 1995 Recovery Plan outlined three recovery objectives: (1) to prevent further reduction of existing wild populations of gulf sturgeon within the range of the subspecies; (2) to establish population levels that would allow delisting of the gulf sturgeon by management units (management units could be delisted by 2023 if required criteria are met); (3) to establish, following delisting, a self-sustaining population that could withstand directed fishing pressure within management units (USFWS 1995a). The most recent gulf sturgeon five-year review recommended that criteria be developed in a revised recovery plan (NMFS and USFWS 2009).

6.2.12 Green Sturgeon and Designated Critical Habitat

The North American green sturgeon, *Acipenser medirostris*, is an anadromous fish that occurs in the nearshore Eastern Pacific Ocean from Alaska to Mexico (Figure 26; Moyle 2002).

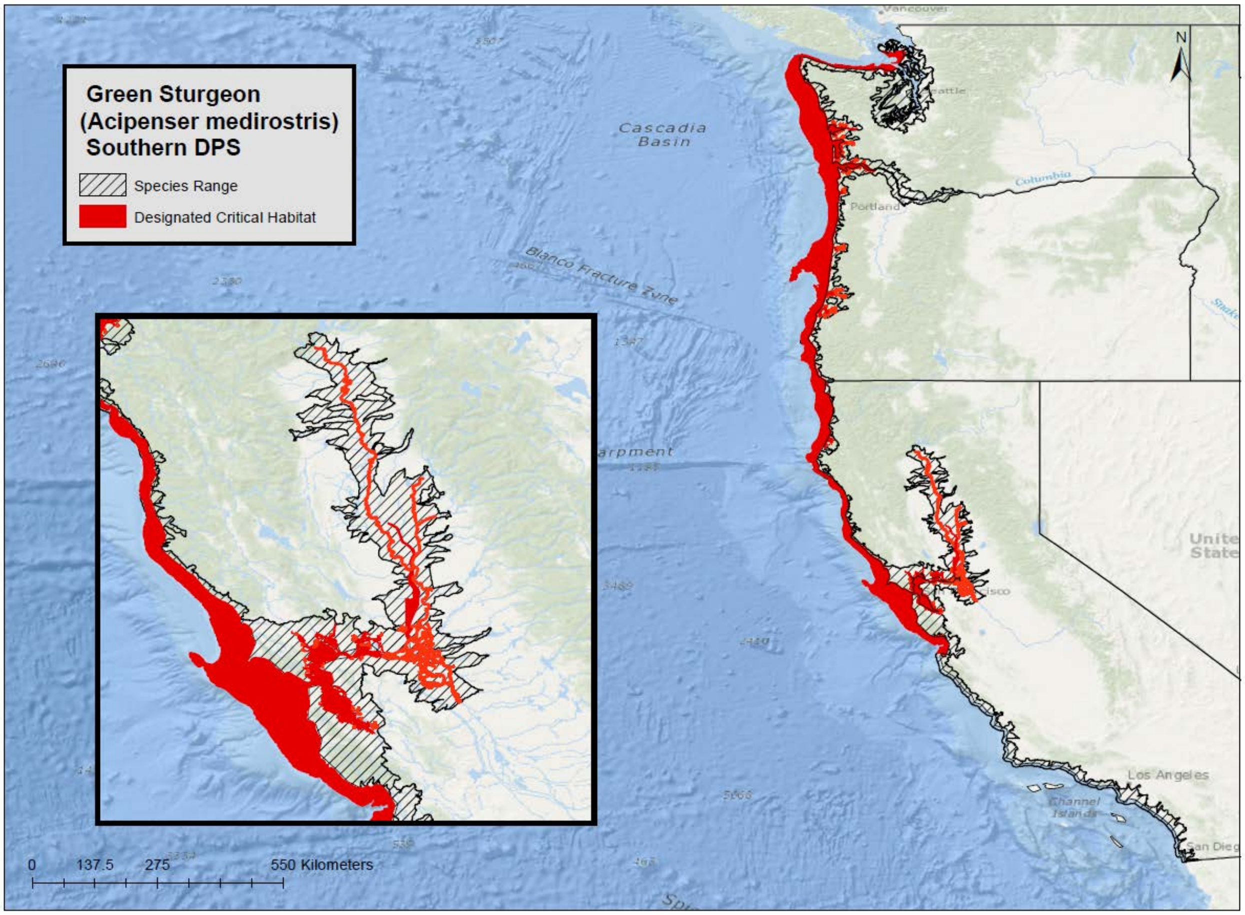

Figure 26. Geographic Range (Within the Contiguous U.S.) and Designated Critical Habitat for Green Sturgeon, Southern DPS

Green sturgeon are long-lived, late-maturing, iteroparous, anadromous species that spawn infrequently in natal streams, and spend substantial portions of their lives in marine waters. Although they are members of the class of bony fishes, the skeleton of sturgeons is composed mostly of cartilage. Sturgeon lack scales; however, they have five rows of characteristic bony plates on their body (called scutes). Green sturgeon have an olive green to dark green back, a yellowish green-white belly (Adams et al. 2002). Green sturgeon have been observed in large concentrations in the summer and autumn within coastal bays and estuaries along the west coast of the U.S., including the Columbia River estuary, Willapa Bay, Grays Harbor, San Francisco Bay and Monterey Bay (Huff et al. 2012; Moser and Lindley 2007; Lindley et al. 2008; Lindley et al. 2011).

NMFS has identified two DPSs of green sturgeon; northern and southern (Israel et al. 2009). In 2006, NMFS determined that the Southern DPS green sturgeon warranted listing as a threatened species under the ESA.

Life History

Green sturgeon reach sexual maturity at approximately fifteen years of age (Van Eenennaam et al. 2006), and may spawn every three to five years throughout their long lives (Tracy 1990). Southern DPS green sturgeon spawn in cool (14 to 17 degrees Celsius), deep, turbulent areas with clean, hard substrates. Six discrete spawning sites have been identified in the upper Sacramento River between Gianella Bridge (river kilometer 320.6) and the Keswick dam (river kilometer 486; Poytress et al. 2013). Spawning has also been confirmed in the Feather River near the Thermalito Afterbay Outlet (river kilometer 95; Seesholtz et al. 2015). Little is known about green sturgeon feeding other than general information. Adults captured in the Sacramento-San Joaquin delta are benthic feeders on invertebrates including shrimp, mollusks, amphipods, and even small fish (Houston 1988; Moyle et al. 1992). Juveniles in the Sacramento River delta feed on opossum shrimp, *Neomysis mercedis*, and *Corophium* amphipods (Radtke 1966).

Green sturgeon are believed to spend the majority of their lives in nearshore oceanic waters, bays, and estuaries. Younger green sturgeon reside in freshwater, with adults returning to freshwater to spawn when they are about 15 years of age and more than four ft (1.3 m) in size. In preparation for spawning, adult Southern DPS green sturgeon enter San Francisco Bay between mid-February and early-May, and migrate rapidly (on the order of a few weeks) up the Sacramento River (Heublein et al. 2009). Spawning occurs from April through early July, with peaks of activity that depend on a variety of factors including water temperature and water flow rates (Poytress et al. 2009;2010). Post-spawn fish typically congregate and hold for several months in a few deep pools in the upper mainstem Sacramento River near spawning sites and migrate back downstream when river flows increase in fall. They re-enter the ocean during the winter months (November through January) and begin their marine migration north along the coast (California Fish Tracking Consortium database).

Green sturgeon larvae are different from all other sturgeon because of the absence of a distinct swim-up or post-hatching stage. Larvae grow fast; young fish grow to 74 mm 45 days after hatching (Deng 2000). Larvae and juveniles migrate downstream toward the Sacramento-San Joaquin Delta/Estuary, where they rear for one to four years before migrating out to the Pacific Ocean as subadults (Nakamoto et al. 1995). Once at sea, subadults and adults occupy coastal waters to a depth of 110 m from Baja California, Mexico to the Bering Sea, Alaska (Erickson and Hightower 2007). Seasonal migrations are known to occur. Fish congregate in coastal bays and estuaries of Washington, Oregon, and California during summer and fall. In winter and spring, similar aggregations can be found from Vancouver Island to Hecate Strait, British Columbia, Canada (Lindley et al. 2008).

Population Dynamics

Population dynamics of Southern DPS green sturgeon focus on abundance; intrinsic growth rates (λ); genetic diversity, drift, and natural selection. Preliminary results from 2010-14 surveys indicated the presence of the following number of adult Southern DPS green sturgeon in the Sacramento River (95 percent confidence interval): 2010: 164 ± 47; 2011: 220 ± 42; 2012: 329 ± 57; 2013: 338 ± 61; 2014: 526 ± 64. Based on these numbers and estimates of mean spawning periodicity, the total number of adults in the Southern DPS population is estimated to be 1,348 ± 524 (NMFS 2015e; Mora 2015).

Attempts to evaluate the status of Southern DPS green sturgeon have been met with limited success due to the lack of reliable long-term data. No estimate of λ is available for Southern DPS green sturgeon.

Genetic Diversity

The available genetic data do not change the status of the species or the imminence or magnitude of any threat; data only confirm the DPS structure and add detail to the DPS composition in different estuaries during the sampling periods (NMFS 2015e). Green sturgeon stocks from the DPSs have been found to be genetically differentiated (Israel et al. 2009; Israel et al. 2004).

Distribution

This species is found along the west coast of Mexico, the U.S., and Canada. Green sturgeon are the most broadly distributed, wide-ranging, and most marine-oriented species of the sturgeon family. The green sturgeon ranges from Mexico to at least Alaska in marine waters, and is observed in bays and estuaries up and down the west coast of North America. Tagged Southern DPS green sturgeon subadults and adults have been detected in coastal marine waters from Monterey Bay, California to Graves Harbor, Alaska, including the Strait of Juan de Fuca and Puget Sound (AquaMaps 2016; Lindley et al. 2011), Washington estuaries within the action area for the Navy ship tow program. Lindley et al. (2011) reported that green sturgeon use the Puget Sound estuary at a low rate, but fish were detected within this estuary in both winter and summer months. More recent preliminary tag results (i.e., data that are in the process of being validated) suggest that green sturgeon may be using the Puget Sound estuary at a higher rate than previously thought. Up to 70 transmitters having tag codes identified as green sturgeon were detected at Admiralty Inlet, the entrance to Puget Sound, between October 2012 and February 2018 (M. Moser, NMFS, pers. comm. to R. Salz, NMFS, February 7, 2019). Although not confirmed, based on the proportions of meta-populations found in other Pacific Northwest estuaries (i.e., Columbia River, Willapa Bay, and Grays Harbor; Israel et al. 2009; Moser et al.

2016; Schreier et al. 2016), we anticipate the green sturgeon detected at Admiralty Inlet likely represent a mix of both Southern DPS and Northern DPS green sturgeon.

Status

Attempts to evaluate the status of Southern DPS green sturgeon have been met with limited success due to the lack of reliable long term data, however based on available scientific data (Adams et al. 2007) and ongoing conservation efforts, NMFS concluded in the final rule designating this species that Southern DPS green sturgeon were likely to become endangered in the foreseeable future throughout all of its range. The final rule listing Southern DPS green sturgeon indicates that the principle factor for the decline in the DPS is the reduction of spawning to a limited area in the Sacramento River caused primarily by impoundments, but they also face threats from water temperature, water flow, and commercial and recreational bycatch. Climate change has the potential to impact Southern DPS green sturgeon in the future, but it is unclear how changing oceanic, nearshore and river conditions will affect the Southern DPS overall (NMFS 2015e).

Green sturgeon have been observed in large concentrations in the summer and autumn within coastal bays and estuaries, including San Francisco Bay, which is part of the designated critical habitat for the ESA-listed Southern DPS. This species may be present from California to Alaska and tagged subadults and adults from the Southern DPS have been found in Puget Sound. Juveniles rear and feed in fresh and estuarine waters for up to four years before dispersing to marine waters as subadults (Nakamoto et al. 1995).

Designated Critical Habitat

Critical habitat was designated for the Southern DPS green sturgeon on October 9, 2009 (74 FR 52300), and includes marine, coastal bay, estuarine, and freshwater areas (Figure 26). In freshwater, designated critical habitat is: the mainstream Sacramento River downstream of Keswick Dam (including the Yolo and Sutter bypasses), the Feather River below Oroville Dam, the Yuba River below Dagueere Point Dam, and the Sacramento-San Joaquin Delta. In coastal bays and estuaries, designated critical habitat is: San Francisco Bay Estuary and Humboldt Bay in California; Coos, Winchester, Yaquina, and Nehalem Bays in Oregon; Willapa and Grays Harbor, and the Lower Columbia River Estuary from the mouth to river km 74. In marine waters, designated critical habitat is: areas to the 60 fathom (110 m) depth isobaths from Monterey Bay to the U.S.-Canada border.

Recovery Goals

The final recovery plan for Southern DPS green sturgeon was released in August 2018 (NMFS 2018b). The recovery plan indicates that the recovery potential for Southern DPS green sturgeon is likely high; however, certain life history characteristics (e.g., long-lived, delayed maturity) coupled with on-going sources of mortality and activities that decrease habitat quality and quantity, particularly in spawning and rearing habitat, indicate recovery could be limited. According to the recovery plan, the objective is to increase Southern DPS green sturgeon abundance, distribution, productivity, and diversity by alleviating significant threats, which include destruction, modification, or curtailment of habitat or range; overutilization for recreational, commercial, scientific, or educational purposes; disease and predation; inadequacy of existing regulatory mechanisms; and other factors such as competition for habitat by native and non-native species, electromagnetic fields, and entrainment/impingement of larvae (NMFS 2018b).

6.2.13 Shortnose Sturgeon

The shortnose sturgeon is the smallest of the three sturgeon species that occur in eastern North America; they grow up to 4.7 ft (1.4 m) and weigh up to 50.7 pounds (23 kilograms). It has a short, conical snout with four barbells in front of its large underslung mouth. Five rows of bony plates occur along its body: one on the back, two on the belly, and one on each side. The body coloration is generally olive-yellow to gray or bluish on the back, and milky-white to dark yellow on the belly. Shortnose sturgeon occur along the Atlantic Coast of North America from the St. John River in Canada to the St. Johns River in Florida (Figure 27).

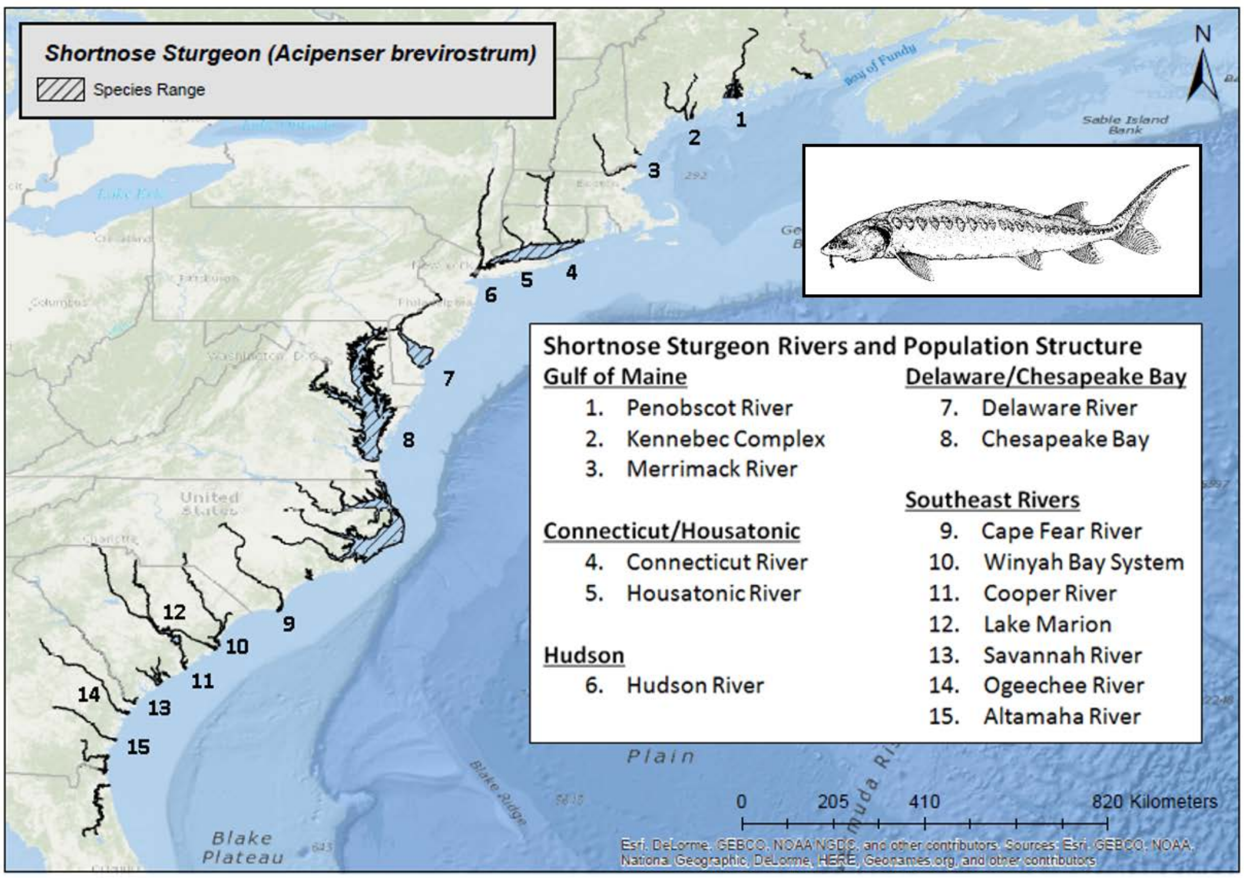

Figure 27. Geographic Range of Shortnose Sturgeon

This section provides general information on the shortnose sturgeon coast-wide population, including information about the species life history, population dynamics, and status.

Life History

The shortnose sturgeon is a relatively slow growing, late maturing, and long-lived fish species. The maximum recorded size of shortnose sturgeon was collected from the Saint John River, Canada, measuring 143 cm total length and weighing 23 kilograms (Dadswell et al. 1984). Shortnose sturgeon typically live longer in the northern portion of their range compared to the southern portion (Dadswell et al. 1984; Gilbert 1989). The maximum ages reported of female shortnose sturgeon by river system include 67 years for the St. John River (New Brunswick), 40 years for the Kennebec River, 37 years for the Hudson River, 34 years for the Connecticut River, 20 years for the Pee Dee River, and ten years for the Altamaha River (Dadswell et al. 1984; Gilbert 1989). Female shortnose sturgeon generally outlive and outgrow males, which seldom exceed 30 years of age (Dadswell et al. 1984; Gilbert 1989). Shortnose sturgeon also exhibit sexually dimorphic growth and maturation patterns across latitudes (Dadswell et al. 1984). In the northern parts of its range, males reach maturity at five to 11 years, while females mature between seven and 18 years. Shortnose sturgeon in southern rivers typically grow faster, mature at younger ages (two to five years for males and four to five for females), but attain smaller maximum sizes than those in the north which grow throughout their longer lifespans (Dadswell et al. 1984).

Shortnose sturgeon are amphidromous, inhabiting large coastal rivers or nearshore estuaries within river systems (Buckley and Kynard 1985; Kieffer and Kynard 1993). They spawn in upper, freshwater areas, and feed and overwinter in both fresh and saline habitats. During the summer and winter months, adults occur primarily in freshwater tidally influenced river reaches (Buckley and Kynard 1985). Older juveniles or subadults tend to move downstream in the fall and winter as water temperatures decline and the salt wedge recedes. In the spring and summer, they move upstream and feed mostly in freshwater reaches; however, these movements usually occur above the saltwater/freshwater river interface (Dadswell et al. 1984; Hall et al. 1991). While shortnose sturgeon do not undertake the long marine migrations documented for Atlantic sturgeon, telemetry data indicate that shortnose sturgeon do make localized coastal migrations (Dionne et al. 2013). Non-spawning movements include rapid, directed post-spawning movements to downstream feeding areas in the spring, and localized, wandering movements in the summer and winter (Dadswell et al. 1984; Buckley and Kynard 1985). Young-of-the-year shortnose sturgeon are believed to move downstream after hatching (Dovel 1983) but remain within freshwater habitats.

Shortnose sturgeon have been found in waters with temperatures as low as two to 3ºC (Dadswell et al. 1984) and as high as 34ºC (Heidt and Gilbert 1979). However, temperatures above 28ºC are thought to adversely affect shortnose sturgeon (Kynard 1997). Shortnose sturgeon are known to occur at a wide range of depths from a minimum depth of 0.6 m up to 30 m (Dadswell et al. 1984; Dadswell 1979). Shortnose sturgeon exhibit tolerance to a wide range of salinities from freshwater (Taubert 1980) to waters with salinity of 30 parts-per-thousand (Holland and Yelverton 1973). Shortnose sturgeon typically occur in the deepest parts of rivers or estuaries where suitable oxygen and salinity levels are present (Gilbert 1989).

Spawning occurs from late winter/early spring (southern rivers) to mid to late spring (northern rivers) depending upon location and water temperature. Shortnose sturgeon spawning migrations are characterized by rapid, directed and often extensive upstream movement (NMFS 1998). Mature males will spawn every other year or annually depending on the river they inhabit (NMFS 1998; Dadswell 1979). Age at first spawning for females is around five years post-maturation, with spawning occurring approximately every three to five years (Dadswell 1979). Shortnose sturgeon are believed to spawn at discrete sites within their natal river (Kieffer and Kynard 1996), typically at the farthest upstream reach of the river, if access is not obstructed by dams (NMFS 1998). Spawning occurs over channel habitats containing gravel, rubble, or rock-cobble substrates (NMFS 1998; Dadswell 1979). Additional environmental conditions associated with spawning activity include decreasing river discharge following the peak spring freshet, water temperatures ranging from 6.5 to 18ºC, and bottom water velocities of 0.4 to 0.8 meters/second (Dadswell 1979; Hall et al. 1991; Kieffer and Kynard 1996; NMFS 1998).

Estimates of annual egg production for shortnose sturgeon are difficult to calculate and are likely to vary greatly in this species because females do not spawn every year. Fecundity estimates that have been made range from 27,000 to 208,000 eggs/female, with a mean of 11,568 eggs/kg body weight (Dadswell et al. 1984). At hatching, shortnose sturgeon are seven to 11 mm long and resemble tadpoles (Buckley and Kynard 1981). In nine to 12 days, the yolk sac is absorbed and the sturgeon develops into larvae which are about 15 mm total length (Buckley and Kynard 1981). Sturgeon larvae are believed to begin downstream migrations at about 20 mm total length.

Shortnose sturgeon are benthic omnivores that feed on crustaceans, insect larvae, worms, mollusks (Moser and Ross 1995; Savoy and Benway 2004), oligochaete worms (Dadswell 1979) and off plant surfaces (Dadswell et al. 1984). Subadults feed indiscriminately, consuming aquatic insects, isopods, and amphipods along with large amounts of mud, stones, and plant material (Dadswell 1979; Bain 1997).

Population Dynamics

Historically, shortnose sturgeon are believed to have inhabited nearly all major rivers and estuaries along the entire east coast of North America. NMFSs Shortnose Sturgeon Recovery Plan identifies 19 populations based on the fish’s strong fidelity to natal rivers and the premise that populations in adjacent river systems did not interbreed with any regularity (NMFS 1998).

The 2010 Shortnose Sturgeon Status Review Team (SSSRT) conducted a three-step risk assessment for shortnose sturgeon at a riverine scale: (1) assess population health, (2) populate a "matrix of stressors" by ranking threats, and (3) review assessment by comparing population health scores to stressor scores. The Hudson River had the highest estimated adult abundance (30,000 to 61,000), followed by the Delaware (12,000), Kennebec Complex (9,000), and Altamaha (6,000; SSSRT 2010). The SSSRT found evidence of an increasing abundance trend for the Kennebec Complex and ACE Basin populations; a stable trend for the Merrimack, Connecticut, Hudson, Delaware, Winyah Bay Complex, Cooper, Savannah, Ogeechee, and Altamaha populations; and a declining trend only for the Cape Fear population (all other populations had an unknown trend) (SSSRT 2010).

The SSSRT summarized continuing threats to the species in each of the 29 identified populations (SSSRT 2010). Dams represent a major threat to seven shortnose sturgeon populations and a moderate threat to seven additional populations. Dredging represents a major threat to one shortnose sturgeon population (Savannah River), a moderately high threat to three populations, and a moderate threat to seven populations. Fisheries bycatch represents a major threat to one shortnose sturgeon population (Lakes Marion and Moultrie in Santee-Cooper Reservoir System), a moderately high threat to four populations, and a moderate threat to ten populations (SSSRT 2010). Water quality represents a major threat to one shortnose sturgeon population (Potomac River), a moderately high threat to six populations, a moderate threat to 13 populations, and a moderately low threat to one population. Specific sources of water quality degradation affecting shortnose sturgeon include coal tar, wastewater treatment plants, fish hatcheries, industrial waste, pulp mills, sewage outflows, industrial farms, water withdrawals, and non-point sources. Impingement/entrainment at power plants and treatment plants was rated as a moderate threat to two shortnose sturgeon populations (Delaware and Potomac).

The SSSRT examined the relationship between population health scores and associated stressors/threats for each shortnose sturgeon riverine population and concluded the following: 1) despite relatively high stressor scores, the Hudson and Kennebec River populations appear relatively healthy; 2) shortnose sturgeon in the Savannah River appear moderately healthy, but their status is perilous; 3) shortnose in the ACE system are of moderate health with low stress and may be most able to recover (SSSRT 2010). Climate warming has the potential to reduce abundance or eliminate shortnose sturgeon in many rivers, particularly in the South (Kynard et al. 2016).

The SSSRT reported results of an age-structured population model using the RAMAS® software (Akçakaya and Root 2007) to estimate shortnose sturgeon extinction probabilities for three river systems: Hudson, Cooper, and Altamaha. The estimated probability of extinction was zero for all three populations under the default assumptions, despite the long (100-year) horizon and the relatively high year-to-year variability in fertility and survival rates. The estimated probability of a 50 percent decline was relatively high (Hudson 0.65, Cooper 0.32, Altamaha 0.73), whereas the probability of an 80 percent decline was low (Hudson 0.09, Cooper 0.01, Altamaha 0.23; SSSRT 2010). The largest shortnose sturgeon adult populations are found in the Northeastern rivers: Hudson 56,708 adults (Bain et al. 2007); Delaware 12,047 (ERC 2006); and Saint Johns greater than 18,000 adults (Dadswell 1979). Shortnose sturgeon populations in southern rivers are considerably smaller by comparison.

Genetic Diversity

Both mitochondrial DNA and nuclear DNA analyses indicate effective (with spawning) coastal migrations are occurring between adjacent rivers in some areas, particularly within the Gulf of Maine and the Southeast (King et al. 2014). The currently available genetic information suggests that shortnose sturgeon can be separated into smaller groupings that form regional clusters across their geographic range (SSSRT 2010). Both regional population and metapopulation structures may exist according to genetic analyses and dispersal and migration patterns (Wirgin et al. 2010; King et al. 2014).

The SSSRT concluded shortnose sturgeon across their geographic range includes five genetically distinct groupings each of which have geographic ecological adaptations: 1) Gulf of Maine; 2) Connecticut and Housatonic Rivers; 3) Hudson River; 4) Delaware River and Chesapeake Bay; and 5) Southeast (SSSRT 2010). Two additional geographically separate populations occur behind dams in the Connecticut River (above the Holyoke Dam) and in Lake Marion on the Santee-Cooper River system in South Carolina (above the Wilson and Pinopolis Dams).

Although these populations are geographically isolated, genetic analyses suggest individual shortnose sturgeon move between some of these populations each generation (Quattro et al. 2002; Wirgin et al. 2005; Wirgin et al. 2010). The SSSRT recommended that each riverine population be considered as a separate management/recovery unit (SSSRT 2010).

Distribution

Shortnose sturgeon occur along the East Coast of North America in rivers, estuaries and the sea. They were once present in most major rivers systems along the Atlantic coast (Kynard 1997). Their current distribution extends north to the Saint John River, New Brunswick, Canada, and south to the St. Johns River, FL (NMFS 1998). The distribution of shortnose sturgeon is disjointed across their range, with northern populations separated from southern populations by a distance of about 400 km near their geographic center in Virginia. Some river systems host populations which rarely leave freshwater while in other areas coastal migrations between river systems are common. Spawning locations have been identified within a number of river systems (SSSRT 2010).

Of the Navy's origination and destination ports for the action, shortnose sturgeon are found in the port of Philadelphia on the Delaware River and in the port of Mayport on the St. Johns River in northern Florida.

Status

The decline in abundance and slow recovery of shortnose sturgeon has been attributed to pollution, overfishing, bycatch in commercial fisheries, and an increase in industrial uses of the nation's large coastal rivers during the 20th century (e.g., hydropower, nuclear power, treated sewage disposal, dredging, construction; SSSRT 2010). In addition, the effects of climate change may adversely impact shortnose sturgeon by reducing the amount of available habitat, exacerbating existing water quality problems, and interfering with migration and spawning cues (SSSRT 2010). Without substantial mitigation and management to improve access to historical habitats and water quality of these systems, shortnose sturgeon populations will likely continue to be depressed. This is particularly evident in some southern rivers that are suspected to no longer support reproducing populations of shortnose sturgeon (SSSRT 2010). The number of river systems in which spawning has been confirmed has been reduced to around 12 locations (SSSRT 2010).

Designated Critical Habitat

No critical habitat has been designated for the shortnose sturgeon.

Recovery Goals

The Shortnose Sturgeon Recovery Plan was developed in 1998. The long-term recovery objective, as stated in the Plan, is to recover all 19 discrete populations to levels of abundance at which they no longer require protection under the ESA (NMFS 1998). To achieve and preserve minimum population sizes for each population segment, essential habitats must be identified and maintained, and mortality must be monitored and minimized. Accordingly, other key recovery tasks discussed in the Plan are to define essential habitat characteristics, assess mortality factors, and protect shortnose sturgeon through applicable federal and state regulations.

6.2.14 Atlantic Salmon Gulf of Maine Distinct Population Segment and Designated Critical Habitat

The Atlantic salmon is an anadromous fish, occupying freshwater streams in North America. There are three Atlantic salmon distinct population segments in the U.S.: Long Island Sound, Central New England, and the Gulf of Maine (Fay et al. 2006). The Gulf of Maine DPS Atlantic salmon are found in watersheds throughout Maine (Figure 28).

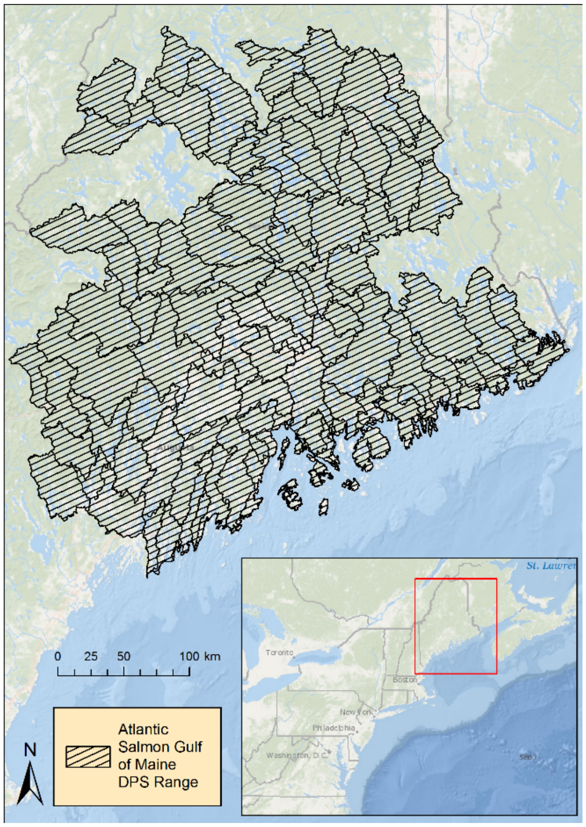

Figure 28. Range Map for the Atlantic Salmon Gulf of Maine DPS

Adult Atlantic salmon are silver-blue with dark spots. The Gulf of Maine DPS was first listed as endangered by the U. S. Fish and Wildlife Service and NMFS on November 17, 2000 (65 FR 69459). The listing was refined by the Services on June 19, 2009 (74 FR 29344) to include all anadromous Atlantic salmon whose freshwater range occurs in the watersheds from the Androscoggin River northward along the Maine coast to the Dennys River, and wherever these fish occur in the estuarine and marine environment.

Life History

Adult Atlantic salmon typically spawn in early November and juveniles spend about two years feeding in freshwater until they weigh approximately two ounces and are six inches in length. Smoltification (the physiological and behavioral changes required for the transition to salt water) usually occurs at age two for Gulf of Maine DPS Atlantic salmon. Gulf of Maine DPS Atlantic salmon migrate more than 4,000 km in the open ocean to reach feeding areas in the Davis Strait between Labrador and Greenland. The majority of Gulf of Maine DPS Atlantic salmon (about ninety percent) spend two winters at sea before reaching maturity and returning to their natal rivers, with the remainder spending one or three winters at sea. At maturity, Gulf of Maine DPS

Atlantic salmon typically weigh between eight to fifteen pounds and average thirty inches in length.

Population Dynamics

The conservation hatchery program plays a significant role in the persistence of Gulf of Maine DPS Atlantic salmon. In 2015, four million juvenile salmon (eggs, fry, parr and smolts) and 4,271 adults were stocked in the Connecticut, Merrimack, Saco, Penobscot and five other coastal rivers in Maine (USASAC 2016). The total number of returns to U.S. rivers was 921, and the majority (eighty percent) of the adult returns were of hatchery origin. The fact that so few of the returning adults are naturally-reared is concerning to managers; the reliance on hatcheries can pose risks such as artificial selection, inbreeding depression and outbreeding depression (Fay et al. 2006).

Adult returns of Gulf of Maine DPS Atlantic salmon captured in six Maine rivers from 1997 to 2004 ranged from 567 to 1,402. These counts include both wild and hatchery origin fish. Each year, the majority (ninety-two to ninety-eight percent) of adult returns were found in the Penobscot River; the Narraguagus River supported between 0.8 to 4.1 percent of adult returns during those years (Fay et al. 2006).

There is no population growth rate available for Gulf of Maine DPS Atlantic salmon. However, the consensus is that the DPS exhibits a continuing declining trend (NOAA 2016).

The Gulf of Maine DPS Atlantic salmon is genetically distinct from other Atlantic salmon populations in Canada, and can be further delineated into stocks by river. The Downeast Coastal stocks include the Dennys, East Machias, Machias, Pleasant and Narraguagus rivers. The Penobscot Bay stock and the Merrymeeting Bay (Sheepscot). The hatchery supplementation programs for the Penobscot and Merrymeeting Bays stocks use river-specific broodstock (USASAC 2016).

Distribution

Gulf of Maine DPS Atlantic salmon can be found in at least eight rivers in Maine: Dennys River, East Machias River, Machias River, Pleasant River, Narraguagus River, Ducktrap River, Sheepscot River, Cove Brook, Penobscot River, Androscoggin River and the Kennebec River.

Status

Historically, Atlantic salmon occupied U.S. rivers throughout New England, with an estimated 300,000 to 500,000 adults returning annually (Fay et al. 2006). Of the three DPSs found in the U.S., native salmon in the Long Island Sound and Central New England DPSs were extirpated in the 1800s. Several rivers within these DPSs are presently stocked with Gulf of Maine DPS salmon. The Gulf of Maine DPS Atlantic salmon was listed as endangered in response to population decline caused by many factors, including overexploitation, degradation of water quality and damming of rivers, all of which remain persistent threats (Fay et al. 2006). Coastal development poses a threat as well, as artificial light can disrupt and delay fry dispersal (Riley et al. 2013). Climate change may cause changes in prey availability and thermal niches, further threatening Atlantic salmon populations (Mills et al. 2013). Even with current conservation efforts, returns of adult Atlantic salmon to the Gulf of Maine DPS rivers remain extremely low, with an estimated extinction risk of nineteen to seventy-five percent in the next one hundred years (Fay et al. 2006). Based on the information above, the species would likely have a low resilience to additional perturbations.

Critical Habitat

On June 19, 2009, NMFS and the USFWS designated critical habitat for Atlantic salmon (74 FR 29300). The critical habitat includes all anadromous Atlantic salmon streams whose freshwater range occurs in watersheds from the Androscoggin River northward along the Maine coast northeastward to the Dennys River, and wherever these fish occur in the estuarine and marine environment (Figure 29). Primary constituent elements were identified within freshwater and estuarine habitats of the occupied range of the Gulf of Maine DPS and include sites for spawning and incubation, juvenile rearing, and migration. The Rule also identified three salmon habitat recovery units to identify geographic and population-level factors to aid in managing the habitat: Merrymeeting Bay, Penobscot, and Downeast. Critical habitat and primary constituent elements were not designated within marine environments because of the limited knowledge of the physical and biological features that the species uses during the marine phase of its life.

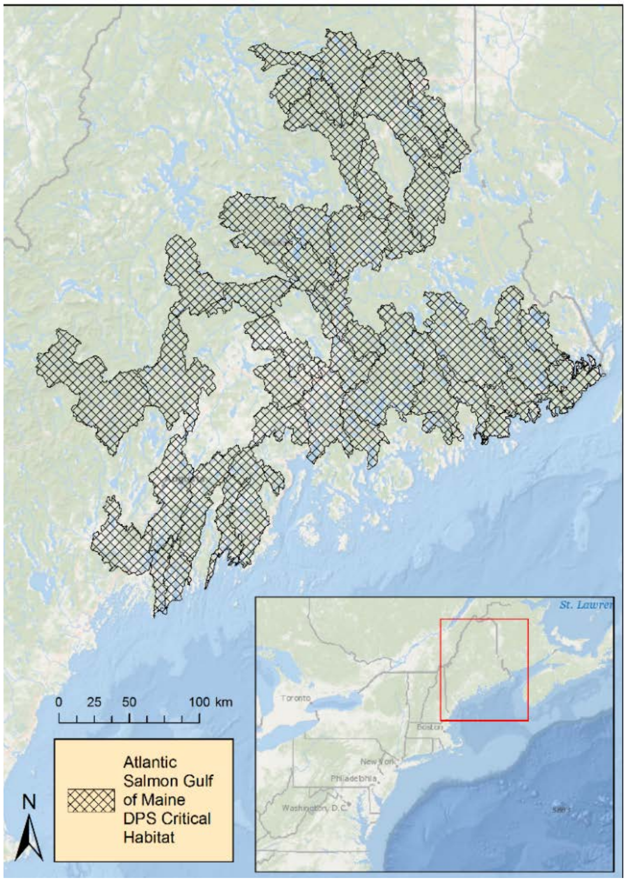

Figure 29. Designated Critical Habitat for the Gulf of Maine DPS of Atlantic Salmon

Recovery Goals

See the 2016 Draft Recovery Plan for the Gulf of Maine DPS Atlantic Salmon, for complete down listing/delisting criteria for each of their respective recovery goals. The following items were the top recovery actions identified to support in the Draft Recovery Plan:

1. Enhance connectivity between the ocean and freshwater habitats important for salmon recovery
2. Maintain the genetic diversity of Atlantic salmon populations over time
3. Increase adult spawners through the conservation hatchery program
4. Increase adult spawners through the freshwater production of smolts
5. Increase Atlantic salmon survival through increased ecosystem understanding and identification of spatial and temporal constraints to salmon marine productivity to inform and support management actions that improve survival
6. Consult with all involved Tribes on a government-to-government basis
7. Collaborate with partners and engage interested parties in recovery efforts for the Gulf of Maine DPS.

6.2.15 Chinook Salmon and Designated Critical Habitat for Central Valley Spring-Run, Puget Sound, and Sacramento River Winter Run Evolutionarily Significant Units

Chinook salmon, also referred to as king salmon, are the largest of the Pacific salmon. Spawning adults are olive to dark maroon in color, without conspicuous streaking or blotches on the sides. Spawning males are darker than females, and have a hooked jaw and slightly humped back. They can be distinguished from other spawning salmon by the color pattern, particularly the spotting on the back and tail, and by the dark, solid black gums of the lower jaw (Moyle 2002). Historically, spring-run chinook salmon occurred in the headwaters of all major river systems in the Central Valley where natural barriers to migration were absent.

On September 16, 1999, NMFS listed the Central Valley ESU of spring-run chinook salmon as a “threatened” species (FR 64 50394). Historically, spring-run chinook salmon occurred in the headwaters of all major river systems in the Central Valley where natural barriers to migration were absent. The only known streams that currently support self-sustaining populations of non-hybridized spring-run chinook salmon in the Central Valley are Mill, Deer and Butte creeks. Each of these populations is small and isolated (NMFS 2014b). On March 24, 1999, NMFS listed the Puget Sound ESU of chinook salmon as “threatened” (64 FR 14308) and this listing was revisited and confirmed in 2005 (70 FR 37160). On January 4, 1994, NMFS listed the Sacramento River winter-run ESU of chinook salmon as Endangered (59 FR 440).

Central Valley Spring-Run ESU

The chinook salmon, Central Valley spring-run ESU includes naturally spawned spring-run chinook salmon originating from the Sacramento River and its tributaries, and spring-run chinook salmon from the Feather River Hatchery Spring-run Chinook Program (Figure 30).

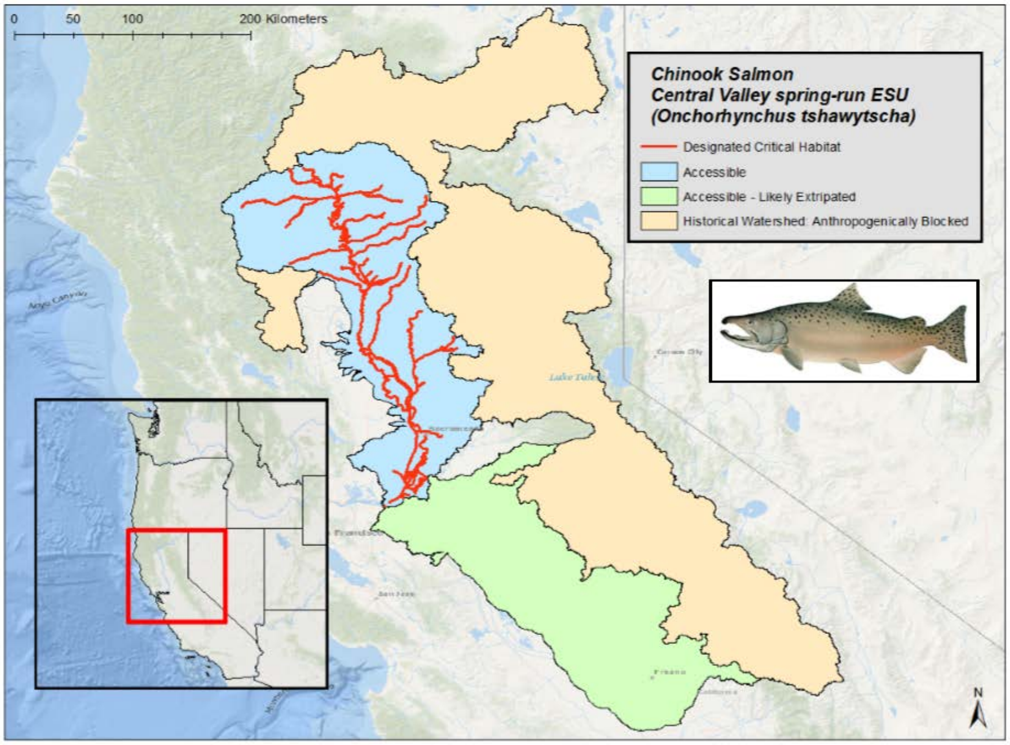

Figure 30. Geographic Range and Designated Critical Habitat for Central Valley Spring-Run, Chinook Salmon

Puget Sound ESU

The Puget Sound ESU includes naturally spawned chinook salmon originating from rivers flowing into Puget Sound from the Elwha River (inclusive) eastward, including rivers in Hood Canal, South Sound, North Sound and the Strait of Georgia. Twenty-six artificial propagation programs are included as part of the ESU (Figure 31).

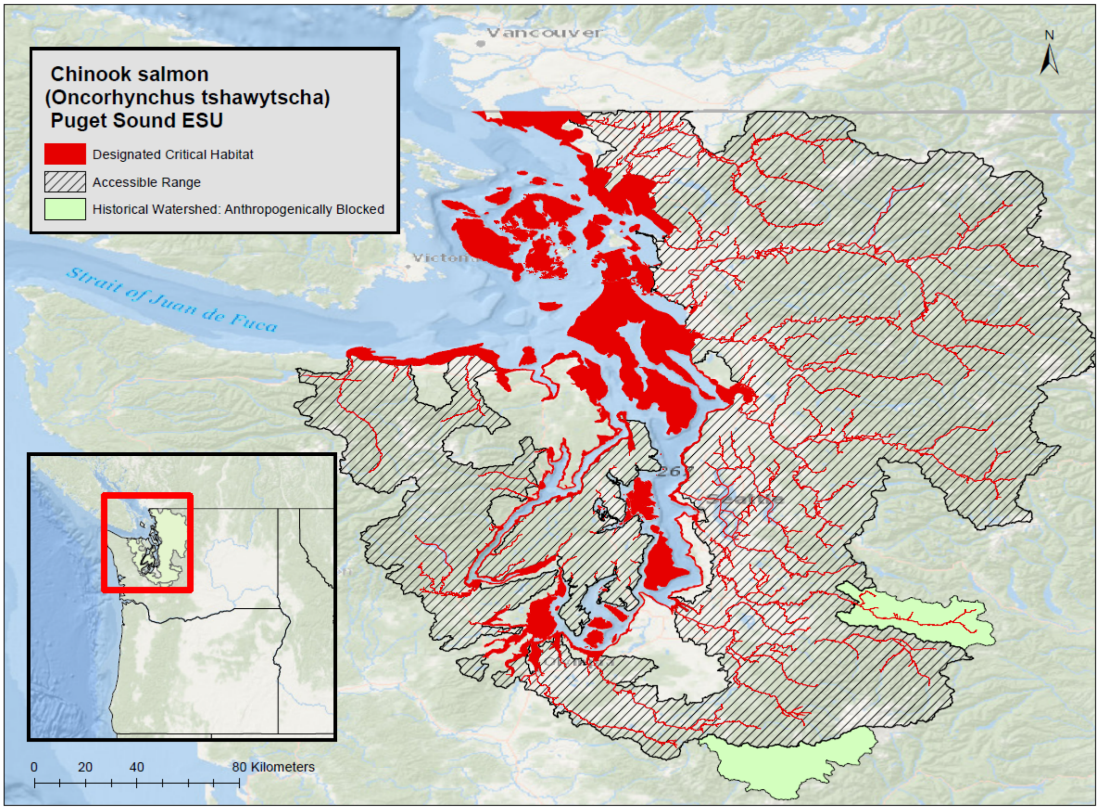

Figure 31. Geographic Range and Designated Critical Habitat of Chinook Salmon, Puget Sound ESU

Sacramento River Winter-Run ESU

The Sacramento River winter-run chinook salmon ESU includes winter-run chinook salmon spawning naturally in the Sacramento River and its tributaries, as well as winter-run chinook salmon that are part of the conservation hatchery program at the Livingston Stone National Fish Hatchery (LSNFH; Figure 32).

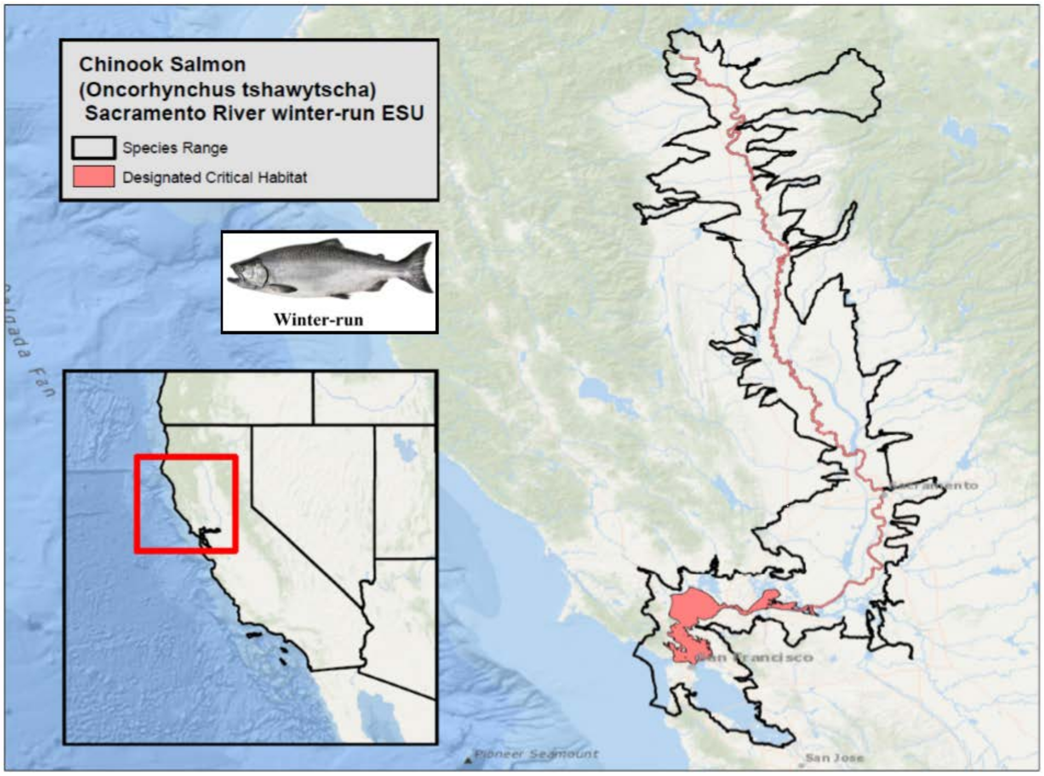

Figure 32. Geographic Range and Designated Critical Habitat of Sacramento River Winter-Run ESU, Chinook Salmon

Life History

Juvenile chinook salmon forage in shallow areas with protective cover, such as tidally influenced sandy beaches and vegetated zones (Healey 1991). Cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1982; MacFarlane and Norton 2002; Sommer et al. 2001). Upon reaching the ocean, juvenile chinook salmon feed voraciously on larval and juvenile fishes, plankton, and terrestrial insects (Healey 1991). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Central Valley Spring-Run ESU

Adult Central Valley spring-run chinook salmon leave the ocean to begin their upstream migration in late January and early February, and enter the Sacramento River between March and September, primarily in May and June (Moyle 2002; Yoshiyama et al. 1998). Spring-run chinook salmon generally enter rivers as sexually immature fish and must hold in freshwater for up to several months before spawning. While maturing, adults hold in deep pools with cold water. Spawning normally occurs between mid- August and early October, peaking in September (Moyle 2002).

The length of time required for embryo incubation and emergence from the gravel is dependent on water temperature. For maximum embryo survival, water temperatures reportedly must be between 41°F and 55.4°F and oxygen saturation levels must be close to maximum. Under those conditions, embryos hatch in 40 to 60 days and remain in the gravel as alevins (the life stage between hatching and egg sack absorption) for another four to six weeks before emerging as fry. Spring-run fry emerge from the gravel from November to March (Moyle 2002). Juveniles may reside in freshwater for 12 to 16 months, but some migrate to the ocean as young-of-the- year in the winter or spring months within eight months of hatching.

Puget Sound ESU

Puget Sound chinook salmon populations exhibit both early-returning (August) and late-returning (mid-September and October) adult chinook salmon spawners (Healey 1991). Juvenile chinook salmon within the Puget Sound generally exhibit an “ocean-type” life history. However, substantial variation occurs with regard to juvenile residence time in freshwater and estuarine environments. Juvenile chinook that remain in freshwater longer have a “river-type” life history. Some rivers may have chinook that exhibit both strategies. In addition, Hayman (Hayman et al. 1996) described three juvenile life histories for chinook salmon with varying freshwater and estuarine residency times in the Skagit River system in northern Puget Sound. In this system, 20 percent to 60 percent of sub-yearling migrants rear for several months in freshwater habitats while the remaining fry migrate to rear in the Skagit River estuary and delta (Beamer et al. 2005). Juveniles in tributaries to Lake Washington exhibit both a stream rearing and a lake rearing strategy. Lake rearing fry are found in highest densities in nearshore shallow (less than one m) habitat adjacent to the opening of tributaries or at the mouth of tributaries where they empty into the lake (Tabor et al. 2006). Puget Sound chinook salmon also has several estuarine rearing juvenile life history types that are highly dependent on estuarine areas for rearing (Beamer et al. 2005). In the estuaries, fry use tidal marshes and connected tidal channels including dikes and ditches developed to protect and drain agricultural land. During their first ocean year, immature chinook salmon use nearshore areas of Puget Sound during all seasons and can be found long distances from their natal river systems (Brennan et al. 2004).

Upon entering Puget Sound, juvenile chinook salmon are nearshore obligate foraging in shallow areas with protective cover, such as tidally influenced sandy beaches and zones vegetated with eelgrass (*Zostera marina*; Healey 1991). Cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1982; MacFarlane and Norton 2002; Sommer et al. 2001). In the marine environment, as juvenile chinook salmon increase in size, they move away from the shore into deeper water and feed voraciously on zooplankton, larval crustaceans and juvenile fishes, plankton, and terrestrial insects (Healey 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Sacramento Winter-Run

Winter-run chinook salmon are unique because they spawn during summer months when air temperatures usually approach their yearly maximum. As a result, winter-run chinook salmon require stream reaches with cold water sources that will protect embryos and juveniles from the warm ambient conditions in summer. Adult winter-run chinook salmon immigration and holding (upstream spawning migration) through the Delta and into the lower Sacramento River occurs from December through July, with a peak during the period extending from January through April (USFWS 1995b). Winter-run chinook salmon are sexually immature when upstream migration begins, and they must hold for several months in suitable habitat prior to spawning. Spawning occurs between late-April and mid-August, with a peak in June and July as reported by California Department of Fish and Wildlife (CDFW) annual escapement surveys (2000-2006).

Winter-run chinook salmon embryo incubation in the Sacramento River can extend into October (Vogel et al. 1988). Winter-run chinook salmon fry rearing in the upper Sacramento River exhibit peak abundance during September, with fry and juvenile emigration past Red Bluff Diversion Dam (RBDD) primarily occurring from July through November (Poytress and Carrillo 2010;2011;2012). Emigration of winter-run chinook salmon juveniles past Knights Landing, located approximately 155.5 river miles downstream of the RBDD, reportedly occurs between November and March, peaking in December, with some emigration continuing through May in some years (Snider and Titus 2000).

Population Dynamics

Central Valley Spring-Run ESU

The Central Valley as a whole is estimated to have supported spring-run chinook salmon runs as large as 600,000 fish between the late 1880s and 1940s. The only known streams that currently support self-sustaining populations of non-hybridized spring-run chinook salmon in the Central Valley are Mill, Deer and Butte creeks. Abundance and trend estimates for these streams, as well as streams supporting dependent populations, indicate population declines in many of the reaches (NMFS 2014b).

Cohort replacement rates (CRR) are indications of whether a cohort is replacing itself in the next generation. The majority of CV spring-run chinook salmon are found to return as three-year-olds; therefore, looking at returns every three years is used as an estimate of the CRR. In the past the CRR has fluctuated between just over 1.0 to just under 0.5, and in the recent years with high returns (2012 and 2013), CRR jumped to 3.84 and 8.68 respectively. CRR for 2014 was 1.85, and the CRR for 2015 with very low returns was a record low of 0.14. Low returns in 2015 were further decreased due to high temperatures and most of the CV spring-run chinook salmon tributaries experienced some pre-spawn mortality. Butte Creek experienced the highest prespawn mortality in 2015, resulting in a carcass survey CRR of only 0.02 (NMFS 2014b).

Threats to the genetic integrity of spring-run chinook salmon was identified as a serious concern to the species when it was listed in 1999 (FR 64 50394; Myers et al. 1998). Three main factors compromised the genetic integrity of spring-run chinook salmon: (1) the lack of reproductive isolation following dam construction throughout the Central Valley resulting in introgression with fall-run chinook salmon in the wild; (2) within basin and inter-basin mixing between spring and fall broodstock for artificial propagation, resulting in introgression in hatcheries; and (3) releasing hatchery-produced juvenile chinook salmon in the San Francisco estuary, which contributes to the straying of returning adults throughout the Central Valley (NMFS 2014b).

Puget Sound ESU

Estimates of the historic abundance range from 1,700 to 51,000 potential Puget Sound chinook salmon spawners per population. During the period from 1996 to 2001, the geometric mean of natural spawners in populations of Puget Sound chinook salmon ranged from 222 to just over 9,489 fish. Thus, the historical estimates of spawner capacity are several orders of magnitude higher than spawner abundances currently observed throughout the ESU (Good et al. 2005).

Available data on total abundance since 1980 indicate that although abundance trends have fluctuated between positive and negative for individual populations, there are widespread negative trends in natural-origin chinook salmon spawner abundance across the ESU (NMFS NWFSC 2015). Productivity remains low in most populations, and hatchery-origin spawners are present in high fractions in most populations outside of the Skagit watershed. Available data now shows that most populations have declined in abundance over the past seven to ten years. Further, escapement levels for all populations remain well below the Technical Recovery Team (TRT) planning ranges for recovery, and most populations are consistently below the spawner-recruit levels identified by the TRT as consistent with recovery (NMFS NWFSC 2015).

Current estimates of diversity show a decline over the past 25 years, indicating a decline of salmon in some areas and increases in others. Salmon returns to the Whidbey Region increased in abundance while returns to other regions declined. In aggregate, the diversity of the ESU as a whole has been declining over the last 25 years.

Sacramento River Winter-Run ESU

Over the last ten years of available data (2003-2013), the abundance of spawning winter-run chinook adults ranged from a low of 738 in 2011 to a high of 17,197 in 2007, with an average of 6,298 (NMFS 2011b).

The population declined from an escapement of near 100,000 in the late 1960s to fewer than 200 in the early 1990s (Good et al. 2005). More recent population estimates of 8,218 (2004), 15,730 (2005), and 17,153 (2006) show a three-year average of 13,700 returning winter-run chinook salmon. However, the run size decreased to 2,542 in 2007 and 2,850 in 2008. Monitoring data indicated that approximately 5.6 percent of winter-run chinook salmon eggs spawned in the

Sacramento River in 2014 survived to the fry life stage (three to nearly ten times lower than in previous years). The drought in 2015 made this another challenging year for winter-run chinook salmon (NMFS 2016g).

The rising proportion of hatchery fish among returning adults threatens to increase the risk of extinction. Lindley et al. (2007) recommend that in order to maintain a low risk of genetic introgression with hatchery fish, no more than five percent of the naturally-spawning population should be composed of hatchery fish. Since 2001, hatchery origin winter-run chinook salmon have made up more than five percent of the run, and in 2005 the contribution of hatchery fish exceeded 18 percent (Lindley et al. 2007).

Distribution

Central Valley Spring-Run ESU

The Central Valley (CV) Technical Recovery Team delineated 18 or 19 historic independent populations of CV spring-run chinook salmon, and a number of smaller dependent populations, that are distributed among four diversity groups (southern Cascades, northern Sierra, southern Sierra, and Coast Range; Lindley et al. 2004). Of these independent populations, only three are extant (Mill, Deer, and Butte creeks) and they represent only the northern Sierra Nevada diversity group. Of the dependent populations, CV spring-run chinook salmon are found in Battle, Clear, Cottonwood, Antelope, Big Chico, and Yuba creeks, as well as the Sacramento and Feather rivers and a number of tributaries of the San Joaquin River including Mokelumne, Stanislaus, and Tuolumne rivers.

Puget Sound ESU

The Puget Sound TRT identified twenty-two extant populations, grouped into five major geographic regions, based on consideration of historical distribution, geographic isolation, dispersal rates, genetic data, life history information, population dynamics, and environmental and ecological diversity.

Sacramento River Winter-Run ESU

The range of winter-run chinook salmon has been greatly reduced by Keswick and Shasta dams on the Sacramento River and by hydroelectric development on Battle Creek. Currently, winter-run chinook salmon spawning is limited to the main-stem Sacramento River between Keswick Dam (River Mile [RM] 302) and the RBDD (RM 243) where the naturally-spawning population is artificially maintained by cool water releases from the dams. Within the Sacramento River, the spatial distribution of spawners is largely governed by water year type and the ability of the CVP to manage water temperatures (NMFS 2014b).

Status

Central Valley Spring-Run ESU

Although spring-run chinook salmon were probably the most abundant salmonid in the Central Valley, this ESU has suffered the most severe declines of any of the four chinook salmon runs in the Sacramento River Basin (Fisher 1994). The ESU is currently limited to independent populations in Mill, Deer, and Butte creeks, persistent and presumably dependent populations in the Feather and Yuba rivers and in Big Chico, Antelope, and Battle creeks, and a few ephemeral or dependent populations in the Northwestern California region (e.g., Beegum, Clear, and Thomes creeks). The Central Valley spring-run chinook salmon ESU is currently faced with three primary threats: (1) loss of most historic spawning habitat; (2) degradation of the remaining habitat; and (3) genetic introgression with the Feather River fish hatchery spring-run chinook salmon strays. The potential effects of climate change are likely to adversely affect spring-run chinook salmon and their recovery (NMFS 2014b).

Puget Sound ESU

All Puget Sound chinook salmon populations are well below escapement abundance levels identified as required for recovery to low extinction risk in the recovery plan. In addition, most populations are consistently below the productivity goals identified in the recovery plan as necessary for recovery. Although trends vary for individual populations across the ESU, most populations have declined in total natural origin recruit abundance since the last status review; while natural origin recruit escapement trends are severely depressed, they have remained mostly stable since 1995. Several of the risk factors identified in the previous status review (Good et al. 2005) are still present, including high fractions of hatchery fish in many populations and widespread loss and degradation of water quality and habitat. Although this ESU's total abundance is a greatly reduced from historic levels, recent abundance levels do not indicate that the ESU is at immediate risk of extinction. This ESU remains relatively well distributed over 22 populations in five geographic areas across the Puget Sound. Although current trends are concerning, the available information indicates that this ESU remains at moderate risk of extinction (NMFS 2011a).

Sacramento River Winter-Run ESU

Like many other populations of Chinook salmon in the Central Valley, the Sacramento River winter-run Chinook salmon ESU has declined in abundance since 2005 and the 10-year trend in abundance is negative (NMFS 2014b). The average run or population size still satisfies the low risk criterion although the latest return estimate for 2010 (1,533 adults) falls into the moderate risk criterion 16 ($N < 2500$) based on the extinction criteria in Lindley et al. 2007. Although ESU abundance has declined over the past ten years, it has not yet triggered the population decline criterion. Since 2000, the proportion of the ESU spawning in the Sacramento River that are of hatchery origin has generally ranged between 5-10 percent of the total population, but in 2005 it did reach approximately 20 percent of the population. This is generally consistent with the USFWS's goal to manage the LSNFH program such that hatchery origin fish are less than 20 percent of the total in-river escapement. In 2010, hatchery fish were estimated to be 12 percent of the total in-river spawners based on carcass surveys (CDFG 2010).

Designated Critical Habitat

Central Valley Spring-Run ESU

NMFS published a final rule designating critical habitat for Central Valley spring-run chinook on September 2, 2005 (70 FR 52488) . Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, nearshore marine areas, and offshore marine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival.

The current condition of PBFs of the CV Spring-run chinook salmon critical habitat indicates that PCEs are not currently functioning or are degraded; their conditions are likely to maintain a low population abundance across the ESU. Spawning and rearing PBFs are degraded by high water temperature caused by the loss of access to historic spawning areas in the upper watersheds, which maintained cool and clean water throughout the summer. The rearing PBF is degraded by floodplain habitat being disconnected from the mainstem of larger rivers throughout the Sacramento River watershed, thereby reducing effective foraging. Migration PBF is degraded by lack of natural cover along the migration corridors. Juvenile migration is obstructed by water diversions along the Sacramento River and by two large state and federal water-export facilities in the Sacramento-San Joaquin Delta.

Puget Sound ESU

Critical habitat was designated for Puget Sound chinook salmon on September 2, 2005 (70 FR 52629). It includes 1,683 km of stream channels, 41 square kilometers of lakes, and 3,512 kilometers of nearshore marine habitat (Figure 31). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, nearshore marine areas, and offshore marine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. The Puget Sound Naval Shipyard at Naval Base Kitsap was excluded from critical habitat designation for Puget Sound chinook salmon.

Forestry practices have heavily impacted migration, spawning, and rearing PBFs in the upper watersheds of most rivers systems within critical habitat designated for the Puget Sound chinook salmon. Degraded PBFs include reduced conditions of substrate supporting spawning, incubation and larval development caused by siltation of gravel; and degraded rearing habitat by removal of cover and reduction in channel complexity. Urbanization and agriculture in the lower alluvial valleys of the Puget Sound Basin and the Strait of Juan de Fuca have reduced channel function and connectivity, reduced available floodplain habitat, and affected water quality.

Hydroelectric development and flood control also obstruct Puget Sound chinook salmon migration in several basins. The most functional PBFs are found in northwest Puget Sound: the Skagit River basin, parts of the Stillaguamish River basin, and the Snohomish River basin where federal land overlap with critical habitat designated for the Puget Sound chinook salmon. However, estuary PBFs are degraded in these areas by reduction in the water quality from contaminants, altered salinity conditions, lack of natural cover, and modification and lack of access to tidal marshes and their channels from the installation of floodgates and other structures.

Sacramento River Winter-Run ESU

NMFS designated critical habitat for the Sacramento winter-run chinook on June 16, 1993 (58 FR 33212; Figure 32). Physical and biological features that are essential for the conservation of Sacramento winter-run chinook salmon, based on the best available information, include (1) access from the Pacific Ocean to appropriate spawning areas in the upper Sacramento River; (2) the availability of clean gravel for spawning substrate; (3) adequate river flows for successful spawning, incubation of eggs, fry development and emergence, and downstream transport of juveniles; (4) water temperatures between 42.5 and 57.5 °F (5.8 and 14.1 degrees Celsius (°C)) for successful spawning, egg incubation, and fry development; (5) habitat and adequate prey free of contaminants; (6) riparian habitat that provides for successful juvenile development and survival; and (7) access of juveniles downstream from the spawning grounds to San Francisco Bay and the Pacific Ocean (58 FR 33212) .

The current condition of PBFs for the Sacramento River Winter-run chinook salmon indicates that they are not currently functioning or are degraded. Their conditions are likely to maintain low population abundances across the ESU. Spawning and rearing PBFs are especially degraded by high water temperature caused by the loss of access to historic spawning areas in the upper watersheds where water maintain lower temperatures. The rearing PBF is further degraded by floodplain habitat disconnected from the mainstems of larger rivers throughout the Sacramento River watershed. The migration PBF is also degraded by the lack of natural cover along the migration corridors. Rearing and migration PBFs are further affected by pollutants entering the surface waters and riverine sediments as contaminated stormwater runoff, aerial drift and deposition, and via point source discharges. Juvenile migration is obstructed by water diversions along the Sacramento River and by two large state and federal water-export facilities in the Sacramento-San Joaquin Delta.

Recovery Goals

Central Valley Spring-Run ESU

Recovery goals, objectives and criteria for the Central Valley spring-run chinook are fully outlined in the 2014 Recovery Plan (NMFS 2014b). The ESU delisting criteria for the spring-run chinook are:

1. One population in the Northwestern California Diversity Group at low risk of extinction
2. Two populations in the Basalt and Porous Lava Diversity Group at low risk of extinction
3. Four populations in the Northern Sierra Diversity Group at low risk of extinction
4. Two populations in the Southern Sierra Diversity Group at low risk of extinction
5. Maintain multiple populations at moderate risk of extinction.

Puget Sound ESU

The recovery plan consists of two documents: the Puget Sound salmon recovery plan (Shared Strategy for Puget Sound 2007) and a supplement by NMFS (2006c). The recovery plan adopts ESU and population level viability criteria recommended by the Puget Sound TRT (Ruckelshaus et al. 2002). The TRT's biological recovery criteria will be met when all of the following conditions are achieved:

1. The viability status of all populations in the ESU is improved from current conditions, and when considered in the aggregate, persistence of the ESU is assured;
2. Two to four chinook salmon populations in each of the five biogeographical regions of the ESU achieve viability, depending on the historical biological characteristics and acceptable risk levels for populations within each region;
3. At least one population from each major genetic and life history group historically present within each of the five biogeographical regions is viable;
4. Tributaries to Puget Sound not identified as primary freshwater habitat for any of the 22 identified populations are functioning in a manner that is sufficient to support an ESU-wide recovery scenario; Production of chinook salmon from tributaries to Puget Sound not identified as primary freshwater habitat for any of the 22 identified populations occurs in a manner consistent with ESU recovery; and
5. Populations that do not meet the viability criteria for all parameters are sustained to provide ecological functions and preserve options for ESU recovery.

Sacramento River Winter-Run ESU

Recovery goals, objectives and criteria for the Sacramento River winter-run chinook are fully outlined in the 2014 Recovery Plan (NMFS 2014b). In order to achieve the downlisting criteria, the species would need to be composed of two populations – one viable and one at moderate extinction risk. Having a second population would improve the species' viability, particularly through increased spatial structure and abundance, but further improvement would be needed to reach the goal of recovery. To delist winter-run chinook salmon, three viable populations are needed. Thus, the downlisting criteria represent an initial key step along the path to recovering winter-run chinook salmon.

6.2.16 Chum Salmon and Designated Critical Habitat for Hood Canal Summer-Run Evolutionarily Significant Unit

The chum salmon, Hood Canal summer-run ESU includes naturally spawned summer-run chum salmon originating from Hood Canal and its tributaries as well as from Olympic Peninsula Rivers between Hood Canal and Dungeness Bay (inclusive; Figure 33). The summer-run chum salmon ESU also includes four artificial propagation programs.

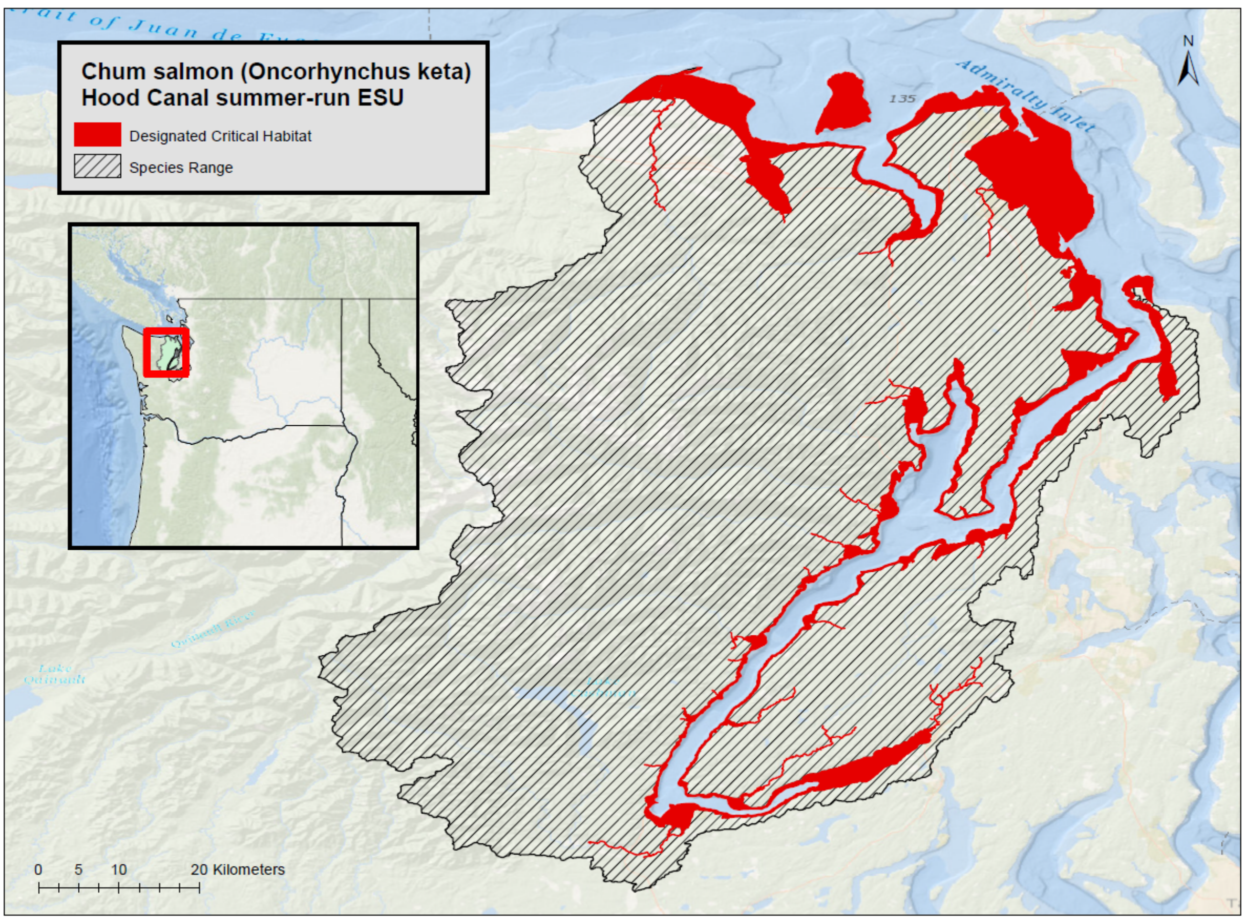

Figure 33. Geographic Range and Designated Critical Habitat of Chum Salmon, Hood Canal Summer-run ESU

Chum salmon are an anadromous (i.e., adults migrate from marine to freshwater streams and rivers to spawn) and semelparous (i.e., they spawn once and then die) fish species. Adult chum salmon are typically between eight and fifteen pounds, but they can get as large as 45 pounds and 3.6 ft long. Males have enormous canine-like fangs and a striking calico pattern body color (front two-thirds of the flank marked by a bold, jagged, reddish line and the posterior third by a jagged black line) during spawning. Females are less flamboyantly colored and lack the extreme dentition of the males. Ocean stage chum salmon are metallic greenish-blue along the back with black speckles. Chum salmon have the widest natural geographic and spawning distribution of the Pacific salmonids.

This section provides general information on the chum salmon population, including information about the species life history, population dynamics, and status.

Life History

Most chum salmon mature and return to their birth stream to spawn between three and five years of age, with 60 to 90 percent of the fish maturing at four years of age. Age at maturity appears to follow a latitudinal trend (i.e., greater in the northern portion of the species' range). Chum salmon typically spawn in the lower reaches of rivers, with redds usually dug in the mainstem or in side channels of rivers from just above tidal influence to 100 km from the sea. Juveniles out-migrate to seawater almost immediately after emerging from the gravel covered redds (Salo 1991). The survival and growth in juvenile chum salmon depend less on freshwater conditions (unlike stream-type salmonids which depend heavily on freshwater habitats) than on favorable estuarine conditions. Chum salmon form schools, presumably to reduce predation (Pitcher 1986), especially if their movements are synchronized to swamp predators (Miller and Brannon 1982).

Chum salmon spend two to five years in feeding areas in the northeast Pacific Ocean, which is a greater proportion of their life history compared to other Pacific salmonids. Chum salmon distribute throughout the North Pacific Ocean and Bering Sea, although North American chum salmon (as opposed to chum salmon originating in Asia), rarely occur west of 175 E longitude (Johnson et al. 1997). North American chum salmon migrate north along the coast in a narrow band that broadens in southeastern Alaska, although some data suggests that chum may travel directly offshore into the North Pacific Ocean (Johnson et al. 1997).

Population Dynamics

Of the sixteen populations that comprise the Hood Canal Summer-run chum ESU, seven are considered “functionally extinct” (Skokomish, Finch Creek, Anderson Creek, Dewatto, Tahuya, Big Beef Creek and Chimicum). NMFS examined average escapements (geometric means) for five-year intervals and estimated trends over the intervals for all natural spawners and for natural-origin only spawners. For both populations, abundance was relatively high in the 1970s, lowest for the period 1985-1999, and high again for the most recent ten years (NMFS NWFSC 2015).

The overall trend in spawning abundance is generally stable for the Hood Canal population (all natural spawners and natural-origin only spawners) and for the Strait of Juan de Fuca population (all natural spawners). Productivity rates, which were quite low during the five-year period from 2005 to 2009 (NMFS NWFSC 2015), increased from 2011-2015 and were greater than replacement rates from 2014-2015 for both major population groups (NMFS NWFSC 2015).

Genetic Diversity

There were likely at least two ecological diversity groups within the Strait of Juan de Fuca population and at least four ecological diversity groups within the Hood Canal population. With the possible exception of the Dungeness River aggregation within the Strait of Juan de Fuca population, Hood Canal ESU summer chum spawning groups exist today that represent each of the ecological diversity groups within the two populations (NMFS 2017a). Diversity values (Shannon diversity index) were generally lower in the 1990s for both independent populations within the ESU, indicating that most of the abundance occurred at a few spawning sites (NMFS NWFSC 2015). Although the overall linear trend in diversity appears to be negative, the last five-year interval shows the highest average value for both populations within the Hood Canal ESU.

Distribution

The Hood Canal summer-run chum salmon ESU includes all naturally spawned populations of summer-run chum salmon in Hood Canal and its tributaries as well as populations in Olympic Peninsula Rivers between Hood Canal and Dungeness Bay, Washington. The nine populations are well distributed throughout the ESU range except for the eastern side of Hood Canal (Johnson et al. 1997). Two independent major population groups have been identified for this ESU:

1. spawning aggregations from rivers and creeks draining into the Strait of Juan de Fuca
2. spawning aggregations within Hood Canal proper (Sands 2009).

Status

The two most recent status reviews indicate some positive signs for the Hood Canal summer-run chum salmon ESU. Diversity has increased from the low levels seen in the 1990s due to both the reintroduction of spawning aggregates and the more uniform relative abundance between populations; considered a good sign for viability in terms of spatial structure and diversity (NMFS NWFSC 2015). Spawning distribution within most streams was also extended further upstream with increased abundance. At present, spatial structure and diversity viability parameters for each population nearly meet the viability criteria (NMFS NWFSC 2015). Spawning abundance has remained relatively high compared to the low levels observed in the early 1990's (NMFS NWFSC 2015). Natural-origin spawner abundance has shown an increasing trend since 1999, and spawning abundance targets in both populations were met in some years

(NMFS NWFSC 2015). Despite substantive gains towards meeting viability criteria in the Hood Canal and Strait of Juan de Fuca summer chum salmon populations, the ESU still does not meet all of the recovery criteria for population viability at this time (NMFS NWFSC 2015). Overall, the Hood Canal Summer-run chum salmon ESU remains at a moderate risk of extinction.

Designated Critical Habitat

NMFS designated critical habitat for Hood Canal Summer-run chum salmon in 2005. Critical habitat includes freshwater spawning, freshwater rearing, freshwater migration, estuarine areas free of obstruction, nearshore marine areas free of obstructions and offshore marine areas with good water quality. The PBFs (formerly *primary constituent elements*) that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity (Figure 33).

The spawning PBF is degraded by excessive fine sediment in the gravel, and the rearing PBF is degraded by loss of access to sloughs in the estuary and nearshore areas and excessive predation. Low river flows in several rivers also adversely affect most PBFs. In the estuarine areas, both migration and rearing PBFs of juveniles are impaired by loss of functional floodplain areas necessary for growth and development of juvenile chum salmon.

Recovery Goals

The recovery strategy for Hood Canal Summer-run chum salmon focuses on habitat protection and restoration throughout the geographic range of the ESU, including both freshwater habitat and nearshore marine areas within a one-mile radius of the watersheds' estuaries (NMFS 2007b). The recovery plan includes an ongoing harvest management program to reduce exploitation rates, a hatchery supplementation program, and the reintroduction of naturally spawning summer chum aggregations to several streams where they were historically present. The Hood Canal plan gives first priority to protecting the functioning habitat and major production areas of the ESU's eight extant stocks, keeping in mind the biological and habitat needs of different life-history stages, and second priority to restoration of degraded areas, where recovery of natural processes appears to be feasible (HCCC 2005). For details on Hood Canal Summer-run chum salmon ESU recovery goals, including complete down-listing/delisting criteria, see the Hood Canal Coordinating Council 2005 recovery plan (HCCC 2005) and the NMFS 2007 supplement to this recovery plan (NMFS 2007b).

6.2.17 Coho Salmon Central California Coast Evolutionarily Significant Unit and Designated Critical Habitat

This ESU includes naturally spawned coho salmon originating from rivers south of Punta Gorda, California to and including Aptos Creek, as well as such coho salmon originating from tributaries to San Francisco Bay. Also, coho salmon from three artificial propagation programs (Figure 34).

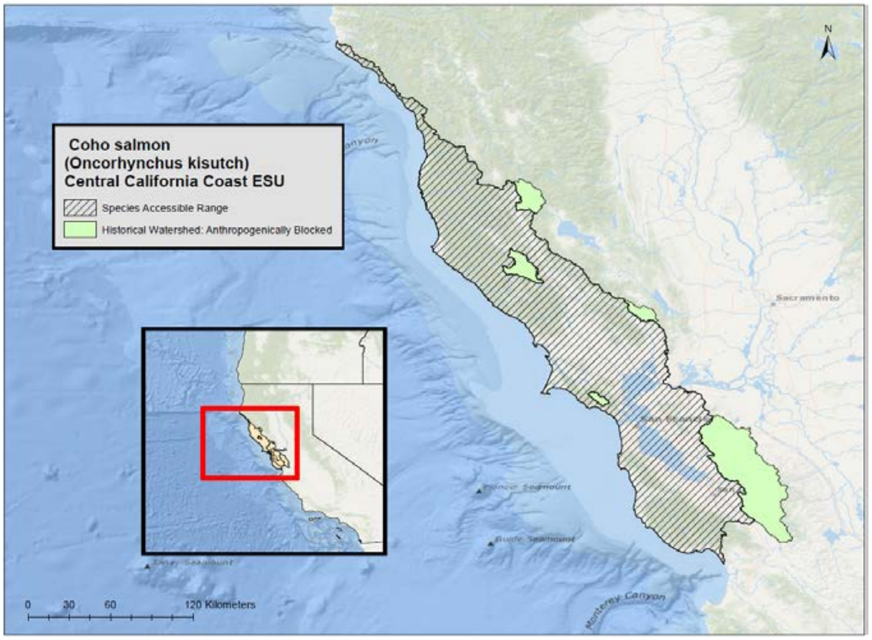

Figure 34. Geographic Range of Central California Coast ESU, Coho Salmon

Coho salmon are an anadromous species. Adult coho salmon are typically about two ft long and eight pounds. Coho have backs that are metallic blue or green, silver sides, and light bellies; spawners are dark with reddish sides; and when coho salmon are in the ocean, they have small black spots on the back and upper portion of the tail. Central California Coast ESU, coho salmon, was listed as threatened under the ESA on October 31, 1996 (64 FR 56138). NMFS re-classified the ESU as endangered on June 28, 2005 (70 FR 37160).

Life history

Central California Coast coho salmon typically enter freshwater from November through January, and spawn into February or early March (Moyle 2002). The upstream migration towards spawning areas coincides with large increases in stream flow (Hassler 1987). Coho salmon often are not able to enter freshwater until heavy rains have caused breaching of sand bars that form at the mouths of many coastal California streams. Spawning occurs in streams with direct flow to the ocean, or in large river tributaries (Moyle 2002). Female coho salmon choose a site to spawn at the head of a riffle, just downstream of a pool where water flow changes from slow to turbulent, and where medium to small size gravel is abundant (Moyle 2002).

Eggs incubate in redds from November through April, and hatch into “alevins” after a period of 35-50 days (Shapovalov and Taft 1954). The period of incubation is inversely related to water temperature. Alevins remain in the gravel for two to ten weeks then emerge into the water column as young juveniles, known as “fry”. Juveniles, or fry, form schools in shallow water along the undercut banks of the stream to avoid predation. The juveniles feed heavily during this time, and as they grow they set up individual territories. Juveniles are voracious feeders, ingesting any organism that moves or drifts over their holding area. The juvenile’s diet is mainly aquatic insect larvae and terrestrial insects, but small fish are taken when available (Moyle 2002).

After one year in freshwater juvenile coho salmon undergo physiological transformation into “smolts” for outmigration to the ocean. Smolts may spend time residing in the estuarine habitat prior to ocean entry, to allow for the transition to the saline environment. After entering the ocean, the immature salmon initially remain in the nearshore waters close to their natal stream. They gradually move northward, generally staying over the continental shelf (Brown et al. 1994). After approximately two years at sea, adult coho salmon move slowly homeward. Adults begin their freshwater migration upstream after heavy fall or winter rains breach the sandbars at the mouths of coastal streams (Sandercock 1991) and/or flows are sufficient to reach upstream spawning areas.

Population Dynamics

Limited information exists on abundance of coho salmon within the Central California Coast coho salmon ESU. About 200,000 to 500,000 coho salmon were produced statewide in the 1940s (Good et al. 2005). This escapement declined to about 99,000 by the 1960s with approximately 56,000 (56 percent) originating from streams within the Central California Coast coho salmon ESU. The estimated number of coho salmon produced within the ESU in 2011 was between 2,000 and 3,000 wild adults (Gallagher et al. 2010).

Within the Lost Coast – Navarro Point stratum and the Navarro Point – Gualala Point stratum, most independent populations show positive but non-significant population trends. Dependent populations within these stratum have declined significantly since 2011. In the Russian River and Lagunitas Creek watersheds, which are the two largest within the Central Coast strata, recent coho salmon population trends suggest limited improvement, although both populations remain well below recovery targets. Recent sampling within Pescadero Creek and San Lorenzo River, the only two independent populations within the Santa Cruz Mountains strata, suggest coho salmon have likely been extirpated within both basins.

Genetic studies show little homogenization of populations, *i.e.,* transfer of stocks between basins have had little effect on the geographic genetic structure of central California coast coho salmon (Sonoma County Water Agency (SCWA) 2002). This ESU likely has considerable diversity in local adaptations given that the ESU spans a large latitudinal diversity in geology and ecoregions, and include both coastal and inland river basins.

Distribution

The TRT identified 11 “functionally independent”, one “potentially independent” and 64 “dependent” populations in the Central California Coast coho salmon ESU (Bjorkstedt et al. 2005 with modifications described in Spence et al. 2008). The 75 populations were grouped into five Diversity Strata. The Russian River is of particular importance for preventing the extinction and contributing to the recovery of Central California Coast coho salmon (NOAA 2013). The Russian River population, once the largest and most dominant source population in the ESU, is now at high risk of extinction because of low abundance and failed productivity (Spence et al. 2008). The Lost Coast and Navarro Point contain the majority of coho salmon remaining in the ESU.

Status

The low survival of juveniles in freshwater, in combination with poor ocean conditions, has led to the precipitous decline of central California coast coho salmon populations. Most independent populations remain at critically low levels, with those in the southern Santa Cruz Mountains strata likely extirpated. Data suggests some populations show a slight positive trend in annual escapement, but the improvement is not statistically significant. Overall, all populations remain, at best, a slight fraction of their recovery target levels, and, aside from the Santa Cruz Mountains strata, the continued extirpation of dependent populations continues to threaten the ESU’s future survival and recovery.

Designated Critical Habitat

Critical habitat for the Central California Coast coho salmon ESU was designated on May 5, 1999 (64 FR 24049; Figure 35). Critical habitat includes juvenile summer and winter rearing areas, juvenile migration corridors, areas for growth and development to adulthood, adult migration corridors, and spawning areas. The physical or biological features that characterize these sites include substrate, water quality, water quantity, water temperature, water velocity, cover/shelter, food, riparian vegetation, space, and safe passage conditions.

NMFS (2008a) evaluated the condition of each habitat attribute in terms of its current condition relative to its role and function in the conservation of the species. The assessment of habitat for this species showed a distinct trend of increasing degradation in quality and quantity of all PBFs as the habitat progresses south through the species range, with the area from the Lost Coast to the Navarro Point supporting most of the more favorable habitats and the Santa Cruz Mountains supporting the least. However, all populations are generally degraded regarding spawning and incubation substrate, and juvenile rearing habitat. Elevated water temperatures occur in many streams across the entire ESU.

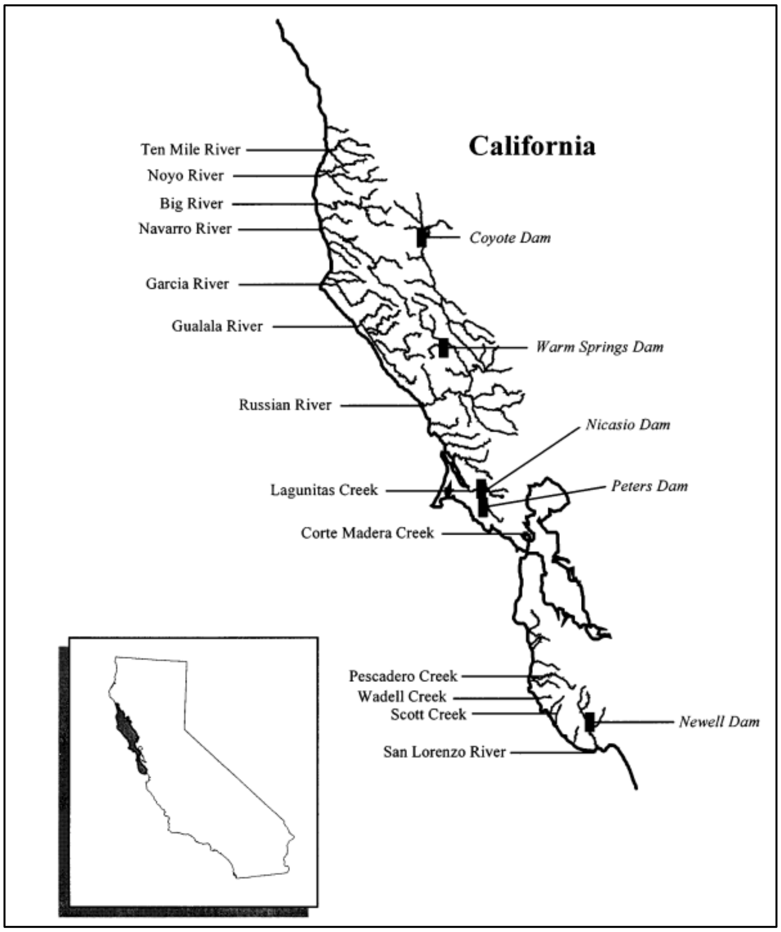

Figure 35. Designated Critical Habitat for the Central California Coast ESU, Coho Salmon

Recovery Goals

See the 2012 Recovery Plan for complete down listing/delisting criteria for each of the following recovery goals (NMFS 2012d):

1. Prevent extinction by protecting existing populations and their habitats
2. Maintain current distribution of coho salmon and restore their distribution to previously occupied areas essential to their recovery
3. Increase abundance of coho salmon to viable population levels, including the expression of all life history forms and strategies
4. Conserve existing genetic diversity and provide opportunities for interchange of genetic material between and within meta populations
5. Maintain and restore suitable freshwater and estuarine habitat conditions and characteristics for all life history stages so viable populations can be sustained naturally

6. Ensure all factors that led to the listing of the species have been ameliorated

7. Develop and maintain a program of monitoring, research, and evaluation that advances understanding of the complex array of factors associated with coho salmon survival and recovery and which allows for adaptively managing our approach to recovery over time.

6.2.18 Steelhead Trout and Designated Critical Habitat for California Central Valley, Central California Coast, and Puget Sound Distinct Population Segments and Designated Critical Habitat

Steelhead (*Onchorynchus mykiss*) are dark-olive in color, shading to silvery-white on the underside with a speckled body and a pink-red stripe along their sides. Those migrating to the ocean develop a slimmer profile, becoming silvery in color, and typically growing larger than rainbow trout that remain in fresh water. Steelhead trout grow to 55 pounds (25 kg) in weight and 45 inches (120 cm) in length, though average size is much smaller.

California Central Valley DPS

This DPS includes naturally spawned anadromous steelhead originating below natural and manmade impassable barriers from the Sacramento and San Joaquin Rivers and their tributaries; excludes such fish originating from San Francisco and San Pablo Bays and their tributaries. This DPS includes steelhead from two artificial propagation programs (Figure 36). On March 19, 1998 NMFS listed the California Central Valley DPS of steelhead as threatened (63 FR 13347) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 834).

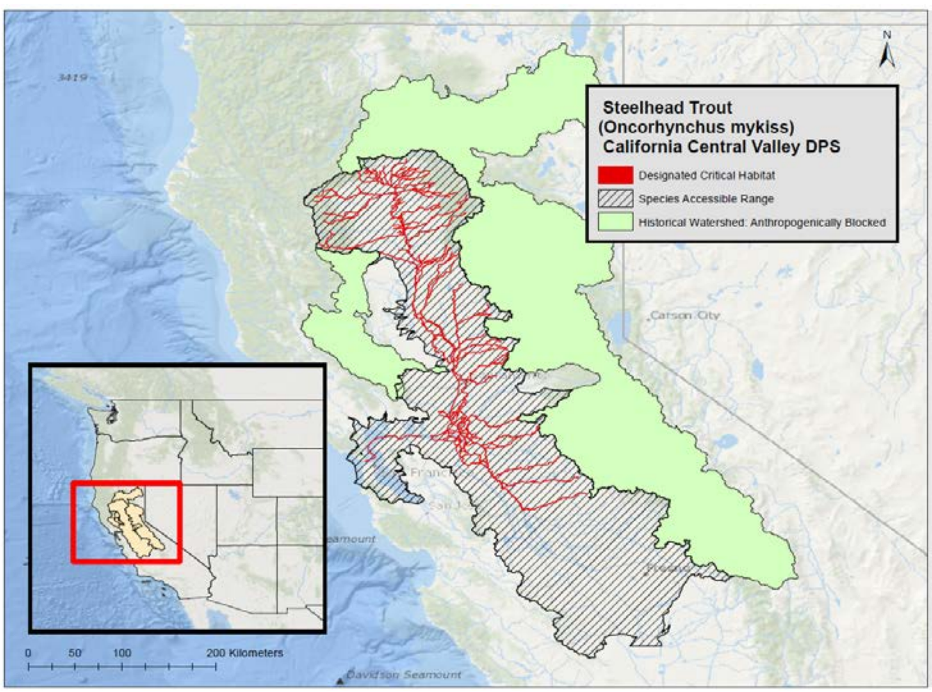

Figure 36. Geographic Range and Designated Critical Habitat of California Central Valley Steelhead Trout

Central California Coast DPS

This DPS includes all naturally spawned populations of steelhead (and their progeny) in streams from the Russian River to Aptos Creek, Santa Cruz County, California (inclusive). It also includes the drainages of San Francisco and San Pablo Bays (Figure 37). On August 18, 1997 NMFS listed the Central California Coast DPS of steelhead as threatened (62 FR 43937) and reaffirmed the DPS's status as threatened on January 5, 2006 (71 FR 834).

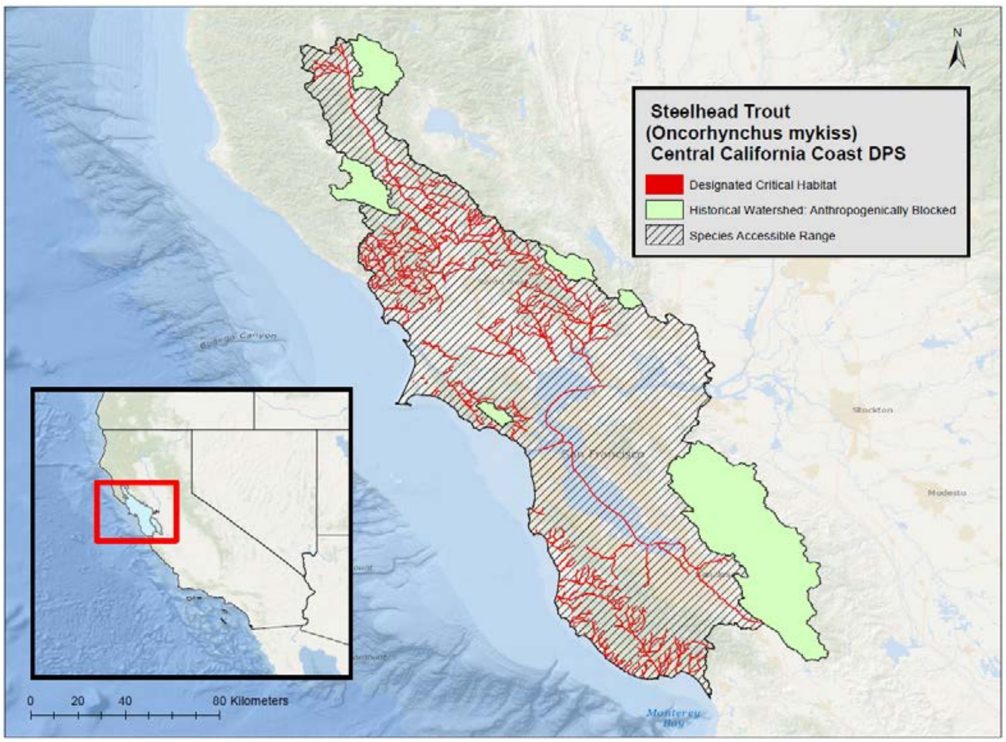

Figure 37. Geographic Range and Designated Critical Habitat for Central California Coast Steelhead Trout

Puget Sound DPS

This DPS includes naturally spawned anadromous steelhead originating below natural and manmade impassable barriers from rivers flowing into Puget Sound from the Elwha River (inclusive) eastward, including rivers in Hood Canal, South Sound, North Sound and the Strait of Georgia (Figure 38). Also, steelhead from six artificial propagation programs are included in this DPS. On May 11, 2007, NMFS listed the Puget Sound DPS as threatened (72 FR 26722).

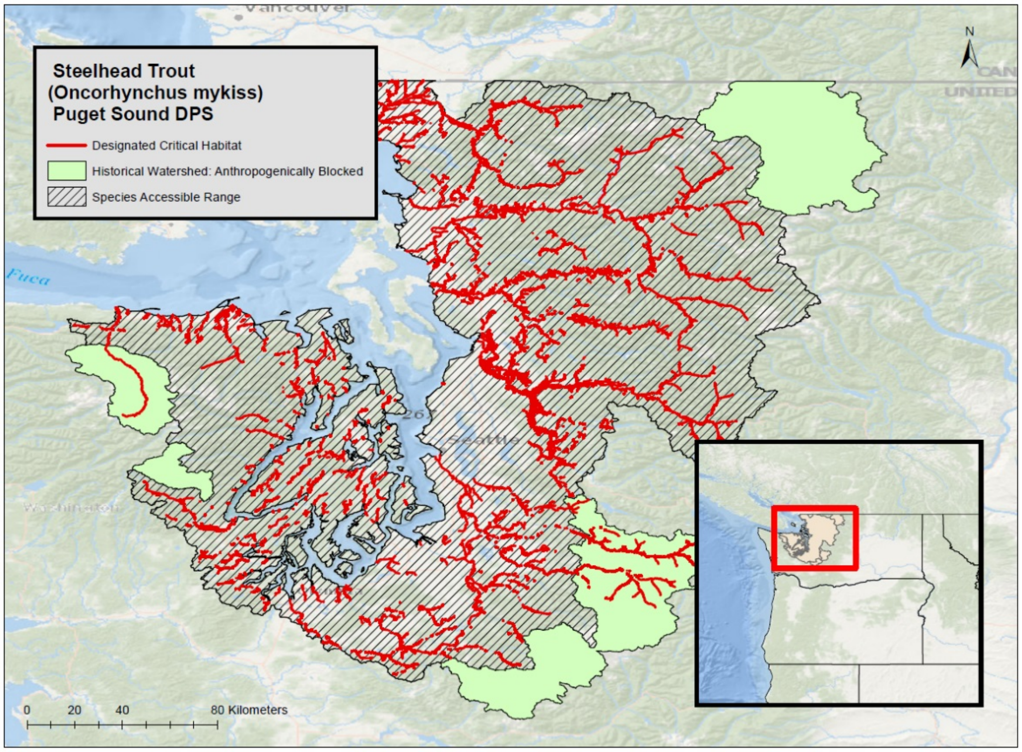

Figure 38. Geographic Range and Designated Critical Habitat for the Puget Sound DPS of Steelhead Trout

Life History

The female steelhead selects a site with good intergravel flow, digs a redd with her tail, usually in the coarse gravel of the tail of a pool or in a riffle, and deposits eggs while an attendant male fertilizes them. The preferred water temperature range for steelhead spawning is reported to be 30°F to 52°F (CDFW 2000). The eggs hatch in three to four weeks at 50°F to 59°F, and fry emerge from the gravel four to six weeks later (Shapovalov and Taft 1954). Regardless of life history strategy, for the first year or two of life steelhead are found in cool, clear, fast flowing permanent streams and rivers where riffles predominate over pools, there is ample cover from riparian vegetation or undercut banks, and invertebrate life is diverse and abundant (Moyle 2002). The smallest fish are most often found in riffles, intermediate size fish in runs, and larger fish in pools.

Steelhead young usually rear in freshwater for one to three years before migrating to the ocean as smolts, but rearing periods of up to seven years have been reported. Migration to the ocean usually occurs in the spring. Steelhead may remain in the ocean for one to five years (two to three years is most common) before returning to their natal streams to spawn (Busby et al. 1996). The distribution of steelhead in the ocean is not well known. Interannual variations in climate, abundance of key prey items (e.g., squid), and density dependent interactions with other salmonid species are key drivers of steelhead distribution and productivity in the marine environment (Atcheson et al. 2012).

Recent information indicates that steelhead originating from central California use a cool, stable, thermal habitat window (ranging between 8-14 °C) in the marine environment characteristic of conditions in northern waters above the 40th parallel to the southern boundary of the Bering Sea (Hayes et al. 2012). Steelhead typically begin returning to the Bay and the Central Valley rivers in late fall, with most immigration occurring from December through February. Spawning takes place from January through April. Adult steelhead typically migrate from the ocean to freshwater between December and April, peaking in January and February (Fukushima and Leah 1998). Ocean maturing steelhead enter fresh water with well-developed gonads and spawn shortly after river entry. Unlike Pacific salmon, steelhead are capable of spawning more than once before they die. However, it is rare for steelhead to spawn more than twice before dying, and most that do so are females (Moyle 2002).

Juvenile steelhead migrate as smolts to the ocean from January through May, with peak migration occurring in April and May (Fukushima and Leah 1998). Barnhart (1986) reports steelhead smolts in California typically range in size from 140 to 210 millimeter (mm, fork length). Steelhead of this size can withstand higher salinities than smaller fish (McCormick and Björnsson 1994), and are more likely to occur for longer periods in tidally influenced estuaries, such as San Francisco Bay. Steelhead smolts in most river systems must pass through estuaries prior to seawater entry.

California Central Valley DPS

Central Valley steelhead spawn downstream of dams on every major tributary within the Sacramento and San Joaquin River systems. Currently, Central Valley steelhead are considered “ocean-maturing” (also known as winter) steelhead, although summer steelhead may have been present prior to construction of large dams (Moyle 2002). Central Valley steelhead enter fresh water from August through April. They hold until flows are high enough in tributaries to enter for spawning (Moyle 2002). Steelhead adults typically spawn from December through April, with peaks from January through March in small streams and tributaries where cool, well oxygenated water is available year-round (Hallock et al. 1961; McEwan 2001).

Central California Coast DPS

This DPS is entirely composed of winter-run fish. Adults return to the Russian River and migrate upstream from December to April, and smolts emigrate between March and May (Shapovalov and Taft 1954; Hayes et al. 2004). Most spawning takes place from January through April. While age at smoltification typically ranges for one to four years, recent studies indicate that growth rates in Soquel Creek likely prevent juveniles from undergoing smoltification until age two (Sogard et al. 2009).

Puget Sound DPS

The Puget Sound steelhead DPS contains both winter-run and summer-run steelhead. Adult winter-run steelhead generally return to Puget Sound tributaries from December to April (NMFS 2005). Spawning occurs from January to mid-June, with peak spawning occurring from mid-April through May. Prior to spawning, maturing adults hold in pools or in side channels to avoid high winter flows. Less information exists for summer-run steelhead as their smaller run size and higher altitude headwater holding areas have not been conducive for monitoring. Based on information from four streams, adult run time occur from mid-April to October with a higher concentration from July through September (NMFS 2005).

Smoltification and seaward migration occur from April to mid-May. The ocean growth period for Puget Sound steelhead ranges from one to three years in the ocean (Busby et al. 1996). Juveniles or adults may spend considerable time in the protected marine environment of the fjord-like Puget Sound during migration to the high seas.

Population Dynamics

California Central Valley DPS

Historic Central Valley steelhead run size may have approached one to two million adults annually (McEwan 2001). By the early 1960s, the steelhead run size had declined to about 40,000 adults (McEwan 2001). Over the past 30 years, the naturally spawned steelhead populations in the upper Sacramento River have declined substantially. Based on catch ratios at Chipps Island in the Delta and using some generous assumptions regarding survival, the average number of CV steelhead females spawning naturally in the entire Central Valley during the years 1980 to 2000 was estimated at about 3,600 (Good et al. 2005).

California Central Valley steelhead lack annual monitoring data for calculating trends. However, the RBDD counts and redd counts up to 1993 and later sporadic data show that the DPS has had a significant long-term downward trend in abundance (NMFS 2009a).

Central California Coast DPS

Historically, the entire Central California Coast steelhead DPS may have consisted of an average runs size of 94,000 adults in the early 1960s (Good et al. 2005). Information on current Central California Coast steelhead populations consists of anecdotal, sporadic surveys that are limited to only smaller portions of watersheds. Presence-absence data indicated that most (82 percent) sampled streams (a subset of all historical steelhead streams) had extant populations of juvenile *O. mykiss* (Adams 2000; Good et al. 2005).

Though the information for individual populations is limited, available information strongly suggests that no population is viable. Long-term population sustainability is extremely low for the southern populations in the Santa Cruz Mountains and in the San Francisco Bay (NMFS 2008a). Declines in juvenile southern populations are consistent with the more general estimates of declining abundance in the region (Good et al. 2005). Data on abundance trends do not exist for the DPS as a whole or for individual watersheds. Thus, it is not possible to calculate long-term trends or lambda.

The interior Russian River winter-run steelhead has the largest runs with an estimate of an average of over 1,000 spawners; it may be able to be sustained over the long-term but hatchery management has eroded the population's genetic diversity (Bjorkstedt et al. 2005; NMFS 2008a).

Puget Sound DPS

Abundance of adult steelhead returning to nearly all Puget Sound Rivers has fallen substantially since estimates began for many populations in the late 1970s and early 1980s. Inspection of geometric means of total spawner abundance from 2010 to 2014 indicates that nine of 20 populations evaluated had geometric mean abundances fewer than 250 adults and 12 of 20 had fewer than 500 adults.

Smoothed trends in abundance indicate modest increases since 2009 for 13 of the 22 populations. Between the two most recent five-year periods (2005 to 2009 and 2010 to 2014), the geometric mean of estimated abundance increased by an average of 5.4 percent. For seven populations in the Northern Cascades subregion, the increase was three percent; for five populations in Central and South Puget Sound subregion, the increase was ten percent; and for six populations in Hood Canal and Strait of Juan de Fuca subregion, the increase was 4.5 percent. However, several of these upward trends are not statistically different from neutral, and most populations remain small. Long-term (15-year) trends in natural spawners are predominantly negative (NMFS NWFSC 2015).

Only two hatchery stocks genetically represent native local populations (Hamma and Green River natural winter-run). The remaining programs, which account for the vast preponderance of production, are either out-of-DPS derived stocks or were within-DPS stocks that have diverged substantially from local populations. The Washington Department of Fish and Wildlife (WDFW) estimated that 31 of the 53 stocks were of native origin and predominantly natural production (Washington Department of Fish and Wildlife (WDFW) 1993).

Distribution

California Central Valley DPS

The Central Valley steelhead distribution ranges over a wide variety of environmental conditions and likely contains biologically significant amounts of spatially structured genetic diversity (Lindley et al. 2006). The loss of populations and reduction in abundances have reduced the large diversity that existed within the DPS. The genetic diversity of the majority of steelhead spawning runs within this DPS is also compromised by hatchery-origin fish.

Central Valley steelhead spawn downstream of dams on every major tributary within the Sacramento and San Joaquin River systems.

Central California Coast DPS

This DPS includes all naturally spawned populations of steelhead (and their progeny) in streams from the Russian River to Aptos Creek, Santa Cruz County, California (inclusive). It also includes the drainages of San Francisco and San Pablo Bays.

Puget Sound DPS

Fifty-three populations of steelhead have been identified in this DPS, of which 37 are winter-run. Summer-run populations are distributed throughout the DPS but are concentrated in northern Puget Sound and Hood Canal; only the Elwha River and Canyon Creek support summer-run steelhead in the rest of the DPS. The Elwha River run, however, is descended from introduced Skamania Hatchery summer-run steelhead. Historical summer-run steelhead in the Green River and Elwha River were likely extirpated in the early 1900s.

Status

California Central Valley DPS

Many watersheds in the Central Valley are experiencing decreased abundance of California Central Valley steelhead. Dam removal and habitat restoration efforts in Clear Creek appear to be benefiting steelhead as recent increases in non-clipped (wild) abundance have been observed. Despite the positive trend in Clear Creek, all other concerns raised in the previous status review remain, including low adult abundances, loss and degradation of a large percentage of the historic spawning and rearing habitat, and domination of smolt production by hatchery fish. Many other planned restoration and reintroduction efforts have yet to be implemented or completed, or are focused on chinook salmon, and have yet to yield demonstrable improvements in habitat, let alone documented increases in naturally produced steelhead. There are indications that natural production of steelhead continues to decline and is now at a very low levels. Their continued low numbers in most hatcheries, domination by hatchery fish, and relatively sparse monitoring makes the continued existence of naturally reproduced steelhead a concern. California Central Valley steelhead is likely to become endangered within the foreseeable future throughout all or a significant portion of its range.

Central California Coast DPS

The Central California Coast steelhead consisted of nine historic functionally independent populations and 23 potentially independent populations (Bjorkstedt et al. 2005). Of the historic functionally independent populations, at least two are extirpated while most of the remaining are nearly extirpated. Current runs in the basins that originally contained the two largest steelhead populations for the DPS, the San Lorenzo and the Russian Rivers, both have been estimated at less than 15 percent of their abundances just 30 years earlier (Good et al. 2005). The Russian River is of particular importance for preventing the extinction and contributing to the recovery of Central California Coast steelhead (NOAA 2013). Steelhead access to significant portions of the upper Russian River has also been blocked (Busby et al. 1996; NMFS 2008a).

Puget Sound DPS

The 2007 Biological Review Team concluded that this DPS was likely to become at risk of extinction in the foreseeable future due to the following major risk factors: widespread declines in abundance and productivity for most natural populations in the DPS (including those in Skagit and Snohomish rivers, previously considered strongholds for steelhead in Puget Sound); low abundance of all summer-run populations; and continued releases of out-of-DPS hatchery fish from Skamania River-derived summer-run and highly domesticated Chambers Creek-derived winter-run stocks. Most of the populations in the DPS are small, and recent declines in abundance of natural fish have persisted despite widespread reductions in harvest of natural steelhead in the DPS since the mid-1990s (NMFS NWFSC 2015; Ford et al. 2011). Low population viability is widespread throughout the DPS based on evidence of diminished abundance, productivity, diversity, and spatial structure. The DPS's current status, particularly with respect to abundance and productivity, is considered to be well below the targets needed to achieve delisting and recovery (NMFS NWFSC 2015). Particular aspects of diversity and spatial structure, including limited use of suitable habitat, are still likely to be limiting viability of most Puget Sound steelhead populations.

Designated Critical Habitat

California Central Valley DPS

NMFS designated critical habitat for California Central Valley steelhead on September 2, 2005 (70 FR 52488; Figure 36). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival.

The spawning PBFs are subject to variations in flows and temperatures, particularly over the summer months. The rearing PBFs are degraded by the channelized, leveed, and riprapped river reaches and sloughs that are common in the Sacramento-San Joaquin system and which typically have low habitat complexity, low abundance of food organisms, and offer little protection from either fish or avian predators. Stream channels commonly have elevated temperatures. Both migration and rearing PBFs are affected by dense urbanization and agriculture along the mainstem and in the Delta which contribute to reduced water quality by introducing several contaminants. In the Sacramento River, the migration corridor for both juveniles and adults is obstructed by the RBDD gates which are down from May 15 through September 15. The migration PBF is also obstructed by complex channel configuration making it more difficult for steelhead to migrate successfully to the western Delta and the ocean. The estuarine PBF, which is present in the Delta, is affected by contaminants from agricultural and urban runoff and release of wastewater treatment plants effluent.

Central California Coast DPS

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630; Figure 37). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival.

Streams throughout the critical habitat have reduced quality of spawning PBFs; sediment fines in spawning gravel have reduced the ability of the substrate attribute to provide well oxygenated and clean water to eggs and alevins. High proportions of fines in bottom substrate also reduce forage by limiting the production of aquatic stream insects adapted to running water. Elevated water temperatures and impaired water quality have further reduced the quality, quantity and function of the rearing PBFs within most streams. These impacts have diminished the ability of designated critical habitat to conserve the Central California Coast steelhead.

Puget Sound DPS

NMFS designated critical habitat for Puget Sound steelhead on February 2, 2016 (81 FR 9251; Figure 38). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, nearshore marine areas, and offshore marine areas. The essential PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Of 70 assessed watersheds, 41 were assigned a high and 18 were assigned a medium conservation value. The remaining watersheds were either of low conservation value, or have been proposed to be excluded for economic considerations. The Puget Sound Naval Shipyard at Naval Base Kitsap was excluded from critical habitat designation for Puget Sound DPS steelhead.

Recovery Goals

California Central Valley DPS

See the 2014 recovery plan for the California Central Valley steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species. The delisting criteria for this DPS are:

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

Central California Coast DPS

See the 2016 recovery plan for the Central California Coast steelhead DPS for complete down-listing/delisting criteria for recovery goals for the species. Recovery plan objectives are to:

- Reduce the present or threatened destruction, modification, or curtailment of habitat or range;
- Ameliorate utilization for commercial, recreational, scientific, or educational purposes;
- Abate disease and predation;
- Establish the adequacy of existing regulatory mechanisms for protecting Central California Coast steelhead now and into the future (i.e., post-delisting);
- Address other natural or manmade factors affecting the continued existence of Central California Coast steelhead;
- Ensure Central California Coast steelhead status is at a low risk of extinction based on abundance, growth rate, spatial structure and diversity.

Puget Sound DPS

A recovery plan has not yet been developed for the Puget Sound DPS of steelhead.

7. ENVIRONMENTAL BASELINE

The "environmental baseline" refers to the condition of the listed species or its designated critical habitat in the action area, without the consequences to the listed species or designated critical habitat caused by the proposed action. The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process. The consequences to listed species or designated critical habitat from ongoing agency activities or existing agency facilities that are not within the agency's discretion to modify are part of the environmental baseline (50 C.F.R. §402.02; 84 FR 44976 published August 27, 2019).

The environmental baseline for this opinion includes natural stressors and stressors from human activities that affect the survival and recovery of ESA-listed Southern resident killer whales; and loggerhead (North Atlantic Ocean DPS) sea turtles; elkhorn, staghorn, rough cactus, pillar, lobed star, mountainous star, and boulder star corals; Johnson's seagrass; bocaccio (Puget Sound/Georgia Basin DPS); yelloweye rockfish (Puget Sound/Georgia Basin DPS); chinook, chum, and coho salmon; steelhead trout; Altantic, gulf, green, and shortnose sturgeon; Atlantic salmon; and designated critical habitat for these species or specific DPSs and ESUs of these species. We describe these stressors below.

7.1 Climate Change

There is a large and growing body of literature on past, present, and future impacts of global climate change, exacerbated and accelerated by human activities. Effects of climate change include sea level rise, increased frequency and magnitude of severe weather events, changes in air and water temperatures, and changes in precipitation patterns, all of which are likely to affect ESA resources. NOAA's climate information portal provides basic background information on these and other measured or anticipated climate change effects (see https://www.climate.gov).

In order to evaluate the implications of different climate outcomes and associated impacts throughout the 21st century, many factors have to be considered with greenhouse gas emissions and the potential variability in emissions serving as a key variable. Developments in technology, changes in energy generation and land use, global and regional economic circumstances, and population growth must also be considered.

A set of four scenarios was developed by the Intergovernmental Panel on Climate Change (IPCC) to ensure that starting conditions, historical data, and projections are employed consistently across the various branches of climate science. The scenarios are referred to as representative concentration pathways (RCPs), which capture a range of potential greenhouse gas emissions pathways and associated atmospheric concentration levels through 2100 (IPCC 2014). The RCP scenarios drive climate model projections for temperature, precipitation, sea level, and other variables: RCP2.6 is a stringent mitigation scenario; RCP2.5 and RCP6.0 are intermediate scenarios; and RCP8.5 is a scenario with no mitigation or reduction in the use of fossil fuels. IPCC future global climate predictions (2014 and 2018) and national and regional climate predictions included in the Fourth National Climate Assessment for U.S. states and territories (USGCRP 2018) use the RCP scenarios.

The increase of global mean surface temperature change by 2100 is projected to be 0.3 to 1.7°C under RCP2.6, 1.1 to 2.6°C under RCP4.5, 1.4 to 3.1°C under RCP6.0, and 2.6 to 4.8°C under RCP8.5 with the Arctic region warming more rapidly than the global mean under all scenarios (IPCC 2014). The Paris Agreement aims to limit the future rise in global average temperature to

2°C, but the observed acceleration in carbon emissions over the last 15 to 20 years, even with a lower trend in 2016, has been consistent with higher future scenarios such as RCP8.5 (Hayhoe et al. 2018).

The globally-averaged combined land and ocean surface temperature data, as calculated by a linear trend, show a warming of approximately 1.0°C from 1901 through 2016 (Hayhoe et al. 2018). The IPCC Special Report on the Impacts of Global Warming (in press) noted that human-induced warming reached temperatures between 0.8 and 1.2°C above pre-industrial levels in 2017, likely increasing between 0.1 and 0.3°C per decade. Warming greater than the global average has already been experienced in many regions and seasons, with most land regions experiencing greater warming than over the ocean (Allen et al. 2018). Annual average temperatures have increased by 1.8°C across the contiguous U.S. since the beginning of the 20th century with Alaska warming faster than any other state and twice as fast as the global average since the mid-20th century (Jay et al. 2018). Global warming has led to more frequent heatwaves in most land regions and an increase in the frequency and duration of marine heatwaves (Hoegh-Guldberg et al. 2018). Average global warming up to 1.5°C as compared to pre-industrial levels is expected to lead to regional changes in extreme temperatures, and increases in the frequency and intensity of precipitation and drought (Hoegh-Guldberg et al. 2018).

Several of the most important threats contributing to the extinction risk of ESA-listed species, particularly those with a calcium carbonate skeleton such as corals and mollusks as well as species for which these animals serve as prey or habitat, are related to global climate change. The main concerns regarding impacts of global climate change on coral reefs and other calcium carbonate habitats generally, and on ESA-listed corals and mollusks in particular are the magnitude and the rapid pace of change in greenhouse gas concentrations (e.g., carbon dioxide and methane) and atmospheric warming since the Industrial Revolution in the mid-19th century. These changes are increasing the warming of the global climate system and altering the carbonate chemistry of the ocean (ocean acidification; IPCC 2014). As carbon dioxide concentrations increase in the atmosphere, more carbon dioxide is absorbed by the oceans, causing lower pH and reduced availability of calcium carbonate. Because of the increase in carbon dioxide and other greenhouse gases in the atmosphere since the Industrial Revolution, ocean acidification has already occurred throughout the world's oceans, including in the Caribbean, and is predicted to increase considerably between now and 2100 (IPCC 2014).

The Atlantic Ocean appears to be warming faster than all other ocean basins except perhaps the southern oceans (Cheng et al. 2017). In the western North Atlantic Ocean surface temperatures have been unusually warm in recent years (Blunden and Arndt 2017). A study by Polyakov et al. (2010) suggests that the North Atlantic Ocean overall has been experiencing a general warming trend over the last 80 years of 0.031±0.0006 degrees Celsius per decade in the upper 2,000 m (6,561.7 ft) of the ocean. Additional consequences of climate change include increased ocean stratification, decreased sea-ice extent, altered patterns of ocean circulation, and decreased ocean oxygen levels (Doney et al. 2012). Since the early 1980s, the annual minimum sea ice extent (observed in September each year) in the Arctic Ocean has decreased at a rate of 11 to 16 percent per decade (Jay et al. 2018). Further, ocean acidity has increased by 26 percent since the beginning of the industrial era (IPCC 2014) and this rise has been linked to climate change. Climate change is also expected to increase the frequency of extreme weather and climate events including, but not limited to, cyclones, tropical storms, heat waves, and droughts (IPCC 2014).

Climate change has the potential to impact species abundance, geographic distribution, migration patterns, and susceptibility to disease and contaminants, as well as the timing of seasonal activities and community composition and structure (Macleod 2009; Robinson et al. 2008; Kintisch and Buckheit 2006; Learmonth et al. 2006; Mcmahon and Hays 2006; Evans and Bjørge 2013; IPCC 2014). Though predicting the precise consequences of climate change on highly mobile marine species is difficult (Simmonds and Eliott 2009), recent research has indicated a range of consequences already occurring. For example, in sea turtles, sex is determined by the ambient sand temperature (during the middle third of incubation) with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25 to 35°C (Ackerman 1997). These impacts will be exacerbated by sea level rise. The loss of habitat because of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Antonelis et al. 2006; Baker et al. 2006).

Changes in the marine ecosystem caused by global climate change (e.g., ocean acidification, salinity, oceanic currents, DO levels, nutrient distribution) could influence the distribution and abundance of lower trophic levels (e.g., phytoplankton, zooplankton, submerged aquatic vegetation, crustaceans, mollusks, forage fish), ultimately affecting primary foraging areas of ESA-listed species including marine mammals, sea turtles, and fish. Marine species ranges are expected to shift as they align their distributions to match their physiological tolerances under changing environmental conditions (Doney et al. 2012). Hazen et al. (2012) examined top predator distribution and diversity in the Pacific Ocean in light of rising sea surface temperatures using a database of electronic tags and output from a global climate model. They predicted up to a 35 percent change in core habitat area for some key marine predators in the Pacific Ocean, with some species predicted to experience gains in available core habitat and some predicted to experience losses. Notably, leatherback turtles were predicted to gain core habitat area, whereas loggerhead turtles and blue whales were predicted to experience losses in available core habitat. Mcmahon and Hays (2006) predicted increased ocean temperatures will expand the distribution of leatherback turtles into more northern latitudes. The authors noted this is already occurring in the Atlantic Ocean. Macleod (2009) estimated, based upon expected shifts in water temperature, 88 percent of cetaceans will be affected by climate change, with 47 percent predicted to experience unfavorable conditions (e.g., range contraction). Willis-Norton et al. (2014)

acknowledged there will be both habitat loss and gain, but overall climate change could result in a 15 percent loss of core pelagic habitat for leatherback turtles in the eastern South Pacific Ocean.

Similarly, climate-related changes in important prey species populations are likely to affect predator populations. For example, blue whales, as predators that specialize in eating krill, are likely to change their distribution in response to changes in the distribution of krill (Payne et al. 1990; Clapham 1999). Pecl and Jackson (2008) predicted climate change will likely result in squid that hatch out smaller and earlier, undergo faster growth over shorter life-spans, and mature younger at a smaller size. This could have negative consequences for species such as sperm whales, whose diets can be dominated by cephalopods. For ESA-listed species that undergo long migrations, if either prey availability or habitat suitability is disrupted by changing ocean temperatures regimes, the timing of migration can change or negatively impact population sustainability (Simmonds and Eliott 2009).

Macleod (2009) estimated that, based upon expected shifts in water temperature, 88 percent of cetaceans would be affected by climate change, 47 percent would be negatively affected, and 21 percent would be put at risk of extinction. Hazen et al. (2012) examined top predator distribution and diversity in the Pacific Ocean in light of rising sea surface temperatures using a database of electronic tags and output from a global climate model. He predicted up to a 35 percent change in core habitat area for some key marine predators in the Pacific Ocean, with some species predicted to experience gains in available core habitat and some predicted to experience losses. Notably, leatherback sea turtles were predicted to gain core habitat area, whereas loggerhead sea turtles and blue whales were predicted to experience losses in available core habitat. Such range shifts could affect marine mammal and sea turtle foraging success as well as sea turtle reproductive periodicity (Birney et al. 2015; Pike 2014).

Blue whales, as predators that specialize in eating krill, are likely to change their distribution in response to changes in the distribution of krill (Payne et al. 1990; Clapham et al. 1999; Payne et al. 1986) associated with ocean acidification. Krill have been shown to suffer decreased larval development and survival under lower pH conditions (McLaskey et al. 2016). Krill also have lower metabolic rates after both short-term and long-term exposure to low pH (Cooper et al. 2016). Increased ocean acidification may also have serious impacts on fish development and behavior (Raven et al. 2005), including sensory functions (Bignami et al. 2013) and fish larvae behavior that could impact fish populations (Munday et al. 2009) and piscivorous ESA-listed species that rely on those populations for food.

The distribution, abundance and migration of baleen whales reflects the distribution, abundance and movements of dense prey patches (e.g., copepods, euphausiids or krill, amphipods, shrimp), which have in turn been linked to oceanographic features affected by climate change (Learmonth et al. 2006). Changes in plankton distribution, abundance and composition are closely related to ocean climate, including temperature. Curran et al. (2003) analyzed ice-core samples from 1841-

1995 and concluded Antarctic sea ice cover had declined by about 20 percent since the 1950s. Atkinson et al. (2004) linked sea ice loss to severe decreases in krill populations over the past several decades in some areas of the Antarctic. Blue whales, as predators that specialize in eating krill, are likely to change their distribution in response to changes in the distribution of krill (Payne et al. 1990; Clapham et al. 1999; Payne et al. 1986).

Within the action area for this opinion, female biased green sea turtle sex ratios have been reported at foraging locations in San Diego Bay, California (Allen et al. 2017). For the Hawaii green sea turtle population, Chaloupka et al. (2008) reported no gender bias in stranding data from 1982-2003. The most recent (2014) published sea turtle stranding report for Hawaii also indicates little to no apparent bias in green sea turtle sex ratio (50 females, 43 males, 155 unknown/indeterminable; NMFS 2015b). However, preliminary (unpublished) data from Allen et al. (2017) suggests there may be a female biased sex ratio in this population. Genetic analyses and behavioral data suggest that populations with temperature-dependent sex determination may be unable to evolve rapidly enough to counteract the negative fitness consequences of rapid global temperature change (Hays 2008 as cited in Newson et al. 2009). Altered sex ratios have been observed in sea turtle populations worldwide (Mazaris et al. 2008; Reina et al. 2008; Robinson et al. 2008; Fuentes et al. 2009). This does not yet appear to have affected population viabilities through reduced reproductive success, although average nesting and emergence dates have changed over the past several decades by days to weeks in some locations (Poloczanska et al. 2009). A fundamental shift in population demographics may lead to increased instability of populations that are already at risk from several other threats. In addition to altering sex ratios, increased temperatures in sea turtle nests can result in reduced incubation times (producing smaller hatchling), reduced clutch size, and reduced nesting success due to exceeded thermal tolerances (Fuentes et al. 2010; Fuentes et al. 2011; Fuentes et al. 2009; Azanza-Ricardo et al. 2017).

Global climate change may affect the ESA-listed fish species and DPSs considered in this opinion. Thermal changes of just a few degrees Celsius can substantially alter fish protein metabolism (McCarthy and Houlihan 1997), response to aquatic contaminants (Reid 1997), reproductive performance (Van Der Kraak and Pankhurst 1997), smolt development (McCormick et al. 1997), species distribution limits (McCarthy and Houlihan 1997), and community structure of fish populations (Schindler 2001). Apart from direct changes to anadromous fish survival, increased water temperatures may alter habitat (e.g., Mantua et al. 2010; Crozier et al. 2014).

Shortnose and Atlantic sturgeon are tolerant to water temperatures up to approximately 28° C; these temperatures are experienced naturally in some areas of rivers during the summer months. If temperature rises beyond thermal limits for extended periods, habitat could be lost; this could be the case if southern habitats warm, resulting in range loss (Lassalle et al. 2010). As water temperatures increase, juvenile sturgeon may experience elevated mortality due to lack of cooler water refuges. The Atlantic salmon Gulf of Maine DPS may be particularly vulnerable to elevated water temperature regimes since Maine is near the southern extent species' range in North America (Fay et al. 2006). Rising temperatures could also exacerbate existing water quality problems associated with DO and temperature.

Studies examining the effects of long-term climate change to salmon and steelhead populations have identified a number of common mechanisms by which climate variation is likely to influence sustainability of steelhead populations (NMFS 2016d). Climate effects on salmonids tend to be negative across multiple life-stages (Wade et al. 2013; Wainwright and Weitkamp 2013). Considering the action area for this opinion, we focus here on the effects of climate change on steelhead in the marine environment. Northward range shifts are a climate response expected in many marine fish species, including salmon (Cheung et al. 2015). Steelhead marine migration patterns could be affected by climate-induced contraction of thermally suitable habitat. Abdul-Aziz et al. (2011) modeled changes in summer thermal ranges in the open ocean for Pacific salmon and steelhead under multiple Intergovernmental Panel on Climate Change warming scenarios. For steelhead, they predicted contractions in suitable marine habitat of 30-50 percent by the 2080s under the medium and high greenhouse gas emissions scenarios.

Numerous researchers have reported that salmon and steelhead marine survival is highly variable over time and often correlated with large-scale climate indices (Litzow et al. 2014; Stachura et al. 2013; Sydeman et al. 2014; Petrosky and Schaller 2010). Many fish communities, including key salmon and steelhead prey and predators, experience changes in abundance and distribution during warm ocean periods (Cheung et al. 2009; Pearcy 2002). However, food chain dynamics in the open ocean are flexible and difficult to predict into the future, and in the case of steelhead poorly understood (Grimes 2007). To what extent a future warmer ocean will mimic historic conditions of warm-ocean, low-survival periods is not known. Current indications are that a warmer Pacific Ocean is generally less productive at mid latitudes, and hence likely to be less favorable for salmon and steelhead (NMFS 2016d). The full implications of ocean acidification on salmon are not known at this time (National Research Council 2010). Olfaction and predator-avoidance behavior are negatively affected in some fish species, including pink salmon (Ou et al. 2015; Leduc et al. 2013). Pink salmon also showed reductions in growth and metabolic capacity under elevated carbon dioxide conditions (Ou et al. 2015). Some high-quality salmon prey (e.g., krill) might be negatively affected by ocean acidification, but there are several possible pathways by which higher trophic levels might compensate for changes at a lower trophic level and impacts could conceivably be positive (Busch et al. 2013).

Because habitat for many shark and ray species is comprised of open ocean environments occurring over broad geographic ranges, large-scale impacts such as global climate change that affect ocean temperatures, currents, and potentially food chain dynamics, may impact these species. Chin et al. (2010) conducted an integrated risk assessment to assess the vulnerability of several shark and ray species on the Great Barrier Reef to the effects of climate change.

Scalloped hammerheads were ranked as having a low overall vulnerability to climate change, with low vulnerability to each of the assessed climate change factors (i.e., water and air temperature, ocean acidification, freshwater input, ocean circulation, sea level rise, severe weather, light, and ultraviolet radiation). In another study on potential effects of climate change to sharks, Hazen et al. (2012) used data derived from an electronic tagging project and output from a climate change model to predict shifts in habitat and diversity in top marine predators in the Pacific out to the year 2100. Results of the study showed significant differences in habitat change among species groups but sharks as a whole had the greatest risk of pelagic habitat loss. Marine populations that are already at risk due to other threats are particularly vulnerable to the direct and indirect effects of climate change. Several ESA-listed species and habitats considered in this opinion have likely already been impacted by this threat through the pathways described above.

Salinity plays an important role in the movement and distribution of some nearshore and estuarine fish species (Simpfendorfer et al. 2011). Rising sea levels associated with climate change will likely shift the salt wedge upstream in affected rivers. Given the importance of salinity, changes in freshwater flow regimes into estuaries as a result of climate change will affect fish populations by potentially changing their distributions. Anadromous fish species (e.g., sturgeon and salmon) spawn in fresh water reaches of rivers because early life stages have little to no tolerance for salinity. If the salt wedge moves further upstream, sturgeon spawning and rearing habitat could be restricted. In river systems with dams or natural falls that are impassable by sturgeon and salmonids, the extent that spawning or rearing may be shifted upstream to compensate for the shift in the movement of the salt wedge would be limited. Simpfendorfer et al. (2011) found that juvenile smalltooth sawfish moved farther inland into estuary reaches within their preferred salinity range. Sea level rise will also likely impact important sawfish mangrove habitats as sediment surface elevations for mangroves will not keep pace with conservative projected rates sea level rise (Gilman et al. 2008).

In the NMFS final rule to list 20 coral species as threatened (79 FR 53851), ocean warming and acidification, associated with climate change, were identified as two of the most important threats to the current or expected future extinction risk of reef building corals. Reef building organisms are predicted to decrease the rate at which they deposit $CaCO_3$ in response to increased ocean acidity and warmer water temperatures (Raymundo et al. 2008). Further, the most severe coral bleaching events observed to date have typically been accompanied by ocean warming events such as the El Niño-Southern Oscillation (Glynn 2001). Bleaching episodes result in substantial loss of coral cover, and result in the loss of important habitat for associated reef fishes and other biota. Corals can typically withstand mild to moderate bleaching, but severe or prolonged bleaching events can lead to coral colony death (79 FR 53851). While the susceptibility to ocean warming and acidification associated with climate change is expected to vary by species and specific coral colony (based on latitude, depth, bathymetry, etc.; 79 FR

53851), climate change is expected to have major impacts on the coral species considered in this opinion.

At the margins of temperate and tropical bio-regions and within tidally-restricted areas where sea grasses are growing at their physiological limits, increased temperatures may result in losses of seagrasses and/or shifts in species composition (Short et al. 2007). The response of seagrasses to increased water temperatures will depend on the thermal tolerance of the different species and their optimum temperature for photosynthesis, respiration, and growth (Short and Neckles 1999). With future climate change and potentially warmer temperatures, there may be a 1-5 m rise in the seawater levels by 2100 when taking into account the thermal expansion of ocean water and melting of ocean glaciers. Rising sea levels may adversely impact seagrass communities due to increases in water depth above present meadows, thus reducing available light. Climate change may also reduce light by shifting weather patterns to cause increased cloudiness. Changing currents may cause erosion, increased turbidity and seawater intrusions higher up on land or into estuaries and rivers, which could increase landward seagrass colonization (Short and Neckles 1999). A landward migration of seagrasses with rising sea levels is a potential benefit, so long as suitable substrate is available for colonization.

It is uncertain how Johnson's seagrass will adapt to rising sea levels and temperatures. Much depends on how much and how quickly temperatures increase. For example, Johnson's seagrass that grows intertidally (e.g., in some parts of the Lake Worth Lagoon) may be affected by a slight change in temperature (since it may already be surviving under less than optimal conditions). However, this may be ameliorated with rising sea levels, assuming Johnson's seagrass would migrate landward with rising sea levels and assuming that suitable substrate would be available for a landward migration. However, rising sea levels could also adversely impact seagrass communities due to increases in water depths above existing meadows reducing available light.

7.2 Anthropogenic Sound

The ESA-listed species that occur in the action area are regularly exposed to multiple sources of anthropogenic sounds. Anthropogenic sound is generated by commercial and recreational vessels, aircraft, sonar, ocean research activities, dredging, construction, offshore mineral exploration, military testing and training activities, seismic surveys, and other human activities. These activities occur within the action area to varying degrees throughout the year. ESA-listed species have the potential to be impacted by increased levels of both background sound and high intensity, short-term sounds. Sources of anthropogenic noise are becoming both more pervasive and more powerful, increasing both oceanic background sound levels and peak intensity levels (Hildebrand 2004).

Sounds are often considered to fall into one of two general types, impulsive and non-impulsive, which differ in the potential to cause physical effects to animals (see Southall et al. 2007 for in-depth discussion). Impulsive sound sources produce brief, broadband signals that are atonal transients and occur as isolated events or repeated in some succession. They are characterized by a relatively rapid rise from ambient pressure to a maximal pressure value followed by a rapid decay period that may include a period of diminishing, oscillating maximal and minimal pressures, and generally have an increased capacity to induce physical injury. Non-impulsive sounds can be tonal, narrowband, or broadband, brief or prolonged, and may be either continuous or non-continuous. Some can be transient signals of short duration but without the essential properties of pulses (e.g., rapid rise time). The duration of non-impulsive sounds, as received at a distance, can be greatly extended in a highly reverberant environment.

Anthropogenic sound within the marine environment is recognized as a potential stressor that can harm marine animals and significantly interfere with their normal activities (NRC 2005). The species considered in this opinion may be impacted by anthropogenic sound in various ways. Damage to marine mammal hearing and mass stranding events due to high-intensity sound exposure have been documented (Hildebrand 2004). Anthropogenic sounds may also produce a behavioral response including, but not limited to, changes in habitat to avoid areas of higher sound levels, changes in diving behavior, or (for cetaceans) changes in vocalization (MMC 2007). Many researchers have described behavioral responses of marine mammals to the sounds produced by boats and vessels, as well as other sound sources such as helicopters and fixed-wing aircraft, and dredging and construction. Most observations have been limited to short-term behavioral responses, which include temporary cessation of feeding, resting, or social interactions. Habitat abandonment can lead to more long-term effects, which may have implications at the population level. Interference, or masking, occurs when a sound is a similar frequency and similar to or louder than the sound an animal is trying to hear (Francis 2013). Masking can interfere with an individual's ability to gather acoustic information about its environment, such as predators, prey, conspecifics, and other environmental cues (Richardson 1995). Masking can reduce the range of communication, particularly long-range communication, such as that for blue and fin whales. Recent scientific evidence suggests that marine mammals, including blue and fin whales, compensate for masking by changing the frequency, source level, redundancy, or timing of their signals, but the long-term implications of these adjustments are currently unknown (Parks 2009;2003; Mcdonald et al. 2006).

There are limited studies on the hearing abilities of sea turtles, their uses of sounds, and their vulnerability to sound exposure. Some evidence suggests that sea turtles are able to detect (Bartol et al. 1999; Bartol and Ketten 2006; Ridgway et al. 1969; Martin et al. 2012) and behaviorally respond to acoustic stimuli (DeRuiter and Doukara 2012; McCauley et al. 2000; Moein et al. 1995; O'Hara and Wilcox 1990). Sea turtles may use sound for navigation, locating prey, avoiding predators, and general environmental awareness (Dow Piniak et al. 2012).

For fishes, the effects of anthropogenic sound have been well documented. However, due to the sheer diversity and numbers of fish, much remains unknown about fishes' abilities to detect and respond to sound. Sensitivity to sound also varies among fishes, and many fish species have developed sensory mechanisms that enable them to detect, localize, and interpret sounds in their environment. When considering the effects of anthropogenic sound on fishes, it is those sound sources that have the potential to cause physical injury and mortality to the individual or disrupt essential behavioral patterns; and whether or not these effects pose a risk to the population of a particular species that are a great concern. These would be acute or limited in duration sound exposures such as those sounds generated during construction activities, use of explosives, and seismic surveys. However, chronic and continuous sound sources such as those produced from vessels or alternative energy sources are also a concern, especially if they could result in fitness consequences and decrease survival and recovery of fishes. Thus, understanding of how fishes detect and respond to sound needs to be tied to ecologically relevant factors such as fish physiology and specific life stage needs, in conjunction with spatial patterns and distribution within the habitats they occupy.

Despite the potential impacts on individual ESA-listed marine mammals and sea turtles, information is not currently available to determine the potential population level effects of cumulative anthropogenic sound sources in the marine environment (MMC 2007). For example, we currently lack empirical data on how sound impacts growth, survival, reproduction, and vital rates, nor do we understand the relative influence of such effects on the population being considered. As a result, the consequences of anthropogenic sound on ESA-listed marine mammals and sea turtles at the population or species scale remain uncertain.

7.2.1 Vessel Noise

Much of the increase in sound in the ocean environment over the past several decades is due to increased shipping, as vessels become more numerous and of larger tonnage (Mckenna et al. 2012; Hildebrand 2009; NRC 2003a). Shipping constitutes a major source of low-frequency sound in the ocean (Hildebrand 2004), particularly in the Northern Hemisphere where the majority of vessel traffic occurs. The northeastern U.S. hosts some of the busiest commercial shipping lanes in the world, including those leading into Boston, Providence, Newark, and New York. While commercial shipping vessels contribute a large portion of oceanic anthropogenic noise, other sources of maritime traffic can be present in large numbers and impact the marine environment. These include recreational boats, whale-watching boats, research vessels, and ships associated with oil and gas activities. Individual vessels produce unique acoustic signatures, although these signatures may change with vessel speed, vessel load, and activities that may be taking place on the vessel. Sound levels are typically higher for the larger and faster vessels. Peak spectral levels for individual commercial vessels are in the frequency band of ten to 50 Hz and range from 195 dB re: μPa^2-s at one m for fast-moving (greater than 20 knots) supertankers to 140 dB re: μPa^2-s at one m for smaller vessels (NRC 2003a). Although large vessels emit predominantly low frequency sound, studies report broadband sound from large cargo vessels above two kHz, which may interfere with important biological functions of cetaceans (Holt

2008). At frequencies below 300 Hz, ambient sound levels are elevated by 15 to 20 dB when exposed to sounds from vessels at a distance (McKenna et al. 2013).

7.2.2 Pile Driving and Construction Sound

Industrial activities and construction both in the ocean and along the shoreline can contribute to underwater noise. Pile driving is commonly used for the construction of foundations for a large number of structures including bridges, buildings, retaining walls, harbor facilities, offshore wind turbines, and offshore structures for the oil and gas industry. Impact hammer pile driving during construction activities is of particular concern because it generates noise with a very high source level. During pile installation, noise is produced when the energy from construction equipment is transferred to the pile and released as pressure waves into the surrounding water and sediments. The impulsive sounds generated by impact pile driving are characterized by a relatively rapid rise time to a maximal pressure value followed by a decay period that may include a period of diminishing, oscillating maximal and minimal pressures (Illingworth and Rodkin Inc. 2001; Illingworth and Rodkin 2007; Reyff 2012). The amount of noise produced by pile driving depends on a variety of factors, including the type and size of the impact hammer, size of the pile, the properties of the sea floor, and the depth of the water. The predominant energy in pile impact impulses is at frequencies below approximately 2000 Hz, with the majority of the sound energy associated with pile driving is in the low frequency range, less than 1,000 Hz (Laughlin 2006; Reyff 2009; Reyff 2012; Illingworth and Rodkin Inc. 2001;2004; NMFS 2018a). Pressure levels from 190-220 decibels referenced to one micropascal root mean square (dB re: one µPa rms) were reported for piles of different sizes in a number of studies (NMFS 2018a). The majority of the sound energy associated with pile driving is in the low frequency range (<1,000 Hz; Illingworth and Rodkin Inc. 2001;2004; Reyff 2003). Impact pile driving occurs over small spatial and temporal scales and produces high-intensity, low-frequency, impulsive sounds with high peak pressures that can be detected by mammals, sea turtles and other marine species such as fish (Dow Piniak et al. 2012). Injury to the inner ear of marine mammals and fishes is caused by pressure damage to hair cells in the inner ear, ear canals, or eardrums. Barotrauma can also result in fishes and result in both lethal and non-lethal physical injuries. No specific studies have been conducted on hearing effects from pile driving exposure for sea turtles, but anatomical similarities of the inner ear in sea turtles to both fish and marine mammals makes it probable that they could experience similar effects on their ears and hearing from this sound source Moreover, sea turtles have been shown capable of detecting and responding to other impulsive sound sources such as airguns (McCauley et al. 2000; Popper et al. 2014). Vibratory pile driving produces a continuous sound with peak pressures lower than those observed in impulses generated by impact pile driving (Popper et al. 2014).

7.3 Military Training and Testing Activities

The Navy conducts training, testing, and other military readiness activities on range complexes throughout coastal and offshore areas of the action area. Activities are conducted off the Atlantic coast and in the Gulf of Mexico, in the Hawaiian and Southern California Range Complexes, in the Pacific Northwest, and in the high seas. The Navy has been conducted training and testing activities in these locations for decades and proposes to continue conducting similar activities. The USCG also conducts some training activities in the action area, such as gunnery training for polar icebreakers in the Pacific Northwest. For this reason, military training and testing activities in the action area are mentioned here as part of the baseline.

During training, activities include routine gunnery, missile, surface fire support, amphibious assault and landing, bombing, sinking, torpedo, tracking, and mine exercises. Testing activities are conducted for different purposes and include at-sea research, development, evaluation, and experimentation. The Navy performs testing activities to ensure that its military forces have the latest technologies and techniques available to them.

Military activities produce sound and visual disturbances to ESA-listed marine mammals, sea turtles, and fishes throughout the action area. Impacts from harassment due to Navy activities include changes from foraging, resting, milling, and other behavioral states that require lower energy expenditures to traveling, avoidance, and behavioral states that require higher energy expenditures. Sound produced during Navy training and testing activities results in instances of TTS for fishes, marine mammals, and sea turtles, and permanent threshold shift (PTS) in hearing sensitivity of marine mammals and sea turtles. The Navy training and testing activities constitute a federal action and take of ESA-listed marine mammals, sea turtles, fishes, and invertebrates considered for these Navy activities have previously undergone section 7 consultations. Through these consultations with NMFS, the Navy has implemented monitoring and conservation measures to reduce the potential effects of underwater sound from military training and testing activities on ESA-protected resources in the training and testing areas in Hawaii and Southern California, the Pacific Northwest, the Atlantic Ocean, the Gulf of Mexico, and in open ocean. Conservation measures include employing visual observers and implementing mitigation exclusion zones when training and testing activities use harmful sound sources such as active sonar or explosives.

The Air Force has also conducted training and testing activities in the action area in the past, and these activities are ongoing and expected to continue into the future. Air Force activities generally involve the firing or dropping of munitions (e.g., bombs, missiles, rockets, and gunnery rounds) from aircraft towards targets located on the surface, though Air Force training exercises may also involve boats. These activities affect ESA-listed species through physical disturbance, boat strikes, debris, and effects from sound and pressure produced by detonations and the use of vessels and in-water sonar and other equipment for navigation or other purposes. Air Force training and testing activities constitute a federal action and take of ESA-listed species resulting from these Air Force activities have previously undergone separate section 7 consultations.

The USCG conducts gunnery training and other training activities in the action area, including in the Pacific Northwest from polar icebreakers. Gunnery training activities involve the use of small arms loaded with practice rounds (non-explosive) and firing from the vessel toward floating or aerial targets. The USCG also engages in activities in its areas of responsibilities that may involve the use of guns as part of law enforcement activities, and deployment of spill response equipment as part of its environmental response activities, among other. As for other military activities, ESA-listed species in the action area may be affected by physical disturbance, vessel strikes, marine debris, and noise. ESA section 7 consultations have been conducted for USCG activities in the action area to address effects to ESA-listed species and their designated critical habitat.

7.3.1 Surveillance Towed Array Sensor System Low Frequency Active Sonar

The Navy's Surveillance Towed Area Sensor System (SURTASS) sonar has a vertical line array of 18 elements operating between 100 and 500 Hz. The typical low-frequency active sonar signal is not a constant tone but consists of various waveforms that vary in frequency and duration. A complete sequence of sound transmissions (waveforms) is referred to as a wavetrain (also known as a ping). These wavetrains last between six and 100 seconds, with an average length of 60 seconds. Within each wavetrain, a variety of signal types can be used, including continuous wave and frequency-modulated signals. The duration of each continuous-frequency sound transmission within a wavetrain is no longer than ten seconds. The interval between transmissions varies between six and 15 minutes. SURTASS low-frequency active sonar has a coherent low frequency signal with a duty cycle of less than 20 percent. Prior to 2017, the Navy has only used SURTASS low-frequency active sonar in the western and central North Pacific Ocean. However, in 2017 the Navy requested programmatic section 7 consultation for the operation of SURTASS low-frequency active sonar from August 2017 through August 2022 in the non-polar region of the world's oceans (including within the action area). The Navy's program will allow each ship (of which there are four) with SURTASS to utilize the system a maximum of 255 hours per year per ship (or 1,050 hours total). ESA section 7 consultation was concluded in August 2017 (NMFS 2017d) and considered the effects of Navy's SURTASS low-frequency active program as well as specific SURTASS low-frequency active annual activities on ESA-listed species within the action area for this UNDS consultation.

7.4 Dredging

Nearshore and offshore coastal areas are often dredged to support commercial shipping, recreational boating, construction of infrastructure, and marine mining. Hydraulic dredging can directly harm large marine animals (e.g., sea turtles and sturgeon) by lethally entraining them through the dredge drag-arms and impeller pumps. Large animals that are entrained in hydraulic dredges rarely survive the encounter. Hopper dredges, in particular, are capable of moving relatively quickly compared to turtles and fishes, which can be overtaken and entrained in the suction draghead of the advancing dredge.

Dredging operations also emit sounds at levels that could potentially disturb individuals of many marine taxa. Depending on the type of dredge, peak SPLs from 100 to 140 dB re: one µPa were reported in one study (Clarke et al. 2003). As with pile driving, most of the sound energy associated with dredging is in the low-frequency range, less than 1000 Hz (Clarke et al. 2003). Based on a literature review of the impacts of dredging activities on marine mammals, Todd et al. (2014) found that dredging is unlikely to cause physiological damage to marine mammal auditory systems, but is more likely to lead to masking and behavioral disturbances, especially for baleen whales, which are more at risk than other taxa. An estimated 609 incidental takes (lethal or sublethal interactions) of sea turtles were documented from hopper dredging activity in the southeastern U.S. from 1980 through 2006 (Dickerson et al. 2007). Dickerson (2006) reported 15 Atlantic sturgeon taken in dredging activities conducted by the U.S. Army Corps of Engineers from 1990 to 2010, most captured by hopper dredge. Notably, these reports include only those trips when an observer was on board to document capture.

Dredging projects within the action area mainly occur in the harbors, ports, and nearshore coastal areas. Considering the locations of past and ongoing dredging, the species most likely affected by dredging within the action area are sea turtles and ESA-listed corals (in areas such as South Florida), as well as Johnson's seagrass. Dredging can also indirectly affect marine species through habitat modification, changes in prey availability, and water quality degradation, including changes in DO and salinity gradients (Jenkins et al. 1993; Secor and Niklitschek 2001; Campbell and Goodman 2004). Designated critical habitat can also be directly and indirectly affected by dredging activities. For example, dredging and filling can impact sturgeon habitat features by disturbing benthic fauna, eliminating deep holes, and altering rock substrates (Smith and Clugston 1997). As benthic omnivores, sturgeon are particularly sensitive to modifications of the benthos that affect the quality, quantity and availability of prey species. Hatin et al. (2007) reported avoidance behavior by Atlantic sturgeon during dredging operations and McQuinn and Nellis (2007) found that Atlantic sturgeon were substrate dependent and avoided dredge spoil dumping grounds.

The dredging of bottom sediments to maintain, or in some cases create, inlets, canals, and navigation channels may affect seagrasses and corals by direct removal, light limitation due to turbidity, and burial from sedimentation. The disturbance of sediments can also destabilize the benthic community and sediment resuspension may release nutrients and other contaminants to the water column, which could result in over-enrichment and/or reduced DO levels, among other effects. Altering benthic topography or burying the plants and/or corals may remove them from the photic zone and the altered shape and depth of the bottom within the dredged footprint may affect future growth.

In addition to dredging, the construction of docks, marinas, bridges, and other in-water structures can impact Johnson's seagrass and corals through direct removal but also indirectly through habitat effects (e.g., shading and increased turbidity). Similar to dredging, installation of piles for docks or bridges can result in increased turbidity that can negatively impact water transparency over short durations. Species within the construction footprint are directly impacted and, if they cannot be transplanted to another site, are lost. Completed structures can have long-term effects on the availability of habitat for the species in the surrounding area because of the shade they produce. While shading does not affect water transparency directly, it does affect the amount and/or duration of sunlight that can reach the bottom.

7.5 Water Quality and Marine Debris

Several different types of anthropogenic pollution resulting from past, present and ongoing human activities adversely affect ESA-listed species and habitats within the action area. For this opinion, we focus on three primary categories of marine and estuarine pollutants: contaminants and pesticides; nutrient loading and algal blooms; and marine debris. This section provides a general discussion of the three major pollutant categories above, including the stressor pathways and anticipated effects on ESA- protected resources, with an emphasis on geographic areas, habitats or species within the action area is an area of high-density offshore oil extraction with chronic, low-level spills and occasional massive spills. Oil spills remain a significant threat to marine ecosystems in the Gulf of Mexico due to the large amount of extraction and refining activity in the region. There are approximately 4,000 oil and gas structures in the northern Gulf of Mexico, 90 percent of which are off Louisiana and Texas (USN 2009).

The largest spill within the action area occurred in April of 2010 as a result of a fire and explosion aboard the semisubmersible drilling platform Deepwater Horizon roughly 80 km southeast of the Mississippi Delta (NOAA 2010). Once the platform sank, the riser pipe connecting the platform to the wellhead on the seafloor broke in multiple locations, initiating an uncontrolled release of oil from the exploratory well. Over the next three months, oil was released into the Gulf of Mexico, resulting in oiled regions of Texas, Louisiana, Mississippi, Alabama, and Florida and widespread oil slicks throughout the northern Gulf of Mexico that are closed more than one-third of the Gulf of Mexico Exclusive Economic Zone to fishing due to contamination concerns. Apart from the widespread surface slick, massive undersea oil plumes formed, possibly through the widespread use of dispersants, and reports of tarballs washing ashore throughout the region were common. NOAA has estimated that 4.9 million barrels of oil were released (Lubchenco et al. 2010).

Oil released into the marine environment contains aromatic organic chemicals known to be toxic to a variety of marine life (Yender et al. 2002). Oil spills can impact wildlife directly through three primary pathways: (1) ingestion—when animals swallow oil particles directly or consume prey items that have been exposed to oil, (2) absorption—when animals come into direct contact with oil, and (3) inhalation—when animals breathe volatile organics released from oil or from "dispersants" applied by response teams in an effort to increase the rate of degradation of the oil in seawater. Direct exposure to oil can cause acute damage including skin, eye, and respiratory irritation, reduced respiration, burns to mucous membranes such as the mouth and eyes, diarrhea, gastrointestinal ulcers and bleeding, poor digestion, anemia, reduced immune response, damage to kidneys or liver, cessation of salt gland function, reproductive failure, and death (NOAA 2003;2010; Vargo et al. 1986c; Vargo et al. 1986a;b). Nearshore spills or large offshore spills that reach shore can oil beaches on which sea turtles lay their eggs, causing birth defects or mortality in the nests (NOAA 2010;2003). Disruption of other essential behaviors, such as breeding, communication, and feeding may also occur. The loss of invertebrate communities due to oiling or oil toxicity would also decrease prey availability for ESA-listed sea turtles, fishes, and whales (NOAA 2003). Sea turtles species which commonly forage on crustaceans and mollusks may be vulnerable to oil ingestion due to oil adhering to the shells of these prey and the tendency for these organisms to bioaccumulate toxins found in oil (NOAA 2003). Furred marine mammals such as pinnipeds may not be able to thermoregulate normally if oil coats their fur. Seagrass beds may be particularly susceptible to oiling as oil contacts grass blades and sticks to them, hampering photosynthesis and gas exchange (Wolfe et al. 1988). If spill cleanup is attempted, mechanical damage to seagrass can result in further injury and long-term scarring. Loss of seagrass due to oiling would be important to green sea turtles, as this is a significant component of their diets (NOAA 2003). Sea turtles are known to ingest and attempt to ingest tar balls, which can block their digestive systems, impairing foraging or digestion and potentially causing death (NOAA 2003).

Availability of light is one of the most significant environmental factors affecting the survival, growth, abundance, and distribution of seagrasses (Bulthuis 1983; Dennison 1987; Abal et al. 1994; Kenworthy and Fonseca 1996). Light availability also affects corals that do not receive energy from their zooxanthellae if these cannot photosynthesize. Water quality and the penetration of light are affected by turbidity (suspended solids), color, nutrients, and chlorophyll. Increases in color and turbidity values throughout the range of Johnson’s seagrass are generally caused by high flows of freshwater discharged from water management canals, which can also reduce salinity. Wastewater and storm water discharges, land runoff, and subterranean sources are also causes of increased turbidity throughout the range of Johnson’s seagrass and Atlantic/Caribbean corals. Degradation of water quality due to increased land use and poor water management practices continues to threaten the welfare of seagrass and coral communities. Declines in water quality are likely to worsen, unless water management and land use practices can curb or eliminate freshwater discharges and minimize inputs of sediments, nutrients, and other contaminants. Degradation of water quality also occurs in areas where maritime activities are concentrated due to leaching of materials, such as anti-fouling coatings, and discharges (accidental or otherwise) from vessels.

7.5.1 Contaminants and Pesticides

Coastal habitats are often in close proximity to major sources of pollutants and contaminants, which make their way into the marine environment from land-based industrial, domestic and agricultural sources. Sources include wastewater treatment plants, septic systems, industrial facilities, agriculture, animal feeding operations, and improper refuse disposal. Agricultural discharges, as well as discharges from large urban centers, contribute contaminants as well as coliform bacteria to coastal watersheds. Contaminants can be carried long distances from terrestrial or nearshore sources and ultimately accumulate in offshore pelagic environments (USCOP 2004). Global oceanic circulation patterns result in a considerable amount of pollutants that are scattered throughout the open ocean and accumulating in gyres and other places due to circulation patterns (Crain et al. 2009).For example, the Gulf of Mexico portion of the action area is a major sink for pollution from a variety of marine and terrestrial sources, which ultimately can interfere with ecosystem health and particularly that of ESA-listed species and their habitats. The Mississippi River drains 80 percent of the U.S. cropland (including the fertilizers, pesticides, herbicides, and other contaminants that are applied to it) and discharges into the Gulf of Mexico (MMS 1998). Vessel discharges, including those from private, commercial, and military vessels also contribute to pollutant loading to marine waters, particularly in harbors and other areas where vessels concentrate.

Chemical contaminants, particularly those that are persistent in the environment, are a particular concern for marine animals that often occupy high trophic positions. Persistent organic pollutants, which include legacy pesticides (e.g., dichlorodiphenyltrichloroethane [DDT], chlordane), legacy industrial-use chemicals (e.g., polychlorinated biphenyls), and emerging contaminants of concern (e.g., polybrominated diphenyl ethers and perfluorinated compounds), accumulate in fatty tissues of marine organisms and are magnified through the food web leading high exposure levels in upper trophic predators (National Academies of Sciences and Medicine 2016). Ocean contamination resulting from chemical pollutants is a concern for cetacean conservation and has been the subject of numerous studies (Desforges et al. 2016; Fair et al. 2010; Krahn et al. 2007; Moon et al. 2010; Ocean Alliance 2010). High concentrations of polychlorinated biphenyls (PCBs) and DDT have been reported in tissues of marine mammals in most parts of the world, particularly in coastal regions adjacent to heavy coastal development and/or industry. These legacy persistent organic pollutants have been linked to a number of adverse health effects including endocrine disruption, reproductive impairment or developmental effects, and immune dysfunction or disease susceptibility (National Academies of Sciences and Medicine 2016). Polybrominated diphenyl ethers commonly used as flame retardants, are another class of persistent organic pollutants that have spread globally in the environment and have also been reported in a broad array of marine mammal species (National Academies of Sciences and Medicine 2016).

Savery et al. (2014) documented detectable lead concentration in 93 percent of 337 blubber biopsies from sperm whales sampled throughout the world. Ylitalo et al. (2008) analyzed blubber and blood samples for organochlorines from 158 Hawaiian monk seals at four of their six primary breeding colonies in the Northwestern Hawaiian Islands. They found that the health and fitness of Hawaiian monk seals from three of the four subpopulations may be at risk from elevated contaminant levels. Lopez et al. (2012) examined concentrations of a large suite of persistent organic pollutants in blubber and serum of juvenile and adult monk seals from the Main Hawaiian Islands. Adult females had the lowest blubber levels of most persistent organic pollutants, whereas adult males had the highest levels. Contaminant levels from the Main Hawaiian Islands were at similar or lower levels than those from remote Northwestern Hawaiian Island populations. In an analysis of cetacean blubber samples obtained from animals stranded in Hawaii between 1997 and 2011, higher levels of persistent organic pollutants were found in killer whale and false killer whale, as opposed to baleen whales, which had lower levels (Bachman et al. 2014).

Levels of chromium in North Atlantic right whale tissues are sufficient to be mutagenic and cause cell death in lung, skin, or testicular cells and are a concern for the species' recovery (Wise et al. 2008; Chen et al. 2009). The organochlorines DDT, dichlorodiphenyldichloroethylene, PCBs, dieldrin, chlordane, hexachlorobenzene, and heptachlor epoxide have been isolated from blubber samples and reported concentrations may underestimate actual levels (Woodley et al. 1991). Mean PCB levels in North Atlantic right whales are greater than any other baleen whale species thus far measured, although less than one-quarter of the levels measured in harbor porpoises (Van Scheppingen et al. 1996; Gauthier et al. 1997). Flame retardants such as polybrominated diphenyl ethers (known to be carcinogenic) have also been measured in North Atlantic right whales (Montie et al. 2010).

The chemical components of pesticides used on land flow as runoff into the marine environment and can bioaccumulate in the bodies of marine mammals, which can then be transferred to their young through mother's milk (Fair et al. 2010). There is growing evidence that the presence of chemical contaminants in their tissues puts marine mammals at greater risk for adverse health effects and potential impact on their reproductive success (Fair et al. 2010; Godard-Codding et al. 2011; Krahn et al. 2007).

Despite the vast evidence indicating that marine animals are exposed to anthropogenic, as well as natural, chemicals capable of producing significant toxic effects, only a few studies have actually examined the impacts on population survival or reproductive rates. Such observational assessments are inherently challenging due to the difficulty in controlling for confounding or interacting variables, as well as the sublethal but chronic nature of chemical contaminant effects, and the difficulty of observing mortality or reproductive endpoints, particularly in long-lived species such as cetaceans and sea turtles (National Academies of Sciences and Medicine 2016).

Polycyclic aromatic hydrocarbons (PAHs) represent another group of organic compounds that can result in adverse effects on marine species. Anthropogenic sources of PAHs include crude oil, fumes, vehicle exhaust, coal, organic solvents, and wildfires. Exposure may be continual, associated with run-off from impervious cover in developed coastal regions, or natural seeps that produce low-level but steady exposure. Acute events such as oil spills may produce pulses of more significant exposure. Vessels regularly discharge small amounts of PAHs and may also be responsible for large discharges due to accidental spills. Depending on the route of exposure (inhalation/aspiration, ingestion, direct dermal contact), PAHs can produce a broad range of health effects including lung disease, disruption of the hypothalamic-pituitary-adrenal axis, and altered immune response (National Academies of Sciences and Medicine 2016). Although PAHs are more rapidly metabolized and do not accumulate, as is the case with persistent organic pollutants, the toxic effects (lung disease, hypothalamic-pituitary-adrenal axis damage) may be long-lasting and initiate chronic disease conditions.

A variety of heavy metals have been found in sea turtles' tissues in levels that increase with turtle size. These include arsenic, barium, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, selenium, silver, and zinc, (Godley et al. 1999; Fujihara et al. 2003; Storelli et al. 2008; García-Fernández et al. 2009; Barbieri 2009). Cadmium has been found in leatherbacks at the highest concentration compared to any other marine vertebrate (Gordon et al. 1998). Newly emerged hatchlings have higher concentrations than are present when eggs are laid, suggesting that metals may be accumulated during incubation from surrounding sands (Sahoo et al. 1996). Arsenic has been found to be very high in green turtle eggs (Van De Merwe et al. 2009). Sea turtle tissues have been found to contain organochlorines, including chlorobiphenyl, chlordane, lindane, endrin, endosulfan, dieldrin, perfluorooctane sulfonate, perfluorooctanoic acid, DDT, and PCB (Gardner et al. 2003; Keller et al. 2005; Alava et al. 2006; Storelli et al. 2007; Oros et al. 2009). PCB concentrations are reportedly equivalent to those in some marine mammals, with liver and adipose levels of at least one congener being exceptionally high (Oros et al. 2009; Davenport et al. 1990). Levels of PCBs found in green sea turtle eggs have exceeded recommended levels for are considered far higher than what is fit for human consumption (Van De Merwe et al. 2009).

Several studies have reported correlations between organochlorine concentration level and indicators of sea turtle health or fitness. Organochlorines have the potential to suppress the immune system of loggerhead sea turtles and may affect metabolic regulation (Keller et al. 2006; Oros et al. 2009). Accumulation of these contaminants can also lead to deficiencies in endocrine, developmental and reproductive health (Storelli et al. 2007). Balazs (1991) suggested that environmental contaminants are a possible factor contributing to the development of the viral disease fibropapillomatosis in sea turtles by reducing immune function. Day et al. (2007) investigated mercury toxicity in loggerhead sea turtles by examining trends between blood mercury concentrations and various health parameters. They concluded that subtle negative impacts of mercury on sea turtle immune function are possible at concentrations observed in the wild. Keller et al. (2004) investigated the possible health effects of organochlorine contaminants, such as PCBs and pesticides on loggerhead sea turtles. Although concentrations were relatively low compared with other species, they found significant correlations between organochlorine contaminants levels and health indicators for a wide variety of biologic functions, including immunity and homeostasis of proteins, carbohydrates, and ions.

Many pollutants in the environment, such as brevotoxins, heavy metals, and PCBs, have the ability to bioaccumulate in fish species. Exposure to chemical pollutants may act in an additive or synergistic manner with other stressors resulting in significant population level consequences (Desforges et al. 2016). The life histories of sturgeon species (i.e., long lifespan, extended residence in estuarine habitats, benthic foraging) predispose them to long-term, repeated exposure to environmental contamination and potential bioaccumulation of heavy metals and other toxicants (Dadswell 1979). Heavy metals and organochlorine compounds accumulate in sturgeon tissue, but their long-term effects are not well studied (Ruelle and Keenlyne 1993). Shortnose sturgeon collected from the Delaware and Kennebec Rivers had total toxicity equivalent concentrations of polychlorinated dibenzo-p-dioxins, polychlorinated dibenzofurans, PCBs, dichlorodiphenyldichloroethylene, aluminum, cadmium, and copper all above adverse effect concentration levels reported in the literature (Brundage III 2008). Dioxin and furans were detected in ovarian tissue from shortnose sturgeon caught in the Sampit River/Winyah Bay system (South Carolina).

High levels of contaminants, including chlorinated hydrocarbons, in several other fish species are associated with reproductive impairment (Giesy et al. 1986; Cameron et al. 1992; Billsson 1998; Hammerschmidt et al. 2002), reduced survival of larval fish (Willford et al. 1981; McCauley et al. 2015), delayed maturity and posterior malformations (Billsson 1998). Pesticide exposure in fishes may affect anti-predator and homing behavior, reproductive function, physiological maturity, swimming speed, and distance (Beauvais et al. 2000; Scholz et al. 2000; Waring and Moore 2004). Sensitivity to environmental contaminants also varies by life stage. Early life stages of fish appear to be more susceptible to environmental and pollutant stress than older life stages (Rosenthal and Alderdice 1976). Early life stage Atlantic and shortnose sturgeon are vulnerable to PCB and Tetrachlorodibenzo-p-dioxin toxicities of less than 0.1 parts per billion (Chambers et al. 2012). Increased doses of PCBs and Tetrachlorodibenzo-p-dioxin have been correlated with reduced physical development of Atlantic sturgeon larvae, including reductions in head size, body size, eye development and the quantity of yolk reserves (Chambers et al. 2012). Juvenile shortnose sturgeon raised for 28 days in North Carolina’s Roanoke River had a nine percent survival rate compared to a 64 percent survival rate at non-riverine control sites (Cope et al. 2011). The reduced survival rate could not be correlated with contaminants, but significant quantities of retene, a paper mill by-product with dioxin-like effects on early life stage fish, were detected in the river (Cope et al. 2011).

Dwyer et al. (2005) compared the relative sensitivities of common surrogate species used in contaminant studies to 17 ESA-listed species including Atlantic sturgeon. The study examined 96-hour acute water exposures using early life stages where mortality is an endpoint. Chemicals tested were carbaryl, copper, 4-nonphenol, pentachlorophenal and permethrin. Of the ESA-listed species, Atlantic sturgeon were ranked the most sensitive species tested for four of the five chemicals (Atlantic and shortnose sturgeon were found to be equally sensitive to permethrin).

Additionally, a study examining the effects of coal tar, a byproduct of the process of destructive distillation of bituminous coal, indicated that components of coal tar are toxic to shortnose sturgeon embryos and larvae in whole sediment flow-through and coal tar elutriate static renewal (Kocan et al. 1993).

7.5.2 Nutrient Loading and Algal Blooms

Industrial and municipal activities can result in the discharge of large quantities of nutrients into coastal waters. Excessive nutrient enrichment results in eutrophication, a condition associated with degraded water quality, algal blooms, oxygen depletion, loss of seagrass and coral reef habitat, and in some instances the formation of hypoxic "dead zones" (USCOP 2004). Hypoxia (low DO concentration) occurs when waters become overloaded with nutrients such as nitrogen and phosphorus, which can enter the marine environment from agricultural runoff, sewage treatment plants, bilge water, atmospheric deposition, and other sources. An overabundance of nutrients can stimulate algal blooms resulting in a rapid expansion of microscopic algae (phytoplankton). When excess nutrients are consumed, the algae population dies off and the remains are consumed by bacteria. Bacterial consumption decreases the DO level in the water which may result in mortality of fishes and crustaceans, reduced benthic and demersal organism abundance, reduced biomass and species richness, and abandonment of habitat to areas that are sufficiently oxygenated (Craig et al. 2001; Rabalais et al. 2002). Higher trophic level species (e.g. turtles and marine mammals) may be impacted by the reduction of available prey as a result of hypoxic conditions. For example, high nutrient loads from the Mississippi River create a massive hypoxic "dead zone" in Northern Gulf of Mexico each year. This hypoxic event occurs annually from as early as February to as late as October, spanning from the Mississippi River Delta to Galveston, Texas. In 2017, NOAA estimated that the Gulf of Mexico Dead Zone covered over 8,000 square miles, an area about the size of New Jersey.

Marine algal toxins are produced by unicellular algae that are often present at low concentrations but that may proliferate to form dense concentrations under certain environmental conditions (National Academies of Sciences and Medicine 2016). When high cell concentrations form, the toxins that they produce can harm marine life, and this is referred to as a harmful algal bloom. Marine mammals can be exposed to harmful algal bloom toxins directly by inhalation or indirectly through food web transfer, and these toxins can cause severe neurotoxic effects (Van Dolah 2005). Mortality and morbidity related to harmful algal bloom toxins have been increasingly reported over the past several decades, and biotoxicosis has been a primary contributor to large scale die-offs across marine mammal taxa (Van Dolah 2005; Simeone et al. 2015). Domoic acid has also been detected in tissues of marine mammals along the southeast U.S. coast (Twiner et al. 2011), but perhaps of greater concern in this area are the brevetoxins produced by Gulf of Mexico red tides. Brevetoxin has been implicated in multiple die-offs involving common bottlenose dolphins, as well as the endangered Florida manatee (Flewelling et al. 2005; Twiner et al. 2012; Simeone et al. 2015). Capper et al. (2013) found that both turtles and manatees were exposed to multiple HAB toxins (okadaic acid, brevetoxins, saxitoxins, and likely others) in Florida. A recent survey of the peer reviewed literature on marine mammal diseases and reports of marine mammal mass mortality events suggests an increase in the frequency of marine mammal die-offs resulting from exposure to harmful algal blooms over the past 40 years (Gulland and Hall 2007).

California coastal harmful algal bloom problems are dominated by two organisms: *Alexandrium catenella* which produces saxitoxin, the causative agent of paralytic shellfish poisoning, and several *Pseudo-nitzschia* species whose toxic strains produce domoic acid, the causative agent for Amnesic Shellfish Poisoning (alternately called Domoic Acid Poisoning; Anderson et al. 2008). Prior to 2000, toxic blooms were considered rare and unusual in southern California (Lange et al. 1994). In 2006, Busse et al. (2006) reported the presence of domoic acid in San Diego during elevated abundances of toxic Pseudo-nitzschia and concurrently in fishes and mussels. This study provides evidence for the transfer of domoic acid from a local algal source in San Diego to higher trophic levels. Unlike many other ecosystems impacted by harmful algal blooms, the physical, chemical, and ecological characteristics of the coastal waters of California are largely dominated by upwelling. Consequently, upwelling circulation overrides both the nutrient limitation of stratified waters and the light limitation of well-mixed waters, and generally nourishes these waters with macronutrients in excess of anthropogenic sources (Anderson et al. 2008). This does not, however, preclude the possibility that the growth of these algae, their toxicity, and the frequency or duration of toxic events may be exacerbated by anthropogenic nutrient inputs once these populations reach nearshore waters (Anderson et al. 2008).

Red tides have been reported off the coast of southern California for over a century (McGowan et al. 2017). Red tides occur when blooms of marine phytoplankton reach such high concentrations that the sea surface becomes noticeably discolored. In La Jolla, California, blooms are often caused by bioluminescent dinoflagellates (e.g., *Lingulodinium polyedrum*) (McGowan et al. 2017). Red tides and other algal blooms in southern California can be caused by toxic algal species, resulting in fish and shellfish mortality (Lewitus et al. 2012). Regardless of toxicity, the sheer concentrations of organisms can lead to oxygen depletion and fish kills when blooms persist over extended periods.

7.5.3 Marine Debris

Marine debris has become a widespread threat for a wide range of marine species that are increasingly exposed to it on a global scale. Plastic is the most abundant material type worldwide, accounting for more than 80 percent of all marine debris (Poeta et al. 2017). The most common impacts of marine debris are associated with ingestion or entanglement. Both types of interactions can result in injury or death of many different marine species taxa. Ingestion occurs when debris items are intentionally or accidentally eaten (e.g. through predation on already contaminated organisms or by filter feeding activity, in the case of large filter feeding marine organisms, such as whales) and enter in the digestive tract. Ingested debris can damage digestive systems and plastic ingestion can also facilitate the transfer of lipophilic chemicals (especially persistent organic pollutants) into the animal's bodies. Entanglement in fishing gear also represents a major, on-going threat to many marine species. An estimated 640,000 tons of fishing gear is lost, abandoned, or discarded at sea each year throughout the world's oceans (Macfadyen et al. 2009). These "ghost nets" drift in the ocean and can fish unattended for decades (ghost fishing), killing, injuring or impairing large numbers of marine animals through entanglement.

Marine debris is a significant concern for ESA-listed species, particularly sea turtles and marine mammals. The initial developmental stages of all turtle species are spent in the open sea. During this time both juvenile turtles and their buoyant food are drawn by advection into fronts (convergences, rips, and driftlines). The same process accumulates large volumes of marine debris, such as plastics and lost fishing gear, in ocean gyres (Carr 1987). An estimated four to twelve million metric tons of plastic enter the oceans annually (Jambeck et al. 2015). It is thought that sea turtles eat plastic because it closely resembles jellyfish, a common natural prey item (Schuyler 2014). Ingestion of plastic debris can block the digestive tract which can cause turtle mortality as well as sub-lethal effects including dietary dilution, reduced fitness, and absorption of toxic compounds (Laist et al. 1999; Lutcavage et al. 1997). Santos et al. (2015) found that a surprisingly small amount of plastic debris was sufficient to block the digestive tract and cause death. They reported that 10.7 percent of green turtles in Brazilian waters were killed by plastic ingestion, while 39.4 percent had ingested enough plastic capable of killing them. These results suggest that debris ingestion is a potentially important source of turtle mortality, one that may be masked by other causes of death.

(Gulko and Eckert 2003) estimated that between one-third and one-half of all sea turtles ingest plastic at some point in their lives. A more recent study by (Schuyler et al. 2015) estimates that 52 percent of sea turtles globally have ingested plastic debris. Schuyler et al. (2016) synthesized the factors influencing debris ingestion by turtles into a global risk model, taking into account the area where turtles are likely to live, their life history stage, the distribution of debris, the time scale, and the distance from stranding location. They found that up to 52 percent of sea turtles globally have ingested plastic debris and oceanic life stage turtles are at the highest risk of debris ingestion. Based on their model, olive ridley turtles are the most at-risk species; green, loggerhead, and leatherback turtles were also found to be at a high and increasing risk from plastic ingestion (Schuyler et al. 2016). This study also found the North Pacific gyre, which encompasses portions of the action area in Hawaii and Southern California, to be a regional hotspot for sea turtle debris ingestion.

The North Pacific Subtropical gyre is a clockwise circular pattern of four prevailing ocean currents (North Pacific, California, North Equatorial, and Kuroshio currents) where debris from around the North Pacific Rim gathers and circulates (PIFSC 2016). The reefs and islands of

Northwestern Hawaiian Islands, in particular, act as a filter amassing marine debris that presents potentially lethal entanglement hazards and ingestion threats to numerous birds and marine animals within the action area. From 1996 through 2014, nearly 837 metric tons (1.8 million lbs) of marine debris, primarily derelict fishing gear, have been removed from the shallow reefs and shorelines of the Northwestern Hawaiian Islands (PIFSC 2016). The regions of highest risk to global turtle populations are off the east coasts of the U.S., Australia, and South Africa; the east Indian Ocean, and Southeast Asia. In addition to ingestion risks, sea turtles also become entangled in marine debris such as fishing nets, monofilament line, and fish-aggregating devices (Laist et al. 1999; Lutcavage et al. 1997; NRC 1990). Turtles are particularly vulnerable to ghost nets due to their tendency to use floating objects for shelter and as foraging stations (Dagorn et al. 2013; Kiessling 2003).

Marine mammals are also highly susceptible to the threats associated with marine debris and many cases of ingestion and entanglement have been reported around the world (Poeta et al. 2017). Baulch and Perry (2014) found that the proportion of cetacean species ingesting debris or becoming entangled in debris is increasing. Based on stranding data, they found that recorded rates of ingestion have increased by a factor of 1.9 and rates of entanglement have increased by a factor of 6.5 over the last forty years (1970-2010). Ingestion of marine debris can also have fatal consequences for large whales. In 2008, two male sperm whales stranded along the northern California coast with large amounts of fishing net scraps, rope, and other plastic debris in their stomachs. One animal had a ruptured stomach, the other was emaciated, and gastric impaction was suspected as the cause of both deaths (Jacobsen et al. 2010). de Stephanis et al. (2013) also describe a case of mortality of a sperm whale related to the ingestion of large amounts of marine debris in the Mediterranean Sea.

Marine debris has the potential to impact ESA-listed fishes through ingestion or entanglement, as well as impacts to habitat used by these species. Ingested debris may lead to blockage in the stomach or intestines of animals, affecting their ability to continue feeding, and can also cause ruptures in the stomach or intestines. Entanglement in debris causes injury or mortality of fish species as they struggle to free themselves.

Marine debris may also impact coral reef ecosystems. For example, Chiappone et al. (2002) conducted surveys of the Florida Keys and documented marine debris entanglement in reef areas. The authors documented damage from marine debris on coral reef habitat, including damage to scleractinian corals (likely inclusive of ESA-listed corals such as elkhorn and staghorn coral). Similarly, Johnson’s seagrass, which is in an area with extensive residential development including in-water structures for recreational vessels, as well as commercial and other uses, is also impacted from debris associated with the construction and operation of these facilities, and due to transport of debris during storms when in-water structures suffer damage from winds and storm surge.

7.6 Whaling

Whale populations within the action area have historically been impacted by aboriginal subsistence hunting, small-scale commercial whaling and, more recently, large-scale commercial whaling using factory ships. From 1864 through 1985, at least 2,400,000 baleen whales (excluding minke whales) and sperm whales were killed worldwide (Gambell 1999). From 1900 to 1965 nearly 30,000 humpback whales were taken in the Pacific Ocean, with an unknown number of additional animals taken prior to 1900 (Perry et al. 1999). Sei whales were estimated to have been reduced to 20 percent (8,600 out of 42,000) of their pre-whaling abundance in the North Pacific (Tillman 1977). In addition, 9,500 blue whales and 25,800 sperm whales were reported killed by commercial whalers in the North Pacific between 1910-1965 (Ohsumi and Wada 1972; Barlow et al. 1997). Many of the whaling numbers reported in the 20th century likely represent minimum estimates, as illegal or underreported catches are not included. For example, recently uncovered Union of Soviet Socialists Republics catch records indicate extensive illegal whaling activity between 1948 and 1979 (Ivashchenko et al. 2014).

Prior to current prohibitions on whaling, most large whale species were significantly depleted to the extent it was necessary to list them as endangered under the Endangered Species Preservation Act of 1966. Since the end of large-scale commercial whaling, the primary threat to these species has been eliminated, although many whale species have not yet fully recovered from those historic declines. Although commercial whaling no longer targets the large, endangered whales in the action area, historical whaling may have altered the age structure and social cohesion of these species in ways that continue to influence them.

In 1982, the International Whaling Commission issued a moratorium on commercial whaling, which went into effect in 1986. There is currently no legal commercial whaling by International Whaling Commission Member Nations party to the moratorium; however, whales are still killed commercially by countries that filed objections to the moratorium. Presently three types of whaling take place: (1) aboriginal subsistence whaling to support the needs of indigenous people; (2) special permit whaling; and (3) commercial whaling conducted either under objection or reservation to the International Whaling Commission moratorium (i.e., Iceland and Norway). Some of the whales killed in these fisheries are likely part of the same population of whales occurring within the action area for this consultation. Whale populations in the action area are likely at reduced numbers due to historic whaling.

7.7 Directed Harvest of Sea Turtles

Sea turtles have been harvested throughout history as both a protein source (for meat or eggs) and as raw material in the manufacture of ornaments and artifacts. An additional threat unique to hawksbill turtles is the tortoiseshell trade. Tortoiseshell is made from hawksbill scutes and is used to produce products such as sunglasses, bracelets, and ornamental boxes that are often sold on the black market (Shattuck 2011).

For centuries, the harvest of sea turtles and turtle eggs was primarily limited to small-scale, artisanal and subsistence fisheries. In many parts of the world, the customs and traditions associated with the harvest, consumption and artistic use of sea turtle products have been passed from generation to generation and have developed cultural meaning and significance over time (Campbell 2003). Historically, green turtles have played a large role in Polynesian and Micronesian cultures. In addition to being used as a food source, native peoples all over the Pacific utilized all parts of the turtle making tools and jewelry out of the bones, and containers and utensils out of the carapace.

Although small-scale turtle fisheries still exist today, by the mid-20th century directed turtle harvest was dominated by large-scale commercial operations with access to global markets (Stringell et al. 2013). The Hawaiian green turtle was in a steep decline as of the 1970s because of direct harvest of both turtles and eggs by humans. By the late 1960s, the global capture of sea turtles had peaked at an estimated 17,000 tons (FAO 2011). Based on Japanese commercial import data, between 1970 and 1986 an estimated two million turtles (mostly hawksbills, greens, and olive ridleys) were harvested to satisfy the demand for turtle products in Japan alone (Milliken and Tokunaga 1987). To maximize efficiency, commercial harvesting effort was often concentrated at mass nesting sites or arribadas with high densities of breeding adult turtles.

Increased conservation awareness at the international scale has led to greater protection of marine turtles in recent decades. The CITES, which went into effect in 1975, helped to reduce demand and promote regional cooperation in increasing turtle populations. All six ESA-listed sea turtles are listed in CITES Appendix I, which provides the greatest level of protection, including a prohibition on commercial trade. Marine turtle species have also been listed on the International Union for Conservation of Nature Red List of Threatened Species since 1982 (IUCN 2017). In 1981, Ecuador, one of the two largest turtle harvesting nations at the time, banned the export of sea turtle products. In 1990, following international pressures, Mexico, the other major turtle exporter, closed commercial fisheries and instituted a moratorium on the take of turtles and eggs (Senko et al. 2014).

Humber et al. (2014) documented the change in the legal take of sea turtles over the past three decades. Just considering the 46 countries that still allow sea turtle directed take (including the four with current moratoria), turtle harvest has decreased by more than 60 percent over the past three decades. The average number of turtles killed in these fisheries annually has declined steadily over time: 116,420 in the 1980s; 68,844 turtles in the 1990s; and 45,387 in the 2000s (Humber et al. 2014). While legal directed take of sea turtles has declined significantly, illegal harvest may still represent a significant source of sea turtle mortality, one that is more difficult to estimate. The scale of global illegal take is likely to be severely underreported due to the inherent difficulty in collecting data on such activity (Humber et al. 2014), including within portions of the action area such as the U.S. Caribbean.

7.8 Commercial Fisheries

Depending on the gear type, commercial fisheries can result in damage to habitat used by ESA-listed species, as well as to the species themselves. In the case of ESA-listed Atlantic/Caribbean corals, bottom gear types can directly affect coral colonies, as well as causing habitat degradation. The placement of bottom-tending gear in many areas is now regulated by fishery management councils in order to minimize potential impacts to benthic habitats, including corals, but the loss of gear, such as fish traps, during storms, the use of unauthorized gear, or the placement of gear in unauthorized areas still have some degree of impacts to corals. Surveys of marine debris in St. Thomas found ghost traps in many areas containing coral, apparently associated with unauthorized fishing in a closed area off the south coast of the island. In other portions of the action area, bottom-tending gear types can damage critical habitat, as well as causing bycatch and entanglement of ESA-listed species including whales, sea turtles, and fishes.

7.8.1 Sea Turtle Interactions with Fisheries

Bycatch of ESA-listed sea turtles occurs in a diversity of fisheries throughout the broad geographic oceanic ranges of these species. Sea turtle bycatch occurs in both large-scale commercial fishing operations as well as small-scale and artisanal fisheries throughout the action area. Fishing gears that are known to interact with sea turtles include trawls, longlines, purse seines, gillnets, pound nets, dredges and to a lesser extent, pots and traps (Finkbeiner et al. 2011; Lewison et al. 2013). Sea turtle bycatch rates (i.e., individuals captured per unit of fishing effort) and mortality rates (i.e., individuals killed per number captured) can vary widely both within and across particular fisheries due to a combination of factors. These include gear types and gear configurations, fishing methods (e.g., depth fished, soak times), fishing locations, fishing seasons, time fished (i.e., day versus night), and turtle handling and release techniques used (Wallace et al. 2010; Lewison et al. 2013). Entanglement in fishing gear and/or plastics can result in severe ulcerative dermatitis, and amputation of flippers (Orós et al. 2005). If mortality is not directly observed during gear retrieval, it may occur after the turtle is released due to physiological stress and injury suffered during capture. Recent studies indicate that underwater entrapment in fishing gear (i.e., trawls and gillnets) followed by rapid decompression when gear is brought to the surface may cause gas bubble formation within the blood stream (i.e., embolism) and tissues leading to organ injury, impairment, and even post-release mortality in some bycaught turtles (Garcia-Parraga et al. 2014; Fahlman et al. 2017).

Lewison et al. (2014) used the bycatch data from 1990-2008 to identify global hotspots of turtle bycatch intensity. High-intensity sea turtle bycatch was most prevalent in three regions: the eastern Pacific Ocean, southwest Atlantic Ocean, and Mediterranean Sea. Spotila et al. (2000) reported a conservative estimate of annual leatherback fishery-related mortality (from longlines, trawls and gillnets) in the Pacific Ocean during the 1990s of 1,500 animals. He estimated that this represented about a 23 percent mortality rate (or 33 percent if most mortality was focused on the East Pacific population). Lewison et al. (2004) estimated between 2,600 and 6,000 loggerhead turtles were captured and killed in Pacific Ocean longline fisheries in 2000.

West Coast Fisheries

The west coast longline fishery operates in the north Pacific ocean, mainly from the U.S. Exclusive Economic Zone (EEZ) west to 140 degree west longitude and from the equator to 35 degree north (NMFS 2016c). This fishery primarily targets bigeye tuna, although other tuna and non-tuna species are caught and retained. As of 2016 there was only one boat participating in this fishery, although fishing effort is expected to increase in the future (NMFS 2016c). Sea turtle incidental take authorized over a ten-year period (starting in 2016) as provided for in the ITS of the 2016 opinion (NMFS 2016c) on this fishery is as follows: one green sea turtle (lethal or non-lethal), East Pacific DPS and Central North Pacific DPS; four total, up to two lethal takes of leatherback sea turtle; one loggerhead (lethal or non-lethal), North Pacific DPS; and six total (lethal or non-lethal) olive ridley sea turtles.

The west coast drift gillnet fishery targets swordfish and thresher sharks in the U.S. EEZ and adjacent high seas off the coasts of California, Oregon, and Washington (NMFS 2013). In 2001, NMFS established Pacific Sea Turtle Conservation Areas that prohibit drift gillnet fishing in large portions of the historical fishing grounds, either seasonally or conditionally, to protect endangered leatherback and loggerhead sea turtle. The primary turtle species captured in U.S. fisheries in Atlantic and Gulf of Mexico fisheries is the loggerhead (Moore et al. 2009). The southeastern U.S. comprises one of the largest aggregate nesting rookeries for loggerhead sea turtles in the world, and the continental shelf provides critical ontogenetic habitats for this population. Thus, because a large number of individuals are present throughout areas of high fishing activity, loggerheads interact with a greater number of fishing fleets and gear types in the Atlantic than other sea turtle species (Moore et al. 2009).

Entanglement in fishing gear represents an important source of injury and mortality in sea turtles. Fisheries interactions are likely to have significant demographic effects on many populations (66 FR 44549; August 24, 2001). Oregon and Washington state laws currently prohibit landings caught with drift gillnet gear, although vessels still fish drift gillnets in federal waters off these states and land their catch in California. The drift gillnet fishery can also be closed during El Niño events in order to reduce bycatch of loggerhead turtles that move further north on the warm El Niño currents from Mexico into U.S. waters (72 FR 31756, June 8, 2007).

In 2013, NMFS issued an opinion on the continued authorization of the west coast drift gillnet fishery (NMFS 2013). Sea turtle incidental take authorized over a five-year period in the ITS of the opinion is as follows: two total, up to one lethal of green sea turtles; ten total, up to seven lethal of leatherback sea turtles; seven total, up to four lethal of loggerhead sea turtles; and two total, up to one lethal of olive ridley turtles.

Hawaii Pelagic Longline

Domestic longline fishing around Hawaii consists of two separately managed fisheries: a deep-set fishery that primarily targets bigeye tuna and a shallow-set fishery that targets swordfish. The shallow-set fishery operates almost entirely north of Hawaii. The deep-set fishery operates primarily to the south of Hawaii, although in some years this fishery expands northward and overlaps with the shallow-set fishery.

In 1999, the shallow-set longline fishery targeting swordfish was closed by court order due to high levels of sea turtle bycatch. Before the closure took effect, an estimated 417 loggerheads and 110 leatherbacks (McCracken 2000) were captured annually (with about 40 percent mortality Gilman et al. 2007) in Hawaii's longline fisheries (shallow and deep-set combined). Subsequent court orders led to regulations in 2001 prohibiting all Hawaii longline vessels from targeting swordfish until 2004. When the shallow-set fishery was reopened in 2004, it was restricted to considerably less fishing effort than pre-2001 levels. As a result, the deep-set fishery targeting tuna made up an increasingly larger proportion of Hawaii's longline fishing effort since 2004. A final rule published in 2004 (69 FR 17329) established a limited shallow-set swordfish fishery and required the use of circle hooks with mackerel-type bait, a combination that had proven effective at reducing interactions with leatherback and loggerhead turtles in the Atlantic longline fishery (Watson et al. 2005). The use of circle hooks with mackerel-type bait reduced sea turtle interaction rates by approximately 90 percent for loggerheads and 83 percent for leatherbacks compared to the previous period 1994-2002 when the shallow-set fishery was operating without these requirements (Gilman et al. 2007). Annual sea turtle bycatch limits (17 loggerhead or 16 leatherback turtles) were also established for the swordfish fishery as part of the 2004 rule. From 2005 through 2014, the Hawaii-based longline fisheries resulted in an estimated total of 15 loggerhead and 17 leatherback mortalities in the shallow-set fishery, and 16 loggerhead, 45 leatherback, and 264 olive ridley mortalities in the deep-set fishery (NMFS 2014a).

In addition to gear restrictions and bycatch limits, Hawaii longline vessel operators are required to take an annual NMFS protected species workshop that instructs fishers in mitigation, handling, and release techniques for sea turtles, seabirds, and marine mammals. Longline fishermen must carry and use specific equipment, and follow certain procedures for handling and releasing sea turtles, seabirds, and marine mammals that may be caught incidentally.

In 2012, NMFS issued an opinion on the continued operation of the Hawaii shallow-set longline fishery (NMFS 2012a). Sea turtle incidental take authorized over a continuous two-year calendar period in the ITS of the opinion is as follows: six total takes, including up to two lethal of green sea turtles; 52 total, including up to 12 lethal of leatherback sea turtles; 68 total including up to 14 lethal of loggerhead sea turtles, North Pacific DPS; and four total, including up to two lethal of olive ridley sea turtles.

In 2014, NMFS issued an opinion on the continued operation of the Hawaii deep-set longline fishery (NMFS 2014a). Sea turtle incidental take authorized over a three-year period in the ITS of the opinion is as follows: nine total, potentially all lethal of green sea turtles; 72 total, including up to 27 lethal of leatherback sea turtles; nine total, potentially all lethal, of loggerhead sea turtles, North Pacific DPS; and 99 total, up to 96 lethal of olive ridley sea turtles.

Southeast Fisheries

The Southeast shrimp trawl fishery in the Atlantic and Gulf of Mexico has historically accounted for the overwhelming majority (up to 98 percent) of sea turtle bycatch in U.S. fisheries (Finkbeiner et al. 2011). Regulations that went into effect in the early 1990's require shrimp trawlers in the Atlantic and Gulf of Mexico to modify their gear with turtle excluder devices (TEDs) designed to allow turtles to escape trawl nets and avoid drowning. Analyses by Epperly and Teas (2002) indicated that, while early versions of TEDs were effective for some species, the minimum requirements for the escape opening dimension were too small for larger sea turtles, particularly loggerheads and leatherbacks. NMFS implemented revisions to the TED regulations in 2003 to address this issue (68 FR 8456, February 21, 2003). The revised TED regulations were estimated to reduce shrimp trawl related mortality by 94 percent for loggerheads and 97 percent for leatherbacks (NMFS 2014d). Finkbeiner et al. (2011) compared sea turtle bycatch estimated before and after the 2003 TED enlargement regulations. In the late 1990's, the southeast shrimp trawl fishery resulted in an estimated 340,500 sea turtle interactions and 133,400 mortalities. By comparison, by 2007 this fishery resulted in an estimated 69,300 interactions and 3,700 mortalities (Finkbeiner et al. 2011). The decline in sea turtle bycatch over this period can be attributed to a combination of the revised TED regulations and a significant decrease in fishing effort. Time-area closures have also been implemented to reduce sea turtle bycatch in shrimp trawl fisheries operating in particularly sensitive areas.

Although mitigation measures have greatly reduced the impact on sea turtle populations, the shrimp trawl fishery is still responsible for large numbers of turtle mortalities each year. The Gulf of Mexico fleet accounts for a large percentage of the sea turtle bycatch in this fishery. In 2010, the Gulf of Mexico shrimp trawl fishery had an estimated bycatch mortality of 5,166 turtles (18 leatherback, 778 loggerhead, 486 green and 3,884 Kemp's ridley). By comparison, the southeast Atlantic fishery had an estimated bycatch mortality of 1,033 turtles (eight leatherback, 673 loggerhead, 28 green and 324 Kemp's ridley) in 2010 (NMFS 2014d).

In 2014, NMFS issued an opinion for reinitiation of the section 7 consultation on the southeast shrimp trawl fishery (NMFS 2014d). Unlike most other fisheries, conventional observer programs are not effective for determining the numbers of sea turtle interactions and mortalities in this fishery. As a result, the ITS for the opinion is based on monitoring fishing effort and TED compliance rate as a surrogate for monitoring take. The baseline effort levels for this fishery, as established in the ITS, are 132,900 days fished in the Gulf of Mexico and 14,560 trips in the South Atlantic. The baseline TED compliance level is 88 percent.

Other federal fisheries within the action area that result in sea turtle bycatch and have undergone recent section 7 consultation include the coastal migratory pelagics fishery in the Atlantic and Gulf of Mexico (NMFS 2015d), the South Atlantic commercial snapper-grouper fishery (NMFS 2006b), reef fish fisheries in the Gulf of Mexico (NMFS 2011d) and Caribbean (NMFS 2011c), the spiny lobster fisheries operating in the Gulf of Mexico, South Atlantic, and the Caribbean (NMFS 2011e;2009b), and the Gulf of Mexico stone crab fishery (NMFS 2009c). Various fishing gears (e.g., trawls, pots, pound nets and gillnets) used in state waters from Maine through Texas are known to incidentally take sea turtles. However, information on turtle bycatch in these coastal, nearshore fisheries is often sparse. Although the past and current effects of state managed fisheries on sea turtles is currently not determinable, NMFS believes that ongoing state fishing activities may be responsible for seasonally high levels of observed sea turtles strandings in state waters on both the Atlantic and Gulf of Mexico coasts.

Atlantic Fisheries

The U.S. Atlantic pelagic longline fishery began in the early 1960s. This fishery is currently comprised of five distinct fishing sectors: Gulf of Mexico yellowfin tuna fishery; southern Atlantic swordfish fishery; Mid-Atlantic and New England swordfish and tuna fishery; U.S. Atlantic Distant Water swordfish fishery; and the Caribbean tuna and swordfish fishery. The pelagic longline fishery mainly interacts with leatherback sea turtles and pelagic juvenile loggerhead sea turtles. The estimated average annual bycatch in this fishery (all geographic areas combined) between 1992-2002 was 912 loggerhead interactions (including seven captured dead) and 846 leatherback interactions (including 11 captured dead NMFS 2004). These mortality estimates do not account for post-release mortality, which historically was likely substantial (NMFS 2014d). NMFS has taken numerous steps to reduce sea turtle bycatch and bycatch mortality in domestic longline fisheries. In 2001, NMFS implemented requirements for U.S. flagged vessels with pelagic longline gear on board to have line clippers and dipnets to remove gear on incidentally captured sea turtles (66 FR 17370). Specific handling and release guidelines designed to minimize injury to sea turtles were also implemented. In 2004, NMFS issued an opinion for the reinitiation of section 7 consultation on the Atlantic pelagic longline fishery (NMFS 2004). This opinion concluded that the pelagic longline fisheries were likely to jeopardize the continued existence of leatherback sea turtles. A Reasonable and Prudent Alternative was provided to avoid jeopardy that included take reduction measures related to fishing gear, bait, disentanglement gear, and training. NMFS published a final rule in 2004 to implement management measures to reduce sea turtle bycatch and bycatch mortality in the Atlantic pelagic longline fishery (69 FR 40734). Since 2004, bycatch estimates for both loggerheads and leatherbacks in pelagic longline gear have been well below the average numbers prior to implementation of gear regulations under the Reasonable and Prudent Alternative (NMFS 2012b). The pelagic longline fishery resulted in an estimated 259 loggerhead and 268 leatherback sea turtle interactions in 2014 (NMFS, 2015).

In 2012, NMFS issued an updated opinion on the federal shark fisheries managed under the Consolidated Highly Migratory Species Fishery Management Plan (NMFS 2012b). Gears used to capture sharks in these fisheries include bottom longlines, gillnets (drift, strike, and sink nets), and commercial and recreational rod-and-reel and handlines. The ITS exempted take of ESA-listed sea turtle species as follows: up to 57 captures every three years of which 24 could be lethal of green sea turtles, North Atlantic DPS; up to 18 captures every three years of which nine could be lethal of hawksbill sea turtles; up to 36 captures every three years of which 15 could be lethal of Kemp's ridley sea turtles; up to 18 captures every three years of which nine could be lethal of leatherback sea turtles; and up to 126 captures every three years of which 78 could be lethal of loggerhead sea turtles, Northwest Atlantic DPS.

Sea turtles overlap seasonally with the Atlantic sea scallop fishery in the Mid-Atlantic region from Cape Cod to southern Virginia when turtles migrate to this area to forage in early summer (Murray 2015). Loggerheads account for the large majority of interactions with this fishery. An estimated 200 interactions between loggerheads and scallop dredge fishing gear occurred on average annually from 2001 to early 2006 (Murray 2011). Subsequent fishing effort reductions and gear modifications implemented in this fishery reduced these interactions to less than 100 per year from late 2006 to 2008, and to an estimated 22 per year from 2009 to 2014 (Murray 2015).

Gillnets and bottom trawls are commonly used gears by many of the commercial fisheries operating in the northeastern U.S. Atlantic EEZ from North Carolina through Maine. These fisheries are also known to interact with large numbers of sea turtles, particularly loggerheads. waters from the North Carolina/South Carolina border to Chincoteague, Virginia. In 2013, NMFS issued a "batched" section 7 biological opinion on the following fisheries: Northeast multispecies; monkfish; spiny dogfish; Atlantic bluefish; Northeast skate complex; mackerel/squid/butterfish; and summer flounder /scup/black sea bass (NMFS 2013). Gillnet gear is used by five of the seven fisheries, and bottom trawl gear is used by six of the seven fisheries covered by the 2013 opinion. The "batched" fishery management plan opinion includes an ITS (amended March 10, 2016) that exempts the following take of Northwest Atlantic DPS loggerhead: up to 1,345 over any consecutive five-year period in gillnet gear, of which up to 835 may be lethal; up to 1,020 individuals over any consecutive five-year period in trawl gear, of which up to 335 may be lethal. Small numbers of leatherback, Kemp's ridley, and green sea turtles were also exempted in this ITS.

The most effective way to monitor sea turtle bycatch is to place trained observers aboard fishing vessels. Although observer programs have increased in recent decades, many fisheries still lack the level of observer coverage necessary to produce reliable estimates of bycatch and associated mortalities needed to assess fishery impacts on ESA-listed species. In 2007, NMFS established a new regulation (72 FR 43176) to annually review sea turtle interactions across fisheries, identify those that require monitoring, and require fishermen to accommodate observers if requested.

This annual process should help NMFS and the fishing industry learn more about sea turtle interactions with fishing operations, continually evaluate existing measures to reduce sea turtle takes, and determine whether additional measures to address prohibited sea turtle takes may be necessary to avoid exceeding established take limits.

Estimating sea turtle interactions and mortality rates associated with commercial fisheries globally remains challenging because a relatively small proportion of fisheries worldwide adequately monitor bycatch (Long and Schroeder 2004). Wallace et al. (2010) compiled a global database of reported marine turtle bycatch from 1990 to 2008 in gillnet, longline, and trawl fisheries. They concluded that bycatch is a moderate or high threat for more than three-fourths of all sea turtle regional management units, and represents the greatest overall threat to sea turtles globally (Wallace et al. 2010). Lewison et al. (2014) used the same 1990-2008 bycatch database as Wallace et al. 2010 to identify global hotspots of turtle bycatch intensity. High-intensity sea turtle bycatch was most prevalent in three regions: the eastern Pacific Ocean, southwest Atlantic Ocean, and Mediterranean Sea. In 1989, the U.S. passed legislation aimed at reducing the impact of global shrimp trawl fisheries bycatch on sea turtle populations. Section 609 of Public Law 101-162 prohibits the import of shrimp harvested with technology that may adversely affect certain species of sea turtles (16 U.S.C. 1537). The shrimp import prohibition does not apply if the Department of State certifies to Congress that the harvesting nation has a regulatory program and an incidental take rate comparable to that of the U.S. (i.e., require and enforce the use of TEDs), or, alternatively, that the fishing environment in the harvesting nation does not pose a threat of the incidental taking of sea turtles (64 FR 36946).

7.8.2 Marine Mammal Fishery Interactions

Entrapment and entanglement in commercial fishing gear is one of the most frequently documented sources of human-caused injury and mortality of marine mammal species. For some marine mammal populations, the impacts from fisheries likely have significant demographic effects of marine mammals (Read et al. 2006). Bycatch mortality is estimated globally to exceed hundreds of thousands of marine mammals each year (Read et al. 2006). Many marine mammals that die from entanglement in commercial fishing gear tend to sink rather than strand ashore, thus making it difficult to fully assess the magnitude of this threat. When not immediately fatal, entanglement or ingestion of fishing gear can impede the ability of marine mammals to feed and can cause injuries that eventually lead to infection and death (Cassoff et al. 2011; Moore and Van der Hoop 2012; Wells et al. 2008). Other sublethal effects of entanglement include increased vulnerability to additional threats, such as predation and ship strikes, by restricting agility and swimming speed. There are also costs likely to be associated with nonlethal entanglements in terms of energy and stress (Moore and Van der Hoop 2012).

In 1994, the Marine Mammal Protection Act (MMPA) was amended to formally require the development of a take reduction plan when bycatch exceeds a level considered unsustainable and would lead to marine mammal population declines if not mitigated. At least in part as a result of the MMPA bycatch amendment, estimates of bycatch in the Pacific declined by a total of 96 percent from 1994 to 2006 (Geijer and Read 2013). Cetacean bycatch declined by 85 percent from 342 in 1994 to 53 in 2006, and pinniped bycatch declined from 1,332 to 53 over the same time period.

From 2000 to 2012, an average of eight large whale entanglements were observed and reported per year in California (Saez et al. 2013). Confirmed reports of entangled animals likely represent only a small fraction of the total number of entanglements that are actually occurring. Humpback whales and gray whales are the most commonly entangled cetacean species off California. Other species reported over this time frame include sperm whales, minke whales and fin whales. Traps and pots are the most common fishing gears reported as entangling west coast whales, accounting for about 45 percent of entanglements (Saez et al. 2013). The number of large whale entanglements may be increasing over time. In 2016, 66 separate cases of entangled whales were reported off the coast of California, 51 of which were humpback whales (NMFS 2017b). About 20 percent of reported entanglements in 2016 were from Southern California (Santa Barbara, Los Angeles, Orange, and San Diego counties), including ten humpbacks, two blue whales and one gray whale.

Insufficient data exist on the incidental bycatch of Guadalupe fur seals in fishing gear, although some juvenile seals have been documented with entanglement injuries. There were 16 records of human-related deaths or serious injuries to Guadalupe fur seals from stranding data for the five-year period 2010-2014 (Carretta et al. 2017). These strandings included entanglement in marine debris and gillnet of unknown origin, and shootings. Observed human-caused mortality and serious injury for this stock very likely represents a fraction of the true impacts because not all cases are reported or documented (Carretta et al. 2017).

The total number of confirmed larges whales reported entangled in Hawaii from 2002 to 2014 was 88 or about seven per year (Lyman 2014). All but three of these reports (one sei whale and two sperm whales), were humpback whales. The most commonly reported gears associated with entanglements in Hawaii are fish pots (50 percent) and longlines (23 percent).

False killer whales in Hawaiian waters have been seen taking catches from commercial longlines and trolling lines There is a strong spatial component to bycatch of marine mammals, with ‘hotspots’ influenced by marine mammal density and fishing intensity (Lewison et al. 2014). In the Atlantic Ocean, marine mammal bycatch occurs in a diversity of fisheries and is most important in various gillnet and trawl fisheries of New England and the Mid-Atlantic coast, and in the pelagic longline fisheries of the Atlantic, Gulf of Mexico, and Caribbean. Entanglement in fishing gear has been identified as one of the leading causes of North Atlantic right whale mortality and has been identified as a factor inhibiting recovery of the species (Knowlton et al. 2012). The prevalence of scars on right whales associated with entanglements indicates the persistent and repetitive nature of this threat. Knowlton et al. (2012) reported that from 1980-2009, 519 out 626 photo-identified right whales (82.9 percent) had been entangled at least once and 306 of the 519 (59.0 percent) had been entangled more than once. Of the 50 reported North Atlantic right whale deaths between 1986 and 2002, there were 18 (six confirmed and 12 presumed) cases of fatal gear entanglement (Kraus et al. 2005). Entanglement in fishing gear is also a significant threat to humpback whales in the Northwest Atlantic. Robbins (2009) found 64.9 percent of the Gulf of Maine humpback population to have entanglement scarring when first assessed in 2003, acquiring new scarring at an average annual rate of 12.1 percent.

In addition to the threats of entanglement and entrapment, fisheries operations can also result in changes to the structure and function of marine ecosystems that adversely affect marine mammals, including loss of prey species and alteration of benthic structure. Overfishing of many fish stocks results in significant changes in trophic structure, species assemblages, and pathways of energy flow in marine ecosystems (Jackson et al. 2001; Myers and Worm 2003). These ecological changes may have important, and likely adverse, consequences for populations of marine mammals (DeMaster et al. 2001). For instance, depletion of preferred prey could lead to a less nutritional diet and decreased reproductive success.

In 2011 and 2012, NMFS and Department of Fisheries and Oceans Canada appointed an independent science panel to review the effects of salmon harvest on Southern Resident killer whales (NMFS 2016f). The panel concluded that at a broad scale, salmon abundance will likely influence the recovery of the whales, but that there was a great deal of uncertainty about whether current fisheries remove enough salmon to have a meaningful influence on the whales' status (NMFS 2016f). The report also provided valuable recommendations on future analysis and research that could be done to fill data gaps and reduce uncertainty.

7.8.3 ESA-Listed Fish Species Interactions with Fisheries

Atlantic Salmon

Commercial bycatch is not thought to be a major source of mortality for Gulf of Maine DPS Atlantic salmon. Beland (1984, cited in Fay et al. 2006) reported that fewer than 100 salmon per year were caught incidental to other commercial fisheries in the coastal waters of Maine. A more recent study found that bycatch of Maine Atlantic salmon in herring fisheries is not a significant mortality source (ICES 2005). Commercial fisheries for white sucker, alewife, and American eel conducted in state waters also have the potential to incidentally catch Atlantic salmon.

Recreational angling occurs for many freshwater fish species throughout the range of the Gulf of Maine DPS Atlantic salmon. As a result, Atlantic salmon can be incidentally caught (and released) by anglers targeting other species such as striped bass or trout. Studies on the effects of catch and release on trout and salmon have concluded that exhaustive exertion may result in significant physiological disturbances including mortality (Graham et al. 1982; Wood et al. 1983; Brobbel et al. 1996). Interactions with these fisheries operations can result in injury, including disfigurement to dorsal fins (Forney and Kobayashi. 2007; Nitta and Henderson 1993; Shallenberger et al. 1981; Baird and Gorgone 2005; Zimmerman 1983; McCracken and Forney

2010). Carretta et al. (2013) estimated that less than one (0.5) individual per year from the Main Hawaiian Islands insular false killer whale stock are killed or seriously injured during the course of fishing operations in the Hawaiian EEZ. NMFS published a final rule to implement the False Killer Whale Take Reduction Plan on November 29, 2012 (77 FR 71260). The final rule includes gear requirements ("weak" circle hooks and strong branch lines) in the deep-set longline fishery, longline closure areas, and training and certification for vessel owners and captains in marine mammal handling and release.

Conditions that contribute to Atlantic salmon post-release mortality include elevated water temperatures, exposure of the fish to air after capture, extremely soft water, low oxygen levels, low river flow and improper handling (Booth et al. 1995). The potential also exists for anglers to misidentify juvenile Atlantic salmon as brook trout, brown trout, or landlocked salmon (i.e., non-anadromous). A maximum length for landlocked salmon and brown trout (25 inches) has been adopted in Maine in an attempt to avoid the accidental harvest of sea-run Atlantic salmon due to misidentification.

Atlantic and Shortnose Sturgeon

Atlantic and shortnose sturgeon are taken incidentally in fisheries targeting other species in rivers, estuaries, and marine waters throughout their range (Collins et al. 1996; ASSRT 2007). Sturgeon are benthic feeders and as a result they are generally captured near the seabed unless they are actively migrating (Moser and Ross 1995). Sturgeon are particularly vulnerable to being caught in commercial gill nets. Therefore, fisheries using this type of gear account for a high percentage of sturgeon bycatch and bycatch mortality. Sturgeon have also been documented in the following gears: otter trawls, pound nets, fyke/hoop nets, catfish traps, shrimp trawls, and recreational hook and line fisheries.

Estimated rates of Atlantic sturgeon caught as bycatch in federal fisheries are highly variable and somewhat imprecise due to small sample sizes of observed trips. An estimated 1,385 individual Atlantic sturgeon were killed annually from 1989 to 2000 as a result of bycatch in offshore gill net fisheries operating from Maine through North Carolina (Stein et al. 2004). From 2001-2006 an estimated 649 Atlantic sturgeon were killed annually in offshore gill net and otter trawl fisheries. From 2006 to 2010 an estimated 391 Atlantic sturgeon were killed (out of 3,118 captured) annually in Northeast federal fisheries (Miller and Shepherd 2011).

Several federally regulated fisheries that may encounter Atlantic sturgeon have fishery management plans that have undergone section 7 consultation with NMFS. On December 16, 2013, NMFS issued a "batched" section 7 biological opinion on the following fisheries: Northeast multispecies; monkfish; spiny dogfish; Atlantic bluefish; Northeast skate complex; mackerel/squid/butterfish; and summer flounder /scup/black sea bass. The majority (73 percent) of all Atlantic sturgeon bycatch mortality in New England and Mid-Atlantic waters is attributed to the monkfish sink gill net fishery (ASMFC 2007). Observer data from 2001 to 2006 shows

224 recorded interactions between the monkfish fishery and Atlantic sturgeon, with 99 interactions resulting in death, a 44 percent mortality rate. For all seven fisheries combined, the following take of Atlantic sturgeon was authorized annually: 1,331 trawl interactions of which 42 may be lethal and 1,229 gill net interactions of which 155 may be lethal. The 2012 NMFS opinion on the Southeast shrimp trawl fishery exempted the take of Atlantic sturgeon as follows: 1,731 total interactions, including 243 captures of which 27 are expected to be lethal every three years. In 2012, NMFS provided an updated opinion on the Federal shark fisheries, including the smoothhound fishery on ESA-listed species. For the federal smoothhound fishery and shark fisheries combined, NMFS exempted the take of 321 Atlantic sturgeon over a three-year span, with 66 of those takes expected to be lethal.

Given the high prevalence of gill net and otter trawl use in nearshore coastal and inland fisheries, state managed fisheries may have a greater impact on Atlantic and shortnose sturgeon than federal fisheries using these same gear types. Commercially important state fisheries that interact with sturgeon include those targeting shrimp, Atlantic croaker, weakfish, striped bass, black drum, spot, shad, and spiny dogfish.

Gulf Sturgeon

Gulf sturgeon are susceptible to capture in commercial fisheries directed at other species that employ various trawling and entanglement gears. Gulf sturgeon are occasionally incidentally captured in state managed shrimp fisheries in bays and sounds along the northern Gulf of Mexico. Gulf sturgeon bycatch has also been documented in entanglement gear (trammel and gill nets) used to target gar in the Pearl River in southeast Louisiana, where (NMFS and USFWS 2009). While state regulations prohibit the taking or possession of gulf sturgeon (including roe), there is no available data to determine bycatch capture or mortality rates (NMFS and USFWS 2009).

Relocation trawling, associated mostly with the removal of sea turtles to avoid interactions with channel dredging and beach nourishment projects, has successfully moved several gulf sturgeon in recent years. These captures in near-shore waters illustrate the relative vulnerability of gulf sturgeon to incidental bycatch in fisheries that use trawls (NMFS and USFWS 2009).

The Florida “net ban”, approved by voter referendum in November 1994 and implemented in July 1995, made unlawful the use of entangling nets (i.e., gill and trammel nets) in Florida state waters. Other forms of nets (i.e., seines, cast nets, and trawls) were restricted, but not totally eliminated. Implementation of the net ban in Florida has likely benefited gulf sturgeon as they are residents of near-shore waters during much of their life span.

Federal fisheries that NMFS authorizes in the Gulf of Mexico have likely had a minor impact on gulf sturgeon. This is because gulf sturgeon occur in the Gulf of Mexico only during winter months and during that time, most migrate alongshore and to barrier island habitats within shallower state waters (NMFS and USFWS 2009).

Bocaccio and Yelloweye Rockfish

Rockfish are unintentionally captured as part of fishing activities targeting other species. Although fishers may return these fish to the water, the mortality rate of these fish is extremely high (Parker et al. 2006). Although there are some methods available that could lower the mortality rates of discarded rockfish (summarized by Palsson et al. 2009), application of these methods in the Puget Sound fishery would be difficult (Palsson et al. 2009). The Washington Department of Fish and Wildlife considers bycatch of rockfish to be a “high impact stressor” on rockfish populations (Palsson et al. 2009).

Palsson et al. (2009) report that more than 3,600 pieces of abandoned fishing gear (especially gill nets) have been located in Puget Sound. About 35 percent of this derelict gear has been removed. Derelict nets continue fishing and are known to kill rockfish. While the total impact of this abandoned gear has not been fully enumerated, the Washington Department of Fish and Wildlife has concluded that derelict gear is likely to moderately affect local populations of rockfish (Palsson et al. 2009).

Green Sturgeon

Take of Southern DPS green sturgeon in federal fisheries was prohibited as a result of the ESA 4(d) protective regulations issued in June of 2010 (75 FR 30714). Green sturgeon are occasionally encountered as bycatch in Pacific groundfish fisheries (Al-Humaidhi 2011), although the impact of these fisheries on green sturgeon populations is estimated to be small (NMFS 2015e). NMFS (2015e) estimates between 86 and 289 Southern DPS green sturgeon are annually encountered as bycatch in the state-regulated California halibut bottom trawl fishery.

Approximately 50 to 250 green sturgeon are encountered annually by recreational anglers in the lower Columbia River (NMFS 2015e), of which 86 percent are expected to be Southern DPS green sturgeon based on the higher range estimate of Israel et al. (2009). Green sturgeon are also caught incidentally by recreational anglers fishing in Washington outside of the Columbia River (NMFS 2015e). Southern DPS green sturgeon are also captured and released by California recreational anglers. Based on self-reported catch card data, an average of 193 green sturgeon were caught and released annually by California anglers from 2007 to 2013 (NMFS 2015e). Recreational catch and release can potentially result in indirect effects on green sturgeon, including reduced fitness and increased vulnerability to predation. However, the magnitude and impact of these effects on Southern DPS green sturgeon are not well studied.

Pacific Salmon

Commercial and recreational fisheries also result in “non-landed mortality” on chinook and other species which varies by the type of gear. Even fisheries designed to be selective either for species or to harvest specially marked hatchery fish will have some mortality associated with the hooking and handling of the released fish. These include fish that are brought to the boat but are released because they are too small (may die from hooking trauma), fish that are hooked but drop off before they are brought to the boat, and fish that die from entanglement in gillnet or purse seine gear and drop out before being landed. For each type of fishery (commercial troll, recreational, net, etc.), harvest managers add between five and 50 percent to the total catch to account for fish deaths due to release, drop-off and other harvest related impacts (PSTT and WDFW 2004).

Puget Sound chinook salmon are captured in fisheries that occur in Alaskan and Canadian waters, ocean fisheries off the West Coast of the contiguous U.S., and within the marine waters and freshwater tributaries of the Strait of Juan de Fuca and Puget Sound. These fisheries are conducted for commercial purposes, for sport/recreational catch, or for tribal ceremonial and subsistence objectives. Puget Sound chinook are captured through fisheries that are directed at the harvest of chinook but are intended to catch populations that are not threatened, such as hatchery-origin fish; or they may be harvested as incidental catch during fisheries for coho and other species of salmon. Chinook are captured using “troll” gear (hook and line) or they may be taken in a variety of net gear types. The impact of these fisheries varies by area, by season and for different individual populations of chinook (NMFS 2007b).

Although fisheries are not directed on Hood Canal summer chum, a sizeable number of Hood Canal summer chum have been harvested incidentally during fisheries directed at chinook and coho, which have overlapping run timing (NMFS 2007b). Substantial incidental catches in Strait of Juan de Fuca and Hood Canal fisheries in the 1980s prompted the NMFS Biological Review Team to consider past harvest levels to be a factor of decline for the Hood Canal summer chum in its 1998 status review (NMFS BRT 1997). During the high harvest years, harvest rates on individual summer chum populations averaged 20 percent (NMFS BRT 2003). Summer chum salmon are also harvested incidentally in British Columbia in pink and sockeye salmon fisheries in the Strait of Juan de Fuca, Johnstone and Georgia Straits; and in troll fisheries off the west coast of Vancouver Island. Canadian harvest declined in the 1990s due to significant reductions in coho and sockeye fishing. Chum salmon are regulated in the same major harvest management forums as chinook. In 1991, coho salmon fishing in the main part of Hood Canal was closed by the co-managers to protect natural coho runs, and modifications were made to the remaining coho and chinook fisheries throughout Puget Sound to protect summer chum. As a result of these efforts, exploitation rates on summer chum in Hood Canal have declined greatly, and have dropped to a cumulative average (including Canadian fisheries) of five percent or less in recent years. Additional information on the effects of harvest management on Hood Canal Summer Chum is contained in the Summer Chum Conservation Initiative (WDFW and PNPTT 2000) and the Hood Canal/ Eastern Strait of Juan de Fuca Summer Chum Salmon Recovery Plan (in progress) by the Hood Canal Coordinating Council (NMFS 2007b).

Steelhead Trout

Puget Sound steelhead are harvested in terminal tribal gillnet fisheries and in recreational fisheries (NMFS 2016d). Fisheries are directed at hatchery stocks, but some harvest of natural origin steelhead occurs incidentally to hatchery-directed fisheries. Winter-run hatchery steelhead production is primarily of Chambers Creek stock, which for several generations has been selected for earlier run timing than natural stocks to minimize fishery interactions. Hatchery production of summer-run steelhead is primarily of Skamania River (lower Columbia River Basin) stock, which has been selected for earlier spawn timing than natural summer-run steelhead to minimize interactions on the spawning grounds. In recreational fisheries, retention of wild steelhead is prohibited, so all harvest impacts occur as the result of release mortality and non-compliance. In tribal net fisheries, most fishery impacts occur in fisheries directed at salmon and hatchery steelhead (NMFS NWFSC 2015).

Most Puget Sound streams have insufficient catch and escapement data to calculate exploitation rates for natural steelhead (NMFS 2016d). Populations with sufficient data include those in the Skagit, Green, Nisqually, Puyallup, and Snohomish rivers. Exploitation rates differ widely among the different rivers, but all have declined since the 1970s and 1980s. Exploitation rates on natural steelhead during the earlier period averaged between ten percent and 40 percent, with some populations in the central and south parts of Puget Sound, such as the Green and Nisqually river populations, experiencing exploitation rates over 60 percent. Exploitation rates on natural steelhead over the past decade have been stable and generally less than five percent. Current exploitation rates are low enough that they are unlikely to substantially reduce spawner abundance for most steelhead populations in Puget Sound, and these rates are expected to continue for the near future (NMFS NWFSC 2015).

7.8.4 Aquaculture

Marine aquaculture systems are diverse, ranging from highly controlled land-based systems to open water cages that release wastes directly into the environment. Species produced in the marine environment are also diverse, and include seaweeds, bivalve mollusks, echinoderms, crustaceans, and finfish (Langan 2004). Aquaculture supplies more than 50 percent of all seafood produced for human consumption globally (NOAA Marine Aquaculture website https://www.fisheries.noaa.gov/topic/aquaculture). The National Offshore Aquaculture Act of 2005 (S. 1195) promoted offshore aquaculture development within the EEZ and established a permitting process that encourages private investment in aquaculture operations, demonstrations, and research. Marine aquaculture is expected to expand in the U. S. EEZ due to increased demand for domestically grown seafood, coupled with improved technological capacity to farm in the open ocean. Hawaii is the first state to successfully operate commercial open ocean aquaculture cages in the U.S.

Open-ocean aquaculture encompasses a variety of infrastructure designs; in the U.S., submersible cages are the model used for offshore finfish production (Naylor 2006). Aquaculture cages are anchored to the sea floor but can be moved within the water column. Cages are tethered to buoys that contain an equipment room and feeding mechanism and can be large enough to hold hundreds of thousands of fish in a single cage. One of the negative effects attributed to finfish culture is enrichment of the water column with dissolved nutrients, resulting from the decomposition of uneaten feed, and from metabolic wastes produced by the fish (Langan 2004). There is growing interest in marine aquaculture systems that combine fed aquaculture species (e.g. finfish), with inorganic extractive aquaculture species (e.g. seaweeds) and organic extractive species (e.g. suspension- and deposit-feeders) cultivated in proximity to mitigate these negative effects. One type of offshore aquaculture system that is expected to grow is longline mussel aquaculture. At a typical commercial mussel farm, multiple backbone lines are arrayed in parallel rows submerged several meters (5– 20m) below the surface using a system of anchors and buoys (Price et al. 2016). The longlines may be 150– 300m in length. Submerged floats keep the vertical lines running up from the anchors and the horizontal longlines properly oriented in the water column and prevent the lines from becoming entangled with each other. In many parts of the world, a single farm may include several hundred longlines covering hundreds of acres. Currently in the U.S., farms are typically being permitted at smaller scales (less than 100 acres), though it is anticipated that scaling up will follow once the domestic industry expands in the near future (Price et al. 2016). Aquaculture companies in Hawaii have also been experimenting with drifting, unanchored cages for open ocean fish production.

The growth of the aquaculture industry has drawn attention to the potential environmental impacts of offshore aquaculture, including impacts to protected species. Although aquaculture has the potential to relieve pressure on ocean fisheries, it can also threaten marine ecosystems through the introduction of exotic species and pathogens, effluent discharge, the use of wild fish to feed farmed fish, and habitat destruction. The potential escape of farmed fish either due to cage failure or operator error may also pose a threat to native fish populations due to the potential introduction of diseases, competition, and/or interbreeding depending on the species being farmed and those naturally occurring in area. Marine aquaculture operations have the potential to displace marine mammals from their foraging habitats or cause other disruptions to their behavior (Markowitz et al. 2004).

The large amount of fixed gear (e.g., nets, cages, lines, buoys) used for open water aquaculture could also represent an entanglement risk for some protected species. Entanglement in nets or lines around fish and mussel farms may cause injury, stress or death to marine mammals. It is generally thought that echolocating marine mammals (toothed whales, dolphins and porpoises) can effectively perceive mussel and fish farms and, in most cases, navigate through or around them (Llyod 2003; Markowitz et al. 2004). Species of baleen whales are not evolved to echolocate and rely on visual and audio queues, which may put them at higher risk of entanglement (Llyod 2003). Global reports of cetacean interactions with aquaculture gear include humpback whales is Australia, Canada and Iceland, Bryde's whales in New Zealand, right whales in South Korea, Argentina, and the North Atlantic Ocean (Price et al. 2016). There are three known incidents involving leatherback sea turtles being entangled in mussel ropes in Notre Dame Bay, Newfoundland from 2009 through 2013 (Price et al. 2016). One leatherback was documented entangled in shellfish aquaculture gear in the Greater Atlantic Region of the U.S. This animal was entangled in the vertical line associated with the anchoring system. We found no published reports on sharks being entangled in aquaculture gear, and there is little published information about the interactions of sharks and marine farms (Price et al. 2016). Despite these reported incidents of entanglement, a literature review conducted by Price et al. (2016) does not indicate significant impacts to marine mammals, sea turtles or ESA-listed fish species from marine aquaculture structures and activities. The authors note that it is unclear if this is because aquaculture is relatively benign and poses little risk, or because the number and density of farms is so low that the detection level for harmful interactions is also very small (Price et al. 2016). Because of the projected growth of aquaculture in the action area in both EEZ and state/territorial waters, as well as the presence of existing operations, there are likely to be impacts to ESA-listed fishes, marine mammals, and sea turtles related to the threats associated with aquaculture described in this section such as entanglement, release of nutrients, and escape of farmed species.

7.9 Commercial and Private Whale Watching

Studies investigating the behavioral responses of cetaceans to vessels suggest that individual whales experience stress responses to approaching vessels. While this type of stimulus is often stressful, the fitness consequences of this stress on individual whales remains unknown (Baker et al. 1983; Baker and Herman 1987). (Beale and Monaghan 2004) concluded that the significance of disturbance was a function of the distance of humans to the animals, the number of humans making the close approach, and the frequency of the approaches. These results would suggest that the cumulative effects of the various human activities in the action area would be greater than the effects of the individual activity.

(Baker et al. 1983) described two responses of whales to vessels: (1) horizontal avoidance of vessels 2,000 to 4,000 m away characterized by faster swimming and fewer long dives; and (2) vertical avoidance of vessels from 0 to 2,000 m away during which whales swam more slowly, but spent more time submerged. Watkins et al. (1981) found that both fin and humpback whales appeared to react to vessel approach by increasing swim speed, exhibiting a startled reaction, and moving away from the vessel with strong fluke motions. Results were different depending on the social status of the whales being observed (single males when compared with cows and calves), but humpback whales generally tried to avoid vessels when the vessels were 0.5 to 1.0 km from the whale. Smaller pods of whales and pods with calves seemed more responsive to approaching vessels (Bauer 1986; Bauer and Herman 1986). Bauer (1986) and Bauer and Herman (1986) noted changes in humpback whale respiration, diving, swimming speed, social exchanges, and other behavior correlated with the number, speed, direction, and proximity of vessels.

Studies of other baleen whales, specifically bowhead and gray whales, document similar patterns of behavioral disturbance in response to a variety of actual and simulated vessel activity and noise (Malme et al. 1983; Richardson et al. 1985). For example, studies of bowhead whales revealed that they orient themselves in relation to a vessel when the engine is on, and exhibit significant avoidance responses when the vessel's engine is turned on even at a distance of about 900 m (3,000 ft). Jahoda et al. (2003) studied the response of 25 fin whales in feeding areas in the Ligurian Sea to close approaches by inflatable vessels and to biopsy samples. They found that close vessel approaches caused the whales to stop feeding and swim away from the approaching vessel. The fin whales studied also tended to reduce the time they spent at the surface and increase their blow rates, suggesting an increase in metabolic rates that might indicate a stress response to the approach. Whales that had been disturbed while feeding remained disturbed for hours after the exposure ended. They recommended keeping vessels more than 200 m from whales and having approaching vessels move at low speeds to reduce visible reactions in these whales.

Although considered by many to be a non-consumptive use of marine mammals with economic, recreational, educational and scientific benefits, whale watching has the potential to harass whales by altering feeding, breeding, and social behavior or even injure them if the vessel gets too close and strikes a whale (New et al. 2015). Another concern is that preferred habitats may be abandoned if disturbance levels from whale watch boats are too high. In the Notice of Availability of Revised Whale Watch Guidelines for Vessel Operations in the Northeastern U.S. (64 FR 29270; June 1, 1999), NMFS noted that whale watch vessel operators seek out areas where whales concentrate, which has led to numbers of vessels congregating around groups of whales, increasing the potential for harassment, injury, or even the death of these animals. Several studies have specifically examined the effects of whale watching on marine mammals, and investigators have observed a variety of short-term responses from animals, ranging from no apparent response to changes in vocalizations, duration of time spent at the surface, swimming speed, swimming angle or direction, respiration rate, dive time, feeding behavior, and social behavior (NMFS 2006a). Responses appear to be dependent on factors such as vessel proximity, speed, and direction, as well as the number of vessels in the vicinity. Foote et al. (2004) found that southern resident killer whale call duration increased by 10-15 percent in the presence of whale watching boats, suggesting the whales compensate for a noisier environment. Disturbance by whale watch vessels has also been noted to cause newborn calves to separate briefly from their mothers' sides, which leads to greater energy expenditures by the calves (NMFS 2006a). Au and Green (2000) concluded that it is unlikely that the levels of sounds produced by whale watching boats in Hawaii would have any grave effects on the auditory system of humpback whales. Although numerous short-term behavioral responses to whale watching vessels are documented, little information is available on whether long-term negative effects result from this activity (NMFS 2006a; New et al. 2015).

By regulation, humpback whales cannot be approached closer than 100 yds (90 m) by vessels in Hawaiian waters (50 C.F.R. 224.103). The only exception to these approach restrictions is for researchers who hold a scientific research permit authorized by NMFS. For all other cetaceans and for monk seals the recommended distance for observation is 50 yds when the animal is on land or in the water. Other guidelines have been issued by NMFS to minimize the impacts of wildlife viewing on marine mammals, including maximum vessel speeds, proper vessel positioning, limiting noise levels, and the use of extra caution in the vicinity of mothers and their young.

In Hawaii, most of the whale watching industry is based around humpback whales which winter in the islands from mid-December to the end of April (Hoyt 2001). Maui is the primary location for boat-based whale watching, but whale watching operations located at most major harbors around the state. The whale watching industry in Hawaii contributes approximately $20 million in total revenues per year. In the Southern California portion of the action area, whale watching companies offer blue whale tours that leave from San Diego Bay from about mid-June through September. We have no information regarding the specific effects of whale watching operations within the action area. We anticipate that at least some short-term effects from whale watching, as described above, are affecting humpback and blue whales within the action area, although the regulations and mitigation measures in place likely reduce those effects to some extent.

7.10 Vessel Strike

Marine habitats occupied by ESA-listed species often feature both heavy commercial and recreational vessel traffic. Vessel strikes represent a recognized threat to several taxa of large air breathing marine vertebrates, including whales and sea turtles. The International Whaling Commission noted that human-induced mortality caused by vessel strikes can be an impediment to cetacean population growth (IWC 2017). Most whales killed by vessel strike likely end up sinking rather than washing up on shore. It is estimated that only 17 percent of vessel strikes of whales are actually detected (Kraus et al. 2005). Therefore, it is likely that the number of documented cetacean mortalities related to vessel strikes is much lower than the actual number of mortalities associated with vessel strikes.

Various types and sizes of vessels have been involved in ship strikes with large whales, including container/cargo ships/freighters, tankers, steamships, USCG vessels, military vessels, cruise ships, ferries, recreational vessels, research vessels, fishing vessels, whale-watching vessels, and other vessels (Jensen and Silber 2004). The majority of vessel strikes of large whales occur when vessels are traveling at speeds greater than approximately ten knots, with faster vessels, especially of large vessels (80 m or greater), being more likely to cause serious injury or death (Jensen and Silber 2004; Laist et al. 2001; Vanderlaan and Taggart 2007; Conn and Silber 2013). Injury is generally caused by the rotating propeller blades, but blunt injury from direct impact with the hull also occurs. Injuries to whales killed by vessel strikes include huge slashes, cuts, broken vertebrae, decapitation, and animals cut in half (Carillo and Ritter 2008). Measures to minimize the risk of ship strikes include re-routing shipping lanes, creating areas to be avoided, and vessel speed limits in areas where collisions are known to occur. From 2007 through May 2017, the Navy reported four whale strikes in the action area (an average of 0.39 per year), with the last strike occurring in 2012. For the 10-year period (1997-2006) prior to the implementation of the original Marine Species Awareness Training in 2007, the Navy reported 15 whale strikes during Navy activities (an average of 1.5 per year) in the action area, which is more than three times the amount reported for 2007-2017. It is likely that the implementation of the Marine Species Awareness Training starting in 2007, and the additional Navy Afloat Environmental Compliance Training Series modules starting in 2014, has contributed to this reduction in strikes.

The west coast of the U.S. has some of the heaviest ship traffic associated with some of the largest ports in the country, including Los Angeles/Long Beach, San Francisco, Seattle, and the Columbia River. Blue, fin, humpback, and gray whales are the most vulnerable species to ship strikes because they migrate along the coast and utilize coastal areas for feeding. In California, gray whales are the most common baleen whale hit by ships, followed (in order of occurrence) by fin, blue, humpback, and sperm whales (Heyning and Dahlheim 1990; NMFS 2011f). NMFS declared an Unusual Mortality Event (UME) on October 11, 2007, because of the number of blue whales (four) struck and killed by vessels during the fall of that year. The magnitude of this threat for large whales populations along the U.S. West Coast could be considerably larger than indicated based on reported incidents due to the unknown number of vessel strikes that go undocumented (NMFS 2011f). For example, Rockwood et al. (2017) estimated ship strike mortality of blue, fin, and humpback whales using an encounter theory model that considered whale density, vessel traffic characteristics, and whale movement patterns. Carretta et al. (2018) estimated that the vessel strike detection rate of blue whales is approximately one percent, fin whales is approximately 3.7 percent, and humpback whales is approximately 12 percent.

A summary of known mortalities and serious injuries related to vessel strikes of ESA-listed cetaceans within U.S. waters in recent years is shown in Table 9. These data represent only known mortalities and serious injuries; more, undocumented mortalities and serious injuries have likely occurred as commercial vessels are not required to report vessel strikes. In addition, these data do not include the recent deaths of North Atlantic right whales associated with the ongoing Unusual Mortality Event.

Table 9. Number of Reported Cetacean Vessel Strikes in U.S. Waters from 2011 to 2015 (2008-2012 for Sperm Whales; Hayes et al. 2017; Henry et al. 2017)

Species	Number of Vessel Strikes*	Annual Average
Blue whales	0	0
Fin whales	8	1.6
North Atlantic right whales	5	1
Sei whales	4	0.8
Sperm whales	1	0.2

Note: None of these strikes involved Navy vessels.

Impact from a boat hull or outboard motor, or cuts from a propeller can kill or severely injure turtles. Vessel strikes have been identified as one of the important mortality factors in several near shore turtle habitats worldwide (Denkinger et al. 2013). Many recovered turtles display injuries that appear to result from interactions with vessels and their associated propulsion systems (Work et al. 2010). Turtles may use auditory cues to react to approaching vessels rather than visual cues, making them more susceptible to strike as vessel speed increases (Hazel et al. 2007). Results from a study by Hazel et al. (2007) suggest that green turtles cannot consistently avoid being struck by vessels moving at relatively moderate speeds (i.e., greater than four kilometers per hour).

Vessel strikes were identified as a source of mortality for green sea turtles in Hawaii waters, although reported incidence rates among stranded turtles are not as high as in the southeastern U.S. Chaloupka et al. (2008) reported that 2.5 percent of green turtles found dead on Hawaiian beaches between 1982 and 2003 had been killed by vessel strike.

High levels of vessel traffic in nearshore areas along the U.S. Atlantic and Gulf of Mexico coasts result in frequent injury and mortality of sea turtles. From 1997 to 2005, nearly 15 percent of all stranded loggerheads in this region were documented as having sustained some type of propeller or collision injury although it is not known what proportion of these injuries were sustained ante-mortem versus post mortem. In one study from Virginia, Barco et al. (2016) found that all 15 dead loggerhead turtles encountered with signs of acute vessel interaction were apparently normal and healthy prior to human-induced mortality. The incidence of propeller wounds of stranded turtles from the U.S. Atlantic and Gulf of Mexico doubled from about ten percent in the late 1980s to about 20 percent in 2004. Singel et al. (2007) reported a tripling of boat strike injuries in Florida from the 1980's to 2005. Over this time period, in Florida alone over 4,000 (~500 live; ~3500 dead) sea turtle strandings were documented with propeller wounds, which represents 30 percent of all sea turtle strandings for the state (Singel et al. 2007). These studies suggest that the threat of vessel strikes to sea turtles may be increasing over time as vessel traffic continues to increase in the southeastern U.S. and throughout the world.

Sturgeon are susceptible to vessel strikes due to their large size and frequent use of coastal waterways with heavy commercial vessel traffic. The factors relevant to determining the risk to sturgeon from vessel strikes are currently unknown, but are likely related to size and speed of the vessels, navigational clearance (i.e., depth of water and draft of the vessel) in the area where the vessel is operating, and the behavior of sturgeon in the area (e.g., foraging, migrating, etc.). The Atlantic Sturgeon Species Recovery Team (ASSRT) determined Atlantic sturgeon in the Delaware River are at a moderately high risk of extinction because of ship strikes, and sturgeon in the James River are at a moderate risk from ship strikes (ASSRT 2007). Balazik et al. (2012) estimated up to 80 sturgeon were killed between 2007 and 2010 in these two river systems. Brown and Murphy (2010) examined 28 dead Atlantic sturgeon from the Delaware River from 2005 through 2008 and found that fifty percent of the mortalities resulted from apparent vessel strikes, and 71 percent of these (ten out of 14) had injuries consistent with being struck by a large vessel. Eight of the fourteen vessel-struck sturgeon were adult-sized fish which, given the time of year the fish were observed, were likely migrating through the river to or from the spawning grounds. Ship strikes may also be threatening Atlantic sturgeon populations in the Hudson River where large ships move from the river mouth to ports upstream through narrow shipping channels. The channels are dredged to the approximate depth of the ships, usually leaving less than six ft of clearance between the bottom of ships and the river bottom. Any aquatic life along the bottom is at risk of being sucked up through the large propellers of these ships.

Large Atlantic sturgeon are most often killed by ship strikes because their size means they are unable to pass through the ship's propellers without making contact. Shortnose sturgeon may not be as susceptible due to their smaller size in comparison to Atlantic sturgeon. There has been only one confirmed incidence of a ship strike on a shortnose sturgeon in the Kennebec River, and two suspected ship strike mortalities in the Delaware River (SSSRT 2010). Smalltooth sawfish may also been susceptible to ship strikes, but there is no available information on this threat to these species.

Commercial and recreational vessel traffic can adversely affect ESA-listed coral colonies and acroporid coral critical habitat through propeller scarring, propeller wash, and accidental groundings. Based on information from the NOAA Restoration Center (RC) and NOAA's ResponseLink, reports of accidental groundings are becoming more common in USVI and Puerto Rico, but numerous vessel groundings are likely not reported. Toller (2002) reported that approximately 13 percent of the Frederiksted Reef System along the west coast of St. Croix, Virgin Islands had been impacted by vessel anchoring, in particular in the area of Frederiksted. Smith et al. (2011) noted impacts to a northern portion of this reef from anchoring of vessels associated with dive and snorkel commercial operations, as well as private vessels. The proliferation of vessels throughout the action area, including within the range of ESA-listed Atlantic/Caribbean corals, means the likelihood of vessel groundings will increase for coral species found in shallow water such as elkhorn, staghorn, and pillar coral.

Propeller scarring and improper anchoring are known to adversely affect seagrasses (Sargent et al. 1995; Kenworthy et al. 2002). These activities can severely disrupt the benthic habitat by uprooting plants, severing rhizomes, destabilizing sediments, and significantly reducing the viability of the seagrass community. Indirect effects to corals and seagrass associated with motor vessels include turbidity from operating in shallow water, dock construction and maintenance, marina expansion, and maintenance dredging. These activities and impacts are likely to increase with predicted increases in boating activity throughout the action area and in South Florida specifically in the case of Johnson's seagrass. There are a number of local, state, and federal statutes to protect seagrass and corals from damage due to vessel impacts, and a number of conservation measures, including the designation of vessel control zones, signage, mooring fields, and public awareness campaigns, are directed at minimizing vessel damage to seagrass and corals. Despite these efforts, vessel damage can have significant local and small-scale (one m^2 to 100 m^2) impacts on seagrasses (Kirsch et al. 2005), but there is no direct evidence that these small-scale local effects are so widespread that they are a threat to the persistence and recovery of Johnson's seagrass.

7.11 Invasive Species

The introduction of non-native species is considered one of the primary threats to at-risk species, including ESA-listed species (Wilcove et al. 1998; Anttila et al. 1998; Pimentel et al. 2004). Clavero and García-Berthou (2005) found that invasive species were a contributing cause to over half of the extinct species in the IUCN database; and invasive species were the only cited cause in 20 percent of those cases. Invasive species consistently rank as one of the top threats to the world's oceans (Wambiji et al. 2007; Terdalkar et al. 2005; Raaymakers and Hilliard 2002; Raaymakers 2003; Pughiuc 2010).

When non-native plants and animals are introduced into habitats where they do not naturally occur, they can have significant impacts on ecosystems and native fauna and flora. Non-native aquatic species can be introduced through infested stock for aquaculture and fishery enhancement, ballast water discharge, and from the pet and recreational fishing industries. Non-native species can reduce native species abundance and distribution, and reduce local biodiversity by out-competing native species for food and habitat. They may also displace food items preferred by native predators, disrupting the natural food web.

An example of indirect predatory effects caused by an invasive species is the European green crab, which has invaded both the east and west coasts of the U.S., resulting in trophic scale effects to ecosystems in both regions (Grosholz and Ruiz 1996). Invasive plants can cause widespread habitat alteration, including native plant displacement, changes in benthic and pelagic animal communities, altered sediment deposition, altered sediment characteristics, and shifts in chemical processes such as nutrient cycling (Wigand et al. 1997; Grout et al. 1997; Ruiz et al. 1999). Introduced seaweeds alter habitat by colonizing previously unvegetated areas, while algae form extensive mats that exclude most native taxa, dramatically reducing habitat complexity and the ecosystem services provided by it (Wallentinus and Nyberg 2007). Invasive algae can alter native habitats through a variety of impacts including trapping sediment, reducing the number of suspended particles that reach the benthos for benthic suspension and deposit feeders, reducing light availability, and adverse impacts to foraging for a variety of animals (Sanchez and Pizarro 2005; Levi and Francour 2004; Britton-Simmons 2004; Gribsholt and Kristensen 2002).

Pathogens and species with toxic effects not only have direct effects on ESA-listed species, but also may affect essential critical habitat features or indirectly affect the species through ecosystem-mediated impacts. There are a number of non-native species that have the potential to either expel toxins at low levels, only becoming problematic for other members of the ecosystem if their population grows to very large sizes, resulting in the release of very large amounts of toxins. Non-native species can be introduced through infested stock for aquaculture and fishery enhancement, ballast water discharge, and from the pet and recreational fishing industries. In general, species located higher within a food web (including most ESA-listed species under NMFS jurisdiction) are more likely to become extinct as a result of an invasion; conversely, species that are more centrally or bottom-oriented within a food web are more likely to establish (Byrnes et al. 2007; Harvey and May 1997). Propagule pressure is generally the reason for this trend as individuals lower in the food web tend to have higher fecundity and lower survival rates (r-selection). This unbalancing of food webs makes subsequent introductions more likely as resource utilization shifts, increasing resource availability, and exploitation success by non-native species (Byrnes et al. 2007; Barko and Smart 1981). Such shifts in the base of food webs fundamentally alters predator-prey dynamics up and across food chains (Moncheva and Kamburska 2002).

Globally, 39 percent of marine NAS invasions were linked to hull fouling, whereas 31 percent of marine NAS invasions were likely to have been transported via ballast water (Molnar et al. 2008). Bax et al. (2003) identified hull fouling as the source of 36 percent of the nonnative coastal marine species established in continental North America (Molnar et al. 2008), whereas ballast water only accounted for 20 percent (Bax et al. 2003). Changes in climate are expected to increase invasion risk, particularly in the Arctic due to a combination of increasing vessel traffic and habitat conditions favoring temperate invading species rather than native species (Carlton 2001).

From 2007 to 2009, Sylvester et al. (2011) surveyed the hulls of 40 large commercial ships (including bulk, container, general cargo carriers, oil and chemical tankers, roll-on/roll-off cargo ships, and one cable layer) in the Canadian ports of Halifax (east coast) and Vancouver (west coast). The average length of these ships was 646 ft, the average sailing speed was 16 knots, and the average time since these ships were last painted was a little under two years. Biosurveys of the hulls showed the average number of species per ship was 33 and the average propagule pressure (i.e. individual organisms per ship) was around 2,860. Thus, even the hulls of active ships painted within the past few years that move at relatively fast speeds can carry a diverse biofouling community with several thousand organisms. Sylvester et al. (2011) found that propagule pressure was highly variable among ships, and increased with time spent in previous ports-of-call and time since last application of antifouling paint. Sylvester et al. (2011) reported high propagule pressures (as high as 600,000 individuals per ship) for commercial ships with particularly long stays in a given port and long periods between hull paintings.

Diver surveys conducted after in-water biofouling removal was conducted on the ex-Independence reported that 99 percent of the hull was free from biofouling organisms post-cleaning (NUWC 2017). Davidson et al. (2008) reported a significant reduction in organism cover on the underwater surfaces (i.e., propeller, rudder, propeller shafts, struts, transverse hull transects and sea chests) of the Orion, a Maritime Administration ship that was inactive for 13 years, post-cleaning. The mean percent area of exposed hull surface increased from 10.9 percent per sample prior to scrubbing to 62.7 percent after scrubbing. A large increase in bare space (a combination of 'hull surface' and 'organism scars' categories) was consistent across all depths of the hull and on the surfaces of the ship's running gear. This increase in bare space coincided clearly with reductions in encrusting species, barnacles, and filamentous biofouling (Davidson et al. 2008). These studies, comparing biofouling levels pre and post hull cleaning, indicate that this mitigation method is highly effective at reducing propagule pressure and, therefore, the likelihood of an NAS being introduced or becoming an ANS. Thus, in-water hull cleaning of inactive ships can significantly reduce but not entirely eliminate invasive species risk, as viable specimens from many species may still be found post-cleaning. While active vessels are more regularly cleaned and painted, these results, along with the results of Sylvester et al. (2011) from active commercial ships indicate that viable NAS may still be present post-cleaning. Anti-fouling coatings are more effective than cleaning for removal of NAS but the application of these coatings is less frequent than hull cleaning and anti-fouling components can cause other effects to ESA-listed species depending on the chemical constituents of the coatings, because these leach over time (e.g., metals, pesticides).

As of 2013 there were 54 documented marine invasive species in San Diego Bay including tunicates (nine species), amphipods (eight species), polychaetes (six species), moss animals (six species), mollusks (five species), and isopods (four species) (four species; Navy 2013). Several of these invasions have resulted in ecosystem level effects. The Japanese mussel (*Musculista senhousia*) has spread rapidly throughout Mission Bay and San Diego Bay, reaching densities up to 27,000 mussels/m^2 in the intertidal zone and up to 178,000 per m^2 carpeting the shallow subtidal bay bottom (Navy 2013). Research has shown that the effect of this species can be both negative and positive (Crooks 1998). While the mussel's dense mats can crowd out native clams and dominate marsh restoration sites, the mats also provide new habitat that supports greater species diversity and densities of native macrofauna than other areas. However, the mussel's dense beds can inhibit growth and vegetative propagation of native eelgrass (Reusch and Williams 1999;1998). Another invasive species in San Diego Bay producing ecosystem-level effects through habitat alteration is the isopod *Sphaeroma quoyanum* (Crooks 1997). High densities (greater than 10,000 per m^2) in some creeks that feed the bay have caused the overlying vegetated marsh flat to slump into the creek and the creek to widen. The ecosystem level changes produced by invasive species within San Diego Bay could potentially have detrimental impacts on ESA-listed green sea turtle habitat and prey, although no studies have specifically addressed this issue. Other bays along the west coast, such as San Francisco Bay, also contain reported invasive species such as zebra mussels that may affect ESA-listed species and their habitat.

There are a total of 333 non-native species, and another 130 cryptogenic species (i.e., unknown origin), documented as part of the marine and estuarine biota of the six largest Hawaiian islands from Kauai to Hawaii (Carlton and Eldredge 2015). The greatest proportion of non-native and cryptogenic species are found in the majors harbors of Oahu, which receive the large majority of all vessel traffic in the Hawaiian Islands (Coles and Eldredge 2002). Approximately 20 percent of the benthic algae, fishes, and macroinvertebrate species found these harbors are either non-native or cryptogenic. Algal species have become nuisance invaders of many Hawaiian reefs (Smith et al. 2002). With the exception of Kaneohe Bay, the largest embayment in Hawaii with a history of urban impact, few nonindigenous fishes or invertebrates have been detected on Hawaiian reefs (Coles and Eldredge 2002). ESA-listed sea turtles and Hawaiian monk seals could be impacted by invasive species in Hawaii, although there are no studies indicating this is occurring.

For example, invertebrates can have major impacts on the ecosystems they invade. Benthic invertebrates, such as mussels, polychaetes, and hydroids can become dominant filter feeders, greatly reducing the amount of organic energy that is available to native taxa in the water column (NMFS 2012c). This transfer of energy from the water column into the benthos fundamentally alters the ecology of the host habitat, resulting in less prey available for other filter feeders. Adverse effects of this include reduced body condition, growth, survival, and/or reproduction of native pelagic organisms at the same or similar trophic level as the invader if the native competitor cannot adapt to another food source. These changes would be manifested up the food chain to higher trophic level organisms in the habitat, including ESA-listed sturgeon and sea turtles (NMFS 2012c). Invasive species may also prey upon ESA-listed species. For example, the crown-of-thorns sea star *Acanthaster planci* can significantly disrupt localized coral reef ecosystems by feeding on live coral (e.g., Colgan 1987; Timmers et al. 2012), including the ESA-listed coral considered in this opinion.

Red tide dinoflagellates have been introduced via ballast water discharges and have the potential to undergo extreme seasonal population fluctuations, potentially resulting in significant adverse effects to ESA-listed species. During bloom conditions, high levels of neurotoxins are released into local and regional surface water and air that can cause illness and death in fishes, sea turtles, marine mammals, and invertebrates (as well as their larvae; McMinn et al. 1997; Lilly et al. 2002; Hallegraeff and Bolch 1992; Hamer et al. 2001; Hamer et al. 2000; Hallegraeff 1998). The brown alga, *Aureococcus anophagefferens*, causes brown tide when it blooms, causing diebacks of eelgrass habitat due to blooms decreasing light availability and failure of scallops and mussels to recruit (Doblin et al. 2004).

Invasive species can adversely affect listed fish species through several mechanisms, including: predation, competition, trophic structure alteration, introgression, and transfer of pathogens (Sanderson et al. 2009). Both positive and negative impacts to fish species have been reported in the literature from the introduction of nonindigenous species (Schlaepfer et al. 2011). For example, channel catfish, small and largemouth bass, and walleye prey on juvenile salmon (Sanderson et al. 2009). Juvenile shad prey heavily on zooplankton, which are also the primary prey for juvenile salmonids (Haskell et al. 2006). Alternatively, some introduced species may serve as a food source for native species in the introduced environment. Vinson and Baker (2008) found that the nonindigenous mudsnail (*Potamopyrgus antipodarum*) was an abundant prey item for native salmonids. However, when native salmonids feed exclusively on mudsnails, this study found they lose 0.5 percent of their body weight per day. This study suggests that, in some cases, even if nonindigenous invertebrate species can provide a new food source, the resulting effect can still be detrimental to native fish species if the nonindigenous prey is not as nutritionally valuable as the native prey items that it is replacing.

7.12 Diseases

Fibropapillomatosis is a neoplastic disease that can negatively affect ESA-listed sea turtle populations. Fibropapillomatosis has long been present in sea turtle populations with the earliest recorded mention from the late 1800s in the Florida Keys (Hargrove et al. 2016). Fibropapillomatosis has been reported in every species of marine turtle but is of greatest concern in green turtles, the only known species where this disease has reached a panzootic status (Williams Jr et al. 1994). Prevalence rates as high as 45 to 50 percent have been reported within some local green turtle populations (Hargrove et al. 2016; Jones et al. 2015). Fibropapillomatosis primarily affects medium-sized immature turtles in coastal foraging pastures.

Fibropapillomatosis is characterized by both internal and external tumorous growths, which can range in size from very small to extremely large. Large tumors can interfere with feeding and essential behaviors, and tumors on the eyes can cause permanent blindness (Foley et al. 2005). Renan de Deus Santos et al. (2017) assessed stress responses (corticosterone, glucose, lactate, and hematocrit) to capture and handling in green sea turtles with different fibropapillomatosis severity levels. Their findings suggest that moderate fibropapillomatosis severity may affect a turtle’s ability to adequately feed themselves (as evidenced by poor body condition), and advanced-stage fibropapillomatosis severity may result in an impaired corticosterone response. Expression of fibropapillomatosis differs across ocean basins and to some degree within basins. Despite some conflicting conclusions, the overwhelming consensus among turtle researchers is that, at present, fibropapillomatosis does not significantly impact the overall survival of sea turtle populations (Hargrove et al. 2016). In Hawaii, tumors have been reported on the internal organs of green sea turtles and oral tumors are common and often severe (Hargrove et al. 2016).

While fibropapillomatosis can result in reduced individual fitness and survival, documented mortality rates in Hawaii are low. The mortality impact of fibropapillomatosis is not currently exceeding population growth rates in some intensively monitored populations (e.g., Florida and Hawaii in the U.S., and the Southern Great Barrier Reef stock in Queensland, Australia) as evidenced by increasing nesting trends despite the incidence of fibropapillomatosis in immature foraging populations (Hargrove et al. 2016). However, fibropapillomatosis cannot be discounted as a potential threat to sea turtle populations (particularly green turtles) as the distribution, prevalence rate, severity, and environmental co-factors associated with the disease have the capacity to change over time (Jones et al. 2015).

Environmental factors likely play a role in the development of fibropapillomatosis. Most sites with a high frequency of fibropapillomatosis tumors are areas with some degree of water quality degradation resulting from altered watersheds (Hargrove et al. 2016). Despite there being a strong positive correlation between the prevalence of fibropapillomatosis in green turtle populations and areas with degraded water quality, it is difficult to identify one specific causal contaminant or a combination of such working synergistically to lead to fibropapillomatosis formation.

Infectious diseases and parasites are a threat to many cetacean populations worldwide. Cetacean morbilliviruses and papillomaviruses as well as *Brucella spp*. and *Toxoplasma gondii* are thought to interfere with population abundance by inducing high mortalities, lowering reproductive success, or by synergistically increasing the virulence of other diseases (Van Bressem et al. 1999). Genital papillomatosis has been observed in sperm whales from Iceland (Lambertsen et al. 1987). Jauniaux et al. (2000) reported evidence for morbillivirus infection in fin whales stranded on the Belgian and French coastlines.

Fish diseases and parasitic organisms occur naturally in the water. Many fish species are highly susceptible to parasites and disease, particularly during early life stages. Native fishes have co-evolved with such organisms and individuals can often carry diseases and parasites at less than lethal levels. While disease organisms commonly occur among wild fish populations, under favorable environmental conditions these organisms are not expected to cause population-threatening epizootics. However, outbreaks may occur when stress from disease and parasites is compounded by other stressors such as diminished water quality, flows, and crowding (Spence and Hughes 1996; Guillen 2003). At higher than normal water temperatures fish species may become stressed and lose their resistance to diseases (Spence and Hughes 1996). Consequently, diseased fish become more susceptible to predation and are less able to perform essential functions, such as feeding, swimming, and defending territories (McCullough 1999). The introduction of non-indigenous fish pathogens to wild fish populations through aquaculture operations also represents a threat to some fish populations. The aquarium industry is another possible source for transfer of non-indigenous pathogens or non-indigenous species from one geographic area to another, primarily through release of aquaria fish into public waters.

Cetaceans have evolved with a group of parasites belonging to the genus *Crassicauda* (order Spirurida; Lambertsen 1992). Infections with these nematodes are endemic in both the toothed and baleen whales. Such infections are a major cause of disease of the urinary, respiratory and digestive systems. Of several known crassicaudid infections, those caused by *Crassicauda boopis* are especially pathogenic. This giant worm infects blue whales, humpback whales, and fin whales (Lambertsen 1992). Anthropogenic environmental changes may increase the prevalence and severity of infectious illnesses and disease in cetaceans. A high prevalence of traumatic injuries or even minor skin lacerations from other stressors (e.g., vessel strike, fisheries interactions), in combination with a compromised immune system create ideal targets for opportunistic pathogens.

The potential population-level impact of infectious disease on Hawaiian monk seals could be severe given their critically endangered status, very low genetic diversity, and that this population has not been previously exposed to many diseases due to the isolation of the Hawaiian Archipelago (PIFSC 2018). Monk seals in the main Hawaiian Islands are often in close proximity to areas of human activity, domestic and feral animals, and agricultural areas, thus increasing the probability of infectious disease transmission. Infectious diseases that pose a risk to the monk seal population include distemper viruses, West Nile virus, *Leptospira spp*., and *Toxoplasma gondii* (PIFSC 2018). Risk factors for Hawaiian monk seals include cetaceans and non-native pinniped species that carry morbillivirus into Hawaiian waters and interactions between monk seals and infected dogs. Toxoplasmosis was first identified infecting a wild Hawaiian monk seal carcass examined in 2004 with disseminated disease and intra- and extracellular tachyzoites and tissue cysts in affected organs (Honnold et al. 2005). Barbieri et al. (2016) reported seven additional cases (eight total) and two suspect cases of protozoal-related mortality in Hawaiian monk seals between 2001 and 2015, including the first record of vertical transmission in this species. *Toxoplasma gondii* was the predominant apicomplexan parasite identified and was associated with 100 percent of confirmed protozoal-related mortalities (n = eight), and 50 percent of suspected cases (Barbieri et al. 2016).

Although the pathogen has not been associated with phocid mortality in the North Pacific to date, morbilliviruses have caused mass die offs of wild phocid populations in other parts of the world (PIFSC 2017). In 2016, NOAA developed the Hawaiian Monk Seal Vaccination Research and Response Plan to proactively address the threat of infectious diseases in this population, particularly for morbillivirus and West Nile virus infections. Studies of Guadalupe fur seals stranding off the coast of California have reported finding hemorrhagic gastroenteritis, nematodes, cestodes (Gerber et al. 1993), septicemia, and bacterial pneumonia (Hanni et al. 1997) in stranded animals.

There is also no information to indicate that disease is a factor affecting populations of scalloped hammerhead or oceanic whitetip sharks (Miller et al. 2014; Young 2018). Like most sharks, these species likely carry a range of external parasites including cestodes, nematodes, leeches, copepods, and amphipods but there are no studies suggesting parasites are negatively affecting the fitness or survival of these species (Miller et al. 2014; Young 2018). At least some oceanic whitetip sharks are infected with highly pathogenic *Vibrio harveyi*. This bacterium is known to cause deep dermal lesions, gastro-enteritis, eye lesions, infectious necrotizing enteritis, vasculitis, and skin ulcers in marine vertebrates (Austin and Zhang 2006). *Vibrio harveyi* is considered to be more serious in immunocompromised hosts, and therefore may act synergistically with the high pollutant loads that oceanic whitetip sharks potentially experience to create an increased threat to the species (Young 2018). However, there is no additional information available regarding the magnitude of impact these parasites may have on the health of oceanic whitetip populations (Young 2018).

Salmonids are susceptible to numerous bacterial, viral, and fungal diseases. The more common bacterial diseases in New England waters include furunculosis, bacterial kidney disease, enteric redmouth disease, coldwater disease, and vibriosis (USFWS and Gaston 1988; Egusa and Kothekar 1992; Olafesen and Roberts 1993). Furunculosis, which is particularly widespread, can be a significant source of mortality in wild Atlantic salmon populations if river water temperatures become unusually high for extended periods (USFWS and Gaston 1988). Whirling disease is a parasitic infection caused by the microscopic parasite *Myxobolus cerebrali*. Infected fish continually swim in circular motions and eventually expire from exhaustion. The disease occurs both in the wild salmonids and in hatcheries. Saprolegnia is a fungal disease of Atlantic salmon and is primarily found in adult males. It invades the epidermis and is associated with the presence of high levels of androsteroids (Olafesen and Roberts 1993; USFWS and Gaston 1988).

In 1996, the first occurrence of the infectious salmon anemia virus in North America was found in an aquaculture facility in New Brunswick, Canada (Fay et al. 2006). The first outbreak of infectious salmon anemia in the U.S. was reported in 2001 in an aquaculture facility in Cobscook Bay, Maine. Approximately 925,000 fish were removed from aquaculture pens throughout the Bay that year, and eventually all cultured salmon in the Bay had to be removed (Fay et al. 2006). While captive fish have the highest risk for transmission and outbreaks of diseases such as infectious salmon anemia, wild fish that must pass near aquaculture facilities are at risk of encountering both parasites and pathogens from hatchery operations. Although substantial progress has been made in recent years to reduce the risks to wild fish posed by aquaculture, this remains a potential threat.

Disease adversely affects various coral life history events by, among other processes, causing adult mortality, reducing sexual and asexual reproductive success, and impairing colony growth. A diseased state results from a complex interplay of factors including the cause or agent (e.g., pathogen, environmental toxicant), the host, and the environment. All coral disease impacts are presumed to be attributable to infectious diseases or to poorly-described genetic defects. Coral disease often produces acute tissue loss. Other forms of "disease" in the broader sense, such as temperature-caused bleaching, are discussed in other threat sections (e.g., ocean warming as a result of climate change).

Coral diseases are a common and significant threat affecting most or all coral species and regions to some degree, although the scientific understanding of individual disease causes in corals remains very poor. The incidence of coral disease appears to be expanding geographically, though the prevalence of disease is highly variable between sites and species. Increased prevalence and severity of diseases is correlated with increased water temperatures, which may correspond to increased virulence of pathogens, decreased resistance of hosts, or both.

Moreover, the expanding coral disease threat may result from opportunistic pathogens that become damaging only in situations where the host integrity is compromised by physiological stress or immune suppression. Overall, there is mounting evidence that warming temperatures and coral bleaching responses are linked (albeit with mixed correlations) with increased coral disease prevalence and mortality.

7.13 Scientific Research and Permits

Information obtained from scientific research is essential for understanding the status of ESA-listed species, obtaining specified critical biological information, and achieving species recovery goals. Research on ESA-listed species is granted an exemption to the ESA take prohibitions of section nine through the issuance of section 10(a)(1)(A) permits. Research activities authorized through scientific research permits can produce various stressors on wild and captive animals resulting from capture, handling, and research procedures. As required by regulation, The ESA requires that research conducted under a section 10(a)(1)(A) research permit cannot operate to the disadvantage of the species. Scientific research permits issued by NMFS are conditioned with mitigation measures to ensure that the impacts of research activities on target and non-target ESA-listed species are as minimal as possible. Section 10(a)(1)(A) permits are also issued to research facilities and educational display facilities for the captive research and educational display of ESA-listed species.

Over time, NMFS has issued dozens of permits on an annual basis for various forms of “take” of marine mammals, sea turtles, and ESA-listed fish species in the action area from a variety of research activities. Authorized research on ESA-listed marine mammals includes close vessel and aerial approaches, photographic identification, photogrammetry, biopsy sampling, tagging, ultrasound, exposure to acoustic activities, breath sampling, behavioral observations, passive acoustic recording, and underwater observation. Only non-lethal “takes” of marine mammals are authorized for research activities.

ESA-listed sea turtle research includes approach, capture, handling, restraint, tagging, biopsy, blood or tissue sampling, lavage, ultrasound, imaging, antibiotic (tetracycline) injections, laparoscopy, captive experiments, and mortality. On average, from 2007 to 2017 approximately 2,370 turtle (all species) takes were reported within the program in any given year. This includes an annual average of 831 sea turtles taken by capture with subsequent procedures, 157 sea turtles taken by conducting procedures only (i.e., capture authorized through different permit), and 1,382 sea turtles taken only during remote surveys. Most authorized take is sub-lethal. Mortality is rarely authorized by the NMFS Permits and Conservation Division in sea turtle research permits and no lethal take was authorized for sea turtle research in the Pacific Ocean basin from 2007-2017. In 2017, NMFS concluded section 7 consultation on a Program for the Issuance of Permits for Research and Enhancement Activities on Threatened and Endangered Sea Turtles Pursuant to Section 10(a) of the ESA (NMFS 2017c). This programmatic consultation allows for the authorization of up to the following number of sea turtle mortalities within the Pacific Ocean basin every ten years: nine green sea turtles (Central West Pacific, Central South Pacific, Central North Pacific, East Pacific DPSs combined); ten hawksbill; two leatherback; 12 loggerhead (North Pacific DPS); and eight olive ridely (NMFS 2017c). This programmatic consultation also includes an ITS that allows for one green sturgeon Southern DPS lethal take every ten years and one lethal take of each of the following ESA-listed fish species every ten years: Atlantic salmon, Atlantic sturgeon, shortnose sturgeon, gulf sturgeon, Nassau grouper, and scalloped hammerhead Eastern Pacific DPS.

The NMFS West Coast Region issues permits for scientific research on threatened salmon, steelhead, and green sturgeon species, DPSs, and ESUs. There are current permits for research work in California, Oregon, Washington, Idaho, and more specific locations within these states including the Willamette River, Lower Columbia River, and Oregon Coast, Snake River and Interior Columbia River, and Puget Sound and Washington Coast.

Since 2006, conservative mitigation measures implemented by NMFS through permit conditions (e.g., reduced soak times at warmer temperatures or lower DO concentrations, minimal holding or handling time) and additional precautions taken by sturgeon researchers have significantly reduced the lethal and sublethal effects of capture in gill, trammel and trawl nets on Atlantic and shortnose sturgeon. From 2006 through 2016, researchers reported only two shortnose sturgeon killed by capture gear out of 7,019 captured, for a capture mortality rate of 0.03 percent. Since they were listed in 2012, the mortality rate associated with Atlantic sturgeon capture in scientific research is 0.22 percent (14 killed out of 6,466 captured). In 2017, the NMFS Permits and Conservation Division implemented a program for the issuance of permits for research and enhancement activities on Atlantic sturgeon and shortnose sturgeon. A section 7 programmatic opinion determined that this action would not likely jeopardize the continued existence of ESA-listed species and would not likely result in the destruction or adverse modification of critical habitat. In addition to the required mitigation measures designed to reduce lethal take every ten years, and sub-lethal effects on sturgeon, the program establishes annual limits on sturgeon mortality resulting from research activities by subpopulation (i.e., spawning stock) and life stage. Relative mortality limits are calculated as a proportion of the estimated population size and are based on the relative health of the population. A health index is calculated by NMFS based on the best available information on the population including abundance, population trends, known threats, and information on spawning activity. For adults/sub-adults and juveniles, relative annual maximum mortality limits are set at 0.4, 0.6, and 0.8 percent of the estimated population size for sturgeon populations with a health index rating of “low,” “medium,” and “high,” respectively. For populations where there is insufficient information to calculate a health index or there is no estimate of population size, the default maximum mortality limit is conservatively established at one fish per year. Maximum annual mortality limits can be exceeded in any given year by up to two times, as long as the five-year moving average is within the established maximum annual mortality limit for that population and life stage.

There are currently three permits issued for research on smalltooth sawfish. The NMFS Permits Division and Interagency Cooperation Division are currently working on a programmatic consultation for the issuance of permits for research and enhancement activities on the U.S. DPS of smalltooth sawfish. Since their listing in 2003, only one smalltooth sawfish mortality has been reported as a result of research authorized under a section 10(a)(1)(A) permit. As with turtles and sturgeon, mitigation measures implemented by NMFS through permit conditions and additional precautions taken by researchers have significantly reduced the lethal and sublethal effects of research activities on smalltooth sawfish.

The USFWS issues section 10(a)(1)(A) permits for Atlantic salmon. For gulf sturgeon, a special rule promulgated at the time of listing (56 FR 49658) gives the states permitting authority to allow taking of this species, in accordance with applicable state laws, for educational purposes, scientific purposes, and enhancement of propagation.

7.14 The Impact of the Environmental Baseline on ESA-Listed Resources

Collectively, the stressors described above in Section 7 have had, and likely continue to have, lasting impacts on the ESA-listed resources considered in this consultation. Some of these stressors result in mortality or serious injury to individual animals (e.g., vessel strike, whaling, entanglement in fishing gear), whereas others result in more indirect (e.g., a fishery that impacts prey availability) or non-lethal impacts (e.g., whale watching). Assessing the aggregate impacts of these stressors on species is difficult and, to our knowledge, no such analysis exists. This becomes even more difficult considering that many of the species in this opinion are wide ranging and subject to stressors in locations throughout the action area and outside the action area.

We consider the best indicator of the aggregate impact of the Environmental Baseline on ESA-listed resources to be the status and trends of those species. As noted in Section 6.2, some of the species considered in this consultation are experiencing increases in population abundance, some are declining, and for others, their status remains unknown. Taken together, this indicates that the Environmental Baseline is impacting species in different ways. The species experiencing increasing population abundances are doing so despite the potential negative impacts of the stressors discussed in the *Environmental Baseline*. Therefore, while the *Environmental Baseline* may slow recovery, their recovery is not being prevented. For the species that may be declining in abundance, it is possible that the suite of conditions described in the *Environmental Baseline* is preventing their recovery. However, it is also possible that their populations are at such low levels (e.g., due to historic commercial whaling or the effects of climate change and disease on some coral species) that even when the species' primary threats are removed, the species may not be able to achieve recovery. At small population sizes, species may experience phenomena such as demographic stochasticity, inbreeding depression, and Allee effects, among others, that cause their limited population size to become a threat in and of itself. A thorough review of the status and trends of each species likely to be adversely affected by the action is presented in the *Status of Species and Critical Habitat Analyzed Further* (Section 6.2) of this opinion.

8. Effects of the Action

"Effects of the action" has been recently revised to mean: all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action. A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur. Effects of the action may occur later in time and may include consequences occurring outside the immediate area involved in the action (see 50 C.F.R. §402.17) 50 C.F.R. §402.02.

This effects analyses section is organized following the stressor, exposure, response, risk assessment framework.

The effects of this action are due to exposure to stressors in the discharges that are consistent with the performance standards of the UNDS. Stressors with indirect effects change habitats or biological communities. The NAS, nutrients, and oxygen demanding substances regulated under the UNDS are indirect stressors because they can create adverse conditions, including alteration of biological communities, eutrophy, and hypoxia. Proximate stressors, those stressors that an organism directly interacts with, can include depleted oxygen, reduced photosynthetic active radiation, predation by/competition with non-native species, and infection by pathogens and parasites. Stressors with indirect effects are assessed applying principles of ecology and environmental chemistry because exposure to the proximate stressors they generate rely on a multitude of other contributing conditions. Accordingly, assessments for stressors with indirect effects include a great deal of uncertainty.

The UNDS also provides performance standards for discharges of uncharacterized mixtures: personal care products in gray water and total petroleum hydrocarbons. Evaluation of uncharacterized mixtures also requires a qualitative assessment as described above, with the attendant uncertainties. However, evaluations of worst-case scenarios for potential high-risk constituents can be informative.

8.1 Uncertainty in Analyses

Stressors with direct effects include toxicants and stressors that cause physical harm, like sediment. For stressors that cause direct effects, exposure intensity determines the severity of effects. Assessments of direct stressors typically compare exposure intensity to concentrations at or below which effects are not likely to occur (i.e., response thresholds). The amount of uncertainty accompanying assessments for stressors with direct effects is largely determined by the quality of the information used and assumptions that needed to be applied to the analysis.

The BE includes a section on uncertainties in its modeling and analysis (Section 5.6, Appendix A) which will not be repeated in this opinion. It is important to note that, as stated by EPA and Navy USEPA and Navy (2018), the analysis in the BE addressed all discharges but may not have addressed all pollutants in these discharges that may affect ESA-listed species under NMFS jurisdiction. This is because ALCs are not available for all constituents that could be present in vessel discharges. The following sections describe the analysis in the BE and include NMFS perspectives and supplemental information and analyses.

Exposure Analysis

Ideally exposure estimates integrate the discharge rate of stressors, dissolution in the environment, and scenarios and concentrations under which organisms would be exposed. This is complicated for UNDS by the diverse and changing populations of vessels discharging within a given RAA and the intent for the modeling to represent exposures in all marine waters where vessels of the armed forces operate. For this reason, the BE used a Level I Screening Model that assumes instantaneous and homogeneous mixing of vessel discharges within the combined waters three miles around facilities and/or bases with homeported vessels of the armed forces in an RAA to estimate pollutant concentrations to which ecological receptors will be chronically exposed based on the comments received from the Navy in response to our draft biological opinion (July 25, 2019). The exception is Pearl Harbor where there is very little area that is not within three miles of a Navy facility with vessel traffic so the entire harbor was included in the calculation. In terms of the calculation, the mass loading for each constituent included the combined mass loading from all vessels that generate the discharge inside the RAA, including those discharges that would occur while operating outside the waters immediately around the facility/base. That mass loading was then mixed into the combined volume of waters within three miles of the facility/base and the harbor-specific mixing characteristics were applied to generate predicted average concentrations in waters near the facilities. The model then selected the highest concentration across all harbors for that constituent and, to be conservative, this high concentration was used to analyze across all of the representative species and critical habitat in all RAAs. The model did not account for incorporation of pollutants into the sediment.

The model does not account for intermittent discharges or gradients of concentrations that would occur within a certain distance from discharge source(s), such as plumes from vessels and other sources or for water bodies with greater residence times where mixing does not occur instantly. For freshwater harbors, exposure concentrations were based on the average annual river flow rate in a basic dilution model. Both models produced estimated worst–case pollutant concentrations for the RAAs, many below analytical detection limits (Gonzalez-Alvarez et al. 2018; Michel and Averty 1991; Paiga et al. 2019; Si et al. 2016; Søndergaard et al. 2015).

The BE acknowledges the risk from exposure to discharge-related pollutants in sediment and pore water as a gap in the assessment and that pollutant behavior is complex and dependent on factors that can fluctuate widely (see the BE, Section 5.6.7, Appendix A). While NMFS acknowledges the uncertainty associated with estimating concentrations of contaminants attributable to UNDS Batch Two discharges, risk exists from sediment exposures, which cannot be determined based on modeled surface water discharges. Level I Screening Modeling results cannot be used to infer the potential for pollutant accumulation in underlying sediments. This modeling assumes homogenous mixing within a portion of the RAA (the area within three miles of facilities for vessels of the armed forces). The approach also assumed maximum biological availability of pollutants and no complexation with water column constituents such as organic acids and organic matter and subsequent precipitation into the sediment.

Climate change has the potential to exacerbate sediment pollutant turnover. Increases in storm frequency and severity under climate change will increase disturbance and redistribution of sediment and constituent pollutants (Fischer Kuh 2012). Ocean acidity will influence metal speciation and partitioning from sediment. In addition, where climate change results in lowered water levels in freshwater, exposure and reinundation of contaminated sediments can create conditions favorable to metal contaminants partitioning into the water column (Lin et al. 2017; Nedrich and Burton 2017). UNDS Batch Two performance standards are expected to reduce the rate at which metals will be discharged to the water column and by extension the rate at which metals partition to sediment, when applicable. However, even low concentrations of metals in discharges could lead to metals being more available depending on changes in acidity. These changes will also be influenced by local physical conditions.

Response Analysis

Ideally, toxicity data is of high enough quality to calculate a minimum response threshold, such as an EC01, at which effects might be considered insignificant. This is rarely the case. Stressor response relationships are typically described by endpoints reporting:

- The concentration at which half of the exposed organisms die (LC50)
- The lowest test exposure concentration at which a given effect did not differ from controls (no observed effects concentration, NOEC)
- The lowest test exposure concentration at which the effect differed significantly from controls (lowest observed effects concentration, LOEC)

In its assessment, the BE identified chronic toxicity effects thresholds (CTET) for each pollutant contained in the Batch Two discharges. The CTETs were typically the geometric mean of the lowest reported LOECs and NOECs for each pollutant. This includes body tissue concentration thresholds at which adverse effects were reported. In some cases, a CTET was calculated using an acute-to-chronic ratio adjustment based on an LC50 or was the recommended chronic ALC from EPA's water quality guidelines (Stephen et al. 1985). Our discussion in Section 5 explained that the use of ALCs to determine which pollutants to select for further analysis in the BE is not expected to protect all species under all circumstances. There may be other pollutants that would adversely affect listed species and designated critical habitats that do not have ALCs.

The BE applies a risk quotient (RQ) approach to determine whether a response may occur. The RQ is calculated by dividing the modeled exposure concentration in water or estimated whole body tissue pollutant residue resulting from in-water estimates by the CTET. The BE characterized risk by applying a sliding scale to the RQs to determine overall ecological health concerns from any given pollutant. An RQ of less than one indicates that the risk to that particular species is either "remote" ($RQ < 0.1$, one order of magnitude difference) or "negligible" ($0.1 \leq RQ < 1.0$). The BE considered an RQ equal to or greater than one, but less than ten, as indicating a "potentially significant" risk and an RQ equal to or greater than ten as indicating a "likely significant" risk.

The BE effect determination matrix prescribed likely to adversely affect (LAA) determinations only for those aspects of the UNDS expected to increase stressor exposures and having risk characterized as potentially or likely significant (Table 10). This means that UNDS standards that decrease discharges but will still result in exposures characterized as having potential or likely significant risk are determined to be NLAA. This also means that any discharges that increase pollutant loading would be determined to be NLAA regardless of pollutant status of the receiving water and with no regard to accumulation or persistence in the environment over the indeterminate lifetime of the rule.

Table 10. The Effect Determination Matrix from the BE

Effect of the Action/Risk	Remote	Negligible	Potentially Significant	Likely Significant
None	No Effect	No Effect	No Effect	No Effect
Decrease Exposure to Stressors	NLAA	NLAA	NLAA	NLAA
Increase Exposure to Stressors	NLAA	NLAA	LAA	LAA

This approach is not consistent with the ESA section 7 consultation requirements. A determination that an action is NLAA requires that the effects are either beneficial with no adverse effects to species, are discountable because they are extremely unlikely to occur, or are insignificant in size or severity such that effects to the species are undetectable, not measurable, or so minor that they cannot be meaningfully evaluated (USFWS and NMFS 1998).

Where the available data show that exposure intensities are orders of magnitude lower (i.e., by 100-fold or more) than marine species' lowest reported response thresholds, it is reasonable to expect effects are extremely unlikely to occur in ESA-listed species. Such effects are discountable and a determination of NLAA is appropriate. Otherwise, an RQ approach only screens exposures against response thresholds. The following response threshold properties need to be considered when evaluating RQs:

- The magnitude of response represented by the response threshold
- The ecological significance of the effect
- The suitability and quality of the study
- The suitability of the surrogate species

For example, a 20 percent response rate for reductions in growth or reproductive success is not likely to be acceptable for imperiled species and an RQ of 0.95 could still result in an LAA determination.

8.2 Mitigation to Minimize or Avoid Exposure

The UNDS rule is intended to enhance the operational flexibility of vessels of the armed forces domestically and internationally, stimulate the development of innovative vessel pollution control technology and practices, and advance the ability of the armed forces to better design and build environmentally sound vessels and manage discharges. The UNDS establishes performance standards that minimize or avoid discharges of pollutants into waters of the U.S. and by extension concentrations of these contaminants in sediments by vessels of the armed forces. These performance standards are discussed in Section 3, Description of the Action. Our purpose in this opinion is to evaluate the UNDS to determine whether these performance standards may result in environmental conditions that result in adverse effects to ESA-listed species and designated critical habitat.

8.3 Exposure and Response Analyses

NMFS feels it is not productive to conduct an independent exposure and response analysis because the BE’s maximum exposure estimates for pollutant concentrations in the RAAs are many orders of magnitude lower than the CTET response thresholds (summarized in Tables 11 and 12) and thresholds identified by NMFS in the open literature and the ECOTOX. This is also true of the BE’s estimated body burden data (summarized in Tables 13 and 14). However, one of the main reasons for these estimates being so low is because the use of a Level I Screening approach provides an estimate reflecting instant dilution into a volume of the harbor within a three-mile area around facilities with vessels of the armed forces (based on information provided by the Navy in response to their review of the draft biological opinion, July 25, 2019), as has been discussed previously in this opinion. This was done to allow extrapolation from the RAAs to other harbors. Given the variability among vessel discharges and behavior, the pollutant concentration estimates for discharges from vessels in each RAA have very broad confidence intervals. In addition, based on information provided by the Navy during a meeting May 8, 2019, the discharge estimates evaluated in the BE are based on an estimate of what may be discharged in port while underway and are typically high because performance standards would limit discharges of graywater and/or bilge water (BE Appendix F; Appendix B this opinion).

Table 11. Response Thresholds for Estuarine and Marine Species Relative to Maximum Modeled Harbor Concentrations Based on Level I Screening Modeling (ug/L; adapted from USEPA and Navy 2018)

Pollutant	Aquatic Life Criteria	Plant Response Threshold	Invertebrate Response Threshold	Vertebrate Response Threshold	Maximum Modeled Harbor Concentration
Cadmium	7.9	9.3	7.009	8.182	0.0000078
Chromium VI		99,300	17.56	921.4	0.00039
Copper	--	49.8	7.9	206.7	0.3
Iron	--	--	141.4	2.69	0.038
Lead	--	8.1	23.85	11.43	0.00082
Mercury	--	2.2	1.89	5	0.00000046
Nickel	--	8.2	21.87	4227	0.0097
Silver	--	0.3	0.31	0.5	0.0000028
Zinc	--	9460	157.5	60.41	0.41
Bis(2-ethylhexyl) phthalate	--	--	111	100	0.014
Tributyl Tin	--	0.06	0.01	0.12	0.00021
CPO (saltwater)	--	7.5	21.81	46.48	0.0037
Oil and Grease	140	1000000	0.24	1.54	0.074
TPH	5.2	--	--	--	0.0000019
Total Phosphorus	5.4	--	--	--	0.0025
Ammonia Nitrogen	--	153.3	2000	2446	0.036
Nitrate/Nitrite	2	--	--	--	0.0011
Total Kjeldahl Nitrogen	--	--	--	--	0.049
Total Nitrogen	50	--	--	--	0.05

Table 12. Response Thresholds for Freshwater Vertebrates Relative to Modeled River Concentrations (from USEPA and Navy 2018)

Pollutant	Freshwater Aquatic Life Criteria	Freshwater Vertebrate CTET	Mean Modeled River Concentration
Cadmium	0.72	0.7962	0.0000000032
Chromium VI		30.11	0.00000016
Copper	--	4.69	0.000067
Iron	--	320	0.0000023
Lead	--	93.62	0.0000003
Mercury	--	0.23	0.0000000000093
Nickel	--	158	0.0000001
Zinc	--	71.98	0.000027
Bis(2-ethylhexyl) phthalate	--	2	0.000000028
Oil and Grease	140	--	0.000028
TPH	5.2	--	0.0000000027
Total Phosphorus	5	--	0.00000032
Ammonia Nitrogen	--	1349	0.000000016
Nitrate/Nitrite	350	--	0.000000048
Total Kjeldahl Nitrogen	100	--	0.00000027
Total Nitrogen	10	--	0.00000032

Table 13. Response Thresholds for Tissue Residues in Marine and Estuarine Species Relative to Estimated Tissue Residues (mg/kg wet weight; adapted from USEPA and Navy 2018)

Pollutant	Invertebrate Residue CTET	Estimated Invertebrate Residue	Vertebrate Residue CTET	Estimated Vertebrate Residue
Cadmium	0.14	0.0000002964	0.04	0.0000028548
Chromium VI	3.2	0.00006201	0.263	0.0000156
Copper	3.9	0.087	4.59	0.087
Iron	68	0.681758	--	0.003572
Lead	260	0.00004018	0.4	0.0001271
Mercury	1.64	0.00000007728	0.1	0.00000025438
Nickel	26.4	0.0004559	--	0.0004559
Silver	0.033	0.0000002996	0.12	0.000000224
Zinc	24	0.01927	--	0.01927
Bis(2-ethylhexyl) phthalate	0.5	0.0378	--	0.01176

Tributyl Tin	0.013	0.00009093	0.27	0.00009093

Table 14. Response Thresholds for Tissue Residues in Freshwater Species Relative to Estimated Tissue Residues (adapted from USEPA and Navy 2018)

Pollutant	Freshwater Vertebrate Tissue CTET	Estimated Freshwater Vertebrate Residue
Cadmium	0.005	0.0000000011712
Chromium VI	1.28	0.0000000064
Copper	3.	0.00001943
Iron	9.	0.0000002162
Lead	0.4	0.0000000465
Mercury	0.048	0.0000000000051429
Nickel	--	0.0000000047
Silver	0.044	0
Zinc	7.7	0.000001269
Bis(2-ethylhexyl) phthalate	0.66	0.00000002352
Tributyl Tin	0.27	0

The BE appendices indicate graywater discharge estimates were calculated using flow weighted averages for graywater constituents, average crew size, and average number of transits by a vessel, but there is no indication of the variability around those estimates, or the number and types of vessels contributing data to those estimates. We do know that the estimates are based on 12 potable water sink samples, 13 galley drain samples, eight sink samples, and 11 scullery samples (USEPA 1999a). It is uncertain whether replicate samples from the same point sources or samples from different vessels or vessel types were used to create these estimates.

Even if the estimate of graywater constituents is a conservative "worst case scenario" estimate, the assumption of complete instantaneous dissolution in an RAA harbor negates the impact of potentially significant amounts of copper in a graywater plume. The flow weighted average estimate of 936 micrograms per liter is greater than 390 times EPA's acute marine water quality guideline of 2.4 micrograms per liter. Meanwhile, EPA's Nature of Discharge report for graywater identifies the highest concentration of copper in vessel graywater as 3,404 micrograms per liter, but does not indicate the range, variance, or total number of vessels used in arriving at this flow-weighted average (USEPA 1999a). A maximum value reported at 3,404 micrograms per liter suggests an extremely broad range of plausible exposure concentrations unless that value is a single outlier or extreme value. On the other hand, there is no laundry, cooking or cleaning dishes, and showering is unlikely while a vessel is underway in port so graywater discharge would be minimal, if it occurs. The Navy also provided information indicating that graywater is only discharged while a vessel is underway and there is expected to be immediate mixing in the vessel's wake as the discharge is displaced with the estimated wake mixing factor being 282,000:1. This means that even if the maximum concentration of 3,404 micrograms per liter were discharged, it would be diluted rapidly in the vessel's wake.

The breadth of uncertainty increases as the discharges are extended to estimate RAA harbor concentrations, and again when these estimates are extrapolated to other areas. The models do not reflect realistic exposure scenarios: spatially and temporally variable pulses, plumes, and gradients. Exposure intensities are also expected to be higher within lanes of travel and with increasing proximity to ports where these vessels aggregate. The modeling also cannot inform an evaluation of the long-term implications of sediment as a pollutant sink and source over the indeterminate period over which the UNDS rule will be in effect. The BE discussion of uncertainty in estimating exposure concentrations (Section 5.6.2, Appendix A) acknowledges that Level I Screening Models do not take into account background pollutant levels and that listed species present may be directly exposed to a vessel discharge plume. Some exposures certainly will occur near the point of discharge over the indeterminate period over which the UNDS rule is in effect. To be consistent with ESA section 7 requirements, we cannot disregard the impacts of exposures that cannot be reliably quantified.

For example, NMFS designation of nearshore critical habitat for ESA-listed Puget Sound/Georgia Basin yelloweye rockfish, rated the Carr Inlet Naval Restricted Area as having a high Uniqueness and Conservation Role for the species (79 FR 68041). Juvenile settlement habitats located in the nearshore with substrates such as sand, rock, and/or cobble compositions that also support kelp are essential for conservation. These features enable forage opportunities and refuge from predators and enable behavioral and physiological changes needed for juveniles to occupy deeper adult habitats. While some waters where DoD activities occur are excluded from the critical habitat designation, specifically DoD facilities that are subject to an Integrated Natural Resources Management Plan (INRMP), take of individuals is not excluded from section 7 consultation. Suitable rearing habitat for the species occur in these waters, so juvenile fishes settling in Carr Inlet waters are expected to be exposed to vessel discharges at higher intensities than organisms in more open areas of Puget Sound. Each of the facilities had INRMPs that contain measures that provide benefits to Puget Sound/Georgia Basin yelloweye rockfish. Examples of the types of beneficial measures incorporated in INRMPs include: implementing actions to protect water quality from land-based infrastructure and vessels; conducting in-water actions during appropriate time periods; and initiating surveys for listed fish. Taken with the UNDS performance standards, these measures may assist in reducing the exposure of ESA-listed species, including fish, in certain RAAs.

Qualitative assessments were also provided in the BE for stressors with indirect effects (i.e., NAS and oxygen demanding substances) and stressors occurring in uncharacterized mixtures (i.e., TPH and Personal Care Products). Nutrients, some of which may be in vessel discharges such as graywater, produce indirect stressors associated with eutrophication. The possibility for this outcome was integrated into the approved and proposed state water quality criteria for phosphorous and nitrogen, including the various nitrogen species. The BE used these nitrogen and phosphorus criteria as response thresholds in a quantitative analysis of nutrients in some discharges.

While the performance standards under the UNDS strive to minimize pollutant loading, the rule does not eliminate loading. The risk of that loading is determined by location-specific factors, with port sediments acting as both a sink and source for UNDS pollutants where armed forces vessels aggregate. The BE acknowledges this as a gap in its analysis and describes the factors that make quantifying such impacts untenable. Regardless of the ability to model and quantify impacts, we must acknowledge that persistent pollutants in those discharges that are limited, but not prohibited, under the rule will accumulate over time in the sediment within any port where vessels of the armed forces operate, and this will radiate into the surrounding harbor system. Resulting effects would occur later in time, but NMFS expects that effects are reasonably certain to occur due to the indeterminate duration of discharges under the rule. It is the actual implementation of the rule, and its control measures that will reduce the impact of these effects on ESA-listed species and designated critical habitat

Given concerns with the modeling approach, gaps in the BE analysis, and the available information on discharges, arriving at exposure estimates to fill those gaps or recalculate exposure scenarios is not a pragmatic approach to assessing risk posed by the UNDS Rule. The assessment in this opinion is necessarily qualitative for all pollutants.

The following sections review the stressors evaluated in the BE, and include supplemental information and NMFS perspective on exposure and response for ESA-listed Southern Resident killer whales and their designated critical habitat, Northwest Atlantic loggerhead sea turtles, ESA-listed Atlantic/Caribbean corals and elkhorn and staghorn coral critical habitat, Johnson's seagrass and its designated critical habitat; and the following ESA-listed fish species and designated critical habitat: Puget Sound/Georgia Basin bocaccio; Puget Sound/Georgia Basin yelloweye rockfish; Central Valley Spring-Run, Puget Sound, and Sacramento River Winter-Run chinook salmon; Hood Canal Summer-Run chum salmon; Central-California Coast coho salmon; California Central Valley, Central California Coast, and Puget Sound steelhead trout; Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic sturgeon; green sturgeon; gulf sturgeon; shortnose sturgeon; and Gulf of Maine Atlantic salmon. We determined these species and designated critical habitats are likely to be adversely affected by discharges from military vessels in the UNDS Batch Two rule because these species and habitats are within RAAs and other port and harbor areas with concentrations of military vessels (see Table 1) and therefore are likely to be exposed to chronic effects of discharges while the UNDS rule is in effect.

We reiterate that, while we did not conduct an independent exposure and response analysis from that included in the BE, we do include supplemental information and our conclusions regarding exposure and response of ESA-listed species and designated critical habitats to UNDS Batch Two discharges, which differ from the final effects determinations in the BE. The rule will authorize discharges containing persistent and elemental pollutants over an indeterminate time. Therefore, the following sections discuss the analysis included in the BE and NMFS assessment of this analysis, including why we believe there are likely to be adverse effects to ESA-listed species and designated critical habitat from constituents in the UNDS Batch Two discharges.

8.3.1 Aquatic Nuisance Species

Ballast water and hull fouling are the primary vectors for NAS introduction to coastal and marine environments (Olenin et al. 2007). Globally, hull fouling and ballast water have been identified as the source of 228 (57 percent) marine NAS invasions (Ruiz et al. 2000). In North America, these vectors were identified as the source for 164 of 316 (52 percent) marine invertebrate and algal invasions (Fofonoff et al. 2003). Biofouling is likely a greater threat than ballast water.

The BE exposure and response analysis applied probabilities over the step-by-step process by which NAS become ANS integrating likelihood of exposure and magnitude of effect (Figure 39).

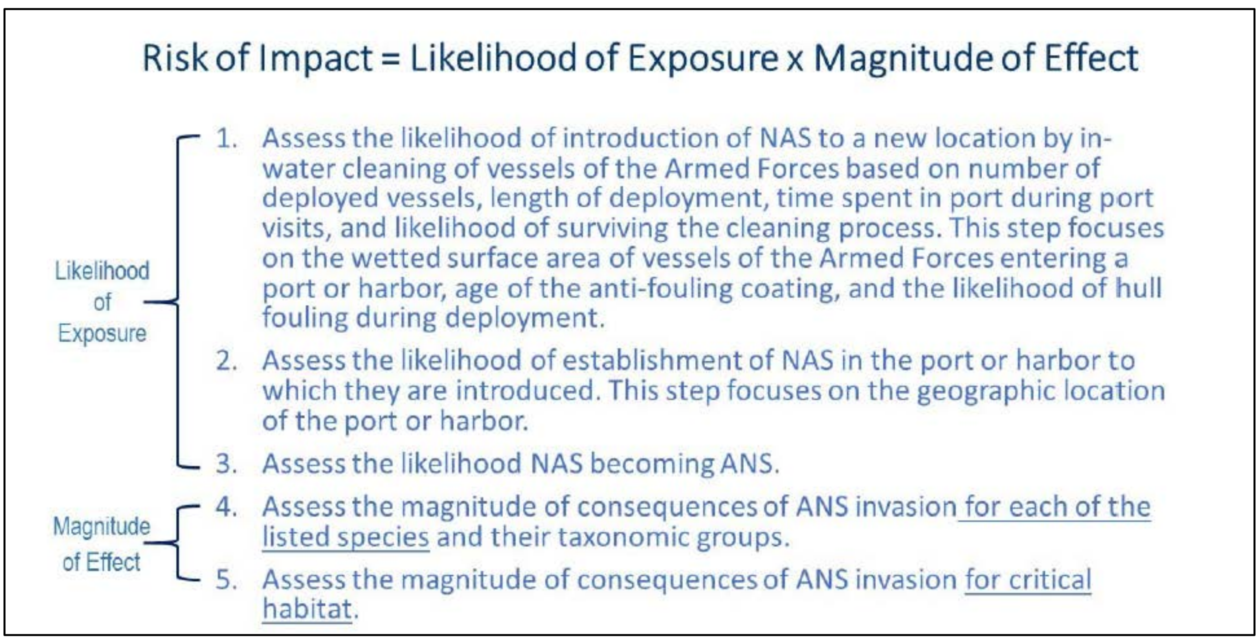

Figure 39. Risk of Impact from ANS Analysis used in the BE (from USEPA and Navy 2018)

The risk classification applied in the BE was as follows:

- Remote – federally listed species are:
  - very unlikely to be exposed to ANS and effects of ANS on a listed species population or its critical habitat are expected to be minor;

  - unlikely to be exposed to ANS and effects of ANS on a listed species or its critical habitat are expected to be minor; or
  - the effects of ANS on a listed species population or its critical habitat are expected to be undetectable.
- Negligible – federally listed species are:
  - very unlikely to be exposed to ANS but effects of ANS on a listed species population or its critical habitat are expected to be moderate or major if an invasion does occur;
  - unlikely to be exposed to ANS but effects of ANS on a listed species population or its critical habitat are expected to be moderate if an invasion does occur; or
  - likely or very likely to be exposed to ANS and ANS are expected to have minor effects on a listed species population or its critical habitat.
- Potentially significant – federally listed species are either:
  - unlikely to be exposed to ANS, but ANS are expected to have major effects on a listed species population or its critical habitat if an invasion does occur; or
  - likely or very likely to be exposed to ANS and ANS are expected to have moderate effects on a listed species population or its critical habitat.
- Likely significant – federally listed species are either likely or very likely to be exposed to ANS and ANS are expected to have major effects on a listed species or its critical habitat.

Among species' groups representing ESA-listed species under NMFS jurisdiction, this analysis identified potentially significant risk for corals, marine snails (i.e., abalone), anadromous salmonids and sturgeon, estuarine and marine fishes, sea turtles, and seagrasses. Effects were expected to be negligible for marine mammals because they feed primarily offshore, so ANS invasions would not be expected to impact habitat quality or food resources for marine mammals.

The BE conclusions for designated critical habitat indicated potentially significant risk for elkhorn and staghorn coral and for Johnson's seagrass because introduction of hull fouling algae or sessile invertebrates to resulting in ANS invasion could have major consequences.

The detailed description of ANS risk in Appendix H of the BE concluded that:

> *Although there is potentially significant risk to some federally listed species from ANS invasions, UNDS (i.e., the action) reduces that risk. Therefore, the action either will not affect federally listed aquatic and aquatic-dependent species or may affect, but is not likely to adversely affect federally listed aquatic and aquatic-dependent species in the action area.*

This reasoning followed the BE's Effect Determination Matrix (Table 10in this opinion). As explained previously, the matrix prescribes LAA determinations only for those aspects of the UNDS expected to increase stressor exposures and having risk characterized as potentially or likely significant. This means that UNDS standards that decrease exposure to discharges, despite resulting in potential or likely significant risk to ESA-listed species or designated critical habitat, are determined to be NLAA. We repeat that this approach is not consistent with ESA section 7 consultation requirements.

The BE acknowledges that, while the UNDS measures reduce risk, they do not eliminate the potential for ANS invasions to occur. The BE integrated the likelihood of overlap of ESA-listed species and designated critical habitat with the RAA's (BE Table 5-9). This provides a location-specific assessment for species that may occur in those RAAs, but cannot be extrapolated to species in other areas affected by the action. Further, the rule will be in effect for an indeterminate amount of time and extrapolations into the future are uncertain due to the effects of climate change on shifts in species distributions and changes in habitat suitability.

The exposure modeling and resulting risk analysis used in the BE does not address all of NMFS concerns with discharges under the UNDS rule, including from ship husbandry activities that may introduce NAS to ports and harbors with the potential to become ANS and adversely affect ESA-listed species and designated critical habitats.

Pimentel et al. (2005) reports that NAS are a major cause of decline and significant impediment to recovery for 42 percent of all listed species. An analysis by Wilcove et al. (1998) identified ANS as the second largest threat to endangered species after habitat loss. Gangloff et al. (2016) identified invasive species as one of the four greatest threats to the world's oceans. Invasive species are recognized among many significant threats contributing to environmental damage and extinction risk (Arthington et al. 2016; Dueñas et al. 2018). Dueñas et al. (2018) conducted a systematic literature review of the available scientific evidence on invasive species' interactions with all endangered and threatened species protected under the ESA. The review found scientific evidence available for 116 endangered or threatened species (8.5 percent of all ESA-listed species), of which 85 species (6.2 percent) were reported as being negatively impacted by invasive species (Dueñas et al. 2018). Of the 85 ESA-listed species reported as being negatively impacted by invasive species, seven were marine species: four sea turtle species adversely impacted by terrestrial invasive predators on nesting beaches; and three salmonids adversely impacted by non-indigenous species of trout (Dueñas et al. 2018).

The introductions of hull-fouling ANS are most likely to have indirect effects on ESA-listed Southern resident killer whales, sea turtles and fish species. Indirect effects may include changes to benthic habitat, changes to prey, and/or competition for food resources. These effects could result from ANS preying upon, outcompeting, or smothering organisms that may be critical to benthic habitat or food chains. The alteration of a species' habitat or food chain could potentially lead to behavioral disturbance in the form of requiring animals to travel farther or could cause fitness consequences if the animal is unable to feed. In some instances, we would expect NAS introduced via hull fouling to suppress an ecologically similar native species low in the food web (e.g., sessile benthic invertebrate) and the impacts of that invasion have a negligible effect to the

ESA-listed species or critical habitats considered in this opinion. For example, loggerhead sea turtles are generalist feeders and it is unlikely additional biofouling species would affect the ability of these species to locate food, even if they were to co-occur with biofouling invasion areas.

Sturgeon feed on a variety of benthic invertebrates including mollusks, polychaete worms, crustaceans, gastropods, shrimps, pea crabs, decapods, amphipods, isopods, and small fishes in the marine environment (Savoy 2007; Savoy and Benway 2004; Collins et al. 2008; Guilbard et al. 2007). Salmonid fry feed largely on plankton in streams while adults eat aquatic and terrestrial insects, mollusks, crustaceans, fish eggs, and small fish. Many of the prey items of ESA-listed fish likely have an ecologically similar function or niche as the hull fouling species that could potentially be introduced via ship hulls as part of the action. Both positive and negative impacts have been reported in the literature from the introduction of nonindigenous benthic invertebrate species on fish species (Schlaepfer et al. 2011). Vinson and Baker (2008) found that the nonindigenous mudsnail (*Potamopyrgus antipodarum*) was an abundant prey item for native fish in the western U.S.. However, when native rainbow trout (*oncorhynchus mykiss*) fed exclusively on mudsnails they lost 0.5 percent of their body weight per day. This study suggests that, in some cases, even if nonindigenous invertebrate species can provide a new food source, the resulting effect can still be detrimental to native species if the nonindigenous prey is not as nutritionally valuable as the native prey items that it is replacing.

Invertebrates can have major impacts on the ecosystems they invade. Benthic invertebrates, such as mussels, polychaetes, and hydroids can become dominant filter feeders, greatly reducing the amount of organic energy that is available to native taxa in the water column (NMFS 2012c). This transfer of energy from the water column into the benthos fundamentally alters the ecology of the host habitat, resulting in less prey available for other filter feeders. Adverse effects of this include reduced body condition, growth, survival, and reproduction of native pelagic organisms at the same or similar trophic level as the invader if the native competitor cannot adapt to another food source. These changes would be manifested to a greater or lesser degree up the food chain to higher trophic level organisms in the habitat, including ESA-listed sturgeon and sea turtles (NMFS 2012c). For example, European green crabs (*Carcinus maenas*) have invaded both the U.S. East and West coasts, including some of the RAAs selected for analysis by the Navy and EPA, having trophic scale effects to both environments. In Massachusetts Bay, green crabs prey upon native mussels and oysters, altering community structure (Grosholz 2002; Lafferty and Kuris 1996; Pimentel et al. 2004).

ANS may also have direct and indirect effects on ESA-listed Atlantic/Caribbean corals and black abalone. Direct effects include preying upon these species. Indirect effects may include competition for food resources and competition for habitat, as well as alteration of habitat such that it is no longer optimal or suitable for settlement and growth of corals or abalone. However, because the vessels of the armed forces in the Miami and San Francisco RAAs are largely vessels smaller than 79 ft in length, which are typically removed from the water for hull cleaning, and because non-Navy vessels larger than 79 ft (such as USCG vessels) rarely travel to other ports, the risk of ANS from these vessels is negligible. For these reasons, it is also likely that hull fouling organisms on vessels in these RAAs are from these RAAs rather than from another port. Some of these organisms may be ANS introduced from non-armed forces vessels but would not be a result of hull cleaning discharges subject to UNDS.

8.3.2 Oil and Grease and Total Petroleum Hydrocarbons (TPH)

Oil and grease are expressed as hexane extractable material (HEM) and silica gel treated hexane extractable material (SGT-HEM). HEM contains relatively nontoxic nonvolatile hydrocarbons, oils, fats and waxes. SGT-HEM contains non-polar petroleum hydrocarbons, including toxic partially combusted hydrocarbons and petrochemical constituents such as polycyclic aromatic hydrocarbons, benzene, toluene, ethylbenzene and xylene (semivolatile organic compounds and volatile organic compounds). The term TPH refers to the hydrocarbons present in SGT-HEM and these terms can be used interchangeably. Several hundred chemical compounds could be part of a TPH mixture, and the composition of the mixture depends on the source, age, and environmental conditions. The amount of TPH found in a sample is a general indicator of petroleum contamination but reveals little about how the particular petroleum hydrocarbons in the sample may affect the environment.

The BE reviews the types of effects that could occur as a result of exposures, but these are associated with the extreme case of oil spills, not incidental discharges. The direct and indirect effects of petroleum oils on many species of fish have been investigated extensively (USEPA 1999b). Oil constituents can be highly toxic and carcinogenic, and may inhibit reproduction and cause organ damage or even mortality (USEPA 1999b). Commonly reported individual effects of petroleum oils in fishes include: impaired reproduction and growth, blood disorders, liver disorders, kidney disorders, malformations, altered respiration or heart rate, altered endocrine function in fishes, altered behavior, increased gill cells, fin erosion, and death. Oils can also act on the epithelial surfaces of fishes, accumulate on gills, and prevent respiration (Howarth 1989). Oil coating surface waters can also interfere with natural processes of re-aeration and increase oxygen demand, depleting the water of oxygen.

While deleterious effects of oil via spills are well known, oil spills are extreme events relative to discharges incidental to vessel operation. It is possible that vessel discharges may cause a sheen or slick immediately surrounding the vessel of the armed forces for a short period of time in a concentration at or near the discharge standard of 15 milligrams per liter. This standard is based on the CWA requirement for “no sheen” and an EPA study defining sheen (Horenstein 1972) and has not been evaluated in terms of adverse effects on biota. It is the water-soluble fraction of TPH that is most likely to cause toxic effects in aquatic organisms. The water-soluble fraction is primarily composed of benzene, toluene, ethylbenzene, and xylene (BTEX). However, this fraction rapidly evaporates to the atmosphere (NRC 2003b). The PAH in the water soluble fractions of TPH are dominated by moderately volatile two- and three-ringed compounds (NRC 2003b). Lighter components, such as gasoline which contains ~80-90 mg BTEX per liter, remains for several minutes to hours, and most lubricating oil evaporates within two days (ATSDR 1999; NRC 2003b). A smaller portion of the heavier oil or grease can remain on the surface marine microlayer for longer periods (days) depending on environmental conditions, including physical, chemical, and biological processes (NRC 2003b).

The BE demonstrates that its modeled concentrations of TPH in the RAAs of 3.70E-08 to 1.90E-06 micrograms per liter are well below the Association of Southeast Asian Nations' marine water quality criterion of 0.14 milligrams per liter. This criterion was derived specifically for the protection of marine aquatic life exposed to the water soluble fraction of TPH in Southeast Asian Waters (Tong et al. 2019). While NMFS does not consider the modeled concentrations to realistically represent actual conditions, it is reasonable to expect components of a TPH discharge meeting the standard of 15 milligrams per liter would rapidly volatilize and dilute to a concentration that is 100-fold, or more, lower in the vicinity of the vessel, thus approaching the Southeast Asian Nations' marine water quality criterion. The NOAA acute screening level for PAHs, which are present in the water-soluble fraction of TPH, is higher than this criterion, at 0.3 milligrams per liter (Buchman 2008). Even though the mass PAH in the water soluble fraction of a TPH mixture varies widely, we would not expect acute exposures to PAH to occur above the NOAA screening level of 0.3 milligrams per liter due to TPH discharges under the UNDS. The BE acknowledged that the available benchmarks do not support a quantitative risk assessment, but concluded that they:

> *"provide confidence that the conclusion of the qualitative assessment that risk to federally listed species from exposure to oil and grease in vessel discharges [is remote] is accurate."*

It is in the implementation of the UNDS rule that ultimately determines the exposure and resulting risk to ESA-listed species and their critical habitat posed by oil and grease.

The performance standards are meant to limit the potential for exposure of ESA-listed species and their habitat to TPH but there will still be some exposure. NMFS expects responses of ESA-listed species to TPH in UNDS-compliant discharges to be insignificant and therefore NLAA based on the expected rapid volatilization, degradation, and dilution of toxic components in the near vicinity of the discharge to concentrations below which effects would occur.

8.3.3 Oxygen Demanding Substances

Elevated loadings of organic material increase levels of oxygen demand as measured by biological oxygen demand (BOD), through microbial break down, and chemical oxygen demand (COD), through the chemical breakdown of particulate and soluble organic matter. Oxygen-demanding substances occur in graywater and underwater ship husbandry discharges. Elevated oxygen demand can lower DO levels in surface water and sediment pore water. Oxygen depletion affects organisms through respiratory stress due to insufficient oxygen. If organisms cannot move and avoid unfavorable DO conditions, direct mortality can result. Avoidance behaviors can influence growth, fecundity and recruitment as organisms expend energy seeking more favorable environments and under circumstances where the amount of habitable space is reduced. Adverse DO conditions affecting food resources in an area, through direct mortality or avoidance by prey, affects growth, fecundity, and recruitment.

Sufficient DO is an important habitat characteristic for many of the species described in Section 6.2. Hypoxia can cause habitat loss, long-term weakening of species, change in species dynamics and even fish kills. Because hypoxia often occurs in estuaries or nearshore areas where the water is poorly mixed, nursery habitat for fish and shellfish is often affected. Without nursery grounds, the young animals cannot find the food or habitat they need to reach adulthood. This causes years of weak recruitment to adult populations and can result in an overall reduction or destabilization of important stocks (Howarth 2008; Wu 2009; Campbell and Rice 2014). While low DO levels, in and of themselves, would not have adverse effects on aquatic plants like seagrasses, disruption and declines in ambient DO promote the generation of toxic sulfides. Sulfide is toxic to seagrass, further reducing metabolism, photosynthesis, and growth (Goodman et al. 1995; Stumm and Morgan 1996; Erskine and Koch 2000; Greve et al. 2003; Pollock et al. 2007).

A study of hypoxia in a semi-enclosed coastal bay in Panama found that hypoxic conditions were in the bottom layer of the stratified water column, leading to mortality of sponges, corals, crustaceans, gastropods, and echinoderms, thick mats of bacteria, and an exclusion of consumers that would normally eat dead organisms below a certain depth (Altieri et al. 2017). Oxygen levels below 0.5 mg/L were observed at bottom sites while all areas with depths of less than four m had DO concentrations above 4.8 mg/L (Altieri et al. 2017). The most hypoxic waters were those with greater depths near land where terrestrial inputs were expected to be greater and included untreated sewage and where exchange with the open ocean was poor. Altieri et al. (2017) noted that, although the event caused mass bleaching and mortality of corals and other organisms, laboratory experiments showed that not all coral species were as sensitive to hypoxia. Hypoxia was also found to play an important role in coral tissue loss during coral-algae interaction processes (Haas et al. 2014). Haas et al. (2014) found that algae were significantly more tolerant to low oxygen concentrations (two to four mg/L) than corals and that corals could only tolerate reduced oxygen concentrations only until a certain combination of exposure time and concentration occurred.

The question that must be addressed in this Opinion is whether UNDS-compliant discharges of oxygen demanding substances will result in the DO conditions described above that would adversely affect ESA-listed species. Bilgewater and graywater discharges can be characterized as discharges with potentially high oxygen demand. Underwater hull husbandry discharges contribute to oxygen demand depending on the age of the AFC and the amount of biofouling. The BE acknowledges that BOD in vessel discharges, specifically greywater and bilgewater, can exceed the lowest World Health Organization (WHO) recommended water quality criterion for BOD of two milligrams oxygen per liter, and COD can exceed the lowest WHO recommended water quality criterion for COD of three milligrams oxygen per liter. Oxygen demand estimates based on the RAA modeling were 0.000096 to 0.19 micrograms per liter for BOD and 0.00026 - 0.28 micrograms per liter for COD. However, NMFS does not consider the modeled concentrations to realistically represent actual conditions.

It is in the implementation of the UNDS rule that ultimately determines the risk posed by oxygen-demanding substances. In port, the graywater system is connected to the blackwater system for offloading to a shore-based facility. Before the ship is underway, they are disconnected. When leaving port, and upon arrival in port, there is “sea and anchor” detail where almost all hands are on deck performing functions associated with departing or arriving until the ship is outside of port or pier side, respectively. This means there is no laundry, minimum showers, and minimum scullery activities. Regarding bilgewater, prior to departing, a ship will offload bilgewater to shore for proper disposal so that, as the ship gets underway, the bilge is dry and there is no need to discharge in port, and it is rare to discharge inside 12 nm. Discharges of graywater and bilgewater are therefore not expected to occur in poorly flushed areas such as a sheltered port that are vulnerable to oxygen demanding substances. Finally, oxygen demanding substances contributed by underwater ship husbandry occurring in port is not expected to be substantial because hull maintenance and cleaning practices under the UNDS do not allow vessels of the Armed Forces to become substantially fouled.

NMFS expects exposures of ESA-listed species to depleted dissolved oxygen conditions resulting from oxygen-demanding substances in UNDS-compliant discharges to be insignificant. Therefore, the effects of exposure will be NLAA. This conclusion is based on the expectation that graywater and bilgewater will not be discharged in poorly flushed areas, such as ports, that are vulnerable to oxygen demanding substances and that oxygen demanding substances discharged in port during underwater ship husbandry will be insignificant because hull grooming occurs before large accumulations of hull fouling organisms are present on hulls of active vessels.

8.3.4 Pharmaceutical and Personal Care Products

Pharmaceutical and personal care products represent thousands of substances, many of which have not been thoroughly evaluated for fate and transport in the marine environment and effects to marine species. The BE describes the breadth of this pollutant group and explains that discharges would be limited to standard personal care products and topical treatments washed off during showers and entering graywater. NMFS expects that these would include sun block containing oxybenzone that is known to be very toxic to coral and insect repellents containing pesticides that are toxic to fishes. Caffeine is also expected to be reliably present in vessel graywater due to washing of coffee makers and teapots and other implements used to make and drink coffee and tea. The BE addressed this pollutant class qualitatively, while our evaluation uses data from domestic discharges to help frame the worst case exposure and associated risk posed by graywater discharges.

Domestic wastewater influent is composed of about 60 percent graywater (Edwin et al. 2014), Oxybenzone, presumably from graywater contributions, occurred at higher concentrations than any other sunblock active ingredient, at concentrations as high as 700 nanograms per liter (Ramos et al. 2016), suggesting graywater concentrations of about 1.2 micrograms per liter. The ultraviolet filters used in personal care products have endocrine disrupting properties. The products that have been evaluated include oxybenzone, avobenzone (butyl methoxydibenzoylmethane), homosalate, octyl dimethyl-p-aminobenzoic acid (padimate O), and octinoxate (Schlumpf et al. 2001). Downs et al. (2016) report that oxybenzone exposures at environmentally plausible concentrations observed at tropical beaches resulted in coral planulae deformation and evidence of bleaching in adult colonies. The LC50s for seven coral species ranged from eight to 340 micrograms per liter with sublethal effects reported at concentrations as low as 6.5 micrograms per liter, or about six–fold the concentration that would be expected in graywater. Based on information from the Navy, crew members use no or minimal amounts of sunscreen because their duties aboard the vessel require they wear clothing that limits the amount of exposed skin for safety reasons. On larger vessels with showers, while the vessel is underway in port or in nearshore areas, showering is limited so this waste stream would not be a measurable contributor to graywater in these areas. Therefore, the concentrations from sunscreen use at the beach are not expected to be comparable to the limited amount of sunscreen use on a vessel of the armed forces. Additionally, the majority of sunscreen use on armed forces vessels would occur on smaller vessels that do not have showers, meaning oxybenzone would not be part of graywater discharges from these vessels (if there are any other sources, which depends on vessel size and type).

The most common pesticide used in insect repellent is N, N-diethyl-m-toluamide (DEET). (Aronson et al. 2012); Olmstead and LeBlanc (2005) estimated average DEET concentration in wastewater reaching water treatment plants to be 1.9 micrograms per liter. Concentrations of DEET in South Florida coastal waters were reported at between 0.005 to 0.048 micrograms per liter (Singh et al. 2010). Unfortunately, data on the effects of DEET on marine organisms is limited to reports on dinoflagellates. Martinez et al. (2016) reported the NOEC for DEET effects growth in *Levanderina fissa* of at 10,000 micrograms per liter. This falls between toxicity data reported for common freshwater laboratory species, which ranges from an Effects Concentration at which five percent of the test organisms die (EC05) of 1,000 micrograms per liter for effects to water flea growth and reproduction (Olmstead and LeBlanc 2005) to a LOEC of 56,000 micrograms per liter for water flea mortality (Forbis and Burgess 1985). As for sunscreen, vessel crew members use no to limited amounts of insect repellant due to the requirement that they be wearing clothing that limits the amount of exposed skin. In addition, the use of insect repellant is more likely to occur on smaller vessels that do not have showers and would therefore not be part of any graywater discharges. Discharges of graywater from larger vessels may contain water from showering, but information from the Navy indicates that this waste stream would be discharged offshore. Thus, while we know other pesticides have toxic effects on listed species such as corals based on studies targeting these organisms, we do not believe insect repellants would have effects to ESA-listed species in areas where military vessels may discharge graywater in accordance with the performance standards.

Caffeine is expected to occur in graywater from vessel galleys, but estimates from the limited number of domestic surrogates vary widely. Caffeine in domestic graywater averaged 1.5+/- 0.78 micrograms per liter in Sneek, Netherlands (Leal 2010) and 850 +/- 360 micrograms per liter in Queensland, Australia (Turner 2017). Meanwhile there is limited information on the toxicity of caffeine to marine organisms, with data for effects to body part regeneration in the solitary tubeworm reported at concentrations as low as 0.5 micrograms per liter (Pires et al. 2016) and growth inhibition in algae ranging from 30 to 75 milligrams per liter (Pollack et al. 2009). It is possible that zooxanthellae, which are an algae, could be affected in a similar way to the algae from the Pollack et al. (2009) study, leading to declines in growth of ESA-listed Atlantic/Caribbean coral species that depend on the relationship with their zooxanthellae for growth.

It is in the implementation of the UNDS rule that ultimately determines the risk posed by products in graywater discharges. Graywater discharges are expected to release pharmaceutical and personal care products, as well as caffeine and other food residues from galley wastewater, to marine waters. However, the performance standards require that graywater not be discharged while vessels are in a port with onshore facilities for disposal of this wastewater, which is the case for all RAAs and other harbors in the U.S. where there are facilities of the armed forces based on information in the BE. In addition, any discharges of graywater while underway in port would not include laundry, dish washing or cooking, and showering would be very limited. The performance standards do allow for discharge to marine waters outside port areas, which include areas containing ESA-listed Atlantic/Caribbean corals. In addition, as discussed previously, wake mixing is expected to dilute the concentrations of constituents in graywater discharges to levels that are not expected to result in measurable effects to ESA-listed species or designated critical habitat. Therefore, we believe graywater discharges under the UNDS Batch Two rule may affect, but are not likely to adversely affect the Southern Resident killer whales, loggerhead sea turtles, ESA-listed Atlantic coral species, Johnson's seagrass, bocaccio and yelloweye rockfish, ESA-listed sturgeon species, ESA-listed Atlantic and Pacific salmonids, or their designated critical habitat.

8.3.5 Chlorine and Chlorine-Produced Oxides

The BE conducted a quantitative assessment of the risks posed by chlorine use associated with hull cleanings, deck cleanings, and graywater. Chlorinated compounds are also volatile and evaporate quickly. As such, their presence in deck runoff and graywater is expected to be minimal to nonexistent. In general, the hypochlorous acid formed during the dissolution of chlorine in natural waters reacts with organic and inorganic materials, ultimately forming chloride ion, oxidized inorganics, chloramines, trihalomethanes, oxygen, and nitrogen. Consequently, chlorine itself (as well as chlorine-produced oxides [CPO] and TRC) does not persist in the aquatic environment (USEPA and Navy 2018). Chlorine species in saltwater are referred to as CPO because, in addition to the chlorine species formed in freshwater, speciation in saltwater also produces hypobromous acid, hypobromous ion, and bromamines (USEPA 1984).

Due to the volatility of chlorinated compounds, exposures of ESA-listed species to chlorine and CPO in discharges from deck wash and graywater are expected to be insignificant. In terms of underwater hull husbandry, the main source of CPO in underwater ship husbandry discharges was from accumulation in the vinyl cover placed around the propulsor when SEAWOLF attack submarines were ported for extended periods. This practice no longer occurs based on information from the Navy. Therefore, NMFS expects exposures of ESA-listed species to CPO discharges from underwater ship husbandry will not occur.

8.3.6 Metals

Metals occur naturally in the marine environment but are present in elevated concentrations in nearly all of the discharges considered during this consultation. The BE reviews factors influencing bioavailability and the potential for metals to accumulate in biota. The CTETs applied in the BE are summarized in Tables 11 to 14 and are the result of calculating the mean of the lowest and NOEC and lowest LOEC or applying an appropriate acute to chronic ratio to an LC50. Risk was determined by comparing these CTETs to exposure estimates, assuming metals were in the dissolved (i.e., most toxic) form. As discussed previously in this section of the opinion, NMFS has concerns regarding the BE exposure estimates and associated assumptions. Data is limited for saltwater exposures (Table 15), so data available for calculation of the CTETs do not fully represent the range of effects to ESA-listed species for each metal.

Table 15. Number of saltwater toxicity test observations for chronic exposures reported in ECOTOX.

Metal	Mortality	Reproduction	Growth	Population	Feeding behavior and avoidance	Development and morphology
Fishes						
Cadmium	7	1	10	no data	no data	12
Copper	60	9	55	no data	3	37
Iron	4	no data	no data	no data	no data	no data
Lead	no data	1	no data	no data	no data	no data
Nickel	3	no data	5	no data	no data	no data
Zinc	4	1	5	no data	7	no data
Chromium VI	23	no data	14	no data	no data	no data
Invertebrates						
Cadmium	78	76	48	26	11	129
Copper	407	278	160	140	29	630
Iron	22	no data	4	52	no data	8
Lead	19	28	12	no data	1	31
Nickel	29	14	19	8	2	30
Zinc	222	70	118	4	12	260
Chromium VI	37	54	12	48	no data	21

While some metals, including copper, nickel, and zinc, are essential to organism function, in excess amounts, these have metal-specific pathways for causing adverse effects. The one pathway to toxic effects that metals have in common is the generation of free radicals that indiscriminately oxidize proteins, nucleic acids, and cell membranes. This damage stimulates defensive measures and repair or replacement of cells. Accumulating damage depletes energy reserves and damages tissues, impairing organ function. Examples of other pathways to adverse effects include copper's interference with fish olfaction, affecting predator detection and cadmium's mimicry of calcium, affecting bone loss and calcium-dependent neuromuscular functions. The pathway to adverse effects does not require actual uptake into body tissues as adverse effects may occur due to metal impairment of gill membranes.

Metals and metalloids that are micronutrients (e.g., selenium, zinc) can be beneficial to coral health and growth when they occur at naturally low levels. However, at increased concentrations, they can be toxic. Heavy metals bioaccumulate in coral host tissues and are most heavily concentrated in the zooxanthellae (Reichelt-Brushett and Mcorist 2003). Tissue body burden may far exceed concentrations found in skeletal material (Bastidas and Garcia 1997; Mcconchie and Harriott 1992), and the contaminants in tissues more directly affect coral physiology.

However, it is difficult to generalize responses to metal contamination because effects can be species-specific or moderated by exposure history. Although bleaching is a generalized stress response, heavy metals can directly induce coral mortality in the absence of bleaching (Mitchelmore et al. 2007). Elevated levels of iron have resulted in expulsion of zooxanthellae from *Porites lutea* (a Pacific coral), but corals exposed to daily runoff enriched with iron had a reduced response, suggesting that corals may be capable of adapting somewhat to exposure (Harland and Brown 1989).

Regardless of form, metal discharges contribute to the net load and existing legacy metals resident in sediments. The proportion of metal in ionic form depends upon the pH, salinity, oxidation state, and other factors influencing solubility. Metals are expected to go into complex or adsorb to particles in a surface water and precipitate out of solution. Sediment is not a static sink for metals. Kalnejais et al. (2010) quantified the potential release of dissolved metals from Boston Harbor sediments due to resuspension and determined that 2-5 percent of sediment bound copper were released when sediments were retained in suspension for 90 hours. Such estimates are dependent on a multitude of factors that are infeasible to measure, but that does not mean the risk from such releases should be dismissed.

It is in the implementation of the UNDS rule that ultimately determines the risk posed by metal discharges. Although there are limited studies on the toxicity of metal to marine organisms, particularly ESA-listed species, discharges could have sublethal effects to ESA-listed species, particularly Atlantic/Caribbean corals and black abalone that are likely to respond similarly to corals given their similar life cycles. Discharges containing metals may also have adverse effects on early life stages of fishes based on existing exposure studies. Southern Resident killer whales have been found to bioaccumulate contaminants with potential consequences to growth and reproduction. Sea turtles, particularly loggerheads, have been shown to bioaccumulate these contaminants but the potential health consequences, if any, of this bioaccumulation are unknown.

8.4 Risk analysis

Here we assess the consequences of the responses to the individuals that will be exposed to the effects of metals and ANS from UNDS Batch Two discharges, specifically hull coating leachate and underwater ship husbandry, the populations those individuals represent, and the species those populations comprise. For designated critical habitat, we assess the consequences of these responses on the value of the critical habitat for the conservation of the species for which the habitat had been designated.

Southern Resident killer whales are present in the Puget Sound area and RAA in the spring through early winter, though densities differ with time of year as they track movements of prey species, including chum and chinook salmon. In winter, these animals are more frequent offshore along the west coast. Prey species include salmonids and other fishes that may be adversely affected by metals from hull coating leachate and underwater ship husbandry and ANS from underwater ship husbandry in the Puget Sound area. Therefore, Southern Resident killer whales are likely to be adversely affected by ingestion of contaminated prey and bioaccumulation with potential health consequences to individuals, as well as impacts to prey from ANS either due to competition or habitat alteration.

The loggerhead is the most common sea turtle in the southeastern U.S. nesting along the Atlantic coast of Florida, South Carolina, Georgia, and North Carolina and along the Florida and Alabama coasts in the Gulf of Mexico. Juveniles inhabit continental shelf waters from Cape Cod Bay, Massachusetts, south through Florida, The Bahamas, Cuba, and the Gulf of Mexico. Thus the Northwest Atlantic Ocean DPS is present in the Norfolk and Miami RAAs, as well as other areas containing ports and harbors with concentrations of military vessels along the east coast and in the Gulf of Mexico. Thus, Northwest Atlantic Ocean loggerhead sea turtles, particularly juveniles, are likely to be adversely affected by metals from military vessels due to impacts to prey from ANS and metals and uptake of contaminants by the animals themselves, as well as exposure to contaminated sediments because they forage in sandy and muddy bottoms.

Larval and juvenile rockfish (that include bocaccio and yelloweye rockfish) are present in Puget Sound (Greene and Godersky 2012; Carr 1983; Haldorson and Richards 1987; Matthews 1990; Love et al. 2002; Yamanaka et al. 2006). Designated critical habitat for yelloweye rockfish (Puget Sound/Georgia Basin DPS) and bocaccio (Puget Sound/Georgia Basin DPS) includes 590.4 square miles (mi^2) of nearshore habitat for bocaccio and 414.1 mi^2 of deepwater habitat for yelloweye rockfish and bocaccio. Habitat and prey species for these fish, as well as juvenile and larval stages of the fish, in the area of the Puget Sound RAA are likely to be adversely affected by ANS and by metals with potential health consequences to different life stages of bocaccio and yelloweye rockfish due to the impacts of ANS on prey and habitat and consumption of contaminated prey and bioaccumulation in the animals by metals. ANS may also consume smaller life stages of bocaccio and yelloweye rockfish.

Certain chinook, coho, and chum salmon ESUs, and steelhead trout DPSs and their designated critical habitats may be found in RAAs on the West Coast, particularly Puget Sound and San Francisco Bay, depending on the species and ESU/DPS. Some of these ESUs/DPSs can also be found in other areas where military vessels operate, such as Alaska, but are present in low numbers in ports and harbors. The geographic areas included in the designated critical habitat for Hood Canal summer-run chum salmon include the Puget Sound subbasin and nearshore marine areas of Hood Canal and the Strait of Juan de Fuca from the line of extreme high tide to a depth of 30 m. ANS and metals in discharges from hull coating leachate and underwater ship husbandry from vessels associated with the Puget Sound and San Francisco RAAs, as well as vessels of the armed forces in other ports and harbors on the west coast and in Alaska, are likely to adversely affect these species and their habitat due to the impacts of ANS to prey and habitat and bioaccumulation of metals from prey consumption in the animals themselves with potential health consequences to individuals. ANS may also consume smaller life stages of Pacific salmonids.

Gulf of Maine Atlantic salmon and its designated critical habitat are present in the Gulf of Maine and associated rivers and estuaries. ANS and metals in discharges from hull coating leachate and underwater ship husbandry from military vessels in ports and harbors within the Gulf of Maine may adversely affect this species and its habitat due to the impacts of ANS to prey and habitat and bioaccumulation of metals from prey consumption in the animals themselves with potential health consequences to individuals. ANS may also consume smaller life stages of Atlantic salmon.

ESA-listed Atlantic/Caribbean corals and elkhorn and staghorn coral designated critical habitat are present in the area of the Miami RAA, as well as in areas such as San Juan, Puerto Rico where there are facilities for vessels of the armed forces. Impacts to the density and distribution of plankton that serve as food resources for corals can affect coral health, growth, and reproduction. Declines in water quality, including contaminants from vessel discharges, such as copper and other chemicals from anti-fouling paints from underwater ship husbandry, can have toxic effects to corals. Resuspension of sediments contaminated by vessel discharges can smother corals and degrade their habitat. Toxic effects and smothering result in decreases in growth and declines in reproduction. Degradation of coral habitat decreases coral settlement and growth to maturity. Therefore, hull coating leachate and underwater ship husbandry discharges from military vessels in ports and harbors within the range of ESA-listed Atlantic/Caribbean corals and elkhorn and staghorn coral critical habitat may adversely affect these species and their habitat due to bioaccumulation of contaminants, hindrance of reproduction and settlement, potential toxic effects of metals to corals with potential health consequences to individuals, and potential competition with corals for settlement habitat or consumption of coral larvae, recruits, or tissue from larger colonies.

Green sturgeon have been observed in large concentrations in the summer and autumn within coastal bays and estuaries, including San Francisco Bay, which is part of the designated critical habitat for the ESA-listed Southern DPS. Tagged subadults and adults from the Southern DPS have been found in Puget Sound. Habitat and prey species for these fish along the west coast in areas with facilities for vessels of the armed forces, as well as juvenile, subadult, and adult stages of the fish in the San Francisco RAA, are likely to be adversely affected by UNDS Batch Two discharges from hull coating leachate and underwater ship husbandry due to impacts to prey and habitat from metals and ANS and bioaccumulation of metals from prey consumption in the ESA-listed animals themselves with potential health consequences to individuals.

Juvenile, subadult and adult gulf sturgeon inhabit coastal rivers, estuaries and bays from Louisiana to Florida, and the Gulf of Mexico, including in port and harbor areas containing concentrations of military vessels. Habitat and prey species for these fishes along the Gulf coast in areas with facilities for vessels of the armed forces, as well as juvenile, subadult, and adult stages of the fishes in these areas, are likely to be adversely affected by UNDS Batch Two discharges from hull coating leachate and underwater ship husbandry due to impacts to prey and habitat from metals and ANS and bioaccumulation of metals from prey consumption in the animals themselves with potential health consequences to individuals.

Shortnose sturgeon were historically found in coastal rivers along the east coast of North America from Canada to Florida, but their distribution is now broken up with a separation between mid-Atlantic and the southern metapopulations. Fishes from both metapopulations may be found in ports and harbors containing facilities for military vessels. One of these areas is the Norfolk RAA. The various DPSs for Atlantic sturgeon are located in areas containing ports and harbors used by vessels of the armed forces and the Chesapeake Bay DPS is in the Norfolk RAA. Prey species for these species along the east coast in areas with facilities for vessels of the armed forces, as well as juvenile, subadult, and adult stages of the fishes in these areas, may be adversely affected by UNDS Batch Two discharges. Each of the Atlantic sturgeon DPSs have designated critical habitat in rivers where these fish spawn from the river mouth to the freshwater spawning sites and larval rearing areas. While most of these rivers are outside areas containing facilities of the armed forces, some of the rivers do have port and harbor facilities and vessels of the armed forces where UNDS Batch Two discharges are reasonably likely to occur. Therefore, Atlantic sturgeon habitats are likely to be adversely affected by hull coating leachate and underwater ship husbandry discharges containing metals and ANS.

All sturgeon species, because they forage by stirring up bottom sediments to look for prey, may be exposed to contaminants including metals from hull coating leachate and underwater ship husbandry discharges that have accumulated in bottom sediments or have resulted in bioaccumulation in prey. Juvenile, subadult, and adult fishes that forage in areas where military vessels are concentrated are likely to be adversely affected by ingestion of contaminated prey and sediment, as well as ANS through competition for prey, with potential health consequences to individuals.

Johnson’s seagrass and its designated critical habitat are present only in the Miami RAA. Declines in water and sediment quality such as from discharges of metals have the potential for toxic effects to Johnson’s seagrass. Sediment resuspension and transport to areas of designated critical habitat may lead to habitat degradation, as well as degradation of the condition of the species itself including through reductions in growth. Underwater ship husbandry discharges containing ANS also have the potential for habitat degradation and direct effects to Johnson’s seagrass such as herbivory. Therefore, hull coating leachate and underwater ship husbandry discharges from military vessels in the Miami RAA in particular within the range of Johnson’s seagrass and its designated critical habitat may adversely affect the species and habitat.

9. 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 C.F.R. §402.02). Future Federal actions that are unrelated to the action are not considered cumulative effects because they require separate consultation pursuant to section 7 of the ESA.

For this consultation, cumulative effects include climate change, anthropogenic sound, military training and testing, dredging, water quality and marine debris, whaling, directed harvest of sea turtles, commercial fisheries, commercial and private whale watching, vessel strike, invasive species, diseases, and scientific research permits. There are ports and harbors throughout the action area with vessel traffic increasing in many locations. Climate change and associated impacts to coastlines may result in increased development of maritime facilities, change in location of some port facilities, or redevelopment of existing facilities to minimize the potential impacts of sea level rise and storm surge. Continuing coastal and maritime development are likely to affect the frequency and extent of dredging projects, as well as water quality and marine debris. Coastal and marine development, and increases in vessel traffic, along with other noise-producing human activities will continue to increase levels of anthropogenic sound in the ocean. Extractive uses, such as hunting and fishing, are expected to continue into the foreseeable future. We are not aware of any proposed or anticipated changes in fishing in particular that would substantially change the impacts of these activities on the whale, sea turtle, coral, and ESA-listed fish species analyzed further in this opinion. Tourism and scientific research activities are also expected to continue at the same or increased levels in the action area.

10. 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 action. In this section, we add the *Effects of the Action* (Section 8) to the *Environmental Baseline* (Section 7) and the *Cumulative Effects* (Section 9) to formulate the agency's biological opinion as to whether the action is likely to: (1) reduce appreciably the likelihood of both the survival and recovery of a ESA-listed species in the wild by reducing its numbers, reproduction, or distribution; or (2) reduce the value of designated or proposed critical habitat for the conservation of the species. These assessments are made in full consideration of the *Status of the Species and Critical Habitat Analyzed Further* (Section 6.2).

10.1 Jeopardy 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 C.F.R. §402.02). Therefore, the jeopardy analysis considers both survival and recovery of the species.

Based on our effects analysis, adverse effects to ESA-listed species are likely to result from vessel discharges under UNDS Phase II Batch Two. Specifically, discharges containing NAS with the potential to become ANS and metals such as copper. Components of discharges from underwater ship husbandry and hull coating leachate may compete with ESA-listed species and their prey or lead to habitat degradation (ANS); or be bioavailable in the water column (metals) or sorb to sediments in the RAAs and other ports and harbors with facilities for vessels of the armed forces (metals). We find it reasonably likely that the following ESA-listed species would be adversely affected by contaminants contained in UNDS Batch Two discharges and resulting sediment contamination as a result of the action given the indeterminate period over which these discharges, and associated exposure to them, will occur: ESA-listed Southern resident killer whales, Northwest Atlantic loggerhead sea turtles, Atlantic/Caribbean corals, Johnson's seagrass, and the following ESA-listed fish species: shortnose sturgeon; Puget Sound/Georgia Basin bocaccio; Puget Sound/Georgia Basin yelloweye rockfish; Central Valley Spring-Run, Puget Sound, and Sacramento River Winter-Run chinook salmon; Hood Canal Summer-Run chum salmon; Central California Coast coho salmon; California Central Valley, Central California Coast, and Puget Sound steelhead trout; Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic sturgeon; green sturgeon; gulf sturgeon; and Gulf of Maine Atlantic salmon. Our Integration and Synthesis section focuses on the anticipated adverse effects associated with these stressors as a result of UNDS Batch Two discharges that will be regulated under the proposed rule over an undetermined period. Specifically, discharges underwater ship husbandry and hull coating leachate containing NAS with the potential to become ANS and metals such as copper that may be bioavailable in the water column, sorb to sediments in the RAAs and other ports and harbors with facilities for vessels of the armed forces. The following discussions summarize the probable risks that water and sediment quality degradation and ANS from UNDS Batch Two hull coating leachate and underwater ship husbandry discharges pose to threatened and endangered species that are likely to be exposed over the undefined lifetime of the rule. These summaries integrate our exposure, response, and risk analyses from Sections 8.3 and 8.4.

10.1.1 Killer Whale

Levels of contaminants in UNDS Batch Two discharges are likely to affect prey species of Southern Resident killer whales particularly during the fall and winter when whales are more likely to be in the Seattle RAA feeding on fish species, including salmon. In order to determine whether the action will jeopardize the continued existence of Southern Resident killer whales, we need to analyze whether these indirect effects on the fitness of Southern Resident killer whales will appreciably reduce the likelihood of survival and recovery of the species.

The need to expend additional effort to find prey species can result in reduced growth rates and lower lifetime fecundity. Chinook salmon populations have declined due to degradation of habitat, hydrology issues, harvest, and hatchery introgression; such reductions may require an increase in foraging effort, which has an energetic burden on the species that may lead to reduced growth and lower fecundity. In addition, the presence of contaminants in these prey species may lead to immune suppression or reproductive impairment. However, the distribution of Southern Resident killer whales will not change due to UNDS Batch Two discharges, as these are not expected to result in degradation of habitat beyond the footprints of facilities for military vessels, changes in hydrology, or harvest of prey species.

Whether the potential effects to reproductive output would appreciably reduce the likelihood of survival of Southern Resident killer whales depends on the probable effect the changes in reproductive output would have relative to current population sizes and trends. The small population size of this species (estimated as 80 individuals), coupled with the apparent pattern of population decline due in part to the population structure (e.g., few reproductive age males and non-calving adult females), means lower fecundity could have population-level effects. If declines in prey species associated with UNDS Batch Two discharges or consumption of contaminated prey occur, these could lead to reduced fecundity of the Southern Resident killer whale population. However, the UNDS Batch Two rule contains performance standards for each of the discharges regulated under the rule, many of which include limits or prohibitions on discharges occurring while vessels are in harbors and ports and requiring discharges be to onshore facilities for receiving wastewater, which include facilities for vessels of the armed forces. With the implementation of these performance standards, we believe the action is not reasonably expected to directly or indirectly cause an appreciable reduction in the likelihood of survival of Southern Resident killer whales in the wild.

The relatively low number of individuals in this population makes it difficult to resist or recover from natural spikes in mortality, including disease and fluctuations in prey availability.

The Final Recovery Plan for the Southern Resident DPS of killer whale (NMFS 2008b) lists the following recovery objectives that are relevant to the impacts of the action:

- Prey Availability: Support salmon restoration efforts in the region including habitat, harvest and hatchery management considerations and continued use of existing NMFS authorities under the ESA and Magnuson-Stevens Fishery Conservation and Management Act to ensure an adequate prey base
- Pollution/Contamination: Clean up existing contaminated sites, minimize continuing inputs of contaminants harmful to killer whales, and monitor emerging contaminants.

As noted, the population of Southern Resident killer whales appears to be in decline and this decline is partially attributed to declines in prey species from stressors that are not associated with contaminants from military vessels. The anticipated non-lethal take of Southern Resident killer whales from indirect effects of UNDS Batch Two discharges could have effects to prey species, including salmonids, but we believe the implementation of performance standards under the rule, which include restricting or prohibiting discharges while vessels are in port, will minimize the potential effects of UNDS discharges. For this reason, we believe the action is not likely to impede the recovery objectives above for Southern Resident killer whales and will not result in an appreciable reduction in the likelihood of the recovery of this species in the wild. We conclude that the action is not likely to jeopardize the continued existence of Southern Resident killer whales in the wild.

10.1.2 Loggerhead Sea Turtles

As discussed in our effects analysis, UNDS Batch Two discharges from vessels of the armed forces are likely to affect habitat and prey for Northwest Atlantic Ocean loggerhead sea turtles, as well as bioaccumulating in turtle tissues with potential fitness consequences. In order to determine whether the action will jeopardize the continued existence of Northwest Atlantic Ocean loggerhead, we need to analyze whether the indirect effects of the discharges on the fitness of this sea turtle species will appreciably reduce its likelihood of survival and recovery.

The need to expend additional effort to find refuge and foraging habitat and prey species can result in reduced growth rates, older age to maturity, and lower lifetime fecundity. However, no reduction in the distribution of the Northwest Atlantic Ocean DPS of loggerhead sea turtles is expected and this turtle species will continue to be present throughout its range where it currently occurs in the action area.

Whether potential effects to reproductive output would appreciably reduce the likelihood of survival of this DPS of loggerhead sea turtles depends on the probable effect the changes in reproductive output would have relative to current population sizes and trends.

The Northwest Atlantic DPS of loggerhead is estimated at 32,000 to 56,000 nesting females with populations in decline or not enough information to make a trend (TEWG 1998; NMFS 2001). The Peninsular Florida Recovery Unit is the largest loggerhead nesting assemblage in the Northwest Atlantic. A near-complete nest census (all beaches including index nesting beaches) undertaken from 1989 to 2007 showed an average of 64,513 loggerhead nests per year, representing approximately 15,735 nesting females per year (USFWS and NMFS 2008). The statewide estimated total for 2016 was 122,706 nests and 18,631 of those from Florida's Gulf coast (FWRI nesting database). Nesting data are the best current indicator of sea turtle population trends, but in-water data also provide some insight. In-water research suggests the abundance of neritic juvenile loggerheads is steady or increasing. Although Ehrhart et al. (2007) found no significant regression-line trend in a long-term dataset, researchers have observed notable increases in catch per unit effort (Ehrhart et al. 2007; Epperly et al. 2007; Arendt et al. 2009). Researchers believe that this increase in catch per unit effort is likely linked to an increase in juvenile abundance, although it is unclear whether this increase in abundance represents a true population increase among juveniles or merely a shift in spatial occurrence. In-water studies throughout the eastern U.S. indicate a substantial decrease in the abundance of the smallest oceanic/neritic juvenile loggerheads, a pattern corroborated by stranding data (TEWG 2009). Therefore, it is not clear whether the population of this DPS is declining.

If declines in refuge and foraging habitat and prey species associated with UNDS Batch Two discharges or consumption of contaminated prey occur, these could lead to reduced fecundity of the Northwest Atlantic Ocean loggerhead sea turtle population. However, the UNDS Batch Two rule contains performance standards for each of the discharges regulated under the rule, many of which include limits or prohibitions on discharges occurring while vessels are in harbors and ports and requiring discharges to onshore facilities for receiving wastewater, which include facilities for vessels of the armed forces. With the implementation of these performance standards, we believe the action is not reasonably expected to directly or indirectly cause an appreciable reduction in the likelihood of survival of loggerhead sea turtles (Northwest Atlantic Ocean DPS) in the wild. The potential reduction in reproduction that could occur as a result of effects from exposure to UNDS Batch Two discharges would not appreciably affect the reproductive output of these sea turtles.

The Final Recovery Plan (USFWS and NMFS 2008) for the Northwest Atlantic Population of Loggerheads lists the following recovery objectives that are relevant to the impacts of the action:

- Ensure the in-water abundance of juveniles in both neritic and oceanic habitats is increasing and is increasing at a greater rate than strandings of similar age classes.
- Manage sufficient feeding, migratory and inter-nesting marine habitats to ensure successful growth and reproduction.
- Minimize trophic changes from fishery harvest and habitat alteration.

The population trends for the DPS of loggerheads considered in this analysis indicate declines from historic nesting populations but more recently potentially slight increases or stability in the juvenile population, although a clear trend of increase or decrease is not clear. The performance standards for each discharge are meant to reduce the potential for introduction of contaminants and ANS to RAAs and other ports and harbors from Batch Two discharges from vessels of the armed forces. Therefore, any non-lethal take of juvenile or adult turtles from this DPS is not likely to impede the recovery objectives identified above and will not result in an appreciable reduction in the likelihood of recovery in the wild. We conclude that the action is not likely to jeopardize the continued existence of the Northwest Atlantic Ocean DPS of loggerhead sea turtles.

10.1.3 Atlantic/Caribbean Corals

ESA-listed Atlantic/Caribbean corals are expected to be adversely affected by the UNDS Batch Two discharges containing contaminants and NAS that may become ANS. Discharges are expected to affect coral growth and reduce fertilization success and recruitment. In order to determine whether the action is likely to jeopardize the continued existence of Atlantic/Caribbean corals, we need to analyze whether the direct and indirect effects of the discharges on the fitness of ESA-listed coral species will appreciably reduce the likelihood of survival and recovery of these species by decreasing their numbers, reproduction, or distribution.

The abundance of elkhorn and staghorn coral is a fraction of what it was before the mass mortality in the 1970s and 80s and recent population models forecast the extirpation of elkhorn coral from some locations over the foreseeable future. Elkhorn coral abundance is at least hundreds of thousands of colonies but likely to decrease in the future with increasing threats. Staghorn coral abundance is at least tens of millions of colonies but likely to decrease in the future with increasing threats. The low density and cover of pillar coral and its natural rarity makes it difficult to extrapolate monitoring data in order to determine trends in abundance. Based on information in our project files from other consultations, pillar coral appears to be more common around Puerto Rico and USVI in general than in South Florida (NOAA National Coral Reef Monitoring Program). Rough cactus coral is naturally uncommon to rare as stated in the listing rule, though the species may be more common in some sites around Puerto Rico and USVI based on data from EPA's bioassessments conducted in 2010 and 2011. Estimates of the populations of pillar and rough cactus corals are not possible due to the limited survey data for these species. The star coral complex (lobed star, mountainous star, and boulder star) has historically been dominant on coral reefs in the Caribbean, though examples from various countries and the Florida Keys, Puerto Rico, and USVI in the U.S. indicate the population has declined. Despite these declines, it is estimated that there are millions of colonies present throughout the range of this species' complex. Therefore, any reduction in numbers as a result of the direct and indirect effects of Batch Two UNDS discharges on ESA-listed Atlantic/Caribbean corals is expected to be minimal, but may result in a loss of reproductive potential in addition to a loss of future recruits.

Despite the potential loss of some reproductive potential resulting from the action, we do not believe that sexually mature colonies of ESA-listed corals will be affected to a degree that will cause short or long-term damage to the ability of each species to sexually reproduce. Therefore, although we believe there may be a small loss of reproductive potential and future colonies, we do not expect that the action will alter the geographic range of the species. We also do not anticipate that the reduction in numbers and reproductive potential resulting from the action would represent a detectable reduction in reproduction of elkhorn, staghorn, pillar, rough cactus, lobed star, mountainous star, and boulder star corals in the portion of the action area that is within their range. No change in distribution of these corals is expected as a result of the action and these species will continue to be present in waters of the action area, including the Miami RAA.

Whether the reduction in numbers and reproduction of ESA-listed coral species would appreciably reduce their likelihood of survival depends on the probable effects these changes would have relative to current population levels and trends. Elkhorn and staghorn coral, pillar coral, rough cactus coral, and all three species from the star coral complex are reported on reefs in Puerto Rico, USVI, and Florida, and corals from the star coral complex are the dominant species in many areas. However, some of these species are now exhibiting symptoms of a relatively new disease, stony coral tissue loss disease that has led to declines in populations of these species on reefs in Florida and now various sites in the Caribbean. As noted previously, we are not able to determine the absolute abundance of pillar and rough cactus coral due to the natural rarity of these species but survey data from around Puerto Rico and USVI indicate that these species are likely more common in the U.S. Caribbean than in Florida. Elkhorn, staghorn, and all three species from the star coral complex are thought to have absolute abundances in the tens of millions based on available data from a few locations such as Florida and St. Croix. Any loss of reproductive potential due to the action will not measurably impact the species' abundance throughout their ranges. Therefore, we believe any loss of colonies and future recruits associated with impacts from the UNDS Batch Two discharges from military vessels will not have any measurable effect on the overall population of ESA-listed coral species and will not appreciably reduce the ability of ESA-listed coral species to survive in the wild.

Next we evaluate whether the expected reduction in reproduction and future recruitment of ESA-listed corals will appreciably reduce the likelihood of the species' recovery in the wild. The recovery plan for elkhorn and staghorn corals notes that elkhorn and staghorn corals continue to decline and are at only a small percentage of their historic abundance throughout their ranges. The recovery plan outlines a recovery strategy for the species as: populations large enough so that successfully reproducing individuals comprise numerous populations across the historic ranges of these species and are large enough to protect their genetic diversity and maintain their ecosystem functions. Threats to these species and their habitat must be sufficiently abated to ensure a high probability of survival into the future (NMFS 2014c). The most relevant recovery criteria to the impacts expected from the action include:

- Ensure Population Viability: Presence of elkhorn and staghorn coral thickets in consolidated reef habitat within forereef zone based on size and density of colonies or live coral benthic cover. Populations with these characteristics should be present throughout the range and maintained for 20 years.
- Recruitment: Observe recruitment rates necessary to achieve Criteria one and two (Genotypic Diversity) over approximately 20 years; and observe effective sexual recruitment (i.e., establishment of new larval derived colonies and survival to sexual maturity) in each species' population across their geographic range.
- Eliminate or Sufficiently Abate Global, Regional, and Local Threats: Address the threat of loss of recruitment habitat through the population viability objective (above or by

maintaining at least 40 percent of the consolidated reef substrate in one – 20 m depth within the forereef free of sediment and macroalgal cover throughout the species' ranges.

- Eliminate or Sufficiently Abate Global, Regional, and Local Threats: Based on five years of data, criteria will be established to reduce sources of nutrients, sediments, and contaminants to levels appropriate for recovery.
- Eliminate or Sufficiently Abate Global, Regional, and Local Threats: Adequate domestic and international regulations and agreements are adopted as necessary to ensure that all threat-based recovery criteria are met.

Pillar, rough cactus, lobed star, mountainous star, and boulder star corals were listed in September 2014, and NMFS does not have an extensive consultation history for these species or a recovery plan. NMFS has developed a recovery outline for these species (available at http://sero.nmfs.noaa.gov/protected_resources/coral/documents/recovery_outline.pdf). The outline is meant to serve as an interim guidance document to direct recovery efforts, including recovery planning, until a full recovery plan is developed and approved. The Summary Assessment in the recovery outline concludes that population trends for rough cactus and pillar coral are unknown but abundance is very low and that populations of star coral species are on the decline. Thus, recovery will depend on successful sexual reproduction and reduction of mortality of existing populations. The key challenges to achieving recovery will be moderating the impacts of ocean warming associated with climate change and decreasing the species' susceptibility to disease, which may be furthered through reduction of local stressors with recovery requiring an ecosystem approach including habitat protection and a reduction in threats caused by human activity.

In terms of the recovery objectives, the action is not expected to reduce the overall abundance of ESA-listed corals though the project is expected to result in a small loss of numbers, particularly in the Miami RAA. The project is expected to result in decreases in reproductive potential and future recruits due to the effects of Batch Two discharges from military vessels containing contaminants or NAS that could become ANS. However, we do not believe there will be an appreciable reduction in the likelihood of recovery in the wild for ESA-listed Atlantic/Caribbean corals. This conclusion is based on the UNDS Phase II Batch Two rule and the performance standards associated with each discharge are meant to reduce discharges to the marine environment by military vessels and will not increase the magnitude of threats that led to listing of all of these coral species. Therefore, we do not believe the action will jeopardize the continued existence of ESA-listed Atlantic/Caribbean corals in the wild.

10.1.4 Johnson's Seagrass

Johnson's seagrass is expected to be adversely affected by the UNDS Batch Two discharges containing contaminants and NAS that may become ANS, which could include plant and algal species that could compete with seagrass for habitat. Discharges are expected to affect growth and habitat quality and availability. In order to determine whether the action is likely to jeopardize the continued existence of Johnson's seagrass, we need to analyze whether the direct and indirect effects of the discharges on the fitness of Johnson's seagrass will appreciably reduce the likelihood of survival and recovery of the species.

Two survey programs have monitored the presence and abundance of Johnson's seagrass within its range. Since the last status review (NMFS 2007a), there has not been any reported reduction in the geographic range of the species but rather a slight increase in the known northern range has been observed (Virnstein and Hall 2009). While the distribution of the species is patchy, in some areas it can cover large areas with a continuous 30-acre patch observed in Lake Worth Lagoon. The population appears to be stable and recent findings of the species outside what was believed to be the northern limit of its range indicate its range is expanding or the complete geographic extent of the species has not yet been determined. Therefore, any reduction in areal coverage as a result of the direct and indirect effects of Batch Two UNDS discharges on Johnson's seagrass is expected to be minimal and will not alter the geographic range of the species.

Based on multiyear surveys, it is believed that Johnson's seagrass does not reproduce sexually. While female flowers have been observed, particularly in areas influenced by the Atlantic Ocean, no male flowers or seeds have ever been observed. This may be due to the male flowers being of small size or being cryptic but has led to the conclusion that the species only reproduces asexually. Therefore, the action will not have any effect on the species ability to sexually reproduce. We do not anticipate that any reductions in cover in the area of the Miami RAA and associated reduction in asexual reproduction resulting from the action would represent a detectable reduction in asexual reproduction of Johnson's seagrass in the portion of the action area that is within their range, namely the area of the Miami RAA. No change in distribution of Johnson's seagrass is expected as a result of the action and the species will continue to be present in waters of the action area, specifically in in the area of the Miami RAA.

Whether a reduction cover of Johnson's seagrass would appreciably reduce the likelihood of survival of the species depends on the probable effects these changes would have relative to current population levels and trends. As noted previously, there are some data indicating the population is stable and that the range may be increasing or is larger than previously thought. While there is no population estimate, the range of the species covers over 22,000 acres so any loss of potential to asexually reproduce due to the action will not measurably impact the species' abundance throughout its range. Therefore, we believe any loss of seagrass cover and future asexual recruits associated with impacts from the UNDS Batch Two discharges from military vessels will not have a measurable effect on the overall population of Johnson's seagrass and will not appreciably reduce the ability of the species to survive in the wild.

Next we evaluate whether the expected reduction in asexual reproduction of Johnson's seagrass will appreciably reduce the likelihood of the species' recovery in the wild. The Final Recovery Plan for Johnson's Seagrass (Dawes et al. ; NMFS 2002) includes the following recovery objectives that are relevant to the effects of the action:

- The species' present geographic range remains stable for at least ten years or increases
- Self-sustaining populations are present throughout the range at distances less than or equal to the maximum dispersal distance to allow for stable vegetative recruitment and genetic diversity
- Populations and supporting habitat in its geographic range have long-term protection (through regulatory action or purchase acquisition).

The action is not expected to reduce the overall abundance of Johnson's seagrass though the project is expected to result in a small loss of cover, particularly in the Miami RAA. The project is expected to result in decreases in asexual reproductive potential due to the effect of Batch Two discharges from military vessels. However, we do not believe there will be an appreciable reduction in the likelihood of recovery in the wild for Johnson's seagrass. This conclusion is based on the UNDS Phase II Batch Two rule and the performance standards associated with each discharge are meant to reduce discharges to the marine environment by military vessels and will not increase the magnitude of threats that led to listing of Johnson's seagrass. Therefore, we do not believe the action will jeopardize the continued existence of Johnson's seagrass in the wild.

10.1.5 Green Sturgeon

As discussed in our effects analysis, UNDS Batch Two discharges from vessels of the armed forces are likely to affect habitat and prey for Southern green sturgeon, as well as bioaccumulating in fish tissues with potential fitness consequences. In order to determine whether the action is likely to jeopardize the continued existence of green sturgeon, we need to analyze whether the indirect effects of the discharges on the fitness of green sturgeon will appreciably reduce the likelihood of survival and recovery of the species.

The need to expend additional effort to find refuge and foraging habitat and prey species can result in reduced growth rates and lower lifetime fecundity and bioaccumulation of contaminants such as heavy metals can result in impacts to sensory organs or reproductive impairment. However, no reduction in the distribution of Southern green sturgeon is expected and the species will continue to be present throughout its range where it currently occurs in the action area, including the San Francisco RAA.

Whether potential effects to reproductive output would appreciably reduce the likelihood of survival of green sturgeon depends on the probable effect the changes in reproductive output would have relative to current population sizes and trends. The estimated total number of adults in the Southern DPS population is 1,348 ± 524 (NMFS 2015e; Mora 2015). Attempts to evaluate the status of Southern DPS green sturgeon have been met with limited success due to the lack of reliable long-term data. The final rule listing Southern DPS green sturgeon indicates that the principle factor for the decline in the DPS is the reduction of spawning to a limited area in the Sacramento River caused primarily by impoundments. Green sturgeon also face threats related to water temperature, water flow, and from commercial and recreational bycatch. If declines in refuge and foraging habitat and prey species associated with UNDS Batch Two discharges or consumption of contaminated prey occur, these could lead to reduced fecundity of Southern green sturgeon populations. However, the UNDS Batch Two rule contains performance standards for each of the discharges regulated under the rule, many of which include limits or prohibitions on discharges occurring while vessels are in harbors and ports and requiring discharges to be to onshore facilities for receiving wastewater, which include facilities for vessels of the armed forces. With the implementation of these performance standards, we believe the action is not reasonably expected to directly or indirectly cause an appreciable reduction in the likelihood of survival of Southern green sturgeon in the wild. The potential reduction in reproduction that could occur as a result of effects from exposure to UNDS Batch Two discharges would not appreciably affect the reproductive output of Southern green sturgeon.

A final recovery plan for Southern DPS green sturgeon is not available but a recovery outline has been prepared. According to the recovery outline, key recovery needs and implementation measures include additional spawning and egg/larval habitat, as well as additional research and monitoring (NMFS 2010a). As noted above, there is no population trend estimate for green sturgeon due to the lack of reliable long-term data. The performance standards for each discharge are meant to reduce the potential for introduction of contaminants and NAS that may become ANS to RAAs and other ports and harbors from Batch Two discharges from vessels of the armed forces. Therefore, any non-lethal take of juvenile or adult Southern green sturgeon is not likely to impede the recovery objectives identified in the recovery outline and will not result in an appreciable reduction in the likelihood of recovery in the wild. We therefore conclude that the action is not likely to jeopardize the continued existence of Southern green sturgeon.

10.1.6 Bocaccio and Yelloweye Rockfish

As discussed in our effects analysis, UNDS Batch Two discharges from vessels of the armed forces are likely to affect habitat and prey for Puget Sound/Georgia Basin bocaccio and yelloweye rockfish, as well as bioaccumulating in fish tissues with potential fitness consequences. In order to determine whether the action is likely to jeopardize the continued existence of bocaccio and yelloweye rockfish (Puget Sound/Georgia Basin DPSs), we need to analyze whether the indirect effects of the discharges on the fitness of bocaccio and yelloweye rockfish will appreciably reduce the likelihood of survival and recovery of these species.

The need to expend additional effort to find refuge and foraging habitat and prey species can result in reduced growth rates and lower lifetime fecundity and bioaccumulation of contaminants such as heavy metals can result in reproductive impairment. However, no reduction in the distribution of Puget Sound/Georgia Basin bocaccio and yelloweye rockfish is expected and the species will continue to be present throughout their range where they currently occur in the action area, particularly the Seattle RAA.

Whether potential effects to reproductive output would appreciably reduce the likelihood of survival of Puget Sound/Georgia Basin bocaccio and yelloweye rockfish depends on the probable effect the changes in reproductive output would have relative to current population sizes and trends. There is no current population abundance estimate for the Puget Sound/Georgia Basin DPS bocaccio. There is a lack of long-term information on this DPS for bocaccio abundance, although among rockfish of the Puget Sound, bocaccio appear to have undergone a particular decline likely because of overfishing and the frequent failure of recruitment classes. In 2008, fishery-independent estimate surveys estimated that 47,407 yelloweye rockfish are present in the San Juan Islands basin. This estimate only includes the San Juan Island basin and so is considered a conservative estimate of Puget Sound/Georgia Basin yelloweye rockfish abundance. Like bocaccio, yelloweye rockfish populations have undergone a decline due largely to overfishing as well as life history characteristics. If declines in refuge and foraging habitat and prey species associated with UNDS Batch Two discharges or consumption of contaminated prey occur, these could lead to reduced fecundity of bocaccio and yelloweye rockfish populations. However, the UNDS Batch Two rule contains performance standards for each of the discharges regulated under the rule, many of which include limits or prohibitions on discharges occurring while vessels are in harbors and ports and requiring discharges be to onshore facilities for receiving wastewater, which include facilities for vessels of the armed forces. With the implementation of these performance standards, we believe the action is not reasonably expected to directly or indirectly cause an appreciable reduction in the likelihood of survival of Puget Sound/Georgia Basin bocaccio and yelloweye rockfish in the wild. The potential reduction in reproduction that could occur as a result of effects from exposure to UNDS Batch Two discharges would not appreciably affect the reproductive output of bocaccio and yelloweye rockfish.

The Draft Rockfish Recovery Plan: Puget Sound/Georgia Basin yelloweye rockfish (*Sebastes ruberrimus*) and bocaccio (*Sebastes paucispinis*; NMFS 2016b) includes the following recovery objectives that are relevant to the action:

- Reduce or eliminate existing threats to listed rockfish from fisheries/anthropogenic mortality.
- Reduce or eliminate existing threats to listed rockfish habitats and restore important rockfish habitat.

As noted above, there are no reliable population trend estimates for these species though data indicate significant declines from historic abundances due largely to fishing pressure. The performance standards for each discharge are meant to reduce the potential for introduction of contaminants and ANS to RAAs and other ports and harbors from Batch Two discharges from vessels of the armed forces. Therefore, any non-lethal take of various life stages of Puget Sound/Georgia Basin bocaccio and yelloweye rockfish is not likely to impede the recovery objectives identified in the draft recovery plan for these species and will not result in an appreciable reduction in the likelihood of recovery in the wild. We therefore conclude the action is not likely to jeopardize the continued existence of Puget Sound/Georgia Basin bocaccio and yelloweye rockfish.

10.1.7 Gulf sturgeon

As discussed in our effects analysis, UNDS Batch Two discharges from vessels of the armed forces are likely to affect habitat and prey for gulf sturgeon, as well as bioaccumulating in fish tissues with potential fitness consequences. In order to determine whether the action is likely to jeopardize the continued existence of gulf sturgeon, we need to analyze whether the indirect effects of the discharges on the fitness of gulf sturgeon will appreciably reduce the likelihood of survival and recovery of the species.

The need to expend additional effort to find refuge and foraging habitat and prey species can result in reduced growth rates and lower lifetime fecundity and bioaccumulation of contaminants such as heavy metals can result in impacts to sensory organs or reproductive impairment. However, no reduction in the distribution of gulf sturgeon is expected and the species will continue to be present throughout its range within the action area, including ports and harbors in the Gulf of Mexico containing facilities for vessels of the armed forces (see Table 1).

Whether potential effects to reproductive output would appreciably reduce the likelihood of survival of gulf sturgeon depends on the probable effect the changes in reproductive output would have relative to current population sizes and trends. Gulf sturgeon abundance trends are typically assessed on a riverine basis for the seven major rivers known to contain reproductive populations. These estimates range from 216 to 14,000 depending on the river. In general, gulf sturgeon populations in the eastern portion of its range appear to be stable or slightly increasing, while populations in the western portion of its range are associated with lower abundances and higher uncertainty (USFWS 2009). If declines in refuge and foraging habitat and prey species associated with UNDS Batch Two discharges or consumption of contaminated prey occur, these could lead to reduced fecundity of gulf sturgeon populations. However, the UNDS Batch Two rule contains performance standards for each of the discharges regulated under the rule, many of which include limits or prohibitions on discharges occurring while vessels are in harbors and ports and requiring discharges be to onshore facilities for receiving wastewater, which include facilities for vessels of the armed forces. With the implementation of these performance standards, we believe the action is not reasonably expected to directly or indirectly cause an appreciable reduction in the likelihood of survival of gulf sturgeon in the wild. The potential reduction in reproduction that could occur as a result of effects from exposure to UNDS Batch Two discharges would not appreciably affect the reproductive output of gulf sturgeon.

The Recovery Plan (USFWS 1995a) included the following recovery objective that is relevant to the action:

- to prevent further reduction of existing wild populations of gulf sturgeon within the range of the subspecies

As noted above, population estimates depend on the river basin where the species occurs and vary widely. Some populations appear to be increasing while a trend cannot be determined for other populations. The performance standards for each discharge are meant to reduce the potential for introduction of contaminants and NAS that may become ANS to RAAs and other ports and harbors from Batch Two discharges from vessels of the armed forces. Therefore, any non-lethal take of juvenile or adult gulf sturgeon is not likely to impede the recovery objectives identified in the recovery plan and will not result in an appreciable reduction in the likelihood of recovery in the wild. We conclude that the action is not likely to jeopardize the continued existence of gulf sturgeon.

10.1.8 Atlantic Sturgeon

As discussed in our effects analysis, UNDS Batch Two discharges from vessels of the armed forces are likely to affect habitat and prey for shortnose sturgeon and Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon, as well as bioaccumulating in fish tissues with potential fitness consequences. In order to determine whether the action is likely to jeopardize the continued existence of Atlantic sturgeon, we need to analyze whether the indirect effects of the discharges on the fitness of shortnose and Atlantic sturgeon will appreciably reduce the likelihood of survival and recovery of the species.

The need to expend additional effort to find refuge and foraging habitat and prey species can result in reduced growth rates and lower lifetime fecundity and bioaccumulation of contaminants such as heavy metals can result in reproductive impairment. However, no reduction in the distribution of shortnose and Atlantic sturgeon is expected. The species will continue to be present throughout their range where they currently occur in the action area, including the Norfolk RAA and other ports and harbors containing facilities for vessels of the armed forces along the east coast.

Whether potential effects to reproductive output would appreciably reduce the likelihood of survival of shortnose and Atlantic sturgeon depends on the probable effect the changes in reproductive output would have relative to current population sizes and trends. The largest shortnose sturgeon adult populations are found in the Northeastern rivers: Hudson 56,708 adults (Bain et al. 2007); Delaware 12,047 (ERC 2006); and Saint Johns greater than 18,000 adults (Dadswell 1979). Shortnose sturgeon populations in southern rivers are considerably smaller by comparison. There are some positive signs for the Gulf of Maine DPS of Atlantic sturgeon, including observations of Atlantic sturgeon in rivers from which they have not been reported for many years and potentially higher catch-per-unit-effort levels than in the past (ASSRT 2007), but long-term abundance is unknown. Precise estimates of population growth rate (intrinsic rates) for the New York Bight DPS of Atlantic sturgeon are unknown due to lack of long-term abundance data. Long-term juvenile surveys indicate that the Hudson River population supports successful annual year classes since 2000 and the annual production has been stable and/or slightly increasing in abundance (ASSRT 2007). The Chesapeake Bay DPS of Atlantic sturgeon once supported at least six historical spawning populations. Today the bay is believed to support at the most four to five spawning populations. Precise estimates of population growth rate (intrinsic rates) for the Chesapeake Bay DPS are unknown due to lack of long-term abundance data. Precise estimates of population growth rate (intrinsic rates) for the Carolina DPS of Atlantic sturgeon are unknown due to lack of long-term abundance data. Precise estimates of population growth rate (intrinsic rates) for the South Atlantic DPS of Atlantic sturgeon are unknown due to lack of long-term abundance data. During the last two decades, Atlantic sturgeon have been observed in most South Carolina coastal rivers, although it is not known if all rivers support a spawning population (Collins 1997).

The UNDS Batch Two rule contains performance standards for each of the discharges regulated under the rule, many of which include limits or prohibitions on discharges occurring while vessels are in harbors and ports and requiring discharges to be to onshore facilities for receiving wastewater, which include facilities for vessels of the armed forces. With the implementation of these performance standards, we believe the action is not reasonably expected to directly or indirectly cause an appreciable reduction in the likelihood of survival of shortnose and Atlantic sturgeon DPSs in the wild. The potential reduction in reproduction that could occur as a result of effects from exposure to UNDS Batch Two discharges would not appreciably affect the reproductive output of shortnose and Atlantic sturgeon.

The Shortnose Sturgeon Recovery Plan (NMFS 1998) has the following long-term recovery objective:

- to recover all 19 discrete populations to levels of abundance at which they no longer require protection under the ESA.

No recovery plans are available for the DPSs of Atlantic sturgeon.

As noted above, there are no reliable population trend estimates for these species though data indicate potentially stable populations in some of the rivers where they are found. Data on populations in marine habitats are extremely limited, making population estimates even more difficult. The performance standards for each discharge are meant to reduce the potential for introduction of contaminants and ANS to RAAs and other ports and harbors from Batch Two discharges from vessels of the armed forces. Therefore, any non-lethal take of juvenile and adult life stages of shortnose and Atlantic sturgeon is not likely to impede the recovery objectives identified in the recovery plan for shortnose sturgeon and will not result in an appreciable reduction in the likelihood of recovery in the wild of either species. We therefore conclude the action is not likely to jeopardize the continued existence of shortnose sturgeon and Atlantic sturgeon (Gulf of Maine, New York Bight, Carolina, Chesapeake Bay, and South Atlantic DPSs).

10.1.9 Pacific Salmonids

As discussed in our effects analysis, UNDS Batch Two discharges from vessels of the armed forces are likely to affect habitat and prey for Central Valley Spring-Run, Puget Sound, and Sacramento River Winter-Run ESUs of chinook salmon; Hood Canal Summer-Run ESU chum salmon; Central California Coast ESU coho salmon; and California Central Valley, Central California Coast, and Puget Sound ESU steelhead trout. Contaminants in discharges may also bioaccumulate in fish tissues with potential fitness consequences. In order to determine whether the action is likely to jeopardize the continued existence of ESA-listed ESUs and DPSs of Pacific salmonids, we need to analyze whether the indirect effects of the discharges on the fitness of these ESUs of chinook, chum, and coho salmon, and DPSs of steelhead trout will appreciably reduce the likelihood of survival and recovery of these species.

The need to expend additional effort to find refuge and foraging habitat and prey species can result in reduced growth rates and lower lifetime fecundity and bioaccumulation of contaminants such as heavy metals can result in reproductive impairment. However, no reduction in the distribution of Pacific salmonids is expected and the species will continue to be present throughout their range where they currently occur in the action area, including the San Francisco and Puget Sound RAAs and other ports and harbors containing facilities for vessels of the armed forces along the west coast.

Whether potential effects to reproductive output would appreciably reduce the likelihood of survival of these ESUs of the species depends on the probable effect the changes in reproductive output would have relative to current population sizes and trends. The Central Valley Spring-run chinook salmon ESU now has only three known streams that currently support self-sustaining populations of non-hybridized spring-run chinook salmon in the Central Valley. Abundance and trend estimates for these streams, as well as streams supporting dependent populations, indicate population declines in many of the reaches (NMFS 2014b). Current estimates of diversity in the Puget Sound ESU of chinook salmon show a decline over the past 25 years, indicating a decline of salmon in some areas and increases in others. In aggregate, the diversity of the ESU as a whole has been declining over the last 25 years. Over the last ten years of available data (2003-2013) for the Sacramento River Winter-run ESU of chinook salmon, the abundance of spawning winter-run chinook adults ranged from a low of 738 in 2011 to a high of 17,197 in 2007, with an average of 6,298 (NMFS 2011b). Like many other populations of Chinook salmon in the Central

Valley, the Sacramento River winter-run Chinook salmon ESU has declined in abundance since 2005 and the 10-year trend in abundance is negative. The two most recent status reviews indicate some positive signs for the Hood Canal Summer-run chum salmon ESU. Productivity rates increased from 2011-2015 and were greater than replacement rates from 2014-2015 for both major population groups (NMFS NWFSC 2015) of this ESU. Data for the Central California Coast ESU of coho salmon suggest some populations show a slight positive trend in annual escapement, but the improvement is not statistically significant. Overall, all populations remain, a slight fraction of their recovery target levels with the estimated number of coho salmon produced within the ESU in 2011 between 2,000 and 3,000 wild adults (Gallagher et al. 2010). The California Central Valley steelhead trout DPS indicates that, over the past 30 years, the naturally spawned steelhead populations in the upper Sacramento River have declined substantially. Based on catch ratios at Chipps Island in the Delta and assumptions regarding survival, the average number of Central Valley steelhead females spawning naturally in the entire Central Valley during the years 1980 to 2000 was estimated at about 3,600 (Good et al. 2005). Though the information for individual populations is limited, available information strongly suggests that no population is viable for the Central California Coast steelhead trout DPS. The interior Russian River winter-run steelhead has the largest runs with an estimate of an average of over 1,000 spawners; it may be able to be sustained over the long-term but hatchery management has eroded the population’s genetic diversity (Bjorkstedt et al. 2005; NMFS 2008a). For the Puget Sound steelhead trout DPS, abundance of adult steelhead returning to nearly all Puget Sound Rivers has fallen substantially since estimates began for many populations in the late 1970s and early 1980s. Smoothed trends in abundance indicate modest increases since 2009 for 13 of the 22 populations. However, several of these upward trends are not statistically different from neutral, and most populations remain small.

The UNDS Batch Two rule contains performance standards for each of the discharges regulated under the rule, many of which include limits or prohibitions on discharges occurring while vessels are in harbors and ports and requiring discharges to be to onshore facilities for receiving wastewater, which include facilities for vessels of the armed forces. With the implementation of these performance standards, we believe the action is not reasonably expected to directly or indirectly cause an appreciable reduction in the likelihood of survival in the wild of chinook, chum, and coho salmon ESUs and steelhead trout DPSs located in RAAs and other ports and harbors along the West coast with concentrations of military vessels. The potential reduction in reproduction that could occur as a result of effects from exposure to UNDS Batch Two discharges would not appreciably affect the reproductive output of these ESUs and DPSs of Pacific salmonid species.

The recovery plan for Central Valley spring-run chinook (NMFS 2014b) contains the following recovery objective that is relevant to the action:

- Maintain multiple populations at moderate risk of extinction.

The recovery plan for the Puget Sound ESU of chinook salmon consists of two documents with the following recovery criteria relevant to the action:

- The viability status of all populations in the ESU is improved from current conditions, and when considered in the aggregate, persistence of the ESU is assured;
- At least one population from each major genetic and life history group historically present within each of the five biogeographical regions is viable;
- Populations that do not meet the viability criteria for all parameters are sustained to provide ecological functions and preserve options for ESU recovery.

The recovery plan for Sacramento River winter-run chinook (NMFS 2014b) contains the following recovery objectives relevant to the action:

- In order to achieve the downlisting criteria, the species would need to be composed of two populations – one viable and one at moderate extinction risk.

The recovery plan for Hood Canal Summer-run chum salmon (NMFS 2007b) includes the following recovery priorities relevant to the action:

- protecting the functioning habitat and major production areas of the ESU's eight extant stocks, keeping in mind the biological and habitat needs of different life-history stages
- restoration of degraded areas, where recovery of natural processes appears to be feasible (HCCC 2005).

The recovery plan for coho salmon (NMFS 2012d) contains the following recovery goals relevant to the action:

- Prevent extinction by protecting existing populations and their habitats
- Maintain and restore suitable freshwater and estuarine habitat conditions and characteristics for all life history stages so viable populations can be sustained naturally
- Ensure all factors that led to the listing of the species have been ameliorated
- Develop and maintain a program of monitoring, research, and evaluation that advances understanding of the complex array of factors associated with coho salmon survival and recovery and which allows for adaptively managing our approach to recovery over time.

The recovery plan for the California Central Valley steelhead DPS (NMFS 2014b) contains the following recovery goals relevant to the action:

- Maintain multiple populations at moderate risk of extinction

The recovery plan for the Central California Coast steelhead DPS (NMFS 2016a) contains the following recovery plan objectives relevant to the action:

- Reduce the present or threatened destruction, modification, or curtailment of habitat or range;
- Establish the adequacy of existing regulatory mechanisms for protecting Central California Coast steelhead now and into the future (i.e., post-delisting);
- Address other natural or manmade factors affecting the continued existence of Central California Coast steelhead.

A recovery plan is not available for the Puget Sound DPS of steelhead trout.

As noted previously, there are no reliable population trend estimates for these species though data indicate dramatic declines from historic abundances with some populations likely not to be viable. The performance standards for each discharge are meant to reduce the potential for introduction of contaminants and ANS to RAAs and other ports and harbors from Batch Two discharges from vessels of the armed forces. Therefore, any non-lethal take of juvenile and adult life stages of Pacific salmonids is not likely to impede the recovery objectives identified in the recovery plans for those species that have one and will not result in an appreciable reduction in the likelihood of recovery in the wild of any of the species. We therefore conclude that the action is not likely to jeopardize the continued existence of Central Valley Spring-Run, Puget Sound, and Sacramento River Winter-Run chinook salmon; Hood Canal Summer-Run chum salmon; Central California Coast coho salmon; and California Central Valley, Central California Coast, and Puget Sound steelhead trout.

10.1.10Atlantic Salmon

As discussed in our effects analysis, UNDS Batch Two discharges from vessels of the armed forces are likely to affect habitat and prey for Gulf of Maine Atlantic salmon, as well as bioaccumulating in fish tissues with potential fitness consequences. In order to determine whether the action is likely to jeopardize the continued existence of Atlantic salmon, we need to determine whether the indirect effects of the discharges on the fitness of Gulf of Maine Atlantic salmon will appreciably reduce the likelihood of survival and recovery of the species.

The need to expend additional effort to find refuge and foraging habitat and prey species can result in reduced growth rates and lower lifetime fecundity and bioaccumulation of contaminants such as heavy metals can result in reproductive impairment. However, no reduction in the distribution of Gulf of Maine Atlantic salmon is expected and the species will continue to be present throughout its range where it currently occurs in the action area, particularly in the Gulf of Maine where facilities for vessels of the armed forces are located (Table 1).

Whether potential effects to reproductive output would appreciably reduce the likelihood of survival of Gulf of Maine Atlantic salmon depends on the probable effect the changes in reproductive output would have relative to current population sizes and trends. In 2015, four million juvenile salmon (eggs, fry, parr and smolts) and 4,271 adults were stocked in the Connecticut, Merrimack, Saco, Penobscot and five other coastal rivers in Maine (USASAC

2016) from hatcheries. The total number of returns to U.S. rivers was 921, and the majority (80 percent) of the adult returns were of hatchery origin. Adult returns of Gulf of Maine DPS Atlantic salmon captured in six Maine rivers from 1997 to 2004 ranged from 567 to 1,402 including both wild and hatchery origin fishes. There is no population growth rate available for Gulf of Maine DPS Atlantic salmon. However, the consensus is that the DPS exhibits a continuing declining trend (NOAA 2016).

If declines in refuge and foraging habitat and prey species associated with UNDS Batch Two discharges or consumption of contaminated prey occur, these could lead to reduced fecundity of the Gulf of Maine Atlantic salmon DPS population. However, the UNDS Batch Two rule contains performance standards for each of the discharges regulated under the rule, many of which include limits or prohibitions on discharges occurring while vessels are in harbors and ports and requiring discharges to be to onshore facilities for receiving wastewater, which include facilities for vessels of the armed forces. With the implementation of these performance standards, we believe the action is not reasonably expected to directly or indirectly cause an appreciable reduction in the likelihood of survival of the Gulf of Maine DPS of Atlantic salmon in the wild. The potential reduction in reproduction that could occur as a result of effects from exposure to UNDS Batch Two discharges would not appreciably affect the reproductive output of Gulf of Maine Atlantic salmon.

The Draft Recovery Plan for the Gulf of Maine DPS Atlantic Salmon (USFWS and NMFS 2019) contains the following recovery actions that are relevant to the action:

- Increase Atlantic salmon survival through increased ecosystem understanding and identification of spatial and temporal constraints to salmon marine productivity to inform and support management actions that improve survival
- Collaborate with partners and engage interested parties in recovery efforts for the Gulf of Maine DPS.

As noted above, data indicate a declining population of this DPS, though there is no population growth rate available due to limited data. The performance standards for each discharge are meant to reduce the potential for introduction of contaminants and NAS that may become ANS to RAAs and other ports and harbors from Batch Two discharges from vessels of the armed forces. Therefore, any non-lethal take of juvenile or adult Gulf of Maine DPS Atlantic salmon is not likely to impede the recovery objectives identified in the recovery outline and will not result in an appreciable reduction in the likelihood of recovery in the wild. We therefore conclude the action is not likely to jeopardize the continued existence of Gulf of Maine DPS Atlantic salmon.

10.2 Destruction and Adverse Modification of Designated Critical Habitat

When determining the potential impacts to critical habitat for this opinion, NMFS relies on the regulatory definition of “destruction or adverse modification” of critical habitat from the final rule issued by NMFS and USFWS (81 FR 7214, February 11, 2016). Under the final rule, destruction or adverse modification 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.

Ultimately, we seek to determine if, with the implementation of the actions, critical habitat would remain functional (or retain the current ability for the essential features to become functional) to serve the intended conservation role for the species with the implementation of the action, or whether the conservation function and value of critical habitat is appreciably diminished through alterations to the physical or biological features essential to the conservation of a species or because of significant delays in the development of these features. This analysis takes into account the geographic and temporal scope of the action, recognizing that "functionality" of critical habitat necessarily means that it must now and must continue in the future to support the conservation of the species and progress toward recovery. The analysis must take into account any changes in amount, distribution, or characters of the critical habitat that will be required over time to support the successful recovery of the species.

Based on our effects analysis, discharges from underwater ship husbandry containing NAS with the potential to become ANS and discharges from underwater ship husbandry containing metals such as copper that will be released to the water column and sorb to sediments in the RAAs and other ports and harbors with facilities for vessels of the armed forces may adversely affect designated critical habitats.

Miami RAA

For the Miami RAA, we find it reasonably likely that the following designated critical habitats would be adversely affected by UNDS Phase II Batch Two discharges as a result of the action: elkhorn and staghorn coral critical habitat and Johnson's seagrass critical habitat. Elkhorn and staghorn coral critical habitat in the Florida and Puerto Rico units could be affected by Batch Two discharges from military vessels at facilities in ports and harbors within these units as well.

The key objective for the conservation and recovery of elkhorn and staghorn corals that is the basis for the critical habitat designation is the facilitation of an increase in the incidence of sexual and asexual reproduction. Recovery cannot occur without protecting the essential feature of coral critical habitat from destruction or adverse modification. Anthropogenic stressors have the greatest impact on habitat quality for elkhorn and staghorn corals because degradation or loss of substrate for sexual and asexual recruits to settle impacts reproductive success, growth, and recovery of these species. The essential feature of critical habitat for elkhorn and staghorn coral is substrate of adequate quantity and quality to allow for settlement and growth where adequate quality refers to the need for hard substrate to be free of high macroalgal growth and sediment cover as these impede the settlement and growth of elkhorn and staghorn corals. Thus, we need to assess whether the discharges and associated contaminant and ANS stressors on elkhorn and staghorn coral critical habitat rise to the level of adversely modifying or destroying the designated critical habitat when considered as a whole. While our analysis indicated that some ANS and contaminants could lead to changes in quality of elkhorn and staghorn coral critical habitat, such as through growth of algae (ANS or due to nutrients in discharges promoting algal growth), these effects are likely to be localized and are not expected to result in the loss or degradation of a large area containing the essential feature in the Florida or Puerto Rico critical habitat units. We base this on the current presence of elkhorn and staghorn corals, including in areas such as the Port of Miami and San Juan Harbor in the action area, which are active ports, and the required performance standards meant to reduce the potential for environmental impacts from discharges. Therefore, we do not expect the action will appreciably diminish the overall value of elkhorn and staghorn coral critical habitat for the conservation of elkhorn and staghorn corals. We conclude that the action will not result in the destruction or adverse modification of elkhorn and staghorn coral critical habitat in the action area.

The key objective for the conservation and recovery of Johnson's seagrass is the maintenance of existing populations, including those where female flowers have been observed. Recovery cannot occur without the maintenance of the essential features including water quality and water transparency, salinity, and stable sediments. Anthropogenic stressors are the greatest threats to these features from both land-based and in-water activities that impact water quality, salinity, water clarity, and bottom substrates. Thus, we need to assess whether the discharges and associated contaminant and ANS stressors on Johnson's seagrass critical habitat rise to the level of adversely modifying or destroying the designated critical habitat when considered as a whole. Our analysis indicated that some ANS and contaminants could lead to changes in quality of Johnson's seagrass critical habitat, such as through competition with the seagrass for bottom substrate (ANS). These effects are likely to be localized and are not expected to result in the loss or degradation of a large area containing the essential feature in the range of Johnson's seagrass critical habitat. We base this on the current presence of Johnson's seagrass, including in areas such as the Port of Miami, which is part of the designated critical habitat of the species and the required performance standards meant to reduce the potential for environmental impacts from discharges. Therefore, we do not expect the action will appreciably diminish the overall value of Johnson's seagrass critical habitat for the conservation of the species. We conclude that the action will not result in the destruction or adverse modification of Johnson's seagrass critical habitat in the action area.

Norfolk RAA

We find it reasonably likely that the following designated critical habitat would be adversely affected by UNDS Batch Two discharges in the Norfolk RAA: Chesapeake Bay DPS Atlantic sturgeon.

While there is no recovery plan for the Chesapeake Bay DPS Atlantic sturgeon or other DPSs of this species, essential features have been identified for Atlantic sturgeon reproduction and recruitment. These features include hard bottom substrate in low salinity waters for development; transitional salinity zones for juveniles; water of appropriate depth and absent physical barriers to passage; unimpeded movement of adults to and from spawning sites; and water quality conditions that support spawning, survival, growth, development, and recruitment. Anthropogenic stressors such as land-based and in-water activities that affect habitat and water quality, as well as the construction of barriers to movement such as dams, particularly in freshwater streams, are the greatest threats to maintenance of these essential features. Thus, we need to assess whether the discharges and associated contaminant and ANS stressors on Chesapeake Bay DPS Atlantic sturgeon critical habitat rise to the level of adversely modifying or destroying the designated critical habitat when considered as a whole. Although some of the ANS and contaminants in UNDS Batch Two discharges from hull coating leachate and underwater ship husbandry have the potential to affect habitats in the port areas where military vessels are located, these effects will occur outside most of the designated critical habitat areas in Chesapeake Bay. The transport of contaminated water or sediment could occur in the bay, leading to changes in quality of critical habitat for Atlantic sturgeon. These effects are likely to be localized and are not expected to result in the loss or degradation of a large area containing the essential feature of Chesapeake Bay DPS Atlantic sturgeon critical habitat. We base this on the location of military vessels in the bay in relation to the location of critical habitat, the presence of sturgeon in the bay despite the long-term presence of military vessels, and the required performance standards meant to reduce the potential for discharges to affect ESA-listed species and their habitats. Therefore, we do not expect the action will appreciably diminish the overall value of the Chesapeake Bay DPS Atlantic sturgeon critical habitat for the conservation of the species. We conclude that the action will not result in the destruction or adverse modification of the Chesapeake Bay DPS Atlantic sturgeon critical habitat in the action area.

Seattle RAA

In the Seattle RAA, we find it reasonably likely that the following designated critical habitats would be adversely affected: Southern resident killer whale critical habitat; green sturgeon; Puget Sound/Georgia Basin bocaccio and yelloweye rockfish critical habitat; Puget Sound ESU chinook salmon critical habitat; Hood Canal Summer-Run chum salmon critical habitat; and Puget Sound DPS steelhead trout critical habitat.

The recovery plan for Southern Resident killer whales identifies the need to ensure an adequate prey base, particularly of salmon species that are key prey of killer whales in the Puget Sound area, and the cleanup of contaminated sites and minimization of continuing inputs of contaminants harmful to killer whales. Critical habitat in the Summer Core Area in Haro Strait and waters around the San Juan Islands, Puget Sound, and the Strait of Juan de Fuca contains essential features of water quality, prey species, and inter-area passage. The Puget Sound Naval

Shipyard at Naval Base Kitsap was excluded from critical habitat designation for Southern Resident DPS killer whale. Anthropogenic stressors, particularly overfishing leading to declines in key prey species and water quality contamination from land-based and in-water activities are threats to the maintenance of the essential features. Thus, we need to assess whether the discharges and associated contaminant and ANS stressors on Southern Resident killer whales in the Seattle RAA rise to the level of adversely modifying or destroying the designated critical habitat when considered as a whole. Although some of the ANS and contaminants in UNDS Batch Two discharges from hull coating leachate and underwater ship husbandry have the potential to affect habitats in the port areas where military vessels are located, a portion of this area (the Puget Sound Naval Shipyard at Naval Base Kitsap) was excluded from the designated critical habitat. Other areas of critical habitat, such as those in the San Juan Islands occur outside the area where military vessels are concentrated in existing facilities. The transport of contaminated water or sediment to critical habitat areas could occur, leading to changes in quality of critical habitat, particularly prey, for Southern Resident killer whale. These effects are likely to be localized and are not expected to result in the loss or degradation of a large area containing the essential features of critical habitat in Puget Sound in particular. We base this on the location of military vessels in the Puget Sound area in relation to critical habitat and the required performance standards meant to reduce the potential for discharges to affect ESA-listed species and their habitats. Therefore, we do not expect the action will appreciably diminish the overall value of the Southern Resident killer whale critical habitat for the conservation of the species. We conclude that the action will not result in the destruction or adverse modification of the Southern Resident DPS killer whale critical habitat in the action area.

The recovery outline for Southern DPS green sturgeon identifies additional spawning and egg/larval habitat as a key recovery need. Designated critical habitat for green sturgeon includes freshwater, bays and estuarine waters, and marine waters to the 110 m depth isobaths from Monterey Bay, California to waters in the Strait of Juan de Fuca, Washington. For estuarine habitats, essential features include food resources, water flow, water quality, and sediment quality. For nearshore marine habitats, essential features include water quality and food resources. Sites owned or controlled by the DoD in the Strait of Juan de Fuca, namely the naval air-to surface weapon range, restricted area; Strait of Juan de Fuca and Whidbey Island naval restricted area; Admiralty Inlet naval restricted area, and Navy three operating area, are excluded from critical habitat designation. Anthropogenic stressors including water quality contamination from land-based and in-water activities, declines in prey species, declines in sediment quality, and changes in hydrology are threats to the maintenance of the essential features. Thus, we need to assess whether the hull coating leachate and underwater ship husbandry discharges and associated contaminant and ANS stressors on green sturgeon critical habitat in the Seattle RAA rise to the level of adversely modifying or destroying the designated critical habitat when considered as a whole. Although some of the ANS and contaminants in UNDS Batch Two discharges have the potential to affect habitats in the port areas where military vessels are located, most of these areas are excluded from the designated critical habitat. The transport of contaminated water or sediment to critical habitat areas could occur, leading to changes in quality of critical habitat, including water and sediment quality and prey, for Southern DPS green sturgeon. These effects are likely to be localized and are not expected to result in the loss or degradation of a large area containing the essential features of critical habitat in the Strait of Juan de Fuca or other nearshore marine areas designated as critical habitat for Southern DPS green sturgeon. We base this on the location of military vessels in the Seattle RAA in relation to critical habitat and the required performance standards meant to reduce the potential for discharges to affect ESA-listed species and their habitats. Therefore, we do not expect the action will appreciably diminish the overall value of the Southern DPS green sturgeon critical habitat for the conservation of the species. We conclude that the action will not result in the destruction or adverse modification of the Southern DPS green sturgeon critical habitat in the action area, particularly in the area of the Seattle RAA.

The recovery objectives for the Puget Sound/Georgia Basin DPSs of bocaccio and yelloweye rockfish include reducing or eliminating existing threats to these species from anthropogenic mortality and reducing or eliminating threats to habitats. Designated critical habitat includes nearshore units to support juveniles, particularly for bocaccio as yelloweye rockfish juveniles tend to use deeper waters, and deeper rocky habitat to support adults of both species. Essential features include sufficient prey resources, water quality, and habitat. The Puget Sound Naval Shipyard at Naval Base Kitsap was excluded from critical habitat designation. Anthropogenic stressors such as water quality contamination from land-based and in-water activities leading to declines in prey and habitat quality and quantity are threats to the maintenance of the essential features. Thus, we need to assess whether the discharges and associated contaminant and ANS stressors on Puget Sound/Georgia Basin bocaccio and yelloweye rockfish in the Seattle RAA rise to the level of adversely modifying or destroying the designated critical habitat when considered as a whole. Although some of the ANS and contaminants in UNDS Batch Two discharges from hull coating leachate and underwater ship husbandry have the potential to affect habitats in the areas where military vessels are located, a portion of this area (the Puget Sound Naval Shipyard at Naval Base Kitsap) was excluded from the designated critical habitat. Other areas containing juvenile and adult critical habitat occur outside the area where military vessels are concentrated in existing facilities in Puget Sound. The transport of contaminated water or sediment to critical habitat areas could occur, leading to changes in essential features of water quality, prey, and habitat. These effects are likely to be localized and are not expected to result in the loss or degradation of a large area containing the essential features of critical habitat in Puget Sound in particular. We base this on the location of military vessels in the Puget Sound area in relation to critical habitat and the required performance standards meant to reduce the potential for discharges to affect ESA-listed species and their habitats. Therefore, we do not expect the action will appreciably diminish the overall value of the Puget Sound/Georgia Basin DPSs bocaccio and yelloweye rockfish critical habitat for the conservation of the species. We conclude that the action will not result in the destruction or adverse modification of the Puget Sound/Georgia Basin bocaccio and yelloweye rockfish critical habitat in the action area.

Recovery of the Puget Sound ESU chinook salmon requires improving the viability of existing populations within the ESU, including through maintenance of groups with genetic and life history differences. Recovery of the Hood Canal Summer-Run ESU chum salmon requires protecting the functioning habitat and major production areas of the ESU's existing stocks in keeping with the biological and habitat needs of different life stages. In order to achieve recovery, critical habitat was designated for chinook and chum salmon and steelhead trout in freshwater, estuarine, nearshore marine, and offshore marine areas. The essential features include water quality and quantity, natural cover, and forage for various life stages. The Puget Sound Naval Shipyard at Naval Base Kitsap was excluded from critical habitat designations for Puget Sound chinook salmon and Puget Sound steelhead trout. Anthropogenic stressors such as water quality contamination from land-based and in-water activities leading to declines in habitat quality and quantity are threats to the maintenance of the essential features. Thus, we need to assess whether the discharges and associated contaminant and ANS stressors on Puget Sound chinook salmon and steelhead trout and Hood Canal Summer-run chum salmon in the Seattle RAA rise to the level of adversely modifying or destroying the designated critical habitats for these species when considered as a whole. Although some of the ANS and contaminants in UNDS Batch Two discharges from hull coating leachate and underwater ship husbandry have the potential to affect habitats in the areas where military vessels are located, a portion of this area (the Puget Sound Naval Shipyard at Naval Base Kitsap) was excluded from the designated critical habitat. Other areas containing juvenile and adult critical habitat occur outside the area where military vessels are concentrated in existing facilities in Puget Sound. None of the freshwater critical habitat areas are expected to be affected though the transport of contaminated water or sediment to estuarine areas containing designated critical habitat for these species could occur, leading to changes in essential features of water quality, natural cover, and forage. These effects are likely to be localized and are not expected to result in the loss or degradation of a large area containing the essential features of critical habitats for these Pacific salmonids in Puget Sound in particular. We base this on the location of military vessels in the Puget Sound area in relation to critical habitat and the required performance standards meant to reduce the potential for discharges to affect ESA-listed species and their habitats. Therefore, we do not expect the action will appreciably diminish the overall value of the Puget Sound ESU chinook salmon, Hood Canal Summer-run chum salmon, and Puget Sound DPS steelhead trout critical habitat for the conservation of these species. We conclude that the action will not result in the destruction or adverse modification of the Puget Sound ESU chinook salmon, Hood Canal Summer-run chum salmon, and Puget Sound DPS steelhead trout critical habitat in the action area.

San Francisco RAA

We find it reasonably likely that the following designated critical habitats in the San Francisco RAA would be adversely affected by UNDS Batch Two discharges: green sturgeon critical habitat; Central Valley Spring-Run and Sacramento River Winter-Run chinook salmon critical habitats; Central California Coast coho salmon critical habitat; and California Central Valley and Central California Coast steelhead trout critical habitats. Black abalone critical habitat along other portions of the California coast where facilities for military vessels are present in ports and harbors may also be adversely affected by discharges.

As discussed above for the Seattle RAA, according to the recovery outline for Southern DPS green sturgeon, a key recovery need is additional spawning and egg/larval habitat, which are not the habitats of green sturgeon where military vessels are present in the San Francisco RAA. Designated critical habitat for green sturgeon includes freshwater, bays and estuarine waters, and marine waters to the 110 m depth isobaths from Monterey Bay, California to waters in the Strait of Juan de Fuca, Washington. For estuarine habitats and bays, including the San Francisco Bay Estuary, essential features include food resources, water flow, water quality, and sediment quality. For nearshore marine habitats, essential features include water quality and food resources. Sites owned or controlled by the DoD in the San Francisco RAA, namely Mare Island U.S. Army Reserve Center, San Pablo Bay, are excluded from critical habitat designation. We need to assess whether the discharges and associated contaminant and ANS stressors on green sturgeon critical habitat in the San Francisco RAA rise to the level of adversely modifying or destroying the designated critical habitat when considered as a whole. Some of the ANS and contaminants in UNDS Batch Two discharges from hull coating leachate and underwater ship husbandry have the potential to affect habitats in the port areas where military vessels are located, although one of these areas is excluded from the designated critical habitat. The transport of contaminated water or sediment to critical habitat areas could occur, leading to changes in quality of critical habitat, including water and sediment quality and prey, for Southern DPS green sturgeon. These effects are likely to be localized and are not expected to result in the loss or degradation of a large area containing the essential features of critical habitat in the San Francisco RAA, or other nearshore marine areas designated as critical habitat for Southern DPS green sturgeon. We base this on the required performance standards meant to reduce the potential for discharges to affect ESA-listed species and their habitats. Therefore, we do not expect the action will appreciably diminish the overall value of the Southern DPS green sturgeon critical habitat for the conservation of the species. We conclude that the action will not result in the destruction or adverse modification of the Southern DPS green sturgeon critical habitat in the action area, particularly in the area of the San Francisco RAA.

Recovery of the Central Valley Spring-run ESU chinook salmon requires maintaining multiple populations at moderate risk of extinction. Recovery of the Sacramento River Winter-run ESU chinook salmon requires two populations of the species with one viable and one at moderate extinction risk. Coho salmon recovery requires protection of existing populations and their habitats, maintaining and restoring freshwater and estuarine habitat conditions and characteristics for all life stages, and ameliorating factors that led to the listing of the species. Recovery of the California Central Valley DPS steelhead trout requires maintaining multiple populations at moderate risk of extinction. Recovery of the Central California Coast DPS steelhead trout requires reducing the present or threatened destruction, modification, or curtailment of habitat or range. In order to achieve recovery, critical habitat was designated for these ESUs of chinook and coho salmon and DPSs of steelhead trout in freshwater, estuarine, nearshore marine, and offshore marine areas. The essential features for Sacramento River Winter-run chinook salmon include habitat and adequate prey free of contaminants and access of juveniles downstream from the spawning grounds to San Francisco Bay and the Pacific Ocean. The essential features for coho salmon include substrate, water quality, water quantity, cover/shelter, and food. The essential features for California Central Valley and Central California Coast steelhead trout and Central Valley Spring-run chinook salmon include water quality and quantity, natural cover, and forage for various life stages. Anthropogenic stressors such as water quality contamination from land-based and in-water activities leading to declines in habitat quality and quantity and prey are threats to the maintenance of the essential features. Thus, we need to assess whether the discharges and associated contaminant and ANS stressors on Central Valley Spring-run and Sacramento River Winter-run chinook salmon, Central California Coast coho salmon, and California Central Valley and Central California Coast steelhead trout in the San Francisco RAA rise to the level of adversely modifying or destroying the designated critical habitats for these species when considered as a whole. Although some of the ANS and contaminants in UNDS Batch Two discharges from hull coating leachate and underwater ship husbandry have the potential to affect habitats in the areas where military vessels are located, much of the designated critical habitat areas are outside locations with concentrations of military vessels. None of the freshwater critical habitat areas are expected to be affected though the transport of contaminated water or sediment to estuarine areas containing designated critical habitat for these species could occur, leading to changes in essential features of water quality, natural cover, and forage. These effects are likely to be localized and are not expected to result in the loss or degradation of a large area containing the essential features of critical habitats for these Pacific salmonids in San Francisco Bay in particular. We base this on the location of military vessels in the San Francisco Bay area in relation to critical habitat and the required performance standards meant to reduce the potential for discharges to affect ESA-listed species and their habitats. Therefore, we do not expect the action will appreciably diminish the overall value of the Central Valley Spring-run and Sacramento River Winter-run chinook salmon, Central California Coast coho salmon, and California Central Valley and Central California Coast steelhead trout critical habitat for the conservation of these species. We conclude that the action will not result in the destruction or adverse modification of the Central Valley Spring-run and Sacramento River Winter-run chinook salmon, Central California Coast coho salmon, and California Central Valley and Central California Coast steelhead trout critical habitat in the action area.

Other Ports and Harbors with Facilities for Vessels of the Armed Forces

Other critical habitats that are outside the RAAs selected by the Navy and EPA to analyze the potential effects of Batch Two discharges to ESA-listed species and their habitats are within ports and harbors with military vessels (see Table 1). These include Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs sturgeon critical habitats; gulf sturgeon critical habitat; and Gulf of Maine Atlantic salmon critical habitat.

There is no recovery plan for Atlantic sturgeon DPSs, but essential features have been identified for Atlantic sturgeon reproduction and recruitment as discussed above for the Chesapeake Bay DPS. These features include hard bottom substrate in low salinity waters for and development; transitional salinity zones for juveniles; water of appropriate depth and absent physical barriers to passage; unimpeded movement of adults to and from spawning sites; and water quality conditions that support spawning, survival, growth, development, and recruitment. Anthropogenic stressors such as land-based and in-water activities that affect habitat and water quality, as well as the construction of barriers to movement such as dams, particularly in freshwater streams, are the greatest threats to maintenance of these essential features. Thus, we need to assess whether the discharges and associated contaminant and ANS stressors on Gulf of Maine, New York Bight, Carolina, and South Atlantic DPSs Atlantic sturgeon critical habitat in areas with ports and harbors with facilities for military vessels (see Table 1) rise to the level of adversely modifying or destroying the designated critical habitat when considered as a whole. Although some of the ANS and contaminants in UNDS Batch Two discharges from hull coating leachate and underwater ship husbandry have the potential to affect habitats in the port areas where military vessels are located, these effects are expected to occur outside most of the designated critical habitat areas in these DPSs because large portions of critical habitat are in streams and rivers. The transport of contaminated water or sediment to critical habitat areas could occur, leading to changes in quality of critical habitat for Atlantic sturgeon. These effects are likely to be localized and are not expected to result in the loss or degradation of a large area containing the essential feature of Atlantic sturgeon critical habitat in the Gulf of Maine, New York Bight, Carolina, and South Atlantic DPSs. We base this on the location of military vessels in ports and harbors on the east coast in relation to the location of critical habitat and the required performance standards meant to reduce the potential for discharges to affect ESA-listed species and their habitats. Therefore, we do not expect the action will appreciably diminish the overall value of the Gulf of Maine, New York Bight, Carolina, and South Atlantic DPSs Atlantic sturgeon critical habitat for the conservation of the species. We conclude that the d action will not result in the destruction or adverse modification of the Gulf of Maine, New York Bight, Carolina, and South Atlantic DPSs Atlantic sturgeon critical habitat in the action area.

One of the recovery objectives of the gulf sturgeon recovery plan is to prevent further reductions of wild populations within the range of the species, particularly the seven river systems where the species use for reproduction. These seven riverine areas are designated as critical habitat areas for gulf sturgeon. Essential features for the conservation of gulf sturgeon relevant to the action include abundant food items; areas necessary for normal behavior, growth, and survival; and water and sediment quality necessary for normal behavior, growth, and viability of all life stages. Anthropogenic stressors such as barriers to movement, land-based and in-water activities that lead to declines in water quality, and modifications of rivers all affect the quality and availability of critical habitat. Thus, we need to assess whether the discharges and associated contaminant and ANS stressors on gulf sturgeon critical habitat in areas of the Gulf of Mexico with ports and harbors with facilities for military vessels (see Table 1) rise to the level of adversely modifying or destroying the designated critical habitat when considered as a whole. Although some of the ANS and contaminants in UNDS Batch Two discharges from hull coating leachate and underwater ship husbandry have the potential to affect habitats in the port areas where military vessels are located, these effects are expected to occur outside most of the designated critical habitat areas because large portions of critical habitat are in rivers. The transport of contaminated water or sediment to critical habitat areas could occur, leading to changes in quality of critical habitat for gulf sturgeon. These effects are likely to be localized and are not expected to result in the loss or degradation of a large area containing the essential feature of gulf sturgeon critical habitat in the Gulf of Mexico. We base this on the location of military vessels in ports and harbors in the Gulf in relation to the location of critical habitat and the required performance standards meant to reduce the potential for discharges to affect ESA-listed species and their habitats. Therefore, we do not expect the action will appreciably diminish the overall value of gulf sturgeon critical habitat for the conservation of the species. We conclude that the action will not result in the destruction or adverse modification of gulf sturgeon critical habitat in the action area.

Two of the recovery actions in the Gulf of Maine DPS Atlantic salmon recovery plan are to increase survival through increased ecosystem understanding and identification of spatial and temporal constraints to salmon marine productivity and support management actions that support survival, and to collaborate with partners to support recovery efforts. Critical habitat includes streams in watersheds on the Maine coast and wherever Atlantic salmon occur in the estuarine and marine environment. Essential features for the conservation of Atlantic salmon include freshwater and estuarine habitats within the range of the DPS that include sites for spawning and incubation, juvenile rearing, and migration. Anthropogenic stressors such as barriers to movement, land-based and in-water activities that lead to declines in water quality, and modifications of rivers all affect the quality and availability of critical habitat. Thus, we need to assess whether the discharges and associated contaminant and ANS stressors on Gulf of Maine Atlantic salmon critical habitat in areas of the Gulf of Maine with ports and harbors with facilities for military vessels (see Table 1) rise to the level of adversely modifying or destroying the designated critical habitat when considered as a whole. Although some of the ANS and contaminants in UNDS Batch Two discharges from hull coating leachate and underwater ship husbandry have the potential to affect habitats in the port areas where military vessels are located, these effects are expected to occur outside most of the designated critical habitat areas because large portions of critical habitat are in rivers. While some of the port and harbor facilities, including some for military vessels, are in rivers, they are near the mouth of these water bodies and are not expected to affect upstream spawning and rearing habitat. The transport of contaminated water or sediment to critical habitat areas could occur, leading to changes in quality of critical habitat for Atlantic salmon. These effects are likely to be localized and are not expected to result in the loss or degradation of a large area containing the essential feature of Gulf of Maine DPS Atlantic salmon in the Gulf of Maine. We base this on the location of military vessels in ports and harbors in the Gulf in relation to the location of critical habitat and the required performance standards meant to reduce the potential for discharges to affect ESA-listed species and their habitats. Therefore, we do not expect the action will appreciably diminish the overall value of Gulf of Maine DPS Atlantic salmon critical habitat for the conservation of the species. We conclude that the action will not result in the destruction or adverse modification of Gulf of Maine DPS Atlantic salmon critical habitat in the action area.

11. CONCLUSION

After reviewing the current status of the ESA-listed species and designated critical habitats, environmental baseline, effects of the action, effects of interrelated and interdependent actions, and cumulative effects within the action area, it is NMFS biological opinion that the action is not likely to jeopardize the continued existence of killer whale (Southern Resident DPS), loggerhead sea turtle (Northwest Atlantic Ocean DPS), elkhorn coral, staghorn coral, pillar coral, rough cactus coral, lobed star coral, mountainous star coral, boulder star coral, Johnson's seagrass, bocaccio (Puget Sound/Georgia Basin DPS), yelloweye rockfish (Puget Sound/Georgia Basin DPS), Atlantic sturgeon (Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs), gulf sturgeon, green sturgeon, shortnose sturgeon, Atlantic salmon (Gulf of Maine DPS), chinook salmon (Central Valley Spring-Run, Puget Sound, and Sacramento River Winter-Run ESUs), chum salmon (Hood Canal Summer-Run ESU), coho salmon (Central California Coast ESU), and steelhead trout (California Central Valley, Central California Coast, and Puget Sound DPSs).

It is also our opinion that the action is not likely to destroy or adversely modify designated critical habitat for Southern Resident killer whale, elkhorn and staghorn coral, Johnson's seagrass, bocaccio (Puget Sound/Georgia Basin DPS), yelloweye rockfish (Puget Sound/Georgia

Basin DPS), Atlantic sturgeon (Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs), gulf sturgeon, green sturgeon, Atlantic salmon (Gulf of Maine DPS), chinook salmon (Central Valley Spring-Run, Puget Sound, and Sacramento River Winter-Run ESUs), chum salmon (Hood Canal Summer-Run ESU), coho salmon (Central California Coast ESU), and steelhead trout (California Central Valley, Central California Coast, and Puget Sound DPSs).

This document also represents the NMFS opinion on the effects of these actions on blue, fin, humpback (Central America, Western North Pacific, Arabian Sea, Cape Verde Islands/Northwest Africa, and Mexico Distinct Population Segments [DPSs]), North Atlantic right, Southern right, North Pacific right, sei, bowhead, sperm, gray (Western North Pacific DPS), Bryde's (Gulf of Mexico subspecies; proposed endangered), false killer (Main Hawaiian Islands Insular DPS), and beluga (Cook Inlet DPS) whales; Maui's and South Island Hector's dolphins; ringed (Arctic DPS), Guadalupe fur, Hawaiian monk, bearded (Beringia DPS), and Mediterranean monk seals; Steller sea lion (Western DPS); green (North Atlantic, South Atlantic, East Pacific, Central North Pacific, Central West Pacific, Indian-West Pacific, Southwest Pacific, Central South Pacific, North Indian, Southwest Indian, and Mediterranean DPSs), hawksbill, Kemp’s ridley, leatherback, loggerhead (South Atlantic Ocean, Southeast Indo-Pacific Ocean, Northeast Atlantic Ocean, North Pacific Ocean, South Pacific Ocean, North Indian Ocean, Southwest Indian Ocean, and Mediterranean Sea DPSs), and olive ridley (Mexico's Pacific coast breeding colonies and all other areas DPSs) sea turtles; dusky sea snake; Sakhalin, Adriatic, European, and Chinese sturgeon; smalltooth (U.S. and non-U.S. portion of range DPSs), largetooth, narrow, dwarf, and green sawfish; scalloped hammerhead (Eastern Atlantic, Eastern Pacific, Southwest Atlantic, and Indo-West Pacific DPSs), oceanic whitetip, daggernose, striped smoothhound, narrownose smoothhound, and spiny, smoothback, sawback, Argentine, and common angel sharks; Brazilian, blackchin, and common guitarfish; Nassau, gulf, and island grouper; steelhead trout (Southern California, South-Central California Coast, Northern California, Lower Columbia River, Upper Willamette River, Middle Columbia River, Upper Columbia River, and Snake River Basin DPSs); chinook salmon (California Coastal, Upper Willamette River, Lower Columbia River, Upper Columbia River Spring-Run, Snake River Fall-Run, and Snake River Spring/Summer-Run ESUs), coho (Lower Columbia River, Southern Oregon & Northern California Coasts, and Oregon Coast ESUs), chum (Columbia River ESU), and sockeye (Snake River and Ozette Lake ESUs) salmon; totoaba; eulachon (Southern DPS); African coelacanth (Tanzanian DPS); *Acropora globiceps*, *Acropora jacquelineae*, *Acropora lokani*, *Acropora pharaonis*, *Acropora retusa*, *Acropora rudis*, *Acropora speciosa*, *Acropora tenella*, *Anacropora spinosa*, *Eyphillia paradivisa*, *Isopora crateriformis*, *Montiplora australiensis*, *Pavona diffluens*, *Porites napopora*, *Seriatopora aculeata*, and *Cantharellus noumeae* corals; black abalone; white abalone; and chambered nautilus.

12. INCIDENTAL TAKE STATEMENT

Section nine 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 results in death or injury to ESA-listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering (see 50 C.F.R. §222.102).

Harass is further defined as an act that "creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering" (NMFS 2016e).

Incidental take is defined as take that results from, but is not the purpose of, the carrying out of an otherwise lawful activity (see 50 C.FR. §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 incidental take statement.

Southern resident killer whales, Northwest Atlantic Ocean DPS loggerhead sea turtles, ESA-listed Atlantic/Caribbean corals, Johnson's seagrass, and the following ESA-listed fish species: shortnose sturgeon; Puget Sound/Georgia Basin bocaccio; Puget Sound/Georgia Basin yelloweye rockfish; Central Valley Spring-Run, Puget Sound, and Sacramento River Winter-Run chinook salmon; Hood Canal Summer-Run chum salmon; Central California Coast coho salmon; California Central Valley, Central California Coast, and Puget Sound steelhead trout; Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic sturgeon; green sturgeon; gulf sturgeon; and Gulf of Maine Atlantic salmon will be exposed to UNDS Phase II Batch Two discharges from hull coating leachate and underwater ship husbandry from vessels of the armed forces in the action area that are likely to result in reductions in fitness of these species. Juvenile and adult life stages of Southern Resident killer whales, sea turtles and ESA-listed fish, and all life stages of ESA-listed Atlantic/Caribbean corals are likely to be affected by hull coating leachate and underwater ship husbandry vessel discharges. These two vessel discharges are likely to cause habitat degradation that will affect these species by reducing the availability of suitable prey organisms, refuge and foraging habitat, and future recruitment habitat thereby impairing essential behavioral patterns of feeding and reproduction.

12.1 Amount or Extent of Take

Section 7 regulations require NMFS to specify the impact of any incidental take of endangered or threatened species; that is, the amount or extent of such incidental taking of the species (50 C.F.R. § 402.14(i)(1)(i)). The amount of take represents the number of individuals that are expected to be taken by actions while the extent of take or "the extent of land or marine area that may be affected by an action" may be used as a surrogate if we cannot assign numerical limits for animals that could be incidentally taken during the course of an action (51 FR 19953).

Where it is not practical to quantify the number of individuals that are expected to be taken by the action, a surrogate (i.e., similarly affected species or habitat or ecological conditions) may be used to express the amount or extent of anticipated take (50 C.F.R. §402.14(i)(1)(i)). A surrogate may be used when the following three conditions are met: the ITS: (i) describes the causal link between the surrogate and take of the listed species; (ii) explains why it is not practical to express the amount or extent of anticipated take or to monitor take-related impacts in terms of individuals of the listed species; and (iii) sets a clear standard for determining when the level of anticipated take has been exceeded.

Take is exempted for the following species identified in our effects analysis as likely to be adversely affected by exposure to ANS and metals in UNDS Batch Two discharges from hull coating leachate and underwater ship husbandry: Southern resident killer whales, Northwest Atlantic Ocean loggerhead sea turtles, ESA-listed Atlantic/Caribbean corals, and the following ESA-listed fish species: shortnose sturgeon; Puget Sound/Georgia Basin bocaccio; Puget Sound/Georgia Basin yelloweye rockfish; Central Valley Spring-Run, Puget Sound, and Sacramento River Winter-Run chinook salmon; Hood Canal Summer-Run chum salmon; Central California Coast coho salmon; California Central Valley, Central California Coast, and Puget Sound steelhead trout; Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic sturgeon; green sturgeon; gulf sturgeon; and Gulf of Maine Atlantic salmon. We anticipate this take will be in the form of non-lethal harm or injury to individuals resulting from the effects of exposure to water and/or sediments contaminated with copper and other metals, as well as NAS that may become ANS. The RAAs selected by the Navy and EPA in conducting their exposure, response and risk analysis were assumed to represent marine areas within which a broad range ESA-listed species would be incidentally taken as a result of exposure to UNDS Batch Two hull coating leachate and underwater ship husbandry discharges. In general, we would expect low densities of ESA-listed species in these heavily industrialized, high traffic port locations compared to surrounding areas. The best available scientific and commercial information that exists currently does not include species density information for such localized areas for estimating take in terms of numbers of individuals; however, we do have information indicating which species are present in the various RAAs and other ports and harbors with facilities for vessels of the armed forces. Therefore, specifying the amount of take in numbers of individuals is not practicable. Moreover, monitoring take-related impacts from Batch Two underwater ship husbandry and hull coating leachate vessel discharges to mobile species is not practical due to the following: (1) sampling for ESA-listed fish in waters where facilities with vessels of the armed forces are located due to the small numbers of these fish species likely present in these locations, (2) the highly migratory nature of ESA-listed fish, loggerhead sea turtles (Northwest Atlantic Ocean DPS), and Southern Resident killer whales makes it not feasible to attribute elevated contaminant levels or ANS to a particular site or activity; and (3) even if affected animals are observed, it is unlikely that the exact cause of injury, mortality, or behavioral effects could be attributed to underwater ship husbandry and hull coating leachate vessel discharges, and sublethal effects could manifest at a later time when the animals have left the area. For benthic species such as ESA-listed Atlantic/Caribbean corals, incidental take caused by direct and indirect (habitat-related) effects of the action on these species and future recruits cannot be accurately quantified. The distribution and abundance of these species cannot be attributed solely to their response to the effects of discharges from military vessels. The effects on future recruitment cannot be readily observed without extensive monitoring of reproduction throughout the action area where facilities for military vessels are present or through laboratory experiments. Because it is not practical to express the amount of anticipated take or to monitor take-related impacts from discharges in RAAs and other ports and harbors with military facilities in terms of individuals of the ESA-listed species, we must use a surrogate measure to express the amount or extent of incidental take.

From our *Effects of the Action* (Sections 8.3 and 8.4) and *Integration and Synthesis* (Section 10) analyses and information on the distribution of the species likely to be adversely affected (Section 6.2), we found that the following species would likely be exposed to UNDS Phase II Batch Two vessel discharges from underwater ship husbandry and hull coating leachate in the Miami RAA: ESA-listed Atlantic/Caribbean coral species, and Northwest Atlantic Ocean loggerhead sea turtles. The following species are reasonably likely to be exposed to vessel discharges from hull leachate and underwater ship husbandry in the Norfolk RAA: Northwest Atlantic Ocean loggerhead sea turtles, Chesapeake Bay DPS Atlantic sturgeon, and shortnose sturgeon. The following species are reasonably likely to be affected by vessel discharges from hull leachate and underwater ship husbandry in the Seattle RAA: Southern Resident killer whales, Puget Sound/Georgia Basin bocaccio and yelloweye rockfish, green sturgeon, Puget Sound chinook salmon, Hood Canal Summer-run chum salmon, and Puget Sound steelhead trout. In the San Francisco RAA, the following species are reasonably likely to be exposed to vessel discharges from hull leachate and underwater ship husbandry: green sturgeon, Central Valley Spring-run and Sacramento River Winter-run chinook salmon, Central California Coast coho salmon, and California Central Valley and Central California Coast steelhead trout. Other port and harbor areas containing facilities for vessels of the armed forces where Batch Two discharges from hull coating leachate and underwater ship husbandry may occur (see Table 1) are reasonably likely to lead to exposure of the following species to these discharges: gulf sturgeon, Gulf of Maine Atlantic salmon, and Atlantic sturgeon (Gulf of Maine, New York Bight, Carolina, and South Atlantic DPSs). Additionally, some species, such as Northwest Atlantic Ocean loggerhead sea turtles, ESA-listed Atlantic/Caribbean corals, Pacific salmonids, green and green sturgeon may be exposed to discharges in other ports and harbors within the range of these species along the Pacific and Atlantic coasts and in the U.S. Caribbean.

The RAAs selected by the Navy and EPA for analysis of the risk of exposure to Batch Two vessel discharges from hull coating leachate and underwater ship husbandry represent the most likely areas where ESA-listed species may be taken as a result of exposure to contaminants in the water and sediments and ANS. We find it reasonable to assume there is a positive relationship between the areal extent of RAAs occupied by vessels of the armed forces (in other words, the portion of the RAAs where armed forces' vessels are located) and the associated mixing zone around discharging vessels. Beyond this mixing zone, contaminants in discharges are diluted to concentrations that are less than the response thresholds for ESA-listed species. We anticipate that the larger the contaminated area, the more individuals could be exposed. This area of influence (AoI) is the area most likely to be impacted by contaminants and stressors, including NAS, contained in underwater ship husbandry and hull coating leachate vessel discharges while these Batch Two discharges are occurring. For the recent ship tow consultation (NMFS 2019, Consultation Tracking Number FPR-2017-9228), the AoI was determined by linking the particle-tracking model, General NOAA Operational Modeling Environment (GNOME; NMFS 2018a), to output from hydrodynamics models for the particular water bodies where discharges occur. For the UNDS Batch Two consultation, NMFS proposes defining the AoI based on the area around ports where vessels of the armed forces are concentrated and where surface water concentrations are expected to exceed the expected response thresholds for ESA-listed species. This area will be calculated for each RAA and other ports and harbors in Table 1, where the ESA-listed species in Table 16 occur within six months of promulgation of the UNDS Batch Two rule.

The area around ports where vessels of the armed forces are concentrated and surface water concentrations are expected to exceed the expected response thresholds for ESA-listed species to pollutants such as copper and zinc can be calculated as follows. First, the volume of water within which mass loadings of copper and zinc will result in average concentrations at or below the National Recommended Water Quality Criteria Criterion Continuous Concentration for saltwater chronic exposures can be calculated. This concentration is 3.1 µg/L (0.0000031 kilograms per cubic meter [kg/m^3]) for copper and 81 µg/L (0.000081 kg/m^3) for zinc. Note that an ESA section 7 consultation has not been completed for these criteria so they may not be protective of all ESA-listed species or designated critical habitats. They are used here because they are the best available information at the time this consultation was written.

Consultation on EPA approval of Oregon's proposed chronic saltwater criteria indicated moderate mortality and reproductive effects on ESA-listed salmonids at the copper saltwater chronic criterion and low mortality and reproductive effects for the zinc saltwater chronic criterion. These effects were indicated at the scale of individuals or groups of individuals, but did not influence population attributes. Consultation on EPA approval of the saltwater chronic copper criterion proposed by the US Virgin Islands indicated that application of criterion is not expected to reduce the overall abundance of ESA-listed corals in waters of the US Virgin Islands, though it is expected to cause a small loss of numbers, decreases in reproductive potential and future recruits. The RPM in the consultation on EPA approval of the Virgin Islands Water Quality Standards, including the copper criterion requires monitoring and analysis to “determine the concentrations at which changes in habitat condition are observed to ensure the standards are protective of ESA-listed corals, green and hawksbill sea turtles, and Nassau grouper” (NMFS 2019).

New information indicating a lower exposure threshold for copper or zinc would be necessary to protect ESA-listed species would require revision of the thresholds applied to this incidental take statement.

Copper mass loadings from hull coating leachate will be calculated as:

Mass Load (kg per day) = (release rate in μg/cm^2/day x 0.000000001 μg/kg x 0.0001 cm^2/m^2) x wetted surface area in m^2

To account for times when there is a higher number of vessels in port, the mass load per day will be calculated assuming that 80 percent of the fleet is in port in order to allow a conservative calculation of the AoI beyond which authorized take that is incidental to the proposed action will not be exceeded.

Copper mass loadings from underwater ship husbandry will be calculated for a day when there is a full in-water cleaning as:

Mass load (kg per day) **–** (release rate in grams [g] per m^2 x 0.001 kg per g) x maximum wetted hull surface area cleaned in a day in m^2 per day

The average concentration within a specific volume of water can be calculated as:
$C(eq) = W \times t_F/V$

Where:
C(eq) = average concentration in kg per m^3
W = mass loading in kg per day
t_F = tidal flushing time in days
V = volume in m^3

Rearranging this equation enables a calculation of the volume of water for the mixing area within and around a port with an average concentration of copper equal to or greater than 3.1 ug/L as a function of the mass loading for the port and the flushing rate for the estuary such that:

$V = (W \times t_F)/C(eq)$

Where the flushing times are based on calculations the Navy has already done for the RAAs in the BE and, for vessels of the armed forces in ports outside the four RAAs where species in Table 16 are present, comparable flushing rates will be calculated.

The AoI will then be calculated as the volume of water where take is anticipated divided by the average depth of the port:

AoI (m^2) – volume of water at or above the effects threshold in m^3/Average depth in port in m

Water depths from local charts will be used for the measure of average depth and, because the four RAAs have multiple vessel population areas, the AoI will be calculated for the vessel population area with the greatest number of vessels, which is also expected to be the area with the greatest mass loading. Because vessels may move between port areas within an RAA, the total AoI for an RAA will be calculated as the AoI for the port with the highest mass loading multiplied by the total number of ports in the RAA. The mixing zone for the AoI will be delineated as a contour that is a constant distance from the land contours.

We are able to determine when this surrogate measure of take has been exceeded by calculating changes in mass loading from the discharges when new technologies are proposed and implemented. Greater mass loadings of pollutants and stressors would result in greater exposure concentrations and, consequently, a greater AoI in comparison with an established baseline (i.e., a greater level of exposure). Take can be directly related to the level of exposure, and greater levels of exposure could occur during periods of greater mass loading of pollutants to water and also incorporated into the sediment (e.g., during in-water hull cleaning events). Using the estimated mixing zone, based on the footprint of the area where vessels of the armed forces are concentrated and, therefore, where underwater ship husbandry and hull coating leachate discharges are occurring within an RAA as a surrogate, take would be exceeded if the action resulted in greater levels of exposure than expected under the proposed rule from hull coating leachate and underwater ship husbandry discharges in this area. For the foregoing reasons, the three criteria for using a surrogate have been met, and the mass loading of contaminants such as copper and zinc within a calculated AoI in ports and harbors where vessels of the armed forces are concentrated is a suitable surrogate for specifying the amount or extent of incidental take (Table 16).

Table 16. Surrogate Measure of Take of ESA-Listed Species Expressed as the Contaminant Mass Loading Due to UNDS Phase II Batch Two Hull Leachate and Underwater Ship Husbandry Vessel Discharges

ESA-listed Species	Surrogate for take expressed as the mass loading of contaminants in hull coating leachate and underwater ship husbandry UNDS Phase II Batch Two vessel discharges to water and sediment the area of influence as a result of the action
Green Sturgeon Southern DPS	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the Puget Sound RAA and the San Francisco RAA where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Bocaccio Puget Sound/Georgia Basin DPS	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the Puget Sound RAA where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Yelloweye Rockfish Puget Sound/Georgia Basin DPS	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the Puget Sound RAA where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Chinook Salmon Puget Sound ESU	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the Puget Sound RAA where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Chinook Salmon Central Valley Spring-run and Sacramento River Winter-run ESUs	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the San Francisco RAA where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Chum Salmon Hood Canal Summer-run ESU	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the Puget Sound RAA where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Coho Salmon Central California Coast ESU	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the San Francisco RAA where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Steelhead Trout Puget Sound DPS	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the Puget Sound RAA where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)

Steelhead Trout California Central Valley and Central California Coast ESUs	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the San Francisco RAA where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Atlantic Sturgeon Chesapeake Bay DPS	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the Norfolk RAA where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Atlantic Sturgeon South Atlantic, Gulf of Maine, New York Bight, and Carolina DPS	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the ports and harbors along the east coast (see Table 1) within the range of these DPSs where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Shortnose sturgeon	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the Norfolk RAA where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Atlantic Salmon Gulf of Maine DPS	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the ports and harbors in the Gulf of Maine (see Table 1) within the range of this DPS where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Gulf Sturgeon	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the ports and harbors in the Gulf of Mexico (see Table 1) within the range of this DPS where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Killer Whale Southern Resident DPS	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the Puget Sound RAA where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Loggerhead Sea Turtle Northwest Atlantic Ocean DPS	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the ports and harbors along the east coast (see Table 1) within the range of this DPS, including the Miami and Norfolk RAAs where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Elkhorn Coral	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the ports and harbors within the range of this species (see Table 1), including the Miami RAA, where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)

Staghorn Coral	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the ports and harbors within the range of this species (see Table 1), including the Miami RAA, where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Pillar Coral	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the ports and harbors within the range of this species (see Table 1), including the Miami RAA, where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Rough Cactus Coral	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the ports and harbors within the range of this species (see Table 1), including the Miami RAA, where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Lobed Star Coral	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the ports and harbors within the range of this species (see Table 1), including the Miami RAA, where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Mountainous Star Coral	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the ports and harbors within the range of this species (see Table 1), including the Miami RAA, where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)
Boulder Star Coral	The mass loading of contaminants from hull coating leachate and underwater ship husbandry to the portion of the ports and harbors within the range of this species (see Table 1), including the Miami RAA, where vessels of the armed forces are concentrated where Batch Two vessel discharges are likely to occur (i.e., area of influence)

12.2 Reasonable and Prudent Measures

The measure described below is nondiscretionary, and must be undertaken by the DoD and EPA so that it becomes a binding condition for the exemption in section 7(o)(2) to apply. Section 7(b)(4) of the ESA requires that when an agency action is found to be consistent with section 7(a)(2) of the ESA and the action may incidentally take individuals of ESA-listed species, NMFS will issue a statement that specifies the impact of any incidental taking of endangered or threatened species. To minimize such impacts, reasonable and prudent measures, and terms and conditions to implement the measures, must be provided. Only incidental take resulting from the agency actions and any specified reasonable and prudent measures and terms and conditions identified in the incidental take statement are exempt from the taking prohibition of section 9(a), pursuant to section 7(o) of the ESA.

"Reasonable and prudent measures" (RPMs) are nondiscretionary measures to minimize the amount or extent of incidental take (50 C.F.R. §402.02). NMFS believes the RPM described below is necessary and appropriate to minimize the impacts of incidental take on threatened and endangered species:

1. The DoD shall evaluate changes in biomass of NAS from underwater ship husbandry and changes in mass loadings of metals such as copper from hull coating leachate and underwater ship husbandry from vessels in the calculated surrogate areas within the ports and harbors in U.S. waters where vessels of the armed forces are concentrated (see Table 1). The DoD shall establish data quality objectives (DQOs) and a baseline in coordination with NMFS for the selected RAAs that can be extrapolated to other ports and harbors where the species in Table 16 occur if a baseline cannot be established for these areas. For a given waterbody, potential changes in mass loadings of metals from vessels of the armed forces from hull coating leachate and underwater ship husbandry discharges and potential changes in hull-fouling biomass, as an indicator of NAS, from underwater ship husbandry that could lead to changes in environmental exposure concentrations could result from, among other things:
    a. Changes in hull coatings
    b. Changes in technology used for underwater ship husbandry
    c. Changes in underwater ship husbandry practices
    d. Changes in individual vessel loading over time.

    DoD and EPA shall use this information to evaluate the efficacy of performance standards in order to ensure take of ESA-listed species identified in Table 16 is minimized.

12.3 Terms and Conditions

To be exempt from the prohibitions of section nine of the ESA, the DoD and EPA must comply with the following terms and conditions, which implement the RPM described above. These include the take minimization, monitoring and reporting measures required by the section 7 regulations (50 C.F.R. §402.14(i)). These terms and conditions are non-discretionary. If DoD and EPA fail to ensure compliance with these terms and conditions and their implementing the RPM, the exemption provided by section 7(o)(2) may lapse.

The following Terms and Conditions implement RPM 1:

1. The DoD shall establish a baseline for RAAs and other ports and harbors with concentrations of vessels of the armed forces where the species in Table 16 are present against which mass loading of contaminants such as copper and zinc and hull-fouling biomass, as an indicator of NAS, will be compared in order to determine whether contaminant contributions from hull coating leachate and underwater ship husbandry will increase exposures to harmful levels and track take from hull coating leachate and underwater ship husbandry discharges. The surrogate area baseline will calculated in order to determine take in different ports and harbors. The Navy shall submit a report to the NMFS ESA Interagency Cooperation Division detailing the baseline established for these areas against which future mass loadings will be compared within six months of promulgation of the UNDS Batch Two rule.
2. The DoD shall track the development of hull coatings designed to reduce the mass loading of metals from hull coating leachate and capture technologies designed to reduce the mass loading of contaminants and stressors, including NAS, from underwater ship husbandry.
3. Prior to the implementation of new technologies and practices, the DoD shall assess whether new hull coatings and cleaning technologies would result in effects to ESA-listed species to ensure the effects to ESA-listed species from changes in these discharges would still be minimized. The results of these assessments shall be submitted to NMFS ESA Interagency Cooperation Division within 60 days of completion of the assessment in the form of an electronic report. The use of new hull coatings or cleaning technologies that are likely to result in effects to ESA-listed species may require reinitiation of ESA section 7 consultation. If new technologies and practices that will ensure the minimization of take of ESA-listed species are implemented, the DoD shall evaluate potential/expected changes in mass loadings of contaminants from hull coating leachate and underwater ship husbandry discharges in the areas of influence within the RAAs (i.e., Miami, Norfolk, Puget Sound, and San Francisco). For other ports and harbors outside these RAAs where the species in Table 16 occur, the Navy shall use the estimates from these RAAs to determine how to extrapolate the information to the areas of influence within other ports and harbors.
    a. When a change in hull coating or cleaning technology or protocol is implemented, sampling shall be conducted to measure the change in the mass loading of contaminants from hull coating leachate and/or underwater ship husbandry. The methods used for this sampling shall be developed in coordination with NMFS within six months of the promulgation of the final implementing rule for the UNDS Batch Two discharges. Sampling will include effluent collection from cleaning devices to analyze the concentrations of metals such as copper and zinc, and biomass sampling from the waste stream generated by hull cleaning.

4. Every five years following promulgation of the UNDS Phase II Batch Two rule (i.e., the date the Phase III final rule is considered effective after it is published), the DoD and EPA shall prepare and submit a report on the RPM's efficacy to NMFS Interagency Cooperation Division. This timing corresponds to the five-year review between EPA and DoD already built into the rule and should include any new information that could impact implementation of the rule and its effects on ESA-listed species. The information provided by the Navy and the EPA should be sufficient to determine whether reinitiation of consultation may be required.
    a. If new information that could affect implementation of the rule and its effects on ESA-listed species does not exist based on implementation of the other Terms and Conditions (1 through 3) described above, EPA and DoD shall submit a letter to NMFS stating that no new information exists since the time of promulgation of the implementing rule.
    b. If new technologies or management practices are implemented as part of underwater ship husbandry and hull coating leachate discharges, a report shall be submitted summarizing:
        i. Any changes in mass loadings of metals and biomass of NAS resulting from changes in hull coating and cleaning technologies or practices for facilities in the selected RAAs and other ports and harbors (Table 1) containing ESA-listed species (Table 16); provide an estimate of incidental take (reflected as a change in the AoI) based on changes in mass loading within the surrogate areas calculated for the different RAAs and other ports and harbors; evaluate whether any changes in hull coating and cleaning technologies or practices for a given population of vessels has resulted in increased take; and assess whether determinations of the effectiveness of performance standards can be made for the Batch Two discharges from hull coating leachate and underwater ship husbandry.
        ii. The report shall include a discussion of the implementation of the nondiscretionary Terms and Conditions required by the RPM.
        iii. The report shall also include information regarding proposed changes to performance standards or implementation of new technologies to further reduce potential effects of Batch Two hull coating leachate and underwater ship husbandry vessel discharges.

13. 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. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of an action on ESA-listed species or critical habitat, to help implement recovery plans or develop information (50 C.F.R. §402.02).

1. We recommend that the DoD collaborate with hull coating manufacturers and conduct research on the effectiveness of non-toxic antifouling hull applications including silicon based compounds and epoxy coatings (McClay et al. 2015).
2. We recommend that the DoD continue its research on in-water hull cleaning methods that minimize impacts to surrounding water bodies by collecting organisms, paint particles and other materials dislodged by the cleaning process for upland disposal (McClay et al. 2015).
3. We recommend that the DoD develop a pilot program in coordination with NMFS to address ANS concerns. The program would include the development and implementation of a sampling protocol to assess the effectiveness of capture versus other hull cleaning technologies in reducing or eliminating the release of hull fouling organisms and the percent mortality of organisms that are released to the water column.
4. We recommend that the DoD and EPA make available to the public the database for tracking the development of hull coatings and capture technologies, as well as their assessments of the potential effects of these on ESA resources, to assist NMFS and other entities in assessing alternatives and their potential impacts to ESA resources.
5. We recommend that the DoD and EPA continue exploring new technologies and other alternatives to further refine performance standards and reduce Batch Two discharges to estuarine and marine waters where ESA-listed species and their habitats occur. As part of this, we recommend the DoD evaluate ways to establish the effectiveness of performance standards for Batch Two discharges for different types and ages of vessels of the armed forces.
6. We recommend that, in order to address concerns related to contaminants that may be in some Batch Two discharges that can sorb to sediments, an overall evaluation of the extent to which resuspension of sediments and associated release of contaminants due to sediment oxygenation occurs as a result of vessel discharges that may disturb the bottom be conducted by the DoD. This evaluation should include information regarding the temporal and spatial extent of occurrence within the RAAs.
7. To the extent practicable, we recommend that INRMPs be modified to include surveys of refuge and foraging habitat of ESA-listed species in the selected RAAs and other ports and harbors with facilities for vessels of the armed forces to assess changes over time in the quality and quantity of habitat. Surveys for the species themselves should also be conducted to assess patterns of presence/absence of the species in areas containing facilities for vessels of the armed forces and to assess the abundance of these species, if possible.

In order for NMFS Office of Protected Resources Endangered Species Act Interagency Cooperation Division to be kept informed of actions minimizing or avoiding adverse effects on, or benefiting, ESA-listed species or their critical habitat, DoD and EPA should notify the Endangered Species Act Interagency Cooperation Division of any conservation recommendations they implement in their final action.

14. Reinitiation Notice

This concludes formal consultation for DoD and EPA for the UNDS Phase II Batch Two rule. 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 taking specified in the incidental take statement is exceeded.
(2) New information reveals effects of the agency action that may affect ESA-listed species or critical habitat in a manner or to an extent not previously considered.
(3) The identified action is subsequently modified in a manner that causes an effect to ESA-listed species or designated critical habitat that was not considered in this opinion.
(4) A new species is listed or critical habitat designated under the ESA that may be affected by the action (50 C.F.R. §402.16).

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16. APPENDICES

16.1 Appendix A. Section 5 of the BE, Analysis of Risk of Effects from Pollutants and Stressors in Discharges from Vessels of the Armed Forces Regulated by Uniform National Discharge Standards

5.0 ANALYSIS OF RISK OF EFFECTS FROM POLLUTANTS AND STRESSORS IN DISCHARGES FROM VESSELS OF THE ARMED FORCES REGULATED BY UNIFORM NATIONAL DISCHARGE STANDARDS

As discussed in Section 2.1, the purpose of UNDS is, in part, to stimulate the development of innovative vessel pollution control technology and advance the ability of the Armed Forces to better design and build environmentally sound vessels. Environmentally sound vessels are those that operate in a way that will not have a substantial impact on ecosystems and environmental resources.

For this BE, the EPA and DoD evaluated the effectiveness of UNDS for reducing the accumulation of pollutants from discharges from vessels of the Armed Forces in ports and harbors and, consequently, the potential for impacts to federally listed aquatic and aquatic-dependent species. Risk to listed species from exposure to pollutants and other stressors present in the regulated Batch Two discharges was analyzed. Although there are eleven Batch Two discharges, the evaluation focuses on eight discharges selected for detailed analysis (as explained in Section 3.2.1) because the volume of these discharges and mass loading of pollutants from them has the potential for adverse effects. The other three discharges have lower volume and/or concentrations of pollutants and are are not as likely to have adverse effect. The potential for adverse effects was evaluated both quantitatively and qualitatively. Although there is some uncertainty in quantifying the potential for adverse effects to federally-listed species (see Section 5.5), quantitative evaluations are generally preferable to qualitative evaluations for risk assessments because they allow the likelihood for adverse effects to be estimated. The conclusions from the risk analysis were used to inform the effects determinations presented in Section 8.

The eight Batch Two discharges from vessels of the Armed Forces identified in Section 3.2.1 to require a more detailed analysis are: deck runoff, firemain systems, graywater, hull coating leachate, sonar dome discharge, submarine bilgewater, surface vessel bilgewater/OWS effluent, and underwater ship husbandry. Concentrations of pollutants in these discharges were modeled as described in Section 5.1. The pollutants and other stressors in the eight discharges fall into one of the following pollutant categories: metals, oil and grease, conventional pollutants, nutrients, toxins and non-conventional pollutants with toxic effects, PPCPs, and non-indigenous aquatic species (NAS).

The EPA and DoD conducted a quantitative effects analysis for most of the pollutants in this BE, as described in Section 5.3. However, the EPA and DoD conducted qualitative analyses of risk to listed species from potential ANS introductions, from exposure to PPCPs, and from exposure to O&G as described in Section 5.2. The findings of these analyses are combined with the findings of the quantitative analysis to support the overall effects determinations presented in Section 8.

5.1 Estimation of Harbor Exposure Concentrations

The EPA and DoD selected a flushing-based screening modeling approach to assess the potential for UNDS Batch Two pollutants to affect listed species populations within each RAA. The models were used to calculate exposure concentrations in the receiving waters of harbors selected to be representative RAAs. The modeled concentrations have been used to support both the quantitative and qualitative risk analysis for potential effects to the listed species as described in Sections 5.2 and 5.3. The flushing-based screening modeling approach establishes a simple model considering the hydrodynamic conditions, Armed Forces vessel populations, and species across the seven RAAs. The hydrodynamic conditions of the RAA harbors were the primary force driving the modeling approach (tidal prism or freshwater flush). The modeling approach calculates receiving water concentrations for estuarine harbor scenarios and a dilution model to estimate the receiving water concentrations for a river harbor scenario. The approach to estimating pollutant loads and calculating harbor concentrations is explained in in the following sections. Appendix F provides a detailed explanation of the model selection process, modeling assumptions, and the equations used for the calculations.

The EPA and DoD modeled seven different harbor environments (six coastal and one inland riverine/freshwater) based on receiving water and RAA characteristics. Vessel population data for the vessels of the Armed Forces and data on the receiving water characteristics for each RAA were used as inputs to the model. The EPA and DoD then modeled the concentrations for Batch Two pollutants in each RAA to depict the range of environmental conditions that could potentially be observed. For the coastal harbors, the maximum, mean, and minimum concentrations were modeled for each pollutant across the six modeled harbors. The EPA and DoD modeled one riverine RAA with a high density of vessels of the Armed Forces and most representative of all river harbors with vessels of the Armed Forces present. Because only one freshwater harbor was modeled, a single exposure concentration (EC) for each pollutant was calculated for the inland riverine/freshwater harbor.

5.1.1 Representative Harbor Scenarios

The EPA and DoD identified harbors (Table 5-1) that represent a geographically and environmentally diverse group of water bodies and have a high density of vessels of the Armed Forces regulated by UNDS in an effort to develop model input values that represent realistic environmental conditions.

Table 5-1. Representative Harbors Selected for Model Input Parameter Development

Harbor City	Name	Primary River Input
Coastal (Estuarine) Harbors		
Miami, Florida	Biscayne Bay	Miami River
Norfolk, Virginia	Chesapeake Bay	James River
Pearl Harbor, Hawaii	Pearl Harbor	Halawa Stream and Waikele Stream
San Diego, California (CA)	San Diego Bay	Chollas Creek and Sweetwater River
San Francisco, CA	San Francisco Bay	Sacramento River and San Joaquin River
Seattle, Washington	Puget Sound	Snohomash River and Puyallup River
Inland (Riverine) Harbor		
St. Louis, Missouri	Upper Mississippi River	Mississippi River and Missouri River

These seven harbors were carefully chosen to be a representative subset of the population of vessels of the Armed Forces and contain the widest possible range of vessel classes; physical characteristics (water body flow, flushing time, and salinity); and ecological communities as well as the highest pollutant loads from vessels of the Armed Forces. Specific characteristics of each harbor selected can be found in Appendix F, Table F-1. The RAAs are defined by an area within each harbor where vessels of the Armed Forces and their discharges are most concentrated. In summary, the RAAs within these seven harbors are comprised of the following representative characteristics:

- Include homeports for a total of 2,474 vessels of the Armed Forces (1,825 under 79 feet and 649 over 79 feet) representing approximately 39 percent of the total population of vessels of the Armed Forces.
- Include the three military Homeports with the most vessels of the Armed Forces: Norfolk, VA (974); San Diego, CA (791); and Pearl Harbor, HI (217).
- Reflect a wide geographic range that captures a wide variety of threatened and endangered species within harbors on the east and west coasts, a Pacific island, in northern and southern regions of the United States, and within a riverine system.
- Range in size from a surface area of 41 million square meters to 265 million square meters.
- Range in depth from approximately 3 meters to137 meters.
- Range in total volume from 132 million cubic meters to 29 billion cubic meters.
- Range in river flow from 31 thousand cubic meters per day to 424 million cubic meters per day.
- Range in harbor salinity from 0 practical salinity unit (PSU) to nearly 34 PSU.

The seven RAAs identified fall into two types of systems:

- Riverine; and
- Estuarine where circulation is predominantly influenced by tides, as indicated by higher salinities.

For both the estuarine and riverine harbors, the models developed present the “likely” scenarios that would result in a range of pollutant exposure concentrations from vessel discharges covered by UNDS.

Although some estuarine RAAs have a larger riverine influence than others, none of the estuarine RAAs are represented by estuaries where circulation is predominantly influenced by river flow, as would be indicated by lower salinities. This is largely due to alteration of the harbors where vessels of the Armed Forces are homeported (e.g., dredging and channel widening) or the river systems that feed into them (e.g., damming and water diversion). Therefore, the predicted residence time for pollutants released to each harbor in discharges from vessels of the Armed Forces and the resulting concentrations of any pollutants within a harbor from the discharges, which are affected by the relative tidal and fluvial influences on estuarine circulation, are higher than they would be for a fluvially dominated system. It is assumed that the total mass loading remains suspended in the water column and available for exposure and uptake by direct exposures, filter feeding, and adsorption across gill surfaces rather than becoming bound, and sometimes buried, in the sediments.

For the river harbor model, the dilution equation uses the average annual river flow rate to calculate the receiving water concentration. Although using low flow or base flow conditions would be the most conservative, the average was selected to avoid being overly conservative. The model is intended to be a screening model that represents conditions in most freshwater ports and harbors. The RAA selected already contains a higher density of vessels of the Armed Forces and represents harbors where there will be higher mass loadings. The uncertainties associated with this approach are discussed in Section 5.6.

5.1.2 Representative Action Area Pollutant Loading Estimates

Based on the types of vessels of the Armed Forces, discharges, and corresponding pollutants selected for detailed evaluation in the BE (see Section 3.2.4), the EPA and DoD identified 21 pollutants to include in the modeling analysis because their concentrations in discharges exceed the most stringent available WQC.

Pollutant loads for each of the seven RAAs were estimated by first calculating the vessel-specific pollutant loads from each of the discharges selected for evaluation. Vessel-specific pollutant loads are based on:

- The types of pollutants in each of the discharges resulting from normal operations.
- The concentration of pollutants in the discharge for those pollutants exceeding the most stringent federal or state WQC (WQC were used as a guide to identify pollutants that are likely to impact aquatic and aquatic-dependent species; however, there may be other pollutants in discharges that could have adverse effects, as well).
- The discharge flow rate for each vessel class.
- The number of days per year that a vessel discharges in port where pollutants are expected to accumulate.
- The number of vessels in each class that are home ported in each RAA.

For each discharge, the total mass loading of each pollutant was calculated as the sum of the mass loads for all vessels of the Armed Forces discharging within each of the RAA harbors. The vessel-specific pollutant loads were estimated based on existing data reported predominantly in EPA's published NOD Reports, the Armed Forces Vessel Database, and technical knowledge of vessel-specific discharge flow rates. The Armed Forces Vessel database, which tracks all vessels regulated by UNDS, was used by the EPA and DoD to identify the populations of vessels of the Armed Forces in each RAA. The vessel populations for each of the seven RAAs that are known to produce each of the eight discharges selected for detailed evaluation and contribute to the 21 pollutants identified for this BE are presented in Table 5-2 below. Appendix F provides detailed descriptions of the data sources and assumptions used to estimate the pollutant mass loadings from vessels of the Armed Forces in each RAA. Tables F-3 and F-4 present the total pollutant loads for each of the seven RAAs used to calculate the exposure concentrations for the risk analysis of potentially affected species. The exposure concentrations modeled represent maximum surface water exposure concentrations for waterbodies where circulation is restricted and are used to evaluate risk of impact to ecological receptors throughout the action area, including in open water where concentrations can be expected to be much lower.

Pollutants may also partition to and concentrate in sediments. Partitioning to sediments is a highly complex process that is dependent on each chemical's partitioning coefficient, the availability of ligands, sediment grain size, temperature, salinity, bioturbation and biodeposition, and a number of other processes and variables that are site specific. Because of the uncertainties associated with modeling sediment concentrations and the amount that is bioavailable for uptake, this assessment focuses on exposure to modeled surface water concentrations, and sediment exposures are an information gap in the assessment.

Table 5-2. Estimated Number of Vessels of the Armed Forces That Generate Each of the Discharges in the Representative Action Areas

Discharge Type	Number of Vessels of the Armed Forces Stationed in RAA Harbors Generating Each Discharge							
	Miami, FL	Norfolk, VA	Pearl Harbor, HI	San Diego, CA	San Francisco, CA	Puget Sound/ Seattle, WA	St Louis, MO	Total
Deck Runoff (all vessels)	36	975	217	791	98	337	21	2475
Firemain Systems	3	105	30	65	12	31	1	247
Graywater	2	101	30	64	13	28	0	238
Hull Coating Leachate	11	387	127	261	26	201	2	1015
Sonar Dome Discharge	0	40	24	26	0	12	0	102
Surface Vessel Bilgewater/OWS Effluent	4	285	94	238	33	182	2	838
Submarine Bilgewater	0	8	17	4	0	15	0	44
Underwater Ship Husbandry	11	320	114	150	25	144	2	766

5.1.3 Exposure Concentrations

To calculate exposure concentrations for the 21 pollutants in discharges selected for detailed analysis (nutrients, metals, monocyclic aromatic hydrocarbons (MAHs) and PAHs, and other toxic chemicals), the EPA and DoD considered the following:

- Annual in-port mass loading rate of each pollutant in the RAA from each type of discharge and class of vessel of the Armed Forces
- Harbor flushing rate determined by freshwater inflow and tidal range
- Harbor volume based on mean harbor depth and surface area, defined as the natural boundaries for where the waterbody meets the coastline, ocean and tributaries within a three-mile radius of the home port where vessels of the Armed Forces are located (not the volume of the entire harbor)

Using these criteria, concentrations of pollutants in receiving waterbodies resulting from discharges from vessels of the Armed Forces were modeled for each of the seven selected RAA harbors listed in Table 5-1. The pollutant concentrations modeled reflect the number of days vessels operate in port, the time spent transiting from port to outside of three miles offshore, and when discharges are being transferred to an onshore facility (such as when they are pierside). Mass loading rates are based on the weighted average concentrations of pollutants measured in the discharges and the average discharge flow rate. Although not specifically captured by the modeling, it is important to remember the following standards and prohibitions when vessels are in port:

- Flight deck washdowns are prohibited;
- Weather deck washdowns are minimized, and deck surfaces are broom swept and clear of debris prior to washdowns when the do occur;
- To the greatest extent practicable, firemain system maintenance and training be conducted outside of port and as far away from shore as possible;
- For vessels designed with the capacity to hold graywater, vessels are prohibited from discharging graywater within one mile of shore if an onshore facility is available and, if an onshore facility is not available, production and discharge of graywater must be minimized within one mile of shore;
- The water inside a sonar dome must not be discharged for maintenance activities unless the use of a drydock for the maintenance activity is not feasible;
- Submarine bilgewater discharges must not occur while the submarine is in port when the port has the capability to collect and transfer the bilgewater to an onshore facility;
- The discharge of OWS effluent must not occur in port if the port has the capability to collect and transfer OWS effluent to an onshore facility;
- To the greatest extent practicable, vessel hulls with AF hull coatings must not be cleaned within 90 days after the AFC application; and
- To the greatest extent practicable, rigorous vessel hull cleanings must take place in drydock or at a land-based facility where the removed fouling organisms or spent AF hull coatings can be disposed of onshore.

To be conservative, the harbor flushing rate used to model harbor concentrations represents a higher residence time that would result in higher receiving water concentrations. The EPA and DoD determined the maximum, minimum, and average receiving water concentrations for the estuary harbors modeled; freshwater concentrations were modeled for a single river harbor and conservatively represents a high receiving water concentration for a river harbor (Table 5-3).

In the quantitative effects analysis presented in Section 5.3, the EPA and DoD use the maximum modeled receiving water EC across all harbors to assess risk of effect to federally listed species from the issuance of the UNDS rule. Although the modeled concentrations are only representative of harbors where pollutants tend to concentrate, they were used to assess exposures and the potential for adverse effects to all federally-listed freshwater and marine aquatic and aquatic-dependent species, regardless of where they occur. Concentrations outside of United States ports and harbors are expected to be substantially lower than in harbors because of dilution and the lower density of vessels discharging.

Table 5-3. Estimated Receiving Water Concentrations from Vessels of the Armed Forces Incidental Discharge Loadings in Representative Estuary and River Harbors (Representative Action Areas)

Class	Pollutant	Range of Estimated Estuary Harbor Concentrations (µg/L)	Estimated River Harbor Concentration (µg/L)
Metals	Cadmium	7.40E-08 - 7.80E-06	3.2E-09
	Chromium	3.60E-06 - 0.00039	1.60E-07
	Total Copper	0.0016 - 0.3	0.000067
	Dissolved Copper	Could not be calculated because only measured in some discharges; dissolved copper concentrations assumed to be the same as total copper concentrations	
	Iron	0.000023 - 0.038	2.3E-06
	Lead	8.30E-06 - 0.00082	3.00E-07
	Mercury	5.7E-10 - 4.6E-07	9.3E-12
	Nickel	5.70E-06 - 0.0097	1.00E-07
	Silver	1.4E-09 - 2.80E-06	0
	Total Zinc	0.004 - 0.41	0.000027
	Dissolved Zinc	Could not be calculated because only measured in some discharges; dissolved zinc concentrations assumed to be the same as total zinc concentrations	
Petroleum Hydrocarbons	Oil and Grease	0.00073 - 0.074	0.000028
	TPH	3.70E-08 - 1.90E-06	2.7E-09
Toxics and Non-Conventional Pollutants with Toxic Effects	Bis (2-ethylhexyl) phthalate	1.8E-07 - 0.014	2.8E-08
	Tributyltin	0 - 0.00021	0
	Chlorine Produced Oxidants	0 - 0.0037	0
Nutrients/ Water Quality	Nitrate/Nitrite	2.4E-06 - 0.0011	4.80E-08
	Total Kjeldahl Nitrogen	0.000035 - 0.049	2.7E-07
	Total Nitrogen	0.000037 - 0.05	3.2E-07
	Ammonia as Nitrogen	0.000019 - 0.036	1.6E-08
	Total Phosphorus	1.3E-05 - 0.0025	3.2E-07
	Total Organic Carbon	0 - 0.0039	0
	Total Suspended Solids	0.00014 - 0.28	0
	Biological Oxygen Demand	0.000096 - 0.19	0
	Chemical Oxygen Demand	0.00026 - 0.28	0

See Appendix F for the assumptions and equations used to estimate exposure concentrations.

5.1.4 Aquatic Organism Body Burdens

Bio-concentration factors (BCFs) or bioaccumulation factors (BAFs) are used in the analysis to calculate the concentration (mg/kg) of pollutant expected in tissues of aquatic organisms exposed to the maximum concentration (mg/L) of pollutant modeled in a receiving water of interest (Section 5.1). Bioconcentration refers to uptake across a respiratory surface (the gills of fish and skin of invertebrates) while bioaccumulation refers to uptake via both respiration and ingestion (Gobas et. al, 1999). BCFs typically are determined from controlled laboratory studies; BAFs are based on field-collected data. When lower trophic level prey organisms are exposed to the pollutant via water only, the use of BCFs is the appropriate method of predicting pollutant concentrations in the tissues of aquatic species, which are prey for higher trophic level species. Use of BCFs also is appropriate for predicting tissue residues in higher trophic level aquatic organisms from the concentrations of some metals in water (e.g., aluminum, copper, cadmium, lead, nickel, and zinc) and for organics with low octanol-water partition coefficients (K_{ow}) (e.g., benzene) because the tissue concentrations of these pollutants do not increase through bio-magnification from lower trophic levels to higher trophic levels (EPA, 2003).

Octanol-water partition coefficients are related to the solubility of a substance in water and, for log K_{ow}s less than six, the log K_{ow} is a good relative predictor of a chemical's tendency to bioconcentrate. For substances with log K_{ow}s greater than six, BCFs tend to decrease with increasing log K_{ow} (ICCA, no date). This is likely because chemicals partition to sediment and particulates from water and are strongly bound by sediment and particulates. On the other hand, a higher log K_{ow} is an indicator that an organic compound will biomagnify, or increase, through the food chain. While K_{ow}s and BCFs are useful for predicting tissue concentrations of pollutants with lower K_{ow}s (<6), neither is an accurate predictor of biomagnification. Even still, the log K_{ow} can help predict tissue concentrations of many pollutants in prey that are consumed by higher trophic level receptors and used to estimate dietary doses.

BAFs were used to predict pollutant concentrations in the tissues of invertebrates and fish, which are prey for higher trophic level aquatic organisms for situations where the organisms are exposed to the pollutant through all routes of exposure (i.e., water, diet, cutaneous), and where concentrations in tissue bio-magnify up the food chain. BAFs take tissue lipid concentrations into account, and tissue concentrations are lipid-normalized. Tissue concentrations were not modeled for other taxonomic groups (e.g., seagrass) or specifically for corals because it did not provide any information that could be used for the quantitative risk assessment. This is an uncertainty in many aquatic risk assessments; however, it should be recognized that accumulation of pollutants in these species could have significant impact on those species and the many other species that depend on them for food and habitat.

A similar approach was used for EPA's *Biological Evaluation of Oregon's Water Quality Criteria for Toxics* (Oregon Toxics BE; EPA, 2008), and BCFs and BAFs were identified for several pollutants. This report served as the source of many BCFs and BAFs for EPA's BE for the VGP and was also used as the primary source of BCFs and BAFs for this BE., EPA's ECOTOX database and published peer-reviewed studies served as sources of BCFs and BAF for any pollutants that were not evaluated for the Oregon Toxics BE. All available BCFs and BAFs and their sources are presented in Appendix G.

5.2 Qualitative Effects Analysis

This section discusses the qualitative analysis of potential effects from ANS introductions, oil and grease, and PPCPs in Batch Two discharges from vessels of the Armed Forces that have been selected for detailed evaluation. A qualitative analysis is performed when there are insufficient information or methods for evaluating risk of effects. Although several approaches for quantifying risk from NAS have been developed, none of the currently available methods are applicable across a broad geographic range, range of ecosystems, or types of species. As such, no effective approaches for quantifying risk from NAS introductions or ANS invasions specifically for the action area for this BE currently exist. Further, the data for oil and grease, including food oils, lack species-specific chronic response and bioaccumulation data because the composition of O&G varies by source. Lastly, species-specific response data for PPCPs is limited, and PPCPs in graywater discharges from vessels of the Armed Forces have not been quantified. This lack of data prevents the calculation of a risk quotient to support a quantitative effects analysis. The approaches to the qualitative analyses for ANS and O&G are presented in the following sections.

5.2.1 Qualitative Analysis of Risk of Effects from Aquatic Nuisance Species Introductions

A NIS is any species that occurs outside its native range, usually as a result of human activities. NIS can be either terrestrial or aquatic. NAS were defined in 1991 by the USFWS NAS Program as any member(s) (i.e., individual, group, or population) of a species that enters a body of water or aquatic ecosystem outside its historic or native range where it has evolved to its present form and includes plants, animals, and microbes. While many NAS that are introduced to another region have minimal impact on the ecosystem, a small percentage will outcompete native species and have a negative impact. These species are called ANS. ANS was defined by Public Law 101-636 (Non-indigenous Aquatic Nuisance Prevention and Control Act of 1990) as a NAS that “threatens the diversity or abundance of native species or the ecological stability of infested waters, or commercial, agricultural, aquacultural, or recreational activities dependent on such waters.” ANS invasions occur in three steps: (1) introduction of a non-indigenous aquatic species (NAS) outside of its range, (2) survival and establishment of that NAS outside of its range, and (3) spread/invasion of a NIS at a new location (NAS becomes ANS). Figure 5-1 shows the relationship between NIS, NAS and ANS.

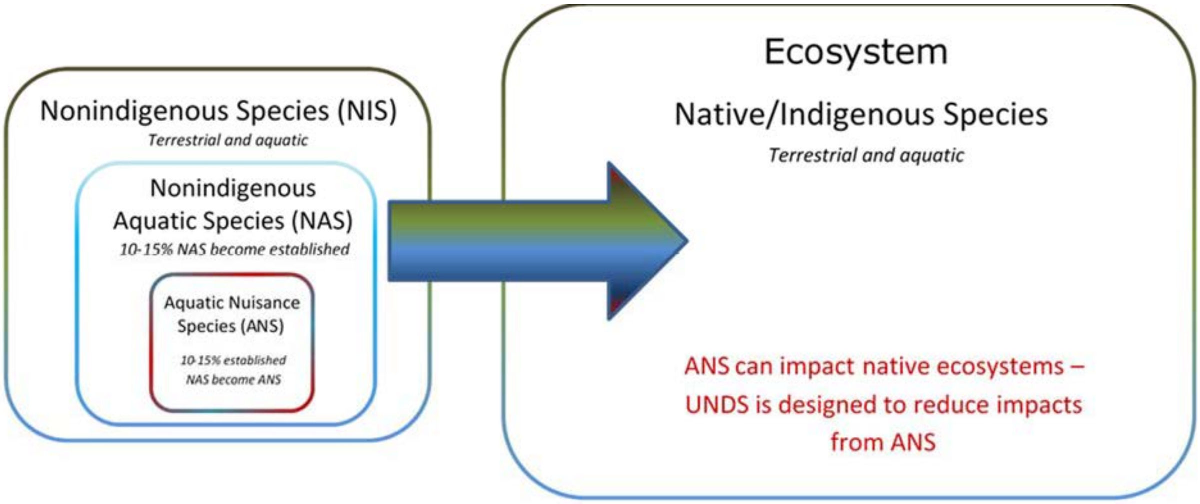

Figure 5-1. Definitional Relationships Between Non-Indigenous Species, Non-Indigenous Aquatic Species, and Aquatic Nuisance Species

5.2.1.1 Vessels of the Armed Forces as Vectors for Non-Indigenous Aquatic Species Introductions

NAS may be introduced through a variety of vessel operations, including ballast water and sediment discharge from ballast tanks; biofouling on the wet surfaces of vessel hulls, anchor chains and chain lockers; and on-board systems that take in seawater and store it for any period of time before discharging overboard. Of all vessel-related vectors, ballast water and biofouling are reported as the primary mechanisms for the transport and introduction of NAS in the modern shipping era (Ruiz et al., 2000; Takata et al, 2011). Biofouling primarily transports species that have sedentary or sessile benthic habits or species that are associated with these communities (i.e., living in, between or on other organisms; Minchin and Gollasch, 2003).

It is important to realize that this is not an assessment of risk from the movement of vessels of the Armed Forces, but rather an assessment of regulated discharges. This BE for the UNDS Batch Two discharges includes the detailed assessment of effects to federally listed species from exposure to eight discharges. One of these discharges, underwater ship husbandry, may affect the potential for a vessel to introduce NAS that could become ANS and is the focus for the assessment of risk to federally listed species. Underwater ship husbandry is defined as the inspection, grooming, maintenance, and repair of hulls and hull appendages performed while a vessel is waterborne. During hull grooming (cleaning), hull-fouling organisms or viable fragments of fouling organisms are dislodged and may be introduced outside their native range. NAS may also be introduced through spawning, independent detachment, or being knocked from vessel hulls by other forces while in port and underway. When NAS are introduced to and become established in a new location, they have the potential for invasion and could threaten the abundance and survival of native species, including those that are federally listed under the ESA.

5.2.1.2 Hull Biofouling Process and Factors Affecting Hull Fouling

In general, there are four sequential stages of biofouling: (1) biochemical conditioning, (2) bacterial colonization, (3) colonization by eukaryotes, and (4) colonization by multicellular eukaryotes, where the fouling in one stage promotes fouling in the next stage (Floerl et al., 2010; Walh 1989 in Kohli 2007). Primary biofouling (stages 1 and 2) begins as soon as the surface of a vessel is submerged in seawater, with biochemical conditioning and the formation of a slime layer consisting mostly of bacteria and microscopic algae. Secondary biofouling (stage 3) usually includes hard encrusting animals such as acorn barnacles, bryozoans and serpulid worms, but may also include soft algal tufts and mobile amphipods. This is then followed by a more general infestation, spreading beyond edges onto the flats of the ship hull. If allowed to progress, tertiary biofouling (stage 4) then follows, generally consisting of larger organisms such as sponges, sea squirts, mussels, oysters and seaweeds that build up on the secondary biofouling layer, and mobile animals such as crabs and sea stars that can live in this growth. Although the time for the biofouling community to develop will vary geographically, in general, biochemical conditioning occurs within 1-2 hours of the surface being submerged, bacterial colonization occurs within 24 hours, spores of macroalgae and protozoa colonize the surface within a week, and larvae of macrofoulers colonize the hull surface within 2-3 weeks (Figure 5-2) (Abarzua and Jakubowski, 1995). Biofouling communities can be extremely diverse, and if these communities become highly developed, they can also provide microhabitats for mobile organisms such as fish.

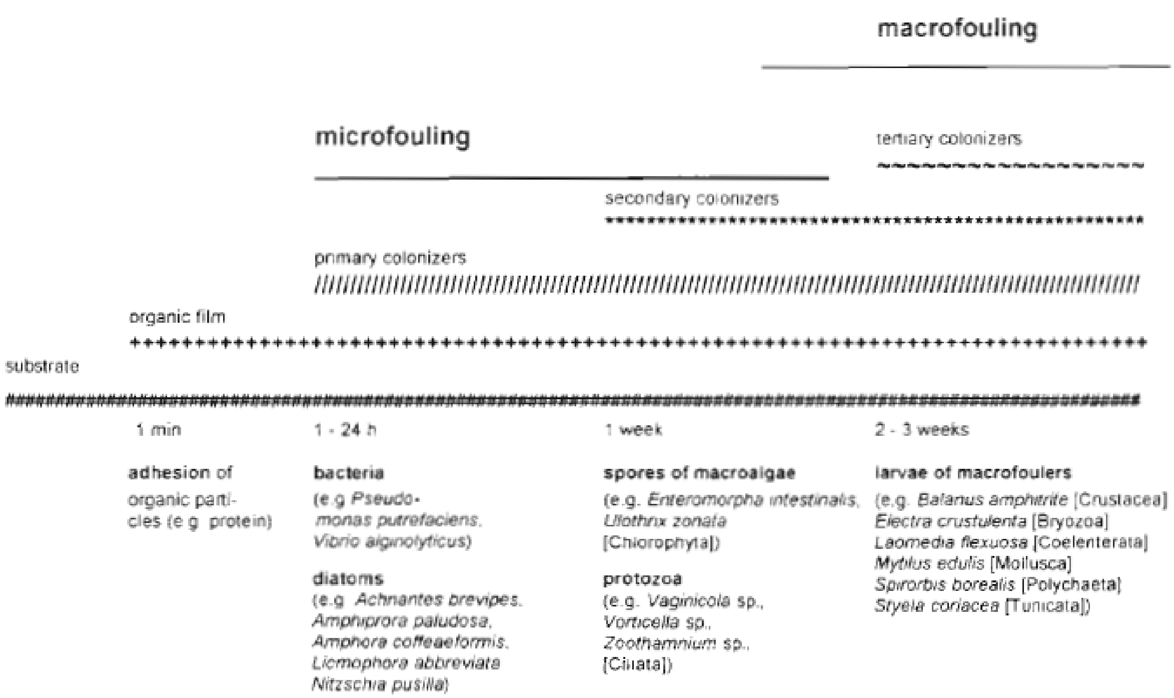

Figure 5-2. Temporal Structure of Settlement (from Abarzua and Jakubowski, 1995)

Recruitment of sessile marine invertebrates on vessel hulls is influenced by a suite of physical, biological, and chemical factors (Butman, 1987; Pawlik and Butman, 1993; Walters et al., 1999).

The factors likely to affect the rate of biofouling, which are also factors associated with the risk of NAS transport and introduction, include the following:

- **Level of Vessel Activity –** Actively operating ships tend to be less fouled than those that spend much of their time stationary and in ports. This is primarily because fouling organisms will attach only to immobile or slow-moving surfaces (Candries, 2009). Long stays in port increase opportunities for organisms to attach to hull surfaces (Davidson et al., 2009, Sylvester and MacIsaac, 2010, both in Sylvester et al., 2011).
- **Environmental Factors (salinity, temperature and nutrients)/Geographic Location -** The degree of aquatic biofouling can be highly variable, depending on geographical location, time of year, and seasonal variations in weather. In general, fouling flourishes in warmer waters and during warmer months and diminishes in colder waters and cooler months. Consequently, vessels in tropical and island ports are more susceptible to biofouling (Vitousek et al., 1987 in Godwin, 2003).
- **Age of the Anti-fouling Coating -** Vessels use AFCs to discourage hull growth that can create drag. AFCs contain or produce bioactive chemical agents. As the vessel remains submerged in seawater, the AFC "ages" and begins to lose effectiveness, allowing secondary biofouling processes to occur as organisms settle on top of the primary biofouling layer. The time since application of the AFC is considered one of the principal risk factors for hull fouling and is strongly related to the diversity and amount of fouling organisms on vessels (Coutts, 1999; Floerl et al., 2005; Ash, 2006b). AF paints used by the Navy can prevent fouling for up to 4-5 years, requiring a frequency of 0.21 cleanings per year (Bendick et al., 2010). Older coatings have been observed to be less effective because the activity of the biocidal toxins decreases and the coating comes off over time.
- **Time Elapsed Since the Last Hull Cleaning -** The amount of time passed since the last hull cleaning is another principal risk factor contributing to fouling on hulls (Cordell et al., 2009). Cleaning the hull refreshes the AFC, allowing it to perform better
- **Structural Complexity of the Hull –** Because a vessel's hull is not a uniform surface, biofouling is not evenly distributed on the submerged portion of a vessel. Fouling tends to be especially predominant in locations that either protrude from, or are recessed into, the hull and have greater structural complexity, providing refuge from exposure and more options for attachment. Military vessels typically have more hull structures that can provide more sheltered areas for fouling species to settle on than typical commercial ships (Global Invasive Species Programme, 2008), and organisms may be concentrated at more protected and structurally complex areas such as rudders, stern tubes, and intake grating.
- **Harbor Configuration -** The extent of vessel fouling is influenced by the configuration of the source harbor in which the vessel is ported. Floer and Inglis (2003) observed that organisms recruit in greater numbers in partially enclosed marinas and enclosed marinas than in open marinas or coastal reference locations

due to water circulation patterns that limit dispersal of planktonic propagules but increase propagule pressure adhering to surfaces, including vessel hulls.

5.2.1.3 Vessels of the Armed Forces Underwater Ship Husbandry Practices

Hull husbandry includes the mechanical removal of biofouling organisms as part of the underwater ship husbandry process. Underwater ship husbandry is defined as the inspection, grooming, maintenance, and repair of hulls and hull appendages performed while a vessel is waterborne. Hull-fouling organisms or viable fragments are dislodged during the process of out-of-water or underwater cleanings. Out-of-water cleanings during dry dock allow for the containment of materials, including fouling organisms that are removed from the vessel hull. In-water cleanings, however, may allow organisms to enter the water column and to be deposited on the seabed.

All vessels of the Armed Forces greater than 40 feet in length undergo regular, periodic underwater ship husbandry. This in-water hull cleaning is performed at nearly all major stateside bases. Hull cleanings can be either full cleanings, interim cleanings, or partial cleanings. Full cleanings are those which include the entire painted underwater hull surface, appendages, propellers and propeller shafts, and openings. Interim cleanings, occurring between full cleanings, are typically a cleaning of the running gear and may include partial cleaning of other ship systems. Partial cleaning includes a limited cleaning of a portion of the hull, appendages (running gear, stabilizers, etc.), or hull openings, and covers approximately 30% of the wetted hull area.

There are two types of hull paint which limit the attachment of organisms. The more comment paint is the ablative copper AFC (99% of vessels). The other is a silicone based foul-release coating. For most vessels, hull cleaning is performed when a very low percentage of the hull is coated. The exception is inactive reserve fleet vessels, which do not receive regular cleanings. Vessel hulls do not receive cleanings for the first three years, on average, after paint application. After the first three years, each active Navy surface ship will receive 0.75 full cleanings a year on average, or one full cleaning every 1.33 years. Vessels greater than 40 feet receive in-water hull cleanings; vessels 40 feet and under do not. Vessels between 40 and 79 feet receive full cleanings each time while vessels greater than 79 feet also receive interim cleanings every 3 months and partial cleanings every 6 months. These schedules vary regionally depending on fouling rates, water temperatures, and the coating service life (Ignacio Rivera-Duarte, personal communication).

Although Navy vessels are on a regular hull-cleaning schedule, the intervals between hull cleanings may be shorter, as effects of fouling on performance vary among ship class, and fouling intensity differs with type of coating, operational profile, and area of operation. The Naval Ships' Technical Manual Chapter 081, Waterborne Underwater Hull Cleaning of Navy Ships (NAVSEA, 2006) gives instruction for underwater hull inspections of coating fouling and damage. The decision to clean is based on regular inspections and performance criteria, including a fouling rating, and the amount of fouling may trigger a cleaning between the regularly scheduled cleaning and as close to deployment as possible when possible. One of the reasons for this is to reduce the likelihood that non-indigenous species will be transported to another location or picked up from another location (a fouled hull is more likely to be colonized by organisms than a clean hull).

UNDS also propose that "[v]essel hulls must be inspected, maintained, and cleaned to minimize the removal and discharge of AF hull coatings and transport of fouling organisms. To the greatest extent practicable, rigorous vessel hull cleanings must take place in drydock or at a land-based facility where the removed fouling organisms or spent AF hull coatings can be disposed of onshore in accordance with any applicable solid waste or hazardous substance management and disposal requirements. The proposed performance standard would also require that vessel hull cleanings be conducted in a manner that minimizes the release of AF hull coatings and fouling organisms (e.g., less abrasive techniques and softer brushes to the greatest extent practicable)." Also, "[f]or vessels less than 79 feet in length, the performance standard would require inspection of vessels before overland transport to a different body of water to control invasive species. For vessels greater than 79 feet in length, the performance standard would require that to the greatest extent practicable, vessel hulls with a copper-based AFC must not be cleaned within 365 days after the AFC application."

5.2.1.4 Factors Affecting the Introduction of Non-Indigenous Aquatic Species from In-water Hull Cleaning of Vessels of the Armed Forces

Underwater hull cleaning using currently-available technologies can discharge (release) viable fouling organisms into receiving waters where the cleaning takes place. If these fouling organisms accumulated on the vessel from different ports, then this discharge may include NAS. The introduction of NAS to a new region is achieved by two processes: (1) the transport of species from one location to another and (2) the establishment in the recipient port as a result of sufficient propagule pressure. There are several factors that affect the introduction and establishment of NAS from in-water hull cleaning including:

- How frequently vessels of the Armed Forces enter a port after deployment to another region – a greater number of deployments means a greater number of in-water cleanings and a greater likelihood that material removed will contain NAS
- How much time the vessel spends in a port of call – vessels that are inactive for longer periods of time in a port of call will have more hull fouling and a greater likelihood of fouling by NAS
- Environmental conditions of the port of call – fouling is more likely to occur in warmer temperate and tropical ports and during summer months
- Vessel travel speed – vessels that have traveled at higher speeds are likely to be less fouled because organisms could not "hold on" during transit (Takata et al., 2011; Minchin and Gollasch, 2003)
- Vessel travel distance – organisms that have survived longer distance transits are likely to be weaker than those that have survived shorter transits (Johnson et al., 2007; Ruiz and Smith, 2005 in Sylvester et al., 2011; Coutts, 1999)
- The amount of hull fouling – the greater the amount of fouling, the greater the propagule pressure in the discharge released during in-water cleaning
- The diversity of NAS – the greater the diversity of NAS, the more likely it is that something will survive during cleaning
- The method used to remove hull-fouling organisms – more abrasive methods will be more damaging to fouling organisms
- The type of hull-fouling organism present – encrusting organisms with an outer shell are more likely to be damaged

- When the vessel is cleaned – organisms on vessels that have recently returned to their home port are likely to be younger and less established than organisms that have had an opportunity to acclimate and grow on the hull in the vessel's home port; also, organisms that are removed from hulls are less likely to survive during colder months.

After NAS have become established in a new location, their ability to invade (i.e., become ANS) depends on that species' tolerance to a range of abiotic and biotic stressors, ability to propagate at the new location, and ability to displace native species. There is greater risk that ANS that have established populations at one location will succeed in invading other locations. The likelihood of invasion is based on:

- Whether the NAS is an ANS in the origination port.
- The number of NAS that have already become established in the destination port.
- Whether the environmental conditions of the destination region are optimal for the NAS to invade.

It has been estimated that approximately 10 – 15% of hull-fouling NAS will become established in a new region, and only up to 15% of NAS that become established will become invasive (OTA, 1993). This means that up to approximately 2 – 3% (15% of 15%) of hull biofouling organisms could pose a threat to federally listed species and their critical habitat. In addition, only about 100 vessels of the Armed Forces will be deployed each year, resulting in a total hull surface area of 0.0046 km^2 returning from deployment each year. Some of the deployed vessels may spend 3 weeks or more in a foreign port of call, allowing for hull-fouling organisms to be brought back to the home port and introduced when the vessel is cleaned. However, because the vessel will have been cleaned prior to deployment, refreshing the AFC, the hull-fouling community will be young (larvae and smaller organisms) and many animals are not likely to survive the transoceanic transport because of lower temperatures, potentially higher salinities, higher wave energy, and limited food resources. Therefore, it is **very unlikely** that hull-fouling NAS will be introduced to and become established in a port by cleaning a single vessel of the Armed Forces, become established, and invade a port or harbor. Increased propagule pressure from multiple vessels (commercial, recreational, and military), through cleaning and other means of introduction, would increase the likelihood of NAS introduction.

Assuming that environmental conditions at a new location are suitable for any hull-fouling NAS, the ability to become established will depend on whether individuals have reached reproductive maturity and numbers of propagules produced (fecundity) (Johnston, 2008). Growth form, the need for protection from biotic and abiotic sources of mortality, and refuge dimensions also influence settlement and short-term post-settlement success (Walters et al., 1996).

Only a small percentage of NAS that are introduced to a region will become invasive and outcompete native species (i.e., become ANS). It is a long-standing presumption in invasion biology that an increase in propagule pressure (inoculums abundance, density, and frequency) will increase the probability of a species establishing a population in a new region (NAS, 2011; SAB, 2011). Conversely, when propagule pressure decreases, the probability of a species becoming established should also decrease. This assumption regarding risk is supported by a wide body of empirical, theoretical, and experimental evidence showing that invasion likelihood increases with an increase in propagule pressure, either by a higher concentration of organisms in an inoculation and/or by an increase in the frequency of inoculations (e.g., Simberloff, 1989, 2009; Ruiz et al., 2000; Kolar and Lodge, 2001, Ruiz and Carlton, 2003; Lockwood et al., 2005; Johnston et al., 2008).

Species traits can also profoundly influence the survival, reproduction and establishment of invading organisms. These include genetics, life history characteristics, population density and abundance, habitat breadth, dispersal and mobility, and environmental matching. ANS (NAS that become invasive) tend to have specific traits or specific combinations of traits that allow them to outcompete native species. In some cases the competition is about rates of growth and reproduction. In other cases species interact with each other more directly. Common invasive species traits include:

- Parthenogenetic (asexual) and sexual reproduction;
- Fast growth to sexual maturity;
- Rapid reproduction;
- High dispersal ability (e.g., lecithotrophic or planktonic-dispersing larva that live off yolk supplied via the egg);
- Tolerance of a wide range of environmental conditions;
- Ability to forage on wide range of food types; and
- Lack of pathogens or predators.

Even if an organism does not become established in the area to which it is introduced and released, it could continue to be transported elsewhere through secondary measures.

5.2.1.5 Threats from Aquatic Nuisance Species Invasions

ANS threaten the diversity or abundance of native species and the ecological stability of the aquatic ecosystems they impact. They can permanently reduce biodiversity by preying on, parasitizing, or out-competing native species, causing or carrying diseases, or altering habitats of native species (Convention on Biological Diversity, 2005). Marine ANS can change food webs and disrupt ecosystem services, including those provided by marine wetland plants (salt marshes, sea grasses). In marine and coastal environments, invasive species have been identified as one of the four greatest threats to the world's oceans (GISP, 2008). ANS have also been cited as the second largest threat to endangered species after habitat loss (Wilcove and Chen, 1998), and the ANS Task Force has cited that 42% of all listed species have been significantly impacted by NIS (http://www.anstaskforce.gov/more_impacts.php accessed on 11 November 2016). However, as understanding of biological invasions has grown, some scientists have come to believe that invasive species have an even greater impact on biodiversity than habitat loss (Johnson et al., 2007).

The abundance and distribution of an ANS and the magnitude of its impact on native species, habitat, and ecosystem function can vary greatly (Olenin et al., 2007). Any ANS may have a range of impacts depending on the specific conditions to which it is introduced. In one region, a species may have no noticeable effect, while elsewhere it may have a strong impact. ANS can impact native species through competition, predation, infection or toxicity that could lead to changes in:

- Community composition (by displacing native species);
- Water quality conditions (release of nutrients by burrowing species or removal of nutrients, plankton and suspended material by filter feeders);

- Food web dynamics;
- Biogeochemical processes, and physical habitat.

Coastal ecosystems (estuaries, bays and lagoons) and inland water bodies (rivers and lakes) are most vulnerable to ANS invasions because they are protected and are more likely to provide environmental conditions that allow NAS to become established. Hull-fouling NAS that are exposed to external environmental conditions are unlikely to survive in offshore environments (Floerl and Inglis, 2003). There are three general sequential processes by which ANS invasions occur: (1) introduction of NAS to a new location; (2) establishment of a NAS population at the new location; and (3) spreading of ANS that impact native species at the new location. Invasion by ANS can potentially lead to exposure of federally listed threatened or endangered species to ANS.

However, determining risk to listed species from ANS introductions by in-water hull cleaning is difficult because of the number of factors that affect the introduction of NAS to a new location, the establishment of a NAS in the new location, and the potential of a NAS population to become invasive and impact native species. For that reason, a qualitative but methodical approach has been developed for assessing the risk of impact to federally listed species from ANS introduced by in-water cleaning of vessels of the Armed Forces using current hull cleaning practices and proposed hull- cleaning standards. The risk assessment approach follows standard methods used for offshore environmental impact assessments and is generally based on the International Finance Corporation and European Union performance standards for the assessment and management of environmental and social risks and impacts and methods used by the International Association for Impact Assessment. The approach described in this BE includes background information that provides an understanding of the factors affecting the likelihood of NAS introductions to inform the five steps of the risk assessment process. That approach is summarized below and detailed in Appendix H.

5.2.1.6 Approach to the Assessment of Risk from Aquatic Nuisance Species Invasion from In-Water Cleaning of Vessels of the Armed Forces

Risk is a function of the likelihood of exposure and the magnitude of potential effect. To summarize:

Risk of Impact = Likelihood of Exposure x Magnitude of Effect

The impact of a non-native species may range from no impact at all to major impacts potentially resulting in extinction. Most NAS will arrive without any noticeable effect on the native community at all (Williamson, 1996 in Olenin, 2007). A measurable effect will only occur when a NAS reaches an abundance that allows it to outcompete native species.

The assessment of risk to federally listed species from ANS introductions by vessels of the Armed Forces using current hull cleaning practices and performance standards is based on the assessment of three factors: (1) the likelihood of introduction and establishment of NAS from in-water hull cleaning, (2) the likelihood of ANS invasion, and (3) the magnitude of potential consequences for federally listed species and their critical habitats should an ANS invasion occur. Risks to listed species from ANS are assessed as a five step process. The first three steps of the process help determine the likelihood of exposure, while steps 4 and 5 determine the potential magnitude of effect. Figure 5-3 summarizes the steps in the process.

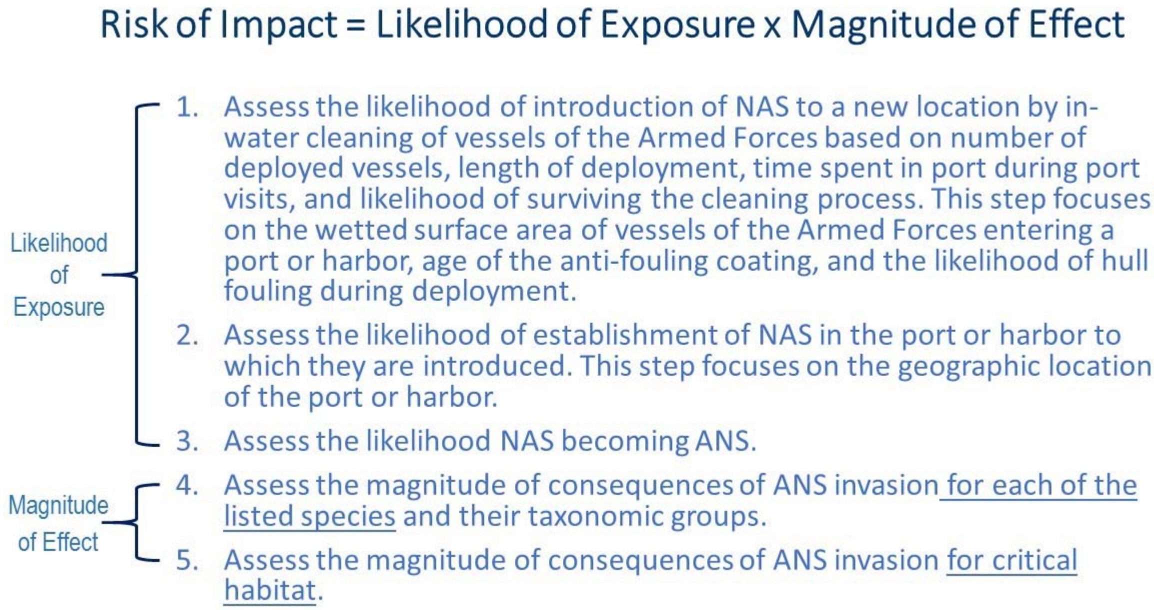

Figure 5-3. Summary of the Five-Step Process for Assessing Risk of Impact to Federally Listed Aquatic and Aquatic-Dependent Species from Aquatic Nuisance Species Introductions by In-Water Hull Cleaning

One means of reducing the risk of transporting NAS to another location, thereby resulting in exposure, is to clean the hull prior to transit. As discussed, vessels of the Armed Forces are inspected prior to deployment and cleaned if necessary. In-water cleaning considered to be an acceptable approach because most of the fouling organisms will be indigenous to the waterbody where the vessel has either ported or remained inactive. Cleaning prior to deployment also refreshes the AFC allowing it to perform more effectively at reducing the rate of biofouling. Vessels typically are not cleaned following port calls prior to returning to their home port.

The active vessels of the Armed Forces carry a low risk of species transport and invasion because they are well maintained, generally are not heavily fouled prior to transit, and the majority of vessels operate within their home port area. The active vessels with the highest likelihood of introducing NAS are the 284 battle forces vessels that could be deployed and spend some time in foreign ports of call. However, the other approximately 850 vessels greater than 79 feet in length could also contribute to risk of NAS introductions if they travel between U.S. ports and harbors and spend any time in a port other than their home port.

Inactive (reserve fleet) vessels that are transported for dismantling or for preservation as a memorial are of special concern because they have spent an extraordinarily long time in ports without regular cleaning and maintenance. As a consequence, their hulls are likely to be heavily fouled by organisms that may include non-native species of concern. Inactive vessels may or may not be cleaned prior to movement. If they are cleaned prior to movement, it can be assumed that all of the fouling organisms removed from the hull of the vessel are from the vessels home port and that there is greater risk from the BOD of the discharge and the AF chemicals in the paint that is also removed during the cleaning.

Because hull-fouling organisms are most likely to attach to a vessel while it is in port and not while it is in transit, and because introductions are most likely to impact protected water bodies such as inland freshwater bodies (lakes and rivers), estuaries, and shallower protected coastal areas, the assessment of risk from ANS invasion focuses on risk of NAS introduction and ANS invasion in ports and harbors, specifically those where vessels of the Armed Forces occur and are cleaned.

For this BE, the challenge of determining risks to listed species from exposure to ANS introduced by vessels of the Armed Forces is that it requires a holistic global approach that is complicated by the number of ports and harbors supporting and visited by vessels of the Armed Forces, the complexity of vessel movement patterns between ports and harbors, the range of environmental conditions among ports and harbors, and the diversity of biofouling organisms and their species-specific environmental tolerances. It is most likely that hull-fouling NAS will be introduced the next time a vessel is cleaned after a deployment. Most hull fouling will occur during inactive periods in a vessel's home port, and hulls are typically inspected and cleaned prior to deployment. Therefore, most fouling organisms will be from the vessel's home port. Vessel hulls are not typically cleaned during port calls while they are deployed; however, port calls are infrequent and usually short (less than five days). Therefore, the likelihood of introducing NAS from a vessel's home port to a port of call is considered to be low.

Cleaning before deployment refreshes the AFC, improving its ability to inhibit fouling, especially when the vessel is underway. However, it is possible for hull fouling to occur while a vessel is deployed should the vessel spend any time in a foreign port or coastal waters. Therefore, risk is assessed from the perspective of introducing NAS, which could potentially become ANS, upon return to a vessel's home port or another port within the action area. A qualitative assessment is the only feasible way to assess the risk of impact to federally listed species in U.S. ports and harbors where vessels of the Armed Forces are home ported and cleaned.

Because this is not an assessment of risk from the movement of vessels but rather risk of introduction from in-water hull cleanings, the assessment approach does not account for differences in the likelihood of NAS establishment and ANS invasion across larger geographic regions. Once deployed, vessels of the Armed Forces could travel anywhere in the world. Therefore, the location, fouling species, and environmental conditions of the destination ports are not considered in the assessment. The approach does not consider what species could be fouling vessel hulls, and it is conservatively assumed that the likelihood of introduction, establishment and invasion is not affected by any differences in environmental conditions between hull-fouling organism origination and destination ports, except that marine and estuarine species can only be introduced to other marine (>30 ppt) or estuarine (0.5 – 30 ppt) ports and freshwater species can only be introduced to other freshwater ports (<0.5 ppt).

The assessment criteria for determining the likelihood of ANS introduction, establishment, and invasion and likelihood definitions are summarized in Table 5-4. A detailed description of the approach to assessing risks to federally listed species from exposure to ANS is provided in Appendix H. It is important to note that the criteria and definitions for this assessment are specific to fouling on vessels of the Armed Forces and may not be applicable to other vessels or introductions via other vectors such as ballast water or other vessel discharges. Although the assessment is qualitative, the likelihood of introduction, establishment and invasion was assessed and ranked categorically using a semi-quantitative approach. For each of the assessment criteria, a score of 1 (very unlikely), 2 (unlikely), 3 (likely), or 4 (very likely) was assigned based on definitions provided in Table 5-4 and the sum total for all of the assessment criteria is divided by the number of assessment criteria for each step of the assessment process. For each step in the process, the average score, also ranging from 1 – 4, reflects the overall likelihood of introduction, establishment and impact from ANS introduced by vessels of the Armed Forces such that:

- A score of 0 – 1 indicates that introduction, establishment or invasion is very unlikely.
- A score of >1 - 2.4 indicates that introduction, establishment or invasion is unlikely.
- A score of 2.5 - 3.5 indicates that introduction, establishment or invasion is likely.
- A score of 3.5 and higher indicates that introduction, establishment or invasion is very likely.

Because limited information is available for quantifying how the different criteria affect the likelihood of NAS introduction and establishment and ANS invasion, the categorical definitions are somewhat subjective and developed to provide a means to reproduce results and allow others to perform a comparable assessment. There is a level of uncertainty associated with the approach that cannot be quantified, but the approach allows a logical, step-wise progression to the assessment and a reasonable level of relative risk to be determined.

Table 5-4. Likelihood of Non-Indigenous Aquatic Species Introduction, Establishment, and Invasion from In-Water Hull Cleaning

Assessment Criteria		Likelihood (Rank)			
		Very Unlikely (1)	Unlikely (2)	Likely (3)	Very Likely (4)
Likelihood of Fouling by NAS and Introduction During Hull Cleaning	How many vessels could potentially be deployed (vessels >79 feet; proxy for wetted surface area and number of in-water cleanings)?	1 – 2	3 – 10	10 – 50	>50
	What is the age of the anti-fouling coating?	0 – 3 years (considered very unlikely based on being well within coating performance term)	>3 years through the term of the dry dock cycle (considered unlikely based on regular inspection and cleaning, in addition to pre-deployment cleaning, and being within coating performance)	Term of the dry dock cycle to 5 years beyond the dry dock cycle (considered likely based on being outside of coating performance term)	>5 years beyond the dry dock cycle (considered very likely based on being well outside of coating performance term)
	How much time has passed since the last hull cleaning prior to deployment?	0 – 1 months	>1 – 3 months	>3 – 12 months	> 12 months
	How much time will the vessel spend in a port of call?	none	0 – 1 weeks	>1 – 3 weeks	>3 weeks
	How much time will the vessel spend in transit?	>21 days	7 – 21 days	3 – 7 days	0 – 2 days
	What in-water hull cleaning method was used?	Cleaning in dry dock	Cleaning methods that capture material	Methods using scrapers	Methods using soft brushes or cloths

Table 5-4. Likelihood of Non-Indigenous Aquatic Species Introduction, Establishment, and Invasion from In-Water Hull Cleaning (Continued)

Assessment Criteria		Likelihood (Rank)			
		Very Unlikely (1)	Unlikely (2)	Likely (3)	Very Likely (4)
Likelihood of Establishment	**Where is the vessel's home port located (latitude)?**	Arctic	Subarctic	Temperate	Tropical
	On what U.S. coast is the home port located?	Alaska	Washington and Oregon	Pacific (other than Washington and Oregon) and Atlantic	Hawaii and Florida
	On what type of freshwater body is the home port located?	Tributary to a Large River >5 Miles from the Larger River	Tributary to a Large River Within 5 Miles of the Larger River	Large Interior River	Large River Near an Estuary or Open Coastline; Lake
	How protected is the port?	Open coastal	Partially protected harbor	Near mouth of a protected harbor or estuary	Inside a protected harbor or estuary
	How many new non-indigenous fouling organisms are likely to occur in the hull-fouling community on the vessel (assuming 15% of hull-fouling species will become established)?[48]	None	Very Few (e.g., 1 – 2)	Some (e.g., 3 – 5)	Many (e.g., >5)
	How invaded is the home port by NAS?	Not Invaded	Slightly Invaded	Moderately Invaded	Highly Invaded

[48] U.S. Congress, Office of Technology Assessment. 1993. Harmful Non-Indigenous Species in the United States, OTA-F-565 (Washington, DC: U.S. Government Printing Office, September 1993).

Table 5-4. Likelihood of Non-Indigenous Aquatic Species Introduction, Establishment, and Invasion from In-Water Hull Cleaning (Continued)

Assessment Criteria		Likelihood (Rank)			
		Very Unlikely (1)	**Unlikely (2)**	**Likely (3)**	**Very Likely (4)**
Likelihood of Invasion (NAS becoming ANS)	**How many introduced species can potentially become invasive (assuming 15% of NAS become ANS)?**	If only 1 new hull-fouling NAS is introduced, there is less than 15% chance that it will become invasive. Therefore, invasions are very unlikely	If 2-3 new hull-fouling NAS are introduced, there is less than 50% chance that any of them will become invasive. Therefore, invasions are unlikely.	If 4-5 new hull-fouling NAS are introduced, there is a greater than 50% chance that one of them will become invasive. Therefore, invasions are likely.	If more than 6 new hull-fouling species are introduced, there is a nearly 100% chance that one of them will become invasive. Therefore, invasions are very likely.

The magnitude of the consequences that an ANS introduction could have on a federally listed species and/or its critical habitat reflects the level of impact threat to that species or critical habitat. In determining the potential magnitude of consequences, the EPA and DoD have considered where the federally listed species occurs (location within the RAA), whether the species is already threatened either directly or by loss of resources as a result of ANS species invasion, how fouling ANS could affect that species (e.g., competition for resources), and how critical habitat could be altered by ANS. The magnitude of consequences that ANS could have on federally listed species and their critical habitat is categorized and defined in Table 5-5.

Table 5-5. Magnitude of Potential Consequences of Aquatic Nuisance Species

Undetectable	Minor	Moderate	Major
Fouling organisms will not directly affect or compete with the species for any resources	Fouling organisms could affect resources, but not in a way that will result in noticeable changes in behavior or population level impacts	Fouling organisms could directly impact or compete with the species in a way that could result in a noticeable change in behavior without affecting current population levels	Fouling organisms are likely to directly impact or compete with the species in a way that leads to a population reduction either locally or throughout its range, extirpation, or extinction

Risk is then determined within a matrix of likelihood of introduction, establishment, and invasion (i.e., exposure) and consequences of the introduction by matching the likelihood and the magnitude of consequence within the matrix shown in Table 5-6. The risk level is given by the matrix cell in which the likelihood row and consequence column intersect.

Table 5-6. Risk Definitions Based on Likelihood of Exposure and Magnitude of Potential Consequences

Likelihood of Exposure to ANS	Magnitude of Consequences from Exposure to ANS			
	Undetectable	Minor	Moderate	Major
Very Unlikely	Remote Risk	Remote Risk	Negligible Risk	Negligible Risk
Unlikely	Remote Risk	Remote Risk	Negligible Risk	Potentially Significant Risk
Likely	Remote Risk	Negligible Risk	Potentially Significant Risk	Likely Significant Risk
Very Likely	Remote Risk	Negligible Risk	Potentially Significant Risk	Likely Significant Risk

Note: The risk matrix and definitions are general and applicable to all qualitative assessments. Although these risk levels are based on likelihood of exposure to ANS and magnitude of consequences from exposure to ANS, it could be based on likelihood of exposure and magnitude of consequences from exposure to any stressor.

Consistent with the Batch One BE and VGP BE, risk levels are defined as:

- Remote – NAS are either:
    - very unlikely to become ANS and effects on a listed species population or its critical habitat are expected to be minor;

    - unlikely to become ANS and effects on a listed species or its critical habitat are expected to be minor; or
    - the effects on a listed species population or its critical habitat are expected to be undetectable.

- Negligible – NAS are:
    - very unlikely to be exposed to ANS but effects on a listed species population or its critical habitat are expected to be moderate or major if invasions do occur; or
    - unlikely to be exposed to ANS but effects on a listed species population or its critical habitat are expected to be moderate if an invasion does occur; or
    - likely or very likely to be exposed to ANS and ANS are expected to have minor effects on a listed species population or its critical habitat.

- Potentially significant – NAS are either:
    - unlikely to become ANS but expected to have major effects on a listed species population or its critical habitat if invasions do occur, or
    - likely or very likely to become ANS and expected to have moderate effects on a listed species population or its critical habitat.

- Likely significant – NAS are either likely or very likely to become ANS and are expected to have major effects on a listed species or its critical habitat.

5.2.1.7 Assessment of Risk to Federally Listed Species from Exposure to Aquatic Nuisance Species Introduced by In-Water Cleaning of Vessels of the Armed Forces

Using this approach, which is detailed in Appendix H, conclusions have been drawn for each of the 111 federally listed species and 26 critical habitats that occur in the RAAs for this BE. The conclusions for these species and critical habitats will be used to help inform the effects determinations for all 674 federally listed aquatic and aquatic-dependent species in the action area for the proposed action.

The likelihood of NAS introduction and establishment and ANS invasion was assessed for each of the RAAs used for the BE. As previously mentioned, it is assumed that the source region is inconsequential to the introduction and establishment of NAS and that NAS have the potential to be introduced from all regions. The likelihood of establishment of NAS and subsequent ANS invasion is largely assessed based on propagule pressure (i.e., the number of vessels that could be deployed, which is a surrogate for the number of hull cleanings), the amount of time a vessel is likely to spend in a port of call, distances traveled, time since the last inspection and cleaning, and other factors. Conclusions of the ANS exposure assessment are summarized in Table 5-7. Risk Conclusions are summarized in Table 5-8.

Table 5-7. Likelihood of Non-Indigenous Aquatic Species Introduction by In-Water Cleaning of Vessels of the Armed Forces, Establishment, and Invasion That Could Result in Exposure of Listed Species to Aquatic Nuisance Species (likelihood rank indicated in parentheses)

<table>
<tr><th colspan="2" rowspan="2">Assessment Criteria</th><th colspan="7">Representative Action Area (RAA)</th></tr>
<tr><th>Miami</th><th>Norfolk</th><th>Pearl Harbor</th><th>Puget Sound</th><th>San Diego</th><th>San Francisco</th><th>St. Louis</th></tr>
<tr><td rowspan="3">Likelihood of Fouling and Introduction</td><td>How many vessels could potentially be deployed (# vessels >79 feet; proxy for wetted surface area entering the RAA and number of in-water cleanings)?</td><td>9
Unlikely (2)</td><td>267
Very Likely (4)</td><td>101
Very Likely (4)</td><td>114
Very Likely (4)</td><td>140
Very Likely (4)</td><td>17
Likely (3)</td><td>1
Very Unlikely (1)</td></tr>
<tr><td>What is the age of the anti-fouling coating (AFC)?</td><td colspan="7">Hull coatings on vessels of the Armed Forces are designed for a 5-, 7- or 12-year dry docking cycle. Most vessels remain free of fouling for the first 3 years after application and then are kept clean by regular hull inspection and in-water cleaning. Most vessels of the Armed Forces are fouled during inactive periods in their home port. Therefore, it is unlikely that even vessels with older AFC will be fouled by species from outside of their home port.
Unlikely (2)</td></tr>
<tr><td>How much time has passed since the last hull cleaning prior to deployment?</td><td colspan="7">Vessels of the Armed Forces are typically inspected and (if they are fouled) cleaned prior to deployment. Any fouling organisms that are removed are most likely from the home port and/or are destroyed during removal. Vessels that have been cleaned within the past 30 days are also expected to be relatively free of fouling organisms. When vessels are relatively free of hull-fouling organisms prior to deployment, they are less likely to become more fouled while deployed. Therefore, it is unlikely that NAS will foul the hull while the vessel is deployed.
Unlikely (2)</td></tr>
</table>

Table 5-7. Likelihood of Non-Indigenous Aquatic Species Introduction by In-Water Cleaning of Vessels of the Armed Forces, Establishment, and Invasion That Could Result in Exposure of Listed Species to Aquatic Nuisance Species (likelihood rank indicated in parentheses) (Continued)

Assessment Criteria		Representative Action Area (RAA)						
		Miami	**Norfolk**	**Pearl Harbor**	**Puget Sound**	**San Diego**	**San Francisco**	**St. Louis**
	How much time will vessels likely spend in a destination port? (Note: Most planned port calls are 1 – 3 days.)	6 – 22 days (Caribbean, Mediterranean, Middle East, North Atlantic and South America) Likely (3)	6 – 22 days (Caribbean, Mediterranean, Middle East, North Atlantic and South America) Likely (3)	11 – 22 days (Middle East, South America, West Pacific) Likely (3)	11 – 22 days (Middle East, South America, West Pacific) Likely (3)	11 – 22 days (Middle East, South America, West Pacific) Likely (3)	11 – 22 days(Middle East, South America, West Pacific) Likely (3)	0 days Very Unlikely (1)
	How much time will vessels of the Armed Forces likely spend in transit, and how far and fast are they likely to travel?	Most vessels that are deployed will be traveling long distances at relatively high speeds to reach their destination and return to their home port (USCG, 2011). In port, vessels will travel at 5 – 7 knots, and vessels will travel at 10 – 12 knots while underway. Although these are not speeds that are likely to knock all biofouling organisms from hulls, because the organisms will be exposed to harsher open water conditions (e.g., low food availability, colder temperatures, greater wave action) for prolonged periods of time, and because the AFC has been refreshed prior to deployment, hull-fouling organisms could become weak and release from the vessel's hull. Travel times for vessels deployed overseas are generally greater than 2 weeks, and transit routes are across open ocean at higher speeds lowering the ability of any fouling organisms to survive the transit. Therefore, it is unlikely that many hull-fouling non-indigenous species will survive and be introduced to the vessel's home port. Unlikely (2)						

Table 5-7. Likelihood of Non-Indigenous Aquatic Species Introduction by In-Water Cleaning of Vessels of the Armed Forces, Establishment, and Invasion That Could Result in Exposure of Listed Species to Aquatic Nuisance Species (likelihood rank indicated in parentheses) (Continued)

<table>
<tr><th colspan="2" rowspan="2">Assessment Criteria</th><th colspan="7">Representative Action Area (RAA)</th></tr>
<tr><th>Miami</th><th>Norfolk</th><th>Pearl Harbor</th><th>Puget Sound</th><th>San Diego</th><th>San Francisco</th><th>St. Louis</th></tr>
<tr><td></td><td>What in-water hull cleaning method was used?</td><td colspan="7">All methods of in-water cleaning are destructive to some degree. Vessels are cleaned prior to deployment and remain relatively clean when deployed because most of their time is spent underway or offshore. Therefore, the diversity of fouling organisms on vessel hulls is relatively low. Although scrapers are most damaging, even brushes and water jets can damage organisms, reducing their likelihood of survival. It is expected that only 10 – 15% of hull-fouling species will be introduced to and become established in a new location (Williamson and Fitter, 1996, OTA, 1993). Therefore, introduction of NAS by in-water hull cleaning is considered to be unlikely.
Unlikely (2)</td></tr>
<tr><td rowspan="3">Likelihood of Establishment (Location-Specific)</td><td>Where is the home port located (latitude)?</td><td>Tropical
Very Likely (4)</td><td>Temperate
Likely (3)</td><td>Tropical
Very Likely (4)</td><td>Temperate
Likely (3)</td><td>Temperate
Likely (3)</td><td>Temperate
Likely (3)</td><td>Temperate
Likely (3)</td></tr>
<tr><td>On what coast is the home port located?</td><td>Atlantic
Likely (3)</td><td>Atlantic
Likely (3)</td><td>Hawaii
Very Likely (4)</td><td>Pacific - Washington and Oregon
Unlikely (2)</td><td>Pacific - Other
Likely (3)</td><td>Pacific - Other
Likely (3)</td><td>NA</td></tr>
<tr><td>On what type of freshwater body is the home port located?</td><td>NA</td><td>NA</td><td>NA</td><td>NA</td><td>NA</td><td>NA</td><td>Upper Mississippi River Basin
Likely (3)</td></tr>
</table>

Table 5-7. Likelihood of Non-Indigenous Aquatic Species Introduction by In-Water Cleaning of Vessels of the Armed Forces, Establishment, and Invasion That Could Result in Exposure of Listed Species to Aquatic Nuisance Species (likelihood rank indicated in parentheses) (Continued)

Assessment Criteria		Representative Action Area (RAA)						
		Miami	**Norfolk**	**Pearl Harbor**	**Puget Sound**	**San Diego**	**San Francisco**	**St. Louis**
	How protected is the port?	Partially Protected Unlikely (2)	Inside Protected Estuary Very Likely (4)	Near Mouth of Estuary Likely (3)	Inside Protected Estuary Very Likely (4)	Inside Protected Estuary Very Likely (4)	Near Mouth of Estuary Likely (3)	Open Very Unlikely (1)
	How many new non-indigenous fouling organisms are likely to occur in the hull-fouling community on the vessel (assuming 15% of hull-fouling species will become established)?[49]	Vessels of the Armed Forces are most likely to be fouled by organisms from their home port rather than non-indigenous species from outside of their home port. Vessels are inspected and, if necessary, cleaned prior to deployment, removing most hull fouling and refreshing the AFC. Cleaning vessels prior to deployment reduces the amount of fouling that will occur while the vessel is deployed, and fouling is most likely to occur if the vessel makes port calls. Because vessels of the Armed Forces are not expected to be substantially fouled by new non-indigenous species upon return to their home port where underwater ship husbandry will occur, it is unlikely that any new non-indigenous species will become established. Unlikely (2)						
	How invaded is the home port by NAS?	Moderately Invaded Likely (3)	Moderately Invaded Likely (3)	Highly Invaded Very Likely (4)	Minimally Invaded Unlikely (2)	Moderately Invaded Likely (3)	Highly Invaded Very Likely (4)	Moderately Invaded Likely (3)

[49] Number of potential hull fouling organisms is based on the maximum number of species and taxonomic groups identified on vessel hull in a study conducted by Davidson et al. (2006). The study team identified 32 unique species or groups of species fouling the ships investigated. "The two most species-rich vessels arrived in the Lower Columbia from overseas, had not spent much time in freshwater prior to docking, and were fouled with organisms, many of which were probably non-indigenous to the Pacific Northwest region." Vessels that operated solely within salt water or freshwater and that had not been cleaned within the last two years, which is generally less frequent than hull-cleaning for vessels of the Armed Forces, tended to have higher levels of fouling.

Table 5-7. Likelihood of Non-Indigenous Aquatic Species Introduction by In-Water Cleaning of Vessels of the Armed Forces, Establishment, and Invasion That Could Result in Exposure of Listed Species to Aquatic Nuisance Species (likelihood rank indicated in parentheses) (Continued)

<table>
<tr><th colspan="2" rowspan="2">Assessment Criteria</th><th colspan="7">Representative Action Area (RAA)</th></tr>
<tr><th>Miami</th><th>Norfolk</th><th>Pearl Harbor</th><th>Puget Sound</th><th>San Diego</th><th>San Francisco</th><th>St. Louis</th></tr>
<tr><td>Likelihood of Invasion</td><td>How many introduced species can potentially become invasive (assuming 15% of hull-fouling species will become established)?</td><td colspan="7">Because home ports occupied by vessels of the Armed Forces are generally more active and, therefore, more highly invaded, they are more vulnerable to NAS introduction and ANS invasion. However, because of hull maintenance practices, vessels of the Armed Forces are less likely to be vectors for new NAS introductions and establishment followed by ANS invasion. Furthermore, some of the species that would be introduced during underwater ship husbandry would also be damaged during the cleaning and unlikely to propagate.
Unlikely (2)</td></tr>
</table>

Table 5-8. Conclusions from Assessing the Likelihood of Non-Indigenous Aquatic Species Introduction by In-Water Cleaning of Vessels of the Armed Forces, Establishment and Impact for Each Representative Action Area (likelihood score in parentheses)

Likelihood Assessment Step	Likelihood (and Average Rank of Assessment Criteria from Table H-8) for Each Representative Action Area (RAA)						
	Miami	Norfolk	Pearl Harbor	Puget Sound	San Diego	San Francisco	St. Louis
Likelihood of Fouling and Introduction	Unlikely (2.1)	Unlikely (2.3)	Unlikely (2.3)	Unlikely (2.3)	Unlikely (2.3)	Unlikely (2.2)	Unlikely (1.8)
Likelihood of Establishment	Likely (2.8)	Likely (3)	Likely (3.4)	Likely (2.6)	Likely (3)	Likely (3)	Unlikely (2.4)
Likelihood of Invasion	Unlikely (2)						
Likelihood of Exposure of a Listed Species to ANS Introduced by Vessels of the Armed Forces (Average of Introduction, Establishment, and Invasion)	Unlikely (2.3)	Unlikely (2.4)	Likely (2.6)	Unlikely (2.3)	Unlikely (2.4)	Unlikely (2.4)	Unlikely (2.0)

In most cases, the act of performing an in-water hull cleaning on vessels of the Armed Forces is unlikely to directly expose federally listed species to invasive species. Exposure could result in an impact. Forunately due to paints, vessel speeds, and changes in aquatic environments the transfer of invasive hull-fouling species is minimized. However, exposure of federally listed species from hull-fouling ANS is considered because of vessel transit paths, the number of vessels being cleaned, port configuration, climate, and level of invasion.

Exposure to biofouling species could potentially affect federally listed species in the 19 taxonomic groups evaluated in this BE. Highest potential impacts to species within each taxonomic group of federally listed species were evaluated based on available information for surrogate species. The likelihood and consequences of ANS introductions for each of the 111 federally listed species in the RAAs and risk conclusions are summarized in Table 5-9. Each aquatic and aquatic-dependent species evaluated has been assigned to a taxonomic group in order to make general conclusions regarding impacts to federally listed species. The results of the impact assessment for each species present in the RAAs was extrapolated to its species group to evaluate risk from ANS introductions to different groups of species. It is important to note that risk is based on the vulnerability of each taxonomic group to an invasive species based on what is already known about the individual species within that taxonomic group. For the assessment, the EPA and DoD considered the worst potential consequences for each species from exposure to any hull-fouling ANS. Table 5-9 summarizes the highest level of risk for each of the 19 major taxonomic groups evaluated for this risk assessment. Although there is “potentially significant” risk to some species and taxonomic groups, this will not necessarily lead to a “may affect, likely to adversely affect” determination. The effect determination will be based on the likehood of exposure to the discharges along with the consequences of the action, which is the implementation of UNDS and designed to mitigate the risk.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors

MIAMI RAA Number of deployable vessels (>79 feet) – 3 Existing ANS[50]: Asian Tiger Shrimp (*Panaeus monodon*), Lionfish (*Pterois volitans*), Green Crab (*Carcinus maenas*), Chinese Mitten Crab (*Eriocheir sinensis*)			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**	**Consequences of ANS Invasion**	**Rationale Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-Fouling ANS			
Bat, Florida bonneted, *Eumops floridanus*	Very Unlikely	Minor	Miami RAA is included in their home range; diet consists of flying insects that have an aquatic life stage that may occur either within or outside of the RAA. Hull-fouling ANS could impact aquatic prey but have only minor consequences for Florida bonneted bat because of their broad feeding range. However, introductions of hull-fouling ANS from in-water cleaning of vessels of the Armed Forces will not impact most of the Florida bonneted bat's feeding range, and exposures are very unlikely to occur.
Butterfly, Miami blue, *Cyclargus (=Hemiargus) thomasi bethunebaker*	Very Unlikely	Undetectable	Does not have an aquatic life stage but is dependent on host plants that occur in coastal wetlands that are not near locations where in-water hull cleanings of vessels of the Armed Forces are performed. Therefore, exposure to ANS introduced by in-water cleanings is very unlikely.

[50] List of ANS is not comprehensive and identifies some of the most notably impactful ANS for each RAA.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

MIAMI RAA **Number of deployable vessels (>79 feet) – 3** **Existing ANS[50]: Asian Tiger Shrimp (*Panaeus monodon*), Lionfish (*Pterois volitans*), Green Crab (*Carcinus maenas*), Chinese Mitten Crab (*Eriocheir sinensis*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**	**Consequences of ANS Invasion**	**Rationale Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-Fouling ANS			
Crocodile, American, *Crocodylus acutus*	Unlikely	Undetectable	Primary habitat is inland mangrove swamps protected from wave action; biggest threats are loss of habitat and egg predation by raccoons. Exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces is unlikely, and consequences of ANS invasions are expected to be undetectable for crocodiles.
Kite, Everglade snail, *Rostrhamus sociabilis plumbeus*	Very Unlikely	Undetectable	Although may be observed near Miami RAA, habitat consists of large, open freshwater marshes and lakes with shallow (< 4 ft) open waters; open water areas without emergent vegetation are required for foraging; diet consist predominantly of apple snails. Most of these areas are not connected to waterbodies where vessels of the Armed Forces are cleaned; therefore, exposure of kites and their food resources to hull-fouling ANS introduced by in-water hull cleaning is very unlikely.
Plover, Piping, *Charadrius melodus*	Unlikely	Minor	The Miami RAA is within the wintering range for this piping plovers. Feeds on worms, fly larvae, beetles, crustaceans, mollusks, and other invertebrates. Composition of diet could be affected by hull-fouling ANS but is unlikely to affect food availability and feeding behavior. In addition, exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces in the Miami RAA is unlikely.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

MIAMI RAA Number of deployable vessels (>79 feet) – 3 Existing ANS[50]: Asian Tiger Shrimp (*Panaeus monodon*), Lionfish (*Pterois volitans*), Green Crab (*Carcinus maenas*), Chinese Mitten Crab (*Eriocheir sinensis*)			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**	**Consequences of ANS Invasion**	**Rationale Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-Fouling ANS			
Rail, Eastern black, *Laterallus jamaicensis ssp. jamaicensis*	Unlikely	Minor	Inhabits fresh and saline marshes, wet meadows and savannas. Occupies marshes with shallower water than other rallids and requires some tall vegetation to escape into. Feeds on terrestrial and aquatic invertebrates. Uses impoundments (managed wetlands) to forage and nest. Composition of diet could be affected by hull-fouling ANS but is unlikely to affect food availability and feeding behavior. In addition, exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces in the Miami RAA is unlikely.
Ray, Giant Manta, *Manta birostris*	Unlikely	Undetectable	Predominantly an ocean-going species and spends most of its life far from land, traveling with the currents and migrating to areas where upwellings of nutrient-rich water increase the availability of zooplankton. Also observed in estuarine waters near oceanic inlets and coral reefs such as Flower Garden Banks which are used as nursery grounds. Hull-fouling organisms are unlikely to thrive in open ocean areas, and vessels of the Armed Forces are not cleaned near ecologically sensitive areas; therefore, exposure to hull-fouling ANS is unlikely. Impacts to the giant manta ray and its food resources are expected to be undetectable.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

MIAMI RAA **Number of deployable vessels (>79 feet) – 3** **Existing ANS[50]: Asian Tiger Shrimp (*Panaeus monodon*), Lionfish (*Pterois volitans*), Green Crab (*Carcinus maenas*), Chinese Mitten Crab (*Eriocheir sinensis*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**	**Consequences of ANS Invasion**	**Rationale Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-Fouling ANS			
Sea turtle, Hawksbill, *Eretmochelys imbricata*	Unlikely	Moderate	Uses a wide range of tropical and subtropical habitats (shallow coastal waters with rocky bottoms, coral reefs, sea grass or algae beds, mangrove-bordered bays and estuaries, and submerged mud flats). Feed predominantly around coral reefs and rock outcroppings and primarily consume sponges. Hull-fouling ANS could have moderate consequences for hawksbill sea turtles if they outcompete their preferred prey in some locations. However, because vessels of the Armed Forces are cleaned only pierside at a few loctions and away from environmentally sensitive areas such as coral reefs, exposure to hull-fouling ANS introduced by in-water hull cleaning is unlikely.
Sea turtle, Leatherback, *Dermochelys coriacea*	Unlikely	Undetectable	An oceanic, deep diving sea turtle that feeds predominantly on jellyfishes, salps and siphonophores. Nesting occurs on tropical sandy beaches, and foraging ranges extend into temperate and sub-polar latitudes. Introductions of hull-fouling ANS by in-water cleaning of vessels of the Armed Forces are unlikely to occur in these locations, and hull-fouling ANS will not affect beach nesting sites or pelagic prey (sea jellies); therefore, consequences of ANS introductions are expected to be undetectable.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

MIAMI RAA Number of deployable vessels (>79 feet) – 3 Existing ANS[50]: Asian Tiger Shrimp (*Panaeus monodon*), Lionfish (*Pterois volitans*), Green Crab (*Carcinus maenas*), Chinese Mitten Crab (*Eriocheir sinensis*)			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**	**Consequences of ANS Invasion**	**Rationale Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-Fouling ANS			
Shark, Scalloped Hammerhead Central and Southwest Atlantic DPS, *Sphyrna lewini*	Unlikely	Minor	A coastal and semi-oceanic pelagic shark, found over continental and insular shelves and in deep water near to them, ranging from the intertidal and surface to at least 275 m depth. Adults inhabit nearshore coastal waters, and juveniles inhabit protected coastal bays. Adults feed on mesopelagic fish and squids; pups and juveniles feed mainly on benthic reef fishes (e.g., scarids and gobiids), demersal fish and crustaceans. ANS introductions in the Miami RAA are unlikely. ANS invasions may affect the composition of the diet for juveniles, however impacts are anticipated to be minor.
Snake, Eastern indigo, *Drymarchon corais couperi*	Unlikely	Undetectable	During the summer prefer wetland edges where food is abundant. Feed on mammals, birds, frogs and other snakes, including rattlesnakes and cottonmouths. Hull-fouling ANS are not expected to have any effects on this species or its resources.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

<table>
<tr><th colspan="4">MIAMI RAA
Number of deployable vessels (>79 feet) – 3
Existing ANS[50]: Asian Tiger Shrimp (Panaeus monodon), Lionfish (Pterois volitans), Green Crab (Carcinus maenas), Chinese Mitten Crab (Eriocheir sinensis)</th></tr>
<tr><th>Species</th><th>Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces</th><th>Consequences of ANS Invasion</th><th>Rationale Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces</th></tr>
<tr><td colspan="4">Species with Remote Risk from Exposure to Hull-Fouling ANS</td></tr>
<tr><td>Sparrow, Cape Sable seaside, Ammodramus maritimus mirabilis</td><td>Unlikely</td><td>Undetectable</td><td>Inhabits seasonally flooded, brushless, fresh to slightly brackish subtropical interior marshes, vegetated by cordgrass, rushes, sawgrass, etc. Hull-fouling ANS introductions by in-water hull cleaning to these locations is unlikely. Is a dietary generalist that primarily feeds on soft-bodied insects such as grasshoppers, spiders, moths, caterpillars, beetles, dragonflies, wasps, marine worms, shrimp, grass and sedge seeds. Appears to shift the importance of prey items in its diet in response to their availability; therefore, although hull-fouling ANS invasions may cause a shift in dietary composition, effects on Cape Sable seaside sparrow are expected to be undetectable.</td></tr>
</table>

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

MIAMI RAA **Number of deployable vessels (>79 feet) – 3** **Existing ANS[50]: Asian Tiger Shrimp (*Panaeus monodon*), Lionfish (*Pterois volitans*), Green Crab (*Carcinus maenas*), Chinese Mitten Crab (*Eriocheir sinensis*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**	**Consequences of ANS Invasion**	**Rationale Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-Fouling ANS			
Stork, Wood, *Mycteria americana*	Very Unlikely	Undetectable	Chiefly inhabits freshwater marshes, swamps, lagoons, ponds, and flooded fields; also occurs in brackish wetlands; nests mostly in upper parts of cypress trees, mangroves, or dead hardwoods over water or on islands along streams or adjacent to shallow lakes. Introduction of hull-fouling ANS from in-water cleaning of vessels of the Armed Forces to most of these habitats is very unlikely. Feeds in freshwater marshes, swamps, lagoons, ponds, flooded pastures and flooded ditches or depressions in marshes. Loss of nesting habitat and food base (small fish) are biggest threats. ANS could have some effect on dietary composition, but there will not be any exposure of wood stork prey to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces, and impacts are expected to be undetectable.
Tern, Roseate, *Sterna dougallii*	Unlikely	Minor	Diet consists almost entirely of small fish. Introduction of hull-fouling ANS by in-water cleaning of vessels of the Armed Forces to the Miami RAA is unlikely, and any impact on forage fish by hull-fouling ANS is expected to be minor.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

MIAMI RAA Number of deployable vessels (>79 feet) – 3 Existing ANS[50]: Asian Tiger Shrimp (*Panaeus monodon*), Lionfish (*Pterois volitans*), Green Crab (*Carcinus maenas*), Chinese Mitten Crab (*Eriocheir sinensis*)			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**	**Consequences of ANS Invasion**	**Rationale Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Negligible Risk from Exposure to Hull-fouling ANS			
Manatee, West Indian, *Trichechus manatus*	Unlikely	Moderate	Herbivore that feeds opportunistically on a wide variety of marine, estuarine, and freshwater plants, including submerged, floating, and emergent vegetation. A fouling species invasion could lead to a change in local vegetation with moderate consequences (change in diet and feeding behavior) for the manatee. However, ANS introductions by in-water cleaning of vessels for the Armed Forces are unlikely.
Sawfish, Smalltooth US DPS (US Portion of Range), *Pristis pectinata*	Unlikely	Minor	Inhabits coastal waters, shallow estuaries, mangroves, and mouths of rivers. Predominantly a benthic feeder with a diet consisting of schooling fish such as mullets and clupeids, as well as crustaceans (lobsters, crabs and shrimp). An ANS invasion could affect dietary composition, but impacts are expected to be minor because of the diversity of food items. Introducion of hull-fouling ANS by in-water cleaning of vessels is unlikely.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

MIAMI RAA **Number of deployable vessels (>79 feet) – 3** **Existing ANS[50]: Asian Tiger Shrimp (*Panaeus monodon*), Lionfish (*Pterois volitans*), Green Crab (*Carcinus maenas*), Chinese Mitten Crab (*Eriocheir sinensis*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**	**Consequences of ANS Invasion**	**Rationale Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Negligible Risk from Exposure to Hull-fouling ANS			
Sea turtle, Kemp's ridley, *Lepidochelys kempii*	Unlikely	Moderate	Occur in bays and coastal waters of the Atlantic Ocean and Gulf of Mexico. Habitat of adults primarily includes shallow coastal and estuarine waters, often over sandy or muddy bottoms where crab are numerous; most activity is benthic. Post-hatchlings spend 1-4 years as surface pelagic drifters in weedlines of offshore currents in the Gulf of Mexico and Atlantic Ocean, then shift to benthic coastal habitats of various types, especially where crabs and other invertebrates are numerous. Feed on mollusks, crustaceans, jellyfish, fish, algae or seaweed, and sea urchins, some of which could be impacted by hull-fouling ANS. Consequences for Kemp's ridley sea turtles from hull-fouling ANS invasions are expected to be moderate, but exposure to ANS introduced by in-water cleaning of vessels of the Armed Forces is unlikely.
Sea Turtle, Loggerhead Northwest Atlantic Ocean DPS, *Caretta caretta*	Unlikely	Moderate	Uses coral reefs, rocky areas, and shipwrecks for feeding, which could be impacted by hull-fouling ANS. Because of the diversity of habitats used, impacts from ANS to loggerhead sea turtles are expected to be only moderate, and introduction of hull-fouling ANS by in-water hull cleaning of vessels of the Armed Forces is unlikely.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

MIAMI RAA **Number of deployable vessels (>79 feet) – 3** **Existing ANS[50]: Asian Tiger Shrimp (*Panaeus monodon*), Lionfish (*Pterois volitans*), Green Crab (*Carcinus maenas*), Chinese Mitten Crab (*Eriocheir sinensis*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**	**Consequences of ANS Invasion**	**Rationale Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Potentially Significant Risk from Exposure to Hull-fouling ANS			
Coral, Elkhorn, *Acropora palmata* Coral, Lobed Star, *Orbicella annularis* Coral, Mountainous Star, *Orbicella faveolata* Coral, Boulder Star, *Orbicella franksii* Coral, Pillar, *Dendrogyra cylindricus* Coral, Rough Cactus, *Mycetophyllia ferox* Coral, Staghorn, *Acropora cervicornis*	Unlikely	Major	Introduction of hull-fouling ANS by in-water hull cleaning of vessels of the Armed Forces could outcompete corals for space and food or overgrow corals, having major consequences. Because in-water cleaning of vessels of the Armed Forces are performed only pierside in a few locations, and because so few vessels in the Miami RAA are deployable, ANS introductions are unlikely.
Grouper, Nassau, *Epinephelus striatus*	Unlikely	Major	Common to offshore rocky bottoms and shallow coral reefs from 30 –90 m. As juveniles, inhabit nearshore shallow waters in macroalgal and seagrass habitats. Population declines have resulted from overfishing. In-water hull cleaning introductions of fouling species that could directly affect seagrass or coral habitat used by grouper are unlikely but could have major consequences (competition and grazing) if they do occur.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

MIAMI RAA **Number of deployable vessels (>79 feet) – 3** **Existing ANS[50]: Asian Tiger Shrimp (*Panaeus monodon*), Lionfish (*Pterois volitans*), Green Crab (*Carcinus maenas*), Chinese Mitten Crab (*Eriocheir sinensis*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**	**Consequences of ANS Invasion**	**Rationale Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Potentially Significant Risk from Exposure to Hull-fouling ANS			
Green sea turtle, North and South Atlantic DPS[51], *Chelonia mydas*	Unlikely	Major	Uses seagrass beds and beaches within the Miami RAA. Hull-fouling ANS could have direct impacts to seagrass beds, and therefore major consequences for green sea turtles. However, in-water cleaning of vessels of the Armed Forces is performed only pierside in a few locations, and there are only three deployable vessels of the Armed Forces in the Miami RAA; therefore, exposure to hull-fouling ANS introduced by underwater ship husbandry discharge is unlikely.
Seagrass, Johnson's, *Halophila johnsonii*	Unlikely	Major	Introductions of hull-fouling algae or invertebrates could lead to competition for space or increased grazing pressure. However, in-water cleaning of vessels of the Armed Forces is performed only pierside in a few locations, and there are only three deployable vessels of the Armed Forces in the Miami RAA; therefore, exposure to hull-fouling ANS introduced by underwater ship husbandry discharge is unlikely.
Species with Likely Significant Risk from Exposure to Hull-fouling ANS **NONE**			

[51] Includes the currently listed Endangered Sea turtle, Green (Florida and Mexico Pacific Coast Breeding Colony)

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

MIAMI RAA Number of deployable vessels (>79 feet) – 3 Existing ANS[50]: Asian Tiger Shrimp (*Panaeus monodon*), Lionfish (*Pterois volitans*), Green Crab (*Carcinus maenas*), Chinese Mitten Crab (*Eriocheir sinensis*)			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**	**Consequences of ANS Invasion**	**Rationale Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with No Risk from Exposure to Hull-fouling ANS			
Thoroughwort, Cape Sable, *Chromolaena frustrata*	No Exposure	Undetectable	Occurs on coastal rock barrens and berms that are very unlikely to become inundated, as well as a variety of terrestrial habitats, that will not be impacted by hull-fouling ANS.
Whale, Blue, *Balaenoptera musculus*	No Exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Fin, *Balaenoptera physalus*	No Exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, North Atlantic right whale, *Eubalaena glacialis*	No Exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Sei, *Balaenoptera borealis*	No Exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Sperm, *Physeter macrocephalus*	No Exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

NORFOLK RAA **Number of deployable vessels (>79 feet) – 165** **Existing ANS[50]: Veined Rapa Whelk (*Rapana venosa*), Colonial Tunicate (*Didemnum vexillum*), Asian Shore Crab (*Hemigrapsus sanguineus*), Lionfish (*Pterois volitans*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Knot, Red, *Calidris canutus rufa*	Unlikely	Minor	For much of the year red knots eat small clams, mussels, snails and other invertebrates, swallowing their prey whole – shell and all. Migrating birds require stopover habitats rich in easily digested foods (e.g., juvenile clams and mussels and horseshoe crab eggs). The Norfolk RAA is not an important stopover for red knots. Impacts to food resources from hull-fouling ANS could occur but are expected to be minor. In addition, introduction of ANS by in-water cleaning of vessels of the Armed Forces is unlikely.
Petrel, Bermuda, *Pterodroma cahow*	Very Unlikely	Undetectable	Nests in natural erosion limestone crevices and artificial burrows on islands off Bermuda. Follows and forages over the Gulf Stream during the non-breeding season. Feeds predominantly on small fish, squid and shrimp-like crustaceans in the offshore where they will not be exposed to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces. Exposure to ANS in nearshore waters is limited.
Plover, Piping, *Charadrius melodus*	Unlikely	Undetectable	Most abundant on expansive sandflats, sandy mudflats, and sandy beaches in close proximity. Feeds on worms, fly larvae, beetles, crustaceans, mollusks, and other invertebrates. Dietary composition could be affected by invasive species but is unlikely to affect food availability; therefore, impacts are expected to be undetectable.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

NORFOLK RAA **Number of deployable vessels (>79 feet) – 165** **Existing ANS[50]: Veined Rapa Whelk (*Rapana venosa*), Colonial Tunicate (*Didemnum vexillum*), Asian Shore Crab (*Hemigrapsus sanguineus*), Lionfish (*Pterois volitans*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Rail, Eastern black, *Laterallus jamaicensis ssp. jamaicensis*	Unlikely	Minor	Inhabits fresh and saline marshes, wet meadows and savannas. Occupies marshes with shallower water than other rallids and requires some tall vegetation to escape into. Feeds on terrestrial and aquatic invertebrates. Uses impoundments (managed wetlands) to forage and nest. Composition of diet could be affected by hull-fouling ANS but is unlikely to affect food availability and feeding behavior. In addition, exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces in the Norfolk RAA is unlikely.
Ray, Giant Manta, *Manta birostris*	Very Unlikely	Undetectable	Predominantly an ocean-going species and spends most of its life far from land, traveling with the currents and migrating to areas where upwellings of nutrient-rich water increase the availability of zooplankton. Also observed in estuarine waters near oceanic inlets and coral reefs such as Flower Garden Banks which are used as nursery grounds. Hull-fouling organisms are unlikely to thrive in open ocean areas, and vessels of the Armed Forces do not operate in ecologically sensitive areas; therefore, exposure is very unlikely. Impacts from hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces are expected to be undetectable for giant manta rays and their food resources.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

NORFOLK RAA **Number of deployable vessels (>79 feet) – 165** **Existing ANS[50]: Veined Rapa Whelk (*Rapana venosa*), Colonial Tunicate (*Didemnum vexillum*), Asian Shore Crab (*Hemigrapsus sanguineus*), Lionfish (*Pterois volitans*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Sea turtle, Leatherback, *Dermochelys coriacea*	Very Unlikely	Undetectable	Less than ten leatherback sea turtles are observed in the Chesapeake Bay each year. Nest on sandy beaches and have a diet of soft-bodied pelagic (open ocean) prey, such as jellyfish and salps. Exposure to hull-fouling ANS from in-water cleaning of vessels of the Armed Forces is very unlikely and impacts to leatherback sea turtles and their pelagic prey from ANS invasions in the Norfolk RAA are expected to be undetectable.
Sea turtle, Hawksbill, *Eretmochelys imbricata*	Unlikely	Undetectable	Predominantly a tropical and sub-tropical species, and extremely rare in the Chesapeake Bay. Diet consist primarily of invertebrates (sponges on coral reefs) but also includes plant material and fishes. Introductions of hull-fouling ANS in the Norfolk RAA from in-water cleaning of vessels of the Armed Forces are not expected to have any detectable impacts on hawksbill sea turtles or their invertebrate and algal food sources if they do occur.
Shark, Scalloped Hammerhead Central and Southwest Atlantic DPS, *Sphyrna lewini*	Unlikely	Undetectable	Present in Virginia coastal waters from June through August. Diet consist of benthic fish and invertebrates including menhaden, mullets, flounders, drums, crustaceans, stingrays, and small sharks. Hull-fouling ANS introductions are unlikely to affect dietary composition, and any impacts are anticipated to be undetectable. In addition, introduction of hull-fouling ANS by in-water cleaning of vessels of the Armed Forces is unlikely.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

NORFOLK RAA **Number of deployable vessels (>79 feet) – 165** **Existing ANS[50]: Veined Rapa Whelk (*Rapana venosa*), Colonial Tunicate (*Didemnum vexillum*), Asian Shore Crab (*Hemigrapsus sanguineus*), Lionfish (*Pterois volitans*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Sturgeon, Atlantic Carolina DPS Chesapeake Bay DPS New York Bight DPS South Atlantic DPS, *Acipenser oxyrinchus*	Unlikely	Minor	Juveniles feed on small benthos over sand river bottom. Adults are generalists and feed on crustaceans, worms, and mollusks. Hull-fouling ANS could cause a shift in dietary composition, but such shifts are expected to only have minor impacts for Atlantic sturgeon because the impacts would only be expected to occur for estuarine benthos.
Sturgeon, Shortnose, *Acipenser brevirostrum*	Unlikely	Minor	Inhabit rivers and estuaries and generally feed on benthic organisms, including crustaceans, worms, and mollusks. Hull-fouling ANS could cause a shift in dietary composition; however, because shortnosed sturgeon are generalists, the impacts are expected to be minor.
Tern, Roseate, *Sterna dougallii*	Unlikely	Minor	Diet consists almost entirely of small fish. Any impact on forage fish by hull-fouling ANS is expected to be minor.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

NORFOLK RAA **Number of deployable vessels (>79 feet) – 165** **Existing ANS[50]: Veined Rapa Whelk (*Rapana venosa*), Colonial Tunicate (*Didemnum vexillum*), Asian Shore Crab (*Hemigrapsus sanguineus*), Lionfish (*Pterois volitans*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Negligible Risk from Exposure to Hull-fouling ANS			
Sea turtle, Green North Atlantic DPS[52], *Chelonia mydas*	Unlikely	Moderate	Uses seagrass beds for feeding and beaches for nesting. There is limited seagrass habitat in the Norfolk RAA, and it is unlikely that green sea turtles and seagrass beds will be exposed to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces. Hull-fouling ANS invasions could have a a noticeable impact on seagrass beds and moderate consequences for green sea turtles in this area.
Sea turtle, Kemp's ridley, *Lepidochelys kempii*	Unlikely	Moderate	Kemp's Ridley sea turtles are the second most common in the Chesapeake Bay, where some individuals feed in the summertime. Feed on mollusks, crustaceans, jellyfish, fish, algae or seaweed, and sea urchins, any of which could be impacted by hull-fouling ANS. Because of the diversity of their diet, ANS impacts are only likely to cause a shift in dietary composition, which could have moderate consequences for Kemp's ridley sea turtles at most, and exposure to ANS introduced by in-water hull cleaning of vessels of the Armed Forces is unlikely.

[52] Includes the currently listed Endangered Sea turtle, Green (Florida and Mexico Pacific Coast Breeding Colony)

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

NORFOLK RAA **Number of deployable vessels (>79 feet) – 165** **Existing ANS[50]: Veined Rapa Whelk (*Rapana venosa*), Colonial Tunicate (*Didemnum vexillum*), Asian Shore Crab (*Hemigrapsus sanguineus*), Lionfish (*Pterois volitans*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Negligible Risk from Exposure to Hull-fouling ANS			
Sea Turtle, Loggerhead Northwest Atlantic Ocean DPS, *Caretta caretta*	Unlikely	Moderate	Mainly juveniles are found in the Chesapeake Bay, foraging on blue crab, horseshoe crab, whelk, fishes, and sea grasses. Virginia coastal beaches are also the northernmost limit for nesting adults. Hull-fouling ANS could cause a shift in dietary composition; however, because of the diversity of prey, impacts are expected to be moderate at most, and exposure to ANS introduced by in-water hull cleaning of vessels of the Armed Forces is unlikely.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

NORFOLK RAA **Number of deployable vessels (>79 feet) – 165** **Existing ANS[50]: Veined Rapa Whelk (*Rapana venosa*), Colonial Tunicate (*Didemnum vexillum*), Asian Shore Crab (*Hemigrapsus sanguineus*), Lionfish (*Pterois volitans*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Potentially Significant Risk from Exposure to Hull-fouling ANS **NONE**			
Species with Likely Significant Risk from Exposure to Hull-fouling ANS **NONE**			
Species with No Risk from Exposure to Hull-fouling ANS			
Bat, Northern long-eared, *Myotis septentrionalis*	No Exposure	Undetectable	Feeds on moths, flies, leafhoppers, caddisflies, and beetles, which they catch while in flight using echolocation, as well as bygleaning motionless insects from vegetation and water surfaces. ANS that are introduced to the Norfolk RAA are likely to bemarine/estuarine species that will not survive in freshwater habitats and, therefore, do not pose any risk to Northern long-eared bats and theirprey.
Whale, Blue, *Balaenoptera musculus*	No Exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Fin, *Balaenoptera physalus*	No Exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, North Atlantic right whale, *Eubalaena glacialis*	No Exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Sei, *Balaenoptera borealis*	No Exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Sperm, *Physeter macrocephalus*	No Exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PEARL HARBOR RAA **Number of deployable vessels (>79 feet) – 60** **Existing ANS[50]: More than half of native species have been replaced with non-native species Gorilla Ogo (*Gracilaria salicornia*); Orange Keyhole Sponge (*Mycale grandis*), Dwarf Hawkfish (*Cirrhitichthys falco*), Coral Hawkfish (*Cirrhitichthys oxycephalus*), Gulf Killifish (*Fundulus grandis*), Mangrove Goby (*Mugilogobius cavifrons*), Blacktail Snapper (*Lutjanus fulvus*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Albatross, Short-tailed, *Phoebastria albatrus*	Unlikely	Undetectable	Feeds offshore, mainly on squid, but also takes shrimp, fish, flying fish eggs and other crustaceans. Because feeding is offshore, exposure to hull-fouling ANS introduced by in water hull cleaning of vessels of the Armed Forces is unlikely. Impacts from hull-fouling ANS on the short-tailed albatross and its prey are expected to be undetectable.
Coot, Hawaiian (alae ke 'oke'o), *Fulica americana alai*	Unlikely	Undetectable	Feeds on seeds and leaves of aquatic plants, snails, crustaceans, small fishes, tadpoles, and insects in various herbacious freshwater and brackish wetlands, including lakes, ponds, reservoirs, irrigation ditches, and marshes. Because feeding is predominantly in freshwater and low-salinity wetlands where marine and estuarine hull-fouling organisms are unlikely to thrive, exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces is unlikely. Impacts to the Hawaiian coot and its prey from hull-fouling ANS invasions are expected to be undetectable.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PEARL HARBOR RAA Number of deployable vessels (>79 feet) – 60 Existing ANS[50]: More than half of native species have been replaced with non-native species Gorilla Ogo (*Gracilaria salicornia*); Orange Keyhole Sponge (*Mycale grandis*), Dwarf Hawkfish (*Cirrhitichthys falco*), Coral Hawkfish (*Cirrhitichthys oxycephalus*), Gulf Killifish (*Fundulus grandis*), Mangrove Goby (*Mugilogobius cavifrons*), Blacktail Snapper (*Lutjanus fulvus*)			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Duck, Hawaiian (koloa), *Anas wyvilliana*	Very Unlikely	Undetectable	Occurs in a wide range of terrestrial and freshwater habitats, from sea level to 3000 m elevation including lowland marshes, reservoirs, taro patches, pastures, drainage ditches, agricultural lands, stream and river valleys in densely wooded areas, mountain pools, mountain bogs, and forest swamps. Is an opportunistic feeder, consuming invertebrates, seeds and plant matter. Because the Hawaiian duck predominantly inhabits freshwater and upland habitats where marine and estuarine hull-fouling organisms are unlikely to thrive, exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces is very unlikely. Impacts from hull-fouling ANS invasion are expected to be undetectable.
Moorhen, Common (alae 'u), *Gallinula chloropus sandvicensis*	Very Unlikely	Undetectable	Nests generally in shallow water in areas of dense emergent vegetation, usually avoiding salt and brackish water. Diet consists of algae, seeds, and other plant material; aquatic insects; and mollusks. Because common moorhens inhabit freshwater wetlands where marine and estuarine hull-fouling organisms are unlikely to thrive, exposure to to hull-fouling ANS introduces by in-water cleaning of vessels of the Armed Forces is very unlikely. Impacts to nesting habitat and food resources from hull-fouling ANS invasions are expected to be undetectable.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PEARL HARBOR RAA **Number of deployable vessels (>79 feet) – 60** **Existing ANS[50]: More than half of native species have been replaced with non-native species Gorilla Ogo (*Gracilaria salicornia*); Orange Keyhole Sponge (*Mycale grandis*), Dwarf Hawkfish (*Cirrhitichthys falco*), Coral Hawkfish (*Cirrhitichthys oxycephalus*), Gulf Killifish (*Fundulus grandis*), Mangrove Goby (*Mugilogobius cavifrons*), Blacktail Snapper (*Lutjanus fulvus*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Petrel, Hawaiian dark-rumped, *Pterodroma phaeopygia sandwichensis*	Likely	Undetectable	Nests in burrows in barren areas high on mountain slopes (2500-3000 m on Maui), commonly in erosional debris at the base of rock outcrops, typically on steep slopes under large rocks in the vicinity of shrub cover. Adults feed mostly on squid, fish and crustaceans and regurgitate food for young. Although exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces during feeding is likely, impacts to Hawaiian dark-rumped petrel and their pelagic prey are expected to be undetectable.
Ray, Giant Manta, *Manta birostris*	Likely	Undetectable	Predominantly an ocean-going species and spends most of its life far from land, traveling with the currents and migrating to areas where upwellings of nutrient-rich water increase the availability of zooplankton. Also observed in estuarine waters near oceanic inlets and coral reefs which are used as nursery grounds. Hull-fouling ANS in underwater ship husbandry discharge will not be introduced to open ocean areas. However, exposure of individuals that enter Pearl Harbor to hull-fouling ANS from in-water cleaning is likely. Impacts to gian manta rays and their planktonic food resources are expected to be undetectable.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PEARL HARBOR RAA Number of deployable vessels (>79 feet) – 60 Existing ANS[50]: More than half of native species have been replaced with non-native species Gorilla Ogo (*Gracilaria salicornia*); Orange Keyhole Sponge (*Mycale grandis*), Dwarf Hawkfish (*Cirrhitichthys falco*), Coral Hawkfish (*Cirrhitichthys oxycephalus*), Gulf Killifish (*Fundulus grandis*), Mangrove Goby (*Mugilogobius cavifrons*), Blacktail Snapper (*Lutjanus fulvus*)			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Sea turtle, Hawksbill, *Eretmochelys imbricata*	Likely	Minor	Uses a wide range of tropical and subtropical habitats (shallow coastal waters with rocky bottoms, coral reefs, sea grass or algae beds, mangrove-bordered bays and estuaries, and submerged mud flats). Feed around coral reefs and rock outcroppings and primarily consume sponges. The majority of hawksbill nesting in the Hawaiian Islands takes place on the Big Island of Hawaii. Although exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces in Pearl Harbor is likely, consequences of introductions of ANS invasions from underwater ship husbandry discharge are expected to be minor because this is not where the majority of feeding occurs.
Sea turtle, Leatherback, *Dermochelys coriacea*	Very Unlikely	Undetectable	Nests on sandy beaches and has a diet of soft-bodied pelagic (open ocean) prey, such as jellyfish and salps. Because leatherback turtles feed in the offshore, exposure to hull-fouling ANS from in-water cleaning of vessels of the Armed Forces is very unlikely, and consequences from ANS invasions are expected to be undetectable.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PEARL HARBOR RAA **Number of deployable vessels (>79 feet) – 60** **Existing ANS[50]: More than half of native species have been replaced with non-native species Gorilla Ogo (*Gracilaria salicornia*); Orange Keyhole Sponge (*Mycale grandis*), Dwarf Hawkfish (*Cirrhitichthys falco*), Coral Hawkfish (*Cirrhitichthys oxycephalus*), Gulf Killifish (*Fundulus grandis*), Mangrove Goby (*Mugilogobius cavifrons*), Blacktail Snapper (*Lutjanus fulvus*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Seal, Hawaiian Monk, *Monachus schauinslandi*	Very Unlikely	Undetectable	Breeding populations occur on atolls outside of the RAA. Forage within atolls in the shallow waters surrounding atolls and islands, and farther offshore at submerged banks and reefs; exposure to and dependence on resources in Pearl Harbor where in-water hull cleanings are performed is limited. Therefore, consequences to Hawaiian monk seals from ANS introductions by underwater ship husbandry discharge from vessels of the Armed Forces are expected to be undetectable.
Duck, Laysan, *Anas laysanensis*	Likely	Minor	Usually occurs in lagoons, tidal pools, and marshes. Feeds and drinks at inland waterbodies at night and is attracted to freshwater seeps. Seeks shelter in vegetation (*Pluchera, Ipomoea, and Sicyos*) during heat of day. Eats mainly insects, including caterpillars, larvae and pupae of flies and beetles around seabird carcasses, and especially Neoscatella flies around saline lakes; also eats crustaceans and other invertebrates in shallow tide pools. Although exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces is likely and hull-fouling species could impact some food resources, diet is diverse enough that impacts to diet are expected to be minor. Therefore, risk is considered to be negligible.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PEARL HARBOR RAA **Number of deployable vessels (>79 feet) – 60** **Existing ANS[50]: More than half of native species have been replaced with non-native species Gorilla Ogo (*Gracilaria salicornia*); Orange Keyhole Sponge (*Mycale grandis*), Dwarf Hawkfish (*Cirrhitichthys falco*), Coral Hawkfish (*Cirrhitichthys oxycephalus*), Gulf Killifish (*Fundulus grandis*), Mangrove Goby (*Mugilogobius cavifrons*), Blacktail Snapper (*Lutjanus fulvus*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Sea Turtle, Loggerhead North Pacific Ocean DPS, *Caretta caretta*	Very Unlikely	Major	Uses coral reefs, rocky areas, and shipwrecks for feeding. Hull-fouling ANS could have major consequences for loggerhead sea turtle feeding areas if ANS outcompete or overgrow food resources. However, loggerhead sea turtles are most often found in pelagic waters in Hawaii where they will not be exposed to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces. Nesting areas are extremely critical to the survival of the loggerhead sea turtle and are likely in Japan for this population.
Sea turtle, Olive ridley (all other areas), *Lepidochelys olivacea*	Likely	Minor	Most often found in shallow water around reefs, bays and inlets. Common prey items include jellyfish, tunicates, sea urchins, bryozoans, bivalves, snails, shrimp, crabs, rock lobsters, and sipunculid worms. Prey items could be impacted by hull-fouling ANS invasions; however, because of the diversity of prey items, consequences for olive ridley sea turtles are expected to be minor. Exposure to hull-fouling ANS from in-water cleaning of vessels of the Armed Forces in Pearl Harbor is likely.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

<table>
<tr><th colspan="4">PEARL HARBOR RAA
Number of deployable vessels (>79 feet) – 60
Existing ANS[50]: More than half of native species have been replaced with non-native species Gorilla Ogo (Gracilaria salicornia); Orange Keyhole Sponge (Mycale grandis), Dwarf Hawkfish (Cirrhitichthys falco), Coral Hawkfish (Cirrhitichthys oxycephalus), Gulf Killifish (Fundulus grandis), Mangrove Goby (Mugilogobius cavifrons), Blacktail Snapper (Lutjanus fulvus)</th></tr>
<tr><th>Species</th><th>Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]</th><th>Consequences of ANS Invasion</th><th>Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces</th></tr>
<tr><th colspan="4">Species with Remote Risk from Exposure to Hull-fouling ANS</th></tr>
<tr><td>Shark, Scalloped Hammerhead Eastern Pacific DPS, Sphyrna lewini</td><td>Likely</td><td>Minor</td><td>Adults inhabit nearshore coastal waters and juveniles inhabit protected coastal bays. Therefore, exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces is likely. Adults feed on mesopelagic fish, squid and stingrays while pups and juveniles feed mainly on benthic reef fishes (e.g., scarids and gobiids), demersal fish, and crustaceans. ANS invasions may affect the composition of the diet for juveniles; however, consequences are anticipated to be minor.</td></tr>
<tr><td>Stilt, Hawaiian (ae'o), Himantopus mexicanus knudseni</td><td>Unlikely</td><td>Moderate</td><td>Frequents mudflats along or near natural or human-made ponds and wetlands, often near coastal areas; feeds in freshwater or tidal wetlands on various aquatic organisms including worms, small crabs, insects, small fishes. Because marine and estuarine ANS introduced by in-water cleaning of vessels of the Armed Forces are not expected to thrive in freshwater and low-salinity habitats, exposure is unlikely. ANS invasion, however, could cause a shift in the Hawaiian stilt diet, having moderate consequences.</td></tr>
</table>

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PEARL HARBOR RAA Number of deployable vessels (>79 feet) – 60 Existing ANS[50]: More than half of native species have been replaced with non-native species Gorilla Ogo (*Gracilaria salicornia*); Orange Keyhole Sponge (*Mycale grandis*), Dwarf Hawkfish (*Cirrhitichthys falco*), Coral Hawkfish (*Cirrhitichthys oxycephalus*), Gulf Killifish (*Fundulus grandis*), Mangrove Goby (*Mugilogobius cavifrons*), Blacktail Snapper (*Lutjanus fulvus*)			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Potentially Significant Risk from Exposure to Hull-fouling ANS			
Coral, Cauliflower, *Pocillopora meandrina*	Unlikely	Major	Common on exposed reef fronts on shallow reefs and amongst coral communities on rocky reefs, at depths from 3-27 m. Usually found where there is strong wave action outside of embayments where hull-fouling ANS are unlikely to invade. Some hull-fouling ANS could overgrow or aggressively graze corals if they are introduced, having major impacts. Therefore, risk is considered to be potentially significant.
Sea turtle, Green Central North Pacific DPS[53], *Chelonia mydas*	Likely	Moderate	Uses seagrass beds for foraging and beaches for nesting. Hull-fouling ANS invasions could have noticeable consequences for seagrass beds and, consequently, green sea turtles. Exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces in Pearl Harbor is likely. However, there is limited seagrass habitat near where vessels of the Armed Forces are cleaned, and consequences of impacts to seagrass habitat for green sea turtles are expected to be moderate, at most.
Species with Likely Significant Risk from Exposure to Hull-fouling ANS **NONE**			

[53] Includes the currently listed threatened Sea turtle, Green (All Others)

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PEARL HARBOR RAA **Number of deployable vessels (>79 feet) – 60** **Existing ANS[50]: More than half of native species have been replaced with non-native species Gorilla Ogo (*Gracilaria salicornia*); Orange Keyhole Sponge (*Mycale grandis*), Dwarf Hawkfish (*Cirrhitichthys falco*), Coral Hawkfish (*Cirrhitichthys oxycephalus*), Gulf Killifish (*Fundulus grandis*), Mangrove Goby (*Mugilogobius cavifrons*), Blacktail Snapper (*Lutjanus fulvus*)**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with No Risk from Exposure to Hull-fouling ANS			
Damselfly, Crimson Hawaiian, *Megalagrion leptodemus*	No Exposure	Undetectable	Occurs only in freshwater habitats that will not be impacted by marine and estuarine hull-fouling ANS.
Damselfly, Orangeblack Hawaiian, *Megalagrion Xanthomelas*	No Exposure	Undetectable	Occurs only in freshwater habitats that will not be impacted by marine and estuarine hull-fouling ANS.
Oʻahu ʻelepaio, *Chasiempis ibidis*	No exposure	Undetectable	Found in native and exotic forests. Eats insects obtained by foliage-gleaning, bark-picking, and aerial sallies below forest canopy. Forages in areas with high foliage density, large bark surface area, and many twigs and branches. There will not be any exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces in areas inhabited by Oʻahu ʻelepaio.
Shearwater, Newell's, *Puffinus auricularis newelli*	No Exposure	Undetectable	Feeds far from shore where hull-fouling ANS from in-water cleaning of vessels of the Armed Forces will not be introduced.
Whale, Blue, *Balaenoptera musculus*	No exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Fin, *Balaenoptera physalus*	No exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Main Hawaiian Islands Insular False Killer, *Pseudorca crassidens*	No exposure	Undetectable	Species occurs predominantly offshore and will not be impacted by ANS invasions.
Whale, Sei, *Balaenoptera borealis*	No exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PEARL HARBOR RAA Number of deployable vessels (>79 feet) – 60 Existing ANS[50]: More than half of native species have been replaced with non-native species Gorilla Ogo (*Gracilaria salicornia*); Orange Keyhole Sponge (*Mycale grandis*), Dwarf Hawkfish (*Cirrhitichthys falco*), Coral Hawkfish (*Cirrhitichthys oxycephalus*), Gulf Killifish (*Fundulus grandis*), Mangrove Goby (*Mugilogobius cavifrons*), Blacktail Snapper (*Lutjanus fulvus*)			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with No Risk from Exposure to Hull-fouling ANS			
Whale, Sperm, *Physeter macrocephalus*	No exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PUGET SOUND RAA Number of deployable vessels (>79 feet) – 100 Existing ANS[50]: At least 74 NIS (Davidson et al., 2014)[54]			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-Fouling ANS			
Albatross, Short-tailed, *Phoebastria (=Diomedea) albatrus*	Unlikely	Undetectable	Feeds offshore. Mainly consumes squid, but also takes shrimp, fish, flying fish eggs, and other crustaceans. Hull-fouling ANS in underwater ship husbandry discharge will not be introduced to feeding areas, and exposure in nearshore areas is unlikely. Consequences of ANS introductions by vessels of the Armed Forces are expected to be undetectable.

[54] Species include European Green Crab (*Carcinus maenas*), Chinese Mitten Crab (*Eriocheir sinensis*), Ectoparasitic Isopod (*Orthione griffenis*), Bamboo Worm (*Clymenella torquata*), Club Tunicate (*Styela clava*), Transparent Tunicate (*Ciona savignyi*), Colonial Tunicate (*Didemnum vexillum*), Encrusting Bryozoan (*Schioporella japonica*), Hydroid (*Cordylophora caspia*), Venus Clam (*Venerupis philippinarum*), Mahogany Clam (*Nuttalia obscurata*), Pacific Oyster (*Crassostrea gigas*), Mediterranean Mussel (*Mytilus galloprovincialis*), Common Slippershell (*Crepidula fornicata*), Japanese Mudsnail (*Batillaria attramentaria*), American Shad (*Alosa sapidissima*) – Native Transplant, Striped Bass (*Morone saxatilis*), Round Goby (*Neogobius melanostomus*), Japanese Wireweed (*Sargassum muticum*), Atlantic Cordgrass (*Spartina alterniflora*), Common Cordgrass (*Spartina anglica*), Dense-flowered Cordgrass (*Spartina densiflora*), Dwarf Eelgrass (*Zostera japonica*)

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PUGET SOUND RAA Number of deployable vessels (>79 feet) – 100 Existing ANS[50]: At least 74 NIS (Davidson et al., 2014)[54]			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Bear, Grizzly, *Ursus arctos horribilis*	Unlikely	Undetectable	Diet consists of a variety of prey, including aquatic species. Because the diet is varied, impacts to aquatic prey items from hull-fouling ANS are expected to have undetectable consequences for grizzly bears. Exposure to hull-fouling ANS from in-water cleaning of vessels of the Armed Forces is unlikely.
Bocaccio, *Sebastes paucispinis*	Unlikely	Undetectable	Piscivorous fish that occurs in nearshore areas over rocky habitats at 12-480 m. Exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces in unlikely, and hull-fouling ANS will not have detectable impacts on bocaccio or their prey.
Eulachon, *Thaleichthys pacificus*	Unlikely	Minor	A small, anadromous smelt that spends most of its adult life in the ocean but returns to its natal freshwater stream or river to spawn and die. Upon hatching, larvae are found near the bottom and are soon carried downstream to salt water and are eventually found in the scattering layer of coastal waters. Adults do not feed while in fresh water. Young fish eat mostly copepod larvae, phytoplankton, copepods, and other zooplankton. Hull-fouling organisms could have an impact on plankton in coastal embayments, but larval fish are carried downstream and out to sea soon after hatching, limiting their time in embayments where hull-fouling ANS from in-water cleaning of vessels of the Armed Forces are likely to occur. Therefore, consequences of ANS introductions by underwater ship husbandry discharge are expected to be minor. In addition, exposure to hull-fouling ANS introduced to the Puget Sound RAA by in-water cleaning of vessels of the Armed Forces is unlikely.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PUGET SOUND RAA Number of deployable vessels (>79 feet) – 100 Existing ANS[50]: At least 74 NIS (Davidson et al., 2014)[54]			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Salmon, Chum Columbia River ESU Hood Canal Summer-Run ESU, *Oncorhynchus keta*	Unlikely	Minor	Juvenile chum salmon feed on zooplankton and insects in freshwater habitats where hull-fouling ANS from in-water cleaning of vessels of the Armed Forces are unlikely to be introduced and become established. Recent studies show that chum salmon also eat comb jellies in estuaries. As adults, they eat smaller fish in the offshore. Hull-fouling organisms are not expected to significantly impact food resources in estuaries and will not be introduced in offshore by in-water hull cleaning. Consequences of ANS invasions for chum salmon in estuaries are expected to be minor.
Sea turtle, Leatherback, *Dermochelys coriacea*	Very Unlikely	Undetectable	Nest on sandy beaches and have a diet of soft-bodied pelagic (open ocean) prey, such as jellyfish and salps. Feed on sea jellies offshore Washington where hull-fouling ANS will not be introduced by in-water cleaning of vessels of the Armed Forces. Consequences of exposure of leatherback sea turtles and their prey are expected to be undetectable.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PUGET SOUND RAA **Number of deployable vessels (>79 feet) – 100** **Existing ANS[50]: At least 74 NIS (Davidson et al., 2014)[54]**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Trout, Bull, *Salvelinus confluentus*	Unlikely	Minor	Species has both migratory and non-migratory populations. Migratory populations may migrate between streams/creeks and larger freshwater bodies or open ocean. An anadromous form of bull trout also exists in the Coastal-Puget Sound population, which spawns in rivers and streams but rears young in the ocean. Resident and juvenile bull trout prey on invertebrates and small fish. Adult migratory bull trout primarily eat fish. The only place where hull-fouling ANS may be introduced by in-water hull cleaning of vessels of the Armed Forces are in estuaries, and exposure of bull trout and their resources is unlikely. Hull-fouling ANS could impact benthos in estuaries, but because the rare marine or amphidromous/anadromous bull trout spends little time in estuaries, consequences from exposure to hull-fouling ANS are expected to be minor.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PUGET SOUND RAA Number of deployable vessels (>79 feet) – 100 Existing ANS[50]: At least 74 NIS (Davidson et al., 2014)[54]			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Trout, Steelhead, California Central Valley DPS Central California Coast DPS Lower Columbia River DPS Middle Columbia River DPS Northern California DPS Puget Sound DPS Snake River DPS South-Central California Coast DPS Southern California DPS Upper Columbia River DPS Upper Willamette River DPS, *Oncorhynchus mykiss*	Very Unlikely	Undetectable	Born in fresh water streams where they spend their first 1-3 years of life. Then migrate to the ocean where most of their growth occurs. After spending between one to four growing seasons in the ocean, return to their native fresh water stream to spawn. Young animals feed primarily on zooplankton. Adults feed on aquatic and terrestrial insects, mollusks, crustaceans, fish eggs, minnows, and other small fishes (including other trout). Because most of their life is spent either in freshwater or the offshore where hull-fouling ANS will not be introduced by in-water hull cleaning, exposure of steelhead and their food resources to hull-fouling ANS is very unlikely. Consequences of exposure of steelhead to hull-fouling ANS are expected to be undetectable.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PUGET SOUND RAA **Number of deployable vessels (>79 feet) – 100** **Existing ANS[50]: At least 74 NIS (Davidson et al., 2014)[54]**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Negligible Risk from Exposure to Hull-Fouling ANS			
Murrelet, Marbled, *Brachyramphus marmoratus*	Unlikely	Moderate	Diet consists of sandlance, herring, other small schooling fish and, in winter, invertebrates. Feeds in near-shore habitats up to 1.4 km offshore, in sheltered waters, lagoons, and sometimes inland lakes. Hull-fouling ANS could impact some food resources, having moderate consequences for marbled murrelet, but exposure to ANS introduced by in-water hull cleaning of vessels of the Armed Forces is unlikely.
Rockfish, Yelloweye, *Sebastes ruberrimus*	Unlikely	Moderate	Associated with rocky reefs, kelp canopies, and artificial structures. Larval rockfish feed on diatoms, dinoflagellates, tintinnids, and cladocerans, and juveniles consume copepods and euphausiids of all life stages. Adults eat demersal invertebrates and small fishes, including other species of rockfish, associated with kelp beds, rocky reefs, pinnacles, and sharp dropoffs. In inland waters, hull-fouling ANS may impact yelloweye rockfish food resources, having moderate consequences, but exposure to ANS introduced by in-water hull cleaning of vessels of the Armed Forces is unlikely.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PUGET SOUND RAA **Number of deployable vessels (>79 feet) – 100** **Existing ANS[50]: At least 74 NIS (Davidson et al., 2014)[54]**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Negligible Risk from Exposure to Hull-Fouling ANS			
Salmon, Chinook California Coastal ESU Central Valley ESU Lower Columbia River ESU Columbia Spring-Run ESU Puget Sound ESU Sacramento River Winter-Run ESU Snake River Fall-Run ESU Snake River Spring/Summer-Run ESU Upper Willamette River ESU, *Oncorhynchus tshawytscha*	Unlikely	Moderate	Juveniles feed opportunistically on aquatic insects and aquatic life stages of insects in freshwater reaches of rivers and streams where hull-fouling organisms will not be introduced by in-water cleaning of vessels of the Armed Forces. As they grow, chinook salmon migrate offshore, where hull-fouling ANS also will not be introduced by in-water cleaning, and they feed on crustaceans as well as other bottom invertebrates. Limited time is spent in estuaries and harbors where hull-fouling ANS are most likely to be introduced by in-water hull cleaning. They rely on eelgrass and seaweeds in estuaries for camouflage (protection from predators), shelter, and foraging habitat as they make their way to the open ocean. Hull-fouling ANS that are aggressive grazers could impact eelgrass and seaweeds. Any impacts from ANS to migrating salmon are expected to occur in estuaries and to be moderate, but exposure to ANS introduced by in-water hull cleaning of vessels of the Armed Forces in the Puget Sound RAA is unlikely.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PUGET SOUND RAA **Number of deployable vessels (>79 feet) – 100** **Existing ANS[50]: At least 74 NIS (Davidson et al., 2014)[54]**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Negligible Risk from Exposure to Hull-Fouling ANS			
Sea Turtle, Loggerhead North Pacific Ocean DPS, *Caretta caretta*	Unlikely	Undetectable	Pacific loggerheads migrate over 7,500 miles (12,000 km) between nesting beaches in Japan and feeding grounds off Mexico. Diet of all life stages is mostly benthic invertebrates (crabs, other crustaceans and mollusks) and occasionally jellyfish. Uses coral reefs, rocky areas, and shipwrecks for feeding. There are only occasional sightings from the coast of Washington, and loggerhead turtles are very unlike to enter Puget Sound where hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces may occur. Impacts to loggerhead turtles from hull-fouling ANS are expected to be undetectable.
Species with Potentially Significant Risk from Exposure to Hull-fouling ANS **NONE**			
Species with Likely Significant Risk from Exposure to Hull-fouling ANS **NONE**			

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PUGET SOUND RAA **Number of deployable vessels (>79 feet) – 100** **Existing ANS[50]: At least 74 NIS (Davidson et al., 2014)[54]**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with No Risk from Exposure to Hull-Fouling ANS			
Frog, Oregon spotted, *Rana pretiosa*	No Exposure	Undetectable	Inhabits quiet freshwater habitats where hull-fouling ANS will not be introduced by in-water cleaning of vessels of the Armed Forces.
Howellia, Water, *Howellia aquatilis*	No Exposure	Major	Aquatic annual that grows submerged, rooted in bottom sediments of ponds and sloughs. Occurs in small vernal wetlands such as shallow, low-elevation glacial pothole ponds and former river oxbows that are inundated by spring rains and snowmelt runoff and typically dry out by end of the growing season. Hull-fouling ANS invasions could have major consequences for water howellia through competition for space, grazing pressure, or water quality changes. However, in-water cleaning of vessels of the Armed Forces will not occur in these habitats.
Whale, Blue, *Balaenoptera musculus*	Unlikely	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Fin, *Balaenoptera physalus*	Unlikely	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Humpback, Mexico DPS and Central America DPS, *Megaptera novaeangliae*	Unlikely	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Killer (southern resident), *Orcinus orca*	Unlikely	Undetectable	Species occurs predominantly offshore and will not be impacted by hull-fouling ANS invasions.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

PUGET SOUND RAA **Number of deployable vessels (>79 feet) – 100** **Existing ANS[50]: At least 74 NIS (Davidson et al., 2014)[54]**			
Species	**Likelihood of Exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with No Risk from Exposure to Hull-fouling ANS			
Whale, Sei, *Balaenoptera borealis*	Unlikely	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Sperm, *Physeter macrocephalus*	Unlikely	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN DIEGO RAA **Number of deployable vessels (>79 feet) – 101** **Existing ANS[50, 55]**			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Albatross, Short-tailed, *Phoebastria albatrus*	Unlikely	Undetectable	Feeds offshore. Mainly consumes squid, but also takes shrimp, fish, flying fish eggs, and other crustaceans. Hull-fouling ANS in underwater ship husbandry discharge will not be introduced to feeding areas, and exposure in nearshore areas is unlikely. Consequences of ANS introductions by vessels of the Armed Forces are expected to be undetectable.
Bird's-beak, Salt marsh, *Cordylanthus maritimus ssp. maritimus*	Unlikely	Undetectable	Occurs in upper terraces and higher edges of coastal salt marshes where tidal inundation is periodic. Species occurs in areas that will not be iinvaded by hull-fouling ANS. Therefore, impacts are expected to be undetectable.
Bocaccio, *Sebastes paucispinis*	Unlikely	Undetectable	Piscivorous fish that occurs in nearshore areas over rocky habitats at 12-480 m. Hull-fouling ANS are will not have detectable impacts on bocaccio or its food resources.

[55] Devil Weed (*Sargassum horneri*), Japanese Wireweed (*Sargassum muticum*), Asian Kelp (*Undaria pinnatifida*), Serpulid Tubeworm (*Hydroides elegans*), Lacy Tube Worm (*Filograna implexa*), Spaghetti Bryozoan (*Amathia verticillata*), Erect Bryozoan (*Bugula neritina*), Encrusting Bryozoan (*Watersipora subtorquata*), Anemone (*Bunodeopsis sp. A*), Pink-Mouthed Hydroid (*Ectopleura crocea*), Rope Grass Hydroid (*Garveia franciscana*), Amphipod (*Ampithoe valida*), Gammarid Amphipod (*Jassa marmorata*), Mud Tube Amphipod (*Laticorophium baconi*), Striped Acorn Barnacle (*Amphibalanus amphitrite*), Red Swamp Crayfish (*Procambarus clarkia*), Wood Boring Isopod (*Limnoria tripunctata*), Burrowing Isopod (*Sphaeroma quoianum*), Striped Bass (*Morone saxatilis*), Sailfin Molly (*Poecilia latipinna*), Chameleon Goby (*Tridentiger trigonocephalus*), Pacific Oyster (*Crassostrea gigas*), Blacktip Shipworm (*Lyrodus pedicellatus*), Mediterranean Mussel (*Mytilus galloprovincialis*), Naval Shipworm (*Teredo navalis*), Japanese Bubble-Shell Snail (*Haminoea japonica*), Colonial Tunicate (*Diplosoma listerianum*), Solitary Tunicate (*Styela plicata*)

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN DIEGO RAA **Number of deployable vessels (>79 feet) – 101** **Existing ANS[50, 55]**			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Ray, Giant Manta, *Manta birostris*	Unlikely	Undetectable	Predominantly an ocean-going species and spends most of its life far from land, traveling with the currents and migrating to areas where upwellings of nutrient-rich water increase the availability of zooplankton. Also observed in estuarine waters near oceanic inlets and coral reefs, which are used as nursery grounds. Hull-fouling ANS will not be introduced to open ocean areas by in-water cleaning of vessels of the Armed Forces. Impacts to the giant manta ray and its food resources from exposure to hull-fouling ANS are expected to be undetectable.
Sea Turtle, Hawksbill, *Eretmochelys imbricata*	Unlikely	Minor	Uses a wide range of tropical and subtropical habitats (shallow coastal waters with rocky bottoms, coral reefs, sea grass or algae beds, mangrove-bordered bays and estuaries, and submerged mud flats). Diet consist primarily of invertebrates (crabs, sea urchins, shellfish, jellyfish, etc.) but also includes plant material and fishes. Although introductions of hull-fouling species could have impacts on coastal habitats and invertebrate and algal food sources if they do occur however, because hawksbill sea turtles us a variety of habitats and food resources the impacts to these turtles are expected to be minor. In addition, exposure of hawksbill sea turtles and their food resources to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces in San Diego Bay is unlikely.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN DIEGO RAA Number of deployable vessels (>79 feet) – 101 Existing ANS[50, 55]			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Sea Turtle, Leatherback, *Dermochelys coriacea*	Very Unlikely	Undetectable	Nest on sandy beaches and have a diet of soft-bodied pelagic (open ocean) prey, such as jellyfish and salps. Because hull-fouling ANS will not become established on sandy beaches or be introduced to open ocean areas by in-water hull cleaning of vessels of the Armed Forces, exposure of leatherback sea turtles is very unlikely. Consequences of hull-fouling ANS for leatherback sea turtles are expected to be undetectable.
Shark, Scalloped Hammerhead Eastern Pacific DPS, *Sphyrna lewini*	Unlikely	Minor	Adults inhabit nearshore coastal waters and juveniles inhabit protected coastal bays. Adults feed offshore at night on a diet that consists predominantly of pelagic fish and squid. Juveniles feed on small fish and crustaceans. ANS invasions may affect the composition of the diet for juveniles, however impacts are anticipated to be minor. Exposure to hull-fouling ANS introduced to San Diego Bay from in-water cleaning of vessels of the Armed Forces is unlikely.
Tern, California Least, *Sterna antillarum browni*	Unlikely	Undetectable	Feeds in coastal areas mainly on small fishes such as anchovy, topsmelt, surf-perch, killifish, and mosquitofish obtained by diving from air into shallow water. Because food resources are pelagic, they are unlikely to be impacted by hull-fouling ANS and consequences of exposure of California least tern and its food resources to hull-fouling ANS are expected to be undetectable. In addition, exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces in San Diego Bay is unlikely.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN DIEGO RAA Number of deployable vessels (>79 feet) – 101 Existing ANS[50, 55]			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Trout, Steelhead South-Central California Coast DPS Southern California DPS, *Oncorhynchus mykiss*	Unlikely	Undetectable	Born in fresh water streams where they spend their first 1-3 years of life. Then migrate to the ocean where most of their growth occurs. After spending between one to four growing seasons in the ocean, return to their native fresh water stream to spawn. Young animals feed primarily on zooplankton. Adults feed on aquatic and terrestrial insects, mollusks, crustaceans, fish eggs, minnows, and other small fishes (including other trout). Because most of their life is spent either in freshwater or the offshore where hull-fouling ANS will not be introduced by in-water hull cleaning, exposure of steelhead and their food resources to hull-fouling ANS is very unlikely. Consequences of exposure of steelhead to hull-fouling ANS are expected to be undetectable.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN DIEGO RAA Number of deployable vessels (>79 feet) – 101 Existing ANS[50,55]			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Negligible Risk from Exposure to Hull-fouling ANS			
Grouper, Gulf, *Mycteroperca jordani*	Unlikely	Moderate	Found on rocky reefs and in kelp beds. Large adults feed on fishes. Reported to prey on juvenile hammerhead sharks. Although hull-fouling ANS could overgrow rocky reefs, graze kelp beds, and outcompete kelp for space, the consequences for Gulf grouper are expected to be moderate at most, and risk to Gulf grouper from ANS introduced by in-water cleaning of vessels of the Armed Forces is considered to be negligible.
Murrelet, Marbled, *Brachyramphus marmoratus*	Unlikely	Moderate	Diet consists of sandlance, herring, other small schooling fish and, in winter, invertebrates. Feeds in near-shore habitats up to 1.4 km offshore, in sheltered waters, lagoons, and sometimes inland lakes. Hull-fouling ANS could impact some food resources, having moderate consequences for marbled murrelet, but exposure to ANS introduced by in-water hull cleaning of vessels of the Armed Forces is unlikely.
Plover, Western snowy, *Charadrius nivosus nivosus*	Unlikely	Minor	Feeds on worms, fly larvae, beetles, crustaceans, mollusks, and other invertebrates. Hull-fouling ANS could affect the composition of the Western snowy plover diet but is unlikely to affect food availability. Consequences for Western snowy plover are expected to be minor. In addition, introduction of hull-fouling ANS from in-water cleaning of vessels of the Armed Forces is unlikely.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN DIEGO RAA Number of deployable vessels (>79 feet) – 101 Existing ANS[50,55]			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Negligible Risk from Exposure to Hull-fouling ANS			
Rail, Light-footed clapper, *Rallus longirostris levipes*	Unlikely	Moderate	Nests under clumps of pickleweed, on the ground or in cordgrass slightly above ground level. Mainly feeds on crabs, but also eats other crustaceans, small fishes, tadpoles, snails, insects, and some plant material. May probe in mud or sand in or near shallow water or pick prey items from substrate. ANS invasions could impact invertebrate food resources, and consequences for clapper rails could be moderate if a preferred prey item is impacted. However, exposure to ANS introduced by in-water hull cleaning of vessels of the Armed Forces is unlikely.
Sea turtle, Green East Pacific DPS[56], *Chelonia mydas*	Unlikely	Moderate	Found off the coast of California from July through September. Offshore, the turtles likely consume red crab, squid and discarded fish, rather than the invertebrates, macroalgae and seagrass typically eaten by green turtles in nearshore habitats. Within San Diego Bay, green turtles can most often be seen within the South San Diego Bay National Wildlife Refuge where seagrass beds provide a protected foraging and resting area. Introductions of hull-fouling species could have direct impacts to seagrass beds leading to moderate consequences for green sea turtles, if they occur. However, exposure to ANS introduced by in-water hull cleaning of vessels of the Armed Forces is unlikely.

[56] Includes the currently listed threatened Sea turtle, Green (All Others)

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN DIEGO RAA **Number of deployable vessels (>79 feet) – 101** **Existing ANS[50,55]**			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Negligible Risk from Exposure to Hull-fouling ANS			
Sea Turtle, Loggerhead North Pacific Ocean DPS, *Caretta caretta*	Unlikely	Minor	Inhabit pelagic waters, continental shelves, bays, lagoons, and estuaries, foraging in the highly productive coastal waters located on continental shelves. Commonly uses coral reefs, rocky areas, and shipwrecks for feeding. The diet of all life stages is mostly invertebrates and occasionally jellyfish. Because most feeding occurs in open water where hull-fouling ANS will not be introduced by in-water cleaning of vessels of the Armed Forces, exposure to ANS introduced by underwater ship husbandry discharge is unlikely. Impacts to loggerhead sea turtles from exposure to hull-fouling ANS are expected to be minor.
Sea turtle, Olive ridley Mexico's Pacific coast breeding colonies All other areas, *Lepidochelys olivacea*	Unlikely	Minor	Common prey items include jellyfish, tunicates, sea urchins, bryozoans, bivalves, snails, shrimp, crabs, rock lobsters, and sipunculid worms. Hull-fouling ANS could impact some prey items, causing a shift in dietary composition for olive ridley sea turtles. Because of the varied diet and limited occurrence of olive ridley sea turtles in San Diego Bay, consequences for olive ridley sea turtles are expected to be minor. Exposure of olive ridley sea turtles to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces in San Diego Bay is unlikely.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN DIEGO RAA **Number of deployable vessels (>79 feet) – 101** **Existing ANS[50,55]**			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Potentially Significant Risk from Exposure to Hull-fouling ANS			
Abalone, Black, *Haliotis cracherodii*	Unlikely	Major	Slow-moving marine gastropod; found in rocky intertidal and shallow subtidal reefs, ranging from the high tide line to a depth of up to five meters; clamps onto rock substrates and feeds on algal matter. Because they occur in coastal habitats, black abalone can withstand extreme variations in temperature, salinity, moisture, and wave action that most fouling organisms cannot, and hull-fouling organisms from vessels of the Armed Forces are not likely to be invade these habitats or be introduced to these habitats by in-water hull cleaning which is performed in relatively protected areas of ports and harbors. However, ANS invasions could have major consequences for black abalone by competing for food resources and space.
Abalone, White, *Haliotis sorenseni*	Unlikely	Major	Herbivorous gastropods found in open low and high relief rock or boulder habitat that is interspersed with sand channels. Found at depths greater than 26 meters amongst rocky reefs with understory kelps. Inhabits deeper water than the other California abalones. Introductions of hull-fouling ANS from in-water cleaning of vessels of the Armed Forces are unlikely to occur because cleaning will take place in more protected ports and harbors and NAS are unlikely to survive in offshore areas. However, ANS invasions could have major consequences for white abalone by competing for food resources and space.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN DIEGO RAA Number of deployable vessels (>79 feet) – 101 Existing ANS[50,55]			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Likely Significant Risk from Exposure to Hull-Fouling ANS **NONE**			
Species with No Risk from Exposure to Hull-Fouling ANS			
Fairy Shrimp, San Diego, *Branchinecta sandiegonensis*	No Exposure	Major	Occurs in vernal pools and similar ephemeral wetland types, including artificial habitats. Presumed to feed on bacteria, protozoa, rotifers, and bits of organic matter. Hull-fouling ANS that are filter feeders could have major impacts for fairy shrimp food resources; however, hull-fouling ANS from in-water cleaning of vessels of the Armed Forces will not be introduced to these ephemeral habitats.
Flycatcher, Southwestern Willow, *Empidonax trailii extimus*	No Exposure	Moderate	Nests in riparian vegetation and feeds on emergent aquatic insects in freshwater habitats. Hull-fouling ANS could impact the quantity or quality of food resources and have moderate impacts for the flycatcher; however, hull-fouling ANS will not be introduced to these habitats from in-water cleaning of vessels of the Armed Forces. Therefore, there is no risk to the southwestern willow flycatcher.
Seal, Guadalupe fur, *Arctocephalus townsendi*	No Exposure	Undetectable	Feeds offshore, and neither offshore resources nor onshore breeding habitat will be impacted by hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces.
Vireo, Least Bell's, *Vireo bellii pusillus*	No Exposure	Undetectable	Occurs in dense brush, mesquite, willow-cottonwood forest, streamside thickets, and scrub oak, in arid regions but often near water. Obtain prey primarily by foliage gleaning (picking prey from leaf or bark substrates), and hovering (removing prey from vegetation surfaces while fluttering in the air). Hull-fouling ANS are not expected to impact least Bell’s vireo habitat or prey.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN DIEGO RAA Number of deployable vessels (>79 feet) – 101 Existing ANS[50,55]			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with No Risk from Exposure to Hull-Fouling ANS			
Whale, Blue, *Balaenoptera musculus*	No exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Fin, *Balaenoptera physalus*	No exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Humpback, Mexico DPS and Central America DPS, *Megaptera novaeangliae*	No exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Sei, *Balaenoptera borealis*	No exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Sperm, *Physeter macrocephalus*	No exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN FRANCISCO RAA Number of deployable vessels (>79 feet) – 13 Existing ANS[50]: More than 200 non-indigenous species[57]			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Bird's-beak, Soft, *Cordylanthus mollis ssp. mollis*	Unlikely	Undetectable	Occurs in the upper reaches of salt grass/pickleweed marshes at or near the limits of tidal action; can tolerate somewhat brackish soil but has its limits. Hull-fouling ANS are unlikely to invade the upper reaches of marshes. Consequences for soft bird's-beak are expected to be undetectable.

[57] Species include Asian Kelp (*Undaria pinnatifida*), Green Sea Fingers (*Codium fragile*), Japanese Wireweed (*Sargassum muticum*), Atlantic Cordgrass (*Spartina alterniflora*), Salt Meadow Cordgrass (*Spartina patens*), Spaghetti Bryozoan (*Amathia verticillata*), Erect Bryozoan (*Amathia verticillata*), Encrusting Bryozoan (*Watersipora subtorquata*), Pink-Mouthed Hydroid (*Ectopleura crocea*), Rope Grass Hydroid (*Garveia franciscana*), Moon Jelly (*Aurelia* sp.), Tube-building Amphipod (*Ampelisca abdita*), Japanese Skeleton Shrimp (*Caprella mutica* and *Caprella scaura*), Wood-boring Amphipod (*Chelura terebrans*), Asian Clams (*Potamocorbula amurensis* and *Corbicula fluminea*), Overbite Clam (*Corbula amurensis*), Mediterranean Mussel (*Mytilus galloprovincialis*), Atlantic Ribbed Mussel (*Geukensia demissa*), Naval Shipworm (*Tereda navalis*), Japanese Bubble Snail (*Haminoea japonica*), Eastern Mudsnail (*Ilyanassa obsoleta*), Solitary Tunicate (*Ciona robusta*), Colonial Tunicates (*Didemnum vexillum* and *D. listerianum*), South African Sabellid Worm (*Terebrasabella heterouncinata*), Bar-Gilled Mud Worm *Streblospio benedicti*), Calanoid Copepods (*Mytilicola orientalis* and *Pseudodiaptomus forbesi*), Burrowing Isopod (*Sphaeroma quoianum*), Chinese Mitten Crab (*Eriocheir sinensis*), European Green Crab (*Carcinus maenas*), Red Swamp Crayfish (*Procambarus clarkia*), Inland Silverside (*Menidia beryllina*), Yellowfin Goby (*Acanthogobius flavimanus*), American Shad (*Alosa sapidissima*), Wakasagi (*Hypomesus nipponensis*), Mississippi Silversides (*Menidia audens*), Striped Bass (*Morone saxatilis*), Shimofuri Goby (*Tridentiger bifasciatus*)

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN FRANCISCO RAA **Number of deployable vessels (>79 feet) – 13** **Existing ANS[50]: More than 200 non-indigenous species[57]**			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Eulachon, *Thaleichthys pacificus*	Unlikely	Undetectable	Spend most of their adult lives in nearshore ocean bottom habitat and coastal inlets but return to their natal freshwater streams and rivers to spawn and die. Larvae are found near the bottom and carried downstream soon after hatching to salt water and eventually found in the scattering layer of coastal waters at 20 to greater than 200 m. Adults feed on krill and juveniles feed on copepod larvae, phytoplankton, copepods and other zooplankton. Hull-fouling organisms could have an impact on plankton in coastal embayments, but larval fish are carried downstream and out to sea soon after hatching, limiting their time in embayments where hull-fouling ANS from in-water cleaning of vessels of the Armed Forces are likely to occur. Therefore, consequences of ANS introductions by underwater ship husbandry discharge are expected to be minor. In addition, exposure to hull-fouling ANS introduced to San Francisco Bay by in-water cleaning of vessels of the Armed Forces is unlikely.
Plover, Western snowy, *Charadrius nivosus nivosus*	Unlikely	Undetectable	Feeds on worms, fly larvae, beetles, crustaceans, mollusks, and other invertebrates. Hull-fouling ANS could affect the composition of the Western snowy plover diet but is unlikely to affect food availability. Consequences for Western snowy plover are expected to be minor. In addition, introduction of hull-fouling ANS from in-water cleaning of vessels of the Armed Forces is unlikely.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN FRANCISCO RAA Number of deployable vessels (>79 feet) – 13 Existing ANS[50]: More than 200 non-indigenous species[57]			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Sea turtle, Green East Pacific DPS[58], *Chelonia mydas*	Very Unlikely	Moderate	Uses seagrass beds for feeding and beaches for nesting. Introductions of fouling species that could have direct impacts to seagrass beds could have moderate consequences for green sea turtles. However, green sea turtles rarely enter San Francisco Bay and are only occasionally seen off the coast; therefore, exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces is very unlikely.
Sea turtle, Leatherback, *Dermochelys coriacea*	Very Unlikely	Undetectable	Nest on sandy beaches and have a diet of soft-bodied pelagic (open ocean) prey, such as jellyfish and salps. Hull-fouling ANS are unlikely to impact food resources, and consequnces for leatherback sea turtles will be undetectable. Further, leatherback sea turtles have only been seen offshore and have not been sighted in San Francisco Bay; therefore, exposure to hull-foulinge ANS introduced by in-water cleaning of vessels of the Armed Forces is very unlikely. Therefore, risk from hull-fouling ANS is considered to be remote.

[58] Includes the currently listed threatened Sea turtle, Green (All Others)

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN FRANCISCO RAA Number of deployable vessels (>79 feet) – 13 Existing ANS[50]: More than 200 non-indigenous species[57]			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Sea Turtle, Loggerhead North Pacific Ocean DPS, *Caretta caretta*	Unlikely	Undetectable	Uses coral reefs, rocky areas, and shipwrecks for feeding, which could be impacted by hull-fouling ANS. The North Pacific population nests along the coast of Japan and travels up to 7,000 miles to feed in coastal Mexico and Southern California. Loggerheads are occasionally seen in San Francisco Bay Area waters, but do not nest or extensively feed. Therefore, exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces is unlikely. Consequences for loggerhead sea turtles and their food resources from exposure to hull-fouling ANS are expected to be undetectable. Therefore, risk from ANS introductions by in-water hull cleaning are considered to be remote.
Sea turtle, olive ridley (all other areas), *Lepidochelys olivacea*	Very Unlikely	Minor	Common prey items include jellyfish, tunicates, sea urchins, bryozoans, bivalves, snails, shrimp, crabs, rock lobsters, and sipunculid worms, any of which could be impacted by hull-fouling ANS, having only minor consequences for olive ridley sea turtles because of their varied diet. Olive ridley sea turtles are rarely seen in the San Francisco Bay Area, so exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces is very unlikely. Therefore, risk from ANS introductions by in-water hull cleaning are considered to be remote.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN FRANCISCO RAA **Number of deployable vessels (>79 feet) – 13** **Existing ANS[50]: More than 200 non-indigenous species[57]**			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Shark, Scalloped Hammerhead Eastern Pacific DPS, *Sphyrna lewini*	Unlikely	Minor	A coastal and semi-oceanic pelagic shark, found over continental and insular shelves and in deep water near to them, ranging from the intertidal and surface to at least 275 m depth. Adults inhabit nearshore coastal waters and juveniles inhabit protected coastal bays. Adults feed on mesopelagic fish and squids; pups and juveniles feed mainly on benthic reef fishes (e.g., scarids and gobiids), demersal fish, and crustaceans. Exposure to ANS introduced by in-water cleaning of vessels of the Armed Forces is unlikely. ANS invasions may affect the composition of the diet for juveniles. However, impacts for sharks are anticipated to be minor.
Trout, Steelhead, California Central Valley DPS Central California Coast DPS *Oncorhynchus mykiss*	Very Unlikely	Undetectable	Born in fresh water streams where they spend their first 1-3 years of life. Then migrate to the ocean where most of their growth occurs. After spending between one to four growing seasons in the ocean, return to their native fresh water stream to spawn. Young animals feed primarily on zooplankton. Adults feed on aquatic and aquatic life stages of insects, mollusks, crustaceans, fish eggs, minnows, and other small fishes (including other trout). Because most of their life is spent either in freshwater or the offshore where hull-fouling ANS will not be introduced by in-water hull cleaning, exposure of steelhead and their food resources to hull-fouling ANS is very unlikely. Consequences of exposure of steelhead to hull-fouling ANS are expected to be undetectable.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN FRANCISCO RAA **Number of deployable vessels (>79 feet) – 13** **Existing ANS[50]: More than 200 non-indigenous species[57]**			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Negligible Risk from Exposure to Hull-fouling ANS			
Mouse, Salt marsh harvest, *Reithrodontomys raviventris*	Unlikely	Moderate	Habitat consists of salt and brackish marshes where plants provide a dense mat of cover, ideally around 30-50 cm high with a high percentage (e.g., 60%) of pickleweed (*Salicornia*) and complex structure of Atriplex and other species. Eats salt grass (*Distichlis*) and pickleweed as well as some seeds. Aquatic margins of habitat could be impacted by some species of hull-fouling ANS, if they are introduced. Exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces is unlikely.
Murrelet, Marbled, *Brachyramphus marmoratus*	Unlikely	Moderate	Diet consists of sandlance, herring, other small schooling fish and, in winter, invertebrates. Feeds in near-shore habitats up to 1.4 km offshore, in sheltered waters, lagoons, and sometimes inland lakes. Hull-fouling ANS could impact some food resources, having moderate consequences for marbled murrelet, but exposure to ANS introduced by in-water hull cleaning of vessels of the Armed Forces is unlikely.
Otter, Southern sea, *Enhydra lutris nereis*	Unlikely	Moderate	Keystone predator that feeds in shallow (up to 20 m) coastal areas. Feeds predominantly on sea urchins, crabs, and a variety of mollusks, but fish are important food items at high population densities. Predation on herbivores determines structure of off-shore kelp communities where they seek refuge. Both herbivorous prey and kelp could be impacted by hull-fouling ANS. However, consequences for sea otters are expected to be moderate at most, and exposure to ANS introduced by in-water hull cleaning of vessels of the Armed Forces is unlikely.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN FRANCISCO RAA Number of deployable vessels (>79 feet) – 13 Existing ANS[50]: More than 200 non-indigenous species[57]			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Negligible Risk from Exposure to Hull-fouling ANS			
Thistle, Suisun, *Cirsium hydrophilum var. hydrophilum*	Unlikely	Undetectable	Coastal saltmarsh plant that is being displaced by *Lepidium latifolium* (perennial pepperweed). Occurs at or above the mean high water mark. Fully aquatic fouling species are unlikely to invade this part of the intertidal zone; therefore, exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces is unlikely. Consequences from exposure to hull-fouling ANS are expected to be undetectable.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN FRANCISCO RAA Number of deployable vessels (>79 feet) – 13 Existing ANS[50]: More than 200 non-indigenous species[57]			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Potentially Significant Risk from Exposure to Hull-fouling ANS			
Abalone, Black, *Haliotis cracherodii*	Unlikely	Major	Slow-moving marine gastropod found in the intertidal zone ranging from the high tide line to a depth of up to five meters. Clamps onto rock substrates and feeds on algae. Because they occur in coastal habitats, black abalone can withstand extreme variations in temperature, salinity, moisture, and wave action that many fouling organisms cannot. Hull-fouling organisms are not likely to be introduced to these habitats by in-water hull cleaning of vessels of the Armed Forces, which is performed in relatively protected areas of ports and harbors. However, ANS invasions could have major consequences for black abalone by competing for food resources and space.
Abalone, White, *Haliotis sorenseni*	Unlikely	Major	Herbivorous gastropods found in open low and high relief rock or boulder habitat that is interspersed with sand channels; clamps onto rock substrates and feeds on algal matter. Found at depths greater than 26 meters amongst rocky reefs with understory kelps. Inhabits deeper water than the other California abalones. Introductions of hull-fouling ANS from in-water cleaning of vessels of the Armed Forces are unlikely to occur in these locations because cleaning will take place in more protected ports and harbors, and NAS are unlikely to survive in offshore areas. However, ANS invasions could have major consequences for white abalone by competing for food resources and space.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN FRANCISCO RAA **Number of deployable vessels (>79 feet) – 13** **Existing ANS[50]: More than 200 non-indigenous species[57]**			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Potentially Significant Risk from Exposure to Hull-fouling ANS			
Goby, Tidewater, *Eucyclogobius newberryi*	Unlikely	Major	Found primarily in brackish coastal lagoons, estuaries, and marshes. Nest in burrows dug in coarse sandy substrate. Hull-fouling ANS could compete with tidewater gobies for food and space, and consequences could be major. However, exposure to ANS introduced by in-water hull cleaning of vessels of the Armed Forces is unlikely.
Smelt, Delta, *Hypomesus transpacificus*	Unlikely	Major	Euryhaline species that typically lives only one year and inhabits open waters of bays, tidal rivers, channels, and sloughs; rarely occurs in water with salinity of more than 10-12 ppt. Spawns in freshwater rivers. Feeds mainly on plankton (copepods, cladocerans) and amphipods, but also eats insect larvae and opossum shrimp (*Neomysis* sp.). Most important food organism for all sizes appears to be the euryhaline copepod, *Eurytemora affinis*. Introduction of invasive hull-fouling species from in-water cleaning of vessels of the Armed Forces is unlikely. However, the introduction of filter feeding species could lead to major consequences because of competition for resources and the small numbers of smelt remaining.
Sturgeon, Green, *Acipenser medirostris*	Unlikely	Major	Spend most of their lives in coastal marine waters, estuaries, and the lower reaches of large rivers, ascending rivers to spawn. Adults feed on bottom invertebrates and small fish, and introduction and invasion by hull-fouling species could result in a change in dietary composition, potentially having major consequences for juvenile fish. However, introduction of hull-fouling ANS from in-water cleaning of vessels of the Armed Forces is unlikely.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN FRANCISCO RAA Number of deployable vessels (>79 feet) – 13 Existing ANS[50]: More than 200 non-indigenous species[57]			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Likely Significant Risk from Exposure to Hull-fouling ANS **NONE**			
Species with No Risk from Exposure to Hull-fouling ANS			
Beetle, Delta Green Ground, *Elaphrus viridis*	No exposure	Undetectable	Mostly found along the margins of vernal pools within 1.5 meters of the water and consume soft-bodied prey (predominantly springtails). Vessels of the Armed Forces do not operate in these locations.
Clover, Showy Indian, *Trifolium amoenum*	No exposure	Undetectable	Typically occurs in low, wet swales in grasslands. Also, on grassy hillsides at up to about 400 m elevation. Open, sunny sites, sometimes on serpentine soil in coastal bluff scrub and valley and foothill grassland. Vessels of the Armed Forces do not operate in these locations.
Frog, California red-legged, *Rana draytonii*	No Exposure	Minor	Requires a variety of habitat elements with aquatic breeding areas embedded within a matrix of riparian and upland dispersal habitats. Breeding sites are in aquatic habitats including pools and backwaters within streams and creeks, ponds, marshes, springs, sag ponds, dune ponds, lagoons, and artificial impoundments. Hull-fouling ANS could have minor impacts to water quality and food resources but will not be introduced to these locations from in-water cleaning of vessels of the Armed Forces.
Goldfields, Contra Costa, *Lasthenia conjugens*	No Exposure	Major	Occurs in vernal pools in open grassy areas at elevations up to 470 m. ANS could overgraze goldfields or outcompete goldfields for space. However, vessels of the Armed Forces do not operate in these locations.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN FRANCISCO RAA **Number of deployable vessels (>79 feet) – 13** **Existing ANS[50]: More than 200 non-indigenous species[57]**			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with No Risk from Exposure to Hull-Fouling ANS			
Manzanita, San Francisco, *Arctostaphylos franciscana*	No Exposure	Undetectable	Lowland scrub species that does not occur in locations where it will be impacted by hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces.
Meadowfoam, Sebastopol, *Limnanthes vinculans*	No Exposure	Undetectable	Occurs in vernally or permanently wet meadows subjected to periodic inundation by heavy rains. Hull-fouling ANS will not be introduced to these habitats by in-water hull cleaning of vessels of the Armed Forces.
Potentilla, Hickman's, *Potentilla hickmanii*	No Exposure	Undetectable	Currently known to occur in two coastal locations. The key to the habitat for this species is the decomposed granite substrate that lies directly under the very fine-grained grassland topsoil. Hull-fouling ANS from in-water cleaning of vessels of the Armed Forces will not be introduced to these habitats.
Salamander, California tiger, *Ambystoma californiense*	No Exposure	Major	Lives in vacant or mammal-occupied burrows (e.g., California ground squirrel, valley pocket gopher), and occasionally other underground retreats, throughout most of the year in grassland, savanna, or open woodland habitats. Lays eggs on submerged stems and leaves, in shallow ephemeral or semi-permanent pools and ponds that fill during heavy winter rains or in permanent ponds; adults spend little time in breeding sites. ANS could prey on salamander eggs or graze vegetation used for breeding, having major consequences for tiger salamanders. However, hull-fouling ANS will not be introduced to these locations by in-water cleaning of vessels of the Armed Forces.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN FRANCISCO RAA Number of deployable vessels (>79 feet) – 13 Existing ANS[50]: More than 200 non-indigenous species[57]			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with No Risk from Exposure to Hull-Fouling ANS			
Seal, Guadalupe fur, *Arctocephalus townsendi*	No Exposure	Undetectable	Feeds offshore, and neither offshore resources nor onshore breeding habitat will be exposed to or impacted by hull-fouling ANS.
Shrimp, California freshwater, *Syncaris pacifica*	No exposure	Moderate	Detritivore that occurs in coastal freshwater creeks. Invasions by non-indigenous algae could change the quality of habitat and/or food resources, and invasions by other crustaceans could result in competition for resources. Consequences for the California freshwater shrimp would be expected to be moderate. However, hull-fouling ANS will not be introduced to these locations by in-water cleaning of vessels of the Armed Forces.
Shrimp, Conservancy fairy, *Branchinecta conservatio*	No Exposure	Major	Inhabits turbid, slightly alkaline, large, deep, vernal pools and winter lakes in California grassland areas. Consumes detritus and invertebrates. Hull-fouling ANS could impact food resources, having major consequences for the conservancy fairy shrimp. However, vessels of the Armed Forces do not operate in these locations and hull-fouling ANS will not be introduced by in-water cleanings.
Shrimp, Vernal pool fairy, *Branchinecta lynchi*	No Exposure	Major	Inhabits vernal pools and similar ephemeral wetlands. It is most commonly found in grass or mud bottomed pools or basalt flow depression pools in unplowed grasslands. Consumes detritus and invertebrates. ANS could impact food resources, having major consequences for the conservancy fairy shrimp. However, vessels of the Armed Forces do not operate in these locations, and hull-fouling ANS will not be introduced by in-water cleanings.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN FRANCISCO RAA Number of deployable vessels (>79 feet) – 13 Existing ANS[50]: More than 200 non-indigenous species[57]			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with No Risk from Exposure to Hull-Fouling ANS			
Shrimp, Vernal pool tadpole, *Lepidurus packardi*	No Exposure	Minor	Found in a variety of natural, and artificial, seasonally ponded habitat types including: vernal pools, swales, ephemeral drainages, stock ponds, reservoirs, ditches, backhoe pits, and ruts caused by vehicular activities. Wetland habitats vary in size from very small (2 square meters) to very large (356,253 square meters) and exhibit extremes in depth (2-15 cm) and volume (23-9,262,573 cubic meters). Adults are omnivorous, foraging on detritus, vegetation and other aquatic invertebrates when available. ANS could impact food resources, but consequences for the vernal pool tadpole shrimp are expected to be minor because of the varied diet. Vessels of the Armed Forces do not operate in these areas, and there will not be any introduction of hull-fouling ANS to these habitats by in-water cleaning of vessels.
Snake, Giant garter, *Thamnophis gigas*	No Exposure	Undetectable	Highly aquatic species that primarily inhabits freshwater marshes and sloughs, sometimes low-gradient streams, ponds, and small lakes, with cattails, bulrushes, willows, or other emergent or water-edge vegetation used for basking and cover. Feeds primarily on aquatic fish, frogs and tadpoles. Historical prey has been extirpated in much of this snake's range, leaving it to consume introduced fish and bullfrogs. Hull-fouling ANS from in-water cleaning of vessels of the Armed Forces will not be introduced to these habitats.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN FRANCISCO RAA Number of deployable vessels (>79 feet) – 13 Existing ANS[50]: More than 200 non-indigenous species[57]			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with No Risk from Exposure to Hull-Fouling ANS			
Sonoma Alopecurus, *Alopecurus aequalis var. sonomensis*	No Exposure	Moderate	Occurs in riparian freshwater marshes. Unlikely invasion by hull-fouling species could lead to increased grazing pressure. Hull-fouling ANS will not be introduced to Sonoma alopecurus habitat from in-water cleaning of vessels of the Armed Forces.
Sunshine, Sonoma, *Blennosperma bakeri*	No Exposure	Undetectable	Vernal pools in valley grassland. Vessels of the Armed Forces do not operate in theses locations, and hull-fouling ANS will not be introduced by in-water hull cleaning.
Thistle, Fountain, *Cirsium fontinale var. fontinale*	No Exposure	Undetectable	Found in serpentine seeps, chaparral (openings), cismontane woodland habitats, meadows and seeps, and valley and foothill grassland where hull-fouling ANS from in-water cleaning of vessels of the Armed Forces will not be introduced.
Whale, Blue, *Balaenoptera musculus*	No Exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Fin, *Balaenoptera physalus*	No Exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Humpback Mexico DPS Central America DPS, *Megaptera novaeangliae*	No Exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Sei, *Balaenoptera borealis*	No Exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.
Whale, Sperm, *Physeter macrocephalus*	No Exposure	Undetectable	Species occurs offshore and will not be impacted by ANS invasions.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

SAN FRANCISCO RAA **Number of deployable vessels (>79 feet) – 13** **Existing ANS[50]: More than 200 non-indigenous species[57]**			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with No Risk from Exposure to Hull-Fouling ANS			
Whipsnake, California, *Masticophis lateralis euryxanthus*	No Exposure	Minor	Occurs in chaparral foothills, shrublands with scattered grassy patches, rocky canyons and watercourses, and adjacent habitats. Preys on insects, frogs, lizards, snakes, small mammals, and birds. Although hull-fouling ANS could impact some of the whipsnake's food resources, consequences would be expected to be minor. However, vessels of the Armed Forces do not operate in locations inhabited by California whipsnake.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

<table>
<tr><th colspan="4">ST. LOUIS RAA
Number of deployable vessels (>79 feet) – 1
Existing ANS[50]: Asian Carp (Cyprinus carpio), Grass Carp (Ctenopharyngodon idella), Freshwater Jellyfish (Craspedacusta sowerbyi), Asian Clam (Corbicula fluminea), Quagga Mussel (Dreissena rostriformis bugensis), Zebra Mussel (Dreissena polymorpha), Cattail (Typha angustifolia)</th></tr>
<tr><th>Species</th><th>Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]</th><th>Consequences of ANS Invasion</th><th>Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces</th></tr>
<tr><th colspan="4">Species with Remote Risk from Exposure to Hull-fouling ANS</th></tr>
<tr><td>Bat, Gray, Myotis grisescens</td><td>Unlikely</td><td>Minor</td><td>The St. Louis RAA is within home range of the gray bat. Feeds on aquatic insects that emerge predominantly from waterbodies that will not be exposed to underwater ship husbandry discharge from in-water cleaning of vessels of the Armed Forces. Although hull-fouling ANS could cause a change in dietary composition for the gray bat, consequences are expected to be minor and exposure of gray bats and their prey to hull-fouling ANS from vessels of the Armed Forces is unlikely.</td></tr>
<tr><td>Bat, Indiana, Myotis sodalis</td><td>Unlikely</td><td>Minor</td><td>Diet consists of flying insects and reflects prey present in available foraging habitat. Forages along river and lake shorelines, in the crowns of trees in floodplains, and in upland forest. It is unlikely that ANS will be introduced by in-water cleaning of vessels of the Armed Forces; if ANS invasions do occur, however, invasions could lead to shifts in dietary composition with minor consequences for the Indiana bat.</td></tr>
<tr><td>Bat, Northern long-eared, Myotis septentrionalis</td><td>Unlikely</td><td>Minor</td><td>Feeds on moths, flies, leafhoppers, caddisflies, and beetles caught while in flight using echolocation, as well as by gleaning motionless insects from vegetation and water surfaces. It is unlikely that ANS will be introduced by in-water cleaning of vessels of the Armed Forces; if they do occur, however, invasions could alter dietary composition with minor consequences for the Northern long-eared bat.</td></tr>
</table>

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

ST. LOUIS RAA **Number of deployable vessels (>79 feet) – 1** **Existing ANS[50]: Asian Carp (*Cyprinus carpio*), Grass Carp (*Ctenopharyngodon idella*), Freshwater Jellyfish (*Craspedacusta sowerbyi*), Asian Clam (*Corbicula fluminea*), Quagga Mussel (*Dreissena rostriformis bugensis*), Zebra Mussel (*Dreissena polymorpha*), Cattail (*Typha angustifolia*)**			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Remote Risk from Exposure to Hull-fouling ANS			
Massasauga, Eastern (=rattlesnake), *Sistrurus catenatus*	Unlikely	Undetectable	Habitat includes sphagnum bogs, fens, swamps, marshes, peatlands, wet meadows, and floodplains in fall, winter and spring and open savannas, prairies, old fields, and dry woodlands in summer. Habitat loss and fragmentation are the biggest threats. Hull-fouling species are unlikely to be introduced by in-water cleaning of vessels of the Armed forces, and any ANS invasions will not affect Eastern massasauga habitat.
Rail, Eastern black, *Laterallus jamaicensis ssp. jamaicensis*	Very Unlikely	Minor	Although historically known from Missouri, there are no recent records along the Mississippi River. Presently are reliably located within the Arkansas River Valley of Colorado,in south central Kansas, and has a patchy distribution in Oklahoma. Inhabits wet sedge meadows with dense cover. Feeds on terrestrial and aquatic invertebrates, as well as small seeds. Composition of diet could be affected by hull-fouling ANS but is unlikely to affect food availability and feeding behavior. Because of its limited interior distribution and because only one vessel of the Armed Forces could be cleaned in water, exposure to hull-fouling ANS introduced by in-water cleaning of vessels of the Armed Forces in the Mississippi RAA is very unlikely.
Tern, Least, *Sterna antillarum*	Unlikely	Minor	Interior populations of least terns feed almost entirely on cyprinids (minnows) but will also consume insect larvae. Introductions of hull-fouling ANS could lead to changes in dietary composition with minor consequences for terns. However, it is unlikely that ANS invasions will occur as a result of in-water cleaning of vessels of the Armed Forces, and risk is considered to be remote.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

ST. LOUIS RAA **Number of deployable vessels (>79 feet) – 1** **Existing ANS[50]: Asian Carp (*Cyprinus carpio*), Grass Carp (*Ctenopharyngodon idella*), Freshwater Jellyfish (*Craspedacusta sowerbyi*), Asian Clam (*Corbicula fluminea*), Quagga Mussel (*Dreissena rostriformis bugensis*), Zebra Mussel (*Dreissena polymorpha*), Cattail (*Typha angustifolia*)**			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Negligible Risk from Exposure to Hull-fouling ANS			
Aster, Decurrent false, *Boltonia decurrens*	Unlikely	Undetectable	Colonizes riverine habitats characterized by moist, sandy soil and regular disturbance. Prefer areas of periodic flooding, which maintains open areas with high light levels. Introduction of hull-fouling ANS from in-water cleaning of vessels of the Armed Forces is unlikely and impacts from ANS introduced by vessels of the Armed Forces are not expected.
Plover, Piping, *Charadrius melodus*	Unlikely	Moderate	Nests on sandbars in the Missouri River from April to September and feeds in low flow backwater areas where macroinvertebrates are most abundant. Biggest threats are from loss of floodplain habitat suitable for feeding. Invasive fouling species introductions could have moderate consequences for piping plover if ANS alter the availability or composition of invertebrate food resources. However, introduction of ANS by in-water cleaning of vessels of the Armed Forces is unlikely.
Red Knot, *Calidris canutus rufa*	Unlikely	Moderate	Makes long migrations between nesting areas in mid- and high arctic latitudes and southern nonbreeding habitats as far north as the coastal United States (low numbers) and southward to southern South America. Use a variety of habitats including herbaceous wetlands, river mouth/tidal river habitats, and tidal flats/shores. Feed on a variety of invertebrates. Although hull-fouling ANS could impact food resources, consequences for red know are expected to be moderate because their diet is varied. Hull-fouling ANS are unlikely to be introduced to Mississippi River by in-water cleaning of vessels of the Armed Forces, and risk is considered to be negligible.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

ST. LOUIS RAA **Number of deployable vessels (>79 feet) – 1** **Existing ANS[50]: Asian Carp (*Cyprinus carpio*), Grass Carp (*Ctenopharyngodon idella*), Freshwater Jellyfish (*Craspedacusta sowerbyi*), Asian Clam (*Corbicula fluminea*), Quagga Mussel (*Dreissena rostriformis bugensis*), Zebra Mussel (*Dreissena polymorpha*), Cattail (*Typha angustifolia*)**			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Negligible Risk from Exposure to Hull-fouling ANS			
Sturgeon, Pallid, *Scaphirhynchus albus*	Unlikely	Minor	Bottom-oriented, large river fish inhabiting the Missouri and Mississippi rivers and some tributaries from Montana to Louisiana. Appear to use a mixture of sand, gravel and rock substrates during the winter and spring and underwater sand dunes during the summer and fall. Fry likely feed on zooplankton and/or small invertebrates, and fouling species invasion could lead to a change in dietary composition with moderate consequences for pallid sturgeon. However, it is unlikely that ANS invasions will occur as a result of in-water cleaning of vessels of the Armed Forces.
Species with Potentially Significant Risk from Exposure to Hull-fouling ANS			
Mussel, Scaleshell, *Leptodea leptodon*	Unlikely	Major	Occurs in medium to large rivers with low to moderate gradients in a variety of stream habitats including gravel, cobble, boulders, and occasionally mud or sand substrates. Bivalve species that could potentially attach to the hull of vessels of the Armed Forces could have major consequences for the scaleshell mussels because of competition for food and space. However, it is unlikely that ANS invasions will occur as a result of in-water cleaning of vessels of the Armed Forces.
Spectaclecase (mussel), *Cumberlandia monodonta*	Unlikely	Major	Adults of this species are essentially sessile. Occurs in large rivers and is a habitat-specialist, relative to other mussel species. It seems to most often inhabit riverine microhabitats that are sheltered from the main force of current. Hull-fouling ANS, particularly other bivalves, could compete with the spectaclecase for food and space and could have major consequences. However, it is unlikely that ANS will be introduced by in-water cleaning of vessels of the Armed Forces.

Table 5-9. Summary of Risk Conclusions for Federally Listed Species from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

ST. LOUIS RAA **Number of deployable vessels (>79 feet) – 1** **Existing ANS[50]: Asian Carp (*Cyprinus carpio*), Grass Carp (*Ctenopharyngodon idella*), Freshwater Jellyfish (*Craspedacusta sowerbyi*), Asian Clam (*Corbicula fluminea*), Quagga Mussel (*Dreissena rostriformis bugensis*), Zebra Mussel (*Dreissena polymorpha*), Cattail (*Typha angustifolia*)**			
Species	**Likelihood of exposure to ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces[4]**	**Consequences of ANS Invasion**	**Conclusions Regarding Risk of Impact from ANS Introduced by In-Water Cleaning of Vessels of the Armed Forces**
Species with Likely Significant Risk from Exposure to Hull-Fouling ANS **NONE**			
Species with No Risk from Exposure to Hull-fouling ANS			
Orchid, Eastern prairie fringed, *Platanthera leucophaea*	No Exposure	Undetectable	Occurs in a wide variety of habitats from mesic prairies to wetlands such as sedge meadows, marsh edges, and bogs. Found in wet terrestrial habitats. Vessels of the Armed Forces do not operate in these locations.

Table 5-10. Potential Magnitude of Impacts for Taxonomic Groups of Federally Listed Species if Exposed to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors

Taxonomic Group	Highest Risk of Impact from ANS	Presumptions
Saltwater Corals	Potentially Significant	Because of their location in environmentally more favorable climates, ports and harbors inhabited by corals are more susceptible to non-indigenous species introductions and invasions. Invasion by certain hull-fouling species, particularly algae that can overgrow coral reefs or sessile invertebrates that can settle on available substrates and compete with corals for space, can have major consequences for corals.
Unionid Mussels	Potentially Significant	Invasion by hull-fouling species, particularly non-indigenous bivalves, could have major consequences for unionid mussels. These other species could compete for food and space, and several ANS have already significantly impacted unionid mussels. Additional introductions could lead to further impact.
Freshwater Snails	Potentially Significant	Hull-fouling ANS introduced by vessels of the Armed Forces could outcompete gastropods for food and space and have major consequences for gastropod populations.
Saltwater Mollusks	Potentially Significant (saltwater snails only)	Hull-fouling ANS introduced by vessels of the Armed Forces could outcompete gastropods for food and space and have major consequences for gastropod populations. Marine gastropods that are found in the intertidal zone and graze on rocky substrates are particularly vulnerable to impact.
Freshwater and Saltwater Shrimp/Crustaceans	Remote	Federally listed freshwater shrimp and other crustaceans occur in vernal pools and springs where vessels of the Armed Forces do not operate.
Freshwater Fish/Inland Salmonids	Negligible	Inland salmonid species have both migratory and non-migratory populations. Migratory populations may migrate between streams/creeks and either larger freshwater bodies (lakes) or open ocean. Inland salmonids feed predominantly on insect larvae and small fish. An ANS invasion could result in a shift in dietary composition for juvenile and/or adult fish that could have moderate impacts to inland salmonid populations.
Anadromous Salmonids	Potentially Significant	These species spend most of their lives in coastal marine waters, estuaries, and the lower reaches of large rivers, ascending rivers to spawn. Adults feed on bottom invertebrates and small fish, and introduction and invasion of ANS by hull-fouling species could result in a change in dietary composition and quality, having potentially significant impacts.
Freshwater Fish/Inland Sturgeon	Remote	These are generally bottom-oriented, large river fish that prefer sandy and gravel substrates. Fry likely feed on insect larvae, and adult fish feed on fish and some insect larvae. In the event of invasion by hull-fouling species introduced by vessels of the Armed Forces, the composition and quality of the diet could change.

Table 5-10. Potential Magnitude of Impacts for Taxonomic Groups of Federally Listed Species if Exposed to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Taxonomic Group	Highest Risk of Impact from ANS	Presumptions
Anadromous Sturgeon	Potentially Significant	Sturgeon spend most of their lives in coastal marine waters, estuaries, and the lower reaches of large rivers, ascending rivers to spawn. Adults feed on bottom invertebrates and small fish, and introduction and invasion by hull-fouling species could result in a change in dietary composition.
Estuarine/Marine Fish	Potentially Significant	Fish species that inhabit nearshore coastal waters for all or part of their life cycle could be impacted by hull-fouling species invasions if it leads to changes in dietary composition or habitat. However, these species are generally more motile and can change location to find alternative habitat and food resources if they are unable to adapt to the changes.
Beetle and Aquatic/Aquatic Dependent Insects	Remote	Federally listed beetles and aquatic or aquatic dependent insects occur in vernal pools, small streams, or wetland areas where vessels of the Armed Forces do not operate.
Amphibian	Remote	Amphibian species occur in quiet freshwater habitats where vessels of the Armed Forces are very unlikely to operate.
Snakes and Other Reptiles	Remote	Some reptile species are highly aquatic, and introductions of ANS could lead to increased predatory pressure or changes in dietary composition.
Sea Turtles	Potentially Significant	ANS invasions are more likely to occur in areas inhabited by sea turtles. Some hull-fouling species, such as algae or sessile or colonial invertebrates, could overgrow coral reefs or outcompete corals and seagrass beds that provide habitat and food for sea turtles, having major consequences for turtle populations.
Coastal/Marine Birds	Negligible	Aquatic-dependent birds that are most likely to be impacted by ANS are those that are highly dependent on nearshore coastal and floodplain habitats. Invasions by hull-fouling organisms could lead to a change in community composition, particularly invertebrate species, and a change in the quality of available prey. However, changes in dietary composition are expected to have only minor impacts, if any, unless a highly important prey item is eliminated.
Marine Mammals	Negligible	Most marine mammals occur and feed primarily offshore, and hull-fouling ANS invasions would not be expected to impact marine mammal habitat quality or food resources. However, ANS could have moderate consequences for marine mammals that feed in coastal waters, such as the manatee.
Terrestrial Mammals	Remote	Although some terrestrial mammals are dependent on aquatic habitats, their diet and habitat is sufficiently variable that they would not be impacted by changes brought on by the unlikely introduction of hull-fouling ANS by vessels of the Armed Forces.

Table 5-10. Potential Magnitude of Impacts for Taxonomic Groups of Federally Listed Species if Exposed to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Taxonomic Group	Highest Risk of Impact from ANS	Presumptions
Seagrasses	Potentially Significant	Seagrasses could be outcompeted by the invasion of hull-fouling algae or experience increased grazing pressure by sessile invertebrates.
Freshwater and Saltwater Wetland and Aquatic Plants	Negligible	Some fouling species could cause a change in water quality, compete with emergent vegetation for space.

5.2.1.8 Assessment of Risk to Critical Habitat from Aquatic Nuisance Species Invasion Resulting from In-water Cleaning of Vessels of the Armed Forces

Equally as important as assessing potential impacts to listed species is assessing potential impacts to critical habitat. Critical habitat refers to the specific areas, both within and outside of the geographic area, occupied by a threatened or endangered species, including physical and biological features that are essential to the conservation of the species (ESA section 3). Essential features of critical habitat, or primary constituent elements (PCEs), are specific to the requirements for survival of each listed species. Essential critical habitat features could include food and prey, but also include habitat conditions (e.g., habitat and water quality, substrate) that may also be affected by invasion of hull-fouling ANS on vessels of the Armed Forces. Table 5-11 summarizes the results of the assessment of risk to critical habitat in the RAAs from ANS invasion.

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-fouling ANS	Risk of Impact
Coral, Elkhorn *Acropora palmata* Coral, Staghorn *Acropora cervicornis*	Miami, FL	• 1,329 square miles offshore of Palm Bach in Broward, Miami-Dade, and Monroe Counties, FL • Substrate of suitable quality and availability to support larval settlement and recruitment and reattachment and recruitment of asexual fragments • Suitable substrate defined as natural consolidated hard substrate or dead coral skeleton that is free from fleshy or turf macroalgae cover and sediment cover	Unlikely	Major • Potential overgrowth of reefs by algae or tunicates • Competition for suitable substrate	**Potentially Significant Risk** Introduction of hull-fouling algae or sessile invertebrates to coral critical habitat by in-water cleaning of vessels of the Armed Forces is unlikely because cleanings will not be performed near sensitive areas. Further, vessels of the Armed Forces are inspected and, if necessary, cleaned prior to deployment, refreshing their AFC and reducing the likelihood of fouling by NAS while deployed. In addition, NAS on the hulls of vessels are likely to be damaged during in-water cleaning and unlikely to invade (i.e., become ANS). However, ANS invasion could have major consequences for corals; therefore, risk to critical habitat is potentially significant
Crocodile, American *Crocodylus acutus*	Miami, FL	• All land and water (excluding structures) along the Florida coast south of Turkey Point, Biscayne Bay on the east coast and the northernmost point of Nine Mile Pond on the west coast • No PCEs identified, however, greatest threats are the availability of suitable nesting sites and interaction with humans in nesting areas, which could result in nest abandonment	No Exposure	Undetectable	**No Risk** Critical habitat for the American crocodile is terrestrial and will not be impacted by invasions of any hull-fouling species that may be introduced by in-water cleaning performed on vessels of the Armed Forces

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-fouling ANS	Risk of Impact
Manatee, West Indian *Trichechus manatus*	Miami, FL	• Coastal areas south of Jacksonville on the east coast of Florida and south of Tampa on the west coast • No PCEs identified, however, greatest threats are habitat loss and degradation, and mortality from boat collisions, hunting, fishing, red tide poisoning, entrapment in water control structures, entanglement in fishing gear, and exposure to cold temperatures.	Unlikely	Major • Introduction of toxic algae could lead to increased mortality • Increased grazing pressure could lead to a reduction in food resources	**Potentially Significant Risk** Hull-fouling ANS invasion of West Indian manatee critical habitat could have major consequences for manatee populations. However, vessels of the Armed Forces are inspected and, if necessary, cleaned prior to deployment, refreshing their AFC and reducing the likelihood of fouling by NAS while deployed. In addition, NAS on the hulls of vessels are likely to be damaged during in-water cleaning and unlikely to invade (i.e., become ANS).

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-fouling ANS	Risk of Impact
Seagrass, Johnson's *Halophila johnsonii*	Miami, FL	• Critical habitat includes areas where persistent flowering populations occur • No specific PCEs identified, however, the greatest threats are destruction from dredge and fill, turbidity, eutrophication, and thermal pollution due to high population pressure along this segment of the coast	Unlikely	Major • Some ANS may increase sediment suspension and water turbidity that leads to a reduction in seagrass coverage • Introductions could lead to increased grazing pressure	**Potentially Significant Risk** Vessels of the Armed Forces are inspected and, if necessary, cleaned prior to deployment, refreshing their AFC and reducing the likelihood of fouling by NAS while deployed. In addition, NAS on the hulls of vessels are likely to be damaged during in-water cleaning and unlikely to invade (i.e., become ANS). Therefore, introduction of hull-fouling ANS by in-water hull cleaning is unlikely. However, some ANS could have major direct impacts to Johnson's seagrass bed critical habitat if they occur; therefore, risk is potentially significant.

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-fouling ANS	Risk of Impact
Sturgeon, Atlantic *Acipenser oxyrinchus oxyrinchus*		• Hard bottom substrate (e.g., rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (i.e., 0.0–0.5 parts per thousand range) for settlement of fertilized eggs, refuge, growth, and development of early life stages; • Aquatic habitat with a gradual downstream salinity gradient of 0.5 up to as high as 30 parts per thousand and soft substrate (e.g., sand, mud) between the river mouth and spawning sites for juvenile foraging and physiological development; • Water of appropriate depth and absent physical barriers to passage (e.g., locks, dams, thermal plumes, turbidity, sound, reservoirs, gear, etc.) between the river mouth and spawning sites necessary to support: o Unimpeded movement of adults to and from spawning sites; o Seasonal and physiologically dependent movement of juvenile Atlantic sturgeon to appropriate salinity zones within the river estuary; and o Staging, resting, or holding of subadults or spawning condition adults.			

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-fouling ANS	Risk of Impact
Sturgeon, Atlantic *Acipenser oxyrinchus oxyrinchus*	Norfolk, VA	Water depths in main river channels must also be deep enough (e.g., at least 1.2 meters) to ensure continuous flow in the main channel at all times when any sturgeon life stage would be in the river; • Water, between the river mouth and spawning sites, especially in the bottom meter of the water column, with the temperature, salinity, and oxygen values that, combined, support: o Spawning; o Annual and inter-annual adult, subadult, larval, and juvenile survival; and o Larval, juvenile, and subadult growth, development, and recruitment (e.g., 13 to 26 °C for spawning habitat and no more than 30 °C for juvenile rearing habitat, and 6 milligrams per liter (mg/L) or greater dissolved oxygen for juvenile rearing habitat). • Does not include areas owned or controlled by: o The DoD, or designated for its use, that are subject to an integrated natural resource management plan	Unlikely	Moderate: Hull-fouling ANS could impact juvenile food resources	**Negligible Risk** It is unlikely that ANS will be introduced to Atlantic Sturgeon critical habitat by in-water cleaning of vessels of the Armed Forces because of vessel management practices; should an ANS invasion occur, there could be impacts to juvenile Atlantic sturgeon prey; however, because of the broad feeding range of this species, impacts to the prey based from hull-fouling ANS are expected to be minor; therefore, risk to Atlantic sturgeon critical habitat is remote

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
Seal, Hawaiian monk *Monachus schauinslandi*	Pearl Harbor, HI	• Northwestern Hawaiian Islands: Includes all beach areas, sand spits and islets, including all beach crest vegetation to its deepest extent inland, lagoon waters, inner reef waters, and including marine habitat through the water's edge, including the seafloor and all subsurface waters and marine habitat within 10 m of the seafloor, out to the 200-m depth contour line • Main Hawaiian Islands: Defined in the marine environment by a seaward boundary that extends from the 200-m depth contour line (relative to mean lower low water), including the seafloor and all subsurface waters and marine habitat within 10 m of the seafloor, through the water's edge into the terrestrial environment where the inland boundary extends 5 m (in length) from the shoreline • Terrestrial areas and adjacent shallow, sheltered aquatic areas with characteristics preferred by monk seals for pupping and nursing • Marine areas from 0 to 200 m in depth that support adequate prey quality and quantity for juvenile and adult monk seal foraging. Inshore, benthic and offshore teleosts, cephalopods, and crustaceans are commonly described as monk seal prey items.	Unlikely	Minor • ANS could impact inshore and benthic prey items through competition for space and food resources	**Remote Risk** ANS are unlikely to be introduced to Hawaiian monk seal critical habitat because most of the critical habitat occurs either onshore, along exposed coasts, or in deeper offshore waters where in-water hull cleaning is not likely to be performed. Further, vessels of the Armed Forces are inspected and, if necessary, cleaned prior to deployment, refreshing their AFC and reducing the likelihood of fouling by NAS while deployed. In addition, NAS on the hulls of vessels are likely to be damaged during in-water cleaning and unlikely to invade (i.e., become ANS). Therefore, introduction of hull-fouling ANS by in-water hull cleaning is unlikely. However, if hull-fouling ANS are introduced, they may compete with some inshore and nearshore benthic invertebrate species, but reductions in prey for Hawaiian monk seal are expected to be minor and not have a significant impact on monk seal critical habitat

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
		• Significant areas used by monk seals for hauling out, resting or molting			
Bocaccio *Sebastes paucispinis*	Puget Sound, Seattle, WA	• Approximately 590.4 square miles of nearshore habitat and 414.1 square miles of deepwater habitat in Washington state • For adults, benthic habitats or sites deeper than 30 m (98ft) that possess or are adjacent to areas of complex bathymetry consisting of rock and or highly rugose habitat are essential to conservation because these features support growth, survival, reproduction, and feeding opportunities by providing the structure for rockfishes to avoid predation, seek food and persist for decades. • For juveniles, settlement habitats located in the nearshore with substrates such as sand, rock and/or cobble that also support kelp are essential for conservation because these features enable forage opportunities and refuge from predators and enable behavioral and physiological changes needed for juveniles to occupy deeper adult habitats.	Unlikely	Major • Hull-fouling ANS could increase grazing pressure on or outcompete kelp, leading to a reduction in habitat for juvenile bocaccio	**Potentially Significant Risk** It is unlikely that ANS will be introduced to bocaccio critical habitat by in-water cleaning of vessels of the Armed Forces because cleaning practices are designed to reduce hull-fouling. Vessels of the Armed Forces are inspected and, if necessary, cleaned prior to deployment, refreshing their AFC and reducing the likelihood of fouling by NAS while deployed. In addition, NAS on the hulls of vessels are likely to be damaged during in-water cleaning and unlikely to invade (i.e., become ANS). However, should an ANS invasion occur, there is the potential for major impacts to juvenile bocaccio critical habitat by altering substrate and kelp beds.

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
Murrelet, Marbled *Brachyramphus marmoratus*	Puget Sound, Seattle, WA San Francisco, CA	• Areas designated in Washington, Oregon, and central and northern California • Coastal hemlock/tanoak habitat • Individual trees with potential nesting platforms • Forested areas within 0.8 kilometers (0.5 miles) of individual trees with potential nesting platforms, and with a canopy height of at least one-half the site-potential tree height. This includes all such forest, regardless of contiguity.	Unlikely - Likely	Undetectable	**Remote Risk** Marbled murrelet critical habitat is terrestrial; therefore, no impacts from hull-fouling ANS are expected.
	Puget Sound, Seattle, WA	• 438.5 sq. mi. (1,135.7 sq. km) of deepwater critical habitat within the Whidbey Basin, Main Basin, South Puget Sound, and Hood Canal	Unlikely	Major • ANS that are filter feeders have the potential to improve water quality	**Potentially Significant Risk** Introduction of hull-fouling ANS to rockfish critical habitat by in-water cleaning of vessels of the Armed Forces is unlikely because of hull cleaning practices. Vessels of the Armed Forces are inspected and, if necessary, cleaned

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
Rockfish, Yelloweye *Sebastes ruberrimus*	Puget Sound, Seattle, WA	• Appropriate quantity and quality of prey species available to support individual survival, growth, and reproduction • Sufficient levels of dissolved oxygen and water quality to support individual survival, growth, reproduction, and feeding opportunities • Sufficient type and amount of structure and substrate complexity to support predator aversion and feeding opportunities		• Some ANS could compete with rockfish prey species, causing a shift in dietary composition and, potentially, quality • Introduction of algae could lead to algal blooms and, ultimately, decreased oxygen levels	prior to deployment, refreshing their AFC and reducing the likelihood of fouling by NAS while deployed. In addition, NAS on the hulls of vessels are likely to be damaged during in-water cleaning and unlikely to invade (i.e., become ANS).However, because these are demersal fish, should hull-fouling ANS be introduced to rockfish critical habitat, there could be major consequences for rockfish populations from a shift in the prey base. Therefore, risk from ANS introduced by vessels of the Armed Forces is potentially significant

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
Salmon, Chinook, All ESUs *Oncorhynchus tshawytscha*	Puget Sound, Seattle, WA San Francisco, CA	• Areas of varying size within each ESU's location • Freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation and larval development • Freshwater rearing sites with sufficient water quantity and floodplain connectivity, water quality, and natural cover • Freshwater migration corridors free of obstruction and excessive predation with water quantity and quality conditions and natural cover • Estuarine and/or nearshore marine areas free of obstruction and excessive predation with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; natural cover, and juvenile and adult prey (invertebrates and fish)	Unlikely	Moderate • Hull-fouling ANS could alter dietary composition by competing with prey species	**Negligible Risk** Hull-fouling ANS will not be introduced to the freshwater elements of salmon critical habitat by in-water hull cleaning. It is also unlikely that hull-fouling ANS will be introduced to the estuarine and marine components of salmon critical habitat. Vessels of the Armed Forces are inspected and, if necessary, cleaned prior to deployment, refreshing their AFC and reducing the likelihood of fouling by NAS while deployed. In addition, NAS on the hulls of vessels are likely to be damaged during in-water cleaning and unlikely to invade (i.e., become ANS). Should an ANS invasion occur, there could be moderate impacts to nearshore prey. However, risk is considered to be negligible.
Sea turtle, Leatherback *Dermochelys coriacea*	Puget Sound, Seattle, WA San Francisco, CA	• Areas along the west coast of the U.S. from the northernmost point on the Washington coast to Cape Blanco, WA and from Point Arena, CA to Point Arguello, CA • Occurrence of prey species, primarily Scyphomedusae (sea jellies) of the order Semaeostomeae, of sufficient condition, distribution, diversity, abundance and density	Unlikely	Minor • Hull-fouling ANS could affect leatherback sea turtle prey densities	**Remote Risk** It is unlikely that ANS will be introduced to leatherback sea turtle critical habitat by in-water cleaning of vessels of the Armed Forces because critical habitat occurs in coastal areas outside of ports and harbors, and vessel hull cleaning is performed dockside in ports and harbors. In addition,

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
		necessary to support individual as well as population growth, reproduction, and development		through competition and predation, but impacts are expected to be minor	vessels of the Armed Forces are inspected and, if necessary, cleaned prior to deployment, refreshing their AFC and reducing the likelihood of fouling by NAS while deployed. When vessels are cleaned, NAS are likely to be damaged and unlikely to invade (i.e., become ANS). Should an ANS invasion occur, there could be impacts to leatherback turtle prey; however, because of the number of species in the order Semaeostomeae and their predominantly pelagic habitat, impacts to this prey based from hull-fouling ANS are expected to be minor; therefore, risk to leatherback sea turtle critical habitat is remote
Sturgeon, Green, All DPSs *Acipenser medirostris*	Puget Sound, Seattle, WA San Francisco, CA	• Coastal U.S. marine waters within 60 fathoms (110 m) depth from Monterey Bay, CA, north to Cape Flattery, WA, including the Strait of Juan de Fuca, to the U.S. Canadian boundary; the Sacramento-San Joaquin Delta and Suisun, San Pablo, and San Francisco bays in California; the lower Columbia River estuary; and certain coastal bays and estuaries in California (Humboldt Bay), Oregon (Coos Bay, Winchester Bay, Yaquina Bay, and Nehalem Bay), and Washington (Willapa Bay and Grays Harbor)	Likely	Moderate • Because of sturgeon benthic feeding habits, hull-fouling ANS have the potential to impact sturgeon food resources	**Potentially Significant Risk** Hull-fouling ANS will not be introduced to freshwater elements of green sturgeon critical habitat by in-water cleaning because smaller vessels moved between freshwater bodies are removed from the water and cleaned prior to deployment at a new location. Further, any fouling that occurs in marine/estuarine environments will not be transferred to freshwater environments. However, it is likely that ANS will be introduced to green sturgeon estuarine elements of critical habitat by

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
		• Riverine systems with sufficient food resources, substrate type for egg deposition, water flow, water quality, migration corridors, holding pool depth (>5 m), and sediment quality • Estuarine habitats with sufficient food resources, water flow, water quality, migration corridors, sediment quality, and a variety of water depths			cleaning vessels of the Armed Forces. Vessels are inspected and, if necessary, cleaned prior to deployment, refreshing their AFC and reducing the likelihood of fouling by NAS while deployed. When vessels are cleaned, NAS are likely to be damaged and generally considered to be unlikely to invade (i.e., become ANS). However, because of the number of vessels being cleaned in San Francisco Bay and the number of non-indigenous species already present, ANS invasion from in-water cleaning is considered to be likely. Should an ANS invasion occur, ANS have the potential to impact water quality and compete with other species that may be food resources for sturgeon, altering their dietary composition and risk to green sturgeon critical habitat is considered to be potentially significant.
Trout, Bull *Salvelinus confluentus*	Puget Sound, Seattle, WA	• Areas throughout Idaho and Washington • Springs, seeps, groundwater sources, and subsurface water connectivity (hyporheic flows) to contribute to water quality and quantity and provide thermal refugia • Migration habitats with minimal physical, biological, or water quality impediments between spawning, rearing, overwintering,	Unlikely	Moderate • Hull-fouling ANS have the potential to impact bull trout food resources by aggressive	**Negligible Risk** Hull-fouling ANS will not be introduced to freshwater elements of bull trout critical habitat by in-water hull cleaning of vessels of the Armed Forces because smaller vessels of the Armed Forces are removed from the water for transport between water bodies, and hulls are thoroughly cleaned before deployment.

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
		and freshwater and marine foraging habitats and an abundant food base • Complex river, stream, lake, reservoir, and marine shoreline aquatic environments with features such as large wood, side channels, pools, undercut banks and unembedded substrates • Water temperatures ranging from 2 to 15 degrees Celsius (°C) (36 to 59 degrees Fahrenheit (°F)), with adequate thermal refugia • In spawning and rearing areas, substrate of sufficient amount, size, and composition to ensure success of egg and embryo overwinter survival, fry emergence, and young-of-the-year and juvenile survival • Natural hydrography • Sufficient water quality to support reproduction, growth and survival • No or low density of nonnative predators		filter feeding or competition for space, both of which can reduce abundances of invertebrates and larval fish	Introduction of hull-fouling ANS to marine areas of bull trout critical habitat by in-water cleaning is unlikely because vessels are inspected and, if necessary, cleaned prior to deployment, refreshing their AFC and reducing the likelihood of fouling by NAS while deployed. When vessels are cleaned, NAS are likely to be damaged and unlikely to invade (i.e., become ANS). However, should an ANS invasion occur, bull trout food resources could be moderately impacted. But because of the low likelihood of invasion from in-water hull cleaning, risk to bull trout critical habitat is considered to be negligible
Trout, Steelhead, All DPS *Oncorhynchus mykiss*	Puget Sound, Seattle, WA San Francisco, CA	• Counties where this species occurs • Freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation and larval development	Unlikely - Likely	Moderate • Hull-fouling ANS have the potential to impact steelhead food resources by	**Negligible to Potentially Significant Risk** Hull-fouling ANS will not be introduced to freshwater elements of steelhead critical habitat by in-water hull cleaning of vessels of the Armed Forces because smaller vessels of the Armed Forces are removed from the water for transport

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
		• Freshwater rearing sites with sufficient water quantity and floodplain connectivity, water quality, and natural cover • Freshwater migration corridors free of obstruction and excessive predation with water quantity and quality conditions and natural cover • Estuarine and/or nearshore marine areas free of obstruction and excessive predation with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; natural cover, and juvenile and adult prey (invertebrates and fish)		aggressive filter feeding or competition for space, both of which can reduce abundances of invertebrates and larval fish	between water bodies, and hulls are thoroughly cleaned before deployment. Introduction of hull-fouling ANS to marine areas of bull trout critical habitat by in-water cleaning in Puget Sound is unlikely because vessels are inspected and, if necessary, cleaned prior to deployment, refreshing their AFC and reducing the likelihood of fouling by NAS while deployed. When vessels are cleaned, NAS are likely to be damaged and unlikely to invade (i.e., become ANS). ANS invasions in San Francisco Bay are considered to be likely because of the number of vessels being cleaned and the number of invasive species already present. Should an ANS invasion occur, hull-fouling species have the potential to moderately impact steelhead food resources, and risk to steelhead trout critical habitat from ANS invasion is considered to be negligible for Puget Sound and potentially significant for San Francisco Bay.
Whale, Killer (southern Resident) *Orcinus orca*	Puget Sound, Seattle, WA	• 2,560 mi.2 (6,630 km2) of marine habitat that includes Haro Strait and the waters around the San Juan Islands, Puget Sound, and the Strait of Juan de Fuca	Unlikely	Undetectable	**Remote Risk** Introduction of hull-fouling ANS to killer whale critical habitat by in-water cleaning of vessels of the Armed Forces is unlikely because vessels are inspected and, if

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
		• Water quality to support growth and development • Prey species of sufficient quantity, quality and availability to support individual growth, reproduction and development, as well as overall population growth • Passage conditions to allow for migration, resting, and foraging			necessary, cleaned prior to deployment. This refreshes the AFC and reduces the likelihood of fouling by NAS while deployed. When vessels are cleaned, NAS are likely to be damaged and unlikely to invade (i.e., become ANS). Although killer whales can be observed in nearshore coastal waters and sounds, they feed on a variety of prey including marine mammals, seabirds, sea turtles, many species of fish (including sharks and rays) and cephalopods, and hull-fouling ANS are unlikely to affect prey species or any other PCE for killer whale critical habitat; therefore, risk is considered to be remote
Plover, Western snowy *Charadrius nivosus nivosus*	San Diego, CA San Francisco, CA	• 24,527 acres of Sandy beaches, dune systems immediately inland of an active beach face, salt flats, mud flats, seasonally exposed gravel bars, artificial salt ponds and adjoining levees, and dredge spoil sites • Areas that are below heavily vegetated areas or developed areas and above the daily high tides • Shoreline habitat areas for feeding, with no or very sparse vegetation, that are between the annual low tide or low water flow and annual high tide or high-water flow, subject to inundation but not constantly under water, that support small invertebrates, such as	Unlikely - Likely	Minor • Hull-fouling ANS could cause a shift in the composition of available food resources	**Remote to Negligible Risk** In San Diego Bay, hull-fouling ANS are unlikely to be introduced by in-water cleaning of vessels of the Armed Forces. Vessels are inspected and, if necessary, cleaned prior to deployment. This refreshes the AFC and reduces the likelihood of fouling by NAS while deployed. When vessels are cleaned, NAS are likely to be damaged and unlikely to invade (i.e., become ANS). The likelihood of invasion in San Francisco Bay is higher because of the greater number of vessels being cleaned and the

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
		crabs, worms, flies, beetles, spiders, sand hoppers, clams, and ostracods, that are essential food sources • Surf- or water-deposited organic debris, such as seaweed (including kelp and eelgrass) or driftwood located on open substrates that supports and attracts small invertebrates for food, and provides cover or shelter from predators and weather, and assists in avoidance of detection (crypsis) for nests, chicks, and incubating adults • Minimal disturbance from the presence of humans, pets, vehicles, or human-attracted predators, which provide relatively undisturbed areas for individual and population growth and for normal behavior			higher level of invasion, making it likely for invasions to occur. Should an ANS invasion occur, hull-fouling species may cause a shift in prey composition but are unlikely to impact any of the other PCEs for Western snowy plover critical habitat; therefore, risk is considered to range from remote to negligible, depending on location.
Bird's-beak, Soft *Cordylanthus mollis ssp. mollis*	San Francisco, CA	• Areas in Contra Costa, Napa, and Solano Counties in California • Persistent emergent, intertidal, estuarine wetland at or above the mean high-water line • Rarity or absence of plants that naturally die in late spring (winter annuals) • Partially open spring canopy cover at ground level, with many small openings to facilitate seedling germination	Likely	Undetectable	**Remote Risk** Because of the number of in-water hull cleanings of vessels of the Armed Forces that are performed in San Francisco Bay, and because of the higher level of invasion, it is likely that hull-fouling ANS will be introduced to soft bird's-beak critical habitat. However, although soft bird's beak is a wetland species, its critical habitat occurs at or above the mean high water level where hull-fouling species are less likely to survive because of exposure. Therefore, they are unlikely

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
					to have detectable impacts on soft bird's-beak critical habitat.
Abalone, Black *Haliotis cracherodii*	San Francisco, CA San Diego, CA	• California coast between the Del Mar Landing Ecological Reserve to the Palos Verdes Peninsula, as well as on the Farallon Islands, Año Nuevo Island, San Miguel Island, Santa Rosa Island, Santa Cruz Island, Anacapa Island, Santa Barbara Island, and Santa Catalina Island • Rocky intertidal and subtidal habitats from the mean higher high water line to approximately 20 ft. (6 m.) deep • Rocky substrate • Habitat for juvenile settlement • Food resources • Suitable nearshore circulation patterns • Sufficient water quality	Unlikely	Major	**Potentially Significant** Should hull-fouling ANS be introduced to black abalone critical habitat, existing black populations could be displaced through competition for space or food. However, critical habitat occurs in the rocky intertidal and subtidal habitats along exposed coastlines where in-water hull cleaning will not be performed; therefore, it is unlikely that black abalone will be exposed to hull-fouling ANS from in-water hull cleaning. Risk is considered to be potentially significant.

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
Shrimp, San Diego Fairy *Branchinecta sandiegonensis*	San Diego, CA	• Vernal pools with shallow to moderate depths (2 in (5 cm) to 12 in (30 cm)) that hold water for sufficient lengths of time (7 to 60 days) necessary for incubation, maturation, and reproduction of the San Diego fairy shrimp, in all but the driest years • Topographic features characterized by mounds and swales and depressions within a matrix of surrounding uplands that result in complexes of continuously, or intermittently, flowing surface water in the swales connecting the pools, providing for dispersal and promoting hydroperiods of adequate length in the pools (i.e., the vernal pool watershed) • Flat to gently sloping topography, and any soil type with a clay component and/or an impermeable surface or subsurface layer known to support vernal pool habitat (including Carlsbad, Chesterton, Diablo, Huerhuero, Linne, Olivenhain, Placentia, Redding, and Stockpen soils)	No Exposure	Undetectable	**No Risk** Hull-fouling ANS from in-water hull cleaning of vessels of the Armed Forces will not be introduced to or affect any of the PCEs for San Diego fairy shrimp critical habitat.
Frog, California red-legged *Rana draytonii*	San Francisco, CA	• Areas throughout coastal California • Aquatic breeding habitat including natural and manmade ponds, slow-moving streams or pools within streams, and other ephemeral or permanent water bodies that typically	Very Unlikely	Undetectable	**Negligible Risk** California red-legged frog critical habitat occurs in areas where smaller vessels of the Armed Forces could operate but where in-water hull cleanings will not be performed. Smaller vessels that are

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
		become inundated during winter rains and hold water for a minimum of 20 weeks. • Non-breeding aquatic and riparian habitat consisting of shallow (non-lacustrine) freshwater features not suitable as breeding habitat, such as streams, small seeps, and ponds that dry too quickly to support breeding, • Upland areas associated with riparian and aquatic habitat that provide food and shelter sites • Dispersal habitat that connects breeding and non-breeding aquatic habitat that is free of barriers			transported between freshwater bodies will be removed from the water and cleaned prior to deployment. Should hull-fouling ANS be introduced by in-water cleaning, the availability of food and shelter sites could be altered by voracious grazers or filter feeders, having major consequences for frogs. However, risk to the California red-legged frog is considered to be negligible.
Whipsnake, Alameda (=striped racer) *Masticophis lateralis euryxanthus*	San Francisco, CA	• Alameda, Contra Costa, San Joaquin, and Santa Clara counties, California • Scrub/shrub vegetation dominated by low- to medium-stature woody shrubs with a mosaic of open and closed canopy, as characterized by the chamise, chamise-eastwood manzanita, chaparral whitethorn, and interior live oak shrub vegetation series occurring at elevations from sea level to approximately 3,850 feet (1,170 meters). Such scrub/shrub vegetation within these series form a pattern of open and closed canopy used by the Alameda whipsnake for shelter from predators; temperature regulation, because it provides sunny and	No Exposure	Undetectable	**No Risk** Hull-fouling ANS will not be introduced to or impact any of the critical habitat PCEs for the Alameda whipsnake because they are predominantly terrestrial

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
		shady locations; prey-viewing opportunities; and nesting habitat and substrate. These features contribute to support a prey base consisting of western fence lizards and other prey species such as skinks, frogs, snakes, and birds • Woodland or annual grassland vegetation series comprised of one or more of the following: Blue oak, coast live oak, California bay, California buckeye, and California annual grassland vegetation series. This mosaic of vegetation supports a prey base consisting of western fence lizards and other prey species such as skinks, frogs, snakes, and birds, and provides opportunities for: Foraging, by allowing snakes to come in contact with and visualize, track, and capture prey (especially western fence lizards, along with other prey such as skinks, frogs, birds); short and long distance dispersal within, between, or adjacent to areas containing essential features; and contact with other Alameda whipsnakes for mating and reproduction • Lands containing rock outcrops, talus, and small mammal burrows. These areas are used for retreats (shelter), hibernacula, foraging, and dispersal, and provide additional prey population support functions			

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
Goby, Tidewater *Eucyclogobius newberryi*	San Francisco, CA	• 10,003 acres (4,050 ha) in Del Norte, Humboldt, Mendocino, Sonoma, Marin, San Mateo, Santa Cruz, Monterey, San Luis Obispo, Santa Barbara, Ventura, and Los Angeles Counties, California. • Persistent, shallow (in the range of approximately 0.3 to 6.6 ft. (0.1 to 2 m)), still-to-slow-moving lagoons, estuaries, and coastal streams with salinity up to 12 ppt • Sand, silt and mud substrate suitable for constructing burrows for reproduction • Submerged and emergent vegetation that provides protection from predators and high flow events • Presence of a sandbar(s) across the mouth of a lagoon or estuary during the late spring, summer, and fall that closes or partially closes the lagoon or estuary, thereby providing relatively stable water levels and salinity	Unlikely	Major • Herbivorous hull-fouling ANS could potentially graze submerged vegetation that provides refuge for tidewater goby •	**Potentially Significant Risk** Because of the number of vessels of the Armed Forces being cleaned in San Francisco Bay and the level of invasion, this estuary is likely to have additional invasions. However, vessels are inspected and, if necessary, cleaned prior to deployment. This refreshes the AFC and reduces the likelihood of fouling by NAS while deployed. When vessels are cleaned, NAS are likely to be damaged and unlikely to invade (i.e., become ANS). Further, all tidewater goby critical habitat lies outside of the San Francisco Bay estuary, and it is unlikely that any ANS introduced by in-water cleaning will reach these other protected areas. However, hull-fouling ANS introductions could potentially have major consequences for tidewater goby critical habitat; therefore, risk is considered to be potentially significant

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
Goldfields, Contra Costa *Lasthenia conjugens*	San Francisco, CA	• Topographic features characterized by isolated mound and intermound complex within a matrix of surrounding uplands that result in continuously, or intermittently, flowing surface water in the depressional features including swales connecting the pools, providing for dispersal and promoting hydroperiods of adequate length in the pools • Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water or whose soils are saturated for a period long enough to promote germination, flowering, and seed production of predominantly annual native wetland species and typically exclude both native and nonnative upland plant species in all but the driest years. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands	No Exposure	Undetectable	**No Risk** Hull-fouling ANS will not be introduced to or affect any of the PCEs for Contra Costa goldfields critical habitat.

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
Manzanite, Franciscan *Arctostaphylos franciscana*	San Francisco, CA	• Areas on or near bedrock outcrops often associated with ridges of serpentine or greenstone, mixed Franciscan rocks, or soils derived from these parent materials • Areas having soils originating from parent materials identified above in PCE 1 that are thin, have limited nutrient content or availability, or have large concentrations of heavy metals • Areas within a vegetation community consisting of a mosaic of coastal scrub, serpentine maritime chaparral, or serpentine grassland characterized as having a vegetation structure that is open, barren, or sparse with minimal overstory or understory of trees, shrubs, or herbaceous plants, and that contain and exhibit a healthy fungal mycorrhizae component • Areas that are influenced by summer fog, which limits daily and seasonal temperature ranges, provides moisture to limit drought stress, and increases humidity	No Exposure	Undetectable	**No Risk** Hull-fouling ANS will not be introduced to or impact any of the PCEs of Franciscan manzanite critical habitat

Table 5-11. Summary of Risk Conclusions for Federally Listed Species Critical Habitat from Exposure to Hull-Fouling Aquatic Nuisance Species Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Critical Habitat	RAA in Which Species Occurs	Location and Primary Constituent Elements (PCEs)	Likelihood of Hull-fouling ANS Invasions by In-Water Cleaning of Vessels of the Armed Forces	Potential Consequences of Hull-Fouling ANS	Risk of Impact
Smelt, Delta *Hypomesus transpacificus*	San Francisco, CA	• Areas of all water and all submerged lands below ordinary high water and the entire water column bounded by and contained in Suisun Bay (including the contiguous Grizzly and Honker Bays), the length of Montezuma Slough, and the existing contiguous waters contained within the Delta, as defined by section 12220 of the State of California's Water Code of 1969 • Shallow, fresh or slightly brackish backwater sloughs and edgewaters for spawning • Protection of the Sacramento and San Joaquin Rivers and their tributary channels from physical disturbance to ensure that delta smelt larvae are transported from the area where they are hatched to shallow, productive rearing or nursery habitat • Maintenance of the 2 parts per thousand (ppt) isohaline and suitable water quality (low concentrations of pollutants) within the estuary to provide delta smelt larvae and juveniles a shallow, protective, food-rich environment. • Unrestricted access to suitable spawning habitat in a period that may extend from December to July	Likely	Undetectable	**Negligible Risk** Because of the number of vessels of the Armed Forces being cleaned in San Francisco Bay and the level of invasion, this estuary is likely to have additional invasions. However, vessels are inspected and, if necessary, cleaned prior to deployment. This refreshes the AFC and reduces the likelihood of fouling by NAS while deployed. When vessels are cleaned, NAS are likely to be damaged and unlikely to invade (i.e., become ANS). Further, all delta smelt critical habitat in the San Francisco Bay estuary is at or below 2 ppt salinity, and it is unlikely that any marine/estuarine ANS introduced by in-water cleaning will reach and thrive in these other protected areas. Even still, invasion in conservatively considered to be likely, but hull-fouling ANS will not impact any of the PCEs, which are physical and chemical in nature, for delta smelt critical habitat.

5.2.1.9 Conclusions of the Assessment of Risk to Federally Listed Species from Exposure to Aquatic Nuisance Species Introduced by In-water Cleaning of Vessels of the Armed Forces

Estuaries, bays and rivers throughout the U.S. have become highly invaded by NAS and ANS, which are responsible for impacts to approximately 42% of all federally listed threatened and endangered species (Pimental et al., 2000). This is largely the result of changes in the shipping industry. However, the majority of NAS introductions and ANS invasions are from commercial shipping, which makes up the majority of wetted hull surface area entering the U.S. annually. A relatively small percentage of wetted surface area entering U.S. ports and harbors is represented by vessels of the Armed Forces. Furthermore, only a few ports and harbors are home ports to vessels of the Armed Forces that will be deployed to other regions. Only those that are deployed to other regions could serve as vectors for new NAS. The largest home ports from which the majority of vessels of the Armed Forces are deployed include Norfolk, VA, San Diego Bay, CA, Puget Sound, WA, and Pearl Harbor, HI, which are all moderately to highly invaded.

Vessels of the Armed Forces are most likely to experience fouling in their home ports when they are in port for prolonged periods of time. Vessels are typically cleaned prior to deployment, increasing their energy efficiency, reducing the likelihood that hull-fouling organisms will be transported from the home port to other locations, improving the performance of the AFC, and reducing the amount of fouling that occurs while the vessel is deployed. Although NAS may foul vessels of the Armed Forces when they are deployed, hull cleaning practices are expected to reduce the level of fouling and propagule pressure from in-water hull cleanings. These management practices (detailed in Appendix H) reduce the likelihood that in-water cleaning of active vessels of the Armed Forces will serve as vectors for NAS that could become ANS.

When vessels are cleaned in their home port, most of the hull-fouling community will consist of species that occur in the home port because that is where most of the fouling will occur. Any NAS that have fouled the hull are likely to be damaged or destroyed during the cleaning process. Therefore, propagule pressure will be low and, in most cases, it is **unlikely** that NAS will become established in ports and harbors where in-water cleaning of vessels of the Armed Forces is performed and that NAS will subsequently invade, negatively impacting native species (i.e., become ANS). The likelihood of ANS invasion may be higher for ports and harbors from which more vessels of the Armed Forces are deployed and where in-water hull cleaning is performed more frequently. The likelihood of invasion is also higher in ports and harbors that are already more highly invaded by NAS.

The approach to the assessment of risk from ANS provides a methodical, reproducible means of assessing risk to listed species depending on the likelihood for ANS exposure and the potential magnitude of effects, should an exposure occur. In some cases, risk to federally listed species and their critical habitat is **remote** or **negligible** either because of geographic location which limits the likelihood of exposure (either the species does not inhabit a port or harbor where ANS could be introduced OR the location is in a more harsh environment where ANS are less likely to survive), or life history characteristics (the species is not a benthic species that it more likely to be impacted by fouling organisms). In other cases there is **potentially significant risk** to some federally listed species from exposure to ANS introduced during in-water cleaning of vessels of the Armed Forces. However, UNDS helps reduce the likelihood that ANS invasions from underwater ship husbandry will occur by establishing clear practices for in-water cleaning, including inspection and cleaning prior to deployment and an hierarchical preference of cleaning methods where they order of preference is cleaning in dry dock, capture, and no capture. All of these methods are preferred over not cleaning.

At the moment, the EPA and DoD are using the best available options, and the implementation of standards to control these discharges would not increase this risk of ANS introduction. Although vessel maintenance and cleaning practices and UNDS reduce the likelihood that NAS will be introduced to military ports and harbors, they do not eliminate the potential for NAS introductions and ANS invasions to occur, and risk to federally listed species is potentially significant if an invasion does occur. The conclusions of this risk assessment will inform the effects determination for all federally listed species and critical habitat included in the BE.

5.2.2 Qualitative Effects Analysis for Oil and Grease and Total Petroleum Hydrocarbons

TPH, which may include oil and grease, is a term used to describe a broad family of several hundred chemical compounds that originally come from crude oil and is defined as the measurable amount of petroleum-based hydrocarbon in an environmental media. There are several hundred individual hydrocarbons that could be part of a TPH mixture, and the composition of the mixture depends on the source. One reason for this is that crude oil varies in its composition. Crude oils can vary in how much of each chemical they contain, and so can the petroleum products that are made from crude oils.

Quantification of potential adverse effects from exposure to either TPH or O&G is complicated because they represent groups of compounds that may have varying compositions depending on the source and types of their constituents, as well as the analytical methods used for measurement. Because TPH is a measured gross quantity without identification of its constituents, TPH measurement is not a direct indicator of risk to humans or to the environment. The amount of TPH found in a sample is useful as a general indicator of petroleum contamination; however, this TPH measurement or number reveals little about how the particular petroleum hydrocarbons in the sample may affect people, animals, and plants. This qualitative effects analysis for TPH and O&G considers both the potential for exposure and likelihood of negative effects on threatened and endangered species.

O&G are measured as hexane extractable material (HEM), while petroleum hydrocarbons are measured as silica gel treated hexane extractable material (SGT-HEM). Toxic petroleum and gasoline constituents, as well as degradants such as PAHs, benzene, ethylene, toluene, and xylenes, are components of O&G that may be measured by SGT-HEM. The adverse effects from O&G (HEM) and TPH are assessed qualitatively in this section. Because of the variability in composition of the various petroleum hydrocarbons that could be present in the Batch Two discharges, a quantitative assessment of individual hydrocarbons is not presented in this BE.

The behavior of petroleum hydrocarbons in the environment depends on the number and configuration of carbon molecules within their molecular structure. Shorter-chain petroleum hydrocarbons are clear or light-colored liquids that evaporate easily, and more complex and longer-chain petroleum hydrocarbons are thick, dark liquids or semi-solids that do not evaporate. When petroleum and non-petroleum hydrocarbons are released directly to water, the lighter fractions will float in water and form thin surface films, while heavier fractions will accumulate in the sediment at the bottom of the water, which may affect bottom-feeding fish and organisms. Some organisms found in the water (primarily bacteria and fungi) may break down some of the hydrocarbon fractions.

It is possible that vessel discharges may cause a sheen or slick immediately surrounding the vessel of the Armed Forces for a short period of time when O&G concentrations in a discharge are near 15 ppm (mg/L). Studies have shown that evaporation and dissolution are the most important fate mechanisms. Lighter components (gasoline) remain only for several minutes to hours, and most lubricating oil evaporates within 2 days (NRC 2003). A smaller portion of the heavier oil or grease can remain on the surface marine microlayer for longer periods (days) depending on environmental conditions, including physical, chemical, and biological processes (NRC 2003).

Concentrations of TPH in surface vessel bilgewater/OWS effluent have the potential to exceed the UNDS maximum allowable concentration of 15 mg/L. Concentrations of oil and grease, which can be a component of TPH, in deck runoff and graywater also have the potential to exceed the current UNDS standard of 15 mg/L. Graywater may contain O&G from foods.

Both petroleum and non-petroleum oils have similar properties that can harm wildlife when released to the environment (62 FR § 54508, 1997). Potential impacts to higher trophic levels (e.g., marine mammals and birds) could occur from direct exposure to oil or grease, inhalation of toxic fumes, and ingestion of oil or grease, directly or indirectly through food. Primary physical and physiological effects could include skin irritation, inflammation, or tissue necrosis, and chemical burns of skin, eyes, and mucous membranes (IPIECA 2004).

For example, birds and mammals may be poisoned when they inadvertently ingest oil while cleaning themselves or when fur or feathers become oiled. Because oil destroys the insulating properties of fur-bearing mammals and the water repellency of birds' feathers, birds and mammals can die from hypothermia in oil-polluted waters. Birds with oiled feathers can lose the ability to fly, dive for food, or float on the water, which could lead to drowning. Furthermore, birds may become easy prey, starve or dehydrate as they are either unable to fly away from predators or search for food with oil-matted feathers. Ingestion of oiled birds also poses risk to predators.

Potential secondary stressors for wildlife, such as immune and reproductive system responses, physiological stress, declining physical condition, and death, also can occur. Behavioral responses to oil or grease and TPH can include displacement of animals from habitat, disruption of social structure, changes in prey availability and foraging distribution and/or patterns, changes in reproductive behavior/productivity, and changes in movement patterns or migration. Tertiary impacts, such as reproductive failure, also can occur in marine mammals exposed to most hydrocarbon products.

Fish and invertebrates may not be exposed to oil immediately upon release, but they can come into contact with oil if it is mixed into the water column or attaches to floating or submerged vegetation (e.g., floating *Sargassum* mats or submerged eelgrass beds). When adult fish are exposed to oil, the fish may experience reduced growth, enlarged livers, changes in heart and respiration rates, fin erosion, and reproduction impairment. Oil can also adversely affect eggs and larval survival (NOAA, 2014). Multiple studies following the Exxon Valdez oil spill identified several effects in fish embryos from exposure to weathered crude oil, including skeletal and pericardial malformations, genetic damage, mortality, and decreased size and inhibited swimming (Carls et al., 1999; Hose et al., 1996). Effects of petroleum hydrocarbons on corals include decreased abundance, diversity and cover, increased bleaching and infection, tissue swelling, mucus production, and decreased reproduction (Turner and Renegar, 2017).

While sea birds, marine mammals, and turtles can be adversely affected by crude oil and bunker fuels, refined oil products are more commonly the cause of such effects (AMSA, 2008). Most research on the qualitative effects of oil in the marine environment is focused on major oil spills. Very little data are available regarding the chronic effects from numerous small releases of oil from small boat engine wet exhaust discharges from vessels of the Armed Forces in ports or harbors (ABP Research, 1999).

Although it is possible that deck runoff could cause a sheen immediately surrounding a vessel of the Armed Forces for a short period of time, it is uncommon for there to be a persistent sheen. Furthermore, the UNDS for deck runoff requires management practices that will serve to minimize the oil and, to the degree possible, prevent a release that results in a sheen. The likelihood of a listed species coming into direct contact with an oil sheen caused by deck runoff is remote because of the small, localized area covered by the sheen and the tendency of fish and wildlife to move, rather than stay in one location for a long period of time.

Due to regular deck cleaning and other BMPs described by UNDS, including the cleanup of oil and other substances spilled during routine maintenance and the removal of residues on decks prior to deck washdowns, the amount of O&G that could be discharged in deck runoff is reduced as much as possible. In addition, flight deck washdowns are prohibited in port, and weather deck washdowns in port are limited. However, these typically small but frequent and widespread releases contribute to the overall quantity of petroleum that enters the sea (NRC, 2003). The more volatile components of O&G in deck runoff evaporate and dissipate over a short period of time (less than 2 days), whereas the less volatile and more viscous components undergo normal weathering and transport processes, such as spreading, evaporation, natural dispersion, aggregation, biodegradation, and photochemical degradation (NRC, 2003). Because of vessel management practices, the amount of O&G in deck runoff is expected to be small, and UNDS requires the amount of O&G in deck runoff to be less than 15 ppm and not produce a visible sheen. Further, the limited discharge allowed in port and weathering processes limit the amount of O&G that will accumulate in ports and harbors from deck runoff discharged by vessels of the Armed Forces, and O&G is unlikely to accumulate to high concentrations in receiving waters. Therefore, the likelihood of producing a negative effect on a protected species from exposure to O&G in deck runoff is considered to be “negligible”.

All surface vessels greater than 400 gross tons must be equipped with an OWS. Currently, DoD requires that bilgewater/OWS effluent not be discharged in port if the port has the capability to collect and transfer OWS effluent to an onshore facility. According to the IMO Global Integrated Shipping Information System (GISIS) Port Reception Facility Database (PRFD), there are 2,179 port reception facilities throughout the United States. Although all ports where vessels of the Armed Forces operate have facilities that can receive bilgewater, other discharges, and other wastes, facilities may not be immediately available for waste transfer. This could result in unintentional release in an emergency situation, but these situations do not occur frequently. Outside of port, the EPA and DoD propose to require that surface vessels equipped with an OWS must not discharge bilgewater and must only discharge OWS effluent through an oil content monitor. In addition, the discharge of OWS effluent must be minimized (i.e., discharged only when necessary) within one mile of shore, must occur at speeds greater than six knots if the vessel is underway, and must be minimized (i.e., discharged only when necessary) in federally-protected waters. For surface vessels not equipped with an OWS, the EPA and DoD propose to require that bilgewater not be discharged if the vessel has the capability to collect, hold, and transfer to an onshore facility.

The EPA and DoD propose to require that vessels prohibit flight deck washdowns and minimize deck washdowns while in port and in federally-protected-waters. In addition to other performance standards, when feasible, machinery on deck must have coamings or drip pans where necessary to collect any oily discharge that may leak from machinery and prevent spills. The drip pans must be drained to a waste container for proper disposal onshore.

Most vessels of the Armed Forces hold graywater for onshore disposal. In addition, under UNDS the EPA and DoD require that large quantities of cooking oils (e.g., from deep fat fryers), including animal fats and vegetable oils, must not be added to graywater systems. The EPA and DoD further require that the addition of smaller quantities of cooking oils (e.g., from pot and dish rinsing) to the graywater system must be minimized when the vessel is within three miles of shore. The EPA and DoD require that graywater discharges not contain oil in quantities that cause a film or sheen upon or discoloration of the surface of the water or adjoining shorelines; or cause a sludge or emulsion to be deposited beneath the surface of the water or upon adjoining shorelines; or contain an oil content above 15 ppm. Ships operating under the UNDS provisions for graywater would not generate O&G in quantities that would cause it to be a pollutant of concern (i.e., discharge of graywater would not produce a visible sheen and has O&G levels below 15 ppm).

TPH concentrations in ports and harbors resulting from the potential discharge of surface vessel bilgewater/OWS effluent were estimated by multiplying vessel class-specific discharge volume of each vessel by the 15 mg/L standard (see Section 5.1). The total mass loading was determined for each RAA and input to the harbor flushing and tidal prism models to determine the maximum concentration of TPH. The maximum modeled concentration was compared with screening benchmarks developed by the Massachusetts Department of Environmental Protection (MADEP). By dividing TPH into groups of petroleum hydrocarbons that act alike in the in environmental media (called petroleum hydrocarbon fractions), scientists can better understand what happens to them. Each fraction contains many individual compounds. The MADEP has developed sediment screening benchmarks for different hydrocarbon fractions using equilibrium partitioning theory. Both bioaccumulation and toxicity of hydrocarbons increase as the octanol-water partition coefficient (Kow) of the hydrocarbon increases (Battelle, 2007).

The comparison of modeled TPH concentrations with the lowest of the final chronic receiving water values developed by MADEP to estimate sediment toxicity benchmarks shows that concentrations are well below the screening benchmark (Table 5-12). Therefore, because releases of TPH and O&G from vessels of the Armed Forces are being managed to minimize them (i.e., keep them below 15 ppm), and because of the low likelihood that a federally listed species will come into direct contact with TPH or O&G released in any of the discharges, risk to federally listed species is “remote”.

The conclusion of this qualitative assessment is further supported by a comparison of modeled concentrations of O&G in an estuarine and freshwater harbor from deck runoff and graywater (using methods described in 5.1) with risk-based WQC developed by Tong et al. (1999) and results of the toxicity studies used to develop the WQC (Table 5-12). Concentrations were estimated by first determining the total weather deck area and amount of rainfall running off weather deck surfaces for all vessels in each of the RAAs. Measured concentrations in samples collected to represent deck runoff were multiplied by the volume of deck runoff to estimate mass loadings of pollutants to each harbor. The mass loadings were then used in harbor flushing and tidal prism models to determine the maximum residence time for pollutants and estimate maximum harbor concentrations assuming flushing but no degradation. The maximum harbor concentration among all estuarine harbors and the mean harbor concentration estimated for the freshwater harbor were used to compare with WQC.

Modeled concentrations of O&G are well below the Association of Southeast Asian Nations (ASEAN) marine water quality criterion (AMWQC) of 0.14 mg/L proposed for O&G for the protection of aquatic life in the ASEAN marine environment and the National Oceanic and Atmospheric Administration marine screening benchmark of 0.3 mg/L for PAHs. Modeled concentrations are also well below the lowest chronic toxicity LC50s. Although these comparisons are not appropriate for a quantitative risk assessment, they do provide confidence that the conclusion of the qualitative assessment that risk to federally listed species from exposure to O&G in vessel discharges is accurate.

Table 5-12. Comparison of Modeled Oil and Grease Concentrations with Risk-Based Water Quality Criteria

Modeled Harbor Type	Estuarine	Freshwater
Total Petroleum Hydrocarbons (TPH)		
TPH Concentration (µg/L)	1.9E-06 (maximum modeled)	2.7E-09
Lowest MADEP Chronic Toxicity Threshold (µg/L)	5.2	
Oil and Grease (O&G)		
O&G Concentration (µg/L)	0.074 (maximum modeled)	0.000028
ASEAN Marine Water Quality Criterion (µg/L)	140	
TNational Oceanic and Atmospheric Administration SQuiRT Value (µg/L)	NA	300
Lowest Chronic Toxicity Value for Invertebrates (µg/L)[1]	3.9	NA
Lowest Chronic Toxicity Value for Vertebrates (µg/L)[2]	1.54	NA

[1]Early life stage study LC50 for newly hatched Malaysian giant prawn (*Macrobrachium rosenbergii*) as cited in Tong et al., 1999.
[2]Study of juvenile seabass (*Lates calcarifer*) as cited in Tong et al., 1999; LC50 divided by ACR of 15 (MADEP, 2007)
NA = not available for freshwater species

5.2.3 Qualitative Effects Analysis for Biological Oxygen Demand and Chemical Oxygen Demand

BOD is a measure of the oxygen used by microorganisms to oxidize the organic matter present in water. Organic matter in wastewater exerts an oxygen demand in receiving waters, thereby depressing dissolved oxygen concentrations. Low dissolved oxygen levels (hypoxia) can impair animal growth or reproduction, and the complete lack of oxygen (anoxia) will kill aquatic organisms. If there is a large quantity of organic waste in water, there also will be substantial bacteria present working to decompose this waste. In this case, the demand for oxygen will be high, so the BOD level will be high. As the waste is consumed or dispersed through the water, BOD levels will begin to decline.

COD is a measure of the amount of oxygen in water that can be consumed during the decomposition of organic matter and the oxidation of inorganic chemicals such as ammonia and nitrite. Like BOD, if there a high load of organic chemicals or reduced chemicals, oxygen will be consumed. COD is related to BOD; however, BOD only measures the amount of oxygen consumed by microbial oxidation and is most relevant to waters rich in organic matter. COD and BOD do not necessarily measure the same types of oxygen consumption.

Vessel discharges that exert relatively high oxygen demand and the associated potential to depress dissolved oxygen concentrations in receiving waters include bilgewater and graywater.

Underwater hull husbandry discharge can also contribute to oxygen demand, and the amount is dependent upon the age of the AFC and the amount of biofouling. BOD in these discharges can exceed the lowest World Health Organization (WHO) recommended water quality criterion for BOD of 2 mg O_2/L, and COD can exceed the lowest WHO recommended water quality criterion for COD of 3 mg O_2/L. Maximum harbor concentrations were modeled based on measured BOD and COD in bilgewater/OWS effluent and graywater, numbers of vessels of the Armed Forces in each RAA, volume of discharge in port, the calculated mass loading of BOD and COD, and harbor flushing characteristics as explained in Section 5.1 above and Appendix F. Due to discharge standards and management practices, concentrations of BOD and COD are not expected to exceed 0.19 µg O_2/L and 0.28 µg O_2/L (0.00019 and 0.00028 mg/L), respectively, as a result of discharges from vessels of the Armed Forces. These concentrations are each four orders of magnitude lower than the WHO recommended WQC. Therefore, there is “remote” risk, as defined in Table 5-6, that water quality and habitat quality for federally listed species will deteriorate as a result of discharges from vessels of the Armed Forces as regulated under UNDS.

5.2.4 Qualitative Effects Analysis for Pharmaceutical and Personal Care Products

5.2.4.1 Pharmaceutical and Personal Care Products and Their Sources

PPCPs describe a large class of chemical contaminants that can originate from human usage and excretions. There are more than 3,000 different pharmaceutical products, and thousands of personal care products and illicit drugs, that are in currently in use (Richardson et al., 2005). Many of these products have been determined to be endocrine disruptors (EDs) and/or toxic. For example, personal care products such as aftershave, cologne, perfume and antibacterial soaps contain alcohol and are toxic, and pharmaceuticals such as hormones are endocrine disruptors. PPCPs include, but are not limited to:

- Aftershave
- Antibacterial soap
- Antibiotics
- Anti-fungal products
- Antihistamines
- Antimicrobial agents
- Antiperspirant deodorant
- Antiseptics such as iodine, betadine, and alcohol based hand wipes
- Aspirin, acetaminophen, and ibuprofen
- Beta-blockers
- Cosmetics
- Dandruff shampoo
- Decongestants
- Dental care products
- Fire retardants
- Hair styling products
- Hormone treatments (e.g., contraceptives and impotence treatments)
- Insect repellents

- Lipid regulators
- Lotion containing vitamins
- Nicotine patches
- Pain Medications
- Products that contain lidocaine
- Psoriasis and eczema topical treatments
- Stimulants (e.g., caffeine)
- Sunscreen (e.g., oxybenzone)

Humans excrete a combination of intact and metabolized pharmaceuticals, many of which have been discharged to the aquatic environment with little evaluation of possible risks or consequences to humans and the environment. Personal care products enter wastewater and the aquatic environment after regular use during showering or bathing. The environmental fate and effects of many cosmetic ingredients are poorly known, although considerable persistence and bioaccumulation in aquatic organisms have been reported for some compounds (*see* Daughton and Ternes, 1999).

Compounds from PPCPs can be found in municipal wastewater and sewage. On vessels of the Armed Forces, pharmaceutical compounds are most likely to be found in vessel sewage, which is not permitted for discharge within 3 miles of shore and is not subject to UNDS. Personal care products can be expected to be found in graywater sources, albeit at relatively low concentrations. Graywater on vessels of the Armed Forces is water that comes from three main sources: (1) galley or kitchen areas, (2) passenger/crew accommodations, and (3) laundry facilities.

PPCPs, after they are excreted, washed off the human body, or directly disposed to the sewage system, enter surface water mainly through insufficiently treated wastewater effluent. On vessels, pharmaceuticals that are ingested go into the onboard blackwater holding tank, while personal care products applied topically get washed off the body and into the graywater system. Because of the nature of operations performed on vessels of the Armed Forces, use of personal care products is different in nature from other populations (e.g., residential populations) and generally consist of standard toiletries and topical treatments.

5.2.4.2 Fate and Effects of Pharmaceutical and Personal Care Products

The major concerns about the presence of PPCPs in freshwater and coastal aquatic environments where they could accumulate are persistence, bioaccumulation, and toxicity. The physicochemical properties of some PPCPs, means that some are not easily removed by conventional water treatment processes, as demonstrated by their presence in drinking water (Snyder, 2008). Both those PPCPs that are persistent and those that have greater use will have a greater ability to result in environmental effects. The continuous use and release of less degradable PPCPs to the environment makes them “pseudo-persistent”, and it is suggested that these pharmaceuticals have greater potential for environmental persistence than other organic contaminants like pesticides because their source continually replenishes even when acted on by environmental processes such as biodegradation, photodegradation, and particulate sorption (Houtman et al., 2004). Pharmaceuticals and pharmaceutical metabolites have been detected in freshwater and coastal waters at concentrations ranging from 0.01 to 6800 ng/L (Ebele et al., 2017; Prichard and Granek, 2016, Bu et al., 2013).

Some biologically active PPCPs have been shown to accumulate in aquatic organisms. A study conducted by Coogan and others (2007) detected the presence of two widely used antimicrobial agents - triclocarban (TCC) and triclosan (TCS), as well as its metabolite methyl-triclosan (M-TCS), in algae samples collected around a wastewater treatment plant in Texas. This means that aquatic and aquatic-dependent receptors can receive unintentional doses of pharmaceuticals through dietary exposure. Other studies have also demonstrated the bioaccumulation of pharmaceuticals in coastal fish and shellfish (Muir et al., 2017; Moreno-Gonzalez et al., 2016, Ramirez et al., 2009).

The biggest concern about the toxic implications of pharmaceuticals is that they are designed specifically to maximize their biological activity at low doses and to target certain metabolic, enzymatic, or cell-signaling mechanisms. Most pharmaceuticals and a few personal care products are designed to interact with a target in humans and animals, such as a specific receptor, enzyme, or biological process, to deliver the desired therapeutic effect. If these targets are present in organisms in the natural environment, exposure to some PPCPs might elicit effects in those organisms, as well. This mode of action (MoA) concept can be applied to all aquatic biota, which are unintentionally exposed to pharmaceuticals in their natural environment, thus raising the risk of ecotoxicological effects (Fabbri and Franzellitti, 2016).

Although the fate and environmental toxicology of PPCPs is not well understood, several effects cause concern, such as feminization or masculinization by hormones and xenoestrogens, synergistic toxicity from complex mixtures at low concentrations, potential creation of resistant strains in natural bacterial populations, and other potential concerns for human and ecological health. One of the major concerns regarding PPCPs in the aquatic environment is their ability to interfere with the endocrine system and consequently causes adverse health effects in an organism, its progeny or sub-population. EDs include a vast group of chemicals from natural (e.g., mycotoxins and phytoestrogens) and synthetic origin (e.g., diethylstilbesterol and Bisphenol A) in a variety of consumer products (e.g., PPCPs, cleaning products, antimicrobials, food preservatives and phthalates) (Mohapatra, 2016).

As noted by Richardson et al. (2005), “Adverse effects which have been noted on aquatic organisms include: (a) green algae toxicity (ciprofloxacin; Halling-Sørensen et al., 2000); (b) endocrine disruption in fish (ethynylestradiol (EE2) and 4 alkylphenols, Jobling et al., 1996); (c) amphipod population effects and sex-ratio changes (EE2, Watts et al., 2002); (d) inhibition of cytochrome P4501A and other P450 enzymes of gizzard shad liver cells (Levine et al., 1997) and (e) spotted sea trout estrogen receptor antagonist (tamoxifen, Thomas and Smith, 1993).” Studies have shown that concentrations of pharmaceuticals and PPCPs at environmentally relevant concentrations can have adverse effects on a variety of aquatic receptor populations. Some of these studies of PPCPs that may be found in vessel graywater are summarized in Table 5-13.

Table 5-13. Examples of Studies Demonstrating Ecological Effects of Pharmaceutical and Personal Care Products in Populations of Aquatic Organisms

PCCP	Species	Effect	Effect Concentration	Study
Diclofenac	German brown trout	Kidney and gill integrity and selected immune parameters	5-50 mg/L	Hoeger, 2005
Mixture of caffeine, ibuprofen, carbamazepine, and novobiocin	Crab *Carcinus maenas*	Decrease in lysosomal membrane stability	0.1–50 µg/L, 28-day exposures	Aguirre-Martínez et al., 2013
Sunscreen and UV filter exposure	*Millepora complanata*, *Stylophora pistillata*, and *Acropora* sp.	Bleaching	10–100 µg/L, 2 – 48 hours	Donovaro et al., 2008
Propanol	*Gammarus* spp.	Decreases in respiration and ammonium excretion	1000 µg/L, 4 weeks	Oskarsson et al., 2012
Propanol	Brown algae *Fucus vesiculosus*	Decreased chlorophyll fluorescence	100 µg/L	Oskarsson et al., 2012

Research indicates that chronic exposure to levels of PPCPs currently detected in the environment can result in sublethal effects. Subcellular biomarkers are indicators of organismal stress, which can affect reproduction, metabolism, locomotion, behavior, and stress responses in the whole organism with potential effects on other species in the community. However, some response biomarkers do not show a response at all ages or developmental stages of an organism, and for many biomarkers, it is difficult to translate sublethal effects to organismal or ecological effects. Yet PPCP-induced changes at the subcellular and cellular levels may have far-reaching consequences, culminating in effects on populations, communities, and ecosystems (Prichard and Granek, 2016).

5.2.4.3 *Assessment of Environmental Effects of Pharmaceutical and Personal Care Products in Discharges Regulated by Uniform National Discharge Standards*

Most reported instances of detectable concentrations of PPCPs in rivers, streams, lakes, and coastal waterbodies have largely been the result of discharging treated municipal wastewater effluent or combined sewer overflows following heavy rain events. Vessels of the Armed Forces are prohibited from discharging blackwater (sewage) within 3 miles of shore, which is where the highest concentrations of PPCP compounds occur. Therefore, PPCPs from vessels of the Armed Forces in coastal waters come from graywater discharge only, and vessels of the Armed Forces are not contributing as much to the elevated concentrations of PPCPs in coastal areas where pharmaceutical products could reach concentrations that have ecological impacts. Sewage is regulated under Section 312 of the CWA and is not regulated by UNDS. The remainder of this discussion focuses on graywater discharge.

The PPCPs that are expected to occur in graywater from vessels of the Armed Forces include:

- Deodorant
    - Aluminum compounds (e.g., aluminum zirconium trishlorohydrex, aluminum zirconium tetrachlorohydrex)
- Shaving products
- Shampoos and cleansers
- Dental care products
- Skin care products
    - Benzoyl peroxide
    - Salicyclic acid
- Antiseptic and antibiotic treatments
    - Triclosan
    - Triclocarban
    - Alcohol
    - Peroxide
    - Witch hazel
    - Bacitracin
    - Neomycin
    - Polymyxin B
- Recommended sunscreen products
    - Titanium dioxide
    - Zinc oxide
    - Avobenzone
    - Mexoryl SX
- Sunscreen products considered to be harmful
    - Oxybenzone (endocrine disruptor, banned in Hawaii)
    - Octinoxate (hormone disruptor)
    - Retinyl Palmitate (Vitamin A Palmitate) (break down with sun exposure and produce destructive free radicals that are toxic to cells, damage DNA, and may lead to cancer)
    - Homosalate (bioaccumulative and toxic)
    - Octocrylene (bioaccumulative and toxic)
    - Benzophenone-2 and 4-hydroxyoxybenzone (hormone disruptor)
    - Paraben preservatives (hormone disruption, developmental and reproductive toxicity)
- Muscle Pain Relief
    - Methyl salicylate
    - Menthol
    - Camphor
- Antifungal Treatment
    - Miconazole and miconazole nitrate
    - Clotrimazole

    - Terbafine Hydrochloride
- Dandruff shampoo
    - Pyrithione zinc
    - Ketoconozole
    - Selenium sulfide
- Treatments for dermatitis and other skin conditions
    - Vaseline or Aquaphor
    - Hydrocortisone
    - Tazarotene

Unlike municipal wastewater treatment facilities, wastewater systems on vessels of the Armed Forces handle a relatively small population. Crew sizes on vessels with graywater discharge can range from 10 crew on the smaller patrol boats to 6,275 crew on aircraft carriers. If the vessel has a marine sanitation device (MSD), the MSD may not effectively remove personal care products.

Data for PPCPs in graywater from vessels of the Armed Forces are lacking and is a data gap in this assessment. However, some general conclusions can be made based on the likely use of personal care products, standard operating procedures, and graywater discharge standards. The use of personal care products on vessels of the Armed Forces is expected to be limited in comparison to land-based sources, recreational vessels, and other non-recreational vessels. For example, sunscreen is generally only applied to exposed skin by personnel who work on deck, and unlike recreational vessels or cruise ships, crew members on DoD vessels are generally highly covered by clothing and are working in areas that are not exposed to sunlight. Larger vessels of the Armed Forces with larger crews are few in number, and their total contribution to graywater discharge is limited to times when they are deployed. Further, for vessels designed with the capacity to hold graywater, UNDS requires that graywater must not be discharged in federally-protected waters, the Great Lakes, or within one mile of shore. Within one mile of shore, graywater generation is reduced (i.e., no laundry, minimize use of galley, showers and sinks), and in port, graywater is pumped to a pierside facility, not over the side. These practices are normally followed as a matter of routine operation. For vessels that do not have the capacity to hold graywater, UNDS requires that graywater production be minimized to limit the discharge of graywater to federally-protected waters, the Great Lakes, and within one mile of shore where any personal care products could accumulate (e.g., embayments where circulation is restricted) and were sensitive ecological populations (e.g., corals) are present. Therefore, while PPCPs in graywater vessels of the Armed Forces contribute to the environmental impacts from these emerging contaminants, the relative contribution to risk to ecological receptors, including federally listed species, from exposure to personal care products in graywater discharges from vessels of the Armed Forces is expected to be “negligible,” as defined in Table 5-6[59].

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[59] Although the risk levels defined in the risk matrix in Table 5-6 were defined for exposure to ANS, the risk matrix and definitions are general and applicable to all qualitative assessments. The risk levels defined based on likelihood of exposure and magnitude of consequences are for exposure to any stressor.

5.3 Quantitative Effects Analysis

A quantitative effects analysis was performed for metals, toxic and non-conventional pollutants with toxic effects, and nutrients. This analysis was based on comparison of estimated maximum exposure concentrations that could occur in harbors with toxicological effects concentrations and was conducted as a 5-step process.

1. Concentrations of pollutants in receiving water were modeled as described in Section 5.1.3, and concentrations of pollutants in fish and invertebrate tissue are modeled as described in Section 5.1.4.
2. Chronic Toxicity Effects Thresholds (CTETs) were identified for each pollutant of concern for each general taxonomic group (freshwater plants, freshwater invertebrates, freshwater vertebrates, marine plants, marine invertebrates, marine vertebrates, aquatic-dependent birds and aquatic-dependent mammals) for direct exposures, body burdens from multiple exposure routes, and dietary exposures. Surrogate species for each general taxonomic group were those with the lowest CTET.
3. Risk Quotients (RQs) were calculated for each pollutant, general taxonomic group, and exposure, with RQs > 1 indicating a potential for adverse effects.
4. Each general taxonomic group was linked to one of the 19 taxonomic groups of federally listed species, and the highest level of risk for surrogate species within the general taxonomic group was considered to be the level of risk for the taxonomic group of federally listed species.
5. The level of risk for each taxonomic group was extrapolated to the individual listed species within that taxonomic group to draw risk conclusions for each federally listed species.

The details of these steps are described in the following sections. Pollutant-specific risk summaries are provided in Section 5.4.

5.3.1 Effects Data and Estimation of Chronic Toxicity Effect Thresholds and Representative Risk Quotients

CTETs are the pollutant-specific and receptor-specific ecotoxicological effects thresholds that were selected for RQ calculations. To calculate RQs, estimated exposure concentrations (Section 5.1.3) are compared with CTETs to assess the potential for adverse effects to threatened and endangered species (Equation 5-1). To be consistent with other BEs (e.g., the Batch One BE [NAVSEA, 2016], VGP BE [EPA, 2013] and Biological Evaluation of Oregon's Water Quality Criteria for Toxics [EPA, 2008]), data used in the development of EPA's aquatic life ambient water quality criteria (ALC)[60] were the primary source of aquatic toxicity data used to support the risk analysis for this BE (Section 3.2.3). These data have been reviewed following the strict data quality objectives and secondary data acceptance criteria as defined in EPA's *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms*

[60] Documents containing these data can be accessed by clicking on the chemical name in *National Recommended Water Quality Criteria - Aquatic Life Criteria Table*: https://www.epa.gov/wqc/national-recommended-water-quality-criteria-aquatic-life-criteria-table

and their Uses (Stephan et al., 1985), hereafter referred to as the *1985 Guidelines*. The framework provided by the *1985 Guidelines* established a uniform method for deriving protective criteria for aquatic pollutants that is technically rigorous and has been used for more than 30 years.

When data used to derive ALCs were not available for a pollutant of concern (i.e., no criteria document for that pollutant), EPA's ECOTOX database (http://cfpub.epa.gov/ecotox/) was used as a secondary source of toxicity data to fill key information gaps. The USACE Environmental Residue Effects Database (ERED) was used as an additional source of effects data for critical body burdens of pollutants that have the potential to bioaccumulate in the tissues of exposed organisms. The lowest of the available state nutrient WQC were selected as the effects thresholds for nitrogen and phosphorus. Other reports and peer reviewed studies were used as tertiary sources to fill any remaining data gaps and support risk conclusions. When these sources of information were used, the information was reviewed following the same data quality and acceptance criteria established by the *1985 Guidelines* in order to assure consistency and integrity of the effects data used for assessments.

Consistent with the *1985 Guidelines*, eco-toxicological effects data used for this BE were based on the lowest NOEC and/or lowest observed effects concentration (LOEC) for the assessment endpoints that are most relevant to successful propagation of populations. These assessment endpoints include survival, growth, or reproduction of aquatic species. The lowest (i.e., most conservative) CTET values from a long-term chronic test (water-only and/or dietary exposure) with a representative surrogate species (or closest related taxon of that species) was preferentially selected to calculate RQs in support of the quantitative effects risk analysis. Using the lowest NOECs or LOECs reduced the likelihood that risks to protected species populations would be overlooked. When both a NOEC and LOEC were available, the CTET was calculated as the geometric mean of the two values.

When only acute (lethal, short-term) toxicity data were available for a species, an acute-to-chronic ratio (ACR) was used to estimate a CTET. This is a standard EPA procedure used when insufficient aquatic chronic toxicity data exists (Stephan et al., 1985). The ACR is calculated from paired acute and chronic toxicity data from toxicity tests with a particular pollutant to specific test organism where the concentrations are in the same units. The toxic MoA is not necessarily the same for the expression of acute and chronic toxicity.

The bioavailability of some metals in freshwater is influenced by hardness, measured as calcium carbonate ($CaCO_3$), as well as other factors. For that reason, the calculation of freshwater criteria is dependent on hardness. To account for any difference that exists between the toxicity of a pollutant in a laboratory dilution water and its toxicity in a site water, freshwater CTETs were adjusted to a hardness of 100 mg/L $CaCO_3$. In some cases, this adjustment results in a lower CTET.

When appropriate toxicity test-specific data or ACR-derived chronic aquatic data were not available for a given taxonomic group of interest for a specific pollutant, if a draft or final chronic criterion value (i.e., criterion continuous concentration [CCC]) was available from EPA, then the chronic criterion value was used to calculate the RQ. For example, no toxicity data were identified for marine/estuarine plants for exposure to chlorine produced oxidants (CPO); therefore, a CCC for CPO was used for the risk analysis. All CTETs used to calculate the RQs (and their sources or bases for derivation) are provided in the pollutant-specific tables included in support of the pollutant-specific risk summaries provided in Section 5.3 below.

Critical body burdens (CBRs) were used to calculate RQs for listed aquatic species exposed to pollutants by multiple exposure routes (absorption, respiration, and ingestion). Aquatic species can bioaccumulate pollutants when they uptake pollutants more rapidly than they excrete, eliminate, or metabolize them. CBRs represent a body burden threshold above which adverse effects may be observed. The primary source of CBRs for this BE is the USACE ERED.

Dose-based CTETs were used to calculate RQs for listed aquatic-dependent species (birds and mammals) exposed to pollutants through the ingestion of contaminated food and drinking water (freshwater only). All aquatic-dependent wildlife CTETs are based on chronic (NOAELs from laboratory exposure of surrogate birds and mammals to oral ingestion of contaminated food and drinking water. Only NOAELs based on growth or reproduction are used as CTETs in this BE because these are more sensitive indicators than survival and can be directly linked to population-level effects. The CTETs are largely the same as what was used to support similar analyses in the EPA's *Biological Evaluation of Oregon's Water Quality Criteria for Toxics*, dated January 2008 (EPA, 2008). The primary source for these values was Appendix A of Sample et al. (1996), *Toxicological Benchmarks for Wildlife: 1996 Revision*; however, dietary exposure data used by the EPA to develop Ecological Soil Screening Levels (EcoSSLs) also were used to fill data gaps. Most of these data are based on more common bird and mammal species (i.e., domestic chickens, mallard ducks, mice, and rats).

All available CTETs and their sources are presented in Appendix G. Because the process of selecting CTETs for the risk analysis may overlook toxicity studies with species that are more taxonomically similar to federally listed species, the conclusions of the risk analysis for each taxonomic group of federally listed species were checked against other studies identified for species that are more similar to federally listed species. Risk conclusions were also checked against conclusions drawn from using a CCC adjusted by a correction factor to account for ESA-listed species vulnerability (Section 5.6.4).

5.3.2 Direct Effects Analysis

RQs, calculated using Equation 5-1, were used to determine the potential for direct effects to listed aquatic species from exposure to pollutants in water only or from bioaccumulation of pollutants by uptake from water and prey. Risk quotients also were used to determine the potential for direct effects to aquatic-dependent species from dietary exposure to pollutants in food and drinking water. Upper bound pollutant mass loading rates for various combinations of vessel class and discharge type and various harbor modeling scenarios were used to derive the reasonable maximum potential EC for each Batch Two pollutant. The EPA and DoD assume that by using the highest EC modeled for any given pollutant in the hypothetical "worst case" harbor modeling scenarios (i.e., highest number of vessels of the Armed Forces and lowest flushing rates) see (Section 5. 1), the effects analysis results can be applied to the larger action area of interest in this BE. The harbor models were designed to represent various environmental conditions and pollutant loading scenarios to help determine a range of potential pollutant ECs for numerous areas. Therefore, "worst case" of the modeled exposure scenarios are represented by the highest of the reasonable maximum ECs estimated for any given pollutant in the harbor modeling scenarios. The maximum modeled exposure concentrations were then quantitatively compared with lowest CTET values for each pollutant of concern to estimate the risk for adverse effects to listed species populations as an RQ.

Under this risk model, a higher RQ indicates a higher risk to a species population from exposure to a pollutant. It is important to recognize, however, that the RQ is only a measure of potential for adverse effects and not a measure of actual effect. Different taxa respond differently to any particular pollutant exposure based on the differences in their chronic sensitivity to any given pollutant. For this BE, a sliding scale was developed to characterize risk based on calculated RQ values and to determine overall ecological health concerns from any given pollutant. The RQ scale is presented in Table 5-14. An RQ less than one indicates that the risk to that particular species is either "remote" ($RQ < 0.1$) or "negligible" ($0.1 \leq RQ < 1.0$). An RQ equal to or greater than one but less than ten indicates "potentially significant" risk for that species. An RQ equal to or greater than 10 indicates risk is "likely significant" for that species, hence having a more likely potential for adverse effects.

Table 5-14. Risk Characterization Risk Quotients Scale and Level of Health Concern Determination

RQ Scale	Risk Characterization	Level of Health Concern
$RQ < 0.1$	Remote	Pollutant poses a "remote" risk and is not likely to adversely affect a listed species population.
$0.1 \leq RQ < 1.0$	Negligible	Pollutant poses a "negligible" risk and adverse effects to a listed species population are not likely to be significant.
$1.0 \leq RQ < 10$	Potentially Significant	Pollutant poses a low level of risk and could potentially adversely affect a listed species population.
$RQ \geq 10$	Likely Significant	Pollutant poses a risk to and is likely to adversely affect a listed species population.

5.3.2.1 Basis of Risk Quotients Used to Support Direct Effects Analysis for Aquatic Listed Species from Water-only Exposure

Due to a lack of definitive chronic toxicity data for almost all of the listed aquatic species considered in this BE, CTETs are derived from tests with surrogate species having the most sensitive (i.e., the lowest) effects threshold value for that entire species group. In some cases, the CTET selected was from a pollutant exposure test with a species that is similar to the listed species being evaluated (e.g., rainbow trout, *Oncorhyncus mykiss*, is a closely related surrogate for the listed Distinct Populations Segments (DPSs) of sea-run steelhead trout). In other cases, the CTET selected was for a different surrogate species (e.g., sheepshead minnow) to represent an entire taxonomic group (e.g., freshwater aquatic vertebrates). Surrogate species fell into six general taxonomic groups, with each taxonomic group being linked to one of the 19 taxoonomic groups of federally listed species. Table 5-15 provides a cross-walk between the taxonomic groups for federally listed species, the six general taxonomic groups of federally listed species, and the surrogate species from toxicity studies that resulted in the lowest chronic effects thresholds. Because of differences in sensitivity of organisms to different chemical stressors, the surrogate species selected to represent each of the general taxonomic groups varies for each of the pollutants evaluated. The most sensitive surrogate species identified for each pollutant and general taxonomic group are presented in the CTET tables for each of the pollutant-specific risk summaries in Section 5.4 below.

Table 5-15. Crosswalk for Surrogate Species Type to Listed Species Taxonomic Group

Listed Species Taxonomic Group [a]	Surrogate Species Type	Surrogate Species
Terrestrial Mammal	Mammals (Freshwater)	Norway rat, mink, mouse
Marine Mammal	Mammals (Saltwater)	
Coastal/Marine Bird	Birds (Saltwater)	Mallard, chicken/chick, Japanese quail, American kestrel, mallard, ringed dove
Snake and Other Reptiles	Birds (Freshwater) [b]	
Sea Turtle	Birds (Saltwater) [b]	
Beetle and Aquatic Insects	Freshwater Invertebrate	Amphipod, rotifer, cladoceran, hydra, ramshorn snail, midge, fatmucket
Unionid Mussel	Freshwater Invertebrate	
Freshwater Shrimp/Crustacean	Freshwater Invertebrate	
Amphibian	Freshwater Vertebrate	Rainbow trout, Atlantic salmon, chinook salmon, fountain darter, fathead minnow, brown trout, brook trout, Japanese medaka, guppy, bluegill, flagfish, zebra fish
Freshwater Fish/Inland Salmonid	Freshwater Vertebrate	
Freshwater Fish/Inland Sturgeon	Freshwater Vertebrate	
Anadromous Salmonid (Juvenile life-stages only)	Freshwater Vertebrate	
Anadromous Sturgeon (Juvenile life-stages only)	Freshwater Vertebrate	
Anadromous Salmonid (Adults)	Estuarine/Marine Vertebrate	Striped bass, European flounder, Atlantic silverside, mummichog, sheepshead minnow, grey mullet, mudskipper, cabezon, topsmelt, summer flounder, target fish, chinook salmon, steelhead trout, Indian medaka, inland silverside
Anadromous Sturgeon (Adults)	Estuarine/Marine Vertebrate	
Estuarine/Marine Fish	Estuarine/Marine Vertebrate	

Table 5-15. Crosswalk for Surrogate Species Type to Listed Species Taxonomic Group (Continued)

Listed Species Taxonomic Group [a]	Surrogate Species Type	Surrogate Species
Saltwater Corals	Estuarine/Marine Invertebrate	Mysid shrimp, seastar, polychaete worm, flower crab, copepod, rock oyster, blue mussel, cockle, clam, grass shrimp, sea urchin, dog whelk, rock shell, greenlip abalone
Saltwater Mollusk	Estuarine/Marine Invertebrate	
Freshwater-Saltwater Aquatic and Wetland Plants	Freshwater Plant (Vascular) and Saltwater Plant (Vascular)	Duckweed, esthwaite waterweed, fungus, giant kelp, red algae, green algae, diatom, eelgrass
Seagrass	Saltwater plant (Vascular)	Giant kelp, red algae, diatom, eelgrass

a) Listed species taxonomic groupings are meant to roughly coincide with the Species Group provided in USFWS Environmental Conservation Online System (ECOS), but with further delineation according to life history information and attributes to demonstrate the strong representation given by the subset of 111 RAAs listed species to the 674 aquatic and aquatic-dependent species and their critical habitats that may be affected by the Uniform National Discharge Standards Batch Two vessel discharges.
b) No data are readily available to calculate exposure concentrations for reptiles; therefore, EPA used data from birds as the closest related surrogate.

The following calculation supports the quantitative effects analysis for aquatic species from water-only exposure to a pollutant that is applied to listed species:

$$RQ_{A,W} = \frac{EC_W}{CTET_{A,W}}$$ *Equation 5-1*

Where:

$RQ_{A,W}$ = Pollutant- and species-specific risk quotient for aquatic organisms (basis for letter A designation) based on water-only exposure (basis for letter W designation).
EC_W = Modeled pollutant-specific exposure concentration in the receiving water.
$CTET_{A,W}$ = Chronic toxicity effect threshold for a given aquatic species from water-only chronic exposure to a pollutant.

The $CTET_{A,W}$ in this analysis is derived from the lowest chronic value from an acceptable water-only laboratory chronic toxicity test with either mortality, growth, or reproduction as the endpoint. This analysis is used to estimate the effects on aquatic plants, invertebrates and vertebrates, such as fish and amphibians. Using the lowest $CTET_{A,W}$ reduces the likelihood of overlooking the potential for adverse effects to the listed aquatic species considered in this BE. Although some listed species may be more sensitive than the surrogate species tested, evidence has shown that using the lowest CTET will not overlook the potential for adverse effects (EPA, 2008). Mortality, growth, and reproduction are the three endpoints considered to be the most appropriate in a risk assessment context because they reflect population-level effects.

The test acceptability and data quality criteria provided in the *1985 Guidelines* (Stephan et al., 1985) were used to derive CTETs for this BE's analysis. The *1985 Guidelines* require inclusion of the entire life cycle when conducting chronic tests for invertebrate species, but partial life-cycle and early life-stage (ELS) testing protocols are accepted for fish species. ELS testing protocols do not include reproductive endpoints and are not used when life-cycle or partial life-cycle tests showing more sensitive adverse chronic effects are available. In contrast, all tests conducted with aquatic algae and plants are chronic, but only chronic values from tests with aquatic rooted vascular plants (macrophytes) and for which the measured endpoint was biologically significant (e.g., vegetative growth or reproduction) are considered acceptable for ALC and, consequently, for this BE. Only tests with aquatic macrophytes, as opposed to algae and diatoms, are considered to be acceptable because phytoplankton display a rapid recovery rate when exposed to pollutants with toxic effects, which precludes them from risk analysis as defined in this BE. Hence, the focus of the analysis in this BE is on aquatic macrophytes. Phytoplankton (algae and diatoms) and macrophytes (vascular plants) in general are rarely as sensitive as fish and macroinvertebrates to pollutants with toxic effects.

All CTETs for aquatic organisms considered in this BE were derived from laboratory tests conducted under flow-through conditions (or renewal tests for zooplankton, e.g., cladocerans), and test concentrations were measured and maintained throughout the test. In addition, water quality characteristics (i.e., conductivity or salinity, hardness, pH, and temperature) were well defined and measured throughout the test to ensure validity and defensibility of $CTET_{A,W}$. Tests with water quality parameters outside of the defined range are not considered to be acceptable. Although not representative of real world conditions, these studies provide a reasonable estimate of concentrations resulting in effects that are from each individual pollutant alone with minimal influence from other stressors. The uncertainties associated with the selected CTETs are discussed in Section 5.6.

5.3.2.2 Basis of Risk Quotients Used to Support Direct Effects Analysis for Aquatic Listed Species from Multiple Routes of Exposure

Aquatic species can be simultaneously exposed to toxic pollutants through multiple exposure routes (sediment, diet, and water). Studies that expose aquatic species to pollutants in water exclude other potentially important exposure routes that could also result in toxic effects. However, studies of the effects from these other exposure routes (diet and sediment) are less common, and dietary exposure studies do not have well standardized methods.

Whole body pollutant concentrations (tissue residues) in aquatic organisms reflect the integration of multiple routes of exposure that includes both waterborne and dietary pollutant uptake. Numerous investigations have demonstrated that whole body residues are reasonable surrogates for the pollutant concentration at the site of toxic action (Landrum et al., 1991; Cook et al., 1992; McCarty and Mackay, 1993 in EPA 2008 OR Toxics BE); therefore, whole body pollutant residue can be useful when making screening-level risk predictions from pollutant exposures. As such, the potential risk of adverse effects from exposure to a pollutant can be evaluated by comparing the amount of that pollutant bioaccumulated in the tissues of an organism with the tissue concentration that corresponds with the manifestation of adverse chronic toxicity effects such as reduced growth or reproduction.

The following calculation supports the quantitative effects analysis on aquatic listed species from multiple routes of exposure to a pollutant:

$$RQ_{A,M} = \frac{EC_T}{CTET_{A,T}}$$

Equation 5-2

Where:

$RQ_{A,M}$ = Pollutant- and species-specific risk quotient for aquatic organisms (basis for letter A designation) from multiple routes of exposure (basis for letter M designation).

EC_T = Estimated pollutant-specific exposure concentration in whole body tissue (basis for letter T designation; wet weight basis) of an aquatic organism that has accumulated that pollutant from modeled receiving water concentrations.

$CTET_{A,T}$ = Chronic toxicity effect threshold of a pollutant measured in tissue (mg/kg wet wt.) for an aquatic listed (or representative surrogate) species associated with adverse chronic effects on reproduction and growth.

The approach used to evaluate toxicity to aquatic species from multiple concurrent routes of exposure in this BE is based on similar rationale described in EPA's *Biological Evaluation of Oregon's Water Quality Criteria for Toxics*, dated January 2008 (EPA, 2008), hereafter referred to as the Oregon Toxics BE. The analysis for this BE uses the following calculation to estimate whole body tissue pollutant concentrations accumulated from exposure to the modeled maximum concentrations predicted in receiving water:

$$EC_T = (BCF \text{ or } BAF) \times EC_W$$

Equation 5-3

Where:

EC_T = Estimated pollutant-specific exposure concentration in whole body tissue (basis for letter T designation; wet weight basis) of an aquatic organism that has accumulated that pollutant from modeled receiving water concentrations.

BCF = Bioconcentration factor: the ratio (L/kg of wet tissue weight) of the concentration of a pollutant in the tissue of an aquatic organism to its concentration in water in situations where the organism is exposed through the water only (used in this analysis for all metals, except arsenic and selenium, and organic pollutants with low octanol/water partition coefficient (K_{OW}). For example, it is used for benzene which has a low to moderate log Kow value of 2.13.

BAF = Bioaccumulation factor: the ratio (L/kg of fresh tissue) of the concentration of a pollutant in the tissue of an aquatic organism to its concentration in water in situations where the organism is exposed through all routes of exposure (water, diet, sediment) – used for arsenic, selenium and organic pollutants that biomagnify such as some PAHs do.

EC_W = Modeled pollutant-specific exposure concentration in the receiving water.

All whole body tissue pollutant concentrations in this BE are presented as mg of pollutant/kg whole body, wet weight.

BCF and BAF values used to calculate EC_T in this BE are provided in Appendix G. Whole body tissue pollutant concentrations associated with adverse effects on growth or reproduction (i.e., $CTET_{A,T}$) were as provided in the Oregon Toxics BE from literature available in Jarvinen and Ankley (1999), the Environmental Residue Effects Database (ERED, available at: http://el.erdc.usace.army.mil/ered/), and other sources identified in the primary literature (see also, Appendix G). As noted above in Section 5.2.3.1, the lowest observed effect residue on growth or reproduction for each species (or representative surrogate species) for which information is available was used as the basis for the $CTET_{A,T}$.

5.3.2.3 Basis of Risk Quotients Used to Support Direct Effects Analysis for Aquatic-Dependent Listed Species

Reptiles, birds, and mammals that consume significant quantities of aquatic organisms are considered to be aquatic-dependent wildlife species. An oral ingestion exposure concentration (EC_O) is used to evaluate whether aquatic-dependent wildlife is at risk from ingestion of pollutants in drinking water and in aquatic prey. The wildlife EC_O for a given pollutant is based on exposure of wildlife to the maximum modeled concentrations of that pollutant in receiving water (freshwater only) and tissue concentrations estimated using the BAF or BCF (Equation 5-3). Exposures are expressed as the combined oral dose of a pollutant for an aquatic-dependent bird or mammal. Birds were used as the closest related surrogate to reptiles, as exposure parameters and toxicity data for reptiles are very limited. If the calculated EC_O for any aquatic-dependent species is lower than the chronic toxicity effect threshold ($CTET_O$) for a pollutant based on the oral dose administered to a bird or mammal species in a chronic toxicity study, then the modeled pollutant concentration in surface water is not high enough to result in an ingested oral dose sufficient to elicit adverse effects in aquatic-dependent wildlife.

The following calculation supports the quantitative effects analysis on aquatic-dependent listed species from oral ingestion of a pollutant via ingestion of prey and/or drinking water:

$$RQ_{Wild} = \frac{EC_O}{CTET_O} \qquad \textit{Equation 5-4}$$

Where:

RQ_{Wild} = Pollutant- and species-specific risk quotient for aquatic-dependent (wildlife) species from oral ingestion of a pollutant via ingestion of prey and drinking water (fresh water).

EC_O = Combined oral dose (basis for letter O designation) of pollutant ingested by an aquatic-dependent bird or mammal from consuming prey or drinking water from a surface water with the modeled receiving water concentrations.

$CTET_O$ = Chronic toxicity effect threshold (in this case, a No Observed Adverse Effect Level or NOAEL) for a given surrogate aquatic-dependent (wildlife) species based on the measured concentration of that pollutant administered orally (food and/or water) from studies examining chronic effects on growth or reproduction.

In this analysis, the pollutant exposure is based solely on the oral ingestion pathway because pollutant exposure experienced by wildlife through both the dermal and inhalation pathways is “negligible” (Sample et al., 1997). Furthermore, sediment ingestion rates of many wildlife species are low, being generally 2% or less of the total food ingestion rate (Beyer et al., 1994). Thus, for the purposes of this BE, the EPA and DoD assume that the oral ingestion of a pollutant in water by an aquatic-dependent species can be described simply as the sum of its ingestion of the pollutant in drinking water and aquatic prey items at predicted concentrations of that pollutant in a receiving water. If the aquatic-dependent wildlife species under evaluation is known to feed on aquatic biota, it follows that the oral dose of the pollutant from consumption of contaminated prey ($E_{O,T}$) can be estimated from the product of the predicted concentration of that pollutant in water and a BCF or BAF (see Equation 5-3). If the food and water ingestion rates for a wildlife species are known or can be estimated, and assuming the concentration of the pollutant in prey is proportional to its concentration in water as above, it is possible to calculate the exposure of aquatic-dependent wildlife to a pollutant from its consumption of prey and water. That information can then be used to calculate the oral dose exposure concentration (EC_O) that can be compared to $CTET_O$ to assess risk to wildlife from the combined ingestion of contaminated food and water (Sample et al., 1997; EPA, 2008). The EC_O is calculated as:

$$EC_O = (IRD \times EC_W) + (IRF \times EC_T)$$

Equation 5-5

Where:

EC_O = Oral exposure concentrations (in mg/kg/day)
IRD = Ingestion rate of drinking water calculated as the quotient of the water intake rate and body weight of a surrogate species (L/kg fresh body weight/day)
EC_W = Exposure concentration in drinking water (in mg/L)
IRF = Ingestion rate of food calculated as the quotient of the food intake rate and body weight of a surrogate species (kg/kg fresh body weight/day)
EC_T = Exposure concentration in prey tissue calculated in Equation 5-3 above (mg/kg wet tissue)

Although some marine wildlife species (dolphins, seals, sea turtles, and sea otters) ingest sea water, only ingestion of freshwater is considered for this assessment. With a few exceptions (e.g., sea otters and sea turtles), ingestion of sea water by marine wildlife is not a common behavior, and most water needs are met metabolically and by food ingestion while incidental ingestion of sea water helps maintain electrolyte balance (Ortiz, 2001). For those species that are known to ingest seawater, ingestion rates are not known. Furthermore, ingestion of pollutants in drinking water is a much less significant exposure pathway than ingestion of pollutants in prey. As such, water ingestion was included as a pathway for freshwater aquatic and aquatic-dependent wildlife because water ingestion rates are more constant and measurable. However, saltwater ingestion impacts to aquatic and aquatic-dependent marine/estuarine wildlife is not regular and largely has not been measured; therefore, water ingestion by marine wildlife was not considered and is an uncertainty in the assessment.

Again, for the purposes of this BE, the EPA and DoD use wildlife NOAELs as $CTET_O$ values. Dietary exposure calculations (EC_O) for this BE are provided in Appendix G.

5.3.3 *Indirect Effects Analysis*

Indirect effects are defined by ESA as those “that are caused by the proposed action and are later in time, but still are reasonably certain to occur” (50 CFR §402.02). In this case, the potential for indirect effects was evaluated by assessing future changes to listed species habitat (i.e., vegetative cover) and food resources. The proposed action is not only expected to reduce the deterioration of habitat quality over time by reducing the contribution from discharges from vessels of the Armed Forces, it is expected that it may help improve aquatic habitat quality by enforcing discharge standards as other programs are implemented to reduce discharges from other sources. However, for this assessment the EPA and DoD assumed that modeled surface water concentrations would remain constant over time rather than declining, as expected based on the proposed action.

A semi-quantitative approach using the same RQs calculated to assess direct effects to listed species population was used to assess the indirect effects of pollutant exposure to those species. The RQs also indicate potential impact of pollutants on water quality, shelter, prey/forage items, etc., that are vital for the health and well-being of each listed species, and these indirect effects are assumed to be proportional to the direct effects RQs. The indirect effects analysis assumes that the likelihood of effects on a listed species increases with time as the risk of impact on that species’ food source or habitat increases. A pollutant may be assumed to pose “remote” or negligible indirect risk to a listed species if the RQ for direct exposure of a potential prey item is also “remote” or “negligible”. A pollutant may also be assumed to pose “remote” or “negligible” risk of indirect effects if other information used to support the direct effects analysis indicates risk is “remote” or “negligible”.

5.3.4 *Analysis of Effects on Critical Habitat*

As stated by the USFWS, “[t]he purpose of the ESA is to protect and recover imperiled species and the habitats on which they depend” (USFWS, 2013). Critical habitat refers to the specific areas both within and outside of the geographic area occupied by a threatened or endangered species, including physical and biological features that are essential to the conservation of the species (ESA section 3). Essential features of critical habitat, or PCEs, are specific to the requirements for survival of each listed species. Essential critical habitat features could include food and prey, but also includes habitat conditions (e.g., habitat suitability and water quality) that may also be affected by pollutants in discharges from vessels of the Armed Forces. In addition to the evaluation of potential direct effects to listed species populations from chronic direct exposure to pollutants in vessel discharges, the potential for indirect effects to vital ecological characteristics (e.g., availability of prey items, geographic distribution, and preferred habitat) for each of the 111 RAA listed species evaluated also was considered.

The ESA requires that any action undertaken or authorized by a federal agency does not result in the destruction or adverse modification of designated critical habitat. Critical habitat has been identified within each of the seven RAAs (or within a five mile radius of the RAA boundaries) for 26 of the 111 federally listed or proposed listed aquatic and aquatic-dependent species evaluated. Each critical habitat is a defined area that has the essential physical and/or biological features that are necessary for the conservation of a federally listed species.

Effects on critical habitat were evaluated qualitatively, but methodically. The general location of each of the 26 critical habitats within the RAAs was determined, and PCEs were identified. Risk of impact to critical habitat was evaluated by considering whether UNDS Batch Two discharges could affect the PCEs. When specific PCEs were not identified, the greatest threats to the species, and whether Batch Two discharges contribute to those threats, were considered. Risk to critical habitat was evaluated using the same approach used to assess risk of impact from pollutants in discharges from vessels of the Armed Forces based on likelihood of exposure and magnitude of potential adverse effects. When critical habitat PCEs included habitat (e.g., seagrasses) or food resources for which indirect effects have been evaluated quantitatively, the results of the indirect effects assessment were taken into consideration when determining the magnitude of potential consequences. Risk was defined within a matrix of likelihood of exposure to pollutants in vessels of the Armed Forces and potential magnitude of effect and matching the likelihood and magnitude within the matrix shown in Table 5-16.

Table 5-16. Critical Habitat Risk Definitions Based on Likelihood of Exposure and Magnitude of Potential Adverse Effects

Likelihood of Exposure to Pollutants	Magnitude of Potential Consequences from Exposure to Pollutants			
	Undetectable	Minor	Moderate	Major
Very Unlikely	Remote Risk	Remote Risk	Negligible Risk	Negligible Risk
Unlikely	Remote Risk	Remote Risk	Negligible Risk	Potentially Significant Risk
Likely	Remote Risk	Negligible Risk	Potentially Significant Risk	Likely Significant Risk
Very Likely	Remote Risk	Negligible Risk	Potentially Significant Risk	Likely Significant Risk

Risk of effects on critical habitat were determined to be:

- Remote:
  - Exposure to pollutants in discharges from vessels of the Armed Forces is very unlikely and effects on critical habitat PCEs are expected to be undetectable or minor;
  - Exposure to pollutants in discharges from vessels of the Armed Forces is unlikely and effects on critical habitat PCEs are expected to be undetectable or minor;
  - Exposure to pollutants in discharges from vessels of the Armed Forces is likely or very likely but effects on critical habitat PCEs are expected to be undetectable;
- Negligible:
  - Exposure to pollutants in discharges from vessels of the Armed Forces is very unlikely but effects on critical habitat PCEs, if they occur, are expected to be moderate or major;

- Exposure to pollutants in discharges from vessels of the Armed Forces is unlikely but effects on critical habitat PCEs, if they occur, are expected to be moderate;
- Exposure to pollutants in discharges from vessels of the Armed Forces is likely or very likely but effects on critical habitat PCEs are expected to be minor;

- Potentially significant:
  - Exposure to pollutants in discharges from vessels of the Armed Forces is unlikely but effects on critical habitat PCEs, if they occur, are expected to be major;
  - Exposure to pollutants in discharges from vessels of the Armed Forces is likely or very likely and effects on critical habitat PCEs are expected to be moderate;
- Likely significant :
  - Exposure to pollutants in discharges from vessels of the Armed Forces is likely or very likely and effects on critical habitat PCEs are expected to be major.

5.3.5 Risk Conclusions for All Listed Species

The risk conclusions from the analysis inform the effects determinations presented in Section 8 and summarize the potential risk to the 674 aquatic and aquatic-dependent species and 278 designated critical habitats that have the potential to have substantial exposure to discharges from vessels of the Armed Forces. Effects determinations have been made for 51 mammals, 53 birds, 22 reptiles, 30 amphibians, 155 fishes, 30 crustaceans, 1 cephalopod, 35 snails, 92 unionid mussels, 24 insects, 22 corals, and 159 sea grasses and aquatic and wetland plants. Section 3.3 identified the seven RAAs selected for the BE analysis, and Section 3.4.2 identified the 111 aquatic and aquatic-dependent species and their 26 critical habitats present in the RAAs to be considered in the risk analysis described above (Sections 5.2.1 to Section 5.2.4).

For the qualitative and quantitative risk analyses for pollutants selected for detailed evaluation, the EPA and DoD intend to use the species-specific risk conclusions for the 111 RAA listed species and the effects assessment for the 26 RAA critical habitats to inform the effects determinations. The approach is performed by first grouping the 111 RAA listed species into appropriate taxonomic groups (see Table 5-16). For the analysis, the EPA and DoD established 19 listed species taxonomic groups of potentially impacted aquatic and aquatic-dependent species. Next, each of the 111 listed species evaluated in the BE was assigned one of the 19 listed species taxonomic groups (see Appendix E). The listed species taxonomic group serves as the key link between the risk conclusions from the 111 RAA species and effects determinations for the 674 aquatic and aquatic-dependent animal and plant species and their critical habitats that might be exposed to pollutants discharged from vessels of the Armed Forces. Finally, for each pollutant evaluated in the quantitative analysis (Section 5.3), the EPA and DoD present a risk conclusion for each of the listed species taxonomic groups, taking into consideration both direct and indirect effects from the pollutant to all of the RAA listed species within each taxonomic group. Similarly, the effects assessment for the 26 RAA critical habitats provides a link to the effects determinations for the 191 federally listed aquatic and aquatic dependent species with defined critical habitat within the action area. This approach ensures that the potential for adverse effects is not under-estimated because: (1) the lowest identified effects concentrations for each taxonomic group are used to assess exposure risk, (2) the highest level of risk determined for each taxonomic group is assumed to represent risk for all federally listed species in that taxonomic group, (3) continuous exposure to maximum modeled exposure concentrations is assumed, and (4) critical habitat within the RAAs represents all types of essential features that could be impacted by pollutants in the Batch Two discharges.

Table 5-17. Listed Species Taxonomic Groupings Based on Listed Aquatic and Aquatic-Dependent Species within the Seven Representative Action Areas

Major Taxonomic Grouping	Listed Species Taxonomic Group[a]	Listed Species/Taxa	RAA(s)[b]	Potential Exposure Pathway Evaluated
Mammals	Terrestrial Mammal	Salt Marsh Harvest Mouse (*Reithrodontomys raviventris*)	SFBE	Water + Diet
		Florida Bonneted Bat (*Eumops floridanus*)	M	Water + Diet
		Gray Bat (*Myotis grisescens*)	SL; M	Water + Diet
		Indiana Bat (*Myotis sodalist*)	SL	Water + Diet
		Northern Long-eared Bat (*Myotis septentrionalis*)	N; SL	Water + Diet
		Grizzly Bear (*Ursus arctos horribilis*)	PS	Water + Diet
	Marine Mammal	West Indian Manatee (*Trichechus manatus*)	M	Diet Only
		Hawaiian Monk Seal (*Monachus schauinslandi*)	PH	Diet Only
		Southern Sea Otter (*Enhydra lutris nereis*)	SFBE	Diet Only
		Guadalupe Fur Seal (*Arctocephalus townsendi*)	SFBE, SD	Diet Only
		Blue Whale (*Balaenoptera musculus*)	N; SD; M; PH; PS; SFBE	Diet Only
		Fin Whale (*Balaenoptera physalus*)	N; SD; M; PH; PS; SFBE	Diet Only
		Humpback Whale (*Megaptera novaeangliae*)	SD; PS; SFBE	Diet Only

Table 5-17. Listed Species Taxonomic Groupings Based on Listed Aquatic and Aquatic-Dependent Species within the Seven Representative Action Areas (Continued)

Major Taxonomic Grouping	Listed Species Taxonomic Group[a]	Listed Species/Taxa	RAA(s)[b]	Potential Exposure Pathway Evaluated
Mammals		North Atlantic Right Whale (*Eubalaena glacialis*)	N; M	Diet Only
		Sei Whale (*Balaenoptera borealis*)	N; SD; M; PH; PS; SFBE	Diet Only
		Sperm Whale (*Physeter catodon (=macrocephalus)*)	N; SD; M; PH; PS; SFBE	Diet Only
		False Killer Whale (Main Hawaiian Islands Insular DPS) (*Pseudorca crassidens*)	PH	Diet Only
		Killer Whale (Southern Resident) (*Orcinus orca*)	PS	Diet Only
Birds	Coastal/Marine Birds	Everglade Snail Kite (*Rostrhamus sociabilis plumbeus*)	M	Diet Only
		Cape Sable Seaside Sparrow (*Ammodramus maritimus mirabilis*)	M	Diet Only
		Wood Stork (*Mycteria americana*)	M	Diet Only
		Piping Plover (*Charadrius melodus*)	M; N; SL	Diet Only
		Eastern Black Rail (*Laterallus jamaicensis ssp. Jamaicensis*)	M; N; SL	Diet Only
		Red Knot (*Calidris canutus rufa*)	N	Diet Only
		Bermuda Petrel (*Pterodroma cahow*)	N	Diet Only
		Roseate Tern (*Sterna dougallii dougallii*)	N; M	Diet Only
		Oahu elepaio (*Chasiempis ibidis*)	PH	Diet Only
		Hawaiian Coot (*Fulica americana alai*)	PH	Diet Only
		Hawaiian Duck (=koloa) (*Anas wyvilliana*)	PH	Diet Only
		Laysan Duck (*Anas laysanensis*)	PH	Diet Only
		Hawaiian Common Moorhen (*Gallinula chloropus sandvicensis*)	PH	Diet Only
		Hawaiian Dark-rumpled Petrel (*Pterodroma sandwichensis*)	PH	Diet Only
		Newell's Townsend's Shearwater (*Puffinus auricularis newelli*)	PH	Diet Only

Table 5-17. Listed Species Taxonomic Groupings Based on Listed Aquatic and Aquatic-Dependent Species within the Seven Representative Action Areas (Continued)

Major Taxonomic Grouping	Listed Species Taxonomic Group[a]	Listed Species/Taxa	RAA(s)[b]	Potential Exposure Pathway Evaluated
Birds	Coastal/Marine Birds	Hawaiian Stilt (*Himantopus mexicanus knudseni*)	PH	Diet Only
		Short-tailed Albatross (*Phoebastria (=Diomedea) albatrus*)	PH; PS; SD	Diet Only
		Marbled Murrelet (*Brachyramphus marmoratus*)	PS; SD; SFBE	Diet Only
		Southwestern Willow Flycatcher (*Empidonax traillii extimus*)	SD	Diet Only
		Light-footed Clapper Rail (*Rallus longirostris levipes*)	SD	Diet Only
		California Least Tern (*Sterna antillarum browni*)	SD	Diet Only
		Least Bell's Vireo (*Vireo bellii pusillus*)	SD	Diet Only
		Western Snowy Plover (*Charadrius nivosus nivosus*)	SFBE; SD	Diet Only
		Least Tern (*Sterna antillarum*)	SL	Diet Only
Reptiles and Amphibians	Snake and Other Reptiles	American Crocodile (*Crocodylus acutus*)	M	Water + Diet
		Eastern Indigo Snake (*Drymarchon corais couperi*)	M	Water + Diet
		Eastern Massasauga (=rattlesnake) (*Sistrurus catenatus*)	SL	Water + Diet
		Alameda Whipsnake (*Masticophis lateralis euryxanthus*)	SFBE	Water + Diet
		Giant Garter Snake (*Thamnophis gigas*)	SFBE	Water + Diet
	Sea Turtle[61]	Green Sea Turtle (*Chelonia mydas*), Central North Pacific DPS	PH	Diet Only
		Green Sea Turtle (*Chelonia mydas*), East Pacific DPS	PS; SD; SFBE	Diet Only
		Green Sea Turtle (*Chelonia mydas*), North Atlantic DPS	N; M	Diet Only

[61] Sea turtles do drink seawater, however ingestion rates are unknown and this could not be evaluated as an exposure pathway.

Table 5-17. Listed Species Taxonomic Groupings Based on Listed Aquatic and Aquatic-Dependent Species within the Seven Representative Action Areas (Continued)

Major Taxonomic Grouping	Listed Species Taxonomic Group[a]	Listed Species/Taxa	RAA(s)[b]	Potential Exposure Pathway Evaluated
Reptiles and Amphibians	Sea Turtle[62]	Hawksbill Sea Turtle (*Eretmochelys imbricate*)	PH; SD; M; N	Diet Only
		Kemp's Ridley Sea Turtle (*Lepidochelys kempii*)	M; N	Diet Only
		Leatherback Sea Turtle (*Dermochelys coriacea*)	N; PH; PS; SD; SFBE; M	Diet Only
		Loggerhead Sea Turtle (*Caretta caretta*), Northwest Atlantic Ocean DPS	N; M	Diet Only
		Loggerhead Sea Turtle (*Caretta caretta*), North Pacific Ocean DPS	PH; PS; SFBE; SD	Diet Only
		Olive Ridley Sea Turtle (*Lepidochelys olivacea*), (all other areas)	PH; SD; SFBE	Diet Only
		Olive Ridley Sea Turtle (*Lepidochelys olivacea*), (Mexico's Pacific coast breeding colonies)	SD	Diet Only
		California red-legged frog (*Rana draytonii*)	SFBE	Water + Diet
		Oregon Spotted Frog (*Rana pretiosa*)	PS	Water + Diet
Freshwater Aquatic Vertebrates	Amphibians	California Tiger Salamander (*Ambystoma californiense*)	SFBE	Water + Diet
	Freshwater Fish/ Inland Salmonid	Bull Trout (*Salvelinus confluentus*)	PS	Water + Diet
	Freshwater Fish/ Inland Sturgeon	Pallid Sturgeon (*Scaphirhynchus albus*)	SL	Water + Diet
	Anadromous Salmonid	Chinook Salmon (*Oncorhynchus (=Salmo) tshawytscha*), Puget Sound ESU	PS	Water + Diet
		Chinook Salmon (*Oncorhynchus (=Salmo) tshawytscha*), Central Valley ESU	SFBE	Water + Diet
		Chum Salmon (*Oncorhynchus keta), Hood Canal Summer-Run*	PS	Water + Diet

Table 5-17. Listed Species Taxonomic Groupings Based on Listed Aquatic and Aquatic-Dependent Species within the Seven Representative Action Areas (Continued)

Major Taxonomic Grouping	Listed Species Taxonomic Group[a]	Listed Species/Taxa	RAA(s)[b]	Potential Exposure Pathway Evaluated
Freshwater Aquatic Vertebrates	Anadromous Salmonid	Steelhead Trout (*Oncorhynchus (=Salmo) mykiss*), California Central Valley DPS	SFBE	Water + Diet
		Steelhead Trout (*Oncorhynchus (=Salmo) mykiss*), Central California Coast DPS	SFBE	Water + Diet
		Steelhead Trout (*Oncorhynchus (=Salmo) mykiss*), Puget Sound DPS	PS	Water + Diet
		Steelhead Trout (*Oncorhynchus (=Salmo) mykiss*), Southern California DPS	SD	Water + Diet
	Anadromous Sturgeon	Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*), Carolina DPS	N	Water + Diet
		Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*), Chesapeake Bay DPS	N	Water + Diet
		Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*), New York Bight DPS	N	Water + Diet
		Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*), South Atlantic DPS	N	Water + Diet
		Shortnose Sturgeon (*Acipenser brevirostrum*)	N; M	Water + Diet
		North American Green Sturgeon (*Acipenser medirostris*), Southern DPS	PS; SFBE	Water + Diet
Freshwater Aquatic Invertebrates	Unionid Mussels	Spectaclecase (mussel) (*Cumberlandia monodonta*)	SL	Water + Diet
		Scaleshell Mussel (*Leptodea leptodon*)	SL	Water + Diet
	Freshwater Shrimp/ Crustacean	California Freshwater Shrimp (*Syncaris pacifica)*	SFBE	Water + Diet
		Conservancy Fairy Shrimp (*Branchinecta conservation*)	SFBE	Water + Diet
		Vernal Pool Fairy Shrimp (*Branchinecta lynchi*)	SFBE	Water + Diet
		Vernal Pool Tadpole Shrimp (*Lepidurus packardi*)	SFBE	Water + Diet

Table 5-17. Listed Species Taxonomic Groupings Based on Listed Aquatic and Aquatic-Dependent Species within the Seven Representative Action Areas (Continued)

Major Taxonomic Grouping	Listed Species Taxonomic Group[a]	Listed Species/Taxa	RAA(s)[b]	Potential Exposure Pathway Evaluated
Freshwater Aquatic Invertebrates	Freshwater Shrimp/ Crustacean	San Diego Fairy Shrimp (*Branchinecta sandiegonensis*)	SD	Water + Diet
	Insects	Miami Blue Butterfly (*Cyclargus (=Hemiargus) thomasi bethunebakeri*)	M	Water + Diet
		Delta Green Ground Beetle (*Elaphrus viridis*)	SFBE	Water + Diet
		Orangeblack Hawaiian Damselfly (*Megalagrion xanthomelas*)	PH	Water + Diet
		Crimson Hawaiian Damselfly (*Megalagrion leptodemas*)	PH	Water + Diet
		Taylor's Checkerspot (*Euphydryas editha taylori*)	PS	Water + Diet
Estuarine/ Marine Aquatic Vertebrates	Estuarine/ Marine Fish	Nassau Grouper (*Epinephelus striatus*)	M	Water + Diet
		Gulf Grouper (*Mycteroperca jordani*)	M	Water + Diet
		Giant Manta Ray (*Manta birostris*)	M; N; PH; SD	Water + Diet
		Smalltooth Sawfish (*Pristis pectinate*), US portion of range	M	Water + Diet
		Scalloped Hammerhead Shark (*Sphyrna lewini*), Central and Southwest Atlantic DPS	N; M	Water + Diet
		Scalloped Hammerhead Shark (*Sphyrna lewini*), Eastern Pacific DPS	PH; SD; SFBE	Water + Diet
		Yelloweye Rockfish (*Sebastes ruberrimus*)	PS	Water + Diet
		Bocaccio (*Sebastes paucispinis*)	PS; SD	Water + Diet
		Eulachon (*Thaleichthys pacificus*)	PS; SFBE	Water + Diet
		Tidewater Goby (*Eucyclogobius newberryi*)	SFBE	Water + Diet
		Delta Smelt (*Hypomesus transpacificus*)	SFBE	Water + Diet
Estuarine/ Marine Aquatic Invertebrates	Saltwater Corals	Cauliflower Coral (*Pocillopora meandrina*)	PH	Water + Diet
		Elkhorn Coral (*Acropora palmate*)	M	Water + Diet
		Boulder Star Coral (*Orbicella franksi*)	M	Water + Diet
		Lobed Star Coral (*Orbicella annularis)*	M	Water + Diet

Table 5-17. Listed Species Taxonomic Groupings Based on Listed Aquatic and Aquatic-Dependent Species within the Seven Representative Action Areas (Continued)

Major Taxonomic Grouping	Listed Species Taxonomic Group[a]	Listed Species/Taxa	RAA(s)[b]	Potential Exposure Pathway Evaluated
Estuarine/ Marine Aquatic Invertebrates	Saltwater Corals	Mountainous Star Coral (*Orbicella faveolata)*	M	Water + Diet
		Pillar Coral (*Dendrogyra cylindrus)*	M	Water + Diet
		Rough Cactus Coral (*Mycetophyllia ferox)*	M	Water + Diet
		Staghorn Coral (*Acropora cervicornis*)	M	Water + Diet
		Cauliflower Coral (*Pocillopora meandrina*)	PH	Water + Diet
	Saltwater Mollusk	White Abalone (*Haliotis sorenseni*)	SD; SFBE	Water + Diet
		Black Abalone (*Haliotis cracherodii*)	SFBE; SD	Water + Diet
Plants	Freshwater-Saltwater Aquatic and Wetland Plants	Cape Sable Thoroughwort (*Chromolaena frustrata*)	M	Water Only
		Pu`uka`a (*Cyperus trachysanthos*)	PH	Water Only
		Water Howellia (*Howellia aquatilis*)	PS	Water Only
		Salt Marsh Bird’s-beak (*Cordylanthus maritimus ssp. maritimus*)	SD	Water Only
		Soft Bird’s-beak (*Cordylanthus mollis ssp. mollis*)	SFBE	Water Only
		Franciscan Manzanita (*Arctostaphylos franciscana*)	SFBE	Water Only
		Sonoma Alopecurus (*Alopecurus aequalis var. sonomensis*)	SFBE	Water Only
		Contra Costa Goldfields (*Lasthenia conjugens*)	SFBE	Water Only
		Suisun Thistle (*Cirsium hydrophilum var. hydrophilum*)	SFBE	Water Only
		Sonoma Sunshine (*Blennosperma bakeri*)	SFBE	Water Only
		Sebastopol Meadowfoam (*Limnanthes vinculans*)	SFBE	Water Only
		Hickman’s Potentilla (*Potentilla hickmanii*)	SFBE	Water Only
		Showy Indian Clover (*Trifolium amoenum*)	SFBE	Water Only
		Decurrent False Aster (*Boltonia decurrens*)	SL	Water Only

Table 5-17. Listed Species Taxonomic Groupings Based on Listed Aquatic and Aquatic-Dependent Species within the Seven Representative Action Areas (Continued)

Major Taxonomic Grouping	Listed Species Taxonomic Group[a]	Listed Species/Taxa	RAA(s)[b]	Potential Exposure Pathway Evaluated
Plants		Eastern Prairie Fringed Orchid (*Platanthera leucophaea*)	SL	Water Only
	Seagrass	Johnson's Seagrass (*Halophila johnsonii*)	M	Water Only

a) Listed species taxonomic groupings are meant to roughly coincide with the Species Group provided in USFWS ECOS, but with further delineation according to life history information and attributes to demonstrate the strong representation given by the subset of 111 RAA listed species to the 674 aquatic and aquatic-dependent species and their critical habitats that may be affected by the Uniform National Discharge Standards Batch Two vessel discharges. One federally listed species taxonomic group is not represented by federally listed species in the RAAs but have suitable surrogate species: freshwater snails (freshwater invertebrates as surrogates)

b) RAA designations: M = Miami, FL RAA; N = Norfolk, VA RAA; PH = Pearl Harbor, HI RAA; PS = Puget Sound, WA RAA; SD = San Diego Bay, CA RAA; SFBE = San Francisco Bay Estuary, CA RAA; SL = St. Louis, MO RAA.

5.4 Pollutant-Specific Risk Summaries

Risks to federally listed aquatic and aquatic-dependent species and their critical habitats from exposure to Batch Two pollutants in deck runoff, firemain systems, graywater, hull coating leachate, sonar dome discharge, submarine bilgewater, surface vessel bilgewater/OWS effluent, and underwater ship husbandry are identified based on the approach described in Section 5.2. The subsections below are subdivided by pollutant category (classical, metals, hydrocarbons) and further subdivided by individual pollutants. Each pollutant-specific subsection includes a summary of direct and indirect effects data used to support the analysis and corresponding data tables with the RQs that have been calculated. Risk from direct exposure is evaluated according to major taxonomic group separated according to fresh-and saltwater habitat (i.e., freshwater aquatic vertebrates, freshwater aquatic invertebrates, freshwater aquatic plants, etc.). Each pollutant-specific subsection is followed by an assessment of risk from indirect effects, which is then followed by a table summarizing risk conclusions made for the major listed species taxonomic groupings provided in Table 5-16. Collectively, the risk conclusions conveyed in this subsection are used with other information to inform the EPA and DoD's effects determinations presented in Section 8 of this BE.

5.4.1 Classical Pollutants

Classical pollutants have been defined to include several standard water quality parameters such as conductivity, salinity, temperature, pH, etc., as well as other parameters the EPA defines as conventional (or common) pollutants (e.g., total suspended solids, and total residual chlorine[TRC] or CPO). Only those pollutants with measured toxic effects to aquatic and aquatic-dependent species lend themselves to quantitative analysis. The only classical pollutant of concern for UNDS Batch Two discharges with measured toxic effects is CPO or TRC.

Residual chlorine is generated during hull cleanings. CPO and residual chlorine are also potentially present in deck runoff and graywater, but at much lower concentrations. Miscellaneous solvents are used to clean and maintain topside equipment. These solvents may contain chlorinated compounds. Crew members may also use detergents during freshwater deck washdowns, as well as in galleys and laundry facilities that generate graywater. However, standards require that detergents used onboard be minimally toxic, and any chlorinated compounds are also volatile and evaporate quickly. As such, their presence in deck runoff and graywater is minimal to nonexistent.

Because of the low boiling point of chlorine (-34.04°C), chlorine is a gas under ambient environmental conditions (HSDB, 2009). Chlorine gas released into water first dissolves and then undergoes immediate conversion into two forms of free chlorine: hypochlorous acid (HOCl) and the hypochlorite ion (OCl^-). If the water contains ammonia (e.g., graywater), the solution will likely also contain two forms of combined chlorine: monochloramine and dichloramine. Because all four of these forms of chlorine can be toxic to aquatic organisms, the term “total residual chlorine or TRC” is used to refer to the sum of free chlorine and combined chlorine in fresh water. However, because salt water contains bromide, addition of chlorine also produces hypobromous acid (HOBr), hypobromous ion (OBr^-), and bromamines (USEPA, 1984a); the term “chlorine-produced oxidants or CPO” is used to refer to the sum of these oxidative products in salt water. Both terms, TRC (fresh water) and CPO (salt water), are intended to refer to the sum of free and combined chlorine and bromine as measured by standard methods.
Chlorine is not expected to bioaccumulate in plants or animals since it reacts with the moist tissues of living systems (Compton, 1987; Schreuder and Brewer, 2001; Schmittinger et al., 2006). Also, chlorine is toxic to microbial communities; therefore, biodegradation is not considered to be a relevant fate process (Vetrano, 2001). In general, the hypochlorous acid formed during the dissolution of chlorine in natural waters reacts with organic and inorganic materials, ultimately forming chloride ion, oxidized inorganics, chloramines, trihalomethanes, oxygen, and nitrogen. Consequently, chlorine (as well as CPO and TRC) does not persist in the aquatic environment.

5.4.1.1 Direct Effects from Chlorine Produced Oxidants and Total Residual Chlorine

Table 5-18 summarizes the chronic toxicity data available for calculating $RQ_{A,W}$ for federally listed aquatic species and RQ_O for federally listed aquatic-dependent species exposed to predicted concentrations of CPO and TRC in water receiving Batch Two discharges from vessels of the Armed Forces. Table 5-19 presents the maximum direct exposure concentrations estimated for an estuarine harbor, representative direct exposure concentrations estimated for river harbors, the dietary exposure from ingestion of TRC in freshwater, and the corresponding RQs calculated for each taxonomic group based on the various chronic toxicity threshold values (in this case, $CTET_{A,W}$ and $CTET_O$) available for representative surrogate taxa. Appropriate CTETs were not available for all of the general taxonomic groups of species.

Risk to Freshwater Aquatic Animal and Plant Species – TRC

The overall risk to freshwater aquatic organisms from direct exposure to TRC discharged from vessels of the Armed Forces is "remote". Measured and modeled concentrations from vessel discharges were not detectable. Therefore, all $RQ_{A,W}$s are well below a value of 0.1.

Risk to Saltwater (Estuarine/Marine) Aquatic Animal and Plant Species – CPO

The risk to saltwater aquatic organisms from direct exposure to CPO discharged from vessels of the Armed Forces is also "remote". The RQs for saltwater vertebrate and invertebrate species are substantially below a $RQ_{A,W}$ of 0.1 (0.00008 and 0.00017, respectively). No appropriate toxicity data are available for saltwater vascular plants exposed to CPO; however, the RQ using the EPA CCC indicates "remote" risk, with an $RQ_{A,W}$ of 0.00049.

Risk to Birds and Mammals – TRC

Because TRC does not bioaccumulate in the tissues of prey [food] organisms, the risk to aquatic-dependent birds and mammals from exposure to TRC in discharges from vessels of Armed Forces is via drinking of [fresh] surface water alone. Furthermore, as noted above, TRC has a short half-life in water and, therefore, does not remain readily available to the organism for uptake. The chronic [oral ingestion] toxicity threshold value (in this case, $CTET_O$) for rodents is 15 mg TRC/kg of body weight per day based on the absence of toxicity in rodents that received chlorine as hypochlorite in drinking water for up to 2 years (equivalent to a tolerable daily intake [TDI] of 5 mg/L) WHO (2003). This was used as a surrogate for marine mammals and aquatic-dependent mammals (e.g., grizzly bears) in the absence of toxicity data for marine mammals and aquatic-dependent mammals and provides a conservative estimate of risk because it is based on drinking water consumption. As discussed, consumption of seawater is considered to be negligible. In addition, marine mammals and aquatic-dependent mammals are very unlikely to occur in ports and harbors where TRC could occur at higher concentrations.

No data are available for birds. However, measured and modeled concentrations from vessel discharges were not detectable. Therefore, the RQ_O for both birds and mammals is well below a value of 0.1, indicating "remote" risk.

Table 5-18. Summary of Chronic Toxicity Effect Thresholds for Aquatic and Aquatic-Dependent Organisms Exposed to Total Residual Chlorine (Freshwater) and Chlorine-Produced Oxidants (Saltwater)

Direct Exposure to TRC and Chlorine Produced Oxidants (CPO)					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Effect Endpoint**	**NOEC - LOEC (µg/L)**	**$CTET_{A,W}$ (µg/L)**	**Study (source)**
Freshwater Vertebrate	Fathead minnow, *Pimephales promelas*	LC Test – growth and survival of progeny	6 - 21	11.2	Arthur et al., 1975 (1984 ALC document)

Table 5-18. Summary of Chronic Toxicity Effect Thresholds for Aquatic and Aquatic-Dependent Organisms Exposed to Total Residual Chlorine (Freshwater) and Chlorine-Produced Oxidants (Saltwater) (Continued)

Direct Exposure to TRC and Chlorine Produced Oxidants (CPO)					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Effect Endpoint**	**NOEC - LOEC (µg/L)**	**$CTET_{A,W}$ (µg/L)**	**Study (source)**
Freshwater Invertebrate	Cladoceran, *Daphnia magna*	LC Test - survival	2 - 7	3.742	Arthur et al., 1975 (1984 ALC document)
Freshwater Plant (Vascular)	Eurasian watermilfoil, *Myriophyllum spicatum*	4-day Test – reduced weight gain of total plant and shoots	–	50	Watkins and Hammershlag, 1984 (1984 ALC document)
Estuarine/Marine Vertebrate	Tidewater silverside, *Menidia peninsulae*	ELS Test – growth and survival	40 - 54	46.48	Goodman et al., 1983 (1984 ALC document)
Estuarine/Marine Invertebrate	Eastern oyster, *Crassostrea viginica*	Acute LC50 = 26/estuarine ACR (ACR = 1.192)	–	21.81 (estimated)	Goodman et al., 1983 (estuarine ACR); Roberts and Gleeson, 1978 (LC50)
Estuarine/Marine Plant	Not Available	–	–	7.5 (CCC)	USEPA 1984, ALC document
Dietary Exposure to TRC and CPO					
Surrogate Aquatic-Dependent Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_o$ (mg/kg bw/d)**	**Study (source)**
Mammals	Rodents	2 years drinking water	NOAEL - growth	15	WHO 2003
Birds	Not Available	–	–	–	–

$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures; the geometric mean of the NOEC and LOEC when both a LOEC and a NOEC were available for the same study

$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)

LC = Life Cycle, ELS = Early Life Stage

Table 5-19. Risk Quotients for Total Residual Chlorine and Chlorine-Produced Oxidants Based on Comparison of Exposure Concentrations with Chronic Toxicity Effect Thresholds of Representative Surrogate Taxa

Taxa	EC_W (µg/L)	$CTET_{A,W}$ (µg/L)	$RQ_{A,W}$
Freshwater Vertebrate	0	11.22	Not calculated
Freshwater Invertebrate		3.742	Not calculated
Freshwater Plant (Vascular)		50	Not calculated
Estuarine/Marine Vertebrate	0.0037	46.48	0.00008
Estuarine/Marine Invertebrate		21.81	0.00017
Estuarine/Marine Plant (Vascular)		7.5	0.00049
Taxa	**EC_O (mg/kg bw/d)**	**$CTET_O$ (mg/kg/d)**	**RQ_{wild}**
Mammals (Freshwater)	0	15	Not calculated
Birds (Freshwater)		–	NA
Mammals (Estuarine/Marine)	0.0037	–	NA
Birds (Estuarine/Marine) –		–	NA

EC_W – Exposure concentration for water

EC_O – Exposure concentration via oral ingestion (diet)

$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures

$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)

$RQ_{A,W}$ – Risk quotient from exposure to TRC or CPO in water calculated as EC_W/ $CTET_{A,W}$

RQ_{Wild} – Risk quotient from exposure to TRC or CPO through diet calculated as EC_O/ $CTET_O$

-- Not available

NA – Not applicable

5.4.2 Metals

Exposure to metals at toxic levels (which is partially dependent on the essentiality of the metal) can cause a variety of changes in biochemical, physiological, morphological, and behavioral patterns in aquatic organisms. One of the key factors in evaluating risk from exposure to metals is the bioavailability of the metal to an organism. In the aquatic environment, elevated concentrations of dissolved metals can be toxic to many species of algae, crustaceans, and fish because it is the dissolved form of the metal that is most readily available and can be taken up and internalized by an organism. However, some metals have a strong tendency to adsorb to suspended organic matter and clay minerals, or to precipitate out of solution, thus removing the metal from the water column. The tendency of a given metal to adsorb to suspended particles is typically controlled by the pH and salinity of the waterbody, as well as the organic carbon content of the suspended particles. If the metal is highly sorbed to particulate matter, then it is likely to not be in a dissolved form that aquatic organisms can process (i.e., bioavailable).

Accordingly, metals criteria for the protection of aquatic life are typically expressed as the dissolved metal form. This is true in the cases of several metals in fresh water, for example, cadmium, copper, lead, nickel, and zinc. The influence of water hardness (calcium and magnesium) on metal toxicity is well-known. These and other elements competitively exclude the free metal ion from adhering to binding sites on a biotic ligand, thereby inhibiting the toxic MoA (see Wood et al., 1997). As such, it is appropriate to also normalize the toxicity values for these metals to a standard water hardness (e.g., 100 mg/L expressed as CaCO3).

In contrast, the use of total metals is more important for aquatic-dependent animals for which the primary route of exposure is assumed to be the consumption of aquatic organisms that have bioaccumulated metals in their tissues. The digestive process is assumed to transform all forms of metals to the dissolved phase, thus increasing the amount of biologically available metals. This also is true for aquatic animals exposed to metalloids such as arsenic and selenium.

The bioavailability of metals is a relative term and depends on many factors (Zhang et al., 2014). For example, particulate metals complexed to suspended organic matter or clay minerals may be recycled into the water column and become bioavailable due to physical re-suspension of bed sediments (e.g., from dredging activities or propeller wash) or bioturbation (the stirring or mixing of sediment particles by benthic animals) (Kalnejais et al., 2007; Amato et al., 2016; Fetters et al., 2016). Depending on water column chemistry and microbiological activity within the surficial sediment layers, these physical and biological activities might re-mobilize the metals into the dissolved phase (Gadd, 2010; Zhang et al., 2014), making them bioavailable for potential uptake by aquatic organisms.

Likewise, certain benthic organisms that consume organic material in sediments (so-called "deposit feeders", such as polychaete worms and bivalve mollusks) might consume particulate-bound metals and re-release metals via digestion and excretion or introduce metals into the food chain when consumed by predators (Diaz and Schaffner, 1990; Chen and Mayer, 1999; Croteau et al., 2005; Kalantzi et al., 2014). There is an absence of scientific and commercial data available to accurately account for these types of pathways of potential exposure in this BE for metals.

The Batch Two metals included in this biological evaluation are cadmium, chromium, copper, iron, lead, mercury, nickel, and zinc. Although total concentrations of these metals were measured for all of the discharges evaluated, dissolved concentrations were only measured for copper and zinc in a couple of the discharges. To be conservative, only total concentrations of these metals were evaluated and assumed to be 100% bioavailable. The basis for the various RQs calculated to support this analysis is provided in Appendices G and H and discussed below.

5.4.2.1 Cadmium

Cadmium was detected in deck runoff at concentrations above the screening benchmark. It occurs naturally in the aquatic environment, but it has no known biological use and is considered to be one of the most toxic metals to fish (Sorensen, 1991). Cadmium is bioconcentrated by organisms, but is not biomagnified through the food chain (Eisler, 1985). Toxicity of cadmium to aquatic organisms varies with water hardness, alkalinity, the type and life stage of organisms, presence of organic matter, presence of other toxicants, and the duration of exposure (USEPA, 2001). Cadmium is a known teratogen, carcinogen, and a probable mutagen to freshwater organisms (Eisler, 1985).

One known mechanism of cadmium toxicity to fish is suppression of calcium uptake

(Verbost et al., 1987). Calcium is vital for fish growth (Pelgrom et al., 1997), and bone repair mechanisms are probably inhibited due to the hypocalcemic effect of cadmium (DWAF, 1996). Since cadmium toxicity to freshwater aquatic animals has been shown to be related to water hardness, it is appropriate to normalize chronic toxicity effect thresholds for freshwater aquatic animals to a standard water hardness (i.e., 100 mg/L as $CaCO_3$) for comparative purposes and to support risk calculation (EPA, 2005a).

5.4.2.1.1 Direct Effects from Cadmium

Table 5-20 summarizes the chronic toxicity values available for calculating the risk quotients for aquatic and aquatic-dependent species exposed to cadmium. Table 5-21 presents the maximum exposure concentrations estimated in ambient receiving waters (EC_W), in tissues of aquatic organisms exposed to those concentrations (EC_T), and in oral doses of aquatic- dependent animals ingesting food (prey) and water in those receiving waters (EC_O). Exposure concentrations in ambient receiving water are from the harbor modeling, as described above in Section 5.2.1.

Risk to Freshwater Aquatic Animal and Plant Species from Cadmium

The overall risk to freshwater aquatic organisms as a result of direct exposure to cadmium in ambient waters discharged from vessels of the Armed Forces is “remote”. All $RQ_{A,W}$s are substantially less than 0.1, with the highest RQ of 1.67E-08 estimated for freshwater invertebrates.

The evaluation of potential risks from bioaccumulated cadmium provides a means of identifying potential risks to listed aquatic species via exposure to from all exposure routes combined. The estimated concentration of cadmium in vertebrate and invertebrate tissue from continuous exposure to cadmium in ambient receiving waters is 1.2E-09 and 1.2E-10 mg/kg, respectively based on a modeled receiving water concentration of 3.2E-09 µg/L and a BAF of 366 for vertebrates and 38 for invertebrates. The RQs calculated for freshwater vertebrates and invertebrates based on estimated concentrations accumulated in tissues are 2.3E-07 and 1.0E-09, respectively, indicating “remote” risk to freshwater aquatic animals from cadmium accumulated in tissues (see Table 5-21).

Risk to Saltwater (Estuarine/Marine) Aquatic Animal and Plant Species from Cadmium

The overall risk to saltwater aquatic organisms as a result of exposure to maximum concentrations of cadmium in ambient waters discharged by vessels of the Armed Forces is also “remote”. Based on comparison of the maximum modeled receiving water concentration of 0.00001 µg/L with the selected $CTET_{A,W}$s, all $RQ_{A,W}$s are 1.1E-06 or less.

Again, based on a BAF of 366 for vertebrates and 38 for invertebrates, tissue concentrations for saltwater vertebrates and invertebrates from continuous exposure to maximum concentrations of cadmium in ambient receiving waters are estimated to be 2.9E-06 and 3.0E-07 mg/kg, respectively (see Table 5-21). The corresponding RQs were 7.1E-05 and 2.1E-06, respectively, again indicating “remote” risk to aquatic animals from cadmium accumulated in tissues.

Risk to Birds and Mammals from Cadmium

An evaluation of risk to aquatic-dependent birds and mammals from exposure to cadmium in discharges from vessels of the Armed Forces via consumption of prey items that have bioaccumlated cadmium or drinking ambient surface water indicates that risk to aquatic-dependent wildlife is "remote". The RQs for representative freshwater and saltwater mammals and birds based on the chronic toxicity values in relation to the dietary concentrations calculated for this BE are all low (2.0E-07 to 9.4E-11).

Table 5-20. Summary of Chronic Toxicity Effect Thresholds for Aquatic and Aquatic-dependent Organisms Exposed to Cadmium

Direct Exposure to Cadmium					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Effect Endpoint**	**NOEC - LOEC (µg/L)**	**$CTET_{A,W}$ (µg/L)**	**Study (source)**
Freshwater Vertebrate	Rainbow trout, *Oncorhynchus mykiss*	ELS Test - delayed hatch & growth	0.59 - 1.3	0.8757 (**0.7962***)	Davies and Brinkman, 1994 (2016 ALC doc.)
Freshwater Invertebrate	Cladoceran, *Daphnia magna*	LC Test - reproduction	0.16 – 0.28	0.3545 (**0.1967 ***)	Chapman et al. manuscript, 1980 (2016 ALC doc.)
Freshwater Plant (Vascular)	Duckweed, *Lemna minor*	96-hour - growth	<1 - 1	<1	Megateli et al. 2009 (NODA Review)
Estuarine/Marine Vertebrate	Striped bass, *Morone saxatilis*	Acute LC50 ÷ Estuarine ACR (ACR=9.11)	75.0 ÷ 9.11	8.232 (**8.182** *) (estimated)	Gentile et al. 1982; Nimmo et al. 1977; Lussier et al. 1985 (ACR); Palawski et al. 1985 (LC50)
Estuarine/Marine Invertebrate	Mysid, *Americamysis bahia*	LC Test - reproduction	5.1 – 10	7.141 (**7.099** *)	Gentile et al. 1982; Lussier et al. 1985 (2016 ALC doc.)
Estuarine/Marine Plant	Not Available	–	–	9.3	CCC from USEPA 2001
Accumulation of Cadmium in Tissue from Multiple Exposure Routes					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_{A,T}$ (mg/kg wet wt.)**	**Study (source)**
Freshwater Vertebrate	Atlantic salmon, *Salmo salar*	ELS Test	Reduction in growth	0.0050	Rombough and Garside 1982
Freshwater Invertebrate	Amphipod, *Hyalella azteca*	42-day Mesocosm Study	Significant decrease in weight	0.118	Stanley et al. 2005
Estuarine/Marine Vertebrate	European flounder, *Platichthys flesus*	ELS Test	Reduction in percentage of eggs that hatch	0.04	von Westernhagen and Dethlefsen 1975

Table 5-20. Summary of Chronic Toxicity Effect Thresholds for Aquatic and Aquatic-dependent Organisms Exposed to Cadmium (Continued)

Accumulation of Cadmium in Tissue from Multiple Exposure Routes					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_{A,T}$ (mg/kg wet wt.)**	**Study (source)**
Estuarine/Marine Invertebrate	Starfish, *Asterias ubens*	5 month Semi-field Study	76% reduction in normal embryo development	0.14	den Besten et al. 1989
Dietary Exposure to Cadmium					
Surrogate Aquatic-Dependent Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	$CTET_o$ (mg/kg bw/d)	**Study (source)**
Mammals	Rat	6-wk oral gavage	Reproduction	1	Sample et al. 1996 (Oregon Toxics BE)
Birds	Mallard	90-d oral in diet	Reproduction	1.45	Sample et al. 1996 (Oregon Toxics BE)

$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures

$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue

$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)

Bolded entries are for dissolved cadmium

*Freshwater data normalized to hardness = 100 mg/L CaCO3 using EPA conversion factor of 0.9090; for estuarine/marine data, a conversion factor of 0.994 was used (EPA 1996)

LC = life cycle

ELS = early life stage

Table 5-21. Risk Quotients for Cadmium Based on Comparison of Exposure Concentrations with Chronic Toxicity Effect Thresholds of Representative Surrogate Taxa

Taxa	**EC_W (µg/L)**	**$CTET_{A,W}$ (µg/L)**	**$RQ_{A,W}$**
Freshwater Vertebrate	3.2E-09	0.7962	4.0E-09
Freshwater Invertebrate		0.1969	1.6E-08
Freshwater Plant (Vascular)		1	3.2E-09
Estuarine/Marine Vertebrate	0.00001	8.182	9.5E-07
Estuarine/Marine Invertebrate		7.009	1.1E-06
Estuarine/Marine Plant (Vascular)		9.3	8.4E-07
Taxa	**EC_T (mg/kg wet wt)**	**$CTET_{A,T}$ (mg/kg wet wt)**	**$RQ_{A,M}$**
Freshwater Vertebrate	1.2E-09	0.005	2.3E-07
Freshwater Invertebrate	1.2E-10	0.118	1.0E-09
Estuarine/Marine Vertebrate	2.9E-06	0.04	7.1E-05
Estuarine/Marine Invertebrate	3.0E-07	0.14	2.1E-06

Table 5-21. Risk Quotients for Cadmium Based on Comparison of Exposure Concentrations with Chronic Toxicity Effect Thresholds of Representative Surrogate Taxa (Continued)

Taxa	EC_O (mg/kg bw/d)	$CTET_O$ (mg/kg/d)	RQ_{wild}
Mammals (Freshwater)	9.4E-11	1	9.4E-11
Birds (Freshwater)	1.2E-10	1.45	8.1E-11
Mammals (Estuarine/Marine)	2.3E-07	1	2.3E-07
Birds (Estuarine/Marine)	2.9E-07	1.45	2.0E-10

EC_W – Exposure concentration for water
EC_T – Exposure concentration accumulated in tissue
EC_O – Exposure concentration via oral ingestion (diet)
$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures
$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue
$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)
$RQ_{A,W}$ – Risk quotient from exposure to cadmium in water calculated as EC_W/ $CTET_{A,W}$
$RQ_{A,M}$ – Risk quotient from accumulation of cadmium in tissue calculated as EC_T/ $CTET_{A,T}$
RQ_{Wild} – Risk quotient from exposure to cadmium through diet calculated as EC_O/ $CTET_O$

5.4.2.1.2 Indirect Effects from Cadmium

The RQs summarized above indicate “remote” risk for direct effects of dissolved cadmium to listed aquatic and aquatic-dependent species in this BE. Furthermore, discharge of dissolved cadmium from vessels of the Armed Forces will not result in appreciable concentrations in estuaries and freshwater receiving water bodies. There is, therefore, no basis for assuming detectable indirect effects on listed species from cadmium in discharges from vessels of the Armed Forces. Because cadmium at maximum modeled exposure concentrations has “remote” risk of directly affecting fresh- and saltwater aquatic animals and plants, toxicity-related reduction in available habitat or the prey base [loss of prey] available to federally listed species is unlikely to occur. For this reason, indirect effects typically associated with exposure, including loss of cover and changes in water quality parameters, are not expected for this metal.

5.4.2.1.3 Risk Conclusions for Each Taxonomic Group of Federally Listed Species in the Representative Action Areas from Exposure to Cadmium

Based on the direct and indirect effects assessment from exposure of vertebrate, invertebrate and wildlife species to cadmium in Batch Two discharges, the following risk conclusions are being made for each of the listed species groups in the RAAs (see Table 5-22). This information is used to inform decisions in the effects determinations made in Section 8 of this document.

Table 5-22. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Cadmium Discharged from Vessels of the Armed Forces to Ports and Harbors

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Saltwater Corals	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Surrogate CTET is representative and appropriate
Unionid Mussel	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a 77-day study with zebra mussel (*Dreisena polymorpha*) with an EC50 of 130 µg/L at a hardness of 268 mg/L (70.22 µg/L normalized to 100 mg/L hardness) (Kraak et al., 1992). Resulting RQ is 4.6E-11.
Freshwater Snails	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion confirmed using a chronic early life stage study using *Physa acuta* with a NOEC and LOEC for embryonic growth of 0.32 and 0.50 mg/L, respectively, and a LOEC for hatchability of 0.13 mg/L. Application of a safety factor of 10 to the LOEC for hatchability results in a hatchability NOEC of 0.013 mg/L. Resulting RQ is 2.5E-10.
Saltwater Mollusk	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed via estimation using a chronic study with the common mussel (*Mytilus edulis*) with 6% mortality at 200 µg/L (only conc.) at a hardness of 31 mg/L (415.6 µg/L normalized to 100 mg/L hardness) (Geret et al., 2002). Resulting RQ is 2.4E-08.

Table 5-22. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Cadmium Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Freshwater Shrimp/ Crustacean	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed via estimation using a 24-hr acute study with fairy shrimp (*Streptocephalus proboscideus*) with a LC50 of 250 µg/L at hardness of 250 mg/L (116.1 µg/L normalized to 100 mg/L hardness) (Crisinel et al., 1994). Using an ACR for freshwater invertebrates of 34.30 results in a chronic value of 3.384 µg/L. Resulting RQ is 9.0E-10.
Freshwater Fish/ Inland Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed via estimation using a chronic study with rainbow trout (*O. mykiss*) and an maximum acceptable toxicant concentration (MATC) (growth) of 7.29 µg/L (normalized to 100 mg/L on a dissolved basis) (Mayer et al., 2008). Resulting $RQ_{A,W}$ is 3.0E-06.
Anadromous Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • No chronic toxicity data were identified for salmonids tested in seawater. However, a sub-lethal study with Atlantic salmon (*Salmo salar*) aelvins showed slower growth at concentrations as low as 0.13 µg/L (Rombough and Garside 1982 as cited in Price 2013). Resulting $RQ_{A,W}$ is 2.5E-08.
Freshwater Fish/Inland Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted
Anadromous Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • No chronic toxicity data were identified for sturgeon tested in seawater.

Table 5-22. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Cadmium Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Estuarine/Marine Fish	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • RQ directly applicable because the CTET is based on toxicity for *M. saxatilis* as presented in Table 5-20 above
Beetle and Aquatic Insect	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted
Amphibian	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed via estimation using a 60-d study with American toad (*Bufo americanus*) reduced growth at 5 µg/L (lowest concentration tested) at hardness 51.2 mg/L (7.60 µg/L normalized to 100 mg/L hardness) (James and Little, 2003). Resulting RQ is 4.2E-10.
Snakes and Other Reptiles	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted
Sea Turtle	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted
Coastal/Marine Bird	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted
Marine Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects

Table 5-22. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Cadmium Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
		• Fate of cadmium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted
Terrestrial Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted
Seagrass	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (change in habitat/water quality parameters) at the concentrations predicted
Freshwater - Saltwater Aquatic and Wetland Plants	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of cadmium indicates very low potential for indirect effects (change in habitat/water quality parameters) at the concentrations predicted • RQ directly applicable because the CTET is based on toxicity for *L. gibba* as presented in Table 5-20

5.4.2.2 Chromium

Chromium was found to be present in deck runoff at concentrations above benchmark screening levels. Chromium is a naturally occurring metallic element present in minute concentrations in the environment. Eisler (1986) reported that background levels of chromium in freshwater environments ranged between 1 and 10 µg/L while background levels in seawater ranged from 0 to 5 µg/L. Anthropogenic sources of chromium are predominantly atmospheric.

The most common forms of chromium are trivalent (chromium III) and hexavalent chromium (chromium VI). In aquatic environments chromium is found as chromium (III) and chromium (VI) as water soluble complex anions. Chromium oxidation states range from -2 to +6, but it is most frequently found in the environment in the trivalent (+3) and hexavalent (+6) oxidation states (4, 5, 7). The +2, +4 and +5 forms are unstable and are rapidly converted to +3, which in turn is oxidized to +6 (Eisler, 1986).

In freshwater environments, hydrolysis and precipitation are the most important processes in determining the environmental fate of chromium. Information on the geochemical behavior of chromium in seawater is limited. Bioaccumulation occurs mostly in aquatic biota with gills, and inorganic forms of chromium do not biomagnify (Eisler, 1986).

Chromium (III) occurs naturally in the environment and is an essential nutrient. In contrast, chromium (VI) is generally man-made and more toxic to aquatic organisms because of its oxidizing potential and it easily penetrates biological membranes (Steven et al., 1976, Taylor and Parr, 1978). Chromium toxicity to aquatic biota is significantly influenced by abiotic variables such as water hardness, temperature, pH, and salinity (Eisler, 1986). Sensitivity to chromium varies widely, even among closely related species (Eisler, 1986).

For this analysis, the EPA and DoD assumed that chromium in Batch Two discharges is the more toxic (chromium VI) form of this metal, although the form of chromium measured in discharges was total chromium (III and VI).

5.4.2.2.1 Direct Effects from Chromium

Table 5-23 summarizes the chronic toxicity values available for calculating the risk quotients for aquatic and aquatic-dependent species exposed to dissolved chromium. Table 5-24 presents the maximum chromium exposure concentrations estimated in ambient receiving waters (EC_W), in tissues of aquatic organisms exposed to those concentrations (EC_T), and in oral doses of aquatic-dependent animals ingesting food (prey) and water in those receiving waters (EC_O). Maximum exposure concentrations in ambient receiving water are from the harbor modeling, as described above in Section 5.2.1.

Risk to Freshwater Aquatic Animal and Plant Species from Chromium

The overall risk to freshwater aquatic organisms as a result of direct exposure to chromium discharged from vessels of the Armed Forces to surface water is “remote”. All $RQ_{A,W}$ are 1.0E-07 or lower.

Because chromium is a nutritionally essential inorganic element, chromium has the potential to accumulate in tissues of aquatic animals. Like other essential metals, aquatic organism exposure to chromium concentrations in excess of nutritional needs via other potential routes of exposure in addition to direct waterborne toxicity (e.g. dietary toxicity) may pose a threat to listed species. While chromium does not bioaccumulate to high levels in aquatic animals (Reid, 2012), the EPA and DoD believe it is prudent to incorporate an analysis on multiple routes of exposure in the analysis of this pollutant. The evaluation of potential risks from bioaccumulated chromium provides a means of identifying potential risks to listed species from all exposure routes combined. An $RQ_{A,M}$ was calculated for freshwater vertebrates based on comparison of estimated concentrations accumulated in tissues from continuous exposure to a modeled concentration of chromium in ambient receiving waters of 1.6E-07 μg/L (see Table 5-24). The resulting RQ is 5.0E-09, indicating “remote” risk to freshwater aquatic vertebrates from chromium accumulated in tissues. Although tissue concentrations of chromium in freshwater invertebrates was estimated (2.5E-08 mg/kg), a $CTET_{A,T}$ for freshwater invertebrates was not identified, and risk to freshwater invertebrates from multiple exposure routes could not be quantified.

Risk to Saltwater (Estuarine/Marine) Aquatic Animal and Plant Species from Chromium

The overall risk to saltwater aquatic organisms as a result of direct exposure to chromium discharged from vessels of the Armed Forces to surface water is also "remote". Based on comparison of maximum modeled receiving water concentrations with selected CTET $_{A,WS}$, all $RQ_{A,WS}$ are 0.00002 or less.

An $RQ_{A,M}$ was calculated for saltwater invertebrates based on comparison of estimated concentrations accumulated in tissues from continuous exposure to maximum concentrations of chromium in ambient receiving waters of 0.00039 µg/L (see Table 5-24). Based on a BAF of 40 for vertebrates and 158.5 for invertebrates, the resulting $RQ_{A,M}$s for estuarine/marine invertebrates and vertebrates are 1.9E-05 and 5.9E-05, respectively, indicating "remote" risk.

Risk to Birds and Mammals from Chromium

An evaluation of risk to aquatic-dependent birds and mammals from dietary exposure to chromium in prey items or drinking water was performed because of the apparent ability of some organisms to accumulate chromium in tissues, albeit to low levels. The RQs for surrogate mammals and birds based on comparison of modeled dietary doses with the chronic toxicity values all indicate "remote" risk for aquatic-dependent receptors, ranging from 1.6E-10 to 1.4E-06.

Table 5-23. Summary of Chronic Toxicity Effect Thresholds for Aquatic and Aquatic-Dependent Organisms Exposed to Chromium

Direct Exposure to Chromium					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Effect Endpoint**	**NOEC - LOEC (µg/L)**	**$CTET_{A,W}$ (µg/L)**	**Study (source)**
Freshwater Vertebrate	Atlantic salmon, *Salmo salar*	ELS Test - survival	10.1 - 97	31.30 (**30.11***)	Grande and Andersen 1983 (NODA review)
Freshwater Invertebrate	Cladoceran, *Daphnia magna*	LC Test - Reproduction	0.8 – 3.2	1.6 (**1.539***)	Spehar and Fiandt 1986 (NODA review)
Freshwater Plant (Vascular)	Esthwaite waterweed, *Hydrilla verticullata*	3-day test – Decrease chlorophyll content	10 - 100	31.62 (**30.42***)	Rai et al. 1995 (NODA Review)
Estuarine/Marine Vertebrate	Atlantic silverside, *Menidia menidia*	Acute LC50 ÷ Estuarine ACR (ACR=15.38)	14,271 ÷ 15.38	927.9 (**921.4** *, estimated)	Lussier et al. 1985 (ACR); Cardin 1982; 1985a (LC50) (NODA review)
Estuarine/Marine Invertebrate	Polychaete worm, *Neanthes*	LC Test - Reproduction	12.5 - 25	17.68 (**17.56** *)	Oshida et al. 1981 (NODA review)
Estuarine/Marine Plant (Vascular)	Fungus, *Apergilus flavus*	15-day Biomass Test	100,000 - >100,00	>100,000 (**>99,300***)	Vala et al. 2004 (NODA review)

Table 5-23. Summary of Chronic Toxicity Effect Thresholds for Aquatic and Aquatic-Dependent Organisms Exposed to Chromium (Continued)

Accumulation of Chromium in Tissue from Multiple Exposure Routes					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_{A,T}$ (mg/kg wet wt.)**	**Study (source)**
Freshwater Vertebrate	Chinook salmon, *Oncorhynchus tshawytscha*	134-day Juvenile Exposure (Laboratory)	29.2% weight reduction, morphological changes in Kidney, DNA damage	1.28	Farag et al. 2006
Freshwater Invertebrate	Not Available	–	–	–	–
Estuarine/Marine Vertebrate	Mummichog, *Fundulus heteroclitus*	Larval assay	Reduction in growth	0.263	Roling et al. 2006
Estuarine/Marine Invertebrate	Flower crab, *Portunus pelagicus*	–	Decreased size	3.20	Mortimer and Miller 1994
Dietary Exposure to Chromium					
Surrogate Aquatic-Dependent Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	$CTET_o$ (mg/kg bw/d)	**Study (source)**
Mammals	Rat	1-year oral in water	Body weight and food consumption	3.28	Sample et al. 1996 (Oregon Toxics BE)
Birds	Chick	–	–	8.59	Romoser et al. 1961 (Oregon Toxics BE)

$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures
$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue
$CTET_{O}$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)
Bolded entries are for dissolved chromium.
*Using EPA dissolved conversion factor of 0.962 for freshwater and 0.993 for estuarine/marine data

Table 5-24. Risk Quotients for Chromium Based on Comparison of Exposure Concentrations with Chronic Toxicity Effect Thresholds of Representative Surrogate Taxa

Taxa	EC_W (µg/L)	$CTET_{A,W}$ (µg/L)	$RQ_{A,W}$
Freshwater Vertebrate	1.6E-07	30.11	5.3E-09
Freshwater Invertebrate		1.539	1.0E-07
Freshwater Plant (Vascular)		30.42	5.3E-09
Estuarine/Marine Vertebrate	0.00039	921.4	4.2E-07
Estuarine/Marine Invertebrate		17.56	2.2E-05
Estuarine/Marine Plant (Vascular)		>99,300	3.9E-09

Table 5-24. Risk Quotients for Chromium Based on Comparison of Exposure Concentrations with Chronic Toxicity Effect Thresholds of Representative Surrogate Taxa (Continued)

Taxa	EC_T (mg/kg wet wt)	$CTET_{A,T}$ (mg/kg wet wt)	$RQ_{A,M}$
Freshwater Vertebrate	6.4E-09	1.28	5.0E-09
Freshwater Invertebrate	2.5E-08	Not available	Not calculated
Estuarine/Marine Vertebrate	1.6E-05	0.263	5.9E-05
Estuarine/Marine Invertebrate	6.2E-05	3.2	1.9E-05
Taxa	**EC_O (mg/kg bw/d)**	**$CTET_O$ (mg/kg/d)**	**RQ_{wild}**
Mammals (Freshwater)	5.2E-10	3.28	1.6E-10
Birds (Freshwater)	4.8E-09	8.59	3.6E-08
Mammals (Estuarine/Marine)	1.2E-06	3.28	3.8E-07
Birds (Estuarine/Marine)	1.2E-05	8.59	1.4E-06

EC_W – Exposure concentration for water

EC_T – Exposure concentration accumulated in tissue

EC_O – Exposure concentration via oral ingestion (diet)

$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures

$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue

$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)

$RQ_{A,W}$ – Risk quotient from exposure to chromium in water calculated as EC_W/ $CTET_{A,W}$

$RQ_{A,M}$ – Risk quotient from accumulation of chromium in tissue calculated as EC_T/ $CTET_{A,T}$

RQ_{Wild} – Risk quotient from exposure to chromium through diet calculated as EC_O/ $CTET_O$

5.4.2.2.2 Indirect Effects from Chromium

The RQs summarized above and in Table 5-24 indicate “remote” risk for direct effects of dissolved chromium (expressed as chromium VI) to federal listed aquatic and aquatic-dependent species in this BE. Furthermore, discharge of dissolved chromium from vessels of the Armed Forces will not result in appreciable concentrations in estuaries and freshwater receiving water bodies. There is, therefore, no basis for assuming detectable indirect effects on listed species from chromium in discharges from vessels of the Armed Forces. Because chromium at maximum modeled exposure concentrations has little risk of directly affecting fresh- and saltwater aquatic animals and plants, toxicity-related reduction in available habitat or the prey base [loss of prey] available to federally listed species is unlikely to occur. For this reason, indirect effects typically associated with exposure, including loss of cover and changes in water quality parameters, are not expected for this metal.

5.4.2.2.3 Risk Conclusions for Each Taxonomic Group of Federally Listed Species in the Representative Action Areas from Exposure to Chromium

Based on the direct and indirect effects assessment from exposure of vertebrate, invertebrate and wildlife species to chromium in Batch Two discharges, the following risk conclusions are being made for each of the listed species groups in the RAAs (see Table 5-25). This information is used to inform decisions in the effects determinations made in Section 8 of this document.

Table 5-25. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Chromium Discharged from Vessels of the Armed Forces to Ports and Harbors

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Saltwater Corals	Remote	• Exposed via water column • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Surrogate CTET is representative and appropriate
Unionid Mussel	Remote	• Exposed via water column • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using acute toxicity data for a juvenile mussel (*Andonta imbecillis*) with an LC50 of 39 µg/L (Keller and Zam,1991). Dividing the acute value by the freshwater invertebrate ACR of 5.00 results in a chronic value of 7.8 µg/L. Resulting RQ is 2.1E-08.
Freshwater Snail	Negligible	• Exposed via water column • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion confirmed using chronic early life stage studies with *Radix quadrasi* with a LOEC of 0.0050 mg/L for abnormal development (Factor and Chavez, 2012). Application of a safety factor of 10 results in chronic NOEC of 0.00050 mg/L. Resulting RQ is 3.2E-07.
Saltwater Mollusk	Remote	• Exposed via water column • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a 7-day study with Asian clam (*Potamocorbula amurensis*) with a MATC (survival) of 14,664 µg/L. Resulting RQ is 2.7E-08.

Table 5-25. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Chromium Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Freshwater Shrimp/ Crustacean	Remote	• Exposed via water column • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed via estimation using a 24-hour acute study with fairy shrimp (*Streptocephalus texanus*) and an LC50 of 53.1 μg/L (Crisinel et al. 1994). Dividing by the freshwater invertebrate ACR of 5.00 results in a chronic value of 10.62 μg/L. Resulting RQ is 1.5E-08.
Freshwater Fish/ Inland Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • RQ directly applicable because the CTET is based on toxicity for *S. salar* as presented in Table 5-23 above
Anadromous Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed via estimation using a 15-day study with Coho/silver salmon (*O. kisutch*) and a MATC (survival) of 23,792 μg/L (Holland et al., 1960). Resulting RQ is 1.6E-08.
Freshwater Fish/Inland Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • No chronic toxicity data were identified for sturgeon tested in freshwater.
Anadromous Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • No chronic toxicity data were identified for sturgeon tested in seawater.

Table 5-25. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Chromium Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Estuarine/Marine Fish	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • RQ directly applicable because the CTET is based on toxicity for *M. saxatilis* as presented in Table 5-23 above
Beetle and Aquatic Insect	Remote	• Exposed via water column • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted
Amphibian	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using chronic toxicity data with Montevido tree frog (*Hypsiboas pulchellus*) and a MATC (survival) of 4,243 µg/L (Natale et al., 2006). Resulting RQ is 3.8E-11.
Snakes and Other Reptiles	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted
Sea Turtle	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted
Coastal/Marine Bird	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted
Marine Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects

Table 5-25. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Chromium Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
		• Fate of chromium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted
Terrestrial Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted
Seagrass	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (change in habitat/water quality parameters) at the concentrations predicted • RQ directly applicable because the CTET is based on toxicity for *A. flavus* as presented in Table 5-23 above
Freshwater - Saltwater Aquatic and Wetland Plants	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of chromium indicates very low potential for indirect effects (change in habitat/water quality parameters) at the concentrations predicted • RQ directly applicable because the CTET is based on toxicity for *H. verticillata* as presented in Table 5-23 above

5.4.2.3 Copper

Elevated concentrations of dissolved copper may occur in a variety of vessel discharges, including firemain systems, graywater, hull coating leachate, sonar dome discharge, submarine bilgewater, surface vessel bilgewater/OWS effluent, and underwater ship husbandry.

Copper toxicity is dependent on copper speciation, with free ionic copper (Cu2+) being the main toxic form of copper. In general, free ionic copper, and thus potential toxicity, will be high in waters of low ionic strength, low concentrations of natural organic matter, and low pH (Rivera-Duarte et al., 2005; Grosell, 2012). In general, fish in seawater show greater overall tolerance to copper than those in freshwater (Grosell et al., 2007). In addition recent studies have revealed that euryhaline fish (fish able to adapt to a wide range of salinities, such as herring, salmon, sturgeon, striped bass, and trout) are less sensitive to copper at intermediate salinities (e.g., estuaries), but at extreme salinities (freshwater and fully marine) there is apparent copper induced mortality (Blanchard and Grosell, 2006). Invertebrate studies have shown that the amount of total copper needed to reach an effects threshold increases with increasing salinity, suggesting that the copper complexation capacity at higher salinities in estuaries controls copper toxicity by keeping the concentration of Cu2+ at less toxic levels (Rivera-Duarte et al., 2005).

Even at higher salinities, though, copper can be toxic to more sensitive species and life stages. Copper has been shown to inhibit coral fertilization and metamorphis when exposed either in solution or to inert surfaces (Negri and Hayward, 2001; Victor and Richmond, 2005). One study shows that there could also be a change in sensitivity to copper over the spawning period (Hédouin and Gates, 2013).

Although toxic at higher concentrations, copper is an essential element for all aerobic organisms, as it is utilized by mitochondrial cytochrome c oxidase and is a cofactor for a number of other enzymes (Solomon and Lowery, 1993). The importance of copper as a micronutrient for teleost fish (i.e., ray-finned fishes) is demonstrated by reduced growth under conditions of low ambient and dietary copper (Ogino and Yang, 1980; Gatlin and Wilson, 1986; Kamunde et al., 2002). Biomagnification is not considered a major factor for copper due to the fact that copper uptake is under homeostatic control (i.e., carrier-mediated and a function of concentration in water and diet) (Grosell, 2012). In fact, BCFs (accumulation from dissolved sources) and BAFs (accumulation from dissolved and dietary sources combined) for copper in freshwater organisms, in general, are inversely related to exposure concentrations (McGeer et al., 2003; DeForest et al., 2007).

Because copper toxicity to freshwater aquatic animals has been shown to be related to water hardness, it is appropriate to normalize CTETs for freshwater aquatic animals to a standard water hardness (i.e., 100 mg/L as $CaCO_3$) for comparative purposes and to support risk calculation (EPA, 2005a).

5.4.2.3.1 Direct Effects from Copper

Table 5-26 summarizes the chronic toxicity values available for calculating the risk quotients for aquatic species exposed to dissolved copper in water ($CTET_{A,W}$), for aquatic species that bioaccumulate copper in their tissues ($CTET_{A,T}$), and for aquatic-dependent wildlife species that are exposed to copper orally through their diet ($CTET_O$). Table 5-27 presents the maximum exposure concentrations estimated in ambient receiving waters (EC_W), in tissues of aquatic organisms exposed to those concentrations (EC_T), and in oral doses of aquatic-dependent animals ingesting prey and water in those receiving waters (EC_O). Maximum exposure concentrations in ambient receiving water are estimated from the harbor modeling, as described above in Section 5.2.1. Table 5-27 also presents the RQs calculated as the ratio of each EC to the corresponding CTET.

Risk to Freshwater Aquatic Animal and Plant Species – Copper

The risk to freshwater aquatic organisms from exposure to copper concentrations in ambient receiving water as the result of discharges from vessels of the Armed Forces is expected to generally be low. Based on comparison of modeled receiving water concentrations with chronic effects thresholds (CTETs), the $RQ_{A,W}$s for freshwater vertebrates and the more “chronically copper-sensitive” freshwater invertebrates are 1.4E-05 and 1.7E-05, respectively, and indicate only “remote” risk of adverse effects (Table 5-27). The $RQ_{A,W}$ for freshwater vascular plants is two orders of magnitude lower at approximately 5.9E-07, also indicative of "remote" risk of adverse effects.

Because copper is a nutritionally essential inorganic element for all plants and animals, it is readily accumulated in tissues of aquatic plants and animals. As long as water column or dietary concentrations are not so high as to overwhelm homeostatic mechanisms, aquatic species are able to regulate their internal body burden of copper. Aquatic organism exposure to copper concentrations in excess of nutritional needs via other potential routes of exposure in addition to direct waterborne toxicity (e.g., dietary toxicity) may pose a threat to listed species. However, whole-body concentrations tend to decrease with increasing trophic level, and it is believed copper is regulated or immobilized in many species. Therefore, it is not biomagnified in food chains to any significant extent (CCREM 1987). While copper does not biomagnify, the EPA and DoD believe it is prudent to incorporate an analysis on multiple routes of exposure in their analysis of this pollutant. The evaluation of potential risks from bioaccumulated copper provides a means of identifying potential risks to listed species via exposure to criteria concentrations from all exposure routes combined.

$RQ_{A,M}$ for freshwater aquatic vertebrates and invertebrates based on estimated concentrations accumulated in tissues from continuous exposure to copper by multiple routes at a modeled concentration of 6.7E-05 μg/L in ambient receiving waters are approximately 6.5E-06 and 5.1E-06, respectively, indicating "remote" risk to freshwater aquatic animals from copper accumulated in tissues.

Risk to Saltwater (Estuarine/Marine) Aquatic Animal and Plant Species from Copper

Because modeled concentrations of copper in estuarine/marine ports and harbors is four orders of magnitude higher than modeled concentrations from freshwater ports and harbors, the risk to saltwater aquatic organisms from maximum exposure to dissolved copper in ambient receiving water is greater than the same risk predicted for freshwater organisms based on predicted RQs. However, risk is still "remote" for saltwater vertebrates ($RQ_{A,W}$ = 0.0038), and vascular plants ($RQ_{A,W}$ = 0.016), and risk is "negligible" for estuarine/marine invertebrates ($RQ_{A,W}$ = 0.10).

The risk to saltwater aquatic vertebrates and invertebrates from multiple routes of exposure was calculated based on the comparison of estimated concentrations accumulated in tissues, from continuous copper exposure via multiple routes to a maximum concentration in ambient receiving water of 0.79 μg/L, with critical body burdens ($CTET_{A,M}$). The resulting $RQ_{A,M}$s are 0.050 and 0.060, respectively, indicating "remote" risk to saltwater aquatic animals from copper accumulated in tissues.

Risk to Birds and Mammals from Copper

Risk to aquatic-dependent birds and mammals from exposure to copper via consumption of prey items or drinking ambient surface water is expected to be low because copper is an essential element and the tendency to bioaccumulate copper in tissues is low (Croteau et al., 2005). The RQ_{wild} for representative freshwater and saltwater mammals and birds based on the comparison of estimated dietary doses with chronic toxicity reference values are 0.0027 and 0.0037 for marine mammals and birds, respectively, and 2.3E-07 and 3.1E-07 for freshwater mammals and birds, respectively. These RQ_{wild} indicate “remote” risk of dissolved copper to aquatic-dependent birds and mammals.

Table 5-26. Summary of Chronic Toxicity Effect Thresholds for Aquatic and Aquatic-Dependent Organisms Exposed to Copper

Direct Exposure to Copper in Surface Water					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Effect Endpoint**	**NOEC - LOEC (µg/L)**	**$CTET_{A,W}$ (µg/L)**	**Study (source)**
Freshwater Vertebrate	Fountain darter, *Etheostoma fonticola*	28-day juvenile F,M/ Survival	–	**4.69***	Mayer et al. 2008
Freshwater Invertebrate	Rotifer, *Brachionus plicatilis*	LC Test - Intrinsic growth rate	2.5 - 5.0	3.5 (**3.9***)	Janssen et al. 1994 (2007 ALC doc.)
Freshwater Plant (Vascular)	Duckweed, *Lemna minor*	7-day EC50	–	119 (**114.2***)	Walbridge 1977 (2007 ALC doc.)
Estuarine/Marine Vertebrate	Sheepshead minnow, *Cyprinodon variegatus*	ELS Test - Growth	172 - 362	249 (**206.7***)	Hughes et al. 1989 (2007 ALC doc.)
Estuarine/Marine Invertebrate	Rotifer, *Brachionus calyciflorus*	96-hr Growth	6.1 - 10.3	**7.9** (dissolved)	Arnold et al. 2010 (Cu Dev. Assoc. work)
Estuarine/Marine Plant	Giant kelp, *Macrocystis pyrifera*	96-hr EC50 – Photosynthesis	–	60 (**49.8***)	Clendenning and North 1959 (2003 Draft ALC Update)
Accumulation of Copper in Tissue from Multiple Exposure Routes					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_{A,T}$ (mg/kg wet wt.)**	**Study (source)**
Freshwater Vertebrate	Rainbow trout, *Oncorhynchus mykiss*	ELS Test	Reduction in growth	3.0	Seim et al. 1984
Freshwater Invertebrate	Cladoceran, *Daphnia magna*	LC Test	25% reduction offspring produced	3.8	Schwartz et al. 2004
Estuarine/Marine Vertebrate	Grey mullet, *Chelon labrosus*	10-wk juvenile dietary exposure	Decreased growth and food intake, induced hepatic lipid	4.59	Baker et al. 1998
Estuarine/Marine Invertebrate	Copepod, *Acartia clausi*	–	Reduced egg production	3.8	Moraitou-Apostolopoulos and Verriopoulos 1979

Table 5-26. Summary of Chronic Toxicity Effect Thresholds for Aquatic and Aquatic-Dependent Organisms Exposed to Copper (Continued)

Dietary Exposure to Copper Through Diet					
Surrogate Aquatic-Dependent Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	$CTET_o$ (mg/kg bw/d)	**Study (source)**
Mammals	Mink	357-d oral in diet	Reproduction	11.7	Sample et al. 1996 (Oregon Toxics BE)
Birds	Chicken	10-wk oral in diet	Growth, mortality	47	Sample et al. 1996 (Oregon Toxics BE)

$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures

$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue

$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)

Bolded entries are for dissolved copper.

*Freshwater data normalized to hardness = 100 mg/L CaCO3 using EPA derived acute slope of 0.8545 and EPA dissolved conversion factor of 0.960; for estuarine/marine data, a dissolved conversion factor of 0.83 was used (EPA 1996)

Table 5-27. Risk Quotients for Copper Based on Comparison of Exposure Concentrations with Chronic Toxicity Effect Thresholds of Representative Surrogate Taxa

Taxa	**EC_W (µg/L)**	**$CTET_{A,W}$ (µg/L)**	**$RQ_{A,W}$**
Freshwater Vertebrate	6.7E-05	4.69	1.4E-05
Freshwater Invertebrate		3.9	1.7E-05
Freshwater Plant (Vascular)		114.2	5.9E-07
Estuarine/Marine Vertebrate	0.79	206.7	0.0038
Estuarine/Marine Invertebrate		7.9	0.10
Estuarine/Marine Plant (Vascular)		49.8	0.016
Taxa	**EC_T (mg/kg wet wt)**	**$CTET_{A,T}$ (mg/kg wet wt)**	**$RQ_{A,M}$**
Freshwater Vertebrate	1.9E-05	3.0	6.5E-06
Freshwater Invertebrate		3.8	5.1E-06
Estuarine/Marine Vertebrate	0.23	4.59	0.050
Estuarine/Marine Invertebrate		3.8	0.060
Taxa	**EC_O (mg/kg bw/d)**	**$CTET_O$ (mg/kg/d)**	**RQ_{wild}**
Mammals (Freshwater)	2.7E-06	11.7	2.3E-07
Birds (Freshwater)	1.5E-05	47	3.1E-07
Mammals (Estuarine/Marine)	0.032	11.7	0.0027
Birds (Estuarine/Marine)	0.17	47	0.0037

EC_W – Exposure concentration for water
EC_T – Exposure concentration accumulated in tissue
EC_O – Exposure concentration via oral ingestion (diet)
$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures
$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue
$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)
$RQ_{A,W}$ – Risk quotient from exposure to copper in water calculated as EC_W/ $CTET_{A,W}$
$RQ_{A,M}$ – Risk quotient from accumulation of copper in tissue calculated as EC_T/ $CTET_{A,T}$
RQ_{Wild} – Risk quotient from exposure to copper through diet calculated as EC_O/ $CTET_O$

5.4.2.3.2 Indirect Effects from Copper

The RQs summarized in Table 5-27 indicate "remote" risk for direct effects of copper to most federally listed aquatic and aquatic-dependent species in this BE. Risk to estuarine/marine invertebrates was determined to be negligible. Furthermore, discharge of copper does not appear to result in appreciable concentrations in modeled estuaries and freshwater receiving water bodies under highly conservative scenarios.

The same RQs used to assess direct effects to listed aquatic and aquatic-dependent species were used to assess indirect effects to available resources (water quality, vegetative cover and prey) and critical habitat. The assessment of risk to federally listed species can be extrapolated to other plants and animals because the effects thresholds apply to the most sensitive aquatic species and, therefore, can conservatively be applied to all species. In other words, the same assessment of risk can be applied to habitat quality or food resources of the listed species. As such, the risk calculations indicate a "remote" (piscivores) or "negligible" (invertivores) risk for indirect effects on listed species due to dietary exposure to copper from vessels of the Armed Forces or due to toxicity-related reductions in the amount of food resources. Therefore, because the information summarized in Table 5-27 indicates that copper at maximum modeled exposure concentrations has only "remote" to "negligible" risk of directly affecting freshwater and saltwater aquatic animals and plants, indirect effects typically associated with exposure, including loss of cover or food resources and changes in water quality parameters, are also not expected for this metal.

5.4.2.3.3 Risk Conclusions for Each Taxonomic Group of Federally Listed Species in the Representative Action Areas from Exposure to Copper

Based on the direct and indirect effects assessment from exposure of vertebrate, invertebrate and wildlife species to copper in Batch Two discharges, the following risk conclusions are being made for each of the listed species groups in the RAAs (see Table 5-28). This information is used to inform decisions in the effects determinations made in Section 8 of this document.

Table 5-28. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Copper Discharged from Vessels of the Armed Forces to Ports and Harbors

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Saltwater Corals	Negligible	• Exposed via water column only • Risk quotient indicates very low ("negligible") potential for direct chronic effects to most marine/estuarine invertebrates • Fate of copper indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • NOECs from studies of copper effects on corals range from 2 - 33.5 µg/L, with most NOECs being 10 µg/L or greater • Risk conclusion for corals is confirmed using the minimum and maximum of the range of coral NOECs. Resulting RQs range from 0.024 - 0.40., with most RQs being 0.08 or lower.
Unionid Mussel	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is checked via estimation using acute study with the mussel (*Actinonaias* spp.) SMAV from the 2007 ALC document of 11.33 µg/L (Keller, unpublished) and dividing by the FACR of 3.22, which results in a chronic value of 3.519 µg/L. Resulting RQ is 1.9E-05.
Freshwater Snail	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion confirmed using a chronic study of *Lymnaea stagnalis* with a NOEC for hemolymph osmality of 12 µ/L (Brix et al., 2012). Resulting RQ is 5.6E-08.
Saltwater Mollusk	Negligible	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed via estimation using 7-day study with blue mussel (*Mytilus edulis*) and a LOEC (mortality) of 200 µg/L (167.6 µg/L dissolved; Scott and Major, 1972). The LOEC was divided by the FACR of 3.22, which resulted in a NOEC of 62.1. Resulting RQ is 0.012.

Table 5-28. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Copper Discharged from Vessels of the Armed Forces to Ports and Harbors - Contined

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Freshwater Shrimp/ Crustacean	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Surrogate CTET is considered representative and appropriate
Freshwater Fish/ Inland Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a chronic study with rainbow trout (*O. mykiss*) and a MATC (growth) of 7.29 µg/L (normalized to 100 mg/L on a dissolved basis) (Mayer et al., 2008). Resulting RQ is 9.2E-06.
Anadromous Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed for saltwater/estuarine exposures using a study of sub-lethal effects on migrating Atlantic salmon that showed copper concentrations as low as 16.8 µg/L will inhibit upstream migration and lead to reverse downstream migrations (Sprague et al. 1965, Saunders and Sprague 1967 and Hecht et al. 2007 as cited in Price 2013). Resulting RQ is 0.047.
Freshwater Fish/Inland Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed via estimation using acute toxicity data with the shovelnose sturgeon (*Scaphirhynchus platorynchus*) from the 2007 ALC doc., which has an SMAV of 69.63 µg/L (Dwyer et al., 1999). Dividing by the FACR of 3.22 results in a chronic value of 21.62 µg/L. Calculation results in $RQ_{A,W}$ of approximately 3.1E-06.

Table 5-28. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Copper Discharged from Vessels of the Armed Forces to Ports and Harbors - Contined

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Anadromous Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • No chronic toxicity data were identified for sturgeon tested in seawater.
Estuarine/Marine Fish	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • RQ directly applicable because the toxicity threshold is for *L. variegatus* as presented in Table 5-26 above
Beetle and Aquatic Insect	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted •
Amphibian	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using acute toxicity data for the boreal toad (*Bufo boreas*) from the EPA 2007 ALC document, which has an SMAV of 47.49 µg/L (Dwyer et al., 1999). Dividing by the FACR of 3.22 results in a chronic value of 14.75 µg/L. Resulting RQ is 4.5E-07.
Snakes and Other Reptiles	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Surrogate CTET is representative and appropriate
Sea Turtle	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted

Table 5-28. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Copper Discharged from Vessels of the Armed Forces to Ports and Harbors - Contined

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Coastal/Marine Bird	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted
Marine Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted
Terrestrial Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted
Seagrass	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (change in habitat/water quality parameters) at the concentrations predicted
Freshwater - Saltwater Aquatic and Wetland Plants	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (change in habitat/water quality parameters) at the concentrations predicted • RQ directly applicable because the toxicity threshold is for *L. minor* as presented in Table 5-26 above

5.4.2.4 Iron

Iron is found at levels exceeding screening level benchmarks in firemain discharge, surface vessel bilgewater/OWS effluent, and hull coating leachate from most vessels of the Armed Forces (see Section 3.2.3). By weight, iron is the second most abundant metal and fourth most abundant element in the earth's crust (Taylor 1964 as cited in Xing and Liu 2011). It is an important component of many soils, and iron in water may be present in variable quantities dependent upon the geology of the area and other chemical 'components of the waterway (EPA, 1986). Under natural conditions, iron in surface water primarily comes from the products of weathered rocks and soil around watersheds, controlled by many factors, such as geological process, soil composition, environmental temperature, precipitation, and hydrology (Harris, 1992 as cited in Xing and Liu 2011). Air deposition in another important source of iron to surface water bodies (Winchester and Nifong, 1971; Semb et al., 1995; Ganor et al., 2000; Herut et al., 2001 as cited in Xing and Liu 2011). Because of its low solubility, its concentration in surface water is generally low (Molot and Dillon, 2003 as cited in Xing and Liu 2011).

Iron is an essential trace element required by both plants and animals and affects the biogeochemical cycles of many important elements such as carbon, nitrogen, phosphorus, and various trace metals. It is a micronutrient required for proteins involved in fundamental cellular processes, including both photosynthesis and respiration (Raven et al., 1999). It is a vital oxygen transport mechanism in the blood of all vertebrate and some invertebrate animals (EPA, 1986).

In some waters, iron is a limiting factor for the growth of algae and other plants. This is true especially in some marl lakes where it is precipitated by the highly alkaline conditions (EPA, 1986). Iron (Fe) is an essential micronutrient for marine organisms, and it is now well established that low Fe availability controls phytoplankton productivity, community structure, and ecosystem functioning in vast regions of the global ocean. Approximately 30% of surface waters in the open ocean are known as high nutrient low chlorophyll (HNLC) regions (Boyd et al., 2007). These areas have sufficient levels of the macronutrients nitrate and phosphate, but present lower phytoplankton biomass, in terms of chlorophyll concentrations, than expected from residual macronutrient concentrations. The restriction of phytoplankton growth in these regions is now acknowledged to be the result of iron (Fe) limitation (Martin and Fitzwater, 1988; Boyd et al., 2007).

Iron in the environment occurs in valency states ranging from 2+ to 6+. The ferrous, or bivalent ($Fe2+$ or Fe(II)), and the ferric, or trivalent ($Fe3+$ or Fe(III)) irons, are the primary forms of iron in the aquatic environment, although other forms may be in organic and inorganic wastewater streams. The ferrous ($Fe2+$) form can persist in waters void of dissolved oxygen and usually originates from groundwater or mines when these are pumped or drained. It exists usually as a dissolved ion, although, in the presence of high carbonate, sulphide and orthophosphate levels, it forms insoluble salts. In oxic waters, ferrous iron ($Fe2+$) rapidly oxidizes to ferric iron ($Fe3+$) and, at neutral pH, it forms highly insoluble colloidal Fe oxides and Fe hydroxide precipitates. For practical purposes the ferric form is insoluble. Iron can exist in natural organometallic or humic compounds and colloidal forms. (Xing and Liu, 2011; EPA, 1986)

Iron speciation is highly affected by the chemical composition of surface water, iron inputs and removal processes, as well as internal recycling. The physicochemical speciation of iron, which profoundly influences bioavailability, depends on the relative importance of various competing processes including adsorption-desorption, precipitation-dissolution, ion exchange, complexation-dissociation, and redox reactions. Each of these processes is likely influenced by organic Fe-binding ligands, which complex more than 99% of dissolved Fe. (Gledhill and Buck, 2012; Xing and Liu, 2011)

Iron in surface water is separated into three size fractions: particulate iron (>0.22 μm), colloidal iron (0.025-0.22 μm) and soluble iron (<0.025 μm). The highly reactive colloidal iron may either coagulate or flocculate to form larger particles, or become more soluble, and the formation of colloidal and larger, more refractory iron particles provides a mechanism for removing dissolved iron and other trace metals from the water by adsorption and co-precipitation. (Xing and Liu, 2011) The lowest concentrations of dissolved Fe occur in the open oceans (0.017 to 0.022 μg/L) while concentrations for coastal waters are higher (0.20 to 0.60 μg/L) (Kuma et al. 1998 as cited in Leigh-Smith, 2017), largely due to anthropogenic sources. Dissolved iron concentrations in open ocean surface waters are typically below 0.011 μg/L. However, biotic ligands have the ability to solubilize natural iron, subsequently resulting in increased productivity. (De Baar and DeJong, 2001; Boyd and Ellwood, 2010)

In unpolluted oceanic seawater, concentrations of iron between 2.8-29 ng/L and 224-1,228 ng/L have been reported, although higher concentrations may be found in estuarine waters (Whitehouse et al 1998). Iron concentrations in saltmarsh sediments are frequently much higher than those occurring in the overlying waters. Mean concentrations as high as 20,800 mg/kg appear to be tolerated in coastal saltmarshes which are designated as 'healthy'. It has been shown that iron concentrations are higher in the immediate vicinity of saltmarsh plant roots and in the burrow walls produced by organisms, such as *Arenicola* (Grimwood and Dixon, 1997).

Many metals, including iron, are biologically essential element but can also be toxic. A significant proportion of biological electron transport systems are based on Fe(II)/Fe(III) redox reactions (Rathgeb et al., 2016 as cited in Leigh-Smith, 2017). However, if concentrations exceed physical requirements, Fe may become toxic, acting as an enzyme inhibitor (Frías-Espericueta et al., 2003 as cited in Leigh-Smith, 2017), and causing effects that may affect other biological activities such as behavior, growth and reproduction (Zhou et al., 2014 as cited in Leigh-Smith, 2017). Iron can be acutely toxic at concentrations on the order of mg/L. In general, the toxicity of iron seems to be higher under acidic conditions where Fe (II) predominates. Acute toxic effects in laboratory bioassays have been observed for concentrations as low as 0.32 mg/L; however, the EPA WQC for freshwater aquatic life based on field observations is 1.0 mg/L. Concentrations as low as 2 mg/L can cause gill damage in fish, and elevated cellular concentrations of iron may cause cell degeneration. In most laboratory bioassays, the toxic effects of iron have been attributed to the motion-inhibiting or smothering effects of Fe-hydroxide or Fe-humic precipitates on gills, eggs, or other surfaces. Because these toxic modes of action limit access to essential resources, such as oxygen or food, effects are considered by toxicologists to be indirect, while effects such as reduced cellular function would be considered direct effects. (Vuori, 1995) For the purposes of this BE and by definition under ESA, all of these effects are considered to be direct effects because there is an immediate population-level response.

Marine organisms accumulate iron but also rapidly excrete iron in clean water conditions. Tissue concentrations of iron are typically related to the water and sediment concentrations, but there is considerable variability. The bioaccumulation of iron by marine organisms does not appear to pose a hazard to higher trophic levels (Mance and Campbell, 1988).

5.4.2.4.1 Direct Effects from Iron

Table 5-29 summarizes the chronic toxicity values available for calculating the risk quotients for aquatic species exposed to dissolved iron in water ($CTET_{A,W}$). Table 5-30 presents the maximum exposure concentrations estimated in ambient receiving waters (EC_W) and the RQs calculated as the ratio of EC_W to the corresponding CTET for each taxonomic group. Maximum exposure concentrations in ambient receiving water are estimated from the harbor modeling, as described above in Section 5.2.1.

Risk to Freshwater Aquatic Animal and Plant Species from Iron

Concentrations of iron in freshwater harbors resulting from discharges from vessels of the Armed Forces are expected to be low (approximately 2.3E-06 µg/L). Because iron is an essential element and because iron in oxygenated surface water is not expected to be soluble, the risk to freshwater aquatic organisms from exposure to iron concentrations in ambient receiving water resulting from discharges from vessels of the Armed Forces is also expected to generally be low. The $RQ_{A,W}$ for freshwater vertebrates, invertebrates, and plants range from 2.3E-09 to 7.3E-08, and indicate only "remote" risk of adverse effects (Table 5-30).

Because iron is an essential element, it has the potential to bioaccumulate. Like other essential metals, aquatic organism exposure to iron concentrations in excess of nutritional needs via other potential routes of exposure in addition to direct waterborne toxicity (e.g. dietary toxicity) may pose a threat to listed species. While iron does not bioaccumulate to high levels in aquatic animals, the EPA and DoD believe it is prudent to incorporate an analysis of multiple routes of exposure in the analysis of this pollutant. The evaluation of potential risks from bioaccumulated chromium provides a means of identifying potential risks to listed species from all exposure routes combined. An $RQ_{A,M}$ was calculated for freshwater vertebrates based on comparison of estimated concentrations accumulated in tissues from continuous exposure to a modeled concentration of iron in ambient receiving waters of 2.3E-06 µg/L, resulting in an estimated tissue concentration of 2.2E-07 mg/kg (see Table 5-30). The resulting RQ is 2.4E-08, indicating "remote" risk to freshwater aquatic vertebrates from iron accumulated in tissues. The tissue concentrations of iron in freshwater invertebrates was estimated to be 4.1E-05 mg/kg resulting in an $RQ_{A,M}$. of 1.7E-07, also indicating "remote" risk.

Risk to Saltwater (Estuarine/Marine) Aquatic Animal and Plant Species from Iron

Maximum iron concentrations in estuarine/marine harbors from discharges from vessels of the Armed Forces are estimated to be three orders of magnitude higher than freshwater harbors (0.038 µg/L). Consequently, the risk to saltwater aquatic organisms from maximum exposure to iron in ambient receiving water is greater than the same risk predicted for freshwater organisms based on RQs. However, risk is still “remote” for saltwater vertebrates ($RQ_{A,W}$ = 0.014) and invertebrates ($RQ_{A,W}$ = 0.00027).

No CTET was identified for estuarine/marine vascular plants, but because of the behavior of iron in marine coastal environments, risk to vascular plants is also expected to be “remote”. Concentrations of iron from vessels of the Armed Forces predicted for coastal harbors will not form enough iron hydroxide precipitates to coat seagrass or wetland plant shoots enough to be harmful. Several studies have shown that the addition of free iron to shallow water habitats can reduce the loss of seagrass habitat by ameliorating the deleterious effects of sulfide production resulting from microbial degradation processes (Chambers and Fourqurean, 2000; Ruiz-Halpern et al., 2008). For several decades, scientists have researched and proposed iron fertilization of the open ocean, where iron is a limiting element, to stimulate phytoplankton production that could lead to the sequestration of atmospheric carbon dioxide and the recovery of fisheries.

The $RQ_{A,M}$ for saltwater aquatic invertebrates based on estimated concentrations accumulated in tissues from continuous iron exposure via multiple routes to a maximum concentration in ambient receiving water of 0.038 μg/L is approximately 0.001, indicating “remote” risk to saltwater aquatic animals from iron accumulated in tissues. A $CTET_{A,T}$ for saltwater aquatic vertebrates was not identified, and risk to vertebrates from multiple routes of exposure could not be evaluated. However, if effects thresholds for estuarine/marine vertebrates are similar to those for freshwater vertebrates (minimum NOEC of 9 mg/kg), based on an estimated body burden of 0.0036, risks to estuarine/marine vertebrates are also “remote” (potential $RQ_{A,M}$ of 0.0004).

Risk to Birds and Mammals from Iron

Risk to aquatic-dependent birds and mammals from exposure to iron via consumption of prey items or drinking ambient surface water is expected to be low because iron is an essential element and the tendency to bioaccumulate iron in tissues is expected to be low. The RQ_{wild} for representative freshwater and saltwater mammals and birds based on the chronic toxicity values in relation to the dietary exposure concentrations calculated for this BE are 2.0E-06 and 2.7E-05 for marine mammals and birds, respectively, and 1.2E-10 and 1.6E-09 for freshwater mammals and birds, respectively. These RQ_{wild} indicate “remote” risk from dietary exposure to iron for aquatic-dependent birds and mammals.

Table 5-29. Summary of Chronic Toxicity Effect Thresholds for Aquatic Organisms Directly Exposed to Iron

Direct Exposure to Iron in Surface Water					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Effect Endpoint**	**NOEC - LOEC (μg/L)**	**$CTET_{A,W}$ (μg/L)**	**Study (source)**
Freshwater Vertebrate	Fathead minnow, *Pimephales promelas*	ELS Test - Survival	–	320	Birge et al. 1985
Freshwater Invertebrate	Rotifer, *Lecane quadridentata*	ELS Test - Survival	10 - 100	31.6	Guzman et al. 2010

Table 5-29. Summary of Chronic Toxicity Effect Thresholds for Aquatic Organisms Directly Exposed to Iron (Continued)

Direct Exposure to Iron in Surface Water					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Effect Endpoint**	**NOEC - LOEC (µg/L)**	**$CTET_{A,W}$ (µg/L)**	**Study (source)**
Freshwater Plant (Vascular)	Common Reed, *Phragmites australis*	64-day Growth Test	–	1000	Batty and Younger 2002
Estuarine/Marine Vertebrate	Mudskipper, *Periophthalmus waltoni*	96-hr Survival	–	2.69	Bu-Olayan and Thomas 2008
Estuarine/Marine Invertebrate	Rock oyster, *Saccostrea glomerata*	ELS Test - Development		141.4	Wilson and Hyne 1997
Estuarine/Marine Plant	Not available	–	–	–	–
Accumulation of Iron in Tissue from Multiple Exposure Routes					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_{A,T}$ (mg/kg wet wt.)**	**Study (source)**
Freshwater Vertebrate	Brown trout, *Salmo trutta*	ELS Test	Egg mortality NOEC	9	Anderson 1997
Freshwater Invertebrate	Cladoceran, *Daphnia magna*	ELS Test	EC20 for brood success	250	Van et al. 2002
Estuarine/Marine Vertebrate	Not available	–	–	–	–
Estuarine/Marine Invertebrate	Blue mussel, *Mytilus edulis*	Field	Growth NOEC	68	St-Jean et al. 2003
Dietary Exposure to Iron					
Surrogate Aquatic-Dependent Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_o$ (mg/kg bw/d)**	**Study (source)**
Mammals	Norway rat, *Rattus norvegicus*	61-day dietary exposure of young rats	Growth and morphology NOEL	140	Appel et al. 2001
Birds	Domestic chicken, *Gallus domesticus*	21-day dietary exposure of immature birds	Growth NOEL	100	Jensen and Maurice 1978

$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures

$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue

$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)

Table 5-30. Risk Quotients for Iron Based on Comparison of Exposure Concentrations with Chronic Toxicity Effect Thresholds of Representative Surrogate Taxa

Taxa	EC_W (µg/L)	$CTET_{A,W}$ (µg/L)	$RQ_{A,W}$
Freshwater Vertebrate	2.3E-06	320	7.2E-09
Freshwater Invertebrate		31.6	7.3E-08
Freshwater Plant (Vascular)		1000	2.3E-09
Estuarine/Marine Vertebrate	0.038	2.69	0.014
Estuarine/Marine Invertebrate		141.4	0.00027
Estuarine/Marine Plant (Vascular)		-	Not calculated
Taxa	**EC_T (mg/kg wet wt)**	**$CTET_{A,T}$ (mg/kg wet wt)**	**$RQ_{A,M}$**
Freshwater Vertebrate	2.2E-07	9	2.4E-08
Freshwater Invertebrate	4.1E-05	250	1.7E-07
Estuarine/Marine Vertebrate	0.0036	-	Not calculated
Estuarine/Marine Invertebrate	0.68	68	0.01
Taxa	**EC_O (mg/kg bw/d)**	**$CTET_O$ (mg/kg/d)**	**RQ_{wild}**
Mammals (Freshwater)	1.7E-08	140	1.2E-10
Birds (Freshwater)	1.6E-07	100	1.6E-09
Mammals (Estuarine/Marine)	2.9E-04	140	2.0E-06
Birds (Estuarine/Marine)	0.0027	100	2.7E-05

EC_W – Exposure concentration for water
EC_T – Exposure concentration accumulated in tissue
EC_O – Exposure concentration via oral ingestion (diet)
$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures
$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue
$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)
$RQ_{A,W}$ – Risk quotient from exposure to iron in water calculated as $EC_W / CTET_{A,W}$
$RQ_{A,M}$ – Risk quotient from accumulation of iron in tissue calculated as $EC_T / CTET_{A,T}$
RQ_{Wild} – Risk quotient from exposure to iron through diet calculated as $EC_O / CTET_O$

5.4.2.4.2 Indirect Effects from Iron

The RQs summarized in Table 5-30 indicate "remote" risk for direct effects of iron to federally listed aquatic species evaluated in this BE. Furthermore, discharge of iron does not appear to result in appreciable concentrations in modeled estuaries and freshwater receiving water bodies under highly conservative scenarios.

The same RQs used to assess direct effects of iron to federally listed aquatic species were used to assess indirect effects to available resources (water quality, vegetative cover, and prey). The assessment of risk to federally listed species can be extrapolated to other plants and animals because the effects thresholds apply to the most sensitive aquatic species and, therefore, can conservatively be applied to all species. In other words, the same assessment of risk can be applied to habitat quality or food resources of the listed species. As such, the risk calculations indicate a "remote" risk for indirect effects on listed species due to dietary exposure to iron from vessels of the Armed Forces or due to toxicity-related reductions in the amount of food resources and habitat. Therefore, because the information summarized in Table 5-30 indicates that iron at maximum modeled exposure concentrations has only "remote" risk of directly affecting freshwater and saltwater aquatic animals and plants, indirect effects typically associated with exposure, including loss of cover and changes in water quality parameters, are also not expected for this metal.

5.4.2.4.3 Risk Conclusions for Each Taxonomic Group of Federally Listed Species in Representative Action Areas from Exposure to Iron

Based on the direct and indirect effects assessment from exposure of vertebrate, invertebrate and wildlife species to iron in discharges from vessels of the Armed Forces, the following risk conclusions are being made for each of the listed species groups in the RAAs (see Table 5-31). This information is used to inform decisions in the effects determinations made in Section 8 of this document.

Table 5-31. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Iron Discharged from Vessels of the Armed Forces to Ports and Harbors

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Saltwater Corals	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects to most marine/estuarine invertebrates • Fate of iron indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Surrogate CTET is representative and appropriate • Toxicity studies with iron are limited; however, a recent study of iron toxicity with scleractinian corals resulted in LC50s ranging from 25 to 60 mg/L (Smith et al. 2017). • Risk conclusion is checked using an ELS study of fertilization success with *Acropora spathulata* resulting in a lowest EC10 of 5 mg/L total Fe (0.08 mg/L dissolved Fe) (Leigh-Smith et al. 2017). Resulting RQ is 0.000475.
Unionid Mussel	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a 24-hr sublethal study of filtration with the zebra mussel (*Dreissena polymorpha*) which resulted in an EC1 of 410 µg/L (Kováts et al., 2012). A chronic value of 49.4 was estimated by applying an ACR of 8.3 (Raimondo et al. 2007). Resulting RQ is 4.7E-08.

Table 5-31. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Iron Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Freshwater Snails	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion confirmed using an acute study of *Melanoides tuberculate* with a lower 95% confidence interval on the mean LC50 of 1.58 mg/L (Shuhaimi-Othman et al., 2012). Using the safety factor of 2 used to calculate the CMC from the FAV (i.e., divide by 2) and an ACR of 8.3 (Raimondo et al., 2007) results in a chronic value of 0.095 mg/L. Resulting RQ is 2.4E-08.
Saltwater Mollusk	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • The RQ is directly applicable because the CTET is for *Saccostrea glomerata* (rock oyster), as shown in Table 5-29 above.
Freshwater Shrimp/ Crustacean	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using an acute study with the freshwater prawn *Macrobrachium lanchesteri* that resulted in an LC50 of 0.4 mg/L (Shuhaimi-Othman et al., 2012). Applying an ACR of 8.3 results in a chronic value of 48.2 µg/L. Resulting RQ is 4.8E-08.
Freshwater Fish/ Inland Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a chronic study with brown trout (*Salmo truta*) and LOEC of 56.77 µg/L (Reader et al. 1989). Resulting RQ is 4.1E-08.

Table 5-31. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Iron Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Anadromous Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • No chronic toxicity data were identified for salmonids tested in seawater
Freshwater Fish/Inland Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • No chronic data were identified for inland/freshwater sturgeon.
Anadromous Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • No chronic data were identified for anadromous sturgeon tested in saltwater.
Estuarine/Marine Fish	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • RQ directly applicable because the toxicity threshold is for *P. waltoni* as presented in Table 5-29 above
Beetle and Aquatic Insect	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using data from a field study of taxonomic richness of aquatic insects at varying concentrations of Fe (II) which showed a decrease in taxonomic richness at 200 µg/L (Rasmussen and Lindegaard 1988). Resulting RQA,W is 1.2E-08.

Table 5-31. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Iron Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Amphibian	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of copper indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Exposure through diet is not expected to be excessive • No chronic toxicity studies of effects of iron on amphibians were identified.
Snakes and Other Reptiles	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Exposure through diet is not expected to be excessive • Surrogate CTET is representative and appropriate • No chronic toxicity studies of effects of iron on reptiles were identified.
Sea Turtle	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Iron is an essential element, and exposure through diet is not expected to be excessive
Coastal/Marine Bird	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Iron is an essential element, and exposure through diet is not expected to be excessive
Marine Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Iron is an essential element, and exposure through diet is not expected to be excessive

Table 5-31. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Iron Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Terrestrial Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Iron is an essential element, and exposure through diet is not expected to be excessive
Seagrass	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (change in habitat/water quality parameters) at the concentrations predicted
Freshwater - Saltwater Aquatic and Wetland Plants	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of iron indicates very low potential for indirect effects (change in habitat/water quality parameters) at the concentrations predicted • RQ directly applicable because the toxicity threshold is for *P. australis* as presented in Table 5-29 above

5.4.2.5 Lead

Dissolved lead is found at levels exceeding screening level benchmarks in deck runoff and graywater discharge from vessels of the Armed Forces (see Section 3.2.3). Lead is a naturally occurring, ubiquitous compound in freshwater present at concentrations generally less than 3 µg/L in most streams (Moore and Ramamoorthy, 1984). Measured concentrations in rivers and streams of the United States averaged between 5 and 23 µg/L nationwide (Eisler, 1988). Higher concentrations are associated either with anthropogenic sources or occur in highly mineralized regions.

The solubility of lead compounds in water depends heavily on pH and ranges from about 10,000,000 µg/L of lead at pH 5.5 to 1 µg/L at pH 9.0 (Hem and Durum, 1973). The bioavailability of lead increases in environments with low pH, low organic content, and low metal salt content (Eisler, 1988). Invertebrates tend to have higher lead BCFs than vertebrates (USEPA, 1984b).

The toxicity of lead to aquatic organisms varies with water temperature, pH, water hardness, metal salt concentrations, organic matter, and suspended solid concentration (USEPA, 1984b). Organic lead compounds are generally more toxic than inorganic (Eisler, 1988). Because lead toxicity to freshwater aquatic animals has been shown to be related to water hardness, it is appropriate to normalize chronic toxicity effect thresholds for freshwater aquatic animals to a standard water hardness (i.e., 100 mg/L as $CaCO_3$) for comparative purposes and to support risk calculation (EPA, 2005a).

5.4.2.5.1 Direct Effects from Lead

Table 5-32 summarizes the chronic toxicity values available for calculating the risk quotients for aquatic species exposed to dissolved lead in water ($CTET_{A,W}$). Table 5-33 presents the total lead exposure concentrations estimated in ambient receiving waters (EC_W) and the RQs calculated as the ratio of EC_W to the corresponding CTET for each taxonomic group. Maximum exposure concentrations in ambient receiving water are estimated from the harbor modeling, as described above in Section 5.2.1.

Risk to Freshwater Aquatic Animal and Plant Species from Lead

Concentrations of lead in freshwater harbors resulting from discharges from vessels of the Armed Forces are expected to be low (approximately 3.0E-07 µg/L). The $RQ_{A,W}$ for freshwater vertebrates, invertebrates, and plants from exposure to lead range from 3.8E-11 for plants to 5.0E-08 for invertebrates and indicate only “remote” risk of adverse effects (Table 5-33).

While lead does not bioaccumulate to high levels in aquatic animals, the EPA and DoD believe it is prudent to evaluate risk from multiple routes of exposure in this BE. The evaluation of potential risks from bioaccumulated lead provides a means of identifying potential risks to listed aquatic species from all exposure routes combined. RQs were calculated for freshwater vertebrates and invertebrates based on estimated concentrations of lead accumulated in tissues from continuous exposure to lead in ambient receiving waters at 3.0E-07 µg/L (see Table 5-33). The resulting RQs for aquatic vertebrates and invertebrates were 1.2E-07 and 3.5E-09, respectively, indicating “remote” risk to freshwater aquatic animals from lead accumulated in tissues.

Risk to Saltwater (Estuarine/Marine) Aquatic Animal and Plant Species from Lead

Maximum lead concentrations in estuarine/marine harbors from discharges from vessels of the Armed Forces are estimated to be three orders of magnitude higher than freshwater harbors (8.2E-04 µg/L). Consequently, the risk to saltwater aquatic organisms from maximum exposure to iron in ambient receiving water is greater than the same risk predicted for freshwater organisms based on RQs. However, risk is still “remote” for saltwater vertebrates ($RQ_{A,W}$ = 0.000072), invertebrates ($RQ_{A,W}$ = 0.000034), and plants ($RQ_{A,W}$ = 0.0001). The $RQ_{A,M}$ for saltwater aquatic vertebrates and invertebrates based on estimated concentrations accumulated in tissues from continuous lead exposure via multiple routes are 0.00032 and 1.5E-07, respectively, indicating “remote” risk to saltwater aquatic animals from lead accumulated in tissues.

Risk to Birds and Mammals from Lead

Risk to aquatic-dependent birds and mammals from exposure to lead via consumption of prey or drinking ambient surface water was evaluated because of the potential for prey to bioaccumulate lead. The RQ_{wild} for representative freshwater and saltwater mammals and birds based on comparison of dietary exposure concentrations calculated for this BE with effects thresholds are

1.3 E-06 and 8.4E-05 for marine mammals and birds, respectively, and 4.7E-10 and 3.1E-08 for freshwater mammals and birds, respectively. These RQ_{wild} indicate “remote” risk from dietary exposure to lead for aquatic-dependent birds and mammals.

Table 5-32. Summary of Chronic Toxicity Effect Thresholds for Aquatic Organisms Directly Exposed to Lead

Direct Exposure to Lead in Surface Water					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure/Effect Endpoint**	**NOEC - LOEC (µg/L)**	**$CTET_{A,W}$ (µg/L)**	**Study (source)**
Freshwater Vertebrate	Rainbow trout, *Onchorhynchus mykiss*	ELS Test - Development	13.2 – 27	93.62 (**93.62***)	Goettl et al. 1972; Davies and Everhart 1973; Davies et al. 1976 (1984b ALC document)
Freshwater Invertebrate	Amphipod, *Hyalella azteca*	LC Test	7.9 – 18	11.92 (**6.05***)	Besser et al. 2005a (NODA review)
Freshwater Plant (Vascular)	Duckweed, *Lemna minor*	96-hour EC50 - Growth	–	8000	Wang 1986
Estuarine/Marine Vertebrate	Cabezon, *Scorpaenichthys marmoratus*	Acute LC50 ÷ Estuarine ACR of 124.8	1,500 ÷ 124.8	12.02 (**11.43***)	Dinnel et al. 1989 (LC50); Lussier et al. 1985 (ACR)
Estuarine/Marine Invertebrate	Mysid, *Americamysis bahia*	LC Test - Reproduction	17 – 37	25.08 (**23.85***)	Lussier et al. 1985 (1984 ALC document)
Estuarine/Marine Plant	Red algae, *Champia parvula*	–	–	8.1	CCC from USEPA 1980 (ALC document)
Accumulation of Lead in Tissue from Multiple Exposure Routes					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_{A,T}$ (mg/kg wet wt.)**	**Study (source)**
Freshwater Vertebrate	Brook trout, *Salvelinus fontinalis*	3 generation laboratory study	Reduced hatchability and survival of third generation fish	0.40	Holcombe et al. 1976
Freshwater Invertebrate	Amphipod, *Hyalella azteca*	42-day concurrent exposure in water and diet	Fewer offspring produced	4.20	Besser et al. 2005a
Estuarine/Marine Vertebrate	Not available	–	–	0.40	Holcombe et al. 1976
Estuarine/Marine Invertebrate	Cockle, *Cerastoderma edule*	–	Partial inhibition of burrowing	260	Amiard et al. 1986

Table 5-32. Summary of Chronic Toxicity Effect Thresholds for Aquatic Organisms Directly Exposed to Lead - Contintued

Dietary Exposure to Lead					
Surrogate Aquatic-Dependent Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_o$ (mg/kg bw/d)**	**Study (source)**
Mammals	Norway rat, *Rattus norvegicus*	3 generations (>1 year) oral in diet	Reproduction	8	Sample et al. 1996 (Oregon Toxics BE)
Birds	Japanese quail, *Coturnix japonica*	12-week oral in diet	Reproduction	1.13	Sample et al. 1996 (Oregon Toxics BE)

$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures

$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue

$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)

*Normalized using EPA pooled slope of 1.442 and using dissolved conversion factor of 1.46203-[(ln hardness)(0.145712)] freshwater and 0.951 estuarine/marine

Bolded entries are for dissolved lead.

Table 5-33. Risk Quotients for Lead Based on Comparison of Exposure Concentrations with Chronic Toxicity Effect Thresholds of Representative Surrogate Taxa

Taxa	**EC_W (µg/L)**	**$CTET_{A,W}$ (µg/L)**	**$RQ_{A,W}$**
Freshwater Vertebrate	3.0E-07	93.62	3.2E-09
Freshwater Invertebrate		6.05	5.0E-08
Freshwater Plant (Vascular)		8,000	3.8E-11
Estuarine/Marine Vertebrate	0.00082	11.43	7.2E-05
Estuarine/Marine Invertebrate		23.85	3.4E-05
Estuarine/Marine Plant (Vascular)		8.1	0.0001
Taxa	**EC_T (mg/kg wet wt)**	**$CTET_{A,T}$ (mg/kg wet wt)**	**$RQ_{A,M}$**
Freshwater Vertebrate	4.7E-08	0.40	1.2E-07
Freshwater Invertebrate	1.5E-08	4.2	3.5E-09
Estuarine/Marine Vertebrate	0.00013	0.40	0.00032
Estuarine/Marine Invertebrate	0.00004	260	1.5E-07
Taxa	**EC_O (mg/kg bw/d)**	**$CTET_O$ (mg/kg/d)**	**RQ_{wild}**
Mammals (Freshwater)	3.7E-09	8	4.7E-10
Birds (Freshwater)	3.5E-08	1.13	3.1E-08
Mammals (Estuarine/Marine)	1.0E-05	8	1.3E-06
Birds (Estuarine/Marine)	9.5E-05	1.13	8.4E-05

EC_W – Exposure concentration for water
EC_T – Exposure concentration accumulated in tissue
EC_O – Exposure concentration via oral ingestion (diet)
$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures
$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue
$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)
$RQ_{A,W}$ – Risk quotient from exposure to lead in water calculated as EC_W/ $CTET_{A,W}$
$RQ_{A,M}$ – Risk quotient from accumulation of lead in tissue calculated as EC_T/ $CTET_{A,T}$
RQ_{Wild} – Risk quotient from exposure to lead through diet calculated as EC_O/ $CTET_O$

5.4.2.5.2 Indirect Effects from Lead

The RQs summarized in Table 5-33 indicate “remote” risk for direct effects of lead to federally listed aquatic species evaluated in this BE. Furthermore, discharge of lead does not appear to result in appreciable concentrations in modeled estuaries and freshwater receiving water bodies under highly conservative scenarios.

The same RQs used to assess direct effects of lead to federally listed aquatic species were used to assess indirect effects to available resources (water quality, vegetative cover, and prey). The assessment of risk to federally listed species can be extrapolated to other plants and animals because the effects thresholds apply to the most sensitive aquatic species and, therefore, can conservatively be applied to all species. In other words, the same assessment of risk can be applied to habitat quality or food resources of the listed species. As such, the risk calculations indicate a “remote” risk for indirect effects on listed species due to dietary exposure to lead from vessels of the Armed Forces or due to toxicity-related reductions in the amount of food resources and habitat. Therefore, because the information summarized in Table 5-33 indicates that lead at maximum modeled exposure concentrations has only “remote” risk of directly affecting freshwater and saltwater aquatic animals and plants, indirect effects typically associated with exposure, including loss of cover and changes in water quality parameters, are also not expected for this metal.

5.4.2.5.3 Risk Conclusions for Each Taxonomic Group of Federally Listed Species in Representative Action Areas from Exposure to Lead

Based on the direct and indirect effects assessment from exposure of vertebrate, invertebrate and wildlife species to lead in discharges from vessels of the Armed Forces, the following risk conclusions are being made for each of the listed species groups in the RAAs (see Table 5-34). This information is used to inform decisions in the effects determinations made in Section 8 of this document.

Table 5-34. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Lead Discharged from Vessels of the Armed Forces to Ports and Harbors

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Saltwater Corals	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects to most marine/estuarine invertebrates • Fate of lead indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Surrogate CTET is representative and appropriate • Risk conclusion is checked using an ELS study of larval motility with *Goniastrea aspera* that resulted in a 72-h EC50 value 2900 µg/L (Reichelt-Brushett and Harrison, 2004). Dividing this by an assessment factor of 10 results in a predicted no effect concentration (PNEC) of 290 µg/L. Resulting RQ is 2.8E-06.
Unionid Mussel	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a 28-day chronic study of with the fatmucket (*Lampsilis siliquoidea*) which resulted in an NOEC of 10 µg/L (Wang et al., 2010). Resulting RQ is 3.0E-08.
Freshwater Snail	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion confirmed using an acute study of *Melanoides tuberculate* with a lower 95% confidence interval on the mean LC50 of 2.89 mg/L (Shuhaimi-Othman et al., 2012). Using the safety factor of 2 used to calculate the CMC from the FAV (i.e., divide by 2) and an ACR of 8.3 (Raimondo et al., 2007) results in a chronic value of 0.17 mg/L. Resulting RQ is 1.8E-09.
Saltwater Mollusk	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using an ELS study of lead on the development of *Mytilus edulis* larvae which resulted in an EC50 for abnormal development of 476 µg/L. Dividing this by an assessment factor of 10 results in a PNEC of 47.6. Resulting RQ is 1.7E-05.

Table 5-34. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Lead Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Freshwater Shrimp/ Crustacean	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Surrogate CTET is representative and appropriate
Freshwater Fish/ Inland Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is directly applicable because the CTET uses is for *O. mykiss* as presented in table 5-32 above.
Anadromous Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • No chronic toxicity data were identified for salmonids tested in seawater.
Freshwater Fish/Inland Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • No chronic data were identified for inland/freshwater sturgeon. However, a chronic study with juvenile white sturgeon (*Acipenser transmontanus*) resulted in an EC20 of 1.9 µg/L. Resulting RQ is 1.6E-07.
Anadromous Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • No chronic data were identified for anadromous sturgeon tested in saltwater. However, a chronic study with juvenile white sturgeon (*Acipenser transmontanus*) resulted in an EC20 of 1.9 µg/L. Resulting RQ is 1.6E-07.

Table 5-34. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Lead Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Estuarine/Marine Fish	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • RQ directly applicable because the toxicity threshold is for *S. Marmoratus* as presented in Table 5-32 above
Beetle and Aquatic Insect	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Surrogate CTET identified is relevant and appropriate.
Amphibian	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Exposure through diet is not expected to be excessive • No chronic toxicity studies of effects of lead on amphibians were identified.
Snakes and Other Reptiles	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted
Sea Turtle	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted
Coastal/Marine Bird	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted
Marine Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted

Table 5-34. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Lead Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Terrestrial Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted
Seagrass	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk is confirmed using a 40-day study of lead toxicity to *Thalassia hemprichii* which showed effects on chloroplasts, observed as a change in leaf color, at 10 mg/L and a significant decrease in chlorophyll at 25 mg/L (Purnama et al., 2015). When modeled concentrations of lead are compared with 10 mg/L, the resulting $RQ_{A,W}$ is 8.2E-08.
Freshwater - Saltwater Aquatic and Wetland Plants	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of lead indicates very low potential for indirect effects (change in habitat/water quality parameters) at the concentrations predicted • Risk is confirmed using a growth study with *Juncus effusus* which showed no effect of lead at 103.1 µg/L (Najeeb et al., 2017). The resulting freshwater and saltwater $RQ_{A,W}$s are 2.9E-09 and 8.0E-06, respectively.

5.4.2.6 Mercury

Mercury was identified as a pollutant of concern in surface vessel bilgewater/OWS effluent and graywater discharge from vessels of the Armed Forces (see Section 3.2.3). It is a highly toxic element that is found both naturally and as an introduced contaminant in the environment. Mercury occurs naturally in rock in the earth's crust, including in deposits of coal. It exists in several forms:

- methylmercury and other organic compounds,
- elemental (metallic) mercury, and
- inorganic mercury compounds.

Methylmercury and other organic mercury compounds are formed when mercury combines with carbon. Microscopic organisms convert mercury into methylmercury, which is the most common organic mercury compound found in the environment.

Human activities are responsible for much of the mercury that is released into the environment. The burning of coal, oil and wood as fuel can cause mercury to become airborne, as can burning wastes that contain mercury. Mercury emitted to the atmosphere can be deposited into aqueous environments by wet and dry depositions, and some can be re-emitted into the atmosphere. The amount of mercury deposited in a given area depends on how much mercury is released from local sources.

In the environment, mercury exists as various species, mainly elemental (Hg0) and divalent (Hg2+) mercury depending on its oxidation states in air and water. Another species in the atmosphere is particulate mercury (Hg(p)), which is the mercury species adsorbed by particulate matter. Atmospheric deposition is the primary pathway for inputs of particulate mercury (Hg(p)) and divalent (Hg2+) to natural waters in many cases. The deposited mercury species, mainly Hg2+, can react with various organic compounds in water and sediment by biotic reactions mediated by sulfur-reducing bacteria, and abiotic reactions mediated by sunlight photolysis, resulting in conversion into organic mercury such as methylmercury (MeHg). (Kim and Zoh, 2012)

Mercury’s ability to build up in organisms and up along the food chain is cause for concern. Although all forms of mercury can accumulate to some degree, methylmercury is absorbed and accumulates to a greater extent than other forms. Inorganic mercury can also be absorbed, but is generally taken up at a slower rate and with lower efficiency than is methylmercury. The biomagnification of methylmercury causes it to have a greater impact on higher trophic levels. Nearly 100 percent of mercury that bioaccumulates in predator fish is methylmercury, the most toxic form. Most of the methylmercury in fish tissue is covalently bound to protein sulfhydryl groups. This binding results in a long half-life for elimination (about two years). Under steady state conditions, mercury concentrations in individuals of a given fish species tend to increase with age as a result of the slow elimination of methylmercury and increased intake due to changes in trophic position that often occur as fish grow to larger sizes.

The main pathway of exposure to mercury for humans and wildlife is by eating fish and shellfish that have high levels of methylmercury in their tissues. Birds and mammals that eat fish are have more exposures to methylmercury than other animals in aquatic ecosystems. Because mercury bioaccumulates, predators that eat these birds and mammals are then also at risk. At high levels of exposure, mercury's harmful effects on these animals include reduced reproduction, slower growth and development, neurotoxic effects that can lead to abnormal behavior, and death. Methylmercury is a central nervous system toxin, and the kidneys are the organs most vulnerable to damage from inorganic mercury.

For this BE, it is assumed that total measured and modeled mercury concentrations are methylmercury. Methylmercury is the most biologically available and most toxic form of mercury. Therefore, this assumption leads to the most conservative estimate of risk and likely overestimates risk.

5.4.2.6.1 Direct Effects from Mercury

Table 5-35 summarizes the chronic toxicity values available for calculating the risk quotients for aquatic species exposed to mercury in water ($CTET_{A,W}$). Table 5-36 presents the total mercury exposure concentrations estimated in ambient receiving waters (EC_W) and the RQs calculated as the ratio of EC_W to the corresponding CTET for each taxonomic group. Maximum exposure concentrations in ambient receiving water are estimated from the harbor modeling, as described above in Section 5.2.1.

Risk to Freshwater Aquatic Animal and Plant Species from Mercury

Concentrations of mercury in freshwater harbors resulting from discharges from vessels of the Armed Forces are expected to be low (approximately 9.3E-12 µg/L). The $RQ_{A,W}$ for freshwater vertebrates, invertebrates, and plants from exposure to mercury range from 4.4E-13 for plants to 4.0E-11 for vertebrates and indicate only "remote" risk of adverse effects (Table 5-36).

Risk from exposure to mercury from multiple routes of exposure was evaluated because of the ability for mercury to bioaccumulate and biomagnify through the food chain. RQs were calculated for freshwater vertebrates and invertebrates based on estimated concentrations of mercury accumulated in tissues from continuous exposure to lead in ambient receiving waters (see Table 5-36). The resulting RQs for aquatic vertebrates and invertebrates were 1.1E-10 and 1.8E-12, respectively, indicating "remote" risk to freshwater aquatic animals from lead accumulated in tissues.

Risk to Saltwater (Estuarine/Marine) Aquatic Animal and Plant Species – Mercury

Because mercury is only a pollutant of concern in graywater and surface bilgewater, and because of the much smaller number of vessels releasing these discharges to freshwater ports and harbors, maximum mercury concentrations in estuarine/marine harbors from discharges from vessels of the Armed Forces are estimated to be five orders of magnitude higher than freshwater harbors but still relatively low (4.6E-07 µg/L). Consequently, the risk to saltwater aquatic organisms from maximum exposure to mercury in ambient receiving water is greater than the same risk predicted for freshwater organisms based on RQs. However, risk is still "remote" for saltwater vertebrates ($RQ_{A,W}$ = 9.2E-08), invertebrates ($RQ_{A,W}$ = 2.4E-07), and plants ($RQ_{A,W}$ = 2.1E-07). The $RQ_{A,M}$ for saltwater aquatic vertebrates and invertebrates based on estimated concentrations accumulated in tissues from continuous lead exposure via multiple routes are 2.5E-06 and 4.7E-08, respectively, indicating "remote" risk to saltwater aquatic animals from mercury accumulated in tissues.

Risk to Birds and Mammals – Mercury

Risk to aquatic-dependent birds and mammals from exposure to mercury via consumption of prey or drinking ambient surface water was also evaluated because of the potential for prey to bioaccumulate mercury and the known risk to higher trophic level organisms that consume fish. The RQ_{wild} for representative freshwater and saltwater mammals and birds based comparison of dietary exposure concentrations calculated for this BE with effects thresholds are 8.1E-08 and 1.5E-05 for marine mammals and birds, respectively, and 1.6E-12 and 3.1E-10 for freshwater mammals and birds, respectively. These RQ_{wild} indicate "remote" risk from dietary exposure to mercury for aquatic-dependent birds and mammals.

Table 5-35. Summary of Chronic Toxicity Effect Thresholds for Aquatic Organisms Directly Exposed to Mercury

Direct Exposure to Mercury in Surface Water					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure/Effect Endpoint**	**NOEC - LOEC (µg/L)**	**$CTET_{A,W}$ (µg/L)**	**Study (source)**
Freshwater Vertebrate	Fathead minnow, *Pimephales promelas*	30-day Post-hatch Growth	--	0.23	Call et al. 1983
Freshwater Invertebrate	Rotifer, *Brachionus patulus*	24-day Growth	1.35 – 2.7	1.91	Sarma et al. 2008
Freshwater Plant (Vascular)	Duckweed, *Lemna minor*	7-day Growth	--	21 (EC10)	Naumann et al. 2007
Estuarine/Marine Vertebrate	Mummichog, *Fundulus heteroclitus*	ELS Test - Growth	--	5	Zhou 1997
Estuarine/Marine Invertebrate	Clam, *Meretrix meretrix*	ELS Test - Development	--	1.89	Wang et al. 2009
Estuarine/Marine Plant	Tangleweed, *Laminaria saccharina*	ELS Test – Sporeling Survival	1 – 5	2.24	Thompson and Burrows 1984
Accumulation of Mercury in Tissue from Multiple Exposure Routes					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_{A,T}$ (mg/kg wet wt.)**	**Study (source)**
Freshwater Vertebrate	Medaka, *Oryzias latipes*	Laboratory	NOEC - Mortality	0.048	ERED; Dillon et al.2010.Environ Toxicol Chem 29:2559-2565
Freshwater Invertebrate	Cladoceran, *Daphnia magna*	Laboratory	NOEC – Survival and Reproduction	0.859	ERED; Biesinger et al.1982.Arch Environ Con Tox 11:769-774
Estuarine/Marine Vertebrate	Flathead grey mullet, *Mugil cephalus*	Lab/Field	NOEC - Development	0.1	ERED; Beckvar et al.2005.Environ Toxicol Chem 24:2094-2105
Estuarine/Marine Invertebrate	Shrimp, *Palaemonetes pugio*	Laboratory	LOEC – Decreases avoidance behavior	1.64	ERED; Barthalmus.1977.Mar Pollut Bull 8:87-90

Table 5-35. Summary of Chronic Toxicity Effect Thresholds for Aquatic Organisms Directly Exposed to Mercury (Continued)

Dietary Exposure to Mercury					
Surrogate Aquatic-Dependent Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_o$ (mg/kg bw/d)**	**Study (source)**
Mammals	Norway rat, *Rattus norvegicus*	Diet	Reproduction	0.25	EPA 1995
Birds	American kestrel, *Falco sparverius*	Diet	Reproduction	0.05	Fuchsman et al. 2017

$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures

$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue

$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)

Table 5-36. Risk Quotients for Mercury Based on Comparison of Exposure Concentrations with Chronic Toxicity Effect Thresholds of Representative Surrogate Taxa

Taxa	EC_W (µg/L)	$CTET_{A,W}$ (µg/L)	$RQ_{A,W}$
Freshwater Vertebrate	9.3E-12	0.23	4.0E-11
Freshwater Invertebrate		1.91	4.9E-12
Freshwater Plant (Vascular)		21	4.4E-13
Estuarine/Marine Vertebrate	4.6E-07	5	9.2E-08
Estuarine/Marine Invertebrate		1.89	2.4E-07
Estuarine/Marine Plant (Vascular)		2.24	2.1E-07
Taxa	**EC_T (mg/kg wet wt)**	**$CTET_{A,T}$ (mg/kg wet wt)**	**$RQ_{A,M}$**
Freshwater Vertebrate	5.1E-12	0.048	1.1E-10
Freshwater Invertebrate	1.6E-12	0.859	1.8E-12
Estuarine/Marine Vertebrate	2.5E-07	0.1	2.5E-06
Estuarine/Marine Invertebrate	7.7E-08	1.64	4.7E-08
Taxa	**EC_O (mg/kg bw/d)**	**$CTET_O$ (mg/kg/d)**	**RQ_{wild}**
Mammals (Freshwater)	4.1E-13	0.25	1.6E-12
Birds (Freshwater)	1.5E-11	0.05	3.1E-10
Mammals (Estuarine/Marine)	2.0E-08	0.25	8.1E-08
Birds (Estuarine/Marine)	7.7E-07	0.05	1.5E-05

EC_W – Exposure concentration for water
EC_T – Exposure concentration accumulated in tissue
EC_O – Exposure concentration via oral ingestion (diet)
$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures
$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue
$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)
$RQ_{A,W}$ – Risk quotient from exposure to mercury in water calculated as EC_W/ $CTET_{A,W}$
$RQ_{A,M}$ – Risk quotient from accumulation of mercury in tissue calculated as EC_T/ $CTET_{A,T}$
RQ_{Wild} – Risk quotient from exposure to mercury through diet calculated as EC_O/ $CTET_O$

5.4.2.6.2 Indirect Effects from Mercury

The RQs summarized in Table 5-36 indicate “remote” risk for direct effects of mercury to federally listed aquatic species evaluated in this BE. Furthermore, discharge of mercury does not appear to result in appreciable concentrations in modeled estuaries and freshwater receiving water bodies under highly conservative scenarios.

The same RQs used to assess direct effects of mercury to federally listed aquatic species were used to assess indirect effects to available resources (water quality, vegetative cover, and prey). The assessment of risk to federally listed species can be extrapolated to other plants and animals because the effects thresholds apply to the most sensitive aquatic species and, therefore, can conservatively be applied to all species. In other words, the same assessment of risk can be applied to habitat quality or food resources of the listed species. As such, the risk calculations indicate a “remote” risk for indirect effects on listed species due to dietary exposure to mercury from vessels of the Armed Forces or due to toxicity-related reductions in the amount of food resources and habitat. Therefore, because the information summarized in Table 5-36 indicates that mercury at maximum modeled exposure concentrations has only “remote” risk of directly affecting freshwater and saltwater aquatic animals and plants, indirect effects typically associated with exposure, including loss of cover and changes in water quality parameters, are also not expected for this metal.

5.4.2.6.3 Risk Conclusions for Each Taxonomic Group of Federally Listed Species in Representative Action Areas from Exposure to Mercury

Based on the direct and indirect effects assessment from exposure of vertebrate, invertebrate and wildlife species to lead in discharges from vessels of the Armed Forces, the following risk conclusions are being made for each of the listed species groups in the RAAs (see Table 5-37). This information is used to inform decisions in the effects determinations made in Section 8 of this document.

Table 5-37. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Mercury Discharged from Vessels of the Armed Forces to Ports and Harbors

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Saltwater Corals	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects to most marine/estuarine invertebrates • Fate of mercury indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is checked using a sublethal toxicity study with *Porites asteroides* that showed physiological effects in coral colonies from exposure to mercury at concentrations as low as 4 µg/L (Bastidas and García, 2004). Resulting RQ is 1.2E-07.
Unionid Mussel	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a 21-day growth study with the Unionid mussel *Villosa iris* which resulted in a NOEC of 4 µg/L (ECOTOX; Valenti et al., 2005). Resulting RQ is 1.2E-07.
Freshwater Snails	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion confirmed using a 28-day study with *Bellamya bengalensis* that showed a biochemical stress response to mercury concentrations as low as 0.04 mg/L (Dhara, 2014). Resulting RQ is 2.3E-13.
Saltwater Mollusk	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Surrogate CTET used is representative and appropriate
Freshwater Shrimp/ Crustacean	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a 30-day study with the crayfish *Orconectes limosus* which resulted in an LC50 of 2 µg/L (ECOTOX; Boutet and Chaisemartin, 1973). The resulting RQ is 4.7E-12, indicating that the RQ for the NOEC is also likely to be well below 1.

Table 5-37. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Mercury Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Freshwater Fish/ Inland Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a 24-day study of rainbow trout (*O. mykiss*) that resulted in a LC50 of 5 µg/L (ECOTOX; Birge et al. 1983). Dividing this by an assessment factor of 10 results in a PNEC of 0.5 µg/L. Resulting RQ is 1.9E-11.
Anadromous Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • No chronic toxicity data were identified for salmonids tested in seawater. However, a study of 4-day exposures of Chinook salmon (*O. tshawytscha*) fry resulted in an LC50 of 17 µg/L (ECOTOX; Hamilton and Buhl, 1990). Dividing this by an assessment factor of 10 results in a PNEC of 1.7 µg/L. Resulting RQ is 5.5E-12.
Freshwater Fish/Inland Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • No chronic data were identified for inland/freshwater sturgeon.
Anadromous Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • No chronic data were identified for anadromous sturgeon tested in saltwater.
Estuarine/Marine Fish	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • RQ directly applicable because the toxicity threshold is for *F. heteroclitus* as presented in Table 5-35 above

Table 5-37. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Mercury Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Beetle and Aquatic Insect	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a study using *Chironomus sp*. midge larvae that resulted in an LC50 of 20 µg/L (ECOTOX; Rehwoldt et al. 1973). Dividing this by an assessment factor of 10 results in PNEC of 2 µg/L. Resulting RQ is 4.7E-12.
Amphibian	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Exposure through diet is not expected to be excessive • Risk conclusion is confirmed using an ELS with the African clawed frog (*Xenopus laevis*) which resulted in an LC50 of 0.16 µg/L (ECOTOX; Birge et al. 1979). Dividing this by an assessment factor of 10 results in a PNEC of 0.016 µg/L. Resulting RQ is 5.8E-10.
Snakes and Other Reptiles	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted
Sea Turtle	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted
Coastal/Marine Bird	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted
Marine Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted

Table 5-37. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Mercury Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Terrestrial Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted
Seagrass	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Surrogate CTET is representative and appropriate.
Freshwater - Saltwater Aquatic and Wetland Plants	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of mercury indicates very low potential for indirect effects (change in habitat/water quality parameters) at the concentrations predicted

5.4.2.7 Nickel

For this BE, risks to listed aquatic and aquatic-dependent species from exposure to nickel could occur from sonar dome discharge and surface vessel bilgewater/oil water separator effluent. Nickel occurs naturally in surface water in the United States, generally ranging between 0.4 and 3.8 μg/L but typically less than 10 μg/L (Eisler, 1998). For wildlife, nickel can be carcinogenic, may be mutagenic, but is not teratogenic. Nickel essentiality in plants and microorganisms is well established; but of all the known nickel-containing enzymes that have been isolated from plants or microorganisms, none, as yet, have been isolated in animal tissues (Pyle and Couture, 2012). In fish, the evidence for possible nickel essentiality is based on:

- o The observations that nickel concentrations in fish tissues remain relatively constant despite wide fluctuations in environmental concentrations (Tja¨lve et al., 1988; Ray et al., 1990),
- o Nickel can be reabsorbed by the kidney (Sreedevi et al., 1992; Ptashynski and Klaverkamp, 2002), and
- o Nickel uptake from food and water appears to be regulated (Lapointe and Couture, 2009).

Thus, nickel is bioconcentrated and bioaccumulated by aquatic organisms (Eisler, 1998), but only to low levels.

Toxicity of nickel to aquatic organisms is dependent on water hardness, pH, ionic composition, pollutant form, type and concentration of ligands, presence of mixtures, and availability of solid surfaces for adsorption (Eisler, 1998). Nickel interacts with many compounds to produce altered patterns of accumulation, metabolism, and toxicity (Eisler, 1998). Mixtures of metals containing nickel salts can be more toxic to daphnids and fishes than predicted on the basis of individual components (Enserink et al., 1991). As a conservative assumption for this BE, total concentrations of nickel are considered to be dissolved and, therefore, bioavailable. Also, because nickel toxicity to freshwater aquatic animals has been shown to be related to water hardness, it is appropriate to normalize chronic toxicity effect thresholds for freshwater aquatic animals to a standard water hardness (i.e., 100 mg/L as CaCO3) for comparative purposes and to support risk calculation (EPA, 2005a).

5.4.2.7.1 Direct Effects from Nickel

Table 5-38 summarizes the chronic toxicity values available for calculating the risk quotients for aquatic and aquatic-dependent species exposed to nickel. Table 5-39 presents the exposure concentrations for nickel estimated in ambient receiving waters (EC_W), in tissues of aquatic organisms exposed to those concentrations (EC_T), and in oral doses of aquatic-dependent animals ingesting food and water in those receiving waters (EC_O). Exposure concentrations in ambient receiving water are estimated from the harbor modeling, as described above. Table 5-39 also presents the RQs for nickel calculated as the ratio of each EC and corresponding CTET.

Risk to Freshwater Aquatic Animal and Plant Species – Nickel

The risk to freshwater aquatic organisms from exposure to nickel in ambient receiving water resulting from discharges from vessels of the Armed Forces is “remote”. The $RQ_{A,W}$ for freshwater vertebrates, freshwater invertebrates, and vascular plants are, 6.3E-10, 1.4E-08 and 2.9E-10, respectively.

Because nickel is a nutritionally essential inorganic element, it will accumulate to some degree in tissues of aquatic animals and is metabolically regulated. Like other essential metals, aquatic organism exposure to nickel concentrations in excess of nutritional needs and failure to metabolically regulate body burdens in addition to direct waterborne toxicity (e.g., dietary toxicity) may pose toxicity threats to some species (Jakimska et al., 2011). While nickel does not bioaccumulate to high levels in aquatic animals (Pyle and Couture, 2012), the EPA and DoD believe it is prudent to incorporate an analysis on multiple routes of exposure in their analysis of this pollutant. The evaluation of potential risks from bioaccumulated nickel provides a means of identifying potential risks to listed species via exposure to criteria concentrations from all exposure routes combined.

An $RQ_{A,M}$ was calculated for freshwater invertebrates based on estimated concentrations accumulated in tissues from continuous exposure to a maximum concentration of dissolved nickel in ambient receiving waters of 4.7E-09 μg/L (see Table 5-39). This estimated concentration was compared with a $CTET_{A,T}$ of 1.15 mg nickel/kg wet tissue (see Table 5-39). The corresponding $RQ_{A,M}$ of 4.1E-09, indicates “remote” risk to freshwater aquatic invertebrate populations from nickel accumulated in tissues. A $CTET_{A,T}$ for freshwater vertebrates was not identified; therefore, these exposures were not evaluated.

Risk to Saltwater (Estuarine/Marine) Aquatic Animal and Plant Species from Nickel

The risk to saltwater aquatic organisms from maximum exposure to nickel in ambient receiving water from discharges from vessels of the Armed Forces is also "remote". Using the maximum estimated exposure concentration, $RQ_{A,W}$ for saltwater organisms are approximately 2.3E-06 for marine vertebrates, 0.00044 for marine invertebrates and 0.0012 for vascular plants.

An $RQ_{A,M}$ was calculated for saltwater invertebrates based on estimated concentrations accumulated in tissues from continuous exposure to a maximum concentration of dissolved nickel in ambient receiving waters of 0.01 µg/L (see Table 5-39). The corresponding $RQ_{A,M}$ was 1.7E-05, indicating "remote" risk to saltwater aquatic animals from nickel accumulated in tissues.

Risk to Birds and Mammals from Nickel

An evaluation of risk to aquatic-dependent birds and mammals from exposure via consumption of prey or drinking ambient surface water at the maximum concentrations predicted for dissolved nickel is required because of the ability of (and need for) aquatic organisms to accumulate this essential metal in tissues. The RQ_{wild} estimated for surrogate mammals and birds from estimated dietary exposure to nickel are 9.4E-12 for freshwater mammals, 6.1E-12 for freshwater birds, 9.1E-07 for saltwater mammals and 5.9E-07 for saltwater birds (Table 5-39). These RQ_{wild} indicate "remote" risk to aquatic-dependent birds and mammals from dietary exposure to nickel.

Table 5-38. Summary of Chronic Toxicity Effect Thresholds for Aquatic and Aquatic-Dependent Organisms Exposed to Nickel

Surrogate Aquatic Species Type	Surrogate Species	Exposure Effect Endpoint	NOEC - LOEC (µg/L)	$CTET_{A,W}$ (µg/L)	Study (source)
Freshwater Vertebrate	Rainbow trout, *Oncorhynchus mykiss*	ELS Test -	62 - 134	91.15 (**158.0***)	Nebeker et al. 1985 (1986 ALC doc.)
Freshwater Invertebrate	Cladoceran, *Daphnia magna*	LC Test -	5 - 10 (hardness not provided)	7.071 (**7.050***)	Lazareva 1985 (1986 ALC doc.)
Freshwater Plant (Vascular)	Duckweed, *Lemna minor*	28-d EC50	- (hardness not provided)	340 (**339.0***)	Brown and Rattigan 1979 (1986 ALC doc.)
Estuarine/Marine Vertebrate	Topsmelt, *Atherinops affinis*	LC Test -	3,240 - 5,630	4,270 (**4,227***)	Hunt et al. 2002 (Ni saltwater addendum)
Estuarine/Marine Invertebrate	Mysid, *Mysidopsis intii*	LC Test -	10.0 - 48.8	22.09 (**21.87***)	Hunt et al. 2002 (Ni saltwater addendum)
Estuarine/Marine Plant (Vascular)	Not Available	–	–	8.2	CCC in EPA 1986 (ALC doc.)

Table 5-38. Summary of Chronic Toxicity Effect Thresholds for Aquatic and Aquatic-Dependent Organisms Exposed to Nickel (Continued)

Surrogate Aquatic Species Type	Surrogate Species	Exposure Type	Effect Endpoint	$CTET_{A,T}$ (mg/kg wet wt.)	Study (source)
Freshwater Vertebrate	Not Available	–	–	–	–
Freshwater Invertebrate	Amphipod, *Hyalella azteca*	4-wk Exposure to nickel-spiked sediment	ED25 for reduced growth	1.15	Borgmann et al. 2001
Estuarine/Marine Vertebrate	Not Available	–	–	–	–
Estuarine/Marine Invertebrate	Sea urchin, *Lytechinus pictus*	–	Inhibited normal development of embryos	26.40	Timourian and Watchmaker 1972
Surrogate Aquatic-Dependent Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_o$ (mg/kg bw/d)**	**Study (source)**
Mammals	Rat	3 generations (>1 yr) oral in diet	Reproduction	40	Sample et al. 1996 (Oregon Toxics BE)
Birds	Mallard	90-d oral in diet	Mortality, growth, behavior	77.4	Sample et al. 1996 (Oregon Toxics BE)

$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures

$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue

$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)

Bolded entries are for dissolved nickel.

*Freshwater data normalized to hardness = 100 mg/L CaCO3 using EPA pooled slope of 0.8460 and EPA dissolved conversion factor of 0.997; for estuarine/marine data, a dissolved conversion factor of 0.990 was used (EPA 1996)

Table 5-39. Risk Quotients for Nickel Based on Comparison of Exposure Concentrations with Chronic Toxicity Effect Thresholds of Representative Surrogate Taxa

Taxa	EC_W (µg/L)	$CTET_{A,W}$ (µg/L)	$RQ_{A,W}$
Freshwater Vertebrate	1.0E-07	158	6.3E-10
Freshwater Invertebrate		7.05	1.4E-08
Freshwater Plant (Vascular)		339	2.9E-10
Estuarine/Marine Vertebrate	0.0097	4227	2.3E-06
Estuarine/Marine Invertebrate		21.87	0.00044
Estuarine/Marine Plant (Vascular)		8.20	0.0012

Table 5-39. Risk Quotients for Nickel Based on Comparison of Exposure Concentrations with Chronic Toxicity Effect Thresholds of Representative Surrogate Taxa (Continued)

Taxa	EC_T (mg/kg wet wt)	$CTET_{A,T}$ (mg/kg wet wt)	$RQ_{A,M}$
Freshwater Vertebrate	4.7E-09	NA	NA
Freshwater Invertebrate	4.7E-09	1.15	4.1E-09
Estuarine/Marine Vertebrate	0.00046	NA	NA
Estuarine/Marine Invertebrate	0.00046	26.40	1.7E-05
Taxa	**EC_O (mg/kg bw/d)**	**$CTET_O$ (mg/kg/d)**	**RQ_{wild}**
Mammals (Freshwater)	3.8E-10	40	9.5E-12
Birds (Freshwater)	4.7E-10	77.4	6.1E-12
Mammals (Estuarine/Marine)	.000036	40	9.1E-07
Birds (Estuarine/Marine)	0.000046	77.4	5.9E-07

EC_W – Exposure concentration for water
EC_T – Exposure concentration accumulated in tissue
EC_O – Exposure concentration via oral ingestion (diet)
$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures
$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue
$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)
$RQ_{A,W}$ – Risk quotient from exposure to nickel in water calculated as EC_W/ $CTET_{A,W}$
$RQ_{A,M}$ – Risk quotient from accumulation of nickel in tissue calculated as EC_T/ $CTET_{A,T}$
RQ_{Wild} – Risk quotient from exposure to nickel through diet calculated as EC_O/ $CTET_O$

5.4.2.7.2 Indirect Effects for Nickel

Discharge of dissolved nickel from covered vessels of the Armed Forces will not result in appreciable concentrations in estuaries and freshwater receiving water bodies. As such, the potential for direct effects to aquatic populations in receiving waters is “remote”. Therefore, there is no basis for assuming indirect effects on listed species via dissolved nickel exposure and corresponding toxicity-related reduction in the prey base [loss of prey] available to upper trophic level organisms. Since the information summarized in Table 5-39 indicates that dissolved nickel at maximum modeled exposure concentrations has little risk of directly affecting fresh- and saltwater aquatic animals and plants, indirect effects typically associated with exposure, including loss of cover and changes in water quality parameters, are also not expected for this metal.

5.4.2.7.3 Risk Conclusion for Each Taxonomic Group of Federally Listed Species in Representative Action Areas from Exposure to Nickel

Based on the exposure assessment for aquatic and aquatic-dependent species for direct and indirect effects from nickel, the following risk conclusions are being made for each of the listed species in the RAAs (see Table 5-40). This information is used to inform decisions in the effects determinations made in Section 8 of this document.

Table 5-40. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Nickel Discharged from Vessels of the Armed Forces to Ports and Harbors

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Saltwater Corals	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • Risk conclusion is confirmed using a published study of mortality in *Pocilopor damicornis* that resulted in a NOEC of 1000 µg/L (Goh, 1991). Resulting $RQ_{A,W}$ is 9.7E—06.
Unionid Mussel	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • Risk conclusion is confirmed using a 4-day study with southern fatmucket (*Lampsilis straminea claibornen*) and an LC50 of 132 µg/L (131.6 µg/L dissolved; note: hardness not provided) (Keller, 2000). Dividing by a FACR of 17.99 results in a chronic value of 38.41 µg/L. Resulting $RQ_{A,W}$ is 2.6E-09.
Freshwater Snails	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • Risk conclusion is confirmed using a chronic early life stage study with *Radix quadrasi* with a LOEC of 0.0088 mg/L (Factor and Chavez, 2012). Application of a safety factor of 10 results in a NOEC of 0.00088 mg/L. Resulting RQ is 1.1E-07.
Saltwater Mollusk	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • Risk conclusion is confirmed via estimation using a lifecycle test with red abalone (*Haliotis rufescens*) MATC of 26.43 µg/L (26.17 µg/L dissolved; Hunt et al., 2002). Resulting $RQ_{A,W}$ is 3.8E-09.

Table 5-40. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Nickel Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Freshwater Shrimp/ Crustacean	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET is representative and appropriate
Freshwater Fish/ Inland Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • RQ directly applicable since estimated for *O. mykiss* as presented in Table 5-38 above
Anadromous Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data were identified for salmonids tested in seawater. However, a 144-hr study of yearling Coho salmon did not show any mortality of fish in freshwater or affect survival of fish when introduced to seawater, even at the highest exposure concentration of 5 mg/L (Lorz et al., 1978). Dividing that by an FACR of 17.99 results in an estimated chronic NOEC of 0.277 mg/L (=277 µg/L). Resulting RQ is 4.0E-10.
Freshwater Fish/Inland Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data were identified for sturgeon
Anadromous Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data were identified for sturgeon

Table 5-40. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Nickel Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Estuarine/Marine Fish	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • RQ directly applicable since estimated for *A. affinis* as presented in Table 5-38 above
Beetle and Aquatic Insect	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • Risk conclusion is confirmed using an ELS study with midge (*Culicoides furens*) larvae which resulted in an LC50 of 30 µg/L (ECOTOX; Vedamanikam and Shazilli, 2008). This was divided by a FACR of 17.99 to obtain a chronic LC50 of 1.7 µg/L. A factor of 10 applied to this value results in a predicted no effects concentration (PNEC) of 0.17 µg/L. Resulting RQ is 5.9E-09.
Amphibian	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Risk conclusion is confirmed via estimation using a 7-d study of narrow-mouthed toad (*Gastrophryne carolinensis*) with an EC50 (death and deformity) of 50 µg/L (50.27 µg/L dissolved and normalized to 100 mg/L hardness) (Birge and Black, 1980). This was divided by a FACR of 17.99 to obtain a chronic EC50 of 2.79 µg/L. A factor of 10 applied to this value results in a PNEC of 0.279 µg/L. Resulting $RQ_{A,W}$ is3.6E-07.
Snakes and Other Reptiles	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data were identified for reptiles; surrogate CTET for avian wildlife is considered representative and appropriate

Table 5-40. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Nickel Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Sea Turtle	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data were identified for reptiles; surrogate CTET for avian wildlife is considered representative and appropriate
Coastal/Marine Bird	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET is considered representative and appropriate
Marine Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted
Terrestrial Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET is considered representative and appropriate
Seagrass	Remote	• Exposed via water column only • Risk quotient indicates low potential for direct chronic effects • Fate of nickel indicates low potential for indirect effects (change in habitat/water quality parameters) – at the concentrations predicted

Table 5-40. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Nickel Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Freshwater- Saltwater Aquatic and Wetland Plants	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (change in habitat/water quality parameters) – at the concentrations predicted • RQ directly applicable since estimated for *L. minor* as presented in Table 5-38 above • Risk conclusion is confirmed via estimation using a study with an unknown duration with giant kelp (*Macrocystis pyritera*) and an EC50 (reduction in photosynthesis) of 2,000 µg/L (1,980 µg/L dissolved) (Clendenning and North, 1959). A factor of 10 was applied and resulted in a PNEC of 198 µg/L. Resulting $RQ_{A,W}$ is 5.0E-10.

5.4.2.8 Silver

Silver is a rare element that occurs naturally in the aquatic environment in trace quantities in association with basalt and igneous rock sources. Silver levels are elevated naturally in water from hot springs and steam wells. Background concentrations of dissolved silver are typically less than 0.2 µg/L in surface waters (Eisler 1996). Anthropogenic sources of silver have included mining, industrial and smelting wastes, hazardous waste sites, cloud seeding with silver iodide, waste water treatment plants, jewelry manufacture, electroplating plants, and the production and disposal of photographic material (EPA 1980, 1987; Eisler 1996).

Silver exhibits oxidation states of 0, +1, +2, and +3 but only occurs in the 0 and +1 state to any extent in the environment. Silver is usually found in very low concentrations in the aquatic environment due to its low crustal abundance and low mobility in water. A study of 10 U.S. rivers detected silver in concentrations ranging from 0.092 to 0.55 µg/L (EPA, 1980). Concentrations in 19 estuaries ranged from 0.07 to 4.1 µg/L (Eisler, 1996). The monovalent (+1) species is the form on concern in aquatic habitats, and it may exist either as simple hydrated monovalent ions that are dissociated from anions that, at one time, could have been part of its crystalline salt lattice or associated to varying degrees with inorganic ions (e.g., sulfate, bicarbonate, or nitrate) to form compounds with a range of solubilities. Sorption and precipitation are effective in reducing the concentration of dissolved silver, with sorption by manganese dioxide and precipitation with halides are likely being the dominant processes controlling its mobility. (EPA, 1980)

Silver is one of the most toxic metals to freshwater aquatic life (EPA, 1980). It is not known to be mutagenic, teratogenic, or carcinogenic (Eisler 1996). It bioconcentrates, and it may bioaccumulate (Eisler 1996). Silver toxicity may be altered by a number of factors including pH, organic carbon, cation exchange capacity, presence of mixtures (Ratte, 1999), sulfides, and duration of exposure. Silver, as ionic Ag^+, is one of the most toxic metals known to aquatic organisms in laboratory testing (Nebeker et al., 1983). Aquatic insects concentrate silver in relative proportion to environmental levels (Nehring, 1976), and more efficiently than most fish species (Diamond et al., 1990). Effects of silver toxicity to freshwater algae and phytoplankton include growth inhibition and altered species composition and species succession (Eisler, 1996). Effects of silver toxicity to freshwater invertebrates include inhibited feeding and coordination, reduced growth, elevated oxygen consumption, and reduced survival (Eisler, 1996). Effects of silver toxicity to freshwater fish include inhibited ionic flux across gills, reduced growth, premature hatch, and reduced survival (Eisler, 1996). Interspecies differences in the ability to accumulate, retain, and eliminate silver are large (Baudin et al., 1994).

5.4.2.8.1 Direct Effects from Silver

Table 5-41 summarizes the chronic toxicity values available for calculating the risk quotients for aquatic and aquatic-dependent species exposed to silver. Table 5-42 presents the exposure concentrations for silver estimated in ambient receiving waters (EC_W), in tissues of aquatic organisms exposed to those concentrations (EC_T), and in oral doses of aquatic-dependent animals ingesting food and water in those receiving waters (EC_O). Exposure concentrations in ambient receiving water are estimated from the harbor modeling, as described above. Table 5-42 also presents the RQs for silver calculated as the ratio of each EC and corresponding CTET.

Risk to Freshwater Aquatic Animal and Plant Species – Silver

Vessels of the Armed Forces are not expected to discharge silver. Therefore, the risk to freshwater aquatic organisms from exposure to silver in ambient receiving water resulting from discharges from vessels of the Armed Forces is “remote”.

Risk to Saltwater (Estuarine/Marine) Aquatic Animal and Plant Species from Silver

The risk to saltwater aquatic organisms from exposure to silver in ambient receiving water from discharges from vessels of the Armed Forces is also “remote”. Using the maximum modeled exposure concentration, $RQ_{A,W}$s for saltwater organisms are approximately 0.00001 for marine vertebrates, 0.00001 for marine invertebrates and 9.3E-06 for vascular plants.

An $RQ_{A,M}$ was calculated for saltwater invertebrates based on estimated concentrations accumulated in tissues from continuous exposure to a maximum concentration of dissolved nickel in ambient receiving waters of 2.8E-06 μg/L (see Table 5-42). The corresponding $RQ_{A,M}$s was 1.0E-06 for fish and 9.1E-06 for invertebrates, indicating “remote” risk to saltwater aquatic animals from silver accumulated in tissues.

Risk to Birds and Mammals from Silver

An evaluation of risk to aquatic-dependent birds and mammals from exposure to silver discharged by vessels of the Armed Forces via consumption of prey or drinking ambient surface water at the maximum modeled concentrations indicates that risk is “remote”. The RQ_{wild}s estimated for surrogate mammals and birds from estimated dietary exposure to nickel are 8.1E-10 for marine mammals and 2.2E-09 for marine birds (Table 5-42). These RQ_{wild}s indicate “remote” risk to aquatic-dependent birds and mammals from dietary exposure to nickel.

Table 5-41. Summary of Chronic Toxicity Effect Thresholds for Aquatic and Aquatic-Dependent Organisms Exposed to Silver

Surrogate Aquatic Species Type	Surrogate Species	Exposure Effect Endpoint	NOEC - LOEC (µg/L)	$CTET_{A,W}$ (µg/L)	Study (source)
Freshwater Vertebrate	Guppy, *Poecilia reticulata*	-	-	0.70	EPA 2008
Freshwater Invertebrate	Cladoceran, *Daphnia magna*	-	-	0.03	EPA 2008
Freshwater Plant (Vascular)		-	-	26	EPA 2008
Estuarine/Marine Vertebrate	Summer flounder, *Paralicthys dentatus*	-	-	0.50	EPA 2008
Estuarine/Marine Invertebrate	Pacific oyster, *Crassostrea gigas*	-	-	0.31	EPA 2008
Estuarine/Marine Plant (Vascular)	Multiple	Aquatic plant community composition	0.3 – 0.6 µg/L	0.3	EPA 2008
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_{A,T}$ (mg/kg wet wt.)**	**Study (source)**
Freshwater Vertebrate	Bluegill, *Lepomis macrochirus*	ELS Study	Growth and mortality	0.044	Coleman and Cearley 1974
Freshwater Invertebrate	Cladoceran, *Daphnia magna*	ELS Study	Reproduction	0.23	Glover and Wood 2004
Estuarine/Marine Vertebrate	Target fish, *Terapon jarbua*	ELS Study	Mortality	0.12	Long and Wang 2005
Estuarine/Marine Invertebrate	Copepod, *Acartia tonsa*	-	Reproduction	0.033	Hook and Fisher 2001
Surrogate Aquatic-Dependent Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_o$ (mg/kg bw/d)**	**Study (source)**
Mammals	Rat			22.2	Matuk et al. 1981
Birds	Chick			77.4	Smith and Carson 1977

$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures

$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue

$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)

Table 5-42. Risk Quotients for Silver Based on Comparison of Exposure Concentrations with Chronic Toxicity Effect Thresholds of Representative Surrogate Taxa

Taxa	EC_W (µg/L)	$CTET_{A,W}$ (µg/L)	$RQ_{A,W}$
Freshwater Vertebrate	0	0.70	Not calculated
Freshwater Invertebrate		0.03	Not calculated
Freshwater Plant (Vascular)		26	Not calculated
Estuarine/Marine Vertebrate	2.8E-08	0.50	0.00001
Estuarine/Marine Invertebrate		0.31	0.00001
Estuarine/Marine Plant (Vascular)		0.30	9.3E-06
Taxa	**EC_T (mg/kg wet wt)**	**$CTET_{A,T}$ (mg/kg wet wt)**	**$RQ_{A,M}$**
Freshwater Vertebrate	0	0.044	0
Freshwater Invertebrate	0	0.23	0
Estuarine/Marine Vertebrate	2.2E-07	0.12	1.9E-06
Estuarine/Marine Invertebrate	3.0E-07	0.033	9.1E-06
Taxa	**EC_O (mg/kg bw/d)**	**$CTET_O$ (mg/kg/d)**	**RQ_{wild}**
Mammals (Freshwater)	0	22.2	Not calculated
Birds (Freshwater)	0	77.4	Not calculated
Mammals (Estuarine/Marine)	1.8E-08	22.2	8.1E-10
Birds (Estuarine/Marine)	1.7E-07	77.4	2.2E-09

EC_W – Exposure concentration for water
EC_T – Exposure concentration accumulated in tissue
EC_O – Exposure concentration via oral ingestion (diet)
$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures
$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue
$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)
$RQ_{A,W}$ – Risk quotient from exposure to silver in water calculated as EC_W/ $CTET_{A,W}$
$RQ_{A,M}$ – Risk quotient from accumulation of silver in tissue calculated as EC_T/ $CTET_{A,T}$
RQ_{Wild} – Risk quotient from exposure to silver through diet calculated as EC_O/ $CTET_O$

5.4.2.8.2 Indirect Effects for Silver

Discharge of silver from vessels of the Armed Forces will not result in appreciable concentrations in estuaries and freshwater receiving water bodies. As such, the potential for direct effects to aquatic populations in receiving waters is "remote". Therefore, there is no basis for assuming indirect effects on listed species via dissolved nickel exposure and corresponding toxicity-related reduction in the prey base [loss of prey] available to upper trophic level organisms. Since the information summarized in Table 5-42 indicates that silver at maximum modeled exposure concentrations has little risk of directly affecting fresh- and saltwater aquatic animals and plants, indirect effects typically associated with exposure, including loss of cover and changes in water quality parameters, are also not expected for this metal.

5.4.2.8.3 Risk Conclusion for Each Taxonomic Group of Federally Listed Species in Representative Action Areas from Exposure to Silver

Based on the exposure assessment for aquatic and aquatic-dependent species for direct and indirect effects from silver, the following risk conclusions are being made for each of the listed species in the RAAs (see Table 5-43). This information is used to inform decisions in the effects determinations made in Section 8 of this document.

Table 5-43. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Silver Discharged from Vessels of the Armed Forces to Ports and Harbors

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Saltwater Corals	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of silver indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • Risk conclusion is confirmed using a published ELS study of survival and development of *Acropora japonica* exposed to silver nanocolloids that resulted in a NOEC of 5 µg/L (Suwa et al., 2014). Resulting $RQ_{A,W}$ is 5.6E-07.
Unionid Mussel	Remote	• Exposed via water column only • Substantial concentrations of silver are not expected from vessels of the Armed Forces in freshwater habitats; therefore, risk is remote
Freshwater Snail	Remote	• Exposed via water column only • Substantial concentrations of silver are not expected from vessels of the Armed Forces in freshwater habitats; therefore, risk is remote
Saltwater Mollusk	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of nickel indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • RQ directly applicable since estimated for *C. gigas* as presented in Table 5-41 above
Freshwater Shrimp/ Crustacean	Remote	• Exposed via water column only • Substantial concentrations of silver are not expected from vessels of the Armed Forces in freshwater habitats; therefore, risk is remote
Freshwater Fish/ Inland Salmonid	Remote	• Exposed via water column and diet • Substantial concentrations of silver are not expected from vessels of the Armed Forces in freshwater habitats; therefore, risk is remote

Table 5-43. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Silver Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Anadromous Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of silver indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data were identified for salmonids tested in seawater; however, the surrogate CTET for *P. dentatus* is considered to be relevant and appropriate
Freshwater Fish/Inland Sturgeon	Remote	• Exposed via water column and diet • Substantial concentrations of silver are not expected from vessels of the Armed Forces in freshwater habitats; therefore, risk is remote
Anadromous Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of silver indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data were identified for sturgeon; however, the surrogate CTET for *P. dentatus* is considered to be relevant and appropriate
Estuarine/Marine Fish	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of silver indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • RQ directly applicable since estimated for *P. dentatus* as presented in Table 5-41 above
Beetle and Aquatic Insect	Remote	• Exposed via water column only • Substantial concentrations of silver are not expected from vessels of the Armed Forces in freshwater habitats; therefore, risk is remote
Amphibian	Remote	• Exposed via water column and diet • Substantial concentrations of silver are not expected from vessels of the Armed Forces in freshwater habitats; therefore, risk is remote
Snakes and Other Reptiles	Remote	• Exposed via water column and diet • Substantial concentrations of silver are not expected from vessels of the Armed Forces in freshwater habitats; therefore, risk is remote

Table 5-43. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Silver Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Sea Turtle	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of silver indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data were identified for reptiles; surrogate CTET for avian wildlife is considered representative and appropriate
Coastal/Marine Bird	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of silver indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET is considered representative and appropriate
Marine Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of silver indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted
Terrestrial Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of silver indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET is considered representative and appropriate
Seagrass	Remote	• Exposed via water column only • Risk quotient indicates low potential for direct chronic effects • Fate of silver indicates low potential for indirect effects (change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET based on exposure of multiple species is considered to be representative and appropriate

Table 5-43. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Silver Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Freshwater- Saltwater Aquatic and Wetland Plants	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of silver indicates very low potential for indirect effects (change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET based on exposure of multiple species is considered to be representative and appropriate

5.4.2.9 Zinc

Dissolved zinc is ubiquitous in discharges from vessels of the Armed Forces, often at concentrations that exceed WQC (see Section 3.2.3.5). Of the Batch Two discharges selected for detailed analysis, zinc was detected in surface vessel bilgewater/OWS effluent, graywater, underwater ship husbandry discharge, and hull coating leachate.

Zinc occurs naturally in the aquatic environment, typically as leachate from igneous rocks. Background concentrations in freshwater systems are estimated to range between 0.5 and 40 µg/L (Groth, 1971; Moore and Ramamoorthy, 1984; Eisler, 1993). Although most naturally introduced zinc is adsorbed to sediments, a small amount remains dissolved.

Zinc interacts with many pollutants to produce altered patterns of accumulation, metabolism, and toxicity; some interactions reduce toxicity and others increase toxicity (Eisler, 1993). Most of the zinc introduced into aquatic environments is eventually partitioned into sediments (Eisler, 1993). Zinc bioavailability from sediment is increased under conditions of high dissolved, low salinity, low pH, and high levels of inorganic oxides and humic substances. Zinc bioconcentrates but does not biomagnify in the aquatic food chain (USEPA, 1987b).

Zinc is an essential element that is present in every plant and animal tissue measured (Hogstrand, 2012). There can be thousands of zinc proteins in animals, the majority of which use zinc to create folds in the protein structure or to join different gene products together (Andreini et al., 2005; Passerini et al., 2007; Maret and Li, 2009). In addition to its structural and catalytic roles in proteins, there is also a growing appreciation for zinc as a signaling substance and role as a neuromodulator or transmitter substance (Laity and Andrews, 2007; Hirano et al., 2008; Besser et al., 2009; Hogstrand et al., 2009; Sensi et al., 2009; Hershfinkel et al., 2010).

The toxicity of zinc to aquatic organisms is dependent on water hardness, pH, DO, presence of mixtures, and trophic level (Sorensen 1991, Eisler 1993). Because zinc toxicity to freshwater aquatic animals has been shown to be related to water hardness, it is appropriate to normalize chronic toxicity effect thresholds for freshwater aquatic animals to a standard water hardness (i.e., 100 mg/L as CaCO3) for comparative purposes and to support risk calculation (USEPA, 2005b). Because dissolved zinc was not measured along with total zinc in all Batch Two discharges selected for detailed analysis, total zinc concentrations are used for the risk analysis and assumed to be 100 percent bioavailable.

5.4.2.9.1 Direct Effects from Zinc

Table 5-44 summarizes the chronic toxicity values available for calculating the risk quotients for aquatic and aquatic-dependent species exposed to zinc. Table 5-45 presents the exposure concentrations for zinc estimated in ambient receiving waters (EC_W), in tissues of aquatic organisms exposed to those concentrations (EC_T), and in oral doses of aquatic-dependent animals ingesting food and water in those receiving waters (EC_O). Exposure concentrations in ambient receiving water are estimated from the harbor modeling, as described above. Table 5-45 also presents the RQs for zinc calculated as the ratio of each EC and corresponding CTET.

Risk to Freshwater Aquatic Animal and Plant Species from Zinc

The risk to freshwater aquatic organisms from exposure to zinc in ambient receiving water resulting from discharges from vessels of the Armed Forces is "remote". The $RQ_{A,W}$ for freshwater vertebrates, freshwater invertebrates, and vascular plants are 3.8E-07, 0.0009, and 2.7E-09, respectively.

Because zinc is a nutritionally essential inorganic element, it will accumulate to some degree in tissues of aquatic animals and is metabolically regulated. Like other essential metals, aquatic organism exposure to zinc concentrations in excess of nutritional needs and failure to metabolically regulate body burdens in addition to direct waterborne toxicity (e.g., dietary toxicity) may pose toxicity threats to some species (Jakimska et al., 2011). The evaluation of potential risks from bioaccumulated zinc provides a means of identifying potential risks to listed species via exposure to criteria concentrations from all exposure routes combined.

An $RQ_{A,M}$ was calculated for freshwater invertebrates based on estimated concentrations accumulated in tissues from continuous exposure to zinc in ambient receiving waters at a concentration of 2.7E-05 µg/L (see Table 5-45). This estimated concentration was compared with a $CTET_{A,T}$ of 7.7 mg zinc/kg wet tissue for vertebrates and 17.4 mg zinc/kg wet tissue for invertebrates (see Table 5-45). The corresponding $RQ_{A,M}$ of 1.6E-7 and 7.3E-08, respectively, indicates "remote" risk to freshwater aquatic vertebrate and invertebrate populations from zinc accumulated in tissues.

Risk to Saltwater (Estuarine/Marine) Aquatic Animal and Plant Species from Zinc

The risk to saltwater aquatic organisms from maximum exposure to zinc in ambient receiving water from discharges from vessels of the Armed Forces is also "remote". Using the maximum estimated exposure concentration, $RQ_{A,W}$ for saltwater organisms are 0.0057 for marine vertebrates, 0.0026 for marine invertebrates and 4.3E-05 for vascular plants.

An $RQ_{A,M}$ was calculated for saltwater invertebrates based on estimated concentrations accumulated in tissues from continuous exposure to a maximum concentration of dissolved zinc in ambient receiving waters of 0.41 µg/L (see Table 5-45). The corresponding $RQ_{A,M}$ for invertebrates was 0.0008, indicating “remote” risk to saltwater aquatic animals from zinc accumulated in tissues. No CTET was identified for vertebrates, therefore risk to vertebrates from zinc via multiple exposure routes was not evaluated.

Risk to Birds and Mammals from Zinc

Risk to aquatic-dependent birds and mammals from exposure to elevated concentrations of zinc via consumption of prey or drinking ambient surface water was evaluated because of the ability of (and need for) aquatic organisms to accumulate this essential metal in tissues. The RQ_{wild} estimated for surrogate mammals and birds from estimated dietary exposure to zinc are 6.4E-10 for freshwater mammals, 6.6E-08 for freshwater birds, 9.6E-06 for saltwater mammals and 0.001 for saltwater birds (Table 5-45). These RQ_{wild} indicate “remote” risk to aquatic-dependent birds and mammals from dietary exposure to zinc.

Table 5-44. Summary of Chronic Toxicity Effect Thresholds for Aquatic and Aquatic-Dependent Organisms Exposed to Zinc

Direct Exposure to Zinc					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Effect Endpoint**	**NOEC - LOEC (µg/L)**	**$CTET_{A,W}$ (µg/L)**	**Study (source)**
Freshwater Vertebrate	Flagfish, *Jordanella floridae*	LC Test	26 – 51	36.41 (**71.98***)	Spehar 1976 (1987 ALC document)
Freshwater Invertebrate	Cladoceran, *Daphnia magna*	LC Test	42 – 52	46.73 (**24.47***)	Chapman et al. manuscript (1987 ALC doc.)
Freshwater Plant (Vascular)	Duckweed, *Lemna minor*	Growth EC50	-	10,000 (**9,860***)	Wang 1986 (1987 ALC doc.)
Estuarine/Marine Vertebrate	Cabexon, *Scorpaenichthys marmoratus*	Acute LC50 ÷ Estuarine ACR (ACR=2.997)	191.4 ÷ 2.997	63.86 (**60.41***) (estimated)	Dinnel et al. 1983 (ACR) (1987 ALC document)
Estuarine/Marine Invertebrate	Mysid, *Americamysis bahia*	LC Test – survival and reproduction	120 - 231	166.5 (**157.5***)	Lussier et al. 1985 (1987 ALC document)
Estuarine/Marine Plant (Vascular)	Giant kelp, *Macrocystis pyrifera*	96-hour EC50 – photosynthetic rate	–	10,000 (**9,460***)	Clendenning and North 1959 (1987 ALC document)
Multiple Routes of Exposure to Zinc					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_{A,T}$ (mg/kg wet wt.)**	**Study (source)**
Freshwater Vertebrate	Rainbow trout, *Oncorhynchus mykiss*	Zinc deficiency study	Increased mortality, decreased growth	7.70	Spry et al. 1988

Table 5-44. Summary of Chronic Toxicity Effect Thresholds for Aquatic and Aquatic-Dependent Organisms Exposed to Zinc (Continued)

Multiple Routes of Exposure to Zinc					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_{A,T}$ (mg/kg wet wt.)**	**Study (source)**
Freshwater Invertebrate	Cladoceran, *Daphnia magna*	Zinc accumulation and regulation study	Decreased reproduction, growth, and energy	17.40	Muyssen and Janssen 2002
Estuarine/Marine Vertebrate	Not Available	–	–	–	–
Estuarine/Marine Invertebrate	Amphipod, *Allorchestes compressa*	4-week dietary accumulation study	Reduction in growth	24.00	Ahsanullah and Williams 1991
Dietary Exposure to Zinc					
Surrogate Aquatic-Dependent Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_o$ (mg/kg bw/d)**	**Study (source)**
Mammals	Rat	Days 1 – 16 of gestation, oral in diet	Reproduction	160	Sample et al. 1996 (Oregon Toxics BE)
Birds	White leghorn chicken	44-week oral in diet	Reproduction	14.5	Sample et al. 1996 (Oregon Toxics BE)

$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures

$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue

$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)

Bolded entries are for dissolved zinc.

*Freshwater data normalized to hardness = 100 mg/L CaCO3 using EPA pooled slope of 0.8473 and EPA dissolved conversion factor of 0.986; for estuarine/marine data, a dissolved conversion factor of 0.946 was used (EPA 1996)

Table 5-45. Risk Quotients for Zinc Based on Comparison of Exposure Concentrations with Chronic Toxicity Effect Thresholds of Representative Surrogate Taxa

Taxa	EC_W (µg/L)	$CTET_{A,W}$ (µg/L)	$RQ_{A,W}$
Freshwater Vertebrate	0.000027	71.98	3.8E-07
Freshwater Invertebrate		0.03	0.0009
Freshwater Plant (Vascular)		9860	2.7E-09
Estuarine/Marine Vertebrate	0.41	71.98	0.0057
Estuarine/Marine Invertebrate		157.5	0.0026
Estuarine/Marine Plant (Vascular)		9460	4.3E-05
Taxa	**EC_T (mg/kg wet wt)**	**$CTET_{A,T}$ (mg/kg wet wt)**	**$RQ_{A,M}$**
Freshwater Vertebrate	1.3E-06	7.7	1.6E-07
Freshwater Invertebrate		17.4	7.3E-08
Estuarine/Marine Vertebrate	0.019	NA	NA
Estuarine/Marine Invertebrate		24.0	0.0008
Taxa	**EC_O (mg/kg bw/d)**	**$CTET_O$ (mg/kg/d)**	**RQ_{wild}**
Mammals (Freshwater)	1.0E-07	160	6.4E-10
Birds (Freshwater)	9.5E-07	14.5	6.6E-08
Mammals (Estuarine/Marine)	0.0015	160	9.6E-06
Birds (Estuarine/Marine)	0.014	14.5	0.001

EC_W – Exposure concentration for water
EC_T – Exposure concentration accumulated in tissue
EC_O – Exposure concentration via oral ingestion (diet)
$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures
$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue
$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)
$RQ_{A,W}$ – Risk quotient from exposure to zinc in water calculated as EC_W/ $CTET_{A,W}$
$RQ_{A,M}$ – Risk quotient from accumulation of zinc in tissue calculated as EC_T/ $CTET_{A,T}$
RQ_{Wild} – Risk quotient from exposure to zinc through diet calculated as EC_O/ $CTET_O$

5.4.2.9.2 Indirect Effects for Zinc

Discharge of zinc from vessels of the Armed Forces will not result in appreciable concentrations in estuaries and freshwater receiving water bodies. As such, the potential for direct effects to aquatic populations in receiving waters is "remote". Therefore, there is no basis for assuming indirect effects on listed species via dissolved zinc exposure and corresponding toxicity-related reduction in the prey base [loss of prey] available to upper trophic level organisms. Because the information summarized in Table 5-45 indicates that zinc at maximum modeled exposure concentrations has little risk of directly affecting fresh- and saltwater aquatic animals and plants, indirect effects typically associated with exposure, including loss of cover and changes in water quality parameters, are also not expected for this metal.

5.4.2.9.3 *Risk Conclusion for Each Taxonomic Group of Listed Species in the Representative Action Areas from Zinc*

Based on the exposure assessment for aquatic and aquatic-dependent species for direct and indirect effects from zinc, the following risk conclusions are being made for each taxonomic group of listed species in the RAAs (see Table 5-46). This information is used to inform decisions in the effects determinations made in Section 8 of this document.

Table 5-46. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Zinc Discharged from Vessels of the Armed Forces to Ports and Harbors

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Saltwater Corals	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of zinc indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a published study of fertilization success *Goniastrea aspera* gametes that resulted in a NOEC of 500 µg/L (Reichelt-Brushett and Harrison, 1999). Resulting $RQ_{A,W}$ is 0.00082.
Unionid Mussel	Remote	• Exposed via water column only • Risk quotient indicates low potential for direct chronic effects • Fate of zinc indicates low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • Risk conclusion is confirmed using acute toxicity data for Asiatic clam (*Corbiucla fluminea*) with an LC50 of 6,040 µg/L (8,692 µg/L dissolved and normalized to 100 mg/L hardness) (Cherry et al., 1980; Rodgers et al., 1980). Dividing by the FACR of 2.208 results in a chronic value of 3,936 µg/L. Resulting $RQ_{A,W}$ is 6.9E-09.
Freshwater Snails	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of zinc indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using an acute study using *Physa gyrina* with a NOEL of 0.57 mg/L. Dividing by an ACR of 8.3 (Raimondo et al., 2007) results in a chronic NOEL of 0.069 mg/L. Resulting RQ is 3.9E-07.

Table 5-46. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Zinc Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Saltwater Mollusk	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of zinc indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed via estimation using a 23-day study of Pacific oyster (*Crassostrea gigas*) with an LC50 of 75 µg/L (70.95 µg/L dissolved; Watling, 1983). Dividing this by an assessment factor of 10 results in a PNEC of 7.5 µg/L. Resulting $RQ_{A,W}$ is 0.055.
Freshwater Shrimp/ Crustacean	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of zinc indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Surrogate CTET is representative and appropriate
Freshwater Fish/ Inland Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of zinc indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Risk conclusion is confirmed using a study of avoidance behavior of juvenile rainbow trout (*Oncorhynchus mykiss*) with an EC50 of 8.6 µg/L (Spraque, 1968 as cited in Price 2013). Dividing this by an assessment factor of 10 results in a PNEC of 0.86 µg/L. Resulting RQ is 0.000031.
Anadromous Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates low potential for direct chronic effects • Fate of zinc indicates low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • No chronic toxicity data were identified for salmonids tested in seawater. However, an ELS study with 4 stages of Chinook salmon resulted in a lowest LC50 of 54 µg/L (Chapman, 1978). Dividing this by an assessment factor of 10 results in a PNEC of 5.4 µg/L. Resulting RQ is 5.0E-06.

Table 5-46. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Zinc Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Freshwater Fish/Inland Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of zinc indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data were identified for sturgeon. However, a chronic ELS study of juvenile white sturgeon mortality and immobilization resulted in an EC20 of 99 µg/L (Ingersol and Mebane, 2014). Resulting RQ is 2.7E-07.
Anadromous Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of zinc indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data were identified for sturgeon
Estuarine/Marine Fish	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of zinc indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • RQ directly applicable since estimated for *S. marmoratus* as presented in Table 5-44 above
Beetle and Aquatic Insect	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of zinc indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • Risk conclusion is confirmed using an ELS study with midge (*Tanytarsus dissimilis*) eggs which resulted in an LC50 of 36.8 µg/L (ECOTOX; Anderson et al., 1980). A factor of 10 applied to this value results in a PNEC of 3.68 µg/L. Resulting RQ is 7.3E-06.

Table 5-46. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Zinc Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Amphibian	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of zinc indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Risk conclusion is confirmed using a 7-day study of narrow-mouthed toad (*Gastrophryne carolinensis*) with an EC50 (death and deformity) of 10 µg/L (5.599 µg/L dissolved and normalized to 100 mg/L hardness). Resulting $RQ_{A,W}$ is 4.8E-06.
Snakes and Other Reptiles	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of zinc indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data were identified for reptiles; surrogate CTET for avian wildlife is considered representative and appropriate
Sea Turtle	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of zinc indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data were identified for reptiles; surrogate CTET for avian wildlife is considered representative and appropriate
Coastal/Marine Bird	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of zinc indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET is considered representative and appropriate
Marine Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of zinc indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted

Table 5-46. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Zinc Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Terrestrial Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of zinc indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET is considered representative and appropriate
Seagrass	Remote	• Exposed via water column only • Risk quotient indicates low potential for direct chronic effects • Fate of zinc indicates low potential for indirect effects (change in habitat/water quality parameters) – at the concentrations predicted • RQ directly applicable since estimated for *M. pyrifera* as presented in Table 5-44 above
Freshwater- Saltwater Aquatic and Wetland Plants	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of zinc indicates very low potential for indirect effects (change in habitat/water quality parameters) – at the concentrations predicted • RQ directly applicable since estimated for *L. minor* as presented in Table 5-44 above • RQ directly applicable since estimated for *M. pyrifera* as presented in Table 5-44 above

5.4.2.10 Organometals

TBT is the only organometal of concern for this BE. It occurs in sonar dome discharge and hull coating leachate. TBT, an organotin compound that has been used extensively as a fungicide and bacteriocide in underwater and AF paints. It is extremely toxic to aquatic life and is an endocrine-disrupting chemical that causes severe reproductive effects in aquatic organisms. TBT is extremely stable and resistant to natural degradation in water. Because of its chemical properties, widespread use as an AF agent, and toxicity, concerns have been raised over the risks it poses to both freshwater and saltwater organisms. In 1988, the U.S. banned the use of AFC with TBT on vessels less than 25 meters in length. In 2001, the IMO adopted the International Convention on the Control of Harmful Anti-fouling Systems on Ships (AFS Convention) which bans the use of environmentally-damaging ship hull “AF systems.” AF paints using TBT have not been allowed for use on Navy ships since 1994, and use of TBT in AF has been prohibited globally since 2003. However, a small number of vessels of the Armed Forces still have TBT AF systems that have not been replaced.

TBT compounds may be moderately to highly persistent in the environment. Degradation depends on temperature and the presence of microorganisms and oxygen. Under aerobic conditions, TBT takes 1 to 3 months to degrade (Short, et al. 1986 as cited in EPA, 2008). But under anaerobic conditions, this compound can persist for more than 2 years. The breakdown of TBT leads eventually to the tin ion (Short, et al. 1986 as cited in EPA, 2008). All of the breakdown products are less toxic than TBT itself.

Because of the low water solubility of TBT and other properties, it will bind strongly to suspended material such as organic material or inorganic sediments (Clarkson 1991 as cited in EPA, 2008) and precipitate to the bottom sediment (Short, et al. 1986). Rates of sedimentation vary with location, organic content, particle size, and type of material. Reported half-lives of the compound in freshwater are 6 to 25 days; in seawater and estuarine locations, it is 1 to 34 weeks, depending on the initial concentration (Clark, et al. 1988 as cited in EPA, 2008). Because of the low levels of UV light beyond the topmost few centimeters in aquatic environments, it is unlikely photolysis plays a major role in degradation of TBT compounds (Clark, et al. 1988 as cited in EPA, 2008). Levels up to 0.800 µg/L have been found along the East Coast of the United States. In the Great Lakes, concentrations from 0.020 to 0.840 µg/L have been recorded. Concentrations as high as 1.0 µg/L have been found in San Diego Bay (MDNR 1987 as cited in EPA, 2008).

TBT compounds are highly toxic to many species of aquatic organisms. TBT exposure to non-target aquatic organisms such as mussels, clams, and oysters, at low levels, may cause physiological changes, retard growth, and increase mortality (Huggett et al. 1992 and MDNR 1987 as cited in EPA, 2008). TBT is very highly toxic to crustaceans. Lobster larvae show a nearly complete cessation of growth at just 1.0 µg/L TBT (EPA 1985e as cited in EPA, 2008). Mollusks, used as indicators of TBT pollution because of their high sensitivity to these chemicals, react adversely to levels of TBT ranging as low as 0.06 to 2.3 µg/L. They release TBT very slowly from their bodies after it has been absorbed. Imposex, the development of male characteristics in females, has been observed after TBT exposure in several snail species. In laboratory tests, reproduction was inhibited when female snails exposed to 0.05 µg/L of TBT developed male characteristics (EPA 1985e as cited in EPA, 2008). Imposex was also noted in the mud snail, or dogwhelk, at less than 3 ppt (0.003 µg/L) TBT (EPA 1985e as cited in EPA, 2008). Generally, the larvae of any tested species are more sensitive to TBT exposure than are the adults. Tributyltin oxide (TBTO) has been shown to inhibit cell survival of marine unicellular algae at very low concentrations; the 72-hour EC50 ranges from 0.33 µg/L to 1.03 µg/L (EPA 1985e as cited in EPA, 2008). TBT is lipophilic and tends to accumulate in oysters, mussels, crustaceans, mollusks, fish, and algae. Freshwater species will bioaccumulate more TBT than will marine organisms. Oysters bioaccummulate TBT compounds readily, reach an equilibrium uptake soon after exposure, and are slow to release this chemical. Oysters exposed to very low TBTO concentrations bioaccumulated TBT 1000 to 6000 fold. Juvenile chinook salmon accumulate TBT immediately upon exposure to low TBT concentrations. TBT and its metabolite, DBT, were found in the salmon's muscle tissue (Short, et al. 1986 as cited in EPA, 2008).

For this BE, it is assumed that all organotins measured in discharges from vessels of the Armed Forces are in the form of TBT.

5.4.2.10.1 Direct Effects from Tributyltin

Table 5-47 summarizes the chronic toxicity values available for calculating the risk quotients for aquatic and aquatic-dependent species exposed to TBT. Table 5-48 presents the exposure concentrations for TBT estimated in ambient receiving waters (EC_W), in tissues of aquatic organisms exposed to those concentrations (EC_T), and in oral doses of aquatic-dependent animals ingesting food and water in those receiving waters (EC_O). Exposure concentrations in ambient receiving water are estimated from the harbor modeling, as described above. Table 5-48 also presents the RQs for TBT calculated as the ratio of each EC and corresponding CTET.

Risk to Freshwater Aquatic Animal and Plant Species from Tributyltin

No vessels of the Armed Forces with sonar domes or TBT AF systems occur in freshwater ports or harbors. Therefore, there is no risk to freshwater receptors, including federally listed species, from exposure to TBT in discharges regulated by UNDS.

Risk to Saltwater (Estuarine/Marine) Aquatic Animal and Plant Species from Tributyltin

The risk to saltwater aquatic organisms from exposure to TBT in ambient receiving water from discharges from vessels of the Armed Forces is “remote”. Using the maximum estimated exposure concentration, $RQ_{A,W}$ for saltwater organisms are 0.0018 for marine vertebrates, 0.021 for marine invertebrates and 0.0035 for vascular plants.

An $RQ_{A,M}$ was calculated for saltwater vertebrates and invertebrates based on estimated concentrations accumulated in tissues from continuous exposure to a maximum concentration of TBT in ambient receiving waters of 0.00021 µg/L (see Table 5-48). The corresponding $RQ_{A,M}$ for vertebrates and invertebrates were 0.00034 and 0.007, respectively, indicating “remote” risk to saltwater aquatic animals from TBT accumulated in tissues.

Risk to Birds and Mammals from Tributyltin

Risk to aquatic-dependent birds and mammals from exposure to TBT via consumption of prey or drinking ambient surface water was evaluated because of the ability of aquatic organisms to bioaccumulate TBT. The RQ_{wild} estimated for surrogate mammals and birds from estimated dietary exposure to zinc are 5.8E-07 for saltwater mammals and 0.00001 for saltwater birds (Table 5-48). These RQ_{wild} indicate “remote” risk to aquatic-dependent birds and mammals from dietary exposure to zinc. Because there aren’t any vessels of the Armed Forces in freshwater ports and harbors with sonar domes or TBT AF systems, there is no risk to aquatic wildlife that depend on freshwater habitats from exposure to TBT in discharges from vessels of the Armed Forces.

Table 5-47. Summary of Chronic Toxicity Effect Thresholds for Aquatic and Aquatic-Dependent Organisms Exposed to Tributyltin

Direct Exposure to Tributyltin (TBT)			
Surrogate Aquatic Species Type	**Surrogate Species**	**$CTET_{A,W}$ (µg/L)**	**Study (source)**
Freshwater Vertebrate	Fathead minnow, *Pimephales promelas*	0.15	Brooke et al. 1986 (as cited in Oregon Toxics BE (EPA, 2008))
Freshwater Invertebrate	Hydra, *Hydra oligactus*	0.09	TAI Environmental Sciences, 1989 (as cited in Oregon Toxics BE (EPA, 2008))
Freshwater Plant (Vascular)	Green algae, *Scenedesmus obliquus*	3.4	Oregon Toxics BE (EPA, 2008)
Estuarine/Marine Vertebrate	Chinook salmon, *Oncorhynchus tshawytscha*	0.12	Oregon Toxics BE (EPA, 2008)
Estuarine/Marine Invertebrate	Dog whelk, *Nucella lappilus*	0.01	Oregon Toxics BE (EPA, 2008)
Estuarine/Marine Plant (Vascular)	Diatom, *Skeletonema costatum*	0.06	Oregon Toxics BE (EPA, 2008)

Multiple Routes of Exposure to TBT				
Surrogate Aquatic Species Type	**Surrogate Species**	**Effect Endpoint**	**$CTET_{A,T}$ (mg/kg wet wt.)**	**Study (source)**
Freshwater Vertebrate	Rainbow trout, *Oncorhynchus mykiss*	Reduced weight gain, behavioral changes	0.27	Triebskorn et al. 1994 (as cited in Oregon Toxics BE (EPA, 2008))
Freshwater Invertebrate	Ramshorn snail, *Marisa cornuarietis*	Threshold for imposex induction	0.32	Schulte-Oehlmann et al. 1995
Estuarine/Marine Vertebrate	Steelhead trout, *Oncorhynchus mykiss*	Reduced weight gain, behavioral changes	0.27	Triebskorn et al. 1994 (as cited in Oregon Toxics BE (EPA, 2008))
Estuarine/Marine Invertebrate	Rock shell, *Thais clavigera*	Imposex induction	0.013	Gibbs and Bryan 1987

Dietary Exposure to TBT					
Surrogate Aquatic-Dependent Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_o$ (mg/kg bw/d)**	**Study (source)**
Mammals	Mouse	Days 6 – 15 of gestation, oral intubation	Reproduction	23.4	Sample et al. 1996 (as cited in Oregon Toxics BE (EPA, 2008))
Birds	Japanese Quail	6-week oral in diet	Reproduction	6.8	Sample et al. 1996 (as cited in Oregon Toxics BE (EPA, 2008))

$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures
$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue
$CTET_{O}$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)

Table 5-48. Risk Quotients for Tributyltin Based on Comparison of Exposure Concentrations with Chronic Toxicity Effect Thresholds of Representative Surrogate Taxa

Taxa	EC_W (µg/L)	$CTET_{A,w}$ (µg/L)	$RQ_{A,w}$
Freshwater Vertebrate	0	0.15	Not calculated
Freshwater Invertebrate		0.09	Not calculated
Freshwater Plant (Vascular)		3.4	Not calculated
Estuarine/Marine Vertebrate	0.00021	0.12	0.0018
Estuarine/Marine Invertebrate		0.01	0.021
Estuarine/Marine Plant (Vascular)		0.06	0.0035
Taxa	**EC_T (mg/kg wet wt)**	**$CTET_{A,T}$ (mg/kg wet wt)**	**$RQ_{A,M}$**
Freshwater Vertebrate	0	0.27	Not calculated
Freshwater Invertebrate		0.32	Not calculated
Estuarine/Marine Vertebrate	0.000091	0.27	0.00034
Estuarine/Marine Invertebrate		0.013	0.007
Taxa	**EC_O (mg/kg bw/d)**	**$CTET_O$ (mg/kg/d)**	**RQ_{wild}**
Mammals (Freshwater)	0	23.4	Not calculated
Birds (Freshwater)	0	6.8	Not calculated
Mammals (Estuarine/Marine)	0.000014	23.4	5.8E-07
Birds (Estuarine/Marine)	0.000068	6.8	0.00001

EC_W – Exposure concentration for water
EC_T – Exposure concentration accumulated in tissue
EC_O – Exposure concentration via oral ingestion (diet)
$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures
$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue
$CTET_{O}$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)
$RQ_{A,W}$ – Risk quotient from exposure to Tributyltin (TBT) in water calculated as EC_W/ $CTET_{A,W}$
$RQ_{A,M}$ – Risk quotient from accumulation of TBT in tissue calculated as EC_T/ $CTET_{A,T}$
RQ_{Wild} – Risk quotient from exposure to TBT through diet calculated as EC_O/ $CTET_O$

5.4.2.10.2 Indirect Effects for Tributyltin

Discharge of TBT from vessels of the Armed Forces will not result in appreciable concentrations in estuaries and freshwater receiving water bodies. As such, the potential for direct effects to aquatic populations in receiving waters is “remote”. Therefore, there is no basis for assuming indirect effects on listed species via dissolved zinc exposure and corresponding toxicity-related reduction in the prey base [loss of prey] available to upper trophic level organisms. Because the information summarized in Table 5-48 indicates that TBT at maximum modeled exposure concentrations has little risk of directly affecting fresh- and saltwater aquatic animals and plants, indirect effects typically associated with exposure, including loss of cover and changes in water quality parameters, are also not expected for this metal.

5.4.2.10.3 Risk Conclusion for Each Taxonomic Group of Listed Species in the RAAs from Tributyltin

Based on the exposure assessment for aquatic and aquatic-dependent species for direct and indirect effects from TBT, the following risk conclusions are being made for each taxonomic group of listed species in the RAAs (see Table 5-49). This information is used to inform decisions in the effects determinations made in Section 8 of this document.

Table 5-49. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Tributyltin Discharged from Vessels of the Armed Forces to Ports and Harbors

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Saltwater Corals	Negligible	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of Tributyltin (TBT) indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a published study of fertilization success and metamorphosis in *Acropora millepora* that resulted in an IC50 NOEC of 2 µg/L (Negri and Hayward, 2001). Applying a conservative conversion factor of 1000 to estimate a PNEC results in a concentration equivalent to the ANZECC guidelines equal to 0.002 µg/L. The resulting $RQ_{A,W}$ based on the ANZECC guidelines is 0.11. • Risk conclusion changed from "remote" to "negligible" based on comparison of maximum modeled concentrations with the ANZECC guideline.
Unionid Mussel	Remote	• Unionid mussels are not expected to be exposed to TBT released from vessels of the Armed Forces
Freshwater Snail	Remote	• Freshwater snails are not expected to be exposed to TBT released from vessels of the Armed Forces
Saltwater Mollusk	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of TBT indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Surrogate CTET is representative and appropriate
Freshwater Shrimp/ Crustacean	Remote	• Freshwater shrimp/crustaceans are not expected to be exposed to TBT released from vessels of the Armed Forces

Table 5-49. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Tributyltin Discharged from Vessels of th Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Freshwater Fish/ Inland Salmonid	Remote	• Freshwater fish/inland salmonids are not expected to be exposed to TBT released from vessels of the Armed Forces
Anadromous Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates low potential for direct chronic effects • Fate of zinc indicates low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • RQ directly applicable because the CTET used is for chinook salmon (*Oncorhynchus tshawytscha*) as presented in Table 5-47 above
Freshwater Fish/Inland Sturgeon	Remote	• Freshwater fish and inland sturgeon are not expected to be exposed to TBT released from vessels of the Armed Forces
Anadromous Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of TBT indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET is representative and appropriate
Estuarine/Marine Fish	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of TBT indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • RQ directly applicable because the CTET used is for chinook salmon (*Oncorhynchus tshawytscha*) as presented in Table 5-47 above
Beetle and Aquatic Insect	Remote	• Beetles and aquatic insects are not expected to be exposed to TBT released from vessels of the Armed Forces
Amphibian	Remote	• Amphibians are not expected to be exposed to TBT released from vessels of the Armed Forces
Snakes and Other Reptiles	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of TBT indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data were identified for reptiles; surrogate CTET for avian wildlife is considered representative and appropriate

Table 5-49. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Tributyltin Discharged from Vessels of th Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Sea Turtle	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of TBT indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data were identified for reptiles; surrogate CTET for avian wildlife is considered representative and appropriate
Coastal/Marine Bird	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of TBT indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET is considered representative and appropriate
Marine Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of TBT indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted
Terrestrial Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of TBT indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET is considered representative and appropriate
Seagrass	Remote	• Exposed via water column only • Risk quotient indicates low potential for direct chronic effects • Fate of TBT indicates low potential for indirect effects (change in habitat/water quality parameters) – at the concentrations predicted • Risk conclusion confirmed using a 6-week growth study of *Thalassia testudinum* that resulted in a NOEC of 2.23 µg/L (Kelly et al., 1990). This is higher than the $CTET_{A,W}$ used for the risk assessment. Resulting RQ based on this study is 0.000094.

Table 5-49. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Tributyltin Discharged from Vessels of th Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Freshwater- Saltwater Aquatic and Wetland Plants	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects to saltwater wetland plants • Freshwater wetland plants are not expected to be exposed to TBT released from vessels of the Armed Forces • Fate of TBT indicates very low potential for indirect effects (change in habitat/water quality parameters) – at the concentrations predicted • RQ directly applicable since estimated for *L. minor* as presented in Table 5-47 above • Surrogate CTET is considered relevant and appropriate

5.4.3 Volatile Organic Compounds and Semi-volatile Organic Compounds

A number of VOCs and semi-volatile organic compounds (SVOCs), including petroleum hydrocarbons, were selected as pollutants for detailed evaluation in Section 3.2.3.6 of this BE. These pollutants were identified because their levels measured in selected discharges exceeded screening level benchmarks. The pollutants that were determined to exceed the benchmarks are bis(2-ethylhexyl)phthalate (DEHP), oil and grease, and petroleum hydrocarbons measured as TPH. Risks to ecological receptors from exposure to O&G and TPH are evaluated qualitatively in Section 5.1.2. Risks to ecological receptors from exposure to DEHP is evaluated quantitatively below.

5.4.3.1 Bis(2-ethylhexyl)phthalate

Bis(2-ethylhexyl)phthalate (DEHP) is a compound that is found in firemain system discharge. It is used in the production of polyvinyl chloride (PVC) and is added to plastics to make them flexible. Release of DEHP to the environment occurs during the production, transport, storage, and processing of PVC and non-polymers. Its main sources in the environment is leaching from PVC and other plastic materials that contain DEHP. Plasticisers are not chemically bound to the matrix polymer in flexible PVC (or other materials); therefore the will leach from the final product to some extent during its use and after its final disposal.

DEHP does not hydrolyse in water and appears to be readily biodegradable. Experimental data indicates a biodegradation half-life for DEHP in surface water of 50 days, and 300 days in aerobic sediment. Anaerobic conditions and low temperature further reduce the degradation rate. With a log Kow of 7.5, DEHP is expected to be strongly adsorbed to organic matter and therefore is expected to be found in the solid organic phase in the environment. (IHCP, 2008)

DEHP has been found to bioaccumulate in aquatic organisms, and the highest BCF values are observed for invertebrates (e.g. 2,700 for *Gammarus* compared with a BCF of 840 for fish). This indicates that trophic transfer via the food chain might be an important exposure route (secondary poisoning). There is no indication that DEHP bio-magnifies, and this may be due in part to more effective metabolism in higher organisms. (IHCP, 2008)

DEHP in rivers waters has been reported to vary from below the detection limit to 21 µg/L. In the marine environment, DEHP in coastal surface waters near Norway were reported to be predominantly below 0.1 µg/L.

DEHP, along with other phthalates, is believed to cause endocrine disruption in males. Exposure to DEHP can also affect reproduction, growth, and cardiovascular activity.

5.4.3.1.1 Direct Effects from Bis(2-ethylhexyl)phthalate

Table 5-50 summarizes the chronic toxicity values available for calculating the risk quotients for aquatic and aquatic-dependent species exposed to DEHP. Table 5-51 presents the exposure concentrations for DEHP estimated in ambient receiving waters (EC_W), in tissues of aquatic organisms exposed to those concentrations (EC_T), and in oral doses of aquatic-dependent animals ingesting food and water in those receiving waters (EC_O). Exposure concentrations in ambient receiving water are estimated from the harbor modeling, as described above. Table 5-51 also presents the RQs for DEHP calculated as the ratio of each EC and corresponding CTET.

Risk to Freshwater Aquatic Animal and Plant Species from Bis(2-ethylhexyl)phthalate

The risk to freshwater aquatic organisms from exposure to DEHP in ambient receiving water resulting from discharges from vessels of the Armed Forces is “remote”. The $RQ_{A,W}$ for freshwater vertebrates and invertebrates are 1.4E-08 and 9.3E-08, respectively. No toxicity threshold was identified for freshwater vascular plants; therefore risks to freshwater plants were not evaluated.

Because DEHP has a tendency to bioaccumulate, potential risks from bioaccumulated DEHP were evaluated to provide a means of identifying potential risks to listed species from all exposure routes combined. An $RQ_{A,M}$ was calculated for freshwater vertebrates and invertebrates based on estimated concentrations accumulated in tissues from continuous exposure to DEHP in ambient receiving waters at a modeled concentration of 2.8E-08 µg/L (see Table 5-51). This concentration was estimated to result in a vertebrate body burden (EC_T) of 2.4E-08 mg DEHP/kg (wet weight) and an invertebrate body burden of 7.6E-08 mg DEHP/kg (wet weight). Comparison of these tissue concentrations with their respective $CTET_{A,T}$ results in $RQ_{A,M}$ of 3.6E-08 and 1.4E-07 for freshwater vertebrates and invertebrates, respectively, indicating “remote” risk from DEHP accumulated in tissues.

Risk to Saltwater (Estuarine/Marine) Aquatic Animal and Plant Species from Bis(2-ethylhexyl)phthalate

The risk to saltwater aquatic organisms from maximum exposure to DEHP in ambient receiving water from discharges from vessels of the Armed Forces is also “remote”. Using the maximum estimated exposure concentration, $RQ_{A,W}$ for saltwater organisms are 0.00014 for marine vertebrates and 0.00013 for marine invertebrates. As with freshwater plants, no toxicity threshold was identified for marine vascular plants.

An $RQ_{A,M}$ was calculated for saltwater invertebrates based on estimated concentrations accumulated in tissues from continuous exposure to a maximum concentration of DEHP in ambient receiving waters of 0.014 μg/L (see Table 5-51). The corresponding $RQ_{A,M}$ for invertebrates was 0.076, indicating “remote” risk to saltwater aquatic animals from DEHP accumulated in tissues. No CTET was identified for vertebrates, therefore risk to vertebrates from DEHP via multiple exposure routes was not evaluated.

Risk to Birds and Mammals from Bis(2-ethylhexyl)phthalate

Risk to aquatic-dependent birds and mammals from exposure to elevated concentrations of DEHP via consumption of prey or drinking ambient surface water was evaluated because of the ability of DEHP to bioaccumulate in aquatic organisms. The RQ_{wild} for surrogate mammals and birds from estimated dietary exposure to DEHP are 1.9E-10 for freshwater mammals, 2.4E-08 for freshwater birds, 9.6E-05 for saltwater mammals and 0.012 for saltwater birds (Table 5-51). These RQ_{wild} indicate “remote” risk to aquatic-dependent birds and mammals from dietary exposure to DEHP.

Table 5-50. Summary of Chronic Toxicity Effect Thresholds for Aquatic and Aquatic-Dependent Organisms Exposed to Bis(2-ethylhexyl)phthalate

Direct Exposure to DEHP					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Effect Endpoint**	**NOEC - LOEC (μg/L)**	**$CTET_{A,W}$ (μg/L)**	**Study (source)**
Freshwater Vertebrate	Zebra fish, *Danio rerio*	21-day – reproduction; development of oocytes	-	2[a]	Carnevali et al. 2010
Freshwater Invertebrate	Midge, *Chironomus riparus*	33-day - growth	-	0.30 [a]	Kwak and Lee 2005
Freshwater Plant (Vascular)	Not available	-	-	-	-
Estuarine/Marine Vertebrate	Indian medaka, *Oryzias melastigma*			100 [a]	Ye et al. 2014
Estuarine/Marine Invertebrate	Cladoceran, *Daphnia magna*	21-day - survival	77-160	111 [a]	Staples et al. 1997
Estuarine/Marine Plant (Vascular)	Not available	-	-	-	-
Multiple Routes of Exposure to DEHP					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_{A,T}$ (mg/kg wet wt.)**	**Study (source)**
Freshwater Vertebrate	Bluegill, *Lepomis macrochirus*	–	Mortality NOEC	0.66	
Freshwater Invertebrate	Cladoceran, *Daphnia magna*	ELS Test	Reproduction NOEC	0.53	Wobeser 1975
Estuarine/Marine Vertebrate	Not Available	–	–	–	–
Estuarine/Marine Invertebrate	Shrimp, *Palaemonetes pugio*	ELS Test	Development NOEC	0.5	Laughlin et al.1978 (ERED database)
Dietary Exposure to DEHP					
Surrogate Aquatic-Dependent Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_o$ (mg/kg bw/d)**	**Study (source)**
Mammals	Mouse	105-day oral in diet	Reproduction	18.3	Sample et al. 1996
Birds	Ringed Dove	4-week oral in diet	Reproduction	1.1	Sample et al. 1996

$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures
$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue
$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)
[a]Based on 1,2-Benzenedicarboxylic acid, 1,2-Bis(2-ethylhexyl)ester, an isomer of bis(2-ethylhexyl)phthalate

Table 5-51. Risk Quotients for Bis(2-ethylhexyl)phthalate Based on Comparison of Exposure Concentrations with Chronic Toxicity Effect Thresholds of Representative Surrogate Taxa

Taxa	EC_W (µg/L)	$CTET_{A,W}$ (µg/L)	$RQ_{A,W}$
Freshwater Vertebrate	2.8E-08	2	1.4E-08
Freshwater Invertebrate		0.30	9.3E-08
Freshwater Plant (Vascular)		--	Not calculated
Estuarine/Marine Vertebrate	0.014	100	0.00014
Estuarine/Marine Invertebrate		111	0.00013
Estuarine/Marine Plant (Vascular)		--	Not calculated
Taxa	**EC_T (mg/kg wet wt)**	**$CTET_{A,T}$ (mg/kg wet wt)**	**$RQ_{A,M}$**
Freshwater Vertebrate	2.4E-08	0.66	3.6E-08
Freshwater Invertebrate	7.6E-08	0.53	1.4E-07
Estuarine/Marine Vertebrate	0.012	NA	Not calculated
Estuarine/Marine Invertebrate	0.04	0.5	0.076
Taxa	**EC_O (mg/kg bw/d)**	**$CTET_O$ (mg/kg/d)**	**RQ_{wild}**
Mammals (Freshwater)	3.5E-09	18.3	1.9E-10
Birds (Freshwater)	2.6E-08	1.1	2.4E-08
Mammals (Estuarine/Marine)	0.002	18.3	0.000096
Birds (Estuarine/Marine)	0.0132	1.1	0.012

EC_W – Exposure concentration for water
EC_T – Exposure concentration accumulated in tissue
EC_O – Exposure concentration via oral ingestion (diet)
$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures
$CTET_{A,T}$ – Chronic toxicity effects threshold for accumulation in tissue
$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)
$RQ_{A,W}$ – Risk quotient from exposure to DEHP in water calculated as EC_W/ $CTET_{A,W}$
$RQ_{A,M}$ – Risk quotient from accumulation of DEHP in tissue calculated as EC_T/ $CTET_{A,T}$
RQ_{Wild} – Risk quotient from exposure to DEHP through diet calculated as EC_O/ $CTET_O$

5.4.3.1.2 Indirect Effects for Bis(2-ethylhexyl)phthalate

Discharge of DEHP from vessels of the Armed Forces will not result in appreciable concentrations in estuaries and freshwater receiving water bodies. As such, the potential for direct effects to aquatic populations in receiving waters is “remote”. Therefore, there is no basis for assuming indirect effects on listed species from exposure to DEHP and corresponding toxicity-related reduction in the prey base [loss of prey] available to upper trophic level organisms. Because the information summarized in Table 5-51 indicates that DEHP at maximum modeled exposure concentrations has little risk of directly affecting fresh- and saltwater aquatic animals, indirect effects typically associated with exposure, including changes in water quality, are also not expected for this metal.

5.4.3.1.3 Risk Conclusion for Each Taxonomic Group of Listed Species in the RAAs from Bis(2-ethylhexyl)phthalate

Based on the exposure assessment for aquatic and aquatic-dependent species for direct and indirect effects from DEHP, the following risk conclusions are being made for each taxonomic group of listed species in the RAAs (see Table 5-52). This information is used to inform decisions in the effects determinations made in Section 8 of this document.

Table 5-52. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Bis(2-ethylhexyl)phthalate Discharged from Vessels of the Armed Forces to Ports and Harbors

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Saltwater Corals	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of DEHP indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a chronic study of the freshwater hydra, *Hydra viridissima* (freshwater relative of corals and anemones), which resulted in a NOEC of 1 µg/L (Ganeshakumar, 2009). Resulting RQ is 0.014.
Unionid Mussel	Remote	• Exposed via water column only • Risk quotient indicates low potential for direct chronic effects • Fate of DEHP indicates low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • Studies were not identified to confirm risk conclusions; surrogate CTET is accepted as relevant and appropriate
Freshwater Snail	Remote	• Exposed via water column only • Risk quotient indicates low potential for direct chronic effects • Fate of DEHP indicates low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • Studies were not identified to confirm risk conclusions; surrogate CTET is accepted as relevant and appropriate

Table 5-52. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Bis(2-ethylhexyl)phthalate Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Saltwater Mollusk	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of DEHP indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a study of the mussel *Mytilus galloprovincialis* which demonstrated biochemical effects at 500 µg/L (Carlisle et al., 2009). Resulting RQ is 5.6E-11.
Freshwater Shrimp/ Crustacean	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of DEHP indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Surrogate CTET is considered to be relevant and appropriate
Freshwater Fish/ Inland Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of DEHP indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Risk conclusion confirmed using an ELS test with rainbow trout (*O. mykiss*) that resulted in an LC01 of 150 µg/L (EcoTox; Birge 1978). Resulting RQ is 1.9E-10.
Anadromous Salmonid	Remote	• Exposed via water column and diet • Risk quotient indicates low potential for direct chronic effects • Fate of DEHP indicates low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Surrogate CTET is considered to be relevant and appropriate
Freshwater Fish/Inland Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of DEHP indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET is considered to be relevant and appropriate

Table 5-52. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Bis(2-ethylhexyl)phthalate Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Anadromous Sturgeon	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of DEHP indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET is considered to be relevant and appropriate
Estuarine/Marine Fish	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of DEHP indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • RQ directly applicable because CTET used is for *Oryzias melastigma* as presented in Table 5-50 above
Beetle and Aquatic Insect	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of DEHP indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • RQ directly applicable because CTET used is for *Chironomus riparius* as presented in Table 5-50 above
Amphibian	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of DEHP indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Studies were not identified to confirm risk conclusions; surrogate CTET is accepted as relevant and appropriate
Snakes and Other Reptiles	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of DEHP indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data were identified for reptiles; surrogate CTET for avian wildlife is considered representative and appropriate

Table 5-52. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Bis(2-ethylhexyl)phthalate Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Sea Turtle	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of DEHP indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data were identified for reptiles; surrogate CTET for avian wildlife is considered representative and appropriate
Coastal/Marine Bird	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of DEHP indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET is considered representative and appropriate
Marine Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of DEHP indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted
Terrestrial Mammal	Remote	• Exposed via water column and diet • Risk quotient indicates very low potential for direct chronic effects • Fate of DEHP indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET is considered representative and appropriate
Seagrass	Inconclusive	• Exposed via water column only • Risk could not be evaluated because of a lack of toxicity data
Freshwater- Saltwater Aquatic and Wetland Plants	Inconclusive	• Exposed via water column only • Risk could not be evaluated because of a lack of toxicity data

5.4.4 Nutrient-related Toxicity and Water Quality Effects

Nutrients have been measured in graywater and surface bilgewater/OWS effluent at concentrations that exceed WQC. Nutrient pollution is one of the leading causes of water quality impairment in the nation, primarily because the quantity of nutrients reaching the nation's waters has dramatically increased over the past 50 years (USEPA, 2009a). Nutrient loadings, particularly nitrogen and phosphorus, to waterbodies impact water quality by stimulating plant and algae growth which subsequently may result in depletion of dissolved oxygen, degradation of habitat, development of harmful algal blooms, impairment of the waterbody's designated use, and (for freshwater bodies) impairment of drinking water sources. In cases where the waterbody is shown to be impaired, it is important to identify the causative agent, whether it is nutrients or another factor such as hydrologic conditions, and then determine the limiting nutrient (WERF, 2010).

Generally, nitrogen is most often the limiting nutrient in estuarine waters, and phosphorus is more often limiting in freshwater systems. This means that the growth of phytoplankton is substantially controlled by the concentration and availability of nitrogen in marine/estuarine systems and by phosphorus in freshwater systems. In freshwater systems, increased phosphorus concentrations can lead to changes in composition of flora and fauna present, increased eutrophication of a water body, rates of ecosystem functioning, nutrient uptake, recycling rates of the ecosystem, and decomposition rates (WERF, 2010). Determining risk to aquatic life from excess nutrients (e.g., eutrophication) is complicated because nitrogen and phosphorus are essential for primary production in aquatic ecosystems, and over-enrichment problems involve multiple interrelated variables.
The most visible symptom of eutrophication is the excessive algal growth that reduces water clarity. Eutrophication can also significantly affect phytoplankton community structure resulting in a greater abundance of less desirable taxa such as blue-green algae. These changes in the phytoplankton community can have cascading effects on higher trophic levels and the eventual transfer of organic carbon from the primary producers to less desired species – for example, the replacement of seagrasses with less desirable vegetation types (WERF, 2010).

5.4.4.1 Phosphorus

Total phosphorus is detected in bilgewater and graywater discharge from vessels of the Armed Forces at concentrations that exceed WQC. Increases in phosphorus concentrations in ambient surface water can lead to a decreased photic zone and an increase in turbidity, local extinction of intolerant or specialized aquatic flora, higher pH, and a decrease in dissolved oxygen (WERF, 2010). These changes in turn can cause loss of habitat, change in food resources, and other impacts that could affect aquatic and aquatic-dependent species, including federally listed species, particularly in freshwater habitats. Generally, there would be little, if any, impact to listed species from total phosphorus in brackish estuaries and coastal waters where nitrogen is usually the limiting factor.

Under natural conditions, freshwater ecosystems generally have low phosphorous concentrations (< 100 µg/L), but each waterbody is different, and there are numerous factors that impact how any particular waterbody will respond to excess nutrient loading, including hydraulic residence time, freshwater inflow, clarity and light attenuation, geologic substrate, depth, temperature, and degree of physical alterations, such as channelization. Additionally, increases in phosphorus and the consequences associated with nutrient enrichment are generally widespread, may be manifested at a location that is remote from the source(s), and may not become apparent for some time after inputs to the system have occurred (WERF, 2010).

For these reasons, the EPA and DoD are using the lowest of the available waterbody-specific total phosphorus WQS developed across all of the states as the basis for $CTET_{A,W}$ for the analysis. The lowest estuarine/marine and freshwater total phosphorus WQC identified are 5.4

µg/L and 5.0 µg/L, respectively (https://www.epa.gov/nutrient-policy-data/state-progress-toward-developing-numeric-nutrient-water-quality-criteria accessed September 2017).

5.4.4.1.1 Direct Effects from Total Phosphorus

Table 5-53 summarizes the approved and proposed state WQC values for total phosphorus available for calculating the risk quotients for aquatic and aquatic-dependent species exposed to total phosphorus. Table 5-54 presents the phosphorus exposure concentrations estimated for ambient receiving waters from the harbor modeling. Table 5-54 also presents the RQs calculated for each taxonomic group potentially exposed to phosphorus in relation to the lowest WQC for total phosphorus.

The estimated RQ for the freshwater environment and its aquatic organisms from exposure to total phosphorus discharged from vessels of the Armed Forces is 6.4E-08 indicating that risk is “remote”. For the saltwater environment and its aquatic organisms, the RQ from exposure to maximum concentrations of total phosphorus discharged from vessels of the Armed Forces is 0.000043, also indicating that risk is “remote”. Because phosphorus does not bioaccumulate through the food chain and because phosphorus is an essential nutrient, risk to aquatic-dependent wildlife was evaluated based on the maximum “safe” level for phosphate in drinking water of 5 mg/L for human consumption established by the World Health Organization. The risk from exposure of aquatic-dependent birds and mammals to phosphorus in drinking of surface water alone because total phosphorus is “remote” based on comparison of modeled surface water concentrations with the WHO standard that results in an RQ of 6.4E-08.

Table 5-53. Summary of Approved and Proposed Total Phosphorus Water Quality Criteria for United States States and Territories

State/Territory	Waterbody Type	Min of WQC (mg/L)	Max of WQC (mg/L)
American Samoa	Estuaries	0.015	0.03
	Lakes/Reservoirs	0.15	0.15
	Rivers/Streams	0.15	0.15
Arizona	Lakes/Reservoirs	0.115	0.125
	Rivers/Streams	0.05	2.5
California	Lakes/Reservoirs	0.008	0.3
	Rivers/Streams	0.005	0.25
Colorado	Lakes/Reservoirs	0.0074	0.03
Florida	Estuaries	0.0054	0.31
	Lakes/Reservoirs	0.01	0.05
	Rivers/Streams	0.06	0.49
Georgia	Lakes/Reservoirs	0.092	2.02
Hawaii	All Embayments	0.02	0.075
	Estuaries	0.025	0.2
	Ocean Waters	0.01	0.025
	Open Coastal Waters	0.016	0.06
Illinois	Lakes/Reservoirs	0.007	0.05

Table 5-53. Summary of Approved and Proposed Total Phosphorus Water Quality Criteria for United States States and Territories (Continued)

State/Territory	Waterbody Type	Min of WQC (mg/L)	Max of WQC (mg/L)
Minnesota	Lakes/Reservoirs	0.012	0.09
Missouri	Lakes/Reservoirs	0.007	0.031
	Rivers/Streams	0.012	0.026
Montana	Rivers/Streams	0.02	0.039
Nebraska	Lakes/Reservoirs	0.04	0.05
Nevada	Lakes/Reservoirs	0.025	0.33
	Rivers/Streams	0.05	0.33
	Wetlands	0.1	0.33
New Jersey	Lakes/Reservoirs	0.05	0.05
	Rivers/Streams	0.1	0.1
New Mexico	Lakes/Reservoirs	0.1	0.1
Northern Marianas Islands	Fresh Surface Waters	0.1	0.1
	Marine Waters	0.025	0.05
Oklahoma	Lakes/Reservoirs	0.0141	0.0168
	Rivers/Streams	0.037	0.037
Oregon	Lakes/Reservoirs	0.241	0.241
Puerto Rico	Estuaries	1	1
	Lakes/Reservoirs	0.026	0.026
	Rivers/Streams	0.16	0.16
South Carolina	Lakes/Reservoirs	0.02	0.09
U.S. Virgin Islands	All waters	0.05	0.05
Utah	Aquatic Wildlife	0.05	0.05
	Recreation and Aesthetics	0.05	0.05
Vermont	Lakes/Reservoirs	0.01	0.054
	Rivers/Streams	0.01	0.01
Virginia	Lakes/Reservoirs	0.01	0.04
West Virginia	Lakes/Reservoirs	0.03	0.04
Wisconsin	Lakes/Reservoirs	0.005	0.04
All Marine/Estuarine Waterbodies		0.0054	1.0
All Fresh Waterbodies		0.005	2.5

Table 5-54. Risk Quotients for Total Phosphorus Based on Comparison of Exposure Concentrations with the Lowest State Water Quality Criteria

Taxa	EC_W (µg/L)	Lowest WQC (µg/L)	$RQ_{A,W}$
Freshwater Aquatic Organisms	3.2E-07	5.0	6.4E-08
Marine/Estuarine Aquatic Organisms	0.0025	5.4	0.00046
Aquatic-Dependent Wildlife	3.2E-07	5,000[1]	6.0E-11

[1] Represents the World Health Organization maximum "safe" level for phosphate in drinking water for human consumption

5.4.4.1.2 Indirect Effects from Total Phosphorus

The information summarized in Table 5-54 indicates that there is "remote" risk that maximum total phosphorus concentrations modeled for ambient receiving waters (fresh waterbodies in particular) will have direct effects on the plant and animal communities. As such, it can also be concluded that indirect effects typically associated with the loss of aquatic and riparian vegetation, including loss of cover, loss of food resources, and changes in water quality, are not expected. Therefore, risk to federally listed species from such indirect effects is also "remote".

5.4.4.1.1 Risk Conclusion for Each Taxonomic Group of Listed Species in the Representative Action Areas from Total Phosphorus

Based on the exposure assessment for aquatic and aquatic-dependent species for direct and indirect effects from total phosphorus, the following risk conclusions are being made for each taxonomic group of listed species in the RAAs (see Table 5-55). This information is used to inform decisions in the effects determinations made in Section 8 of this document.

Table 5-55. Summary of Risk Conclusions for Listed Species in Representative Action Areas from Exposure to Total Phosphorus Discharged from Vessels of the Armed Forces to Ports and Harbors

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Saltwater Corals	Remote	• Exposed via water column only • Phosphorus is only limiting in freshwater ecosystems • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Unionid Mussel	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species

Table 5-55. Summary of Risk Conclusions for Listed Species in Representative Action Areas from Exposure to Total Phosphorus Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Freshwater Snail	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Saltwater Mollusk	Remote	• Exposed via water column only • Phosphorus is only limiting in freshwater ecosystems • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Freshwater Shrimp/ Crustacean	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Freshwater Fish/ Inland Salmonid	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Anadromous Salmonid	Remote	• Exposed via water column only • Phosphorus is only limiting in freshwater ecosystems • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Freshwater Fish/Inland Sturgeon	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Anadromous Sturgeon	Remote	• Exposed via water column only • Phosphorus is only limiting in freshwater ecosystems • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species

Table 5-55. Summary of Risk Conclusions for Listed Species in Representative Action Areas from Exposure to Total Phosphorus Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Estuarine/Marine Fish	Remote	• Exposed via water column only • Phosphorus is only limiting in freshwater ecosystems • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Beetle and Aquatic Insect	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Amphibian	Remote	• Exposed directly via water column and through water ingestion • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Snakes and Other Reptiles	Remote	• Exposed directly via water column and through water ingestion • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Sea Turtle	Remote	• Exposed via water column only • Phosphorus is only limiting in freshwater ecosystems • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Coastal/Marine Bird	Remote	• Exposed through water ingestion only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Marine Mammal	Remote	• Exposed through water ingestion only • Phosphorus is only limiting in freshwater ecosystems • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species

Table 5-55. Summary of Risk Conclusions for Listed Species in Representative Action Areas from Exposure to Total Phosphorus Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Terrestrial Mammal	Remote	• Exposed through water ingestion only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Seagrass	Inconclusive	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Freshwater- Saltwater Aquatic and Wetland Plants	Inconclusive	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species

5.4.4.2 Nitrogen

Nitrogen is detected in bilgewater and graywater discharge from vessels of the Armed Forces at concentrations that exceed WQC. Nitrogen is essential for sustaining life, and nitrogen is usually the limiting nutrient for phytoplankton growth in estuarine/marine ecosystems. However, excess nitrogen in water causes health and environmental risks. Three forms of nitrogen in the environment, nitrate, nitrite and ammonia, are toxic. Of these three forms, ammonia is the most toxic. Nitrite is generally more toxic aquatic organisms than nitrate, and freshwater organisms seem to be more sensitive to different forms of nitrogen than marine organisms (Camargo et al., 2005).

Ammonia is produced by the biodegradation of organic material. Ammonia is then either take up by plants or oxidized to form nitrite and then nitrate. Usually ammonia nitrogen is used directly as a nutrient in aquatic systems by bacteria such as *Nitrosomonas*, which oxidize ammonia to nitrite. Blue green algae can fix atmospheric nitrogen to produce ammonium. Many aquatic plants take up ammonium preferentially over nitrate as a nitrogen source (Walstad, 2003 as cited in EPA, 2008).

Toxicity data for nitrogen are fairly limited, with the most information being available for ammonia and the least amount of information being available for nitrite. Although ammonia, nitrate and nitrite can be toxic, the main environmental problem caused by elevated nitrogen in the aquatic environment is eutrophication. In nutrient-enriched waters, the overabundance of nutrients stimulates growth of algae and aquatic plants. This can lead to lead to a decreased photic zone, increased turbidity, and a decrease in dissolved oxygen, all of which can cause loss of habitat, changes in food resources, fish kills, and other impacts that could affect aquatic and aquatic-dependent species, including listed species.

Eutrophication caused by nutrients promotes the deterioration of seagrass meadows, which are an important habitat for many species. Eutrophication can cause the growth of epiphytic and drift microalgae, which inhibit the growth of seagrass. Nitrate toxicity can also inhibit seagrass health (TPW, 1999). Seagrass beds are critical coastal nursery habitat for estuarine fisheries and wildlife and are a direct food sources for fish, birds, and sea turtles. They contribute organic matter to estuarine and marine food webs, participate in nutrient cycling processes, and can help stabilize coastal sedimentation and erosion processes (TPW, 1999).

For these reasons, the EPA and DoD are using the lowest of the available waterbody-specific total nitrogen WQS being developed across all of the states as the basis for $CTET_{A,W}$ for the analysis. The lowest estuarine/marine and freshwater total nitrogen WQC identified are 50 μg/L and 10 μg/L, respectively (https://www.epa.gov/nutrient-policy-data/state-progress-toward-developing-numeric-nutrient-water-quality-criteria accessed September 2017). The toxicity of ammonia nitrogen is discussed further in Section 5.3.4.3 below. This section focuses on the assessment of total nitrogen and discusses the toxicity of nitrate and nitrite. The toxicity and risks from ammonia nitrogen are discussed in Section 5.3.4.3 below.

5.4.4.2.1 Direct Effects from Total Nitrogen

Table 5-56 summarizes the approved and proposed state WQC values for nitrate, nitrite, and total nitrogen available. Only total nitrogen WQC were used for calculating the risk quotients for aquatic and aquatic-dependent species exposed to nitrogen in surface water. Table 5-57 presents the total nitrogen exposure concentrations estimated for ambient receiving waters from the harbor modeling. Table 5-57 also presents the RQs calculated for each taxonomic group potentially exposed to nitrogen in relation to the lowest WQC for total nitrogen.

The estimated RQ for the freshwater environment and its aquatic organisms from exposure to total nitrogen discharged from vessels of the Armed Forces is 3.2E-08, indicating that risk is "remote". For the saltwater environment and its aquatic organisms, the RQ from exposure to maximum concentrations of total nitrogen discharged from vessels of the Armed Forces is 4.6E-06, also indicating that risk is "remote". Because nitrogen does not bioaccumulate through the food chain and because nitrogen is an essential element, risk to aquatic-dependent wildlife was evaluated based on exposure through water consumption. The $CTET_{O}$s used to evaluate risk to aquatic-dependent wildlife are the EPA maximum contaminant levels (MCLs) for nitrate as nitrogen (10 mg N-NO3/L) and nitrite as nitrogen (1 mg N-NO2/L) in drinking water for human consumption. The risk from exposure of aquatic-dependent birds and mammals to total nitrogen in drinking of surface water is "remote" based on comparison of modeled surface water concentrations of total nitrogen (conservatively assuming all nitrogen is either nitrate or nitrite) with the EPA MCL standard that results in an RQ of 3.2E-11 for nitrate and 3.2E-10 for nitrite.

Table 5-56. Summary of Approved and Proposed Nitrogen Water Quality Criteria for United States States and Territories

State/Territory	Waterbody Type	Nitrate (mg/L)		Nitrite (mg/L)		Nitrate + Nitrite (mg/L)		Total Nitrogen (mg/L)	
		Min WQC	Max WQC	Min WQC	Max WQC	Min WQC	Max WQC	Min WQC	Max WQC
American Samoa	Estuaries	--	--	--	--	--	--	0.135	0.2
	Lakes/Reservoirs	--	--	--	--	--	--	0.3	0.3
	Rivers/Streams	--	--	--	--	--	--	0.3	0.3
Arizona	Lakes/Reservoirs	--	--	--	--	--	--	1.6	1.7
	Rivers/Streams	--	--	--	--	--	--	0.1	3
California	Lakes/Reservoirs	5	6	--	--	--	--	0.087	4
	Rivers/Streams	0.25	0.25	--	--	--	--	0.01	10
Florida	Estuaries	--	--	--	--	--	--	0.087998	1.29
	Lakes/Reservoirs	--	--	--	--	--	--	0.51	1.27
	Rivers/Streams	--	--	--	--	--	--	0.67	1.87
Georgia	Lakes/Reservoirs	--	--	--	--	--	--	3	4
Guam	Estuaries[1]	0.1	0.5	--	--	--	--	--	--
	Lakes/Reservoirs[1]	0.1	0.5	--	--	--	--	--	--
	Rivers/Streams[1]	0.1	0.5	--	--	--	--	--	--
Hawaii	All Embayments	--	--	--	--	0.005	0.035	0.15	0.5
	Estuaries	--	--	--	--	0.008	0.07	0.2	0.75
	Marine Waters[2]	--	--	--	--	0.0045	0.0045	0.1	0.1
	Ocean Waters	--	--	--	--	0.0015	0.0035	0.05	0.1
	Open Coastal Waters	--	--	--	--	0.0035	0.025	0.11	0.35
Illinois	Lakes/Reservoirs	10	10	--	--	--	--	--	--
Massachusetts	Lakes/Reservoirs	--	--	--	--	--	--	0.38	0.38
Missouri	Lakes/Reservoirs	--	--	--	--	--	--	0.2	0.616
Montana	Rivers/Streams	--	--	--	--	--	--	0.3	0.3
Nebraska	Lakes/Reservoirs	--	--	--	--	--	--	0.8	1

Table 5-56. Summary of Approved and Proposed Nitrogen Water Quality Criteria for United States States and Territories (Continued)

State/Territory	Waterbody Type	Nitrate (mg/L)		Nitrite (mg/L)		Nitrate + Nitrite (mg/L)		Total Nitrogen (mg/L)	
		Min WQC	Max WQC	Min WQC	Max WQC	Min WQC	Max WQC	Min WQC	Max WQC
Nevada	Lakes/Reservoirs	10	90	0.06	5	--	--	0.25	1
	Rivers/Streams	0.18	100	0.04	10	--	--	0.2	6.1
New Jersey	Lakes/Reservoirs	2	2	--	--	--	--	--	--
	Rivers/Streams	2	2	--	--	--	--	--	--
New York	Lakes/Reservoirs[3]	--	--	0.02	0.1	--	--	--	--
	Rivers/Streams[3]	--	--	0.02	0.1	--	--	--	--
	Wetlands[3]	--	--	0.02	0.1	--	--	--	--
Northern Marianas Islands	Fresh Surface Waters	--	--	--	--	--	--	0.75	1.5
	Marine Waters[1]	0.2	0.5	--	--	--	--	0.4	0.75
South Carolina	Lakes/Reservoirs	--	--	--	--	--	--	0.35	1.5
Utah	Aquatic Wildlife[1]	4	4	--	--	--	--	--	--
	Recreation and Aesthetics[1]	4	4	--	--	--	--	--	--
Vermont	Lakes/Reservoirs	5	5	--	--	--	--	--	--
	Rivers/Streams	0.2	5	--	--	--	--	--	--
Min and Max for All States and Territories		**0.18**	**100**	**0.04**	**10**	**0.0015**	**0.35**	**0.01**	**10**
All Estuarine/Marine		**0.1**	**0.5**	**--**	**--**	**0.0015**	**0.07**	**0.05**	**1.29**
All Freshwater		**0.1**	**100**	**0.02**	**10**	**0**	**0**	**0.01**	**10**

1 Nitrate reported as N only

2 Total Nitrogen is for dissolved form

3 Nitrite reported as N only

Table 5-57. Risk Quotients for Total Nitrogen Based on Comparison of Exposure Concentrations with the Lowest State Water Quality Criteria

Direct Exposure to Total Nitrogen			
Taxa	**EC_W (µg/L)**	**Lowest WQC (µg/L)**	**$RQ_{A,W}$**
Freshwater Aquatic Organisms	3.2E-07	10	3.2E-08
Marine/Estuarine Aquatic Organisms	0.05	50	0.001
Exposure to Total Nitrogen in Drinking Water			
Taxa - Exposure	**EC_W (µg/L)**	**EPA MCL (µg/L)[1]**	**$RQ_{A,W}$**
Aquatic-dependent Wildlife – Nitrate	4.8E-08	10,000	4.8E-12
Aquatic-dependent Wildlife – Nitrite	4.8E-08	1,000	4.8E-11

[1] https://www.wqa.org/Portals/0/Technical/Technical%20Fact%20Sheets/2014_NitrateNitrite.pdf

5.4.4.2.2 Indirect Effects from Total Nitrogen

The information summarized in Table 5-57 indicates that there is "remote" risk that maximum total nitrogen concentrations modeled for ambient receiving waters (fresh waterbodies in particular) will have direct effects on the plant and animal communities. As such, it can also be concluded that indirect effects typically associated with the loss of aquatic and riparian vegetation, including loss of cover, loss of food resources, and changes in water quality, are not expected. Therefore, risk to federally listed species from such indirect effects is also "remote".

5.4.4.2.3 Risk Conclusion for Each Taxonomic Group of Listed Species in the Representative Action Areas from Total Nitrogen

Based on the exposure assessment for aquatic and aquatic-dependent species for direct and indirect effects from total nitrogen, the following risk conclusions are being made for each taxonomic group of listed species in the RAAs (see Table 5-58). This information is used to inform decisions in the effects determinations made in Section 8 of this document.

Table 5-58. Summary of Risk Conclusions for Listed Species in Representative Action Areas from Exposure to Total Nitrogen Discharged from Vessels of the Armed Forces to Ports and Harbors

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Saltwater Corals	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species • Nutrient effects in corals is complicated because increased nitrogen could stimulate growth of coral zooxanthellae until nitrogen and phosphorus become unbalanced (D'Angelo and Weidenmann, 2013)

Table 5-58. Summary of Risk Conclusions for Listed Species in Representative Action Areas from Exposure to Total Nitrogen Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Unionid Mussel	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species • Risk conclusion is confirmed by laboratory and field studies conducted and cited by Douda (2010) that should the distribution of three species of unionid mussels was dependent on N-NO3 concentrations with a NOEC of 2 mg/L. Resulting RQ from comparison of total nitrogen concentrations with the study NOEC is 2.0E-10.
Freshwater Snail	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species • Risk conclusion is confirmed using a study of *Potamopyrgus antipodarum* with a NOEC of 21.4 mg N-NO3/L for movement velocity and a LOEC of 21.4 mg N-NO3/L for reproduction (Alonso and Camargo, 2013). Resulting RQ is 1.5E-11.
Saltwater Mollusk	Remote	• Exposed via water column only • Phosphorus is only limiting in freshwater ecosystems • Risk quotient indicates very low potential for direct chronic effects • A 15-day study of inorganic nitrogen on the growth of juvenile green ormer (*Haliotis tuberculate*) identified a safe level of 250 mg/L (Basuyaux and Mathieu, 1999 as cited in Camargo et al., 2005). Resulting RQ from comparing total nitrogen concentrations with the safe level is 2.0E-07.
Freshwater Shrimp/ Crustacean	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species • Risk conclusion confirmed using a 7-day reproduction study with *Ceriodaphnia dubia* that resulted in an IC25 of 50 mg NO3/L (Elphick, 2011 as cited in CCME, 2012). Resulting RQ from comparing total nitrogen concentrations with the IC25 is 6.4E-12.

Table 5-58. Summary of Risk Conclusions for Listed Species in Representative Action Areas from Exposure to Total Nitrogen Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Freshwater Fish/ Inland Salmonid	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species • Risk conclusion is confirmed using a 30-day study of *O. mykiss* eggs that resulted in a NOEC of 1.1 mg N-NO3/L (Kincheloe et al., 1979 as cited in Camargo et al., 2005). Resulting RQ from comparing total nitrogen concentrations with the NOEC is 2.9E-10
Anadromous Salmonid	Remote	• Exposed via water column only • Phosphorus is only limiting in freshwater ecosystems • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species • No chronic studies were identified for salmonids in saltwater; however, a 30-day study of anadromous *O. mykiss* eggs resulted in a NOEC of 1.1 mg N-NO3/L (Kincheloe et al., 1979 as cited in Camargo et al., 2005). Resulting RQ from comparing total nitrogen concentrations in seawater with the NOEC is 4.5E-05.
Freshwater Fish/Inland Sturgeon	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Anadromous Sturgeon	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Estuarine/Marine Fish	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species • Risk conclusion is confirmed using a 72-day growth and mortality study with anemone fish (*Amphiprion ocellaris*) that resulted in a LOEC of 443 mg NO3/L (Frakes and Hoff Jr., 1982 as cited in CCME 2012). Resulting RQ comparing total nitrogen concentrations with the LOEC is 1.1E-07.

Table 5-58. Summary of Risk Conclusions for Listed Species in Representative Action Areas from Exposure to Total Nitrogen Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Beetle and Aquatic Insect	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species • Risk conclusion is confirmed using a 120-hour study of caddisfly (*Hydropsyche occidentalis*) early instar larvae that resulted in an LC0.01 of 4.5 mg N-NO3/L (Camargo and Ward, 1995 as cited in Camargo et al, 2005). Resulting RQ comparing total nitrogen concentrations with the LC0.01 is 7.1E-11.
Amphibian	Remote	• Exposed directly via water column and through water ingestion • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species • Risk conclusion is confirmed using a 10-week larval study with *Rana temporaria* that resulted in a NOEC of 5.0 mg N-NO3/L (Johansson et al., 2001 as cited in Camargo et al., 2005). Resulting RQ comparing total nitrogen concentrations with the NOEC is 6.4E-11.
Snakes and Other Reptiles	Remote	• Exposed directly via water column and through water ingestion • Risk quotient indicates very low potential for direct chronic effects from water ingestion • In the absence of reptile data, CTET is considered relevant and appropriate
Sea Turtle	Remote	• Exposed via water column only • Effects are expected to be indirect through effects on sea turtle habitat and food resources • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Coastal/Marine Bird	Remote	• Exposed through water ingestion only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species • In the absence of reptile data, CTET is considered relevant and appropriate
Marine Mammal	Remote	• Exposed through water ingestion only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species

Table 5-58. Summary of Risk Conclusions for Listed Species in Representative Action Areas from Exposure to Total Nitrogen Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Terrestrial Mammal	Remote	• Exposed through water ingestion only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Seagrass	Inconclusive	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species
Freshwater- Saltwater Aquatic and Wetland Plants	Inconclusive	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Risk quotient is inclusive of direct and indirect chronic effects to all plant and animal species

5.4.4.3 Total Ammonia Nitrogen

Ammonia (expressed as total ammonia nitrogen) is a constituent found in bilgewater and graywater at concentrations that may exceed WQC (Section 3.2.3.2). Although it is produced by human activity, ammonia also occurs naturally and it is found throughout the environment in the air, soil, and water. Ammonia is excreted by animals, including humans, and is rapidly taken up by plants, bacteria, and animals where it is an important source of nitrogen. Because of this, ammonia does not build up in the food chain, but it does serve as a nutrient for plants and bacteria (USEPA, 2009b).

The pollutant form of ammonia in water consists of two species: a larger component which is the ammonium ion (NH4+), and a smaller component which is the non-dissociated or un-ionized ammonia (NH3) molecule. The sum of the two forms is usually expressed as total ammonia-nitrogen. The ratio of un-ionized ammonia to ammonium ion, which depends upon both pH and temperature, generally increases by 10-fold for each rise of a single pH unit and by approximately 2-fold for each 10°C rise in temperature over the 0-30°C range (Erickson, 1985). Toxicity of ammonia to aquatic life was initially thought to arise largely from the small uncharged NH3 molecule (Wuhrmann and Woker, 1948, Downing and Merkens, 1955), but the ammonium ion has since proven to be toxic as well, particularly at low pH (Armstrong et al., 1978; Tomasso et al., 1980); thus, toxicity of ammonia is best expressed as a function of pH and/or temperature and on the basis of total ammonia nitrogen (USEPA, 2009b).
For this purpose of comparison and risk calculation in this BE, $CTET_{A,WS}$ for freshwater aquatic organisms are expressed on the basis of total ammonia nitrogen at pH 8.0 (freshwater vertebrates), or, pH 8 and 25°C (freshwater invertebrates) where a functional relationship between toxicity and pH and/or temperature has been demonstrated (USEPA, 2009b).

Conversely, $CTET_{A,W}$s for saltwater aquatic vertebrates and invertebrates are expressed simply as total ammonia nitrogen.

5.4.4.3.1 Direct Effects from Total Ammonia Nitrogen

Table 5-59 summarizes the chronic toxicity data available for calculating $RQ_{A,W}$ for aquatic species exposed to modeled concentrations of total ammonia in water receiving from discharges from vessels of the Armed Forces. Table 5-60 presents the estimated exposure concentrations modeled from the harbor modeling, as well as the corresponding $RQ_{A,W}$s calculated for each taxonomic group based on the various chronic toxicity threshold values (in this case, $CTET_{A,W}$) readily available for representative surrogate taxa.

Risk to Freshwater Aquatic Animal and Plant Species from Total Ammonia Nitrogen

The risk to freshwater aquatic organisms from exposure to total ammonia nitrogen in ambient receiving water resulting from discharges from vessels of the Armed Forces is “remote”. The $RQ_{A,W}$ for freshwater vertebrates, invertebrates, and plants are 1.2E-11, 5.3E-11, and 1.3E-11, respectively. Because ammonia does not have a tendency to bioaccumulate, potential risks from bioaccumulated ammonia were not evaluated.

Risk to Saltwater (Estuarine/Marine) Aquatic Animal and Plant Species from Total Ammonia Nitrogen

The risk to saltwater aquatic organisms from exposure total ammonia nitrogen in ambient receiving water from discharges from vessels of the Armed Forces is also “remote”. Using the maximum modeled exposure concentrations, $RQ_{A,W}$ for saltwater organisms are 0.000015 for marine vertebrates and 0.000018 for marine invertebrates. The $RQ_{A,W}$ for saltwater vascular plants is 0.0002, also indicating that “remote”.

Risk to Birds and Mammals from Total Ammonia Nitrogen

Because ammonia does not bioaccumulate in prey, the only pathway by which aquatic-dependent birds and mammals can by exposed to total ammonia nitrogen is via (fresh) drinking water consumption. However, as noted above, total ammonia has a short half-life in water, and therefore, does not remain readily available uptake by organisms. In addition, a chronic toxicity threshold for dietary exposure to ammonia was not identified; therefore, because of the behavior of ammonia in the environment, the EPA and DoD assumes “remote” risk to wildlife via oral ingestion.

Table 5-59. Summary of Chronic Toxicity Effect Thresholds for Aquatic and Aquatic-Dependent Organisms Exposed to Total Ammonia Nitrogen

Direct Exposure to Total Ammonia Nitrogen					
Surrogate Aquatic Species Type	**Surrogate Species**	**Exposure Effect Endpoint**	**NOEC - LOEC (mg/L)**	**$CTET_{A,W}$ (mg/L)**	**Study (source)**
Freshwater Vertebrate	Bluegill sunfish, *Lepomis macrochirus*	ELS Test – biomass	-	1.349 (EC20)	Smith et al. 1984 (2009 Draft Update ALC document)
Freshwater Invertebrate	Fatmucket, *Lampsilis siliquoidea*	28-day Juvenile Test - survival	-	0.3027 (LC20)	Wang et al. 2007 (2009 Draft Update ALC document)
Freshwater Plant (Vascular)	-	-	-	1.2	CCC from ALC Update document (USEPA, 1999)
Estuarine/Marine Vertebrate	Inland silverside, *Menidia berylina*	ELS Test – growth	2.1-4.2	2.970 (2.446)	Miller et al., 1990 (2004 GLEC Update File)
Estuarine/Marine Invertebrate	Greenlip abalone, *Haliotis laevigata*	95-day Test – growth	1.46-2.65	1.97	Harris et al. 1998
Estuarine/Marine Plant (Vascular)	Eelgrass, *Zostera marina*	5-week Test – growth and mortality	-	0.1533	vanKatwijk et al. 1997
Dietary Exposure to Total Ammonia Nitrogen					
Surrogate Aquatic-Dependent Species Type	**Surrogate Species**	**Exposure Type**	**Effect Endpoint**	**$CTET_o$ (mg/kg bw/d)**	**Study (source)**
Mammals	Rodents	-	-	-	-
Birds	Not available	-	-	-	-

$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures

$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)

Table 5-60. Risk Quotients for Total Ammonia Nitrogen Based on Comparison of Exposure Concentrations with Chronic Toxicity Effect Thresholds of Representative Surrogate Taxa

Taxa	EC_W (µg/L)	$CTET_{A,W}$ (µg/L)	$RQ_{A,W}$
Freshwater Vertebrate	1.6E-08	1349	1.2E-11
Freshwater Invertebrate		302.7	5.3E-11
Freshwater Plant (Vascular)		1200	1.3E-11
Estuarine/Marine Vertebrate	0.036	2446	0.000015
Estuarine/Marine Invertebrate - Other		2000	0.000018
Saltwater Vascular Plant		--	Not calculated
Taxa	**EC_O (mg/kg bw/d)**	**$CTET_O$ (mg/kg/d)**	**RQ_{wild}**
Mammals (Freshwater)	3.2E-13	--	Not calculated
Birds (Freshwater)	1.6E-12	--	Not calculated
Mammals (Estuarine/Marine)	0	--	0
Birds (Estuarine/Marine)	0	--	0

EC_W – Exposure concentration for water
EC_T – Exposure concentration accumulated in tissue
EC_O – Exposure concentration via oral ingestion (diet)
$CTET_{A,W}$ – Chronic toxicity effects threshold for ambient water exposures
$CTET_O$ – Chronic toxicity effects threshold for exposure via oral ingestion (diet)
$RQ_{A,W}$ – Risk quotient from exposure to total ammonia nitrogen in water calculated as EC_W/ $CTET_{A,W}$
RQ_{Wild} – Risk quotient from exposure to total ammonia nitrogen through diet calculated as EC_O/ $CTET_O$

5.4.4.3.2 Indirect Effects for Total Ammonia Nitrogen

Discharge of ammonia from vessels of the Armed Forces will not result in appreciable concentrations in estuaries and freshwater receiving water bodies. As such, the potential for direct effects to aquatic populations in receiving waters is “remote”. Therefore, there is no basis for assuming indirect effects on listed species from exposure to total ammonia nitrogen and corresponding toxicity-related reduction in the prey base (loss of prey) available to upper trophic level organisms. Because the information summarized in Table 5-60 indicates that total ammonia nitrogen at maximum modeled exposure concentrations has little risk of directly affecting fresh- and saltwater aquatic animals, indirect effects typically associated with exposure, including changes in water quality, are also not expected for this metal.

5.4.4.3.3 Risk Conclusion for Each Taxonomic Group of Listed Species in the Representative Action Areas from Total Ammonia Nitrogen

Based on the exposure assessment for aquatic and aquatic-dependent species for direct and indirect effects from total ammonia nitrogen, the following risk conclusions are being made for each taxonomic group of listed species in the RAAs (see Table 5-61). This information is used to inform decisions in the effects determinations made in Section 8 of this document.

Table 5-61. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Total Ammonia Nitrogen Discharged from Vessels of the Armed Forces to Ports and Harbors

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Saltwater Corals	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of ammonia indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) because ammonia does not tend to accumulate in the environment • Surrogate CTET is representative and appropriate
Unionid Mussel	Remote	• Exposed via water column only • Risk quotient indicates low potential for direct chronic effects • Fate of ammonia indicates low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • RQ is directly applicable because CTET used is for *Lampsilis siliquoidea* as presented in Table 5-59 above
Freshwater Snail	Remote	• Exposed via water column only • Risk quotient indicates low potential for direct chronic effects • Fate of ammonia indicates low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a 60-day study of *Sphaerium novaezelandiae* with a NOEC for mortality of 0.01 mg N-NH3/L (Hickey and Martin, 1999 as cited in Alonso and Camargo, 2009). Resulting RQ is 1.6E-09.
Saltwater Mollusk	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of ammonia indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • RQ is directly applicable because CTET used is for *Haliotis laevigata* as presented in Table 5-59 above
Freshwater Shrimp/ Crustacean	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of ammonia indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Surrogate CTET is considered to be relevant and appropriate

Table 5-61. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Total Ammonia Nitrogen Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Freshwater Fish/ Inland Salmonid	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of ammonia indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a 62-day study of the hatchability of embryonic sockeye salmon (*Oncorhynchus nerka*) with a chronic value of <4.160 mg/L (Rankin, 1979). Resulting RQ is 3.8E-12.
Anadromous Salmonid	Remote	• Exposed via water column only • Risk quotient indicates low potential for direct chronic effects • Fate of ammonia indicates low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted • No chronic toxicity data are available for salmonids tested in seawater
Freshwater Fish/Inland Sturgeon	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of ammonia indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Risk conclusion is confirmed using acute toxicity data for shortnosed sturgeon (*Acipenser brevirostrum*) with an LC50 of 36.49 mg N/L (Fontenot et al. 1998). Dividing by the FACR of 6.8 (USEPA, 2009b) results in a chronic value of 5.37 mg/L. Resulting RQ is 3.1E-12.
Anadromous Sturgeon	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of ammonia indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • No chronic toxicity data are available for salmonids tested in seawater
Estuarine/Marine Fish	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of DEHP indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • RQ directly applicable because CTET used is for *Menidia berylina* as presented in Table 5-59 above

Table 5-61. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Total Ammonia Nitrogen Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Beetle and Aquatic Insect	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of ammonia indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Risk conclusion is confirmed using a 24-day juvenile study with the stonefly *Pteronarcella badia*, which resulted in a chronic value of 26.27 mg/L (Thurston et al. 1984 as cited in the 2013 ALC Update document). Resulting RQ is 6.0E-10.
Amphibian	Remote	• Exposed via water column and water ingestion • Risk quotient indicates very low potential for direct chronic effects • Fate of ammonia indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted
Snakes and Other Reptiles	Remote	• Exposed via water column and water ingestion • Risk quotient indicates very low potential for direct chronic effects • Fate of ammonia indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • No chronic toxicity data were identified for reptiles
Sea Turtle	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of ammonia indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted
Coastal/Marine Bird	Remote	• Exposed via water column and water ingestion • Risk quotient indicates very low potential for direct chronic effects • Fate of ammonia indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted
Marine Mammal	Remote	• Exposed via water column and water ingestion • Risk quotient indicates very low potential for direct chronic effects • Fate of ammonia indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) at the concentrations predicted

Table 5-61. Summary of Risk Conclusions for Listed Species Taxonomic Groups in Representative Action Areas from Exposure to Total Ammonia Nitrogen Discharged from Vessels of the Armed Forces to Ports and Harbors (Continued)

Listed Species Taxonomic Group	Risk Conclusion	Presumptions
Terrestrial Mammal	Remote	• Exposed via water column and water ingestion • Risk quotient indicates very low potential for direct chronic effects • Fate of ammonia indicates very low potential for indirect effects (loss of prey and change in habitat/water quality parameters) – at the concentrations predicted • Surrogate CTET is considered representative and appropriate
Seagrass	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of ammonia indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • directly applicable because CTET used is for *Zostera marina* as presented in Table 5-59 above
Freshwater- Saltwater Aquatic and Wetland Plants	Remote	• Exposed via water column only • Risk quotient indicates very low potential for direct chronic effects • Fate of ammonia indicates very low potential for indirect effects (loss of food resources and change in habitat/water quality parameters) at the concentrations predicted • Surrogate CTET is considered to be relevant and appropriate

5.5 Summary of Risk Conclusions for Critical Habitat

Table 5-62 lists the critical habitat within each of the RAAs, their essential features, the effects that pollutants in discharges from vessels of the Armed Forces could have on each critical habitat, and the risk that pollutants in vessel discharges could adversely affect critical habitat. Risk was assessed qualitatively as “negligible”, “remote”, “potentially significant”, or “likely significant” based on how likely the pollutants in the Batch Two discharges will affect that critical habitat. Risk of impact to all critical habitat evaluated was either “negligible” or “remote”, with risk of impact to aquatic-dependent species being “remote” and risks to aquatic species generally being “remote” or “negligible”. If any species and its critical habitat occurs only in a location where vessels of the Armed Forces do not operate, the risk is considered to be “remote” because, although there could be risk from other stressors, Batch Two discharges are not a stressor that could affect that species. This information is used to inform decisions in the effects determinations made in Section 8 of this document.

Table 5-62. Potential for Impacts to the Essential Features of Critical Habitat within the Representative Action Areas

Critical Habitat	RAA	Location and Primary Constituent Elements (PCEs)	Potential Effects	Risk of Impact
Bird's-beak, soft *Cordylanthus mollis ssp. mollis*	San Francisco, CA	• Persistent emergent, intertidal, estuarine wetland at or above the mean high-water line (as extended directly across any intersecting channels) • Rarity or absence of plants that naturally die in late spring (winter annuals) • Partially open spring canopy cover (approximately 790 nMol/m^2/s) at ground level, with many small openings to facilitate seedling germination	Batch Two discharges are not expected to have any impact on wetland vegetation, although exposure to pollutants in discharges is likely	Remote
Coral, Elkhorn *Acropora palmata*	Miami, FL	• 1,329 square miles offshore of Palm Bach in Broward, Miami-Dade, and Monroe Counties, FL • Substrate of suitable quality and availability to support larval settlement and recruitment and reattachment and recruitment of asexual fragments • Suitable substrate defined as natural consolidated hard substrate or dead coral skeleton that is free from fleshy or turf macroalgae cover and sediment cover	Batch Two discharges are not expected to have any impact to the suitability of substrate for elkhorn or staghorn coral; pollutants in discharges will not accumulate in consolidated hard substrate or dead coral skeleton that provides substrate to support settlement and recruitment of larvae or asexual coral fragments	Remote
Coral, Staghorn *Acropora cervicornis*	Miami, FL	• 1,329 square miles offshore of Palm Bach in Broward, Miami-Dade, and Monroe Counties, FL • Substrate of suitable quality and availability to support larval settlement and recruitment and reattachment and recruitment of asexual fragments • Suitable substrate defined as natural consolidated hard substrate or dead coral skeleton that is free from fleshy or turf macroalgae cover and sediment cover	Batch Two discharges are not expected to have any impact to the suitability of substrate for elkhorn or staghorn coral; pollutants in discharges will not accumulate in consolidated hard substrate or dead coral skeleton that provides substrate to support settlement and recruitment of larvae or asexual coral fragments	Remote
Crocodile, American *Crocodylus acutus*	Miami, FL	• All land and water (excluding structures) along the Florida coast south of Turkey Point, Biscayne Bay on the east coast and the northernmost point of Nine Mile Pond on the west coast • No PCEs identified, however, greatest threats are the availability of suitable nesting sites and interaction with humans in nesting areas, which could result in nest abandonment	Batch Two discharges are not expected to have any impact on the availability or quality of nesting sites; very limited and dilute petroleum hydrocarbons in bilgewater, graywater and deck runoff could reach coastal nesting sites, but the potential for that to occur is unlikely	Negligible

Table 5-62. Potential for Impacts to the Essential Features of Critical Habitat within the Representative Action Areas (Continued)

Critical Habitat	RAA	Location and Primary Constituent Elements (PCEs)	Potential Effects	Risk of Impact
Whipsnake, California *Masticophis lateralis euryxanthus*	San Francisco, CA	• Scrub/shrub communities with a mosaic of open and closed canopy • Woodland or annual grassland plant communities contiguous to lands containing PCE 1 • Lands containing rock outcrops, talus, and small mammal burrows within or adjacent to PCE 1 and or PCE	Batch Two discharges will not have any impact on the quality or availability of terrestrial habitat	Remote
Sturgeon, Atlantic *Acipenser oxyrinchus oxyrinchus*	Norfolk, VA	• Hard bottom substrate (e.g., rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (i.e., 0.0–0.5 ppt) for settlement of fertilized eggs, refuge, growth, and development of early life stages; • Aquatic habitat for juveninl foraging and development with a gradual downstream salinity gradient of 0.5 - 30 ppt and soft substrate (e.g., sand, mud) between the river mouth and spawning sites; • Water of appropriate depth (at least 1.2 m) and absent of physical barriers to passage (e.g., locks, dams, thermal plumes, turbidity, sound, reservoirs, gear, etc.) between the river mouth and spawning sites necessary to support: o Unimpeded movement of adults to and from spawning sites; o Seasonal and physiologically dependent movement of juvenile to appropriate salinity zones within the river estuary; and o Staging, resting, or holding of subadults or spawning condition adults. • Water, between the river mouth and spawning sites, especially in the bottom meter of the water column, with the temperature, salinity, and oxygen values that, combined, support: o Spawning;	Batch Two Discharges are not expected to impact substrate availability, salinity, water depth, or temperature. Underwater ship husbandry could increase BOD, but the change is not expected to be substantial because hull maintenance and cleaning practices do not allow vessels of the Armed Forces to become substantially fouled.	Negligible

Table 5-62. Potential for Impacts to the Essential Features of Critical Habitat within the Representative Action Areas (Continued)

Critical Habitat	RAA	Location and Primary Constituent Elements (PCEs)	Potential Effects	Risk of Impact
Sturgeon, Atlantic *Acipenser oxyrinchus oxyrinchus* (continued)	Norfolk, VA	o Annual and inter-annual adult, subadult, larval, and juvenile survival; and o Larval, juvenile, and subadult growth, development, and recruitment (e.g., 13 to 26 °C for spawning habitat and no more than 30 °C for juvenile rearing habitat, and 6 milligrams per liter (mg/L) or greater dissolved oxygen for juvenile rearing habitat).		
Frog, California red-legged *Rana draytonii*	San Francisco, CA	• Aquatic Breeding Habitat - Standing bodies of fresh water, including natural and manmade ponds, slow-moving streams or pools within streams, and other ephemeral or permanent water bodies that typically become inundated during winter rains and hold water for a minimum of 20 weeks in all but the driest of years. • Aquatic Non-Breeding Habitat - Freshwater pond and stream habitats, as described above, that may not hold water long enough for the species to complete its aquatic life cycle but which provide for shelter, foraging, predator avoidance, and aquatic dispersal of juveniles and adults. Other wetland habitats considered to meet these criteria include but are not limited to: plunge pools within intermittent creeks, seeps, quiet water refugia within streams during high water flows, and springs of sufficient flow to withstand short-term dry periods. • Upland Habitat - Upland areas adjacent to or surrounding breeding and non-breeding aquatic and riparian habitat up to a distance of 1 mi (1.6 km) including various vegetational types such as grassland, woodland, forest, wetland, or riparian areas that provide shelter, forage, and predator avoidance. Upland features are also essential in that they are needed to maintain the hydrologic, geographic, topographic, ecological, and edaphic features that support and surround the aquatic, wetland, or riparian habitat. Upland habitat should include structural features such as boulders, rocks	Batch Two discharges are not likely to impact any of the aquatic habitats used by the California red-legged frog; pollutants will not accumulate to levels that will have measurable effects on frogs	Remote

Table 5-62. Potential for Impacts to the Essential Features of Critical Habitat within the Representative Action Areas (Continued)

Critical Habitat	RAA	Location and Primary Constituent Elements (PCEs)	Potential Effects	Risk of Impact
Frog, California red-legged *Rana draytonii* (continued)	San Francisco, CA	and organic debris (e.g., downed trees, logs), small mammal burrows, or moist leaf litter. • Dispersal Habitat - Accessible upland or riparian habitat within and between occupied or previously occupied sites that are located within 1 mi (1.6 km) of each other, and that support movement between such sites. Dispersal habitat includes various natural habitats, and altered habitats such as agricultural fields, that do not contain barriers to dispersal.		
Goldfields, Contra Costa *Lasthenia conjugens*	San Francisco, CA	• Topographic features characterized by isolated mound and intermound complex within a matrix of surrounding uplands that result in continuously, or intermittently, flowing surface water in the depressional features including swales connecting the pools, providing for dispersal and promoting hydroperiods of adequate length in the pools • Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water or whose soils are saturated for a period long enough to promote germination, flowering, and seed production of predominantly annual native wetland species and typically exclude both native and nonnative upland plant species in all but the driest years. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands	Batch Two discharges will not impact any of the elements of Contra Costa goldfields critical habitat; pollutants in Batch Two discharges also will not accumulate in these locations	Remote
Manatee, West Indian *Trichechus manatus*	Miami, FL	• Coastal areas south of Jacksonville on the east coast of Florida and south of Tampa on the west coast • No PCEs identified, however, greatest threats are habitat loss and degradation, and mortality from boat collisions, hunting, fishing, red tide poisoning, entrapment in water control structures, entanglement in fishing gear, and exposure to cold temperatures.	Batch Two discharges are not expected to result in additional loss of habitat; the concentrations of pollutants modeled are unlikely to result in significant habitat degradation	Negligible

Table 5-62. Potential for Impacts to the Essential Features of Critical Habitat within the Representative Action Areas (Continued)

Critical Habitat	RAA	Location and Primary Constituent Elements (PCEs)	Potential Effects	Risk of Impact
Seal, Hawaiian Monk	Pearl Harbor, HI	• Terrestrial areas and adjacent shallow, sheltered aquatic areas with characteristics preferred by monk seals for pupping and nursing • Marine areas from 0 to 200 m in depth that support adequate prey quality and quantity for juvenile and adult monk seal foraging • Significant areas used by monk seals for hauling out, resting or molting • Does not include areas with DoD presence	Batch Two discharges are not expected to affect pupping and nursing areas or haul out areas for monk seals. Pollutants in discharges are also not expected to affect prey quantity and quality.	Remote
Murrelet, Marbled *Brachyramphus marmoratus*	Puget Sound, Seattle, WA San Diego, CA San Francisco , CA	• Individual trees with potential nesting platforms • Forested areas within 0.5 mile (0.8 kilometer) of individual trees with potential nesting platforms, and with a canopy height of at least one-half the site-potential tree height. This includes all such forest, regardless of contiguity.	Batch Two discharges are not expected to have any impact on murrelet critical habitat; PCEs will not be exposed to Batch Two discharges	Remote
Seagrass, Johnson's *Halophila johnsonii*	Miami, FL	• Critical habitat includes areas where persistent flowering populations occur • No specific PCEs identified, however, the greatest threats are destruction from dredge and fill, turbidity, eutrophication, and thermal pollution due to high population pressure along this segment of the coast	Batch Two discharges are not expected to have any impact on existing populations; pollutants are modeled at concentrations that are unlikely to lead to increased eutrophication	Remote

Table 5-62. Potential for Impacts to the Essential Features of Critical Habitat within the Representative Action Areas (Continued)

Critical Habitat	RAA	Location and Primary Constituent Elements (PCEs)	Potential Effects	Risk of Impact
Bocaccio *Sebastes paucispinis*	Puget Sound, Seattle, WA	• Approximately 590.4 square miles of nearshore habitat and 414.1 square miles of deepwater habitat in Washington state • For adults, benthic habitats or sites deeper than 30 m (98ft) that possess or are adjacent to areas of complex bathymetry consisting of rock and or highly rugose habitat are essential to conservation because these features support growth, survival, reproduction, and feeding opportunities by providing the structure for rockfishes to avoid predation, seek food and persist for decades. • For juveniles, settlement habitats located in the nearshore with substrates such as sand, rock and/or cobble that also support kelp are essential for conservation because these features enable forage opportunities and refuge from predators and enable behavioral and physiological changes needed for juveniles to occupy deeper adult habitats.	Because of their location, Batch Two discharges will not affect critical habitat for adult bocaccio; Batch Two discharges are unlikely to affect critical habitat for juvenile bocaccio because concentrations pollutants in discharges are not high enough to affect kelp beds	Remote
Rockfish, Yelloweye *Sebastes ruberrimus*	Puget Sound, Seattle, WA	• 438.5 sq. mi. (1,135.7 sq. km) of deepwater critical habitat within the Whidbey Basin, Main Basin, South Puget Sound, and Hood Canal • Appropriate quantity and quality of prey species available to support individual survival, growth, and reproduction • Sufficient levels of dissolved oxygen and water quality to support individual survival, growth, reproduction, and feeding opportunities • Sufficient type and amount of structure and substrate complexity to support predator aversion and feeding opportunities	Batch Two discharges could impact water quality for spawning and prey populations if pollutants in discharges accumulate to high enough concentrations; however, an assessment of risks to plants, invertebrates, and fish using maximum modeled concentration of pollutants and the lowest identified no-effects concentrations indicated that the risks are “remote” to “negligible”	Negligible

Table 5-62. Potential for Impacts to the Essential Features of Critical Habitat within the Representative Action Areas (Continued)

Critical Habitat	RAA	Location and Primary Constituent Elements (PCEs)	Potential Effects	Risk of Impact
Salmon, Chinook, All ESUs *Oncorhynchus tshawytscha*	Puget Sound, Seattle, WA	• Areas of varying size within each ESU's location • Freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation and larval development • Freshwater rearing sites with sufficient water quantity and floodplain connectivity, water quality, and natural cover • Freshwater migration corridors free of obstruction and excessive predation with water quantity and quality conditions and natural cover • Estuarine and/or nearshore marine areas free of obstruction and excessive predation with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; natural cover, and juvenile and adult prey (invertebrates and fish)	Batch Two discharges could impact water quality for spawning and prey populations if pollutants in discharges accumulate to high enough concentrations however, an assessment of risks to plants, invertebrates, and fish using maximum modeled pollutant concentrations and the lowest identified no-effects concentrations indicates that the risks are "remote" to "negligible"	Negligible
Sea Turtle, Leatherback *Dermochelys coriacea*	San Francisco, CA	• Areas along the west coast of the U.S. from the northernmost point on the Washington coast to Cape Blanco, WA and from Point Arena, CA to Point Arguello, CA • Occurrence of prey species, primarily Scyphomedusae (sea jellies) of the order Semaeostomeae, of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development	Batch Two discharges could impact water quality for leatherback turtle prey; however, an assessment of risk to marine invertebrates from exposure to maximum modeled concentrations of pollutants indicates that risk is "negligible"	Negligible

Table 5-62. Potential for Impacts to the Essential Features of Critical Habitat within the Representative Action Areas (Continued)

Critical Habitat	RAA	Location and Primary Constituent Elements (PCEs)	Potential Effects	Risk of Impact
Shrimp, San Diego Fairy *Branchinecta sandiegonensis*	San Diego, CA	• Vernal pools with shallow to moderate depths (2 in (5 cm) to 12 in (30 cm)) that hold water for sufficient lengths of time (7 to 60 days) necessary for incubation, maturation, and reproduction of the San Diego fairy shrimp, in all but the driest years • Topographic features characterized by mounds and swales and depressions within a matrix of surrounding uplands that result in complexes of continuously, or intermittently, flowing surface water in the swales connecting the pools, providing for dispersal and promoting hydroperiods of adequate length in the pools (i.e., the vernal pool watershed) • Flat to gently sloping topography, and any soil type with a clay component and/or an impermeable surface or subsurface layer known to support vernal pool habitat (including Carlsbad, Chesterton, Diablo, Huerhuero, Linne, Olivenhain, Placentia, Redding, and Stockpen soils)	Batch Two discharges are not expected to have any impact on fairy shrimp critical habitat; PCEs will not be exposed to Batch Two discharges	Remote
Sturgeon, Green, All DPSs *Acipenser medirostris*	Puget Sound, Seattle, WA San Francisco, CA	• Coastal U.S. marine waters within 60 fathoms (110 m) depth from Monterey Bay, CA, north to Cape Flattery, WA, including the Strait of Juan de Fuca, to the U.S. Canadian boundary; the Sacramento-San Joaquin Delta and Suisun, San Pablo, and San Francisco bays in California; the lower Columbia River estuary; and certain coastal bays and estuaries in California (Humboldt Bay), Oregon (Coos Bay, Winchester Bay, Yaquina Bay, and Nehalem Bay), and Washington (Willapa Bay and Grays Harbor) • Riverine systems with sufficient food resources, substrate type for egg deposition, water flow, water quality, migration corridors, holding pool depth (>5 m), and sediment quality • Estuarine habitats with sufficient food resources, water flow, water quality, migration corridors, sediment quality, and a variety of water depths	Batch Two discharges may impact water quality and food resources if pollutants in discharges accumulate to high enough concentrations; however, an assessment of risks to invertebrates and fish from exposure to maximum modeled concentrations of pollutants indicates that risks are "remote" to "negligible"	Negligible

Table 5-62. Potential for Impacts to the Essential Features of Critical Habitat within the Representative Action Areas (Continued)

Critical Habitat	RAA	Location and Primary Constituent Elements (PCEs)	Potential Effects	Risk of Impact
Trout, Bull *Salvelinus confluentus*	Puget Sound, Seattle, WA	• Areas throughout Idaho and Washington • Springs, seeps, groundwater sources, and subsurface water connectivity (hyporheic flows) to contribute to water quality and quantity and provide thermal refugia • Migration habitats with minimal physical, biological, or water quality impediments between spawning, rearing, overwintering, and freshwater and marine foraging habitats and an abundant food base • Complex river, stream, lake, reservoir, and marine shoreline aquatic environments with features such as large wood, side channels, pools, undercut banks and unembedded substrates • Water temperatures ranging from 2 to 15 degrees Celsius (°C) (36 to 59 degrees Fahrenheit (°F)), with adequate thermal refugia • In spawning and rearing areas, substrate of sufficient amount, size, and composition to ensure success of egg and embryo overwinter survival, fry emergence, and young-of-the-year and juvenile survival • Natural hydrography • Sufficient water quality to support reproduction, growth and survival • No or low density of nonnative predators	Batch Two discharges may impact water quality if pollutants in discharges accumulate to high enough concentrations; however, an assessment of risks to fish from exposure to maximum modeled concentrations of pollutants indicates that risks are remote	Remote

Table 5-62. Potential for Impacts to the Essential Features of Critical Habitat within the Representative Action Areas (Continued)

Critical Habitat	RAA	Location and Primary Constituent Elements (PCEs)	Potential Effects	Risk of Impact
Trout, Steelhead, All DPS *Oncorhynchus mykiss*	Puget Sound, Seattle, WA San Francisco, CA	• Counties where this species occurs • Freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation and larval development • Freshwater rearing sites with sufficient water quantity and floodplain connectivity, water quality, and natural cover • Freshwater migration corridors free of obstruction and excessive predation with water quantity and quality conditions and natural cover • Estuarine and/or nearshore marine areas free of obstruction and excessive predation with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; natural cover, and juvenile and adult prey (invertebrates and fish)	Batch Two discharges could impact water quality for spawning and prey populations if pollutants in discharges accumulate to high enough concentrations; however, an assessment of risks to plants, invertebrates, and fish using maximum modeled pollutant concentrations and the lowest identified no-effects concentrations indicates that the risks are "remote" to "negligible"	Negligible
Whale, Killer (southern Resident) *Orcinus orca*	Puget Sound, Seattle, WA	• 2,560 mi.2 (6,630 km2) of marine habitat that includes Haro Strait and the waters around the San Juan Islands, Puget Sound, and the Strait of Juan de Fuca • Water quality to support growth and development • Prey species of sufficient quantity, quality and availability to support individual growth, reproduction and development, as well as overall population growth • Passage conditions to allow for migration, resting, and foraging	Batch Two discharges could impact water quality and prey populations if pollutants in discharges accumulate to high enough concentrations; however, an assessment of risks to fish and marine mammals from exposure to maximum modeled pollutant concentrations indicated that risks are remote	Remote

Table 5-62. Potential for Impacts to the Essential Features of Critical Habitat within the Representative Action Areas (Continued)

Critical Habitat	RAA	Location and Primary Constituent Elements (PCEs)	Potential Effects	Risk of Impact
Plover, Western snowy *Charadrius nivosus nivosus*	San Diego, CA San Francisco, CA	• 24,527 acres of Sandy beaches, dune systems immediately inland of an active beach face, salt flats, mud flats, seasonally exposed gravel bars, artificial salt ponds and adjoining levees, and dredge spoil sites • Areas that are below heavily vegetated areas or developed areas and above the daily high tides • Shoreline habitat areas for feeding, with no or very sparse vegetation, that are between the annual low tide or low water flow and annual high tide or high water flow, subject to inundation but not constantly under water, that support small invertebrates, such as crabs, worms, flies, beetles, spiders, sand hoppers, clams, and ostracods, that are essential food sources • Surf- or water-deposited organic debris, such as seaweed (including kelp and eelgrass) or driftwood located on open substrates that supports and attracts small invertebrates for food, and provides cover or shelter from predators and weather, and assists in avoidance of detection (crypsis) for nests, chicks, and incubating adults • Minimal disturbance from the presence of humans, pets, vehicles, or human-attracted predators, which provide relatively undisturbed areas for individual and population growth and for normal behavior	Batch Two discharges will not impact the availability of suitable habitat but could impact the availability of intertidal food resources if shorelines are exposed to pollutants in discharges from vessels of the Armed Forces; however, an assessment of risks to invertebrates from exposure to maximum modeled pollutant concentrations indicates that risk to food resources is "negligible"	Negligible

Table 5-62. Potential for Impacts to the Essential Features of Critical Habitat within the Representative Action Areas (Continued)

Critical Habitat	RAA	Location and Primary Constituent Elements (PCEs)	Potential Effects	Risk of Impact
Franciscan Manzanita *Arctostaphylos franciscana*	San Francisco, CA	• 13 geographic units including the Fort Point Unit, Fort Point Rock Unit, World War II Memorial Unit, Immigrant Point Unit, Inspiration Point Unit, Corona Heights Unit, Twin Peaks Unit, Mount Davidson Unit, Diamond Heights Unit, Bayview Park Unit, McLaren Park East Unit, and McLaren Park West Unit • Areas on or near bedrock outcrops often associated with ridges of serpentine or greenstone, mixed Franciscan rocks, or soils derived from these parent materials • Areas having soils originating from parent materials that are thin, have limited nutrient content or availability, or have large concentrations of heavy metals • Areas within a vegetation community consisting of a mosaic of coastal scrub, serpentine maritime chaparral, or serpentine grassland characterized as having a vegetation structure that is open, barren, or sparse with minimal overstory or understory of trees, shrubs, or herbaceous plants, and that contain and exhibit a healthy fungal mycorrhizae component. • Areas that are influenced by summer fog, which limits daily and seasonal temperature ranges, provides moisture to limit drought stress, and increases humidity	Batch Two discharges will not affect critical habitat because vessels of the Armed Forces do not occur in these areas	Remote

Table 5-62. Potential for Impacts to the Essential Features of Critical Habitat within the Representative Action Areas (Continued)

Critical Habitat	RAA	Location and Primary Constituent Elements (PCEs)	Potential Effects	Risk of Impact
Goby, Tidewater *Eucyclogobius newberryi*	San Francisco, CA	• 10,003 acres (4,050 ha) in Del Norte, Humboldt, Mendocino, Sonoma, Marin, San Mateo, Santa Cruz, Monterey, San Luis Obispo, Santa Barbara, Ventura, and Los Angeles Counties, California. • Persistent, shallow (in the range of approximately 0.3 to 6.6 ft (0.1 to 2 m)), still-to-slow-moving lagoons, estuaries, and coastal streams with salinity up to 12 ppt • Sand, silt and mud substrate suitable for constructing burrows for reproduction • Submerged and emergent vegetation that provides protection from predators and high flow events • Presence of a sandbar(s) across the mouth of a lagoon or estuary during the late spring, summer, and fall that closes or partially closes the lagoon or estuary, thereby providing relatively stable water levels and salinity	Batch Two discharges will not affect any of the PCEs for tidewater goby critical habitat	Remote

Table 5-62. Potential for Impacts to the Essential Features of Critical Habitat within the Representative Action Areas (Continued)

Critical Habitat	RAA	Location and Primary Constituent Elements (PCEs)	Potential Effects	Risk of Impact
Smelt, Delta *Hypomesus transpacificus*	San Francisco, CA	• Areas of all water and all submerged lands below ordinary high water and the entire water column bounded by and contained in Suisun Bay (including the contiguous Grizzly and Honker Bays), the length of Montezuma Slough, and the existing contiguous waters contained within the Delta, as defined by section 12220 of the State of California's Water Code of 1969 • Shallow, fresh or slightly brackish backwater sloughs and edgewaters for spawning • Protection of the Sacramento and San Joaquin Rivers and their tributary channels from physical disturbance to ensure that delta smelt larvae are transported from the area where they are hatched to shallow, productive rearing or nursery habitat • Maintenance of the 2 parts per thousand isohaline and suitable water quality (low concentrations of pollutants) within the estuary to provide delta smelt larvae and juveniles a shallow, protective, food-rich environment. • Unrestricted access to suitable spawning habitat in a period that may extend from December to July	Batch Two discharges could affect water quality in Delta smelt rearing habitat; however, an assessment of risk to fish from exposure to maximum modeled concentrations of pollutants based on lowest available no-effect concentrations indicates that risks are remote	Remote

5.6 Uncertainties in the Quantitative Effects Analysis

This section highlights the primary uncertainties and limitations associated with the approach to the exposure and effects analyses undertaken in this BE. There are six major sources of uncertainty potentially limiting interpretation of results and corresponding conclusions, including:

1. Use of WQC to identify pollutants of concern;
2. Estimation of anticipated exposure concentrations used to support the effects analysis;
3. Selection of chronic toxicity values representative of the sensitivity of federally-listed aquatic and aquatic-dependent species;
4. Use of toxic effects concentrations for surrogate species to represent and extrapolate risks to listed species;
5. Extrapolation of exposure and risk for aquatic-dependent birds and mammals; and
6. Estimation of risk from exposure to individual pollutants instead of pollutant mixtures.

These sources of uncertainty are discussed in detail in the following sections.

5.6.1 Uncertainty Associated with Using Water Quality Criteria to Identify Pollutants of Concern for Evaluation

The most stringent federal and state WQC were used identify pollutants in discharges from vessels of the Armed Forces that are likely to have adverse effects on aquatic and aquatic-dependent species, including those listed under ESA. However, WQC are not available for other constituents that could be present in discharges from vessels of the Armed Forces. Therefore, there could be other pollutants in the discharges that have the potential to cause adverse environmental effects but were not studied. Although this could result in risks being underestimated, the WQC allowed many of the potentially more toxic pollutants in the discharges to be identified.

5.6.2 Uncertainty Associated with Estimating Exposure Concentrations

The action area for the evaluation of Batch Two pollutants identified for detailed consideration is represented by the seven RAAs selected for this BE (Appendix E). The EPA and DoD selected two screening-level models to represent the estuarine (coastal) and riverine (inland) water bodies to estimate the exposure concentrations evaluated in the risk analysis. The purpose of using the screening-level water quality models is to identify major water quality issues, provide valuable information on potential pollutants of concern, and identify data gaps.

Known limitations and uncertainties associated with using the screening-level models include:

- Model pollutant load inputs reflect the limitations in available vessel discharge data (discharge flow, discharge concentrations, and vessel numbers);
- Model outputs assume no background concentrations of pollutants are present in receiving water; and

- The models are not designed to predict concentrations on a fine scale, and they tend to overestimate average concentration increases in a harbor.

Overall, the limitations of the estuarine harbor model very likely overestimate pollutant loadings for the harbors and overestimate pollutant concentrations throughout the harbors resulting from discharges from vessels of the Armed Forces. For example, the mass load modeling to graywater assumes a maximum discharge per person of 45 gallons/day; however, under some circumstances vessels may need to minimize graywater discharge to 9 gallons/day (i.e., showers and laundry not permitted). The models also estimate loadings for the maximum crew size. For all discharges, maximum discharge rates are modeled, and the exposure concentrations used for the risk analysis is the maximum modeled concentration for all 6 estuarine RAAs. However, under all scenarios, modeled concentrations can range over several orders of magnitude. See for example the range of modeled concentrations for graywater discharge presented in Table 5-63.

Table 5-63. Sensitivity Analysis for Range of Concentrations Modeled for Graywater Discharge for Two Discharge Rates and Six Harbor Estuary Scenarios

RAA	Discharge Rate Scenario	Copper (µg/L)	Lead (µg/L)	Mercury (µg/L)	Nickel (µg/L)	Silver (µg/L)	Zinc (µg/L)	TSS (µg/L)	BOD (µg/L)	COD (µg/L)	Oil & Grease (µg/L)	Phosphate (µg/L)	Total Nitrogen (µg/L)
Miami	***9 gal/day***	3.3E-08	8.8E-09	4.6E-11	1.5E-09	2.8E-10	1.8E-08	2.8E-05	1.9E-05	5.1E-05	5.3E-07	2.3E-07	5.1E-06
	45 gal/day	1.7E-07	4.4E-08	2.3E-10	7.4E-09	1.4E-09	8.9E-08	1.4E-04	9.6E-05	2.6E-04	2.7E-06	1.2E-06	2.5E-05
Norfolk	***9 gal/day***	2.1E-05	5.4E-06	2.8E-08	9.2E-07	1.8E-07	1.1E-05	1.8E-02	1.2E-02	3.2E-02	3.3E-04	1.4E-04	3.1E-03
	45 gal/day	1.0E-04	2.7E-05	1.4E-07	4.6E-06	8.8E-07	5.5E-05	8.8E-02	5.9E-02	1.6E-01	1.6E-03	7.1E-04	1.6E-02
Pearl Harbor	***9 gal/day***	6.5E-05	1.7E-05	9.0E-08	2.9E-06	5.6E-07	3.5E-05	5.6E-02	3.8E-02	1.0E-01	1.0E-03	4.5E-04	1.0E-02
	45 gal/day	3.3E-04	8.6E-05	4.5E-07	1.5E-05	2.8E-06	1.7E-04	2.8E-01	1.9E-01	5.0E-01	5.2E-03	2.3E-03	5.0E-02
Puget Sound	***9 gal/day***	3.4E-06	9.0E-07	4.7E-09	1.5E-07	2.9E-08	1.8E-06	2.9E-03	2.0E-03	5.3E-03	5.5E-05	2.4E-05	5.2E-04
	45 gal/day	1.7E-05	4.5E-06	2.4E-08	7.6E-07	1.5E-07	9.1E-06	1.5E-02	9.8E-03	2.6E-02	2.7E-04	1.2E-04	2.6E-03
San Diego	***9 gal/day***	5.9E-05	1.5E-05	8.1E-08	2.6E-06	5.0E-07	3.1E-05	5.0E-02	3.4E-02	9.0E-02	9.4E-04	4.1E-04	9.0E-03
	45 gal/day	2.9E-04	7.7E-05	4.1E-07	1.3E-05	2.5E-06	1.6E-04	2.5E-01	1.7E-01	4.5E-01	4.7E-03	2.0E-03	4.5E-02
San Francisco	***9 gal/day***	1.0E-06	2.7E-07	1.4E-09	4.6E-08	8.8E-09	5.5E-07	8.8E-04	5.9E-04	1.6E-03	1.6E-05	7.1E-06	1.6E-04
	45 gal/day	5.1E-06	1.4E-06	7.1E-09	2.3E-07	4.4E-08	2.7E-06	4.4E-03	3.0E-03	7.9E-03	8.2E-05	3.6E-05	7.8E-04
Minimum		**3.3E-08**	**8.8E-09**	**4.6E-11**	**1.5E-09**	**2.8E-10**	**1.8E-08**	**2.8E-05**	**1.9E-05**	**5.1E-05**	**5.3E-07**	**2.3E-07**	**5.1E-06**
Maximum		**3.3E-04**	**8.6E-05**	**4.5E-07**	**1.5E-05**	**2.8E-06**	**1.7E-04**	**2.8E-01**	**1.9E-01**	**5.0E-01**	**5.2E-03**	**2.3E-03**	**5.0E-02**

Therefore, the estuarine harbor model systematically overestimates exposure concentrations for listed species potentially impacted by discharges associated with vessels of the Armed Forces. Because the harbors selected to represent the action area represent areas where pollutants will tend to concentrate, the maximum modeled concentrations are much higher than what concentrations would be for more open waterbodies. As such, the potential for adverse effects is much lower than what is actually determined based on the RQs.

The freshwater harbor model used average stream flow to model pollutant concentrations in receiving water to represent the most common conditions to which aquatic and aquatic-dependent species could be exposed. However, this could potentially underestimate risk during low flow periods. To assess the potential for underestimating risk the model was used to back-calculate the flow conditions that would result in an RQ of 1 for each pollutant of concern, which would indicate a low potential for adverse effects. To do this, the lowest effects threshold (CTET) was used in the back-calculation. Table 5-64 presents the estimated mass loading for each pollutant selected for detailed analysis, the modeled average river harbor concentration, the target concentration to exceed the lowest CTET (i.e., achieve an RQ slightly above 1), and the flow rate needed to achieve the target concentration. The equation used to achieve the target concentration is:

$$\text{Mass Loading (kg/d) / Target Concentration (kg/L)}$$

The result was converted to units of cubic feet per day.

It was determined that the flow conditions to achieve this would have to drop to approximately 377,000 m3/day (154 ft3/s) (Table 5-64). This is well below the average flow rate used of 424,000,000 m3/day (173,303 ft3/s). The lowest flow conditions over the past 10 years occurred in 2013 and were around 146,794,800 m3/day (60,000 ft3/s) (Figure 5-4). Most flow rates over the past 10 years are above 244,658,000 m3/day (100,000 ft3/s). Since 1938 (earliest records), the lowest flow conditions were 73,397,300 m3/day (30,000 ft3/s) (Figure 5-5), which is still well above the flow conditions that would be needed to have pollutant concentrations begin to exceed effects thresholds.

Table 5-64. Flows at or Below Which Pollutant Concentrations in the Representative Freshwater Harbor Will Exceed Toxicity Effects Thresholds (CTETs)

Pollutant	Mass Load (kg/day)	Average River Harbor Concentration (µg/L)	Target Concentration to Exceed Lowest CTET (µg/L)	Average Flow (m3/day)	Flows to Reach Target (m3/day)	Convert m3/day to ft3/s	Notes
Cadmium	0.0000013	3.15143E-09	0.2	424,000,000	6.5	0.0027	Entirely from deck runoff
Chromium	0.000066	1.55718E-07	1.4	424,000,000	47	0.019	Entirely from deck runoff
Total Copper	0.018	6.73624E-05	3.9	424,000,000	4489	1.8	34% hull husbandry, 65% hull leachate
Iron	0.00091	2.25489E-06	31.6	424,000,000	29	0.012	95% from hull leachate
Lead	0.00013	2.96605E-07	6.1	424,000,000	21	0.0087	Entirely from deck runoff
Mercury	3.9E-09	9.25187E-12	0.23	424,000,000	0.017	0.0000070	Entirely from surface bilgewater
Nickel	0.000042	1.00944E-07	7.1	424,000,000	5.9	0.0024	From firemain, surface bilgewater, and deck runoff
Total Zinc	0.011	2.66753E-05	0.03	424,000,000	376586	154	34% hull husbandry, 66% hull leachate
Oil and Grease	0.0012	2.78067E-05	140	424,000,000	8.6	0.0035	Entirely from deck runoff
Total Petroleum Hydrocarbons	0.0000011	2.67911E-09	5.2	424,000,000	0.21	0.000086	Entirely from surface bilgewater
Bis (2-ethylhexyl) phthalate	0.000012	2.81097E-08	0.3	424,000,000	40	0.016	Entirely from fireman
Nitrate/Nitrite	0.00002	4.8224E-08	350	424,000,000	0.057	0.000023	Entirely from surface bilgewater
Total Kjeldahl Nitrogen	0.00011	2.67911E-07	100	424,000,000	1.1	0.00045	Entirely from surface bilgewater
Total Nitrogen	0.00013	3.16135E-07	10	424,000,000	13	0.0053	Entirely from surface bilgewater
Ammonia as Nitrogen	0.0000068	1.60747E-08	303	424,000,000	0.022	0.0000092	Entirely from surface bilgewater
Total Phosphorus	0.00014	3.2328E-07	5	424,000,000	28	0.011	Entirely from surface bilgewater

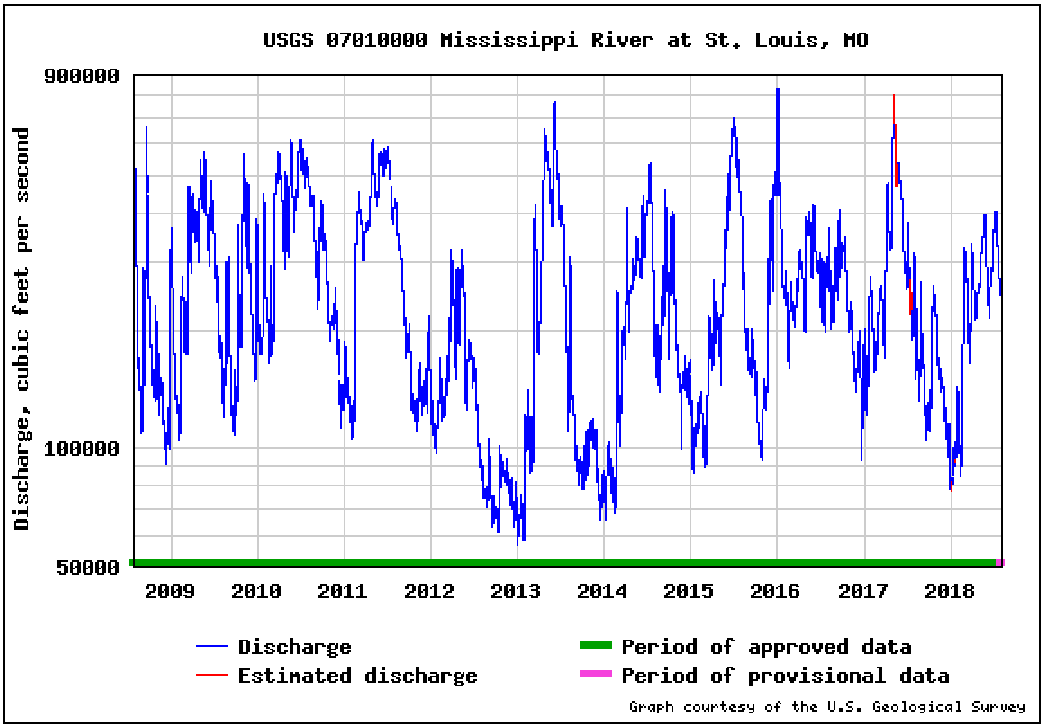

Figure 5-4. St. Louis Stream Gauge Data Over the Past Ten Years

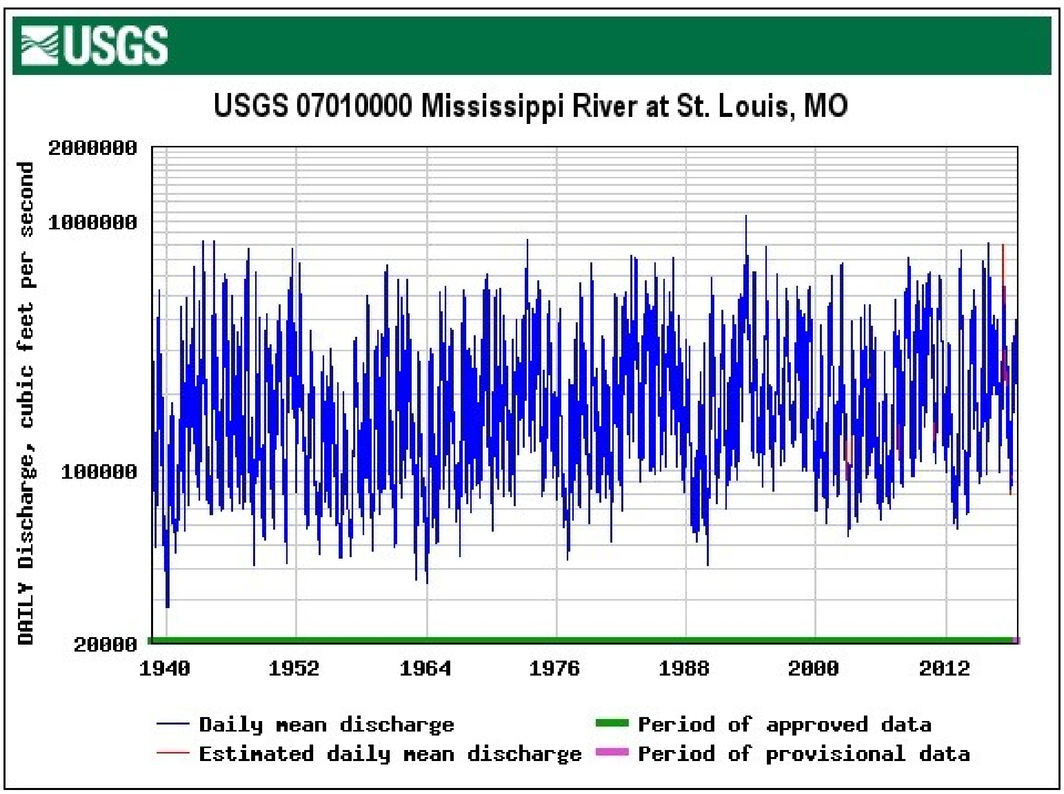

Figure 5-5. St. Louis Stream Gauge Data Since 1938

The EPA and DoD used the best scientific and commercial data available to develop the pollutant loads used in the modeling analysis; however, data gaps exist for the various combinations of vessel types, discharges, pollutants, and vessel populations. Although the EPA and DoD used best professional judgment to extrapolate the pollutant concentrations and flow information among the different vessel types and vessel populations, the extrapolation of information represents a source of uncertainty in the analysis. The uncertainties in the data could either overestimate or underestimate pollutant loads. Data collected as part of the monitoring requirements proposed for Batch Two pollutants in the BE will improve the quality of information used for the UNDS Batch Three analyses.

Both the fraction of freshwater estuary model and the river dilution model assume instantaneous and universal mixing within the harbor. The screening models are not designed to examine impacts on a fine scale, such as for a small embayment or a cove or immediately in the vicinity of the point of discharge; however, they do estimate localized concentrations within 3 nm of the points of discharge within the harbor. Although this approach may underestimate short-term exposure concentrations, those higher exposure concentrations are expected to dissipate rapidly because discharges are limited while in port (i.e., they are either held for transfer to an onshore facility or they are released only while the vessel is underway). The UNDS rule also requires that discharging in sensitive areas (e.g., near coral reefs and marine preserves) be avoided whenever possible.

The EPA and DoD selected model input values to represent the range of conditions likely to occur within the broader action area and, when appropriate, identified model parameters to represent an upper bound in the analysis and reflect a reasonable “worst case” scenario (e.g., minimum river flow in the representative freshwater riverine system; Mississippi River for the St. Louis RAA). The approach used was intentionally conservative, and it likely overestimates pollutant concentrations resulting from discharges from vessels of the Armed Forces throughout the harbors because the maximum modeled concentrations across all of the RAAs was used to represent exposure concentrations throughout the action area.

The population of vessels of the Armed Forces affected by the proposed rule encompasses more than 6,000 vessels distributed among the U.S. Navy, MSC, U.S. Coast Guard, U.S. Army, U.S. Marine Corps, and U.S. Air Force (see Table 3-1). The Armed Forces have vessels that range in design and size from small boats with lengths of less than 20 feet to aircraft carriers with lengths of over 1,000 feet. The model also assumes that the vessels are discharging continuously, and the entire harbor area is used for each respective RAA, which was calculated from surface area metadata (combined surface area of all GIS polygons within defined RAA). The number of vessels in each RAA ranges from 17 (St. Louis, MO RAA) to 728 (Norfolk, VA RAA) (see Table F-3). The RAAs include the two harbors that are most densely populated with vessels of the Armed Forces, and a wide range of flushing conditions was considered (see Section 5.2.1 and Appendix F). Hence, the screening-level model conservatively characterizes chronic exposure concentrations and subsequent pollutant risk. The effects analysis is based on the maximum exposure concentration across all the harbors and very likely overestimates most exposures, excluding the uncommon acute exposures to a discharge plume that may occur immediately following discharge.

The EPA and DoD believe that the analysis also represents vessel discharges to inland and coastal waterways for two primary reasons:

- Several of the major water bodies (e.g., Biscayne Bay, Chesapeake Bay, San Diego Bay, San Francisco Bay, Puget Sound) were considered in developing the receiving characteristics for the estuary model; and
- The daily vessel population in inland waterways is likely significantly lower than in a harbor environment.

Although listed species present in these inland waterways may be directly exposed to a vessel discharge plume, the EPA and DoD expects the exposure duration to be brief because most discharges will only be release while the vessel is underway, with the exception of hull coating leachate, which is a slow continuous release. Although the ECs calculated for the effects analysis do not specifically account for localized effects under conditions where pollutants might accumulate, they do represent the reasonable maximum chronic exposure conditions by assuming that the highest exposure concentrations occur throughout the action area. Further, the EPA and DoD considered the potential risk posed by other non-military vessel discharges in other unique environments, as discussed in the VGP BE, and determined that vessel discharges that are regulated under the VGP and sVGP are unlikely to be a concern for federally listed species (EPA, 2013). For this BE, because maximum exposure concentration are assumed, the evaluation very likely overestimates risks to exposed organisms.

Lakes and reservoirs isolated from the major rivers navigable by vessels regulated under UNDS represent another unique environment that was not specifically modeled in the analysis. However, the EPA and DoD believe that the maximum exposure concentrations estimated in the St. Louis, MO, RAA, UMR Harbor can be used to inform effects determinations for species identified in the lake environment. In a similar study, the EPA conducted internet searches to assess the potential vessel populations that operate on lakes and reservoirs greater than 0.5 square mile (EPA, 2013). The EPA determined that vessel populations operating on lakes and reservoirs are significantly smaller than vessel populations operating in river harbors, and that the vessels also operate and discharge for limited periods of time. Therefore, the EPA concluded from that study that the population of vessels in lake environments is likely to be significantly lower than the vessel populations estimated in river environments. Subsequently, the EPA estimated maximum pollutant concentrations in a lake environment to be lower than the maximum exposure concentration calculated for a representative river harbor (EPA, 2013). Based on the study conducted by the EPA, the EPA and DoD believe that the calculated river EC values and subsequent risk conclusions for freshwater species in this study effectively capture the potential reasonable maximum scenario pollutant exposure concentrations and risks in a lake environment from vessels of the Armed Forces.

Lastly, it is important to realize that the model outputs represent loadings only from vessels of the Armed Forces, and the BE does not quantify the contribution of other sources of pollutants. However, by looking at the magnitude of the RQ, the EPA and DoD can determine whether incidental vessel pollutant discharges covered under the UNDS BE would make notable or sizable contributions to exposure concentrations that exceed pollutant specific thresholds.

5.6.3 Uncertainty Associated with Estimation of Chronic Toxicity

The consideration and selection of appropriate chronic values from empirical laboratory toxicity test data for assessing risk to aquatic organisms is similar to the process used by the EPA for ALC development. The quality and acceptability of the toxicity data depend on the acceptability of test conditions, test duration, an endpoints measured. Chronic toxicity tests in general encompass a variety of specific test types, each with their own associated uncertainties. For example, full and partial life cycle tests contain the lowest uncertainty, but due to the additional time needed to conduct these tests and associated higher costs, these tests are rarely conducted. In contrast, early life-stage (ELS) tests are far more common, but because these tests exclude certain life stages (e.g., adult) they cannot be used to quantify secondary effects such as reproduction.

With all chronic tests, using wild-caught test organisms introduces greater uncertainty than using test organisms reared in the laboratory, although certain species cannot be reared under laboratory conditions. Uncertainty is also associated with the selection of chronic endpoints. Nearly all chronic tests measure growth and survival, whereas reproduction endpoints can be measured only in full and partial life cycle tests. However, reproduction endpoints are more sensitive to pollutant exposures than survival endpoints, and growth endpoints are generally the most sensitive. The specific selection of endpoints used to measure growth and reproduction can also introduce test uncertainty. For example, measures of fish reproduction can include different combinations of egg production, viability and hatchability. Many reproductive endpoints are known to be highly variable; however, reduced reproduction may be one of the most ecologically significant toxicological responses pertinent to listed species. (Mount et al., 2003)

Growth can be measured as mean individual wet or dry weight (or wet or dry weight gain); standard body, fork or total length; or biomass (total wet or dry weight of survivors). Finally, the statistical method by which chronic toxicity is quantified can introduce uncertainty. Historically, most chronic toxicity data were evaluated using hypothesis testing to derive a no observed effect concentration (NOEC) and lowest observed effect concentration (LOEC) (and resultant MATC) from the test concentration series. The resulting values are highly dependent on the test concentrations selected, and the number of replicates at each treatment level. In many cases, the level of protection afforded by the NOEC-LOEC approach is driven more by study design and data precision than ecological significance (Mount et al., 2003). In contrast, the use of point estimates of chronic toxicity, such as the EC10 or EC20, overcome many of the shortcomings of the NOEC-LOEC approach, although the reporting of point estimates associated with a chronic effect is historically less common, and debate continues with regards to the significance to the individual organism (and population) of an estimated 10% reduction in chronic response, such as growth. Furthermore, with regards to listed species, reproduction and survival are more relevant for determining population-level effects, and because of their sensitivity, NOECs are the desired target value.

In addition to survival, growth and reproduction, there are many more biological responses that have been observed in response to chronic toxicant exposure, both at the whole organism level (e.g., behavior) and at lower levels of biological organization (e.g., biochemical or histological changes). For many of these endpoints, the relationship to the assessment goal for this analysis of “protection of listed aquatic organisms” is less direct. Although consideration of endpoints

not prescribed by the *1985 Guidelines* is an option under any approach, the applicability of the endpoint to the assessment goal and the level of uncertainty surrounding the endpoint, requires careful consideration (Stephan et al., 1985). As a general rule, the use of alternative endpoints is discouraged unless a compelling argument can be made that inclusion would reduce the overall level of uncertainty in the analysis towards achieving the assessment goal (EPA 1998).
The EPA (1998) broadly classifies the uncertainty regarding estimation of chronic toxicity in this analysis into four areas:

- Relevance of test species to the species being protected (e.g., species versus genus versus family);
- Testing protocol (e.g., full or partial life cycle versus early life stage [ELS] test);
- Test endpoint (survival, growth, reproduction or other effects measurement); and
- Method of effect quantification (NOEC-LOEC versus point estimate).

All of these issues strongly influence evaluation of the potential risk of a given exposure concentration. Given the potential uncertainty surrounding these considerations for even the “best” data used in this approach, it is no surprise that chronic values obtained using lower tiered data can exhibit high variability. The EPA and DoD rely on the EPA's decision to use the *1985 Guidelines* data requirements and data selection process to fully vet information for its utility and applicability in this BE analysis (Stephan et al., 1985).

If it were feasible, the least uncertain chronic toxicity effect threshold value would be one determined from numerous field tests with individual pollutants (and their mixtures) added to a wide variety of unpolluted water bodies containing an array of aquatic and aquatic-dependent organisms. It would be necessary to add various amounts of the material to each body of water in order to determine the highest concentration that would not cause any unacceptable long-term or short-term effect on the organisms or their uses (i.e., the no-effects threshold). The lowest of these no-observable effects concentrations (NOECs) would serve as the basis for derivation of a toxicity threshold value that would be protective of all species, including listed species. Because it is not feasible to conduct such field tests, the EPA and DoD believe that the approach used in this analysis provides the best objective, internally consistent, appropriate, and feasible way of determining risk to listed aquatic and aquatic-dependent species with the lowest degree of uncertainty.

Despite these uncertainties in the chronic toxicity data used for the risk evaluation, as previously discussed, listed species present in navigable waters of the U.S. may be directly exposed to a vessel discharge plume, but the exposure duration is expected to be brief. As such, the chronic effects thresholds used in the BE are conservatively low because they represent effects thresholds for an unlikely exposure scenario where a species has been continuously exposed at the test concentration. Actual effects thresholds for invertebrates, fish and wildlife that are intermittently exposed to pollutants in discharges from vessels of the Armed Forces for short periods are likely to be higher. Using CTETs derived for continuous exposures under laboratory conditions very likely overestimates risk to listed species in the wild. In some instances, a CTET for a specific pollutant could not be identified for a taxonomic group and risk from that pollutant could not be characterized. There are also some pollutants, such as petroleum products, that break down to constituents that could be more toxic than the parent compounds, which is not accounted for in this evaluation.

5.6.4 Uncertainty Associated with Using Toxic Effects Concentrations for Surrogate Species

Chronic toxicity data for listed aquatic and aquatic-dependent species largely do not exist. The data that do exist are limited to only a few pollutants (ammonia, copper, carbaryl, 4-nonylphenol, pentachlorophenol, and permethrin) and freshwater/euryhaline fish species (i.e., shortnose and Atlantic sturgeon, bonytail chub, Cape Fear shiner, Colorado pikeminnow, razorback sucker, Apache trout, Lahontan and greenback cutthroat trout, and desert and Leon Spring pupfish). Considering that adequate chronic toxicity data are limited for nearly all pollutants selected for detailed evaluation in this BE, use of surrogate data for representative species is required to perform the risk assessment. For each pollutant evaluated in Section 5.3, the EPA and DoD present the surrogate species data used to calculate the RQ values to support the risk conclusions for the 18 listed species taxonomic groups. The EPA and DoD established the surrogate species types in Table 5-12 to classify the chronic toxicity data and to provide a consistent presentation of the RQ values in the quantitative analysis. Table 5-12 presents the crosswalk for how surrogate species data and the resulting RQs were used to inform the pollutant-specific risk conclusions and effects determinations for the 18 listed species taxonomic groups evaluated. There are two primary areas of uncertainty associated with the use of chronic effects data gleaned from tests with surrogate species. First, the surrogate species selected for use in this BE were not necessarily the taxonomically closest related organism for which information was available. Instead, the most sensitive surrogate species identified from the available toxicological studies for each general taxonomic group was selected, which provides an added measure of conservatism in the risk analysis. Generally, early life stages are the most sensitive. However, if early life stage studies were not available, the chronic lowest effects threshold for all available studies was selected.

From a taxonomic perspective, the best surrogates would be those that are most closely related (within same genus or family), live in similar environments, and consume similar food types as the listed species of interest. However, there may be relatively little toxicity data for species within the same genus or family, and other surrogate species may be considerably more sensitive to a given pollutant. Both of these considerations (quantity and representativeness of toxicity data and sensitivity) influenced the EPA and DoD's decision to select surrogate species for the BE, as was done for the EPA’s VGP BE (EPA 2013). While the sensitivity of specific federally listed species to stressors is largely unknown, the selection of the most sensitive surrogate species and life stage will tend to minimize the potential of underestimating risk of impact from exposure to pollutants in discharges.

Another area of uncertainty has to do with the assumption that listed species may be uniquely sensitive to pollutants compared to surrogate species. For the available data, side-by-side acute and chronic toxicity studies clearly indicate that listed fish and amphibian species generally do not appear universally more sensitive than other common surrogate fish species, such as the rainbow trout, sheepshead minnow, and fathead minnow (Mayer et al., 2008). Such conclusion is assumed for other species groups as well, based on the data used to develop recommended WQC (e.g., listed freshwater unionid mussels are no more sensitive to ammonia and copper than other surrogate freshwater bivalve mollusk species) (EPA, 2005b, 2009).

However, there may be some instances where listed species are more sensitive than the surrogate species for which toxicity data are available. The EPA and DoD are currently not aware of any one factor that could be applied universally to surrogate data to ensure protection to listed aquatic species. However, comparative analysis of the extreme chronic sensitivity of the darter species, *Etheostoma fonticola*, and the generally-sensitive surrogate species rainbow trout, to copper (EPA 2013; Besser et al., 2005b) suggests that a factor of 2.0 may be a reasonable safety factor. In addition to the Besser et al. study, related studies have shown that derivation of an appropriate safety adjustment factor of 2.0 may be applicable with the use of taxonomically-representative and pollutant-sensitive surrogate taxa (EPA, 2013; Dwyer et al., 2005; Gensemer et al., 2007; Mayer et al., 2008). A sensitivity analysis for the Oregon Toxics BE showed that taxonomic groups that are most closely related to listed species were not among the most sensitive ecological receptors (EPA, 2008). Combined, these studies suggest that: (1) listed aquatic vertebrate species are not universally more sensitive to contaminant exposure than other aquatic vertebrates on a toxicological basis; and (2) surrogate fish species appear to represent other listed fish species toxicologically.

In order to assess the ability of the selected $CTET_{A,W}$s to account for differences in sensitivity amongst species, RQs calculated using the selected CTETs were compared with RQs calculated using EPA's CCCs. The CCC is an estimate of the highest concentration of a pollutant in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect and is based on toxicity data for a minimum of eight families of organisms. To account for any possible differences in sensitivity between a listed species and its surrogate species, the EPA and DoD opted to apply a recommended risk adjustment factor of 2.0 (Mayer et al., 2008) to the EPA's CCCs to further evaluate the potential range of risk to aquatic and aquatic-dependent species exposure to Batch Two pollutants. The CCC represents the highest continuous exposure concentration for a specific pollutant in water that is not expected to pose a significant risk to the majority of species in a given environment. Table 5-65 presents the RQ values for water-only exposure of vertebrate fish species to the estimated maximum exposure concentrations in an estuarine harbor and representative exposure concentrations in a river harbor. RQs based on the adjusted CCC and selected CTETs are compared.

Table 5-65. Analysis of Sensitivity of Aquatic Organisms from Direct Exposure to Batch Two Pollutants in Water Based on the Environmental Protection Agency Continuous Criterion Concentrations

Pollutant	EC_W (µg/L)	CCC/2.0 (µg/L)	RQ_{CCC}	RQ_{CTET}
Freshwater Fish				
Cadmium	3.2E-09	0.36	8.9E-09	1.9E-09
Chromium	1.6E-07	5.5	2.9E--08	5.3E-09
Copper	6.7E-05	1.5[a]	1.5E-05	1.4E-05
Iron	2.3E-06	500	4.6E-09	7.2E-09
Lead	3.0E-07	1.25	2.4E-07	3.2E-09
Mercury	9.3E-12	0.39	2.4E-11	4.0E-11
Nickel	1.0E-07	26	4.8E-08	6.3E-10
Tributyltin (TBT)	0	0.036	0	0
Zinc	0.000027	60	4.5E-07	3.8E-07
Saltwater Fish				
Cadmium	0.00001	3.95	2.5E-06	9.5E-07
Chromium	0.00039	25	0.000016	4.2E-07
Copper	0.79	1.6	0.49	0.0038
Iron	0.038	-	-	0.014
Lead	0.00082	4.05	0.0002	0.000072
Mercury	4.6E-07	0.47	9.8E-07	9.2E-08
Nickel	0.0097	4.1	0.076	2.3E-06
TBT	0.00021	0.0037	0.057	0.0018
Zinc	0.41	40.5	0.01	0.0057

[a] From the EPA's 2007 Biotic Ligand Model-based ALC (EPA, 2007).

The CCC-based RQs provided in Table 5-65 generally support the results presented in the risk analysis. With the exception of copper, all of the pollutants analyzed show "remote" (RQ<0.1) risk to listed aquatic species (and likely also to the aquatic-dependent species that may consume them) under maximum exposure concentrations. Risk from exposure to copper could be considered to be negligible.

As discussed in Section 5.2.5, the EPA and DoD uses the risk conclusions for the 111 RAA listed species to inform its effects determinations for the 674 aquatic and aquatic-dependent species and their critical habitats that may be affected by the discharges from vessels of the Armed Forces by establishing 18 listed species taxonomic groups as the link between the two lists (Table 5-12). In general, the listed species taxonomic groupings determined by the subset of 111 RAA species are representative of all the listed species evaluated in this BE. Numerically, the 111 RAA listed species represents approximately 16 percent of the full list of 674 species to which the effects determinations for pollutants identified for detailed evaluation apply.

In terms of overall taxonomic representation, the EPA and DoD estimate that the subset of 111 RAA listed species represents the vast majority of the types of threats, critical life stages, and habitats associated with the full list of 674 listed species that may be affected by discharges from vessels of the Armed Forces. The 19 taxonomic groups established based on the listed species that occur in the RAAs adequately capture several potentially sensitive groups, such as unionid mussels and inland and anadromous fishes, freshwater planktonic crustaceans, amphibians, seagrasses, saltwater corals, bivalve mollusks, coastal and marine birds, and marine and terrestrial mammals.

The list of 111 RAA federally-listed species spans nearly the full range of critical life stages and habitats representative of the species and critical habitats that may be affected by discharges from vessels of the Armed Forces throughout the entire action area. Absent among the 111 RAA listed species are several listed freshwater snails and benthic crustaceans, for which major threats and critical life stages and/or habitat may not be adequately represented. However, in these particular cases, the EPA and DoD believe that the current list of 111 RAA listed species provides adequate overall coverage even among taxa that are only distantly related. Existing and current literature indicate that freshwater unionid mussels are good ecotoxicologically sensitive surrogates for gill-bearing snails; planktonic crustaceans are good sensitive surrogates for all other crustaceans; and salmonids are good surrogates for other fresh and saltwater fish (Mayer and Ellerseick, 1986; Mayer et al., 2008).

5.6.5 Uncertainty Associated with Estimating Dietary Doses

Estimation of risk for aquatic-dependent birds and mammals from exposure to a pollutant in water is complicated because the exposure pathway of primary concern is the oral (ingestion) route. Proper risk analysis would consider potential food web effects (e.g., biomagnification), which would require consideration of a pollutant's fate properties (e.g., water solubility, persistence, octanol: water partition coefficient or log K_{ow}, etc.). The risk analysis performed by the EPA and DoD in this BE as described in Section 5.2.3 incorporates food and water ingestion data available from the literature for wildlife species to estimate an ingested dose for wildlife based on a maximum water exposure concentration. This oral dose was then compared to an oral dose (NOAEL) from the literature that would likely be protective of listed wildlife species.

However, very few wildlife species ingest only one type of prey species. Prey of wildlife species may come from different trophic levels, which in turn may exhibit BAFs that differ from each other. This is particularly true for organic pollutants with large K_{ow} values, which also tend to be the pollutants with the greatest potential for biomagnification. For this BE, macroinvertebrates represent secondary consumers that are also prey items, and vertebrates represent higher level consumers that are consumed by larger predators (e.g., fish). For simplicity, the EPA and DoD applied one BAF for all potential trophic levels to estimate potential pollutant concentrations in prey for calculating the ingested dose to wildlife. With the exception of mercury, it was determined that consideration of bioaccumulation in different trophic levels is largely unwarranted.

Dietary doses for a wildlife species will largely depend on specific caloric needs based on an animal's body weight, level of activity, habitat and climate. Even within a species, daily food intake will depend on life stage and environmental factors such as competition and food availability. All of these factors introduce variability into the dietary exposure estimates, so estimates are based on average literature values.

For this evaluation, only ingestion of freshwater was included in the dietary exposure modeling. Although some marine wildlife species (dolphins, seals, sea turtles and sea otters) ingest sea water, ingestion of sea water by most marine wildlife is not a common behavior, and most water needs are met metabolically and by food ingestion, while incidental ingestion of sea water helps maintain electrolyte balance (Ortiz, 2001). Sea otters and sea turtles are known to ingest seawater, but seawater ingestion rates are unknown, and seawater ingestion was not included in any of the wildlife dietary exposure models. While this may underestimate risk to some degree, it is likely that it is not substantial, and other conservative assumptions likely balance this uncertainty.

Water ingestion was included as a pathway for freshwater aquatic and aquatic-dependent wildlife because water ingestion rates are more constant and measurable. However, although some aquatic and aquatic-dependent marine/estuarine wildlife may ingest seawater, seawater ingestion is not regular and largely has not been measured. Although risks to fully aquatic wildlife species may have been underestimated, ingestion of pollutants in drinking water is much less significant exposure pathway than ingestion of pollutants in prey. As such, underestimates for marine aquatic and aquatic-dependent wildlife are not expected to be substantial.

The dietary exposure estimates were estimated based on surrogate species that may or may not be similar to the federally listed species being evaluated. Exposure parameters for aquatic-dependent birds and mammals are based on the surrogate species for which the CTETs were selected. Exposure and risk estimates were then extrapolated to the larger listed species guild (i.e., aquatic-dependent birds or aquatic-dependent mammals). Although this reduces the uncertainty associated with the assessment of adverse effects because the estimated exposure is then compared with an effects concentration for the same species, the exposure and effects for other bird and mammal species may not be represented as effectively. There may be differences in food ingestion rates and gastrointestinal absorption for pollutants in environmental media and food sources, as well as differences in dietary preferences for different species within the same guild. Using dietary exposure estimates for one species could overestimate or underestimate exposure risks for another species.

The exposure estimates assume that species will be maximally exposed to pollutants in water and prey because they feed entirely within the action area. Although this may be the case for some species, other species are migratory and may spend some time feeding outside of the action area. If receptors prefer to feed in specific areas within the action area (*e.g.*, invertivorous birds feeding in mudflats or piscivorous wildlife feeding around piers where there is greater fish habitat), or have a greater preference to spend more time feeding off-site, this assumption could result in under- or overestimating exposures since chemical concentrations are generally not evenly distributed across a site.

Dietary exposure risk for fish and invertebrates are not assessed because of the lack of information on dietary exposure factors. This is an information gap in this risk assessment, as it is in many others.

Potential inhalation exposure to pollutants, particularly MAHs, was not evaluated because of the difficulty with characterizing these risks. As a result, risks may have been underestimated because this exposure route was not considered. However, given the pollutants and their associated environmental fates, this underestimate is likely to be insignificant.

Lastly, dermal absorption of pollutants was assumed to be a negligible route of exposure for wildlife receptors and was not quantified in this BE. This approach may underestimate the total exposure risk, especially given the potential exposure to surface water for marine birds and mammals. Risks from dermal exposure may be more significant than originally suspected, especially for species that are fully aquatic. While methods are available to assess dermal exposure in humans, currently ecological toxicity data for dermal exposure is limited to pesticides, hence this BE does not address dermal exposure.

5.6.6 Uncertainty and Evaluation of Risk from Possible Exposure to Pollutant Mixtures

Although organisms are simultaneously exposed to multiple pollutants in their natural environment, environmental toxicologists and the EPA have historically assessed the risk from exposure to a single pollutant present in the environment. Assessing mixture toxicity is extremely complicated due to the vast number of combinations of pollutants that could be present in a mixture, the limited amount of information on interactions among pollutants in mixtures, and the differing mechanisms of action among pollutants within a given mixture. In addition, determining the actual dose of each pollutant within the mixture to which an organism is exposed presents a significant amount of uncertainty due to the numerous factors that can control the bioavailability and uptake of an individual pollutant by the organism within the mixture.

Given that real world environments contain a mix of pollutants with differing mechanisms of action, any approach that hopes to accurately predict mixture toxicity must ultimately be based upon toxicity data for all of the individual pollutants within the mixture. Evaluating and quantifying the effects of mixtures is challenging as the toxic effects of pollutants in a mixture can be less than, greater than, or not appreciably different than those predicted for the same pollutants singly. Concentration addition is a common method used to assess mixture toxicity; however, this method is accurate only when the mixture consists solely of pollutants with the same toxicological mechanism of action. Perhaps the best known examples of pollutant mixtures where toxicity is additive is for metals and type 1 narcotics, such as PAHs (EPA, 2005b; McGrath and DiToro, 2009).

As an additional assessment of the risk posed to listed aquatic species from incidental discharges from vessels covered under the VGP, the EPA calculated the additive toxicity of metals and PAHs in mixture from vessel discharges. In this analysis, the EPA and DoD summed the RQs for water-column only exposure based on a mixture of metals to give a total RQ representative of the risk of those mixtures to listed aquatic species. The RQs of individual metals in mixture are provided in Table 5-66 below, along with the sum of RQs representing their mixture. Based on the results of this analysis, the EPA and DoD believe that exposure to the maximum exposure concentrations of metals in mixture presents a “remote” risk to most listed species and “negligible” risk to estuarine/marine invertebrates.

Although the EPA limited the quantitative mixture analysis to metals, the RQ values for all of the pollutants evaluated in the BE were low (i.e., <0.1) suggesting that any additive risk from the pollutants from vessel discharges in mixture is likely not of concern.

Table 5-66. Sum of Risk Quotients for Metals under Uniform National Discharge Standards Batch Two Pollutants

Metal	Estuarine/ Marine Vertebrate	Estuarine/ Marine Invertebrate	Estuarine/ Marine Plant	Freshwater Vertebrate	Freshwater Invertebrate	Freshwater Plant
Cadmium	9.5E-07	1.1E-06	8.4E-07	1.9E-09	1.7E-08	3.2E-09
Chromium	4.2E-07	2.2E-05	3.9E-09	5.3E-09	1.0E-07	5.3E-09
Total Copper	0.0038	0.10	0.016	1.4E-05	1.7E-05	5.9E-07
Iron	0.014	0.00027	--	7.2E-09	7.3E-08	2.3E-09
Total Lead	7.2E-05	3.4E-05	0.00010	3.2E-09	5.0E-08	3.8E-11
Total Nickel	2.3E-06	0.00044	0.0012	6.3E-10	1.4E-08	2.9E-10
Total Zinc	0.0057	0.0026	4.3E-05	3.8E-07	0.00090	2.7E-09
Sum Metal $RQ_{A,WS}$	**0.024**	**0.10**	**0.017**	**0.000015**	**0.00092**	**6.0E-07**

Total RQ does not include mercury, which has a different mode of action and tends to biomagnify.

5.6.7 Uncertainty Associated With Assuming Total Modeled Concentrations Are Bioavailable in Surface Water

Because UNDS is specifically for discharge to receiving waters, the effects analysis focuses on total concentrations of pollutants being in surface water and being 100% bioavailable. It is more likely that some fraction of the pollutants will adhere to particulates in surface water and partition to sediment, making them less bioavailable. Therefore, risk from exposure to pollutants in surface water is very likely overestimated.

However, risk from exposure to the fraction of pollutants that partition to sediment is not accounted for in this assessment. Because uncertainties and inaccuracies are compounded with each additional step in estimating concentrations, there would be a high degree of uncertainty associated with estimating the amount of each pollutant in sediment resulting from discharges from vessels of the Armed Forces. However, had the effects analysis determined that modeled concentrations of pollutants in surface water could pose significant risk to federally listed species, additional effort would have been made to assess risk from estimated concentrations that partition to sediments.

5.6.8 Uncertainty Associated With Sediment and Porewater Exposures

Sediment and porewater exposures to pollutants are an important exposure pathway when evaluating risk to ecological receptors. Sediments could serve as a sink for pollutants discharged to surface water. There could also be movement of pollutants between sediment and porewater. However, exposure concentrations in sediment and porewater resulting from discharges from vessels of the Armed Forces are uncertain, as are the direct and indirect effects from pollutants in sediment and porewater resulting from discharges.

Measured concentrations of pollutants in sediment and porewater could not be used to assess the impact of this regulation because sediment concentrations of pollutants in military ports and harbors come from multiple sources and the relative contribution from Batch Two discharges could not be determined. Freshwater and marine/estuarine sediment and porewater concentrations of pollutants also were not modeled due to the associated uncertainties. The behavior of pollutants in surface water, including partitioning to sediments, is complex and highly site specific. Even for non-degrading pollutants like metals, the behavior in surface water, including partitioning to sediment, is complex and dependent on a number of factors that can fluctuate widely temporally including organic matter, pH, and salinity. In sediments, humic material and acid volatile sulfides are important controls on bioavailability of several metals. For pollutants that degrade, the relationship between surface water and sediment concentrations would be even more difficult to model. Pollutants that partition to sediments are typically adsorbed to organic material and other ligands and are largely not bioavailable. Over time, pollutants in sediment could be resuspended and then redeposited, and in highly depositional environment could become buried below the biologically active zone. Lastly, the level of uncertainty associated with modeling sediment and porewater concentrations from partitioning based would be compounded by the uncertainties associated with modeled surface water concentrations from each of the Batch Two discharges. Therefore, risk from exposure to discharge-related pollutants in sediment and porewater is a gap in this assessment.

5.7 Summary of Risk Conclusions for Listed Aquatic and Aquatic-Dependent Species and Their Critical Habitats Represented Within Representative Action Areas

Both qualitative and quantitative risk analyses were conducted for pollutants and stressors in the Batch Two discharges regulated under UNDS. Those pollutants and stressors for which there were insufficient data and risk assessment models were subjected to a qualitative evaluation. Those included risk from ANS, oil and grease, TPH, BOD, COD, and PPCPs. Risk from ANS invasion was determined to be “remote” to “potentially significant” depending on the listed species and its life history requirements. Although ANS invasion resulting in exposure for federally listed aquatic and aquatic-dependent species was determined to be unlikely in most cases, the potential consequences if an ANS invasion does occur could have major consequences for some species that are more susceptible to the effects of fouling organisms. Further, the implementation of standards to control these discharges would not increase this risk. Although vessel maintenance and cleaning practices and UNDS reduce the likelihood that NAS will be introduced to military ports and harbors, they do not eliminate the potential for NAS introductions and ANS invasions to occur, and risk to federally listed species is potentially significant if an invasion does occur.

Because of the way the Armed Forces manage discharges and propose to manage discharges under UNDS, risk to federally listed species from exposure to oil and grease, TPH, BOD, COD, and PPCPs in Batch Two discharges is “remote”. Regulations prohibit the discharge of petroleum hydrocarbons at concentrations greater than 15 ppm and generally hold discharges for onshore disposal when possible. At a minimum, UNDS prohibits or limits discharging within one mile of shore. In addition, most PPCPs would be found in blackwater, which is not regulated by UNDS, and any products that will be found in graywater would be present at minimal concentrations.

The EPA and DoD quantitatively evaluated the effects of 21 chemical pollutants and determined that the risks to nearly all aquatic species from exposure pollutants in the eight UNDS Batch Two discharges selected for detailed analysis are "remote". An evaluation of risk by taxonomic group determined risk to estuarine/marine invertebrates to be "negligible" based on the risk evaluation for copper exposure. The pollutants that resulted in the highest RQ values (all less than 0. 1, indicating "remote" to "negligible" risk) in the quantitative analysis for listed and surrogate aquatic species from direct water exposures were copper, iron, nickel, zinc, and TBT. The application of a safety adjustment factor of 2.0 to the Agency's CCC (Table 5-65) resulted in a similar conclusion of "remote" to "negligible" risk to aquatic listed species, with risk from exposure to all metals except copper being "remote". The EPA and DoD's analysis of the potential risk of various metals in mixture also resulted in RQ values of 0.1 or lower; thus, the EPA and DoD assume that the risk to aquatic species from mixtures is "remote" to "negligible". Even when body burdens of Batch Two pollutants were estimated using BAFs and BCFs, risks from bioaccumulation of pollutants were "remote", with most RQs being less than 0.1.

Finally, the majority of the RQ_{wild} calculated for aquatic-dependent wildlife species were very low (1.6E-12 to 0.)0027) such that the estimation of risk based on use of CTETs from surrogate species should still be predictive even when extrapolating differences in dietary absorption between listed and surrogate species. Based on these results, the EPA and DoD concluded that the discharge of toxic and conventional pollutants with toxic effects from vessels of the Armed Forces presents "remote" to "negligible" risk to the 111 RAA listed species.

In addition to the qualitative and quantitative risk analysis, the EPA and DoD evaluated and considered the stressors currently affecting the 111 RAA listed species (see Table 4-8). Biological and natural threats (e.g., competition, predation, disease, and genetic threats) with no relationship to vessel discharges were the predominant stressors documented for the 111 RAA listed species. The EPA and DoD identified 21 of the 111 RAA species with known threats from water pollution and other related water quality problems; however, none of the species-specific threat information identified vessel discharges as a specific pollution source of concern. The results of the quantitative analysis estimated RQ values indicate a "remote" risk to listed aquatic and aquatic-dependent species, and an evaluation of risk to PCEs based the same RQs and an assessment of whether Batch Two pollutants in vessel discharges are likely to affect critical habitat elements determined that risk to critical habitat is "remote" to "negligible".

The RAAs evaluated in the effects analysis are impacted by pollutant sources other than vessel discharges, such as industrial discharges, urban storm water, superfund sites, and agricultural runoff (Section 4). The magnitude of the pollutant loads associated with these other pollutant sources in the RAAs is likely greater than the pollutant loads associated with the populations of vessels of the Armed Forces present in RAAs. Given that pollutant loads from vessel discharges are comparatively lower than those from other sources in the RAAs and given that the estimated

RQs from incidental vessel discharges are very low, the EPA and DoD believe that the UNDS Batch Two pollutants in vessel discharges analyzed in this BE are unlikely a primary pollution source for listed species with identified sensitivities to water pollution except in those ports and harbors that are predominantly military ports. Even in those ports that are predominantly military ports (e.g., Pearl Harbor and Apra Harbor), the largest inputs of most pollutants are from land based sources such as runoff and onshore operations rather than vessel discharges.

Given this assessment, the EPA and DoD concluded that, even when known threats are taken into consideration and potentially more vulnerable species are exposed to discharges from vessels of the Armed Forces regulated by UNDS, the risk for adverse effects to federally listed aquatic and aquatic-dependent species and their critical habitat from this action and the resulting exposure to chemical pollutants is “remote” to “negligible”. In addition, the standards reduce/minimize the risk of exposure to ANS and risk from other stressors such as nutrients, oil and grease, and PPCPs for federally listed species. Given this and all of the supporting information provided in Section 5.2, the EPA and DoD are confident that the use of the risk conclusions (Section 5.2.5) presented in Table 5-67 for the 111 RAA listed species to support the effects determinations in Section 8 appropriately captures the species variability in taxonomic group, spatial scale, life history, and environmental stressors for all 674 aquatic and aquatic-dependent listed species and their critical habitats that may be affected by vessels of the Armed Forces.

Table 5-67. Summary of Species-Specific Risk Conclusions for Listed Aquatic and Aquatic-Dependent Species in Representative Action Areas

Common Name (Scientific Name)	RAA(s)[a]	Receptor Type and Exposure Pathway of Concern[b]	Potential for Adverse Effects Via Loss of Prey[c]	Potential for Adverse Effects Via Loss of Habitat	Potential for Adverse Effects Via Decreased Water Quality	Potential for Adverse Effects Via Additional Risk Considerations[d]
			Remote	Remote	Negligible	None
Abalone, white (*Haliotis sorenseni*)	SD; SFBE	Aq Mult – SW	Remote	Remote	Negligible	None
Albatross, short-tailed (*Phoebastria (=Diomedea) albatrus*)	PH; PS; SD	Aq Dep Diet – SW	Remote	Remote	Remote	Pollution sensitivity (oil spills)
Aster, decurrent false (*Boltonia decurrens*)	SL	Aq Water Only - FW	NA	Remote	Remote	None
Bat, Florida bonneted (*Eumops floridanus*)	M	Aq Dep Diet – FW	Remote	Remote	Remote	Limited aquatic-dependence
Bat, gray (*Myotis grisescens*)	SL; M	Aq Dep Diet – FW	Remote	Remote	Remote	Limited aquatic-dependence
Bat, Indiana (*Myotis sodalist*)	SL; M	Aq Dep Diet – FW	Remote	Remote	Remote	Limited aquatic-dependence
Bat, Northern long-eared (*Myotis septentrionalis*)	N; SL	Aq Dep Diet – FW	Remote	Remote	Remote	Limited aquatic-dependence
Bear, grizzly (*Ursus arctos horribilis*)	PS	Aq Dep Diet – FW	Remote	Remote	Remote	None
Beetle, Delta green ground (*Elaphrus viridis*)	SFBE	Aq Dep Diet – FW	Remote	Remote	Remote	None
Bird's-beak, salt marsh (*Cordylanthus maritimus ssp. Maritimus*)	SD	Aq Water Only - SW	NA	Remote	Remote	Pollution sensitivity (oil spills)
Bird's-beak, soft (*Cordylanthus mollis ssp. Mollis*)	SFBE	Aq Water Only - SW	NA	Remote	Remote	Pollution sensitivity (oil spills)
Bocaccio (*Sebastes paucispinis*)	PS; SD	Aq Mult – SW	Negligible	Remote	Remote	None

Table 5-67. Summary of Species-Specific Risk Conclusions for Listed Aquatic and Aquatic-Dependent Species in Representative Action Areas (Continued)

Common Name (Scientific Name)	RAA(s)[a]	Receptor Type and Exposure Pathway of Concern[b]	Potential for Adverse Effects Via			
			Loss of Prey[c]	Loss of Habitat	Decreased Water Quality	Additional Risk Considerations[d]
Butterfly, Miami blue (*Cyclargus (=Hemiargus) thomasi bethunebakeri*)	M	Aq Dep Diet – FW	Remote	Remote	Remote	Limited aquatic-dependence
Clover, showy Indian (*Trifolium amoenum*)	SFBE	Aq Water Only - FW	NA	Remote	Remote	None
Coot, Hawaiian (*Fulica americana alai*)	PH	Aq Dep Diet – FW, SW	Remote	Remote	Remote	None
Coral, cauliflower (*Pocillopora meandrina*)	PH	Aq Mult – SW	Remote	Remote	Negligible	None
Coral, elkhorn (*Acropora palmata*)	M	Aq Mult – SW	Remote	Remote	Negligible	None
Coral, boulder star (*Orbicella franksi*)	M	Aq Mult – SW	Remote	Remote	Negligible	None
Coral, lobed star (*Orbicella annularis*)	M	Aq Mult – SW	Remote	Remote	Negligible	None
Coral, mountainous star (*Orbicella faveolata*)	M	Aq Mult – SW	Remote	Remote	Negligible	None
Coral, pillar (*Dendrogyra cylindricus*)	M	Aq Mult – SW	Remote	Remote	Negligible	None
Coral, rough cactus (*Mycetophyllia ferox*)	M	Aq Mult – SW	Remote	Remote	Negligible	None
Coral, staghorn (*Acropora cervicornis*)	M	Aq Mult – SW	Remote	Remote	Negligible	None
Crocodile, American (*Crocodylus acutus*)	M	Aq Dep Diet – FW, SW	Remote	Remote	Remote	None
Damselfly, crimson Hawaiian (*Megalagrion leptodemas*)	PH	Aq Mult – FW	Remote	Remote	Remote	None

Table 5-67. Summary of Species-Specific Risk Conclusions for Listed Aquatic and Aquatic-Dependent Species in Representative Action Areas (Continued)

Common Name (Scientific Name)	RAA(s)[a]	Receptor Type and Exposure Pathway of Concern[b]	Potential for Adverse Effects Via			
			Loss of Prey[c]	Loss of Habitat	Decreased Water Quality	Additional Risk Considerations[d]
Damselfly, orangeblack Hawaiian (*Megalagrion xanthomelas*)	PH	Aq Mult – FW	Remote	Remote	Remote	None
Duck, Hawaiian (=koloa) (*Anas wyvilliana*)	PH	Aq Dep Diet – FW, SW	Remote	Remote	Remote	None
Duck, Laysan (*Anas laysanensis*)	PH	Aq Dep Diet – FW, SW	Remote	Remote	Remote	None
Elepaio, Oahu (*Chasiempis ibidis*)	PH	Aq Dep Diet – FW	Remote	Remote	Remote	None
Eulachon (*Thaleichthys pacificus*)	PS; SFBE	Aq Mult – FW, SW	Negligible	Remote	Remote	None
Flycatcher, southwestern willow (*Empidonax traillii extimus*)	SD	Aq Dep Diet – FW	Remote	Remote	Remote	Limited aquatic-dependence
Frog, California red-legged (*Rana draytonii*)	SFBE	Aq Mult – FW	Remote	Remote	Remote	None
Frog, Oregon spotted (*Rana pretiosa*)	PS	Aq Mult – FW	Remote	Remote	Remote	None
Gallinule, Hawaiian common (*Gallinula chloropus sandvicensis*)	PH	Aq Dep Diet – FW	Remote	Remote	Remote	None
Goby, tidewater (*Eucyclogobius newberryi*)	SFBE	Aq Mult – FW, SW	Negligible	Remote	Remote	Pollution sensitivity
Goldfields, Contra Costa (*Lasthenia conjugens*)	SFBE	Aq Water Only – FW	NA	Remote	Remote	None
Grouper, Gulf (*Mycteroperca jordani*)	SD	Aq Mult – SW	Negligible	Remote	Remote	None
Grouper, Nassau (*Epinephelus striatus*)	M	Aq Mult – SW	Negligible	Remote	Remote	None

Table 5-67. Summary of Species-Specific Risk Conclusions for Listed Aquatic and Aquatic-Dependent Species in Representative Action Areas (Continued)

Common Name (Scientific Name)	RAA(s)[a]	Receptor Type and Exposure Pathway of Concern[b]	Potential for Adverse Effects Via			
			Loss of Prey[c]	Loss of Habitat	Decreased Water Quality	Additional Risk Considerations[d]
Howellia, water (*Howellia aquatilis*)	PS	Aq Water Only – FW	NA	Remote	Remote	None
Kite, Everglade snail (*Rostrhamus sociabilis plumbeus*)	M	Aq Dep Diet – FW	Remote	Remote	Remote	None
Knot, red (*Calidris canutus rufa*)	N	Aq Dep Diet – SW	Remote	Remote	Remote	None
Manatee, West Indian (*Trichechus manatus*)	M	Aq Mult – FW, SW	Remote	Remote	Remote	None
Manzanita, Franciscan (*Arctostaphylos franciscana*)	SFBE	Aq Water Only - FW	NA	Remote	Remote	None
Massasauga, eastern (=rattlesnake) (*Sistrurus catenatus)*	SL	Aq Dep Diet - FW	Remote	Remote	Remote	None
Meadowfoam, Sebastopol (*Limnanthes vinculans*)	SFBE	Aq Water Only - FW	NA	Remote	Remote	None
Mouse, salt marsh harvest (*Reithrodontomys raviventris*)	SFBE	Aq Dep Diet – SW	Remote	Remote	Remote	None
Murrelet, marbled (*Brachyramphus marmoratus*)	PS; SD; SFBE	Aq Dep Diet – SW	Remote	Remote	Remote	None
Mussel, scaleshell (*Leptodea leptodon*)	SL	Aq Mult – FW	Remote	Remote	Remote	None
Orchid, eastern prairie fringed (*Platanthera leucophaea*)	SL	Aq Water Only - FW	NA	Remote	Remote	None
Otter, Southern sea (*Enhydra lutris nereis*)	SFBE	Aq Mult – SW	Negligible	Remote	Remote	Pollution sensitivity (oil spills)
Petrel, Bermuda (*Pterodroma cahow*)	N	Aq Dep Diet – SW	Remote	Remote	Remote	None

Table 5-67. Summary of Species-Specific Risk Conclusions for Listed Aquatic and Aquatic-Dependent Species in Representative Action Areas (Continued)

Common Name (Scientific Name)	RAA(s)[a]	Receptor Type and Exposure Pathway of Concern[b]	Potential for Adverse Effects Via			
			Loss of Prey[c]	Loss of Habitat	Decreased Water Quality	Additional Risk Considerations[d]
Petrel, Hawaiian dark-rumped (*Pterodroma sandwichensis*)	PH	Aq Dep Diet – SW	Remote	Remote	Remote	None
Plover, piping (*Charadrius melodus*)	M; N; SL	Aq Dep Diet – SW, FW	Negligible	Remote	Remote	Pollution sensitivity (oil contamination)
Plover, western snowy (*Charadrius nivosus nivosus*)	SFBE; SD	Aq Dep Diet – SW, FW	Negligible	Remote	Remote	None
Potentilla, Hickman's (*Potentilla hickmanii*)	SFBE	Aq Water Only - FW	NA	Remote	Remote	None
Pu`uka`a (*Cyperus trachysanthos*)	PH	Aq Water Only - FW	NA	Remote	Remote	None
Rail, Eastern black (*Laterallus jamaicensis ssp. jamaicensis*)	PH	Aq Dep Diet – fW	Negligible	Remote	Remote	None
Rail, light-footed clapper (*Rallus longirostris levipes*)	SD	Aq Dep Diet – SW	Negligible	Remote	Remote	None
Ray, giant manta (*Manta birostis)*	M; N; PH; SD	Aq Mult – SW	Negligible	Remote	Remote	None
Rockfish, Yelloweye (*Sebastes ruberrimus*)	PS	Aq Mult – SW	Negligible	Remote	Remote	None
Salamander, California tiger (*Ambystoma californiense*)	SFBE	Aq Dep Mult – FW	Remote	Remote	Remote	None
Salmon, Chinook Central Valley ESU (*Oncorhynchus (=Salmo) tshawytscha*)	SFBE	Aq Mult – FW, SW	Negligible	Remote	Remote	Remote risk to juvenile FW lifestage; pollution sensitivity
Salmon, Chinook Puget Sound ESU (*Oncorhynchus (=Salmo) tshawytscha*)	PS; SFBE	Aq Mult – FW, SW	Negligible	Remote	Remote	Remote risk to juvenile FW lifestage; pollution sensitivity
Salmon, Chum Hood Canal Summer-Run ESU (*Oncorhynchus keta*)	PS	Aq Mult – FW, SW	Negligible	Remote	Remote	Remote risk to juvenile FW life stage; pollution sensitivity

Table 5-67. Summary of Species-Specific Risk Conclusions for Listed Aquatic and Aquatic-Dependent Species in Representative Action Areas (Continued)

Common Name (Scientific Name)	RAA(s)[a]	Receptor Type and Exposure Pathway of Concern[b]	Potential for Adverse Effects Via			
			Loss of Prey[c]	Loss of Habitat	Decreased Water Quality	Additional Risk Considerations[d]
Sawfish, smalltooth (US Portion of Range) (*Pristis pectinate*)	M	Aq Mult – SW	Remote	Remote	Remote	Pollution sensitive
Green sea turtle, Central North Pacific DPS[Y](*Chelonia mydas*)	PH	Aq Mult – SW	Remote	Remote	Remote	Limited nearshore exposure
Green sea turtle, North Atlantic DPS[X] (*Chelonia mydas*)	N; M	Aq Mult – SW	Remote	Remote	Remote	Limited nearshore exposure
Green sea turtle, East Pacific DPS[Y] (*Chelonia mydas*)	SD; SFBE	Aq Mult – SW	Remote	Remote	Remote	Limited nearshore exposure
Sea turtle, hawksbill (*Eretmochelys imbricate*)	PH; SD; M, N	Aq Mult – SW	Negligible	Remote	Remote	Limited nearshore exposure
Sea turtle, Kemp's ridley (*Lepidochelys kempii*)	N	Aq Mult – SW	Negligible	Remote	Remote	Limited nearshore exposure
Sea turtle, leatherback (*Dermochelys coriacea*)	N; PH; PS; SD; SFBE; M	Aq Mult – SW	Negligible	Remote	Remote	Limited nearshore exposure
Sea Turtle, Loggerhead Northwest Atlantic Ocean DPS (*Caretta caretta*)	N; M	Aq Mult – SW	Negligible	Remote	Remote	Limited nearshore exposure
Sea Turtle, Loggerhead North Pacific Ocean DPS (*Caretta caretta*)	PH; PS; SFBE; SD	Aq Mult – SW	Negligible	Remote	Remote	Limited nearshore exposure
Sea turtle, olive ridley (Mexico's Pacific coast breeding colonies) (*Lepidochelys olivacea*)	SD	Aq Mult – SW	Negligible	Remote	Remote	Limited nearshore exposure

Table 5-67. Summary of Species-Specific Risk Conclusions for Listed Aquatic and Aquatic-Dependent Species in Representative Action Areas (Continued)

Common Name (Scientific Name)	RAA(s)[a]	Receptor Type and Exposure Pathway of Concern[b]	Potential for Adverse Effects Via			
			Loss of Prey[c]	Loss of Habitat	Decreased Water Quality	Additional Risk Considerations[d]
Sea turtle, olive ridley (all other areas) (*Lepidochelys olivacea*)	PH; SD; SFBE	Aq Mult – SW	Negligible	Remote	Remote	Limited nearshore exposure
Seagrass, Johnson's (*Halophila johnsonii*)	M	Aq Water Only - SW	NA	Remote	Remote	None
Seal, Guadalupe fur (*Arctocephalus townsendi*)	SFBE	Aq Mult – SW	Remote	Remote	Remote	Pollution sensitivity (oil spills)
Seal, Hawaiian monk (*Monachus schauinslandi*)	PH	Aq Mult – SW	Remote	Remote	Remote	Pollution sensitivity (oil spills)
Shark, Scalloped Hammerhead Central and Southwest Atlantic DPS (*Sphyrna lewini*)	N; M	Aq Mult – SW	Remote	Remote	Remote	None
Shark, Scalloped Hammerhead Eastern Pacific DPS (*Sphyrna lewini*)	PH; SD; SFBE	Aq Mult – SW	Remote	Remote	Remote	None
Shearwater, Newell's Townsend's (*Puffinus auricularis newelli*)	PH	Aq Dep Diet – SW	Remote	Remote	Remote	None
Shrimp, California freshwater (*Syncaris pacifica*)	SFBE	Aq Mult – FW	Remote	Remote	Remote	Limited to vernal pools; pollution sensitivity
Shrimp, conservancy fairy (*Branchinecta conservation*)	SFBE	Aq Mult – FW	Remote	Remote	Remote	Limited to vernal pools; pollution sensitivity
Shrimp, San Diego fairy (*Branchinecta sandiegonensis*)	SD	Aq Mult – FW	Remote	Remote	Remote	Limited to vernal pools; pollution sensitivity
Shrimp, vernal pool fairy (*Branchinecta lynchii*)	SFBE	Aq Mult – FW	Remote	Remote	Remote	Limited to vernal pools; pollution sensitivity
Shrimp, vernal pool tadpole (*Lepidurus packardi*)	SFBE	Aq Mult – FW	Remote	Remote	Remote	Limited to vernal pools; pollution sensitivity

Table 5-67. Summary of Species-Specific Risk Conclusions for Listed Aquatic and Aquatic-Dependent Species in Representative Action Areas (Continued)

Common Name (Scientific Name)	RAA(s)[a]	Receptor Type and Exposure Pathway of Concern[b]	Potential for Adverse Effects Via			
			Loss of Prey[c]	Loss of Habitat	Decreased Water Quality	Additional Risk Considerations[d]
Smelt, delta (*Hypomesus transpacificus*)	SFBE	Aq Mult – FW, SW	Negligible	Remote	Remote	None
Snake, eastern indigo (*Drymarchon corais couperi*)	M	Aq Dep Diet – FW	Remote	Remote	Remote	None
Snake, giant garter (*Thamnophis gigas*)	SFBE	Aq Dep Diet – FW	Remote	Remote	Remote	None
Sonoma Alopecurus (*Alopecurus aequalis var. sonomensis*)	SFBE	Aq Water Only - FW	NA	Remote	Remote	None
Sparrow, Cape Sable seaside (*Ammodramus maritimus mirabilis*)	M	Aq Dep Diet – FW	Remote	Remote	Remote	None
Spectaclecase (mussel) (*Cumberlandia monodonta*)	SL	Aq Mult – FW	Remote	Remote	Remote	None
Stilt, Hawaiian (*Himantopus mexicanus knudseni*)	PH	Aq Dep Diet – SW	Negligible	Remote	Remote	None
Stork, wood (*Mycteria americana*)	M	Aq Dep Diet – FW	Remote	Remote	Remote	None
Sturgeon, Atlantic Carolina DPS (*Acipenser oxyrinchus oxyrinchus*)	N	Aq Mult – FW, SW	Negligible	Remote	Remote	None
Sturgeon, Atlantic Chesapeake Bay DPS (*Acipenser oxyrinchus oxyrinchus*)	N	Aq Mult – FW, SW	Negligible	Remote	Remote	None
Sturgeon, Atlantic New York Bight DPS (*Acipenser oxyrinchus oxyrinchus*)	N	Aq Mult – FW, SW	Negligible	Remote	Remote	None

Table 5-67. Summary of Species-Specific Risk Conclusions for Listed Aquatic and Aquatic-Dependent Species in Representative Action Areas (Continued)

Common Name (Scientific Name)	RAA(s)[a]	Receptor Type and Exposure Pathway of Concern[b]	Potential for Adverse Effects Via			
			Loss of Prey[c]	Loss of Habitat	Decreased Water Quality	Additional Risk Considerations[d]
Sturgeon, Atlantic South Atlantic DPS (*Acipenser oxyrinchus oxyrinchus*)	N	Aq Mult – FW, SW	Negligible	Remote	Remote	None
Sturgeon, green, North American southern DPS (*Acipenser medirostris*)	PS; SFBE	Aq Mult – FW, SW	Negligible	Remote	Remote	None
Sturgeon, pallid (*Scaphirhynchus albus*)	SL	Aq Mult – FW, SW	Remote	Remote	Remote	None
Sturgeon, shortnose (*Acipenser brevirostrum*)	N; M	Aq Mult – FW, SW	Negligible	Remote	Remote	None
Sunshine, Sonoma (*Blennosperma bakeri*)	SFBE	Aq Water Only – FW	NA	Remote	Remote	None
Tern, California least (*Sterna antillarum browni*)	SD	Aq Dep Diet – SW	Remote	Remote	Remote	None
Tern, least (*Sterna antillarum*)	SL	Aq Dep Diet – SW	Remote	Remote	Remote	None
Tern, roseate (*Sterna dougallii dougallii*)	N; M	Aq Dep Diet – SW	Remote	Remote	Remote	None
Thistle, fountain (*Cirsium fontinale var. fontinale*)	SFBE	Aq Water Only - FW	NA	Remote	Remote	None
Thistle, Suisun (*Cirsium hydrophilum var. hydrophilum*)	SFBE	Aq Water Only - FW	NA	Remote	Remote	None
Thoroughwort, Cape Sable (*Chromolaena frustrata*)	M	Aq Water Only - FW-	NA	Remote	Remote	None
Trout, bull (*Salvelinus confluentus*)	PS	Aq Mult – FW	Remote	Remote	Remote	Pollution sensitivity

Table 5-67. Summary of Species-Specific Risk Conclusions for Listed Aquatic and Aquatic-Dependent Species in Representative Action Areas (Continued)

Common Name (Scientific Name)	RAA(s)[a]	Receptor Type and Exposure Pathway of Concern[b]	Potential for Adverse Effects Via			
			Loss of Prey[c]	Loss of Habitat	Decreased Water Quality	Additional Risk Considerations[d]
Trout, Steelhead, Central California Coast DPS (*Oncorhynchus (=Salmo) mykiss*)	SFBE	Aq Mult – FW, SW	Negligible	Remote	Remote	Remote risk to juvenile FW life stage; pollution sensitivity
Trout, Steelhead, Puget Sound DPS (*Oncorhynchus (=Salmo) mykiss*)	PS	Aq Mult – FW, SW	Negligible	Remote	Remote	Remote risk to juvenile FW life stage; pollution sensitivity
Trout, Steelhead, Southern California DPS (*Oncorhynchus (=Salmo) mykiss*)	SD	Aq Mult – FW, SW	Negligible	Remote	Remote	Remote risk to juvenile FW life stage; pollution sensitivity
Vireo, least Bell's (*Vireo bellii pusillus*)	SD	Aq Dep Diet – FW	Remote	Remote	Remote	None
Whale, blue (*Balaenoptera musculus*)	SD	Aq Mult – SW	Remote	Remote	Remote	Limited nearshore exposure
Whale, False Killer (Main Hawaiian Islands Insular DPS) (*Pseudorca crassidens*)	PH	Aq Mult – SW	Remote	Remote	Remote	None
Whale, fin (*Balaenoptera physalus*)	N; SD; M	Aq Mult – SW	Remote	Remote	Remote	Limited nearshore exposure
Whale, humpback (*Megaptera novaeangliae*)	N; PH; PS; SD; M	Aq Mult – SW	Remote	Remote	Remote	Limited nearshore exposure
Whale, killer (southern Resident) (*Orcinus orca*)	PH; PS; SFBE	Aq Mult – SW	Remote	Remote	Remote	None
Whale, North Atlantic right whale (*Eubalaena glacialis*)	N; M	Aq Mult – SW	Remote	Remote	Remote	Limited nearshore exposure
Whale, Sei (*Balaenoptera borealis*)	N	Aq Mult – SW	Remote	Remote	Remote	Limited nearshore exposure
Whale, sperm (*Physeter catodon (=macrocephalus)*)	N; SD	Aq Mult – SW	Remote	Remote	Remote	Limited nearshore exposure

Table 5-67. Summary of Species-Specific Risk Conclusions for Listed Aquatic and Aquatic-Dependent Species in Representative Action Areas (Continued)

Common Name (Scientific Name)	RAA(s)[a]	Receptor Type and Exposure Pathway of Concern[b]	Potential for Adverse Effects Via			
			Loss of Prey[c]	Loss of Habitat	Decreased Water Quality	Additional Risk Considerations[d]
Whipsnake, Alameda (*Masticophis lateralis euryxanthus*)	SFBE	Aq Dep Diet – FW	Remote	Remote	Remote	None

a) RAA designations: PS=Puget Sound RAA; ULB=Upper and Lower Bay RAA; GB=Galveston Bay RAA; UMR=Mississippi River RAA; SLE=Saint Louis Estuary RAA; SFBE=San Francisco Bay Estuary RAA; BB=Biscayne Bay RAA.

b) FW = Freshwater; SW = Saltwater, Aq = Aquatic, Aq Dep = Aquatic Dependent, Mult = Multiple Routes of Exposure

c) As = arsenic, Se = selenium.

d) DPS = Distinct Population Segment (as classified under the ESA); ESU= evolutionary significant unit for Pacific Salmon, some of which may also be classified as a DPS under the ESA

e) The EPA identified species as sensitive to ANS and or pollution based on the identified threats presented in Table 4-19. The EPA limited the additional risk considerations in this table to ANS and pollution threats that could reasonably be attributed to vessel discharges covered under the VGP. For example, a species identified as sensitive to pollution from entanglement with debris is not designated as sensitive to pollution in this table as discharge of debris is not allowed under the VGP.

16.2 Appendix B. Appendix F of the BE, Methodology for Estuarine and Freshwater Harbor Modeling

Appendix F. Methodology for Estuarine and Freshwater Harbor Modeling

This Appendix provides additional details and descriptions of the methodology for the harbor modeling and exposure concentrations presented in Sections 5.1.1 through 5.1.3.

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Vessels that are regulated by Uniform National Discharge Standards (UNDS) discharge to coastal and inland waters throughout the United States. The Environmental Protection Agency (EPA) and DoD have modeled harbor concentrations of pollutants associated with discharges from vessels of the Armed Forces and compared these concentrations with chronic toxicity effects threshold (CTET) concentrations at, and below which, adverse effects are not likely to be observed. This Appendix describes the methodology for modeling harbor exposure concentrations utilized for this Biological Evaluation (BE). The following topics are discussed in detail:

Section F.1 - Model selection and model equations;
Section F.2 - Model input values and sources;
Section F.3 - Likely loading scenarios for RAAs and concentrations estimated for the RAA loading and flushing scenarios.

F.1. MODEL SELECTION

Estuarine (coastal) models commonly used to assess coastal harbor water quality, consist of two primary components: hydrodynamic (i.e., water transport) processes and pollutant inputs. These two components typically are used to predict pollutant concentrations. Estuarine models are generally classified into the following four levels according to the temporal and spatial complexity of the hydrodynamic component of the model (EPA, 2001):

Level I - Desktop screening models that calculate seasonal or annual mean concentrations based on steady-state conditions and simplified flushing time estimates.
Level II - Computerized steady-state or tidally averaged quasi-dynamic simulation models, which generally use a box or compartment-type network.
Level III - Computerized one-dimensional (i.e., estuary is well-mixed vertically and laterally) and quasi-two-dimensional (i.e., a link-node system describes estuary longitudinal and lateral mixing) dynamic simulation models.
Level IV - Computerized two-dimensional (i.e., represents estuary longitudinal and lateral mixing) and three-dimensional (i.e., represents estuary longitudinal, lateral, and vertical mixing) dynamic simulation models.

Because of the complexity and diversity of the different coastal harbor environments potentially impacted by UNDS regulated vessels, use of higher level models would require separate modeling efforts for each harbor, and the individual RAAs would not be representative of any other ports and harbors where vessels of the Armed Forces are home ported. The EPA and the Department of Defense (DoD) selected a Level I screening model and used conservative assumptions to generalize estimates of harbor pollutant concentrations. The level I screening model provided an opportunity to evaluate different "likely' scenarios under a range of environmental conditions. This screening model estimated dilution by flushing from fresh water and tides using a steady state equation for the estuarine harbor to support the effects analysis further described in Section 5.3.

Similar to estuarine models, river models can range in complexity from simple steady state representations of a river to more dynamic fate and transport models that compartmentalize the river environment and capture spatial and temporal variations in river inputs. The wide range of river environments potentially impacted by vessels regulated by UNDS also means that the EPA and DoD would need to consider wide ranging environmental conditions to support the BE effects analysis. The EPA and DoD selected a steady state dilution model to estimate receiving water concentrations for the riverine (inland) harbor scenario.

F.1.1. Flushing-Based Screening Model for Estuarine Harbor

The flushing-based screening model is a series of equations that represent the harbor environment in zero dimensions and at a steady state (USEPA, 2001). These calculations are zero-dimensional in that they estimate concentrations at a given point in a water body within a specified, spatially homogenous volume. The calculations assume instantaneous and homogeneous mixing of vessel discharges within the defined volume of a given harbor and modeled concentrations represent those to which ecological receptors will be chronically exposed. The model does not account for gradients of concentrations that would occur within a certain distance from discharge source(s) such as plumes from vessels and other sources. Specifying plumes and accounting for locations of numerous discharge sources would require a two- or three- dimensional model, which is beyond this Level I screening-level model.

Another reason the Level I screening model was selected because most vessel discharges do not occur while a vessel is stationary in port but rather while they are underway. Exceptions are hull coating leachate, which is a very slow discharge that will not result in a plume; underwater ship husbandry, which is an occasional discharge consisting predominantly of particulates; and sonar dome discharge, which is an occasional discharge limited to a very small number of vessels and locations. Other discharges released while in port either occur while a vessel is underway or are transferred to an onshore facility when a vessel is stationary in port. While a vessel is underway, any discharges to receiving water are expected to disperse relatively quickly because of the vessel's movement. Modeling of effluent discharged from a cruise ship travelling at a speed of 6 to 10 knots and discharging at a rate of 200 m^3/s showed a dilution by a factor of 600 to 2,500 within minutes (Colonell et al., 2000). However, surveys have shown that effluent from cruise ships travelling at speeds between 9 and 17 knots are rapidly diluted by a factor of 260,000 to 580,000 (Heinen et al., 2003). One study has shown that, even for stationary cruise ships, as effluent leaves the discharge port, it begins an initial dilution characterized by rapid and turbulent entrainment and a vertical rise. Observed dilutions in the near field zone extending out to about 2.5 meters ranged from about 1/4 to 1/12 of the concentrations leaving the discharge port (Alaska DEC, 2009). The same is expected for vessels of the Armed Forces traveling at similar speeds.

Steady state means that the calculations provide an instantaneous estimate of the concentration under the assumption of chemical and physical equilibrium and the assumption that the total modeled concentration is completely available (i.e., no ligands binding pollutants making them biologically unavailable). The assumption of chemical equilibrium implies that the water body salinity and the vessel discharge pollutant concentrations do not change over time, while physical equilibrium means that the volume of water in the water body, tides, currents, water column stratification, and vessel discharge flow rates do not change over time. The general assumption is that every process occurs under the equilibrium conditions; therefore, there isn't any temporal variability in concentrations. Accounting for changes in tides, currents, river flow, stratification, vessel discharge flow rates, and discharge concentrations over time would require a dynamic model, which is beyond this Level I screening model. Level I screening models do not provide details of vertical and horizontal velocity distributions, salinity stratification, pollutant concentrations, or circulation. Because they do not capture spatial and temporal variations in pollutant concentrations, they also do not allow temporal and spatial changes in risk of effects to be evaluated because it is assumed that there is chronic exposure to the maximum modeled concentration at every point and point in time after mixing has occurred.

The objective of this BE is to evaluate the risk of long-term chronic effects to sensitive populations from sustained exposures. While immediate and short-term exposure to relatively high concentrations of pollutants can have immediate population level effects in the form of mortality, long-term exposure to much lower concentrations can have chronic effects on survival, growth and reproduction. This can ultimately lead to decreased population levels. As discussed above, based on previous studies of discharges from cruise ships, pollutant concentrations in discharges are rapidly diluted upon discharge to receiving waters, and the highest concentrations of pollutants occur right at the discharge port where exposures are very limited, if they occur at all. It is the continuous exposure to pollutants that accumulate and persist that have the greatest potential to impact ecological populations. The EPA and DoD have used conservative model parameter inputs (i.e., inputs that may overestimate surface water concentrations) for variables such as harbor surface area, flow, and pollutant loadings. The modeling approach also assumes that estimated maximum concentrations are maintained continuously because vessel discharges are not changing over time.

The flushing-based screening model calculated the pollutant concentration in a harbor resulting from vessel discharges using the following four steps:

Step 1: Calculate vessel discharge pollutant loading rates for each discharge in each of the Representative Action Areas (RAAs) (Equations F-1 through F-15)
Step 2: Calculate the fraction of freshwater in the harbor (Equation F-16)
Step 3: Calculate the harbor flushing time (Equation F-17)
Step 4: Calculate the harbor pollutant concentration under the variety of harbors and conditions analyzed and select the highest concentration (Equation F-18)

The following subsections describe the input requirements, assumptions, and calculations for each step in the model.

F.1.1.1. Step 1: Calculate Vessel Discharge Pollutant Loading Rates

Pollutant-specific total discharge loading rates (We) are required as input values in the flushing-based screening model and the river dilution model to calculate the instantaneous pollutant concentrations in the harbor (Cx). In this analysis, the EPA and DoD generally estimated pollutant loading rates using the following three input parameters for each RAA:

Estimated average pollutant concentrations for each vessel type discharge type;
Estimated flow rate for each discharge type for each vessel type; and
Estimated number of vessels per vessel type present in the harbor.

$$We,c = \sum(Ce,y,c * Qy,c * Nc)$$

Where:

$\sum$=Sum for all vessels in the RAA
We,c= Discharge loading rate for analyte e from vessel class c (mass/time)
Ce,y,c = Average concentration of analyte e in discharge y from vessel class c (mass/volume)
Qy,c= Flow rate for discharge y from vessel class c (volume/time)
Nc= Number of vessel class c present in the harbor

The EPA and DoD calculated the pollutant-specific total discharge loading rate (We) by summing the discharge loading rates (We,c) for that pollutant from each vessel class for each RAA.

$$We = \sum(We,c)$$

Where:

We= Total discharge loading rate for pollutant e from study vessel discharges (mass/time)
We,c= Discharge loading rate for pollutant e from vessel class c (mass/time)

The specific equations for estimating mass loadings for each of the discharges selected for evaluation are presented in the following sections.

Graywater

Equations F-1 and F-2 below explain how the mass loading for each of the pollutants in graywater is calculated for each vessel of the Armed Forces. The total mass loading of each pollutant within each RAA is calculated as the sum of mass loadings from all vessels discharging and is subsequently used to model harbor concentrations (model equations F-15 through F-18).

Mass loading at normal discharge rate (45 gal/person/day)

Max mass loading in lb/yr = # transits/year * 2 hr/transit * (45 gal/person/d)/(24 hr/d) * max crew size * discharge concentration in µg/L * 3.785 L/gal * 0.0000000022 lb/µg

(*Equation F-1*)

Mass loading as minimized discharge rate (9 gal/person/day)

Minimized mass loading in lb/yr = # transits/year * 2 hr/transit * (9 gal/person/d)/(24 hr/d) * max crew size * discharge concentration in µg/L * 3.785 L/gal * 0.0000000022 lb/µg

(*Equation F-2*)

Surface Bilgewater/Oil-Water Separator Effluent

Equation F-3 below explains how the mass loading for each of the pollutants in surface bilgewater/OWS effluent is calculated for each vessel of the Armed Forces discharging. The total mass loading of each pollutant within each RAA is calculated as the sum of mass loadings from all vessels discharging and is subsequently used to model harbor concentrations (model equations F-15 through F-18).

Mass loading in lb/yr = # transits/year * 4 hr/transit * (gallons generated/day)/(24 hr/d) * fraction discharged * discharge concentration * 3.785 L/gal * 0.0000000022 lb/µg

(*Equation F-3*)

Hull Leachate

Equation F-4 below explains how the mass loading for each pollutant in hull leachate is calculated for each vessel of the Armed Forces. The total mass loading of each pollutant within each RAA is calculated as the sum of mass loadings from all vessels discharging and is subsequently used to model harbor concentrations (model equations F-15 through F-18).

Mass Loading in lb/yr = release rate in µg/cm^2/d * 0.0000000022 lb/µg * 929.03 cm^2/ft^2 * wetted surface area in ft^2 * # days in port/yr

(*Equation F-4*)

The number of days in port used for the calculation in Equation F-4 includes both the time that the vessel is pierside and while in transit. Transit time through port is assumed to be two hours.

For hull coating leachate, the mass loading rate was calculated slightly differently from the other discharges. For each RAA, loading rate was calculated as:

$$We,v = \sum(Re,h * SAc * T)$$

Where:

$\sum$=Sum for all vessels in the RAA
We= Discharge loading rate for analyte *e* (mass/time)
Re,h= Release rate of analyte *e* for vessel hull material *h* (mass/unit area/time)
SAc= Wetted hull surface area for vessel class *c*
T= Amount of time vessel *v* spends in port

Wet and Dry Firemain

Equations F-5 and F-6 below explain how annual firemain discharge volume is calculated for each vessel of the Armed Forces.

Upper Bound Estimate Wet Firemain Discharge in gal/yr = flow rate in gal/min * 60 min/hr * 24 hr/d * days within 12 nm/yr

(*Equation F-5*)

Upper Bound Estimate Dry Firemain Discharge in gal/yr = flow rate in gal/min * 10 min/wk * 1 wk/7 d * days within 12 nm/yr

(*Equation F-6*)

Firemain discharge volume for each vessel discharging is then used to calculate pollutant mass loadings. Equation F-7 below explains how the mass loading for each pollutant in wet and dry firemain is calculated for each vessel of the Armed Forces discharging. The total mass loading of each pollutant within each RAA is calculated as the sum of mass loadings from all vessels discharging and is subsequently used to model harbor concentrations (model equations F-15 through F-18).

Mass Loading in lb/yr = annual discharge in gal/yr * discharge concentration in µg/L * 0.0000000022 lb/µg * 3.78541178 L/gal

(*Equation F-7*)

Sonar Dome

Equation F-8 below explains how the annual internal sonar dome discharge volume is calculated for each vessel of the Armed Forces with internal sonar dome discharge.

Discharge volume in gal/yr = discharge volume/event [gal] * # events/yr

(*Equation F-8*)

Sonar dome discharge volume for each vessel discharging is then used to calculate pollutant mass loadings from internal sonar dome discharge. Equations F-9 and F-10 below explain how the mass loading for each pollutant in internal sonar dome discharge is calculated for each vessel of the Armed Forces discharging.

Interior mass loading for metals in lb/yr = discharge volume in gal/yr * discharge concentration in µg/L * 0.0000000022 lb/µg * 3.78541178 L/gal

(*Equation F-9*)

Interior mass loading for COD and TOC in lb/yr = discharge volume in gal/yr * discharge concentration in mg/L * 0.0000022 lb/µg * 3.78541178 L/gal

(*Equation F-10*)

Pollutants also leach from the exterior surface of sonar domes. Equation F-11 explains how the mass loading of TBT from external leaching is calculated for each vessel of the Armed Forces discharging. Only external mass loading of TBT is calculated for sonar domes, specifically rubber sonar domes. Leaching of all other pollutants (e.g., copper) is included in the hull leachate mass loading calculations. Because only Navy surface vessels in a few locations (Norfolk, Pearl Harbor, Puget Sound, and San Diego) have rubber sonar domes and because there are only 3 types/sizes, an average release rate per vessel is used for exterior mass loading calculations instead of a release rate per unit of area. The average was measured for three vessels that represent the larger sonar domes (24,000 gallons) and, therefore, the high end of the range for mass loadings of pollutants.

Exterior TBT mass loading in kg/d = # hulls with rubber sonar domes * mean release rate in g/vessel/d * 0.001 kg/g

(*Equation F-11*)

The total mass loading of each pollutant within each RAA is calculated as the sum of mass loadings from all vessels discharging and is subsequently used to model harbor concentrations (model equations F-15 through F-18).

Deck Runoff

All vessels of the Armed Forces generate deck runoff both within and outside of port. Equation F-12 below explains how deck runoff is calculated for each class of vessels of the Armed Forces discharging within each RAA.

Annual deck runoff in gal/yr = weather deck area in ft^2 * average annual rainfall in in/yr * 1 ft/12 in * 1 yr/365 days * 7.48 gal/ft3 * # vessels in the class * # days in port/yr

(*Equation F-12*)

The annual deck runoff for each vessel class is then used to calculate oil and grease mass loadings. Equation 13 explains how the mass loading for oil and grease in deck runoff is calculated for each class of vessels of the Armed Forces discharging within each RAA. The total mass loading within each RAA is calculated as the sum of mass loadings from all vessel classes and is subsequently used to model harbor concentrations (model equations F-15 through F-18 provided in the next section).

Mass loading in lb/yr = annual deck runoff in gal/yr * average concentration in mg/L * 3.785 L/gal * 0.0000022 lb/mg

(*Equation F-13*)

Underwater Ship Husbandry

Underwater ship husbandry includes hull cleaning, inspections, and hull repairs. The predominant discharge from underwater ship husbandry is from in-water hull cleaning, with the greatest pollutants of concern being those associated with antifouling coatings. Equations F-14a, b and c below explains how mass loadings of copper and zinc from full, partial and interim in-water hull cleanings, respectively, are calculated for each vessel of the Armed Forces. Equation F-14c explains how mass loadings of CPO from SEAWOLF propulsor layup is calculated. The total mass loading per vessel per year from in-water hull cleanings is calculated as the sum of the mass loadings from full, partial and interim cleanings.

Mass loading Cu or Zn from full in-water hull cleanings in kg/yr = (release rate in g/m^2 * 0.001 kg/g) * (wetted hull surface area in ft^2 * 0.092903 m^2/ft^2) * average # full in-water hull cleanings/yr

(*Equation F-14a*)

Mass loading Cu or Zn from partial in-water hull cleanings in kg/yr = (release rate in g/m^2 * 0.001 kg/g) * (0.30 * wetted hull surface area in ft^2 * 0.0929 m^2/ft^2) * average # partial in-water hull cleanings/yr

(*Equation F-14b*)

Mass loading Cu or Zn from interim in-water hull cleanings in kg/yr = (release rate in g/m^2 * 0.001 kg/g) * (0.15 * wetted hull surface area in ft^2 * 0.0929 m^2/ft^2) * average # interim in-water hull cleanings/yr

(*Equation F-14c*)

Mass loading of CPO in kg/yr = release rate in grams/event * 0.001 kg/g * # events/yr

(*Equation F-14d*)

For all discharges, mass loadings are converted to kg/day by dividing the loading rate by 365 days/yr or multiplying by 0.4536 kg/lb, as appropriate.

F.1.1.2. Step 2: Calculate the Fraction of Freshwater in the Estuarine Harbor

The next step is to calculate the fraction of freshwater using Equation F-15, as defined by the EPA (2001). The fraction function, fx, ranging from 0 to 1, measures the degree of freshwater content in the water with 0 representing no fresh water and 1 representing pure fresh water. The fraction of freshwater (*fx*) at any location in the estuary is calculated using the following equation:

$$fx = (So - Sx)/So \qquad \text{(Equation F-15)}$$

Where:

fx = Fraction of Freshwater at location x in the harbor (unit-less)
So = Seaward boundary Salinity at the mouth of the harbor (PSU)
Sx = Salinity at location x in the harbor (PSU)

F.1.1.3. Step 3: Calculate the Estuarine (Coastal) Harbor Flushing Time

Step 3 calculates the flushing time, using Equation F-16. For practical reasons, flushing of the harbor water can be driven by either freshwater inflows or tidal forcing. Tidal prism is defined as volume of water in an estuary or harbor between mean high tide and mean low tide, or the volume of water leaving an estuary at ebb tide. For a conservative assumption, the lower value of freshwater inflow or tidal prism determined the primary factor that influenced the flushing in the harbor. The flushing time (t_F) of the estuarine harbor is calculated using the following equation:

$$t_F = \min\left(\left(fx * V\right)/\mathrm{R}\right) \text{ or } \left(\left(V * T\right) / *V_T + \left(R * T\right)\right) \qquad \textit{Equation F-16a and b}$$

Where:

t_F= Harbor flushing time (days)
fx= Fraction of freshwater at location x in harbor (unit-less)
V= Volume of harbor (m^3)
R= Inflow of freshwater to harbor from the primary river input (m3/day)
V_T= Tidal Prism (m^3)
T = Tidal Cycle/Period (days)

Using this approach, only the primary flushing mechanism (tidal flushing or freshwater flushing) will be used in the pollutant concentration prediction. Other flushing mechanisms are not considered, which tends to over-estimate the predicted concentration.

F.1.1.4. Step 4a: Calculate the Estuarine Harbor Pollutant Concentration

The concentration of a pollutant at location x (Cx) is the pollutant-specific total loading rate (We, mass/time) divided by the flow rate away from location x, described by the volume of the harbor (V) (i.e., the RAA) divided by the flushing time (t_F) (USEPA, 2001):

$$Cx{,}estuary = We/(V/tF) \qquad \textit{Equation F-17}$$

Where:

$Cx{,}estuary$= Instantaneous pollutant concentration at location x in estuary harbor (mass/volume)
We= Pollutant-specific loading rate (mass/time) as calculated in Step 1
V= Volume of the harbor
t_F= Harbor flushing time as calculated in Step 3, Equation F-16a (estuarine) or 16b (freshwater)

F.1.2. <u>Dilution Model for Estimating River (Inland) Harbor Concentrations</u>

The EPA and DoD selected a dilution model to estimate the receiving water concentrations for the river environments potentially impacted by vessel discharges covered by UNDS. After estimating the pollutant-specific discharge load values for the river harbor environment, equation F-18 calculates the receiving water concentration for the river harbor environment based on river flow.

$$C_{x,river} = W_e/Q_{river} \qquad \textit{Equation F-18}$$

Where:

$C_{x,river}$ = Instantaneous pollutant concentration at location x in river harbor (mass/volume)
W_e = Pollutant-specific loading rate (mass/time) as calculated in Step 1
Q_{river}= Average annual river flow rate (volume/time)

The freshwater harbor model used average river flow to model pollutant concentrations in receiving water to represent the most common conditions to which aquatic and aquatic-dependent species could be exposed.

F.2. SELECTION OF MODEL INPUT VALUES

As discussed in Section F.1.1.1, the following information is needed to estimate pollutant loadings from each discharge for the RAAs:

Average concentration of pollutant C_e in discharge y from vessel type z ($C_{e,y,z}$)
Flow rate for discharge y from vessel type z ($Q_{y,z}$)
Number of vessel type z present in the harbor ($N_{,z}$)

The flushing-based screening model used to model the estuary harbors requires the following input parameters to define the water body characteristics:

Seaward boundary salinity at the mouth of the harbor (S_o)
Salinity at location x in the harbor (S_x)
Volume of the harbor (V)
Inflow of freshwater to the harbor (R)
Tidal Prism (V_T)
Tidal Cycle/Period (T)

The EPA and DoD collected data on the input parameters for the estuary harbors selected as RAAs to develop the water body characteristics for the six estuary harbor scenarios. These harbors were carefully chosen to be a representative subset of the total Armed Forces harbor population and contain the widest possible range of water body flow, flushing time, salinity, vessel populations, and loading assumptions. In addition, these harbors have the following representative characteristics:

- Homeports for 2,474 vessels of the Armed Forces (1,825 under 79 feet and 649 over 79 feet) representing approximately 39 percent of the total population of vessels of the Armed Forces
- Contain the three Homeports with the most vessels of the Armed Forces: Norfolk, VA (974); San Diego, CA (791); and Pearl Harbor, HI (217)
- Reflect a wide geographic range that captures a wide variety of T&E Species with harbors on the east and west coasts, a Pacific island, in northern and southern regions of the United States, and within a riverine system
- Reflects a wide variety of sizes and hydrodynamic characteristics, including volume, depth, surface area, freshwater inflow, salinities, and tides.

Table F-1 presents the harbor characteristics identified by the EPA and DoD for each estuary harbor used for the model input parameters. The EPA and DoD selected the input parameters for the volume of each harbor based on the harbor surface area and mean depth. In order to prevent the dilution of pollutant concentrations resulting from discharges from vessels of the Armed Forces, the harbor surface area was based on a three-mile radius of the homeport where vessels of the Armed Forces are located. This focused on the area where vessels of the Armed Forces are more likely to occur rather than the larger estuary. For example, the Chesapeake Bay is substantially larger than the Norfolk RAA where vessels of the Armed Forces are concentrated. The EPA and DoD assumed an average seaward boundary salinity (at the mouth of each harbor) of 35 PSU and the input parameters for the harbors' salinity were selected based on the actual salinity of the particular harbor. The freshwater flow was based on the river inflows to the harbors which directly impact the selected harbor area. The EPA and DoD selected the input parameters for the tidal prism based on the harbors surface area and tidal height. To add conservatism to the model, the tidal height was based on the lowest tide observed at any vessel of the Armed Forces location within an RAA. All tidal cycles were assumed to be 12 hours (0.5 days).

Table F-1. Representative Action Area (RAA) Estuary Harbor Characteristics

Harbor City	Estuary Name	Primary River Flowing into Estuary	Surface Area (m^2)[a]	Mean Depth (m)[b]	Harbor Volume (m^3)[c]	Harbor Salinity (PSU)[d]	River Flow (m^3/day)[e]	Mean Tide (m)[f]	Tidal Prism (m^3)[g]	Tidal Cycle/ Period (days)
Miami, FL	Biscayne Bay	Miami River	45,680,000	2.90	132,271,008	31.00	266,000	0.67	30,605,600	0.5
Norfolk, VA	Chesapeake Bay	James River	265,430,000	6.40	1,698,752,000	19.80	23,818,000	0.78	207,035,400	0.5
Pearl Harbor, HI	Pearl Harbor	Halawa Stream and Waikele Stream	19,340,000	9.10	175,994,000	32.80	103,344	0.38	7,291,180	0.5
San Diego, CA	San Diego Bay	Chollas Creek and Sweetwater River	40,610,000	6.40	259,936,488	33.49	31,436	1.19	48,325,900	0.5
San Francisco, CA	San Francisco Bay	Sacramento River and San Joaquin River	190,450,000	4.27	812,688,240	24.30	41,194,000	1.27	241,871,500	0.5
Seattle, WA	Puget Sound	Snohomash River and Puyallup River	211,890,000	137.16	29,062,832,400	28.20	47,349,152	1.60	339,024,000	0.5

a) Surface Area was calculated from Surface Area metadata (combined surface area of all GIS polygons within defined RAA)

b) The Mean Depths for Miami and Seattle were obtained from Table G-1 of the 2013 VGP BE, The Mean Depth for the remaining harbors were obtain from the following sources :

1. Chesapeake Bay:
   http://www.navfac.navy.mil/content/dam/navfac/Environmental/PDFs/MRA/chesapeake%20bay%20mra%20final%20-%20june%202009.pdf
2. Pearl Harbor:
   http://www.google.com/url?sa=t&rct=j&q=&esrc=s&frm=1&source=web&cd=3&cad=rja&uact=8&ved=0CCgQFjACahUKEwiSpanMx5XJAhUDpR4KHS1iCAw&url=http%3A%2F%2Fwww.dtic.mil%2Fcgi-bin%2FGetTRDoc%3FAD%3DADA607427&usg=AFQjCNEqZYDILZk3rCLQ1rFF3nSrdH192g&bvm=bv.107467506,d.dmo
3. Puget Sound:
   http://www.ecy.wa.gov/puget_sound/overview.html
4. San Diego:
   http://www.cnic.navy.mil/content/dam/cnic/cnrsw/NAVFACSW%20Environmental%20Core/SDBay_Final_INRMP_102913.pdf
5. San Francisco:

http://www.thebayinstitute.org/page/detail/95

c) Harbor Volume was calculated by multiplying the Mean Depth by the calculated Surface Area.
d) Harbor Salinity was obtained from Table G-1 of the 2013 VGP BE, unless another source is identified below:
   1. Pearl Harbor:
      http://www.google.com/url?sa=t&rct=j&q=&esrc=s&frm=1&source=web&cd=3&cad=rja&uact=8&ved=0CCgQFjACahUKEwiSpanMx5XJAhUDpR4KHS1iCAw&url=http%3A%2F%2Fwww.dtic.mil%2Fcgi-bin%2FGetTRDoc%3FAD%3DADA607427&usg=AFQjCNEqZYDILZk3rCLQ1rFF3nSrdH192g&bvm=bv.107467506,d.dmo
   2. San Diego:
      http://www.sandiego.gov/mwwd/pdf/sbwrp_2014_fullrpt.pdf
      *Note: 33.49 is the mean salinity value over the entire water column (1-55 m) from all SBOO stations during 2014.

e) River Flow was obtained from U.S. Geological Survey database: http://waterdata.usgs.gov/
f) Mean Tide was obtained from NOAA database: http://tidesandcurrents.noaa.gov/
g) Tidal Prism was calculated from Surface Area and Tidal Height.

As discussed in Section 5.2.1.1, the dilution equation for the river harbor model requires the average annual river flow rate (*Qriver*) to calculate a receiving water concentration. Table F-2 presents the river flow rates for the river harbor selected as the representative RAA for evaluation.

Table F-2. Representative Action Area (RAA) River Harbor Characteristics

Harbor City	Harbor Name	Primary Rivers Flowing into Harbor	Surface Area (m^2)[a]	River Flow (m^3/day)
St. Louis, MO	Upper Mississippi River	Mississippi River and Missouri River	22,860,000	424,000,000

a) Source: NHD Plus Data.

F.3. DEVELOPMENT OF REPRESENTATIVE ACTION AREA (RAA) LOADING SCENARIOS

The EPA and DoD identified a total of 21 pollutants to include in the quantitative effects analysis based on the vessels, discharges, and corresponding pollutants/constituents selected for detailed evaluation in this BE (see Section 3.2.3). This section presents the mass loading estimates and harbor concentrations based on the assumptions for the flow rates and pollutant concentrations expected with implementation of the UNDS. These flow rates and pollutant concentrations are used to calculate the vessel-specific loading rates (We,z) in Equation F-1.

The Nature of Discharge Reports prepared in 1998 and published in 1999 served as a basis for the pollutant load calculations, along with current expert knowledge of the Navy UNDS discharge leads. Several of the assumptions for discharge flow rates, pollutant concentrations, and vessel classes that produce discharges have changed because of changes in the fleet, available information, and the area of interest, among other reasons.

F.3.1. Graywater Constituent Load Development

Graywater is defined in section 312(a) of the Clean Water Act as wastewater from showers, baths and galleys. On vessels of the Armed Forces, drainage from laundry, interior deck drains, lavatory sinks, water fountains, and miscellaneous shop sinks is often collected together with graywater. Therefore, this discharge covers graywater as well as mixtures of graywater with wastewater from these additional sources.

There are approximately 238 vessels of the Armed Forces discharging graywater within the RAA ports and harbors (Table F-3). While pierside, graywater is pumped onshore for treatment. Graywater is only discharged overboard when the vessel is underway, and the transit time for a vessel to travel from pierside to outside of 3 n.m offshore (the outer boundary of the RAA) is up to two hours. The average per capita discharge rate is assumed to be 45 gal/person/day and is based on the rates and generation sources presented in Table F-4. Both normal and minimized discharge rates are presented, and graywater mass loadings are calculated for both a normal discharge rate of 45 gal/person/day and a “minimized” discharge rate of 9 gal/person/day, the latter representing discharge during specific UNDS-legislated conditions under which vessels are required to minimize graywater production.

Table F-3. Number of Vessels of the Armed Forces Discharging Graywater in Each Representative Action Area (RAA)

RAA	Army	Coast Guard	MSC	Navy	Total
Miami	0	2	0	0	2
Norfolk	15	12	7	67	101
Pearl Harbor	0	0	2	28	30
Puget Sound/Seattle	1	9	1	17	28
San Diego	0	6	3	55	64
San Francisco	8	2	3	0	13
				Grand Total	**238**

Table F-4. Graywater Sources and Normal and Minimized Discharge Rates (based on validated data for DDG class vessels)

Generation Source	Generation Rate		Normal Discharge Rate (gpd)	Minimized Discharge Rate (gpd)	Assumptions
Laundry	40	gal/load	5.7	0	Normal discharge for one load per week. No laundry for minimized discharge.
Showers	3	gal/min	30.0	0	Low flow and one 10 min shower per day. No showers for minimized discharge.
Sinks	1.5	gal/min	0.3	0.025	10 hand washes per day at 10 seconds per hand wash. One hand wash for minimized discharge.
Galley	7	gal per person	7.0	7.0	Roughly used for cooking purposes. Assumed no change with minimize.
Incidental	2	gal per person	2.0	2.0	Water fountains, mop water, etc. Same for normal and minimize.
Total Discharge Rate:			**45.0**	**9.0**	

Each vessel graywater discharge rate depends on vessel crew size and the number of transits per year. The crew size and average number of transits per year assumed for each vessel class are presented in Table F-5.

Table F-5. Crew Size and Average Number of Transits Within 3 Nautical Miles of Port Annually

Vessel Class	Average Crew Size	Average Transits/Year
AO 187	449	20
AOE 6	449	12
ARS 50	95	44
CG 47	348	24
CVN 68	4657	22
DDG 51	393	22
LCU 135	11	6
LCU 2000	13	6
LHA 6	1142	18
LHD 1	1142	26
LPD 17	593	22
LSD 41	398	26
LSD 49	398	26
LSV 1	30	40
LSV 7	30	40
MCM 1	35	56
SSBN 726	149	16
SSN 21	117	16
SSN 688	105	16
SSN 774	117	16
WAGB 399	99	8
WAGB 420	55	8
WHEC 378	116	26
WMEC 210	69	18
WMEC 270	81	18
WPB 110	16	14
WPB 87	10	14
Grand Total	**398**	**18**

The primary pollutants of concern in graywater (i.e., those that exceed the most stringent water quality criteria) are ammonia, copper, lead, mercury, nickel, silver, and zinc. Concentrations of these pollutants reported in the NOD Report (EPA, 1999a) and used in the mass loading calculations are presented in Table F-6. Pharmaceutical and personal care products (PPCPs) are also pollutants of concern but have not been measure in vessel graywater. This is a data gap in the analysis, and PPCPs are not modeled.

Table F-6. Concentrations of Pollutants of Concern in Graywater

Pollutant	NOD Report Flow-Weighted Average (µg/L)
Ammonia	102,300
Copper	936
Lead	247
Mercury	1.3
Nickel	42
Silver	8
Zinc	501

The EPA and DoD determined the vessel populations by using the Armed Forces Vessel Database which tracks all vessels of the Armed Forces regulated by UNDS. To calculate the total pollutant specific load for each of the seven RAA loading scenarios, the EPA and DoD summed the results of the pollutant loads multiplied by the number of vessels. Pollutant loads for each RAA were then used as inputs to the flushing-based estuarine harbor screening model and river harbor dilution model to estimate surface water concentrations in ports and harbors. The maximum modeled surface water concentration for each pollutant of concern was selected as the harbor concentration to represent surface water concentrations in the action area resulting from graywater discharge (see Section F.3).

F.3.2. Surface Bilgewater/Oil-Water Separator Effluent Discharge Constituent Load Estimates

There are approximately 838 vessels of the Armed Forces discharging bilgewater that has been treated with an oil-water separator within the RAA ports and harbors (Table F-7). The Armed Forces do not discharge untreated bilgewater. Under the UNDS regulations, oil concentrations in all discharges must be less than 15 ppm before bilgewater/OWS effluent can be discharged overboard, the concentration above which an oily sheen will be observed.

Table F-7. Number of Vessels of the Armed Forces Discharging Bilgewater/Oil-Water Separator Effluent in Each Representative Action Area (RAA)

RAA	Air Force	Army	Coast Guard	Marine Corps	MSC	Navy	Total
Miami	0	0	4	0	0	0	4
Norfolk	0	35	36	0	22	215	308
Pearl Harbor	0	0	0	0	5	89	94
Puget Sound	2	8	21	0	1	126	158
San Diego	0	0	6	55	9	168	238
San Fancisco	0	9	18	4	2	0	33
St. Louis	0	0	2	0	0	0	2
						Grand Total	**838**

The primary pollutants of concern in bilgwater (i.e., those that exceed the most stringent water quality criteria) are ammonia, nitrate, total Kjeldahl nitrogen (TKN), nitrogen, phosphorus, total petroleum hydrocarbons (TPH), copper, iron, mercury, nickel, and zinc. Measured bilgewater concentrations of these pollutants reported in the NOD (EPA, 1999b) are presented in Table F-8 and are for untreated bilgewater. Because bilgewater is not discharged overboard until it is run through an OWS, the value reported for TPH is the maximum allowed for discharge of OWS effluent.

Table F-8. Concentrations of Pollutants of Concern in Bilgewater

Pollutant	Log-Normal Mean Concentration (µg/L)
Ammonia	90
Nitrate	270
TKN	1500
Total Nitrogen	1770
Total Phosphorus	1810
TPH	15[1]
Total Copper	341
Dissolved Copper	163
Iron	472
Mercury	0
Total Nickel	169
Dissolved Nickel	176
Total Zinc	879
Dissolved Zinc	856

[1]Represents the maximum allowable concentration of TPH in discharge.

The EPA and DoD determined the vessel populations by using the Armed Forces Vessel Database which tracks all vessels of the Armed Forces regulated by UNDS. To calculate the total pollutant specific load for each of the seven RAA loading scenarios, the EPA and DoD summed the results of the pollutant loads multiplied by the number of vessels. Pollutant loads for each RAA were then used as inputs to the flushing-based estuarine harbor screening model and river harbor dilution model to estimate surface water concentrations in ports and harbors. The maximum modeled surface water concentration for each pollutant of concern was selected as the harbor concentration to represent surface water concentrations in the action area resulting from bilgewater/OWS discharge (see Section F.3.).

F.3.3. Submarine Bilgewater Constituent Load Estimates

There are approximately 79 vessels of the Armed Forces that generate submarine bilgewater, 44 of which are home ported in the RAAs. The Armed Forces do not discharge submarine bilgewater within 3 miles of shore. In port, submarine bilgewater is pumped to an onshore facility, and bilgwater is held while the vessel is transitting between 0 – 3 miles. Although some Fast Attack submarines discharge bilgewater once outside of 3 miles, all other submarines hold their bilgewater and do not discharge until they are outside of 12 miles of shore. None of the submarines discharge in port or near shore where pollutants and stressors can accumulate; therefore, mass loadings of pollutants in submarine bilgewater to ports and harbors were not estimated.

F.3.4. Hull Coating Leachate Constituent Load Estimates

In 40 CFR Part 1700, the Uniform National Discharge Standards (UNDS) for vessels of the Armed Forces defined hull coating leachate as "...constituents that leach, dissolve, ablate, or erode from the paint on the hull into the surrounding seawater." There are approximately 1,015 vessels of the Armed Forces within the RAA ports and harbors with hull coatings that leach pollutants into surface waters (Table F-9). The types of pollutants leached depends on the specific hull coating and may include copper, iron, zinc, tributyltin (TBT), resin, rosin, plasticizers, copolymers, and specially formulated biocides such as Sea-Nine 211.

Table F-9. Number of Vessels of the Armed Forces With Hull Coating Leachate in Each Representative Action Area (RAA)

RAA	Air Force	Army	Coast Guard	MSC	Navy	Total
Miami	0	0	11	0	0	11
Norfolk	0	47	20	43	277	387
Pearl Harbor	0	0	0	6	121	127
Puget Sound	2	9	21	4	165	201
San Diego	0	0	8	25	228	261
San Francisco	0	10	14	2	0	26
St. Louis	0	0	2	0	0	2
					Grand Total	**1015**

The release rate for each analyte has been determined to be dependent on several factors including the type of hull coating, vessel movement, and age of the hull coating. The release rates in Table F-10 were used to model release rates for metals and synthetic polymers. Because pollutants will only tend to accumulate when they are in ports and harbors where water circulation is restricted, statics release rates were used, and the number of days in port were used to estimate mass loading rates. Because of concerns with copper toxicity and ecological impacts from releases of copper to sensitive coastal areas, loading rates were conservatively estimated using maximum measured release rates.

Table F-10. Static Pollutant Leaching Rates for Different Hull Coatings

Analyte	Non-Aluminum Rigid Hulls	Flexible Hulls	Aluminum Hulls
Copper (mean)[1]	3.8	3.8	3.8
Copper (min.)[1]	3.0	3.0	3.0
Copper (max.)[1]	9.0	9.0	9.0
Iron	0.44	0.44	--
Zinc	3.6	3.6	17
N-ethylenesulfonomide	0.52	0.52	--
Plasticizer	0.47	0.47	--
Polymer Resin	0.47	0.47	--
Rosin	1.6	1.6	--
Sea-Nine211 (4,5-dichloro2-n-octyl-4-isothiazolin-3-one)	--	--	1.8

Note: Values are from 2003 Hull Leachate Discharge Analysis Report (DAR) (Navy and EPA, 2003) unless otherwise noted.
[1]2000 SSC SD
[2]1999 Hull Leachate Nature of Discharge Report (based on a 1993 study) (EPA, 1999c)

The EPA and DoD determined the vessel populations by using the Armed Forces Vessel Database which tracks all vessels of the Armed Forces regulated by UNDS. To calculate the total pollutant specific load for each of the seven RAA loading scenarios, the EPA and DoD summed the results of the pollutant loads multiplied by the number of vessels. Pollutant loads for each RAA were then used as inputs to the flushing-based estuarine harbor screening model and river harbor dilution model to estimate surface water concentrations in ports and harbors. The maximum modeled surface water concentration for each pollutant of concern was selected as the harbor concentration to represent surface water concentrations in the action area resulting from hull leachate (see Section F.3.).

F.3.5. <u>Firemain Discharge Constituent Load Estimates</u>

Firemain systems distribute seawater for fire fighting and secondary services. The fire fighting services are fire hose stations, seawater sprinkling systems, and foam proportioning stations. Fire hose stations are distributed throughout the ship. Seawater sprinkling systems are provided for spaces such as ammunition magazines, missile magazines, aviation tire storerooms, lubricating oil storerooms, dry cargo storerooms, living spaces, solid waste processing rooms, and incinerator rooms. Foam proportioning stations are located in rough proximity to the areas they protect, but are separated from each other for survivability reasons. Foam proportioners inject fire fighting foam into the seawater, and the solution is then distributed to areas where there is a risk of flammable liquid spills or fire. Foam discharge is covered in the aqueous film forming foam (AFFF) NOD report. The secondary services provided by wet firemain systems are washdown countermeasures, cooling water for auxiliary machinery, eductors, ship stabilization and ballast tank filling, and flushing for urinals, commodes and pulpers.

There are approximately 247 vessels of the Armed Forces within the RAA ports and harbors with firemain systems (Table F-11). There are two types of firemain systems: (1) wet firemain systems that are continuously pressurized so that the system will provide water immediately upon demand and (2) dry firemain systems that are not charged with water and, as a result, do not supply water upon demand. Most vessels in the Navy's surface fleet with firemain systems operate wet firemains while most vessels in the military sealift command (MSC) operate dry firemains (note that this is not consistent with data shown for the RAAs). All U.S. Coast Guard (USCG) and Army vessels operate dry firemains. Submarines use dry firemain systems. Boats and craft are not equipped with firemain systems and generally use portable fire pumps or fire extinguishers for fire fighting. Discharge from dry firemains is about 0.1% of the rate from wet firemain systems.

Table F-11. Number of Vessels of the Armed Forces With Firemain Discharge in Each Representative Action Area (RAA)

RAA	Service	Wet Firemain	Dry Firemain
Miami	Coast Guard	--	3
Norfolk	Army	--	16
	Coast Guard	--	15
	MSC	7	--
	Navy	43	24
Pearl Harbor	MSC	2	--
	Navy	11	17
Puget Sound/Seattle	Army	--	2
	Coast Guard	--	10
	MSC	1	--
	Navy	4	14
San Diego	Coast Guard	--	5
	MSC	4	--
	Navy	39	17
San Francisco	Army	--	8
	Coast Guard	--	2
	MSC	2	--
St. Louis	Coast Guard	--	1
	Total	**113**	**134**
	Grand Total	**247**	

Seawater from the firemain is released to the environment as an incidental discharge during:

- Test and maintenance;
- Training;

- Cooling of auxiliary machinery and equipment, for which the firemain is the normal cooling supply (e.g., central refrigeration plants, steering gear coolers, and the Close In Weapon System);
- Bypass flow overboard from the pump outlet, to prevent overheating of fire pumps when system demands are low; and
- Anchor chain washdown.

Pollutants in the discharge include metals and plasticizers that become dissolved in the seawater while inside the firemain system. The primary pollutants of concern in firemain discharge (i.e., those that exceed the most stringent water quality criteria) are bis(2-ethylhexyl)phthalate, copper, iron, and nickel. Concentrations of these pollutants reported in the NOD Report (EPA, 1999d) are presented in Table F-12.

Table F-12. Concentrations of Pollutants of Concern in Firemain Discharge

Pollutant	Log-normal mean discharge concentration (µg/L)
Bis(2-ethylhexyl)phthalate	22.04
Total Copper	45.59
Dissolved Copper	16.46
Total Iron	21.8
Total Nickel	15.2
Dissolved Nickel	13.8

The EPA and DoD determined the vessel populations by using the Armed Forces Vessel Database which tracks all vessels of the Armed Forces regulated by UNDS. To calculate the total pollutant specific load for each of the seven RAA loading scenarios, the EPA and DoD summed the results of the pollutant loads multiplied by the number of vessels. Pollutant loads for each RAA were then used as inputs to the flushing-based estuarine harbor screening model and river harbor dilution model to estimate surface water concentrations in ports and harbors. The maximum modeled surface water concentration for each pollutant of concern was selected as the harbor concentration to represent surface water concentrations in the action area resulting from firemain discharge (see Section F.3).

F.3.6. Sonar Dome Discharge Constituent Load Estimates

Sonar domes are located on the hulls of submarines and surface ships. Their purpose is to house electronic equipment used for detection, navigation, and ranging. There are approximately 102 vessels of the Armed Forces within the RAA ports and harbors that have sonar domes (Table F-13), 61 of which are rubberized. Sonar domes on Navy surface ships are made of rubber. On submarines, they are made of steel or glass-reinforced plastic (GRP) with a 1/2-inch rubber boot covering the exterior. Military Sealift Command (MSC) T-AGS Class ships have sonar domes made of GRP.

Table F-13. Number of Vessels of the Armed Forces With Sonar Domes in Each Representative Action Area (RAA)

RAA	Service	GRP or Steel Sonar Domes	Rubber/TBT Sonar Domes	Total
Norfolk	MSC	7	--	7
	Navy	7	26	33
Pearl Harbor	Navy	13	11	24
Puget Sound/Seattle	Navy	10	2	12
San Diego	Navy	4	22	26
Total		**41**	**61**	**102**

Pollutants can either leach from the exterior of sonar domes or be discharged with internal sonar dome fluid during maintenance. Vessels with rubberized sonar dome windows are not coated with antifouling paint. These sonar domes are impregnated with TBT, which is the only pollutant of concern for sonar dome external leaching. Other sonar domes are coated with antifouling paint, and release of pollutants is included in estimates for hull leachate discharge. For rubberized sonar domes, it is assumed that TBT is released from rubberized sonar domes at an average rate of 0.27 g/vessel/day (EPA, 1999e). This release rate is based on samples from three Navy surface ships (DDG 53 USS John Paul Jones, CG 59 USS Princeton, and DD 976 USS Merrill).

Prior to 1985, all sonar domes contained tributyltin (TBT) antifoulant on the interior and exterior, to prevent or minimize marine growth. There are only 12 vessels remaining throughout the entire active fleet that have internal TBT coatings, 10 of which occur in the RAAs. Other pollutants of concern in internal sonar dome discharge (i.e., those exceeding the most stringent federal and state water quality criteria) include copper, nickel, tin, zinc, chemical oxygen demand (COD), and total organic carbon (TOC). Concentrations of these pollutants reported in the NOD Report (EPA, 1999e) are presented in Table F-14.

Table F-14. Concentrations of Pollutants of Concern in Interior Sonar Dome Discharge

Pollutant	Mean Concentration
TBT	74 µg/L
Copper	303 µg/L
Nickel	145 µg/L
Tin	194 µg/L
Zinc	1577 µg/L
COD	123 mg/L
TOC	5 mg/L

The EPA and DoD determined the vessel populations by using the Armed Forces Vessel Database which tracks all vessels of the Armed Forces regulated by UNDS. To calculate the total pollutant specific load for each of the seven RAA loading scenarios, the EPA and DoD summed the results of the pollutant loads multiplied by the number of vessels. To estimate exterior TBT leaching from sonar domes, the average release rate of 0.27 g/vessel/day was multiplied by the total number of vessels with rubberized sonar domes. To estimate the pollutant loads from interior sonar dome discharge, concentrations of pollutants in the discharge were multiplied by the number of discharge events each year and the total discharge volume. Pollutant loads for each RAA were then used as inputs to the flushing-based estuarine harbor screening model and river harbor dilution model to estimate surface water concentrations in ports and harbors. The maximum modeled surface water concentration for each pollutant of concern was selected as the harbor concentration to represent surface water concentrations in the action area resulting from sonar dome discharge and external TBT leachate.

F.3.7. Deck Runoff Constituent Load Estimates

All vessels of the Armed Forces generate deck runoff. Deck runoff occurs as the result of weather events and deck washdowns. The amount of in port deck runoff that occurs in ports and harbors depends on:

- The number of vessels in port.
- Weather deck area.
- The number of days a vessel spends in port.
- Annual precipitation.

Table F-15 presents the estimated amount of deck runoff in each of the RAAs that occurs from vessels of the Armed Forces each year.

Table F-15. Amount of Deck Runoff from Vessels of the Armed Forces in Each Representative Action Area (RAA)

RAA	Number of Vessels of the Armed Forces	Weather Deck Area (sq. feet)	Average Annual Precipitation (in/yr)[a]	Total Annual Runoff (gallons per year)
Norfolk	975	2,134,022	45-47	90,823,351.66
Miami	36	10,625	62-67	967,249.51
San Diego	791	1,252,622	10-12	13,099,842.18
San Francisco	98	89,511	20-41	2,193,509.63
Puget Sound/Seattle	337	781,410	20-44	19,924,987.45
Pearl Harbor	217	510,144	17	6,505,653.20
St. Louis	21	4,790	41	75,789.52

[a]Range presented for those RAAs with multiple ports

Pollutants of concern (i.e., those that are likely to exceed the most stringent federal and state water quality criteria) according to the NOD Report (EPA, 1999f) include cadmium, chromium, copper, lead, nickel, silver, zinc, oil and grease, and phenols. However, the data available are for concentrations of pollutants in catapult trough drains prior to processing through an oil-water separator and do not represent concentrations in the majority of deck runoff. Catapult trough drains are only found on aircraft carriers and receive approximately one third of the total deck runoff. Further, concentrations of copper, silver, and zinc were below analytic method detection limits. Because the available data are not considered to be representative of pollutant concentrations in deck runoff, and because oil and grease is one of the pollutants of greatest concern, it is assumed that oil and grease concentrations will be kept at or below 15 ppm, which is the maximum concentration allowed by UNDS in any discharge. Other pollutants were not modeled. Oil and grease loads for each RAA were then used as inputs to the flushing-based estuarine harbor screening model and river harbor dilution model to estimate surface water concentrations in ports and harbors. The maximum modeled oil and grease surface water concentration was selected to represent surface water concentrations in the action area resulting from deck runoff.

F.3.8. Underwater Ship Husbandry Constituent Load Estimates

Underwater ship husbandry includes the following operations:

- hull cleaning,
- fiberglass repair,
- welding,
- sonar dome repair,
- non-destructive test/inspection,
- masker belt repairs,
- paint operations, and
- SEAWOLF propulsor layup.

These operations are typically performed pierside for vessels greater than 40 feet in length. It is assumed that vessels that are 40 feet in length or smaller are removed from the water for inspection, repair and cleaning.

The predominant discharge from underwater ship husbandry is from in-water hull cleaning. The discharge rate based on Submerged Cleaning and Maintenance Platform (SCAMP) intake is 13,500 gal/min (51,100 L/min) at a cleaning rate of 20.8 m^2/min. Cleaning is performed to remove hull-fouling organisms that can reduce operational efficiency and can result in the release of antifouling coatings, cleaners, and non-indigenous hull-fouling organisms. The introduction of non-indigenous aquatic species (NAS) is assessed qualitatively in Section 5.1.1 and is not modeled for this BE. The primary pollutants of concern in-water hull cleaning (i.e., those that exceed the most stringent water quality criteria) are copper, zinc, and chlorine produced oxidants. Concentrations of these pollutants reported in the NOD Report (EPA, 1999g) are presented in Table F-16. The amount of copper or zinc release released per unit area cleaned is calculated as: (concentration in SCAMP effluent) * (flow rate from SCAMP impellers / cleaning rate in m^2/min).

Table F-16. Concentrations of Pollutants of Concern Released During In-Water Hull Cleaning

Pollutant	Concentrations in SCAMP Effluent (µg/L)	NOD Report Flow-Weighted Average Concentration (µg/L)	Release Rate (g/m^2 area cleaned)
Total Copper	1,565 - 2,619	1950	4.8
Dissolved Copper[a]	66 - 146	107	0.26
Zinc[b]	626 - 1,048	780	1.92
	Concentrations in SEAWOLF Propulsor Layup Effluent (µg/L)	**NOD Report Concentration (µg/L)**	**Release Rate (g/event)**
CPO	0-40	40	3.2

[a] Copper passing through a 45 µm filter
[b] Not measured; based on ratio of 2.5 Cu to Zn in paint

Assumptions for calculating mass loadings of pollutants during in-water hull cleaning include the following:

- The life cycle of a hull coating is 12 years.
- Vessels between 40 to 79 feet are periodically cleaned in water as follows:
  - No cleanings first three years after painting.
  - Vessels are cleaned once per year thereafter until paint is 12 years old, for a total of 9 in-water cleanings over 12 years (0.75 cleanings per year), or 1 cleaning every 1.33 years.
  - These in-water cleanings will be assumed full cleanings that include all the wetted hull area.
- Commissioned vessels ≥ 79 feet are cleaned in the water periodically as follows:
  - There are three types of in water cleanings: full, interim, and partial. Full cleaning includes all the wetted hull area, appendages, and openings. Interim cleaning, occurring between full cleanings, is typically a cleaning of the running gear and may include partial cleaning of other ship systems. Partial cleaning includes a limited cleaning of a portion of the hull, appendages, running gear, stabilizers, etc., or hull openings, and will be assumed to cover 30% of the wetted hull area.
  - No full or partial cleanings are performed for the first three years after painting.
  - Full cleanings are performed annually after year 3, for a total of 9 in-water cleanings over 12 years (0.75 cleanings per year), or 1 cleaning every 1.33 years. The final cleaning before repainting occurs in dry dock.
  - Interim cleanings are performed every 3 months for a total of 24 interim cleanings over 12 years.
  - Partial cleanings are performed every 6 months for a total of 13.

- Non-commissioned Navy vessels over 79 feet, such as barges, are not cleaned very often and only have a full cleaning performed approximately once every 5 years for barges.
- Inactive vessels are not cleaned.

The EPA and DoD determined the vessel populations by using the Armed Forces Vessel Database which tracks all vessels of the Armed Forces regulated by UNDS. To calculate the total pollutant specific load for each of the seven RAA loading scenarios, the EPA and DoD summed the results of the pollutant loads multiplied by the number of vessels. Pollutant loads for each RAA were then used as inputs to the flushing-based estuarine harbor screening model and river harbor dilution model to estimate surface water concentrations in ports and harbors. The maximum modeled surface water concentration for each pollutant of concern was selected as the harbor concentration to represent surface water concentrations in the action area resulting from in-water hull cleaning (see Section F.3).

F.4. SELECTION OF HYPOTHETICAL HARBOR CONCENTRATIONS FOR ASSESSMENT OF RISK TO FEDERALLY LISTED THREATENED AND ENDANGERED SPECIES

Modeled concentrations of pollutants resulting from each of the discharges were added to estimate total concentrations in each of the RAAs from all Batch Two discharges from vessels of the Armed Forces. These total concentrations were used as the estimated exposure concentrations. The lowest of the modeled harbor concentrations (minimum), highest of the modeled harbor concentrations (maximum), and mean of the harbor concentrations was determined for each pollutant across the RAAs. The maximum concentration for each pollutant was selected for the exposure concentration for the risk assessment. This allowed a conservative assessment that ensured potential risks to listed species would not be overlooked. Minimum, mean and maximum pollutant concentrations are presented in Table F-17.

Table F-17. Modeled Harbor Concentrations of Each of Pollutant of Concern in UNDS Batch Two Discharges

Class	Pollutant	Range of Estimated Estuary Harbor Concentrations (µg/L)	Estimated River Harbor Concentration (µg/L)
Metals	**Cadmium**	7.40E-08 - 7.80E-06	3.2E-09
	Chromium	3.60E-06 - 0.00039	1.60E-07
	Total Copper	0.0016 - 0.3	0.000067
	Dissolved Copper	**Could not calculate because only measured in some discharges; total copper concentrations assumed to be dissolved**	
	Iron	0.000023 - 0.038	2.3E-06

Table F-17. Modeled Harbor Concentrations of Each of Pollutant of Concern in UNDS Batch Two Discharges (Continued)

Class	Pollutant	Range of Estimated Estuary Harbor Concentrations (µg/L)	Estimated River Harbor Concentration (µg/L)
	Lead	8.30E-06 - 0.00082	3.00E-07
	Mercury	5.7E-10 - 4.6E-07	9.3E-12
	Nickel	5.70E-06 - 0.0097	1.00E-07
	Silver	1.4E-09 - 2.80E-06	0
	Total Zinc	0.004 - 0.41	0.000027
	Dissolved Zinc	Could not calculate because only measured in some discharges; total zinc concentrations assumed to be dissolved	
Petroleum Hydrocarbons	Oil and Grease	0.00073 - 0.074	0.000028
	TPH	3.70E-08 - 1.90E-06	2.7E-09
Toxics and Non-Conventional Pollutants with Toxic Effects	Bis (2-ethylhexyl) phthalate	1.8E-07 - 0.014	2.8E-08
	Tributyltin (TBT)	0 - 0.00021	0
	Chlorine Produced Oxidants	0 - 0.0037	0
Nutrients/ Water Quality	Nitrate/Nitrite	2.4E-06 - 0.0011	4.80E-08
	Total Kjeldahl Nitrogen	0.000035 - 0.049	2.7E-07
	Total Nitrogen	0.000037 - 0.05	3.2E-07
	Ammonia as Nitrogen	0.000019 - 0.036	1.6E-08
	Total Phosphorus	1.3E-05 - 0.0025	3.2E-07
	Total Organic Carbon	0 - 0.0039	0
	Total Suspended Solids	0.00014 - 0.28	0
	Biological Oxygen Demand	0.000096 - 0.19	0
	Chemical Oxygen Demand	0.00026 - 0.28	0

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