Endangered Species Act Section 7 Consultation

Biological and Conference Opinion on the

U.S. Environmental Protection Agency's

Proposed Vessel General Permit and Small Vessel General Permit

National Marine Fisheries Service

Office of Protected Resources

Silver Spring, MD 20910

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National Marine Fisheries Service
Endangered Species Act Section 7 Consultation
Biological Opinion and Conference Opinion

Agency:	United States Environmental Protection Agency
Activities Considered:	Proposed issuance and implementation of the Vessel General Permit and the Small Vessel General Permit, which would authorize discharges incidental to the normal operation of vessels in waters of the United States from 2013 through 2018
Consultation Conducted by:	Endangered Species Act Interagency Cooperation Division of the Office of Protected Resources, National Marine Fisheries Service
Approved by:	
Date:	11/28/12

Section 7(a)(2) of the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. 1536(a)(2)) requires Federal agencies to insure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. When a Federal agency's action "may affect" a protected species, that agency is required to consult formally with the National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (FWS; together, "the Services"), depending upon the endangered species, threatened species, or designated critical habitat that may be affected by the action (50 CFR 402.14(a)). Federal agencies are exempt from this general requirement if they have concluded that an action "may affect, but is not likely to adversely affect" endangered species, threatened species, or designated critical habitat, and if NMFS or the FWS concur with that conclusion (50 CFR 402.14(b)).

The U.S. Environmental Protection Agency (EPA) consulted with the Services on EPA's proposal to authorize discharges incidental to the normal operation of vessels and small vessels into waters of the United States from 2013 through 2018 under the National Pollutant Discharge Elimination System permit program, as authorized by the Clean Water Act. This document represents NMFS' Biological and Conference Opinion (Opinion) on EPA's issuance and implementation of its Vessel General Permit (VGP) and Small Vessel General Permit (sVGP; together the VGPs) and their effects on ESA-listed and proposed species and designated critical habitat. This Opinion is based on our review of the EPA's draft VGPs, EPA's Biological

Evaluation (BE), EPA's fact sheets, reviews of the effectiveness of the National Pollution Discharge Elimination System (NPDES) program and compliance for existing general permits, Section 7 consultations on Clean Water Act actions, species status reviews, listing documents, recovery plans, reports on the status and trends of water quality, past and current research and population dynamics modeling, published and unpublished scientific information, and other sources of information gathered and evaluated during our consultation on the proposed VGPs. This Opinion has been prepared in accordance with section 7 of the ESA, associated implementing regulations, and agency policy and guidance (50 CFR 402; U.S. Fish and Wildlife Service and NMFS 1998).

Consultation History

On March 30, 2005, the U.S. District Court for the Northern District of California (in *Northwest Environmental Advocates et al. v. EPA*) ruled that the EPA regulation excluding discharges incidental to the normal operation of a vessel from NPDES permitting exceeded the Agency's authority under the Clean Water Act (CWA).

On July 23, 2008, the Ninth Circuit upheld the District Court's decision after EPA appealed.

On July 31, 2008, Public Law 110-299 was signed into law and provided a two-year moratorium for non-recreational vessels less than 79 feet in length and all commercial fishing vessels regardless of length, from the requirements of NPDES (with the exception of ballast water). This law also directed the EPA to study the impacts of discharges incidental to the normal operation of those vessels and evaluate potential impacts. Because of this, NMFS did not further pursue EPA's consultation request.

On July 30, 2010, the Congressional moratorium initiated by Public Law 110-299 was extended by Public Law 111-215 until December 18, 2013.

In August 2010, EPA published its final "Report to Congress: Study of Discharges Incidental to Normal Operation of Commercial Fishing Vessels and Other Non-Recreational Vessels Less than 79 Feet." The report noted that many of the pollutants present in the vessel discharges were at end-of-pipe concentrations that exceeded an NRWQC; there is the potential for these discharges to contribute a water quality impact on a more localized scale. The study results indicate that total arsenic and dissolved copper are the most significant water quality concerns for the study vessels as a whole, as they are more likely than other pollutants to contribute to exceedances of water quality criteria.

In 2011, at the request of EPA the National Academies of Sciences published a report on "Assessing the Relationship Between Propagule Pressure and Invasion Risk in Ballast Water." That report noted that data "are not sufficient in present form to characterize a biologically meaningful relationship, much less estimate the associated uncertainty, to be able to identify with confidence the invasion probabilities associated with particular discharge standards."

In a March 8, 2011, settlement with environmental groups, EPA agreed to include in the next VGP numeric effluent limits for discharges of ballast water expressed as organisms per unit of

ballast water volume. They also agreed to issue the final VGP by November 30, 2012, one year before the expiration of the current permit.

EPA published a draft VGP on December 8, 2011.

On November 29, 2011, EPA requested informal Section 7 consultation on its proposed 2013 VGPs.

On July 3, 2012, EPA's Office of Water requested formal Section 7 consultation on the draft VGPs. Their initiation package included the draft permits, a Biological Evaluation (BE), fact sheets, and responses to the Service's comments during informal consultation.

As described in a joint letter sent by the Services on August 6, 2012, NMFS initiated formal consultation effective July 3, 2012. NMFS also sent its analysis plan, including the Services' joint "Approach to the Assessment," which is included in this Opinion.

BIOLOGICAL OPINION

Description of the Proposed Action

Under the Vessel General Permit (VGP) and the Small Vessel General Permit (sVGP, together the VGPs), EPA proposes to authorize discharges incidental to the normal operation of all non-recreational and non-military vessels into waters of the U.S. (defined below in our description of the action area). The EPA expects that most vessels seeking coverage under the VGP will be greater than 79 feet in length, whereas vessels less than 79 feet may be eligible for coverage under the sVGP, which has less stringent restrictions and fewer reporting requirements. The EPA estimates that approximately 72,000 vessels may be eligible for coverage under the VGP, and an additional 138,000 vessels under the sVGP (http://www.epa.gov/npdes/vessels).

The statutory authority for the proposed action is the National Pollution Discharge Elimination System (NPDES) of the Clean Water Act (33 USC §§ 1342 et seq.; CWA). The purpose of the proposed general permits is to satisfy the goals and policies of the CWA (33 USC §§1251). Requiring a CWA permit for these discharges provides EPA with the authority to enforce CWA requirements and provides the ability of citizens to sue for permit violations.

For both permits, EPA includes the following clause for modification of the permits and reinitiation:

Permit modification or revocation will be conducted according to 40 CFR §§ 122.62, 122.63, 122.64, and 124.5. This permit is subject to modification in accordance with 40 CFR §§ 124.5 and 122.62. Grounds for such modification include receipt of new information that was not available at the time of permit issuance (other than revised regulations, guidance, or test

methods) and would have justified the application of different permit conditions at the time of permit issuance. This information would also allow EPA to determine whether to request reinitiation of formal ESA Section 7 consultation as provided in 50 CFR 402.16 and described in the VGPs Fact Sheets.

For the VGP, with respect to ballast water discharges, new information that will be considered in determining whether to modify this permit includes, but is not limited to, data or information from permittees, the general public, states, academia, scientific or technical articles or studies, and results of monitoring conducted under this permit indicating that:

- Treatment technology has improved such that these improved technologies would have justified the application of significantly more stringent effluent limitations or other permit conditions had they been known at the time of permit issuance;
- Treatment technologies known of at the time of permit issuance perform better than understood at the time of permit issuance such that this improved performance would have justified the application of significantly more stringent effluent limitations or other permit conditions had this been understood at the time of permit issuance;
- Scientific understanding of pollutant effects or of invasion biology has evolved such that this new information would have justified the application of significantly more stringent effluent limitations or other permit conditions had this been understood at the time of permit issuance; or
- The cumulative effects of any discharge authorized by the VGP on the environment are unacceptable.

Vessel General Permit (VGP)

The proposed VGP authorizes discharges incidental to the normal operation of a vessel into waters of the U.S. The following narrative describes the discharges that will be authorized or prohibited should the EPA issue the VGP. It also describes the activities that EPA will require of vessel owners/operators for vessels covered under the VGP, including reporting and compliance requirements.

Unless expressly prohibited, all discharges of pollutants into waters of the U.S. incidental to the normal operation of all eligible vessels are to be authorized by the proposed VGP. In this Opinion, we assess the effects of all authorized discharges, which EPA has grouped into the following discharge types:

- Deck Washdown and Runoff, and Above Water Line Hull Cleaning
- Bilgewater and Oily Water Separator Effluent
- Ballast Water
- Anti-fouling Hull Coatings/Hull Coating Leachate
- Aqueous Film Forming Foam (AFFF)
- Boiler/Economizer Blowdown
- Cathodic Protection
- Chain Locker Effluent
- Controllable Pitch Propeller and Thruster Hydraulic Fluid and other Oil Sea Interfaces

- Distillation and Reverse Osmosis Brine
- Elevator Pit Effluent
- Firemain Systems
- Freshwater Layup
- Gas Turbine Washwater
- Graywater
- Motor Gasoline and Compensating Discharge
- Non-Oily Machinery Wastewater
- Refrigeration and Air Condensate Discharge
- Seawater Cooling Overboard Discharge
- Seawater Piping Biofouling Prevention
- Boat Engine Wet Exhaust
- Sonar Dome Discharge.
- Underwater Ship Husbandry
- Welldeck Discharges
- Graywater Mixed with Sewage from Vessels
- Exhaust Gas Scrubber Washwater Discharge
- Fish Hold Effluent

The VGP also contains requirements to address hull foulding, which can result in the discharge of aquatic nuisance species into waters of the U.S. As vessels move from one area to another, they often carry fouling organisms on their hulls or other exposed surfaces. These organisms may release themselves or gametes into ports, harbors, or waterways. Hull fouling is considered to be one of the largest sources of aquatic nuisance species.

Discharges not authorized by the VGP include: sewage, used or spent oil, garbage or trash, photo processing effluent, dry cleaning effluent, medical waste, noxious liquid substance residues, tetrachloroethylene (perchloroethylene) and trichloroethylene degreasers, discharges from vessels not operated as a means of transportation, and discharges covered by other permits.

Discharges to be Authorized by the VGP

For vessels covered under the VGP, EPA requires vessel owners and operators to minimize vessel discharges. They define minimize as to reduce and/or eliminate to the extent achievable using control methods, including best management practices, that are technologically available and economically practicable and achievable in light of best marine practices. They require all vessel owner/operators to ensure adequate training for the master, operator, person-in-charge, and crew who manage or affect incidental discharges.

Water Quality-Based Effluent Limits

For vessels covered under the VGP, EPA requires vessel owners/operators to control their discharges as necessary to meet applicable water quality standards in the receiving water body or another water body impacted by a vessel's discharges. The EPA requires corrective actions, as described below, if they or the vessel owner operators become aware that a vessel's discharges

are causing or contributing to an exceedance of applicable water quality standards. They also require that the vessel owners/operators report the exceedence to EPA. Vessel discharges must be consistent with the assumptions and requirements of a specified waste load allocation (WLA) if the owners/operators discharge to an impaired water with an EPA-approved or established total maximum daily load (TMDL) as indicated by EPA or state TMDL authorities. EPA will inform vessel owners/operators if such a WLA exists and if any additional limits or controls are necessary for discharges to be consistent with the assumptions of any available WLA in the TMDL, or whether an individual permit application is necessary. The EPA and/or state TMDL agencies will inform vessel owners/operators if an applicable TMDL exists either individually or categorically for their vessel or vessel class (including disallowing discharges).

Technology-Based Effluent Limits

For vessels covered under the VGP, EPA will require all vessel owners and operators to meet the effluent limits as described below for each authorized discharge.

Material Storage. For vessels covered under the VGP, EPA requires vessel owners and operators to minimize the potential for cargoes or materials to blow overboard or dissolve in sea spray by minimizing exposure time and using covered storage areas. The EPA does not authorize the discharge of harmful quantities of oily water, which might otherwise be inconsistent with requirements found in Section 311 of the Clean Water Act, the Act to Prevent Pollution from Ships, or under the International Convention for the Prevention of Pollution from Ships, 1973 as modified by the protocol of 1978 (MARPOL 73/78). If water draining from storage areas comes in contact with oily materials, the vessel owners and operators must:

- Use dry cleanup methods or absorbents to clean up the wastewater;
- Store the water for onshore disposal; or
- Run water through oily water separator as required by Coast Guard regulations, or other effective method.

Toxic and Hazardous Materials. For vessels covered under the VGP, EPA requires vessel owners and operators to minimize the potential for discharge of toxic and hazardous materials as follows:

- Store toxic and hazardous materials in appropriate sealed containers constructed of a suitable material, labeled, and secured; the containers must not be overfilled; wastes should not be mixed.
- Locate toxic and hazardous materials in protected areas.
- Report any dischanges.

Fuel Spills/Overflows. For vessels covered under the VGP, EPA prohibits the discharge of oil in harmful quantities (40 CFR Part 110). The VGP does not authorize large-scale fuel spills or overflows, which are not incidental to the normal operation of the vessel. They require vessel owners and operators to:

- Conduct fuel operations using control measures and practices designed to minimize spills and overflows and ensure prompt containment and cleanup.

- Conduct fueling in a manner that prevents overfilling and release from the system to the environment.
- For vessels with air vents from fuel tanks, use spill containment or other methods to prevent or contain any fuel or oil spills.
- Owner/operators shall ensure that any crew members responsible for conducting fueling operations are trained in methods to minimize spills caused by human error and/or the improper use of equipment.
- For fueling of auxiliary vessels, (e.g., lifeboats, tenders or rescue boats), vessel owners and operators must:
    - While fueling, examine the surrounding water for the presence of a visible sheen. If a visible sheen is observed, as a result of fueling, it must be cleaned up immediately.
    - Know the capacity of the fuel tanks before fueling in order to prevent unintentionally overfilling the tank.
    - Prevent overfilling and not top off fuel tanks.
    - When possible, fill fuel tanks while boat is on shore or recovered from the water.
    - When possible, fill portable tanks on shore or on the host vessel, not on the auxiliary vessel.
    - Use an oil absorbent material or other appropriate device while fueling the auxiliary vessel to catch drips from the vent overflow and fuel intake.
    - Regularly inspect the fuel and hydraulic systems for any damage or leaks, for instance during fueling, when performing routine maintenance on the auxiliary vessel, and/or during deployments for testing.

<u>Deck Washdown and Runoff/Above the Water Line Hull Cleaning</u>. The proposed permit authorizes the discharge of deck washdown and runoff from deck and bulkhead areas, associated equipment, areas of the hull, and exteriors of the vessel above the water line as a result of precipitation or from washing. Pollutant discharges may include residues and any soaps or detergents used for cleaning.

For vessels covered under the VGP, EPA requires vessel owner/operators to minimize the introduction of on-deck debris, garbage, residue, and spill into deck washdown and runoff discharges. They require vessel owner/operators to broom clean exposed decks or use comparable managmenet measures to remove all existis debris prior to deck washdown. In addition, EPA requires vessel owner/operators to:

- Minimize the presence of floating solids, visible foam, halogenated phenol compounds, and dispersants, or surfactants in deck washdowns.
- Minimize deck washdowns while in port.
- Maintain their topside surface and other areas above water line portions of the vessel 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.
- Minimize residual paint droplets from entering waters subject to this permit whenever they are conducting maintenance painting. Minimization techniques include, but are not

limited to: avoiding paint spraying in windy conditions or avoiding overapplication of paint. Disposal of unused paint into waters of the U.S. is not permitted under the VGP.

- Deck washdowns or above water line hull cleaning that will result in a discharge must be conducted with biodegradable, "non-toxic," and "phosphate free" cleaners and detergents as defined in Appendix A of the VGP. EPA expects that soaps that are "minimally" or non-toxic will contain little to no nonylphenols.
- Where necessary and feasible, machinery on deck must have coamings or drip pans 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 and/or periodically wiped and cleaned.

Bilgewater/Oily Water Separator Effluent. The proposed permit authorizes the discharge of bilgewater and the residues that accumulate therein. These residues may originate from machinery and from deck drainage etc. Pollutant discharges may include seawater, oil, grease, organic compounds, inorganic salts, and metals.

All bilgewater discharges must be in compliance with the regulations in 40 CFR Parts 110 (Discharge of Oil), 116 (Designation of Hazardous Substances), and 117 (Determination of Reportable Quantities for Hazardous Substances) and 33 CFR §151.10 (Control of Oil Discharges). Under the VGP, EPA prohibits:

- Use of dispersants, detergents, emulsifiers, chemicals, or other substances that remove the appearance of a visible sheen in their bilgewater discharges.
- Addition of substances that drain to the bilgewater and are not produced in the normal operation of a vessel.
- Use of oil solidifiers, flocculants, or other required additives, unless used as part of an oil water separation system during processing and provided they do not alter the chemical make-up of the oils being discharged. The EPA requires vessel owners/operators to minimize any discharge of such materials into waters of the U.S.
- Vessel owners/operators must minimize the discharge of bilgewater by minimizing the production of bilgewater, disposing of bilgewater on shore where adequate facilities exist, or discharging into waters not subject to this permit (i.e., more than 3 nautical miles from shore) following regulations under Annex I of the International Convention for the Prevention of Pollution from Ships as implemented by the Act to Prevent Pollution from Ships and U.S. Coast Guard regulations found in 33 CFR part 151.
- For vessels greater than 400 gross tons, the following will be prohibited:
  - Discharge of untreated oily bilgewater.
  - Discharge of treated bilgetwater within 1 nm of shore, if technologically feasible (e.g., holding does not impact safety and stability, does not contaminate other holds or cargo, or does not interfere with essential operations of the vessel). Any discharge that is not technologically feasible to avoid must be documented and reported to EPA as part of the vessel's annual report.
  - Discharge of treated bilgewater into federally protected waters referenced in Appendix G of the VGP, unless the discharge is necessary to maintain the safety and

stability of the ship. Any discharge of bilgewater into these waters must be documented and reported to EPA as part of the vessel's annual report.
    - For vessels that regularly sail outside the territorial sea (at least once per month), if treated bilgewater is discharged into waters subject to this permit, it must be discharged when the vessel is underway (sailing at speeds greater than 6 knots), unless doing so threatens the safety and stability of the ship. EPA notes that vessel operators may also choose to dispose of bilgewater on shore where adequate facilities exist. Any discharge, which is made for safety reasons, must be documented and reported to EPA as part of the vessel's annual report.

For new builds (on or before December 19, 2013), EPA requires electronic submittal of annual oil content monitoring.

Ballast Water. Ballast water is water taken on to vessels to aid in buoyancy and stability and in increasing or decreasing the vessel's draft. According to EPA, cruise ships have a typical ballast capacity of 1,000 cubic meters and cargo ships may contain up to 93,000 cubic meters. Pollutant discharges from ballast water may include contain rust inhibitors, flocculent compounds, epoxy coating materials and metals. Ballast water may also serve as a vector for biological pollutants including aquatic nuisance species (ANS) and pathogens.

For vessels covered under the VGP, EPA requires certain vessel owner/operators to ensure adequate, prompt training for the master, operator, person-in-charge, and crew who manage, take part in, or affect ballast water discharge. All vessel owners/operators are required to maintain a written training plan, including a description of the training, the date of the training, and the names of those that were trained. They are required to maintain a ballast water management plan developed specifically for their vessel that will ensure that those responsible for the plan's implementation understand and follow the vessel's ballast water management strategy (i.e., how the vessel will meet the requirements of the VGP).

The EPA requires owner/operators of vessels with ballast water tanks to:

- Avoid the discharge or uptake of ballast water in Federally Protected Waters (listed in Appendix G of the VGP).
- Minimize or avoid uptake of water in:
    - Critical habitat of ESA-listed species.
    - Areas known to have infestations or populations of harmful organisms and pathogens (e.g., toxic algal blooms).
    - Areas near sewage outfalls.
    - Areas near dredging operations.
    - Areas where tidal flushing is known to be poor or times when a tidal stream is known to be turbid.
    - In darkness, when bottom-dwelling organisms may rise up in the water column.
    - Where propellers may stir up the sediment.
    - Areas with pods of whales, convergence zones, and boundaries of major currents.

- Clean ballast tanks regularly to remove sediments in mid-ocean (when not otherwise prohibited by applicable law) or under controlled arrangements in port, or at dry dock.
- No discharge of sediments from cleaning of ballast tanks is authorized in waters subject to this permit.
- When discharging ballast water in port, if the vessel is equipped with high and low suction within ballast tanks, utilize the high suction for ballast tank discharge to minimize the discharge of entrained sediment. The low suction may be used to strip sediment from tanks when suitable disposal facilities are available.
- Minimize the discharge of ballast water essential for vessel operations while in the waters subject to this permit.

For vessels covered under the VGP, EPA requires vessel owner/operators to meet the following discharge limits:

- For organisms greater than or equal to 50 micrometers in minimum dimension: discharge must include fewer than 10 living organisms per cubic meter of ballast water.
- For organisms less than 50 micrometers and greater than or equal to 10 micrometers: discharge must include fewer than 10 living organisms per milliliter (mL) of ballast water.
- Indicator microorganisms must not exceed:
  - For Toxicogenic *Vibrio cholerae* (serotypes O1 and O139): a concentration of less than 1 colony forming unit (cfu) per 100 mL.
  - For *Escherichia coli*: a concentration of fewer than 250 cfu per 100 mL.
  - For intestinal enterococci: a concentration of fewer than 100 cfu per 100 mL.

The EPA will continue to explore new technologies with industry and states, and when warranted, will make this numeric limit more stringent in the future. Additionally, EPA encourages and anticipates, as part of this process, that States will continue to work with industry to test and provide opportunities for new technologies.

The EPA authorizes four methods for meeting the above ballast water discharge limits: ballast water treatment system; onshore treatment of ballast water; use of public water supply; or no discharge of ballast water.

The EPA has established dates by which vessels, categorized by size and drydock date, are required to use ballast water treatment systems (BWTS) as the best available technology economically achievable. Under the VGP, EPA requires a BWTS which has been shown to be effective by testing in accordance with the EPA-ETV protocol for the verification of ballast water treatment technology conducted by an independent third party laboratory, test facility or test organization. Following installation of a BWTS, EPA requires the master, owner, operator, agent, or person in charge of the vessel to maintain the BWTS in accordance with all manufacturer specifications. Furthermore, EPA requires all treatment to be conducted in accordance with the BWTS manufacturer's instructions. The BWTS must be used prior to any discharge of ballast water to waters of the U.S. Under the proposed VGP, EPA would require:

- Monthly monitoring of equipment performance to assure the system is fully functional.

- Annual calibration of ballast water monitoring equipment.
- Biannual monitoring from all ballast water systems for selected biological indicators (i.e., heterotrophic bacteria, *E. coli*, and *Enterococci.*
- Biannual monitoring of the ballast water discharge itself for biocides and residuals (e.g. chlorine dioxide, chlorine, ozone, peracetic acid, hydrogen peroxide, etc.) to assure compliance with the effluent limitations.
- Annual electronic submittal of above listed testing results to the EPA.

For those vessels whose design and construction safely allows for the transfer of ballast water to shore, EPA requires vessel owners/operators to ensure that all piping and supporting infrastructure up to the last manifold or valve immediately before the dock manifold connection of the receiving facility or similar appurtenance on a reception vessel must be fully free from any leaks or other avenues whereby untreated ballast may be discharged into waters subject to this permit. The EPA notes that transferring ballast water to a treatment barge for eventual treatment and discharge could constitute "on-shore treatment" for purposes of this subpart. The discharge of treated ballast water (transferred from other vessels) from a treatment barge is not eligible for coverage under the VGP as this is a discharge from an industrial operation, not a discharge incidental to the normal operation of a vessel. Instead, these vessels must apply for individual NPDES permit coverage from the appropriate NPDES permitting authority, generally the State in which they are operating.

For vessels using water from a public water supply (PWS) as ballast, EPA requires that vessel owners/operators maintain a record of which PWS they received the water and a receipt, invoice, or other documentation from the PWS indicating that water came from that system. The EPA requires those vessels to use PWS water exclusively for all ballast water to avoid contamination of the ballast water tank. The EPA requires those vessels to have either:

- Previously cleaned the ballast tanks (including removing all residual sediments) and not subsequently introduced ambient water; or
- Never introduced ambient water to those tanks and supply lines.

The EPA requires vessels utilizing water from a PWS as ballast water to certify in their recordkeeping documentation that they have met all the requirements of this section, including maintaining certification by the master or NOI certifier that one of the two above conditions are met regarding contamination. In the event a vessel that normally uses PWS water as ballast is forced for purposes of vessel safety to take on untreated ballast water from a sea, estuary, lake or river source, EPA prohibits the vessel from returning to using PWS water until the tanks and supply lines have been cleaned, including removal of all residual sediments.

For certain vessels that are not required to meet the ballast water management measures (i.e., those that leave the US Exclusive Economic Zone and those engaged in Pacific nearshore voyages), EPA requires the vessel to exchange ballast water as follows:

- The exchange must occur in waters beyond the U.S. EEZ.
- The exchange must occur in an area more than 200 nautical miles from any shore;
- The exchange must be commenced as early in the vessel voyage as possible, as long as

the vessel is more than 200 nm from any shore.

- For vessels engaged in Pacific nearshore voyages:
  - Ballast water must occur in waters more than 50 nautical miles from any shore (US or otherwise), in waters more than 200 meters deep, prior to discharging ballast water into waters subject to this permit.
  - Exchange should occur as far from the shore, major estuary and oceanic river plumes, subsurface physical features (e.g., seamounts), and known fishery habitats as practicable.

For those tanks which are empty or contain unpumpable residual water, EPA requires vessel owners/operators to either seal the tank so that there is no discharge, or uptake and subsequent discharge of ballast water within waters subject to this permit or conduct saltwater flushing of such tanks in an area 50 nm from any shore and in waters at least 200 meters deep prior to the discharge or uptake and subsequent discharge of any ballast water to or from any waters subject to this permit.

Anti-Fouling Hull Coatings/ Hull Coating Leachate. Vessel hull coatings can contain toxics and may contain biocides designed to prohibit the growth of fouling organisms. These coatings discharge pollutants by leaching toxics into the water. Copper is a common biocide found in hull coatings. Discharges of the biocide tributyltin (TBT) and other organotin compounds are not permitted by the proposed permit due to their persistence in the environment and their high toxicity to marine life.

For vessels covered under the VGP, EPA requires vessel owner/operators to give consideration, as appropriate for vessel class and vessel operations, to the use of hull coatings with the lowest effective biocide release rates, rapidly biodegradable components (once separated from the hull surface), or non-biocidal alternatives, such as silicone coatings, at the time of initial application or scheduled reapplication of anti-fouling coatings. EPA requires that all anti-fouling hull coatings subject to registration under FIFRA (see 40 CFR § 152.15) must be registered, sold or distributed, applied, maintained, and removed in a manner consistent with applicable requirements on the coatings' FIFRA label. For anti-fouling hull coatings not subject to FIFRA registration (i.e., not produced for sale and distribution in the United States), EPA requires that hull coatings must not contain any biocides or toxic materials banned for use in the United States (including those on EPA's List of Banned or Severely Restricted Pesticides). This requirement applies to all vessels, including those registered and painted outside the United States.

Some ports and harbors are impaired by copper, a biocide used commonly in anti-foulant paints. A complete list of such waters may be found at www.epa.gov/npdes/vessels. When vessels spend considerable time in these waters (defined as spending more than 30 days per year), or use these waters as their home port (i.e., house boats, ferries or rescue vessels), EPA requires vessel owners/operators to consider using anti-fouling coatings that rely on a rapidly biodegradable biocide or another alternative rather than copper-based coatings. If after consideration of alternative biocides, vessel operators continue to use copper-based antifoulant paints, they must document in their recordkeeping documentation how this decision was reached.

The discharge of Tributyltin (TBT) from any source (whether used as a biocide or not) or any other organotin compound used as a biocide is prohibited by this permit. Therefore, vessel owners/operators covered by this permit have a zero discharge standard for TBT (whether or not used as a biocide) or any other organotin compound used as a biocide. For vessels covered under the VGP, EPA prohibits the use of an antifoulant coating containing TBT or any other organotin compound used as a biocide. If the vessel has previously been covered with a hull coating containing TBT (whether or not used as a biocide) or any other organotin compound used as a biocide, vessels must be effectively overcoated so that no TBT or other organotin leaches from the vessel hull or the TBT or other organotin coating must have been removed from the vessel's hull. When used as a catalyst, an organotin compound other than TBT (e.g., dibutyltin) is not to be present above 2500 mg total tin per kilogram of dry paint. Furthermore, EPA requires that the coating shall not be designed to slough or otherwise peel from the vessel hull. Incidental amounts of coating discharged by abrasion during cleaning or after contact with other hard surfaces (e.g., moorings) are not prohibited by the VGP.

Aqueous Film Forming Foam (AFFF). Fire fighting foams, known as AFFF, contain fluorosurfactants or fluoroproteins. When employed for this purpose, AFFFs are sprayed from hoses and may be discharged into waters of the U.S. Fluorosurfactants compounds are toxic, persistent in the environment and have been detected in the blood and tissues of humans and wildlife. Fluoroproteins are less toxic than fluorosurfactants and are biodegradable.

For vessels covered under the VGP, EPA authorizes the discharge of AFFF for emergency purposes only, when needed to ensure the safety and security of the vessel and crew. They prohibit maintenance and training discharge of fluorinated AFFF for vessels that sail outside of the territorial sea more than once per month (i.e., any such discharges should be collected and stored for onshore disposal or scheduled when the vessel is outside such waters). Vessel owners/operators must minimize discharge volumes associated with regulatory certification and inspection and a substitute foaming agent (i.e., non-fluorinated) must be used if possible within waters of the U.S. For vessels that do not leave the territorial sea more than once per month, maintenance and training discharges of AFFF must be collected and stored for onshore disposal, unless the vessel uses a non-fluorinated or alternative foaming agent. For all vessels, maintenance and training discharges are not allowed in port. For all vessels, AFFF discharges may not occur in or within 1 nm of Federally protected waters (Appendix G of the VGP) unless they are discharged:

- For emergency purposes;
- By rescue vessels such as fireboats for firefighting purposes; or
- By vessels owned or under contract to do business exclusively in or within 1 nm of those protected areas by the United States government or state or local governments.

If AFFF discharges occur in waters referenced in Appendix G for emergency purposes, a written explanation must be kept in the ship's log.

Boiler/Economizer Blowdown. For vessels with steam generators or steam propulsion, boiler blowdown is employed to control corrosion and to remove sludge and residue by discharging boiler effluent into waters of the U.S. In addition to sludge and residue, discharges from boilers/economizers may contain chemicals used to reduce impurities or prevent scale formation.

For vessels covered under the VGP, EPA requires vessel owners/operators to minimize the discharge of boiler/economizer blowdown in port if chemicals or other additives are used to reduce impurities or prevent scale formation. They would prohibit the discharge of boiler/economizer in Federally protected waters listed in Appendix G of the VGP except for safety purposes. For vessels greater than 400 gross tons, which leave the territorial sea at least once per week, the EPA would prohibit the discharge of boiler/economizer blowdown in U.S. waters, unless:

- The vessel remains within waters subject to this permit for a longer period than the necessary duration between blowdown cycles;
- The vessel needs to conduct blowdown immediately before entering drydock; or
- For safety purposes.

Cathodic Protection. In order to prevent metal corrosion, two types of cathodic protection are employed: sacrificial anodes and Impressed Current Cathodic Protection (ICCP). Sacrificial anodes of zinc or aluminum create a flow of electrons to cathodic metals and prevent them from corroding. Residual zinc or aluminum is then released to the aquatic environment. ICCP employs a direct electrical current into the vessel hull to prevent corrosion.

For vessels covered under the VGP, EPA requires vessel owner/operators to maintain cathodic protection to prevent hull corrosion. They require vessel owner/operators to minimize the flaking of large, corroded portions of zinc, magnesium, and aluminum anodes. They require vessel owners to avoid using sacrificial anodes more than necessary to adequately prevent corrosion of the vessel's hull, sea chest, rudder, and other exposed areas of the vessel. Vessel operators are required to appropriately clean and/or replace these anodes during periods of maintenance (such as drydocking), so that release of these metals to waters is minimized. If vessel operators use sacrificial electrodes, EPA requires that they select electrode devices with metals that are less toxic to the extent technologically feasible and economically practicable and achievable (i.e., magnesium is less toxic than aluminum and aluminum is less toxic than zinc). If a vessel owner/operator selects aluminum, they must document why they made this selection, and why use of magnesium is not appropriate. Likewise, if a vessel owner/operator selects zinc, they must document why they did not select magnesium or aluminum. If vessel owners/operators use Impressed Current Cathodic Protection (which is recommended by EPA), they are required to maintain dielectric shields to prevent flaking.

Chain Locker Effluent. Chain lockers are the storage areas that contain anchor chains. Water can collect in chain lockers and this effluent may be discharged. Chain locker discharges may contain toxics such as metals, paint and grease. These discharges may also serve as a vector for biological pollutants including aquatic nuisance species and pathogens.

For vessels covered under the VGP, EPA requires vessel owners/operators to carefully and thoroughly wash down (i.e., more than a cursory rinse) the anchor chain, as it is being hauled out of the water to remove sediment and marine organisms. They require vessel owners/operators to clean chain lockers thoroughly during dry-docking to eliminate accumulated sediments and any potential accompanying pollutants. For vessels that regularly sail outside waters subject to this permit (at least once per month), vessel owners/operators are required to periodically clean, rinse, and/or pump out the space beneath the chain locker prior to entering waters subject to this permit (preferably mid-ocean) if the anchor has been lowered into any nearshore waters and if technically feasible. For vessels that leave waters subject to this permit at least once per month, EPA prohibits vessel owners/operators from rinsing or pumping out chain lockers in waters of the U.S., unless not emptying them would compromise safety, in which case the safety claim must be documented in the vessel's recordkeeping documentation.

Controllable Pitch Propeller and Thruster Hydraulic Fluid and Other Oil-to-Sea Interfaces
Oil to sea interfaces include any part of a vessel where seals or surfaces may discharge oil into waters of the U.S. These discharges include lubrication discharges from paddle wheel propulsion, stern tubes, thruster bearings, stabilizers, rudder bearings, azimuth thrusters, and propulsion pod lubrication, and wire rope and mechanical equipment subject to immersion in water. Examples include machinery that uses hydraulic oils or has parts that are lubricated with greases or oils. An additional example is the stern tube. Riggings and cables may also have surfaces that are lubricated with oils or greases and may cause discharges of these substances into waters of the U.S. As noted by EPA, these discharges can be substantial.

The EPA does not authorize the discharge of any lubricant in a quantity that may be harmful as defined in 40 CFR Part 110. For vessels covered under the VGP, EPA requires vessel owners/operators to maintain protective seals on oil-to-sea interfaces in good operating order to minimize the leaking of hydraulic oil or other oils. Vessel owners/operators are prohibited from discharging oil in quantities that may be harmful as defined in 40 CFR Part 110 from any oil-to-sea interface. Vessel owners/operators are required to minimize maintenance activities on stern tube seals when a vessel is outside of drydock. If maintenance or emergency repair must occur on stern tubes or other oil-to-sea interfaces which have a potential to release oil in quantities that may be harmful as defined in 40 CFR Part 110, appropriate spill response equipment (e.g., oil booms) must be used to contain any oil leakage. The EPA requires operators of the vessel to have ready access to spill response resources to clean up any oil spills. After applying lubrication to wire rope and mechanical equipment subject to immersion, wire ropes, and other equipment must be thoroughly wiped down to remove excess lubricant. For vessels constructed on or after December 19, 2013, EPA requires the use of an environmentally acceptable lubricant in all oil-to-sea interfaces. "Environmentally acceptable lubricants" means lubricants that are "biodegradable" and "non-toxic" and are not "bioaccumulative" as defined in Appendix A of the VGP. For vessels built before December 19, 2013, unless technically infeasible, EPA requires owners/operators to use an environmentally acceptable lubricant in all oil to sea interfaces or document and report why they are unable to do so and which lubricant was used in their annual report (submitted electronically to EPA).

Distillation and Reverse Osmosis Brine. Discharges of salt brine can result from seawater distillation and reverse osmosis activities used to generate freshwater from seawater. In addition to having high salinity, these discharges may cause thermal pollution as well as contain residual chemicals used in the various processes.

For vessels covered under the VGP, EPA prohibits brine from the distillation system and reverse osmosis reject water from containing or coming in contact with machinery or industrial equipment (other than that necessary for the production of potable water), toxic or hazardous materials, or wastes.

Elevator Pit Effluent. Liquids containing oils, hydraulic fluid and other materials may collect in elevator shaft pits aboard vessels. These liquids may enter waters of the U.S. from simple drainage off the vessel, transference to bilge followed by discharge, or by other means.

For vessels covered under the VGP, EPA prohibits discharges of untreated elevator pit effluent within waters of the U.S., except in cases of emergency. They authorize elevator pit effluent to be discharged if it is managed with the vessel's bilgewater and meets bilgewater discharge requirements. Otherwise, they require that it be treated with an oily-water separator and discharged with an oil content below 15 ppm as measured by EPA Method 1664 or other appropriate method for determination of oil content as accepted by the IMO (e.g., ISO Method 9377) or U.S. Coast Guard. Vessel owners/operators are required to document emergency discharges.

Firemain Systems. Firemain systems take up ambient water to supply fire hoses, sprinklers and other fire suppression systems. These systems may also be used for other purposed such as washing the vessel and equipment, machinery cooling, ballast tank filling etc).

For vessels covered under the VGP, EPA authorizes discharges from firemain systems for emergency purposes to ensure the safety and security of the vessel and her crew and other emergency situations. They authorize testing and inspections of the firemain systems in order to assure its operability in an emergency. They also authorize in port discharges from firemain systems for certification, maintenance, and training requirements if the intake comes directly from the surrounding waters or potable water supplies and there are no additions (e.g., AFFF) to the discharge. The EPA authorizes the use of firemain systems for deck washdown or other secondary uses if the intake comes directly from the surrounding waters or potable water supplies and the discharge meets all relevant effluent limitation associated with that activity. They prohibit the discharge of firemain systems in Federally protected waters listed in Appendix G of the VGP, except in emergency situations or when washing down the anchor chain to comply with anchor wash down requirements.

Freshwater Layup. When steam propulsion systems are shut down, biological growth can occur within the system. Freshwater layup replaces this water and may result in the discharge of metals and other contaminants into waters of the U.S.

For vessels covered under the VGP, EPA requires vessel owners/operators to minimize the amount of disinfection or biocidal agents used in freshwater layup (i.e., the minimum required to prevent aquatic growth).

Gas Turbine Washwater. Gas turbines must be cleaned to remove materials such as salts and spent lubricants. These materials may be discharged into waters of the U.S. and may also include cleaning agents.

For vessels covered under the VGP, EPA prohibits the direct discharge of gas turbine washwater within waters of the U.S. They require vessel owners/operators to prevent commingling of gas turbine washwater and bilgewater, where feasible. They prohibit the discharge of oils, including oily mixtures, from gas turbine washwater in quantities that may be harmful as determined in accordance with 40 CFR Part 110.

Graywater. Graywater is water that has been used for bathing, in sinks, laundry etc., and may contain pathogens, nutrients, soaps, detergents, and other organic pollutants. Some –usually large– vessels collect and store graywater for later treatment and discharge. Other vessels discharge untreated graywater directly into waters of the U.S.

For vessels covered under the VGP, EPA requires vessel owners/operators to minimize the discharge of graywater while in port. For those vessels that cannot store graywater, the owner or operator and their crews are required to minimize the production of graywater in port. Examples of ways to minimize production of graywater include delaying laundry, scullery activities, and restricting length of showers while in port, and using high efficiency faucets and showerheads. Vessel owner/operators are required to minimize the discharge of graywater when the vessel is not underway. Vessels that have the capacity to store graywater are prohibited from discharging graywater in federally protected waters listed in Appendix G of the VGP. For vessels that cannot store graywater, vessel operators must minimize the production of graywater while in those waters. For vessels greater than 400 gross tons that regularly travel more than 1 nm from shore and have the capacity to store graywater for a sufficient period, treated graywater must be discharged greater than 1 nm from shore while the vessel is underway, unless the vessel meets the following treatment standards and requirements:

- The discharge must satisfy the minimum level of effluent quality specified in 40 CFR § 133.102;
- The geometric mean of the samples from the discharge during any 30-day period may not exceed 20 fecal coliform/100 milliliters (ml) and not more than 10 percent of the samples exceed 40 fecal coliform/100 ml; and
- Concentrations of total residual chlorine may not exceed 10.0 micrograms per liter (µg/l).

For vessels that do not travel more than 1 nm from shore, EPA requires vessel owners/operators to minimize the discharge of graywater and, provided the vessel has available graywater storage capacity, and dispose of graywater onshore, if appropriate facilities are available and such disposal is economically practicable and achievable, unless the vessel meets aboved listed treatment standards and requirements. The vessel owners/operators are required to minimize the

introduction of kitchen oils to the graywater system if graywater will be discharged in waters of the U.S. They are required to remove as much food and oil residue as practicable before rinsing dishes. They are prohibited from adding excess oils used in cooking, including animal fats and vegetable oils, to the graywater system. They are prohibited from discharging galley and scullery oil in quantities that may be harmful as defined in 40 CFR Part 110. Under the VGP, EPA requires vessel owners/operators to use phosphate-free and non-toxic soaps and detergents for any purpose if graywater will be discharged into waters subject to this permit; EPA expects that non-toxic soaps will contain little to no nonylphenols. Soaps and detergents must be free from toxic or bioaccumulative compounds and not lead to extreme shifts in receiving water pH (i.e., fall below 6.0 or rise above 9.0 as a direct result of the discharge).

For vessels underway in a nutrient-impaired water, or a water that is impaired as a result of nutrient enrichment (such as waters listed as impaired for phosphorus, nitrogen, or for hypoxia or anoxia [low dissolved oxygen concentrations]), EPA prohibits discharge of graywater, if the vessel has adequate graywater storage capacity. If the vessel does not have adequate storage capacity, graywater production and discharge must be minimized and conducted while the vessel is underway and in areas with significant circulation and depth, to the extent feasible. For vessels constructed on or after December 19, 2013, EPA requires the vessel owners/operators to take two samples per year, at least 14 days apart, and analyze: Biochemical Oxygen Demand (BOD), fecal coliform, suspended solids, pH, and total residual chlorine. They require the sampling and testing to be conducted according to 40 CFR Part 136. They require annual reporting, including:

- Date, exact place, and time of sampling or measurements.
- Individual(s) who performed the sampling or measurements.
- Date(s) analyses were performed.
- Individual(s) who performed the analyses.
- Analytical techniques or methods used and results of such analyses.
- Whether the graywater effluent is treated or untreated.
- Whether the effluent is graywater alone or if it is mixed with another effluent type (e.g., graywater mixed with sewage).

<u>Motor Gasoline and Compensating Discharge</u>. Ambient water is taken up onto vessels to compensate for weight as motor gasoline is used. This water is later discharged into waters of the U.S. and may contain residual oils and other pollutants.

For vessels covered under the VGP, EPA prohibits the discharge of motor gasoline and compensating effluent containing oil in quantities that may be harmful as defined in 40 CFR § 110.3, which includes discharges resulting in a visible sheen, or an oil concentration that exceeds 15 ppm. Determination of oil concentration may be measured by EPA Method 1664 or other appropriate method for determination of oil content as accepted by the IMO (e.g., ISO Method 9377) or U.S. Coast Guard. The EPA authorizes compliance with the 15 ppm oil concentration limitation as established by visual monitoring for an oily sheen. They also require vessel owner/operators to minimize discharge of motor gasoline and compensating discharge in port. If an oily sheen is observed, the vessel operator is required to deploy appropriate oil containment practices. The EPA prohibits motor gasoline and compensating discharges in federally protected

waters (Appendix G, VGP).

Non-Oily Machinery Wastewater. Non-oily machinery wastewater systems contain water that does not come into contact with machinery that uses oil. Examples include distillers, condensate drains, pump drains, potable water tank overflows and leaks from propulsion shaft seals. These wastewater systems may result in discharges of various pollutants into waters of the U.S.

For vessels covered under the VGP, EPA requires non-oily machinery wastewater to be free from oils (in quantities that may be harmful pursuant to 40 CFR Part 110) and any additives that are toxic or bioaccumulative in nature, if discharged directly overboard. They permit non-oily machinery wastewater to be drained to the bilge.

Refrigeration and Air Condensate Discharge. Condensation from refrigeration or air condition systems may discharge into waters of the U.S. and may contain detergents, metals and other pollutants.

For vessels covered under the VGP, EPA prohibits the discharge of refrigeration and air condensate, if it had come into contact with oily or toxic materials. They permit the discharge of refrigeration and air conditioning condensate that is collected and plumbed for internal recycling (e.g., recycled as "technical water") which has commingled with oily water if the commingled discharge meets the discharge requirements for oily mixtures and oily water separator effluent.

Seawater Cooling Overboard Discharge. Seawater cooling systems circulate ambient water through an enclosed system and may be discharged into waters of the U.S. These discharges may contain hydraulic fluid, lubricating oil and metals and may also be of higher than ambient temperature. These discharges include non-contact engine cooling water; hydraulic system cooling water, refrigeration cooling water.

For vessels covered under the VGP, EPA requires vessel owner/operators to maintain all piping and seawater cooling systems must meet the requirements as described below.

Seawater Piping Biofouling Prevention. Seawater cooling systems may contain anti-fouling compounds. These compounds may be discharged into waters of the U.S.

For vessels covered under the VGP, EPA requires vessel owner/operators to use seawater piping biofouling chemicals subject to FIFRA registration (see 40 CFR § 152.15) in accordance with their FIFRA label. They prohibit the discharge of pesticides or chemicals banned for use in the U.S. They require vessel owner/operators to use the minimum amount of biofouling chemicals needed to keep fouling under control and discharges containing active agents to contain as little chlorine as possible. Vessel owner/operators are required to remove fouling organisms from seawater piping on a regular basis (though regular is not defined) and dispose of removed substances in accordance with local, state, and federal regulations. The EPA prohibits the discharge of removed fouling organisms into waters of the U.S.

Boat Engine Wet Exhaust. Boat engines use ambient water for cooling and noise reduction. This wet engine exhaust is discharged into waters of the U.S. and may contain various pollutants.

For vessels covered under the VGP, EPA requires vessel owner/operators to maintain vessel engines generating wet exhaust, ensuring that they are in good operating order, well tuned, and function according to manufacturer specifications to decrease pollutant contributions to wet exhaust. The EPA recommends that vessel owner/operators use four stroke engines instead of two stroke engines for vessels generating wet exhaust. For vessel owner/operators who use two stroke engines, EPA requires the use of environmentally acceptable lubricants (as defined in Appendix A of the VGP), unless technologically infeasible, in which case the vessel owner/operator must document why they are not using environmentally acceptable lubricants.

Sonar Dome Discharge. Sonar equipment may discharge water that contains metals and antifouling compounds.

For vessels covered under the VGP, EPA prohibits the discharge of water inside the sonar dome for maintenance purposes. They prohibit the vessel owner/operators from using biofouling chemicals that are bioaccumulative for the exterior of sonar domes when non-bioaccumulative alternatives are available.

Underwater Ship Husbandry Discharges. Underwater ship husbandry is the maintenance and repair associated with vessel hulls while in water. This can result in the discharge of various pollutants used in repair and maintenance as well as the discharge of antifouling paints used on the hull. In addition, these activities may be a source of introduction of aquatic nuisance species.

Vessel hull coatings can contain toxics and may contain biocides designed to prohibit the growth of fouling organisms. These coatings discharge pollutants by leaching toxics into the water. Copper is a common biocide found in hull coatings. Discharges of the biocide tributyltin (TBT) and other organotin compounds are not permitted by the proposed permit due to their persistence in the environment and their high toxicity to marine life.

For vessels covered under the VGP, EPA requires vessel owners/operators to minimize the transport of attached living organisms when traveling into U.S. waters from outside the U.S. economic zone or between Captain of the Port (COTP) zones. If vessel owners and operators use water-pressure-based systems to clean the hull and remove old paint, they must use facilities which treat the washwater prior to discharging to waters subject to this permit in order to remove the antifouling compound(s) and fouling growth from the washwater. If they use mechanical means (scraping, etc.) to clean the hull and remove old paint, the materials removed from the hull during that process must be collected, disposed of properly (e.g., onshore), and not be allowed to contaminate nearby waters. The EPA requires vessel owners/operators who remove fouling organisms from hulls while the vessel is waterborne to employ methods that minimize the discharge of fouling organisms and antifouling hull coatings, including:

- Use of appropriate cleaning brush or sponge rigidity to minimize removal of antifouling coatings and biocide releases into the water column;

- Limiting use of hard brushes and surfaces to the removal of hard growth; and
- When available and feasible, use of vacuum or other control technologies to minimize the release or dispersion of antifouling hull coatings and fouling organisms into the water column.

The EPA requires vessel owners/operators to minimize the release of copper-based antifoulant paints during vessel cleaning operations. They require cleaning of hull surfaces coated with copper-based antifoulant paint that would not result in any visible cloud or plume of paint in the water; if a visible cloud or plume of paint develops, they require vessel owners/operators to shift to a softer brush or less abrasive cleaning technique. A plume or cloud of paint can be noted by the presence of discoloration or other visible indication that is distinguishable from hull growth or sediment removal. Production of a plume or cloud of sediment or hull growth is normal in some cases during vessel hull cleaning, but this plume or cloud must be substantially paint free (e.g., paint should not be clearly identifiable in the plume or cloud). The EPA requires owners/operators to minimize the release of fouling organisms and antifouling systems (including copper-based coatings) into surrounding waters. They require vessel owners/operators to minimize hull husbandry in critical habitat for aquatic species. Vessels that use copper-based anti-fouling paint must not clean the hull in copper-impaired waters within the first 365 days after paint application unless there is a significant visible indication of hull fouling. EPA maintains a list of copper-impaired waters on its webpage at www.epa.gov/npdes/vessels. If vessel owners or operators clean before 365 days in copper-impaired waters, they must document in their recordkeeping documentation why this early cleaning was necessary.

Welldeck Discharges. The welldeck is a platform used for launching or loading boats and cargo. Discharges from welldecks may include pollutants from equipment washing, and oils and other materials associated with engines and machinery.

For vessels covered under the VGP, EPA prohibits welldeck discharges from washdown of gas turbine engines. Welldeck discharges from equipment and vehicle washdowns must be free from garbage and must not contain oil in quantities that may be harmful as defined in 40 CFR Part 110.

Graywater Mixed with Sewage from Vessels. Except for commercial vessels in the Great Lakes, discharges of graywater that contain sewage are eligible for coverage under the VGP and must meet additional effluent limits as described in the permit as well as any additional requirements applicable to sewage discharges outside of the VGP.

For vessels covered under the VGP, EPA requires that commingled discharge of graywater and sewage must comply with the effluent limits for graywater discharge and meet the requirements set forth in section 312 of the CWA.

Exhaust Gas Scrubber Washwater Discharge. Exhaust Gas Scrubber washwater discharges (EGS washwater discharge) occur from exhaust gas cleaning systems on diesel engines. Some pollutants are transferred to the vessel's sludge tank, but the remainder may be discharged into

waters of the U.S. and may contain selenium, nutrients, oils, polycyclic aromatic hydrocarbons (PAHs) and metals, and may have a different pH than receiving waters.

For vessels covered under the VGP, EPA prohibits the discharge of exhaust gas scrubber washwater containing oil, including oily mixtures, in quantities that may be harmful as determined in accordance with 40 CFR Part 110. They prohibit the discharge of sludge or residues generated in treating exhaust gas scrubber washwater discharge and require these discharges to be delivered ashore to adequate reception facilities. In addition, vessel owners/operators with exhaust gas cleaning systems that result in washwater discharges must meet the numeric effluent limits and the monitoring requirements below:

- The discharge washwater from the exhaust gas scrubber treatment system must have a pH of no less than 6.5 measured at the ship's overboard discharge, with the exception that during maneuvering and transit, the maximum difference between inlet and outlet of 2.0 pH units is allowed.
- The maximum continuous PAH (Polycyclic Aromatic Hydrocarbons) concentration in the washwater must not be greater than 50 µg/L PAHphe (phenanthrene equivalence) above the inlet water PAH concentration for washwater flow rates normalized to 45 t/MWh. MWh refers to the maximum continuous rating (MCR) or 80 percent of the power rating of the fuel oil combustion unit. For the purposes of this criterion, the PAH concentration in the washwater must be measured downstream of the water treatment equipment, but upstream of any washwater dilution or other reactant dosing unit, if used, prior to discharge. The 50-µg/L limit is adjusted upward for lower washwater flow rates per MWh, and vice-versa, and the specific permit limits are listed in the VGP.
- The washwater treatment system must be designed to minimize suspended particulate matter, including heavy metals and ash. The maximum turbidity (monitored continuously) in washwater must not be greater than 25 FNU (formazin nephelometric units) or 25 NTU (nephelometric turbidity units) or equivalent units, above the inlet water turbidity. However, during periods of high inlet turbidity, the precision of the measurement device and the time lapse between inlet measurement and outlet measurement are such that the use of a difference limit is unreliable. Therefore, all turbidity difference readings must be a rolling average over a 15-minute period to a maximum of 25 FNU or NTU. For the purposes of this criterion, the turbidity in the washwater must be measured downstream of the water treatment equipment but upstream of washwater dilution (or other reactant dosing) prior to discharge. For a maximum of one 15-minute period within any 12-hour period, the continuous turbidity discharge limit may be exceeded by 20 percent.
- The washwater treatment system must prevent the discharge of nitrates, plus nitrites beyond that associated with a 12 percent removal of NOx from the exhaust, or beyond 60 mg/l normalized for washwater discharge rate of 45 tons/MWh, whichever is greater. MWh refers to the MCR or 80 percent of the power rating of the fuel oil combustion unit. For the purposes of this criterion, the nitrate concentration in the washwater must be measured downstream of the water treatment equipment, but upstream of any washwater dilution or other reactant dosing unit, if used, prior to discharge. The 60-mg/L limit is

adjusted upward for lower washwater flow rates per MWh, and vice-versa, and the applicable permit limits are listed in the VGP.

- The data recording system must comply with the guidelines in sections 7 and 8 of MEPC.184(59) and must continuously record pH, PAH, and turbidity.
- When the EGC system is operated in waters subject to this permit, the washwater monitoring and recording must be continuous. The values monitored and recorded must include pH, PAH, turbidity, and temperature.
- The pH electrode and pH meter must have a resolution of 0.1 pH units and temperature compensation. The electrode must comply with the requirements defined in BS 2586 or of equivalent or better performance and the meter should meet or exceed BS EN ISO 60746-2:2003.
- The PAH monitoring equipment must be capable of monitoring PAH in water in a range of at least twice the discharge concentration limit given in the table above. A demonstration must be made that the equipment operates correctly and does not deviate more than 5 percent in washwater with turbidity within the working range of the application. For those applications discharging at lower flow rates and higher PAH concentrations, ultraviolet light monitoring technology or equivalent should be used due to its reliable operating range.
- The turbidity monitoring equipment must meet requirements defined in ISO 7027:1999 or USEPA 180.1.
- All continuous monitoring equipment must be calibrated as recommended by probe manufacturers or Exhaust Gas scrubber manufacturers. At a minimum, all probes must be calibrated at least annually.
- In addition to the continuous monitoring, vessel owner/operators must collect and analyze one sample per quarter for each of the following analytes to demonstrate treatment equipment maintenance, probe accuracy, and compliance with this permit. Records of the sampling and testing results must be retained onboard for a period of 3 years.
    - Dissolved and Particulate Metals, including Aluminum, Antimony, Arsenic, Cadmium, Chromium, Copper, Lead, Manganese, Nickel, Selenium, Thallium, and Zinc
    - Vessel owners/operators should have selenium analyzed using methods capable of accurately measuring selenium in saline samples (e.g., Octopole Reaction Cell ICP-MS or Hydride generation AA methods or other methods which reduce matrix interference for selenium)
    - PAHs including Acenaphthylene, Acenaphthene, Anthracene Benz[a]anthracene, Benzo[ghi]perylene, Benzo[a]pyrene, Benzo[b]fluoranthene +, benzo[k]fluoranthene, Chrysene, Dibenz[a,h]anthracene, Fluoranthene, Fluorene, Indeno[1,2,3,c,d]pyrene, Naphthalene, Phenanthrene, and Pyrene
    - Nitrates
    - pH
    - Turbidity
    - Temperature
    - Dissolved Oxygen

- Vessel owners/operators are required to submit monitoring data to EPA annually; all monitoring data must be submitted electronically unless exempted

Fish Hold Effluent. Fish hold effluent is the seawater, ice-melt or ice slurry from fish hold tanks and may be discharged into waters of the US. These discharges may contain biological wastes, metals, nutrients, detergents and disinfectants.

For vessels covered under the VGP, EPA requires vessel owners/operators to minimize the discharge of fish hold water and/or ice while in port. They prohibit the discharge of unused bait overboard, unless that bait was caught in the same water body or watershed. The discharged of unused bait purchased from a bait shop or dealer is prohibited. The EPA requires all reasonable steps to be taken to prevent the discharge of excess fish hold water and ice while the vessel is stationary at the pier. If fish waste is contained in the fish hold effluent, they prohibit fish hold effluent from being discharged while in port, unless a physical separation method is used (e.g., filters or removal of residuals). They prohibit the discharge of fish hold effluent (including dirty ice) if unloading catch at a land-based seafood processing facility or pier. They prohibit the discharge of fish hold effluent (including dirty ice) if a shore-based discharge facility is available and meets the following standards:

- Its use is economically achievable, and
- The facility has a valid NPDES permit, or
- That facility discharges to an NPDES-permitted sewage treatment facility.

VGP Vessel-Class-Specific Requirements

Large and Medium Cruise Ships (100 people or more)

Graywater Management. The EPA requires vessel owners/operators to use appropriate onshore reception facilities for graywater while their vessel is pierside unless they meet stringent treatment requirements found in the VGP. If such facilities are not reasonably available and the vessel does not have the capacity to treat graywater, EPA requires the vessel to hold the graywater until the vessel is underway and not in waters subject to the VGP. The EPA considers appropriate reception facilities those authorized for use by the port authority or local municipality and that treat the discharge in accordance with its NPDES permit. The EPA prohibits the discharges of untreated graywater from all large cruise ships and most medium cruise ships while operating within 3 nm from shore.

While operating in nutrient-impaired waters, EPA requires large cruise ship operators to:

- Not discharge any graywater in nutrient-impaired waters subject to this permit unless the length of voyage in that water exceeds the vessel's holding capacity for graywater; and
- Minimize the discharge of any graywater into nutrient-impaired waters subject to this permit, which may require minimizing the production of graywater; and
- If a vessel's holding capacity for graywater is exceeded, treat such excess graywater (above the vessel holding capacity) by a device to meet the standards prior to discharge into nutrient-impaired waters subject to the VGP; or

- Dispose of the graywater at an onshore facility, which will discharge the effluent under a valid NPDES permit.

Graywater Treatment Standards. For large cruise ships, EPA requires treated graywater discharges to meet the following standards:

- The discharge must satisfy the minimum level of effluent quality specified in 40 CFR § 133.102;
- The geometric mean of the samples from the discharge during any 30-day period may not exceed 20 fecal coliform/100 milliliters (ml) and not more than 10 percent of the samples may exceed 40 fecal coliform/100 ml; and
- Concentrations of total residual chlorine may not exceed 10.0 micrograms per liter (µg/l).

The EPA requires quarterly monitoring of graywater (results to be submitted to EPA annually) for:

- BOD, fecal coliform, suspended solids, pH, and total residual chlorine
- *E. coli*, total phosphorus, ammonia, nitrate/nitrite, and total Kjeldahl nitrogen

Sculleries and Galleys. For large cruise ships, EPA requires the use soaps and detergents that are phosphate-free, non-toxic, and biodegradable; EPA expects that non-toxic will contain little to no nonylphenols. They require non-toxic degreasers if discharged as part of any waste stream.

Other Materials. For large cruise ships, EPA requires that vessel owners/operators prevent waste from mercury-containing products, dry cleaners or dry cleaner condensate, photo processing labs, medical sinks or floor drains, chemical storage areas, and print shops using traditional or non-soy-based inks and chlorinated solvents from entering the ship's graywater, blackwater, or bilgewater systems if water from these systems will be discharged into waters of the U.S. The EPA recommends preventing these wastes from entering these systems by plugging all drains that flow to the graywater, blackwater, or bilge systems in areas where these wastes are produced and creating alternate waste receptacles or replumbing drains to appropriate holding tanks. The EPA prohibits vessel owners/operators from discharging any toxic materials, including products containing acetone, benzene, or formaldehyde into salon and day spa sinks or floor drains if those sinks or floor drains lead to any system which will be discharged into waters of the U.S. This includes using these materials on passengers (or crew) and rinsing residuals into these sinks. Alternate waste receptacles or holding tanks are required to be used for these materials. The EPA considers addition of these materials to any systems that will discharge into waters of the U.S. to be a permit violation.

Pool and Spa Discharges. For large cruise ships, EPA prohibits discharges of pool or spa water to federally protected waters (listed in Appendix G of the VGP). The EPA authorizes discharges from pools and spas are authorized into non-Appendix G waters, provided pool and spa water to be discharged is dechlorinate and/or debrominated, and discharge occurs while the vessel is underway. To be considered dechlorinated, the total residual chlorine in the pool or spa effluent must be less than 100 µg/l if the pool or spa water is discharged without going through an Advanced Wastewater Treatment System (AWTS). To be considered debrominated, the total

residual oxidant in the pool or spa effluent must be below 25 µg/l if the pool or spa water is discharged without going through an AWTS. The EPA allows pool and spa water to be added to the graywater treatment systems; however, any resultant discharge must meet all standards and requirements found in the VGP and must be dechlorinated and/or debrominated as applicable.

Monitoring Requirements. For large cruise ships, EPA requires that vessel owners/operators report the discharge of untreated graywater to waters of the U.S. For treated graywater, EPA requires the following of vessel owners/operators:

- Demonstrate an effective treatment system if discharging graywater within 3 nm of shore.
- Take at least five (5) samples from the vessel on different days over a 30-day period that are representative of the treated effluent to be discharged.
- For initial monitoring, take samples for BOD, fecal coliform, suspended solids, pH, total residual chlorine, *E. coli*, total phosphorus (TP), ammonia, nitrate/nitrite, and Total Kjeldahl Nitrogen (TKN).
- Retain records onboard for three years.
- For maintenance monitoring, collect and analyze one sample per quarter.
- Submit data showing that the graywater standards are achieved by their treatment system to EPA's e-reporting system.
- Maintain records estimating the quantity and quality of all discharges of treated graywater into waters subject to this permit, including date, location and volume discharged, and pollutant concentrations monitored in their recordkeeping. documentation. These records shall be maintained as part of or in combination with the vessel's sewage and graywater discharge record book required under 33 CFR § 159.315.

Even if owners/operators have demonstrated their systems meet the standards, EPA, its authorized representative, or the U.S. Coast Guard retains the right to sample their graywater effluent. If EPA finds that they are not meeting these standards, they will hold the cruise ship owners/operators liable for violating their effluent limits.

For large cruise ships, EPA requires that vessel owners/operators monitor and record chlorine or bromine concentrations (as applicable) in pool or spa water before every discharge event using sufficiently sensitive 40 CFR Part 136 methods if they will discharge these streams directly into waters subject to this permit to ensure that the dechlorination/debromination process is complete.

For large cruise ships, EPA requires that vessel owners/operators to provide the following educational and training requirements to ship personnel:

- The ship's crew members who actively take part in the management of a discharge or who may affect any discharge must receive training regarding shipboard environmental procedures and must be able to demonstrate proficiency in implementing these procedures;
- Advanced training in shipboard environmental management procedures must be provided for those directly involved in managing specific discharge types or areas of the ship and these crew members must be able to demonstrate proficiency in implementing these procedures; and

- Appropriate reprimand procedures must be developed for crew whose actions lead to violations of any effluent limit set forth in this permit or procedures established by the cruise ship operator to minimize the discharge of pollutants.

The EPA also requires large cruise ships to educate passengers on their potential environmental impacts. The goals of these education efforts must include preventing trash from entering any waste stream, eliminating the addition of unused soaps, detergents, and pharmaceuticals to the graywater or blackwater systems, and minimizing production of graywater. This can be accomplished in a variety of ways including, but not limited to, posting signage and informational material in guestrooms and common areas, incorporating environmental information passenger orientation presentations or packages at the start of cruises, incorporating this information into additional lectures and seminars, or broadcasting information via loudspeakers.

Large Ferries

Ferries are vessels for hire that are designed to carry passengers and/or vehicles between two ports, usually in inland, coastal, or near-shore waters. "Large Ferry" means a "ferry" that: a) has a capacity greater than or equal to 100 tons of cargo (e.g., for cars, trucks, trains, or other land-based transportation); or b) is authorized by the U.S. Coast Guard to carry 250 or more people. All large ferries authorized to carry 100 or more tons of cars, trucks, trains, or other land-based transportation must meet the deck water and education/training requirements. Large ferries authorized by the Coast Guard to carry 250 or more people must meet the graywater and education/training requirements.

The EPA prohibits large ferries from discharging untreated below deck water from parking areas or other storage areas for motor vehicles or other motorized equipment into waters of the U.S. without first treating the effluent with an oily water separator or other appropriate wastewater treatment system. The EPA requires large ferry operators to use oil absorbent cloths or other appropriate spill response resources to clean oily spills or substances from deck surfaces. Any effluent created by washing the decks may not be discharged into federally protected waters listed in Appendix G of the VGP.

The EPA requires pierside ferries to discharge graywater using appropriate onshore reception facilities, if available and their use is economically achievable, unless the vessel treats graywater to the limits found in the VGP. If such facilities are not available, they require the owners/operators to hold the graywater if the vessel has the holding capacity and discharge the effluent while the vessel is underway. Appropriate reception facilities are those authorized for use by the port authority or municipality and that treat the discharge in accordance with its NPDES permit. The EPA requires large ferries operating within 3 nm from shore to discharge graywater only while sailing at a speed of at least 6 knots if feasible. If not feasible, EPA requires documentation.

The EPA requires ferry operators to do the following:

- The ship's crew members who actively take part in the management of the discharge or who may affect the discharge must receive training regarding shipboard environmental procedures and must be able to demonstrate proficiency in implementing these procedures;
- Advanced training in shipboard environmental management procedures must be provided for those directly involved in managing specific discharge types or areas of the ship and these crew must be able to demonstrate proficiency in implementing these procedures; and
- Appropriate reprimand procedures must be developed for crew whose actions lead to violations of any effluent limit set forth in this permit or procedures established by the ferry operator to minimize the discharge of pollutants.
- Ferry operators must also educate passengers on their potential environmental impacts. The goals of these education efforts should include eliminating the discharge of trash overboard, minimizing the production of trash from parking areas or other storage areas, eliminating the addition of unused soaps, detergents, and pharmaceuticals to the graywater or blackwater systems, and minimizing production of graywater. This can be accomplished in a variety of ways including, but not limited to, posting signage and informational material in common areas, incorporating environmental information into orientation presentations, or broadcasting information via loudspeakers.

Barges

For barges, EPA requires owners/operators to:

- Minimize the contact of below deck condensation with oily or toxic materials and any materials containing hydrocarbon. Whenever barges are pumping water from below deck, the discharge shall not contain oil in quantities that may be harmful as defined in 40 CFR Part 110. If a visible sheen is noted, vessel operators must initiate corrective action and meet the recordkeeping requirements in the VGP.
- Have spill rails and must mechanically plug their scuppers before any cargo operations if required by vessel class society and/or 33 CFR Parts 155 and/or 156. Additionally, scuppers, when available, must be mechanically plugged during fueling of ancillary equipment (e.g., generators and compressors) located on the deck of the barge. If scuppers are unavailable, other types of secondary containment should be employed. If any spills result during loading or unloading of cargo, or other ancillary equipment fueling operations, vessel owners/operators must completely clean up spills or residue before scuppers are unplugged.
- Clean out cargo residues (i.e., broom clean) such that any remaining residue is minimized before washing the cargo compartment or tank and discharging washwater overboard.
- Conduct a visual sheen test after every instance of pumping water from areas below decks, or immediately following washing down the decks. The visual sheen test is used to detect free oil by observing the surface of the receiving water for the presence of an oily sheen. If a visible sheen is observed, EPA requires the owner/operator to initiate corrective actions and meet recordkeeping/reporting requirements of the VGP.

Oil Tankers, Petroleum Tankers, and Bulk Chemical Carriers

Under the VGP, the EPA authorizes the discharges of water from deck seals if the seals are installed as an integral part of an IGS system. For vessels that have an inert gas system, EPA authorizes the discharge of effluent produced from inert gas scrubbers (IGS) into waters of the U.S. They require owners/operators of oil tankers to plug scuppers during cargo loading and unloading operations to prevent the discharge of oil into waters subject to this permit. Any oil spilled must be cleaned with oil absorbent cloths or another appropriate approach. Additionally, owners/operators of oil tankers must comply with applicable requirements of 33 CFR § 155.310 and 33 CFR Part 156, Subpart A. Vessel owners/operators must minimize the discharge of effluent produced from inert gas scrubbers if feasible for their vessel design. The EPA requires operators to conduct a visual sheen test after every instance of pumping water from areas below decks, or immediately following washing down the decks. The visual sheen test is used to detect free oil by observing the surface of the receiving water for the presence of an oily sheen. If a visible sheen is observed, EPA requires the owner/operator to initiate corrective actions and meet recordkeeping/reporting requirements of the VGP. The EPA also requires tankers to do the following:

- The ship's crew members who actively take part in the management of the discharge or who may affect the discharge must receive training regarding shipboard environmental procedures and must be able to demonstrate proficiency in implementing these procedures;
- Advanced training in shipboard environmental management procedures must be provided for those directly involved in managing specific discharge types or areas of the ship and these crew must be able to demonstrate proficiency in implementing these procedures; and
- Appropriate reprimand procedures must be developed for crew whose actions lead to violations of any effluent limit set forth in this permit or procedures established by the ferry operator to minimize the discharge of pollutants.

Research Vessels

Research vessels are those that are engaged in investigation or experimentation aimed at discovery and interpretation of facts, revision of accepted theories or laws in the light of new facts, or practical application of such new or revised theories or laws. For research vessels, EPA authorizes the discharge of tracers (dyes, fluorescent beads, SF6), drifters, tracking devices and the like, and expendable bathythermograph (XBT) probes, into waters subject to the VGP, provided such discharges are for the sole purpose of conducting research on the aquatic environment or its natural resources in accordance with generally recognized scientific methods, principles, or techniques. The EPA requires owners/operators of research vessels to discharge only the minimal amount of materials necessary to conduct research on the aquatic environment or its natural resources in accordance with generally recognized scientific methods, principles, or techniques.

Emergency and Rescue Vessels

For emergency and rescue vessels, EPA authorizes owners/operators to discharge waste streams in conjunction with training, testing, and maintenance operations, provided that they comply

with all additional requirements of the CWA (e.g., section 311) and the National Contingency Plan (40 CFR Part 300). This part does not relieve vessel operators of any additional responsibilities under the CWA and the National Contingency Plan, which prohibits the discharge of oil for research or demonstration purposes without Administrator approval. The use of foaming agents for oil or chemical fire response must be implemented in accordance with the National Contingency Plan (40 CFR Part 300).

VGP Inspections, Recordkeeping, Reporting, Compliance, and Enforcement

Self Inspections and Monitoring

Routine Visual Inspections. For vessels covered under the VGP, EPA requires vessel owners/operators to conduct a routine visual inspection at least once per voyage (but not more than once per day and not less than once per week). Vessel owners/operators are required to document the findings of each routine visual inspection including: the date and time of inspection, ship locations inspected, personnel conducting the inspection, location of any visual sampling and observations, any potential problems and sources of contamination found, any corrective actions taken, and the signature of the person conducting the inspection, who must be a signatory under 40 CFR § 122.22. These records must be available to EPA or its authorized representative upon request. Routine visual inspections must include

- Inspections of all accessible areas addressed in the VGP, including, but not limited to: cargo holds, boiler areas, machinery storage areas, welldecks, and other deck areas. Vessel owners/operators must ensure:
  - areas are clear of garbage, exposed raw materials, oil, any visible pollutant or constituent of concern that could be discharged in any waste stream
  - pollution prevention mechanisms are in proper working order
  - pollution prevention procedures are in place to minimize the addition of pollutants to any waste stream.
- Verification to the extent feasible that effluent limits and related requirements are being met and document any instances of noncompliance.
- Visual inspection of safely accessible deck and cargo areas and all accessible areas where chemicals, oils, dry cargo, or other materials are stored, mixed, and used, whether or not the areas have been used since the last inspection.
- Visual monitoring of the water around and behind the vessel for visible sheens, dust, chemicals, abnormal discoloration or foaming, and other indicators of pollutants or constituents of concern originating from the vessel during each watch.
- Corrective actions, if pollutants or constituents of concern are discharged in a manner that violates the limitations of the VGP.

Extended Unmanned Period Inspections. For vessels covered under the VGP, the EPA provides vessels with the option of conducting extended unmanned period (EUP) inspections in lieu of vessels conducting routine visual inspections for vessels entering an unmanned period of 13 days or more. The extended unmanned period (EUP) inspection must include:

- A routine visual inspection as described above.

- Ensuring material storage, and toxic and hazardous material requirements are met.
- Ensuring all oils and oily machinery are properly secured, covered, and protected. Any spilled or leaked oils must be cleaned up immediately. If machinery or equipment is leaking oil, the leaks must be stopped or appropriate containment must be in place to capture any leaking oil.
- Documenting whether automatic bilge water pump(s) will be engaged on the vessel during the EUP.
- Documenting the amount of fuel on board.
- Documenting the amount of ballast water on board.
- Documenting the date the EUP began.

While a vessel is in extended lay-up, the owner/operator must examine the outside of the vessel and surrounding waters at least once every two weeks for any evidence of leaks, loss of cargo, or any other spills which might result in an unauthorized discharge. If any deficiencies are observed while the vessel is in EUP, the vessel owner/operator must document those deficiencies and the corrective actions taken to resolve those deficiencies. If a visible sheen is noted on the surface of the surrounding water, the source of the oil must be identified and corrective action must be taken immediately. The vessel owner/operator must notify EPA of the visible sheen. All records must be available to EPA or its authorized representative upon request. Before a vessel reenters service, the EPA requires vessel owners/operators to conduct a post lay-up routine visual inspection. As part of this inspection, the owner/operator must document the date the EUP ended, whether fluids (e.g., fuel, ballast water) are at their pre-EUP levels, and whether any spills or leaks of oily materials are observed. Any deficiencies noted must be corrected before the vessel reenters service.

Comprehensive Annual Vessel Inspections. For vessels covered under the VGP, EPA requires vessel owners/operators to conduct comprehensive vessel inspections at least once every 12 months. These annual inspections must be conducted by qualified personnel, including: the Master or owner/operator of the vessel, if appropriately trained; appropriately trained marine or environmental engineers or technicians; or an appropriately trained representative of a vessel's class society acting on behalf of the owner/operator. Comprehensive annual inspections must cover all areas of the vessel affected by the requirements in this permit that can be inspected without forcing a vessel into dry dock. Areas that inspectors must examine include, but are not limited to:

- Vessel hull for attached living organisms, flaking anti-foulant paint, exposed TBT or other organotin surfaces;
- Ballast water tanks, as applicable;
- Bilges, pumps, and oily water separator sensors, as applicable;
- Oil discharge monitoring system and electronic valve switching function, as applicable;
- Protective seals for lubrication and any hydraulic oil leaks;
- Oil and chemical storage areas, cargo areas, and waste storage areas; and
- All visible pollution control measures to ensure that they are functioning properly.

If any portions of the vessel are not inspectable without the vessel entering drydock, the vessel owner/operator must inspect these areas during their next drydock inspection. The EPA requires vessel owner/operators to document areas not accessible during the annual inspection.

The annual inspections must also include a review of monitoring data collected in accordance with vessel-class-specific requirements, and routine maintenance records to ensure that required maintenance is being performed (e.g., annual tune-ups for small boats that have wet exhaust). The inspection must verify whether all monitoring, training, and inspections are logged and documented according to permit requirements.

The EPA requires vessel owners/operators to take corrective action to resolve deficiencies revealed by inspection, including violation of the effluent limits. Vessel owners/operators are required to record annual inspection results.

Dry Dock Inspections. For vessels covered under the VGP, EPA requires vessel owners/operators to make dry-dock reports available to EPA or an authorized representative of EPA upon request. The dry-dock report must note that:

- The chain locker has been cleaned for both sediment and living organisms;
- The vessel hull, propeller, rudder, thruster gratings, sea chest, and other surface areas of the vessel have been inspected for attached living organisms and those organisms have been removed or neutralized;
- Any antifoulant hull coatings have been applied, maintained, and removed consistent with the FIFRA label if applicable; any exposed existing or any new coating does not contain biocides or toxics that are banned for use in the United States;
- For all cathodic protection, anodes or dialectic coatings have been cleaned and/or replaced to reduce flaking; and
- All pollution control equipment is properly functioning.

Recordkeeping

For vessels covered under the VGP, EPA requires vessel owners/operators to keep records on the vessel or accompanying tug that include the following information:

1. Owner/Vessel information:
    - Name,
    - Owner and Vessel IMO Number (official number if IMO number not issued),
    - Vessel type,
    - Owner or operator company name,
    - Owner or operator certifying official's name,
    - Address of owner/operator,
    - Gross tonnage,
    - Call sign, and
    - Port of Registry (Flag).

2. Voyage Log. Include the dates and ports of arrival, vessel agent(s), last port and country of call, and next port and country of call (when known).

3. If vessel owners/operators have violated any effluent limit, they must document the violation and record:

- A description of the violation,
- Date of the violation,
- Name, title, and signature of the person who identified the violation,
- Name, title, and signature of the person who is recording the violation (if different from person who identified the violation),
- If a Corrective Action Assessment pursuant to Part 3.2 is needed, attach a copy or indicate where the corrective action assessment is stored, and
- If a Corrective Action Assessment was previously conducted pursuant to Part 3.2 (and revisions are not needed for this violation of the effluent limit), a reference to that previous corrective action assessment.

4. Log of findings and any deficiencies and problems identified during routine visual inspections and extended unmanned inspections, including a discussion of any corrective actions, if applicable. Include date, inspector's name, findings, and corrective actions planned or taken. If no deficiencies or problems are found during a routine visual inspection, the vessel owner/operator shall record that the inspection was completed with the inspector's name and date. Routine visual inspections and extended unmanned inspections (if applicable) must be recorded as completed.

5. Analytical results of all monitoring conducted, including sample documentation, results, and laboratory quality assurance (QA) documentation. Log of findings from comprehensive annual vessel inspections conducted, including a discussion of any corrective actions planned or taken required. Include date, inspector's name, findings, and a description of the corrective actions taken.

6. Log of findings from comprehensive annual vessel inspections, including a discussion of any corrective actions planned or taken. Include date, inspector's name, findings, and a description of the corrective actions taken.

7. Log of findings from drydock inspections conducted including a discussion of any corrective actions planned or taken. Include date, inspector's name, findings, and a description of the corrective actions taken.

8. Record of any specific requirements given to your vessel by EPA or its authorized representative and how those requirements have been met.

9. Additional maintenance and discharge information to be recorded and kept in a log on the vessel:

a. Deck maintenance. Record dates, materials used, application process, etc. for any significant maintenance of the deck surface(s) (e.g., more than routine, daily cleaning activities, such as sweeping).
b. Bilgewater. Record dates, location, oil concentration (for MARPOL vessels) or visible sheen observation (non-MARPOL vessels), and estimated volume of bilgewater discharges. Record dates, materials used, application process, etc. for any antifouling paint applied to the vessel.
d. AFFF. Record dates, estimated volumes, and constituents of any discharges of AFFF.
e. Chain locker inspections. Record dates of inspections and any rinsing conducted within waters subject to this permit.
f. Controllable pitch propeller, stern tube, and other oil-to-sea interface maintenance. Record dates and locations of any maintenance of controllable pitch propellers that occurs while the vessel is in waters subject to this permit.
g. Any emergencies requiring discharges otherwise prohibited to waters listed in Appendix G of the VGP.
h. Gas Turbine Water Wash. Record dates and estimated volume of any discharge of gas turbine washwater within waters subject to this permit. If hauled or disposed of onshore, record log hauler and volume.
i. Estimated volume and location of graywater discharged while in waters subject to this permit.

10. All other documentation required pursuant to the VGP.

11. Record of training completed as required by this permit, and where applicable, strategy for passenger training. For purposes of this part, if vessel owners/operators include their training plans as part of their ISM or similar environmental management plans, and they can document that they fully implement those plans, they will meet the recordkeeping requirements of this part.

The EPA requires certification of accurate information for all NOIs, NOTs, the PARI form, and any report (including any monitoring data) submitted to EPA, pursuant to 40 CFR § 122.22. The vessel owner/operator are required to retain copies of all reports, certifications, records, monitoring data, and other information required by this permit, and records of all data used to complete the NOI to be covered by this permit, for a period of at least 3 years from the date that coverage under this permit expires or is terminated. The vessel master, owner/operator, or person in charge is required to make available to EPA or an authorized representative from EPA all records kept under this part upon request.

For vessels equipped with ballast tanks that are bound for a port or place in the United States, EPA requires the recordkeeping requirements of 33 CFR Part 151. The master, owner, operator, or person in charge of a vessel bound for a port or place in the United States is required to keep written records that include the following information:
1. Total ballast water information. Include the total ballast water capacity, total volume of ballast water on board, total number of ballast water tanks, and total number of ballast water

tanks in ballast. Use units of measurements such as metric tons (MT), cubic meters (m3), long tons (LT), and short tons (ST).

2. Ballast water management. Include the total number of ballast tanks/holds that are to be discharged into the waters of the United States or to a reception facility. Indicate whether the vessel has a ballast water management plan and IMO guidelines on board, and whether the ballast water management plan is used.

3. Information on ballast water tanks that are to be discharged into waters subject to this permit or to a reception facility. Include the following:

- The origin of ballast water. This includes date(s), locations(s) (including latitude and longitude and port [if relevant]), volume(s), and temperatures(s). If a tank has been exchanged, list the loading port of the ballast water that was discharged during the exchange.
- The date(s), location(s) (including latitude and longitude), volume(s), method, thoroughness (percentage exchanged if exchange conducted), sea height at time of exchange if exchange conducted, of any ballast water exchanged or otherwise managed.
- Specific records pertaining to treated ballast water (see Part 2.2.3.5 of the permit).
- The expected date, location, volume, and salinity of any ballast water to be discharged into the waters of the United States or a reception facility.

4. Discharge of sediment. If sediment is to be discharged into a facility within the jurisdiction of the United States, include the location of the facility where the disposal will take place.
The ballast water reporting forms must be kept on board the vessel and must be submitted to the National Ballast Information Clearinghouse before arriving to U.S. ports if required by the U.S. Coast Guard. In addition, all vessels that conduct saltwater flushing but do not report saltwater flushing to the NBIC, must instead keep a record of saltwater flushing to meet the requirements of this permit.

Reporting Requirements

For vessels covered under the VGP, the EPA requires vessel owners/operators to submit an Annual Report, i.e., the form available in Appendix H of the VGP. The EPA requires vessel owners/operators to report all instances of noncompliance as part of the Annual Report. The EPA requires vessel owners/operators to submit all analytical monitoring results as part of the Annual Report. Vessel owners/operators are required to answer all questions accurately and completely, and provide the necessary information and/or data to support each response. They are required to submit their Annual Report electronically unless exempted as described in the VGP. The EPA requires owners and operators of vessels greater than 300 gross tons or ballast water capacity of at least 8 cubic meters to submit a Notice of Intent (NOI). They require all other vessel owners/operators to complete a Permit Authorization and Record of Inspection Form (PARI) and keep it onboard at all times.

The following questions will be added to the Annual Report form available in Appendix H of the VGP:

- When is your next scheduled drydocking? (This question is optional for inland vessels less than 300 gross tons and unmanned, unpowered barges)
- When did you last conduct a below water (or drydock) hull inspection? (This question is optional for inland vessels less than 300 gross tons and unmanned, unpowered barges)
- Do you use anti-foulant paint? If so, what type of anti-fouling hull coating? [Select specific product from drop-down menu]. When was it last applied?

Corrective Action

Under the VGP, EPA has the authority to take enforcement action to require any remedy or corrective action necessary to achieve compliance as quickly as possible. They require vessel owners/operators to take action to ensure that the problem is eliminated and will not be repeated if they:

- Violate one or more effluent limits or any other requirement of this permit, or if an inspection or evaluation of your vessel by an EPA official or an official agent acting on EPA's behalf determines that modifications to the control measures are necessary to meet the effluent limits;
- Become aware, or EPA determines, that their measures do not control discharges as stringently as necessary to meet applicable water quality standards; or
- Become aware, or EPA determines, that their pollution control measures or best management practices are not being properly operated and maintained, or are not having the intended effect in minimizing pollutant discharges.

Vessel owners/operators are required to conduct a corrective action assessment into the nature, cause, and potential options for eliminating these problems and to retain the records aboard their vessel. The EPA requires corrective actions to occur within 2 weeks for simple adjustments, 3 months for adjustments that require new parts, and during the next dry dock for large repairs. The EPA considers the initial occurrence of a problem to be a violation of the permit; they consider failing to conduct a corrective action assessment and failing to correct the problem by the applicable deadline to be additional violations of the permit.

Permit Compliance and Enforcement

The EPA requires vessel owners and operators to comply with the VGP requirements. Under the CWA, any person who knowingly falsifies, tampers with, or renders inaccurate any monitoring device or method required to be maintained under the CWA shall, upon conviction, be punished by a fine of not more than $10,000, or by imprisonment for not more than 2 years, or both. If a conviction of a person is for a violation committed after a first conviction of such person under this paragraph, punishment is a fine of not more than $20,000 per day of violation, or by imprisonment of not more than 4 years, or both. Any person who knowingly makes any false material statement, representation, or certification in any application, record, report, plan or other document filed or required to be maintained under the CWA shall, upon conviction, be punished by a fine of not more than $10,000 per violation, or by imprisonment for not more than 6 months per violation, or by both. In addition, false statements or representations, as well as alterations or false entries in documents, may be punishable by more severe criminal penalties pursuant to 18

USC §1001 or 18 USC §1519.

To monitor and insure compliance with its VGP, EPA will develop and implement a monitoring and compliance plan by December 2013 to assess the performance of the permit in protecting listed resources. Currently, the U.S. Coast Guard conducts all inspections of vessels covered under the VGP using EPA's Job Aide, i.e., "Guidelines for Coast Guard Evaluations of Compliance with the US Environmental Protection Agency's (EPA) Vessel General Permit (VGP) for Discharges Incidental to the Normal Operation of Vessels." To insure that the inspections assess new requirements of the proposed VGP, EPA will:

- Work with the U.S. Coast Guard, with the early input of the Services, to update the Job Aide to reflect requirements included in the new VGP.
- Create an online training and/or a webinar training regarding the VGP requirements/prohibitions, within one year of final permit issuance.
- Create a comprehensive checklist of requirements for inspector training in conjunction with the U.S. EPA National Enforcement Training Institute, within one year of final permit issuance.
- Provide and continue to provide the U.S. Coast Guard with points of contact, including the title, office name, current address and phone number for employees at headquarters offices and in each EPA Region (and USCG District) who can be contacted with questions regarding inspection protocols.

In addition, EPA's Office of Enforcement and Compliance Assurance will meet with the Services to discuss enforcement under the VGP. The EPA will gather, review and analyze inspection data submitted by the Coast Guard; the results will be summarized (by type of vessel, location when available, type of deficiencies, and number of deficiencies) and provided to the Services annually.

Small Vessel General Permit (sVGP)

The proposed sVGP authorizes owners/operators of eligible vessels less than 79 feet in length to release discharges incidental to the normal operation of a vessel into waters of the U.S. The following narrative describes the discharges authorized or prohibited should EPA issue the sVGP. It also describes the activities that EPA requires of vessel owners/operators for vessels covered under the sVGP, including reporting and compliance requirements.

For vessels covered under the sVGP, the EPA authorizes discharges incidental to the normal operation of a vessel when operating as a means of transportation, including: anti-foulant hull coating leachate, bilgewater, deck runoff, fish hold effluent, graywater, non-contact engine cooling water, packing gland effluent, and underwater hull husbandry.

The EPA does not authorize discharges that are not incidental to the normal operation of a vessel when operating as a means of transportation. They do not authorize the following discharges: industrial operations waste, medical waste, noxious liquid substance residue, sewage, used or spent oil, garbage or trash, tetrachloroethylene degreasers, and discharges covered by another NPDES permit.

Discharges to be Authorized by the sVGP

For vessels covered under the sVGP, EPA requires vessel owners and operators to minimize vessel discharges. They define minimize as to reduce and/or eliminate to the extent achievable using control methods, including best management practices, that are technologically available and economically practible and achievable in light of best marine practices. The EPA prohibits the addition of any constituents to any discharge that are not incidental to the normal operation of a vessel. They require all vessel owner/operators to ensure adequate training for the master, operator, person-in-charge, and crew who manage or affect incidental discharges. They require that all discharges meet the following standards:

- Minimize the potential for substances or pollutants to accidentally enter the effluent, including spills.
- May not contain visible garbage in the effluent.
- May not use any dispersants, cleaners, chemicals, or other materials or emulsifiers that remove the appearance of a visible sheen.
- Minimize the introduction of constituents of concern or pollutants, such as foam or floating solids.
- Oil, including oily mixtures, may not be discharged in quantities that may be harmful or cause a visible sheen.
- The discharge of antifreeze into waters subject to this permit must be minimized. For vessel engines that have been winterized, minimization can be achieved by draining antifreeze from the engine prior to startup or capturing antifreeze when discharged from the engine upon startup. The discharge of antifreeze with toxic or known carcinogenic additives, such as ethylene glycol and methanol, is prohibited.
- When feasible, cleaning, maintenance, and repair jobs should be done while the vessel is out of the water or in drydock.
- Any soaps, detergents or cleaners used must be non-toxic, phosphate-free, and biodegradable; EPA expects that non-toxic soaps will contain little to no nonylphenols. Phosphate-free soap contains by weight 0.5% or less of phosphates or derivatives of phosphates.
- Any spill of oil or other harmful chemicals that are discharged in a quantity that may be harmful or cause a visible sheen as established under 40 CFR Part 110, 40 CFR Part 117, or 40 CFR Part 302, must be reported immediately to the National Response.

Water Quality-Based Effluent Limits

Impaired waters or "water quality limited segment[s]" are those which have been identified by a State or EPA pursuant to Section 303(d) of the CWA as not meeting applicable State water quality standards. Impaired waters may include both waters with EPA-approved or EPA-established Total Maximum Daily Loads (TMDLs) and those for which EPA has not yet approved or established a TMDL.

The EPA requires vessel owners/operators to comply with the requirements of this section, including any additional requirements that EPA may impose pursuant to this section, if the vessel

discharges to an impaired water without an EPA-approved or established TMDL. This provision also applies to situations where EPA determines that a vessel's discharge is not controlled as necessary to meet water quality standards in another water body, even if its discharge is to areceiving water that is not specifically identified on a Section 303(d) list.

If a vessel discharges to an impaired water with an EPA-approved or established TMDL and EPA or state TMDL authorities have informed the owners/operators that a Waste Load Allocation (WLA) has been established that applies specifically to their vessel's discharges, to discharges from vessels in their vessel class or type, or to discharges from vessels in general if applicable, EPA requires discharges to be consistent with the assumptions and requirements of that WLA. If such a WLA exists, EPA will inform vessel owners/operators if any additional limits or controls are necessary for discharges to be consistent with the assumptions of any available WLA in the TMDL, or whether an individual permit application is necessary. This provision also applies to situations where EPA determines that discharges are covered by the WLA in an EPA- approved or established TMDL for another water body, even if the discharge is to a receiving water that is not specifically identified on a Section 303(d) list. If an applicable TMDL exists either individually or categorically for a vessel or vessel class (including disallowing discharges from your vessel), EPA and/or state TMDL agencies will inform vessel operators of specific requirements via dock side postings, information made available from the Captain of the Port, or by specifically contacting vessel owners/operators.

Additional Technology-Based Effluent Limits

Fuel Management. For vessels covered under the sVGP, the EPA requires vessel owners/operators to:

- Have a functioning fuel-air separator or a fuel tank vent to prevent a fuel spill, for all motorized vessels constructed on or after December 19, 2013.
- Prevent overfilling and refrain from topping off fuel tanks.
- Use an oil absorbent material or other appropriate device while fueling the vessel to prevent any oil from entering waters subject to this permit, for motorized vessels that do not have a functioning fuel-air separator or a fuel tank vent.
- Regularly inspect the fuel and hydraulic systems for any damage or leaks, for instance during fueling and/or when performing routine maintenance.
- Fill portable tanks onshore, instead of on the dock or on your vessel, unless impracticable.

Engine and Oil Control. For vessels covered under the sVGP, the EPA requires vessel owners/operators to:

- Periodically inspect (at least once per quarter) the engine for any loose or leaking hoses, gaskets, and/or seals and if needed, repair or replace damaged parts as soon as possible.
- Place oil absorbent material or other spill response equipment under the vessel engine or use other preventative practices to minimize oil entering the bilgewater.
- Immediately clean up any spill or overflow of oil or other engine fluids. A supply of absorbent pads, pillows, or other materials should be kept onboard as appropriate for use

in addressing or remediating any such spills or overflows, and supplies should be checked quarterly and restocked as necessary.

- Dispose of used oil-absorbent materials onshore in containers designed for oily waste disposal.
- If the vessel has a bilge oily water separator, check for the presence of a visible sheen in surrounding waters periodically while discharging.
- If a visible sheen is observed as a result of this discharge, suspend discharge until the problem is corrected and clean up immediately.
- Use an oil-absorbent material to remove any oil from the bilge before discharging, if a U.S. Coast Guard type-approved bilge oily water separator is not used.
- Any discharge of packing gland or stuffing box effluent must not contain oil, including oily materials, in quantities that may be harmful. These discharges must not produce a visible sheen of oil or oily materials. If accessible, segregated water which contains drips from packing gland effluent or stuffing box effluent must be checked daily for the presence of a visible sheen, while the vessel is operational and manned. If not accessible while the vessel is operational and manned, then the surrounding water must be checked for the presence of a visible sheen (while operating). If a visible sheen is observed, appropriate measures such as the use of oil absorbent materials must be used to remove the presence of oil before the effluent may be discharged. Dispersants or emulsifiers that remove the appearance of a visible sheen must not be used.
- If a visible sheen is observed in surrounding waters as a result of this discharge, suspend the discharge until the problem is corrected and clean up immediately.
- Use environmentally acceptable lubricants in all machinery and equipment, including but not limited to stern tubes, wires, and two-stroke engines, where discharges of oil to surrounding waters are likely to occur, unless technically infeasible.
- Unless infeasible, prior to pumping the bilge, inspect the bilgewater for an oily sheen. While pumping the bilge, examine the surrounding water for the presence of a visible sheen.
- If a visible sheen is observed as a result of this bilge pumping, suspend the discharge until the problem is corrected and clean up immediately.
- Refrain from adding dispersants, detergents, emulsifiers, chemicals or other substances that remove the appearance of a visible sheen to the bilge.

Solid and Liquid Waste Management. For vessels covered under the sVGP, the EPA requires vessel owners/operators to:

- Prevent trash or garbage, including food waste, cigarette butts, bottles, and caps from entering any waste stream covered by this permit.
- Have appropriate receptacles for retaining trash or garbage onboard the vessel. Examples of appropriate receptacles include secured trash bags or coolers, bins, or trash cans with secure lids. Store any used antifreeze, paint, out-of-date flares, or other toxics in secure containers and dispose of them properly at onshore disposal facilities.
- Secure or otherwise prevent loose items on deck from entering any waste stream covered by this permit.

- Prevent monofilament line, fishing nets, lines, lures, rope, bait boxes, and hooks from entering any waste stream covered by this permit.

Deck Washdown and Runoff and Above Water Line Hull Cleaning. For vessels covered under the sVGP, the EPA requires vessel owners/operators to:

- Use all soaps and cleaners as directed by the label.
- Use non-toxic, phosphate-free, and biodegradable soaps, detergents, or cleaners; EPA expects that soaps that are "minimally toxic" will contain little to no nonylphenols. The use of soaps that are labeled toxic or highly toxic is prohibited under the sVGP.
- Prevent the introduction of on-deck debris, garbage, residue, and spills into deck washdown and runoff discharges.
- Minimize the discharge of paint chips and residue, especially during cleaning, maintenance, paint application, and reapplication. When performing these activities, collect and then dispose of chips and residues onshore in accordance with applicable requirements for the facility where operations are performed.

Vessel Hull Maintenance. For vessels covered under the sVGP, the EPA requires vessel owners/operators to:

- Minimize the impact of any anti-foulant system or the discharges resulting from anti-foulant paints on the aquatic environment. Use less toxic alternatives to copper based paints to the extent practicable and available.
- Minimize hull husbandry in designated critical habitat for ESA-listed aquatic species.
- Do not use anti-fouling coatings if not needed, where drying or hull cleaning at haul-out is adequate for managing fouling (e.g., for vessels that are hauled frequently or for over-winter storage).
- Avoid discharges of tributyltin (TBT), i.e., zero-discharge standard by avoiding TBT coatings; or having an effective overcoating that completely eliminates TBT discharges.
- Avoid cleaning anti-foulant paint that releases biocides for the first 90 days after application.
- If the anti-foulant paint requires cleaning, gently clean hulls on a regular basis (this minimizes the need for stronger cleaners and more abrasive brushes).
- When cleaning hulls coated with anti-fouling paint while the vessel is in the water, use only soft sponges. Examine the water while cleaning to avoid forming a plume of paint. Stop immediately if any visible plume of paint appears in the water. Consider hiring a qualified, professional hull cleaner to do the work, and ensure that they follow environmental guidelines.
- When cleaning hulls coated with anti-fouling paint while the vessel is out of the water, always work away from the water in a location where paint chips and dust will not be washed into the water. Place a tarp under the area to catch loose particles, and properly dispose of paint chips, dust, and other particles.
- Vessel hulls must be periodically inspected (at least once per quarter), and if necessary, cleaned to prevent the spread or dispersal of potentially invasive species.

- Minimize the transport of any visible living organism from one waterbody to another by regularly cleaning and maintaining the hull. If a vessel's hull is severely fouled the cleaning has not occurred regularly. Hence, severe fouling could be a permit violation.
- Prior to transporting the vessel from one waterbody to another overland, inspect the visible areas of the vessel for any attached or visible stowaway living organisms. If organisms are found, they must be removed and appropriately discarded onshore. Removed organisms may not be discharged into waters subject to the sVGP.

Graywater. For vessels covered under the sVGP, the EPA requires vessel owners/operators to:

- Minimize graywater discharges in areas that have heavy vessel traffic or heavy recreational use and in marine sanctuaries, national wildlife refuges, national wild and scenic rivers, and national wilderness areas. If the vessel has the capacity to store graywater in these waters, it should be stored and later discharged in other waters or onshore.
- Minimize the production of graywater while the vessel is stationary in confined waters (e.g., marinas, harbors). If the vessel has the capacity to store graywater, it should be stored and discharged at an appropriately equipped onshore facility or discharged while the vessel is underway.
- Use soaps, detergents and cleaners that are phosphate free, non-toxic, and biodegradable for any activities that may result in their introduction into graywater; EPA expects that non-toxic soaps will contain little to no nonylphenols. Excess oils, including animal fats and vegetable oils, used during cooking must not be added to the graywater system or into any other discharge covered by this permit.

Fish Hold Effluent. For vessels covered under the sVGP, the EPA requires vessel owners/operators to:

- Discharge your effluent (including dirty ice) to a shore-based facility instead of discharging to surrounding waters, if a shore-based discharge facility is available and economically achievable.
- Avoid discarding any unused bait overboard, unless caught in that waterbody or watershed. Unused bait purchased from a bait shop or dealer may not be discharged overboard. Though their sVGP includes the previous two sentences, EPA clarified that these statements refer to "live" bait in their comments (dated November 16, 2012) on our draft Opinion.
- Minimize the discharge of fish hold water or ice while in port. All reasonable steps must be taken to prevent the discharge of excess fish hold water and ice while the vessel is stationary at the pier. If solid fish waste is contained in the fish hold effluent, the fish hold effluent may not be discharged while in port, unless a physical separation method is used (e.g. filters or removal of residuals).
- Use non-toxic, phosphate-free, and biodegradable soaps and cleaners when cleaning fish holds. While pierside and stationary, avoid washing any residual solids into surrounding waters.

Ballast Water. For vessels covered under the sVGP, the EPA requires vessel owners/operators to:

- Avoid the discharge or uptake of ballast water in areas within, or that may directly affect, marine sanctuaries, marine preserves, marine parks, shellfish beds, or coral reefs.
- Minimize or avoid uptake of ballast water in the following areas and situations: designated critical habitat for ESA-listed species; areas known to have infestations or populations of harmful organisms and pathogens (e.g., toxic algal blooms); areas near sewage outfalls; areas near dredging operations; areas where tidal flushing is known to be poor or times when a tidal stream is known to be turbid; in darkness, when bottom-dwelling organisms may rise up in the water column; where propellers may stir up the sediment; areas with pods of whales, convergence zones, and boundaries of major currents.
- Discharge only the minimal amount of ballast water essential for vessel operations.
- When feasible, use one of the following measures to reduce the potential for transfer or introduction of organisms to waters of the U.S.: use potable water for ballasting; utilize onshore treatment or disposal methods for ballast water; for vessels that conduct fixed routes, capture and reuse ballast water in each port.

Seawater Cooling Overboard Discharge. For vessels covered under the sVGP, the EPA requires vessel owners/operators to:

- Discharge seawater cooling overboard discharge when the vessel is underway to minimize any thermal impacts to the receiving water, when possible.
- To reduce the production and discharge of seawater cooling overboard discharge, EPA recommends that vessel owner/operators use shore-based power when the vessel is in port if:
    - Shore power is readily available for vessel owner/operators from utilities or port authorities;
    - Shore-based power supply systems are capable of providing all needed electricity required for vessel operations; and
    - The vessel is equipped to connect to shore-based power and such systems are compatible with the available shore power.

sVGP Recordkeeping, Inspections, Compliance, and Enforcement

Recordkeeping Requirements

For vessels covered under the sVGP, the EPA requires vessel owners/operators to read and sign a PARI and keep it on board at all times.

Quarterly Visual Inspection Requirements

For vessels covered under the sVGP, the EPA requires vessel owners/operators to conduct quarterly visual inspection. The inspection must cover all discharges and all applicable areas that impact discharges covered by the permit and at minimum, the visible portions of the hull. The inspection must ensure that all areas are clear of garbage, exposed raw materials, oil, or any

other materials that could be discharged into any waste stream or receiving waters and that there is no uncontrolled or unmanaged sources of these pollutants. The EPA will develop a discharge requirement checklist or other implementation tool for vessel owners/operators to use to assist in their quarterly self-inspections; this tool will be developed within one year of the final sVGP permit issuance (December 2013). The EPA requires vessel owners/operators to:

- Frequently check the area around and behind the vessel to ensure that no visible sheen, dust, chemicals, or discoloration is originating from the vessel, while the vessel engine is operating.
- Check to ensure that all equipment on board is in proper working condition. This equipment includes, as appropriate, oily water separators, monitors, bilges, pumps, and generators.
- Check all protective seals for lubrication and hydraulic oil leaks.
- Document any problem(s) requiring corrective action and corrective actions that were taken to resolve the problem(s). The dates of these inspections must be documented on the sVGP PARI Form along with any corrective actions taken.

Corrective Action

The EPA retains the right to request any information required to determine whether cause exists for modifying, revoking and reissuing, or terminating the sVGP or to determine compliance with the sVGP. They require vessel owners/operators to provide any requested information within a reasonable time following the request. The vessel owner or operator is required to allow EPA or an authorized representative to:

- Inspect any vessel, equipment (including monitoring and control equipment), practices, or operations regulated or required under this permit; and
- Sample or monitor, for the purpose of assuring permit compliance or as otherwise authorized by the Act, any substances or parameters at any location.

Authorized representatives include the U.S. Coast Guard, an authorized contractor acting as a representative of the Administrator or Director, or an appropriate state agency. Authorized representatives should present their credentials to the vessel owner or operator before inspecting or entering any vessel. The EPA retains authorities under Section 308 of the CWA, including the authority to enter, access, inspect, sample, monitor, and obtain information to verify compliance with the CWA.

Permit Compliance and Enforcement

The EPA requires vessel owners and operators to comply with the sVGP requirements. Under the CWA, any person who knowingly falsifies, tampers with, or renders inaccurate any monitoring device or method required to be maintained under the sVGP shall, upon conviction, be punished by a fine of not more than $10,000, or by imprisonment for not more than 2 years, or both. If a conviction of a person is for a violation committed after a first conviction of such person under this paragraph, punishment is a fine of not more than $20,000 per day of violation, or by imprisonment of not more than 4 years, or both. Any person who knowingly makes any false material statement, representation, or certification in any record or other document

submitted or required to be maintained under the sVGP shall, upon conviction, be punished by a fine of not more than $10,000 per violation, or by imprisonment for not more than 6 months per violation, or by both. In addition, false statements or representations, as well as alterations or false entries in documents, may be punishable by more severe criminal penalties pursuant to 18 USC §1001 or 18 USC §1519.

To monitor and insure compliance with its sVGP, EPA will:

- Develop and implement a monitoring and compliance plan within 1 year of finalization of the permit to assess the performance of the permit in protecting listed resources.
- Organize a meeting with their Office of Enforcement and Compliance Assurance for the Services to discuss and provide input to OECA's inspection and enforcement plans for the sVGP.
- Work with USCG, with the input of the Services, to update the inspection and enforcement document inspection checklist (i.e., Job Aide) to reflect sVGP requirements
- Conduct outreach to the states to raise awareness about small vessel requirements; EPA will encourage States to check records of quarterly self-inspection records and visible portions of the hull while inspecting vessels when implementing their own programs.
- Develop trainings covering the small vessel requirements and share these with the Coast Guard and States

The EPA will provide the Services with a yearly summary of the number of inspections (by EPA, the Coast Guard, States or other sources) and results (including number of inspections by type of vessel, location when available, and number and type of deficiencies). During a Senior Manager's Meeting (October 19, 2012), EPA assured the Services that a subset of small vessels would be inspected.

Additional Actions Taken by EPA to Protect Listed Resources

Reinitiation Clause

The VGPs include permit reopener clauses which state that the permits are subject to modification based on the receipt of new information that was not available at the time of permit issuance and would have justified the application of different permit conditions at the time of permit issuance. This new information may include data indicating greater than anticipated ANS invasion rates, new technologies to minimize ANS invasions that were not available at the time of permit issuance, discharge concentrations or volumes, pollutant risks, compliance or lack thereof, and adverse affects to listed resources that were not considered in the BE. This information would also allow EPA to determine whether reinititation of formal ESA Section 7 consultation would be required as provided in 50 CFR 402.16, as described in the Fact Sheets.

VGP Requirements

The EPA will include the following language in the VGP Fact Sheet to minimize exposure and/or adverse affects to listed resources. The EPA requires vessel owners/operators to:

- Regularly inspect the fuel and hydraulic systems for any damages or leaks, for instance during fueling, when performing routine maintenance on the auxiliary vessel, and/or during deployments for testing.
- Avoid discharging ballast water in critical habitat when feasible. (http://www.nmfs.noaa.gov/pr/species/criticalhabitat.htm)
- Minimize hull husbandry in critical habitat for aquatic species. (http://www.nmfs.noaa.gov/pr/species/criticalhabitat.htm)
- Use soaps and detergents that are minimally toxic; EPA expects that soaps that are "minimally toxic" will contain little to no nonylphenols.
- Add the following questions added to the Annual Report:
  - When is your next scheduled drydocking? (This question is optional for inland vessels less than 300 gross tons and unmanned, unpowered barges)
  - When did you last conduct a below water (or drydock) hull inspection? (This question is optional for inland vessels less than 300 gross tons and unmanned, unpowered barges)
  - Do you use anti-foulant paint? If so, what type of anti-fouling hull coating? [Select specific product from drop-down menu]. When was it last applied?

sVGP Requirements

The EPA will include the following language in the sVGP Fact Sheet. The EPA requires vessel owners/operators to:

- Regularly inspect the fuel and hydraulic systems for any damages or leaks, for instance during fueling and/or when performing routine maintenance.
- Periodically inspect (at least once per quarter) the engine for any loose or leaking hoses, gaskets, and/or seals and if needed, repair or replace damaged parts as soon as possible.
- Check for the presence of a visible sheen surrounding waters periodically while discharging bilgewater.
- Visible portions of the vessel hulls must be periodically inspected (at least once per quarter), and if necessary, cleaned to prevent the spread or dispersal of potentially invasive species.
- Minimize the transport of any visible living organism from one waterbody to another by regularly cleaning and maintaining the hull. EPA proposes to add [the following] language to the Fact Sheet, with respect to [sVGP Section] 2.6 (g) Vessel Hull Maintenance: "In this context, EPA believes that to clean 'regularly' means to prevent substantial fouling of the vessel's hull. Regularly will vary from vessel to vessel, but cleaning should be done on a schedule to prevent the accumulation of potential ANS or other pollutants. By leaving regularly to be applied on a site-specific basis, vessel owner/operators can manage their operations efficiently, yet keep their hulls clean. If a vessel's hull is severely fouled, the cleaning has not occurred regularly. Hence, severe fouling could be [a] permit violation."
- (http://www.nmfs.noaa.gov/pr/species/criticalhabitat.htm)
- Avoid discharging ballast water in critical habitat when feasible (http://www.nmfs.noaa.gov/pr/species/criticalhabitat.htm)

- Use soaps and detergents that are minimally toxic; EPA expects that soaps that are "minimally toxic" will contain little to no nonylphenols"
- EPA will develop a discharge requirement checklist or other implementation tool for vessel owners/operators to use to assist in their quarterly self-inspections. This tool will be developed within one year of the final sVGP permit issuance (December 2013).

VGP Compliance

To monitor and insure compliance with its VGP, EPA will develop and implement a monitoring and compliance plan by December 2013 to assess the performance of the permit in protecting listed resources. Currently, the U.S. Coast Guard conducts all inspections of vessels covered under the VGP and uses an EPA/USCG jointly developed Job Aide, i.e., "Guidelines for Coast Guard Evaluations of Compliance with the US Environmental Protection Agency's (EPA) Vessel General Permit (VGP) for Discharges Incidental to the Normal Operation of Vessels." To insure that the inspections assess new requirements of the proposed VGP, EPA will:

- Work with the U.S. Coast Guard, with the early input of the Services, to update the Job Aide to reflect requirements included in the new VGP.
- Create an online training and/or a webinar training regarding the VGP requirements/prohibitions, within one year of final permit issuance.
- Create a comprehensive checklist of requirements for inspector training in conjunction with the U.S. EPA National Enforcement Training Institute, within one year of final permit issuance.
- Provide and continue to provide the U.S. Coast Guard with the title, office name, current address and phone number for points of contact at headquarters offices and in each EPA Region and USCG District.

In addition, EPA's Office of Enforcement and Compliance Assurance will meet with the Services to discuss enforcement under the VGP. The EPA will gather, review and analyze inspection data submitted by the Coast Guard; the results will be summarized (by type of vessel, location when available, type of deficiencies, and number of deficiencies) and provided to the Services annually.

sVGP Compliance

To monitor and insure compliance with its sVGP, EPA will:

- Develop and implement a monitoring and compliance plan within one year of permit issuance to assess the performance of the permit in protecting listed resources.
- Organize a meeting with their Office of Enforcement and Compliance Assurance for the Services to discuss and provide input to OECA's inspection and enforcement plans for the sVGP.
- Work with USCG, with the early input of the Services, to update the inspection and enforcement document inspection checklist (i.e., Job Aide) to reflect sVGP requirements

- Conduct outreach to the states to raise awareness about small vessel requirements; EPA will encourage States to check records of quarterly self-inspection records and visible portions of the hull while inspecting vessels when implementing their own programs.
- Develop trainings covering the small vessel requirements and share these with the Coast Guard and States

The EPA will provide the Services with a yearly summary of the number of inspections (by EPA, the Coast Guard, States or other sources) and results (including number of inspections by type of vessel, location when available, and number and type of deficiencies). During a Senior Manager's Meeting (October 19, 2012), EPA assured the Services that a subset of small vessels would be inspected.

Monitoring of Stressors (ANS and Pollutants)

Annually, EPA will summarize data from Annual Reports (which include all instances of noncompliance and unavoidable discharges authorized by the VGP) and Discharge Monitoring Reports and provide this summary to the Services. The Discharge Monitoring Reports include:

- Ballast water treatment system testing results, including results from:
  - Functionality monitoring (at least monthly)
  - Calibration tests (annually, at a minimum).
  - Biological organism monitoring (2 or 4 times annually, depending on system)
  - Residual biocides (After initial monitoring, 2 or 4 times annually, depending on system)
- Graywater from cruise ships (Quarterly, after initial monitoring)
  - BOD, fecal coliform, suspended solids, pH, and total residual chlorine
  - *E. coli*, total phosphorus, ammonia, nitrate/nitrite, and total Kjeldahl nitrogen
- Bilgewater (annual oil content monitoring, for new builds on or before December 19, 2013)

In addition, EPA has committed to the following:

Copper

Within 1 year of finalization of the VGPs, EPA will develop, in cooperation with the Services, and implement a monitoring and reporting plan for copper. The plan will contain the following elements:

1) EPA, with the technical assistance of the Services, will identify metrics for assessing copper loadings to waters inhabited by ESA-listed species (i.e., using the number of ships reporting copper hull treatments in eNOIs for particular areas as a proxy).
2) EPA will compile eNOI and annual report information regarding the number and type of vessels that use copper-based anti-fouling hull coating and/or exhaust gas scrubbers, their home port/most frequented US port, and US ports anticipated visiting during the permit term.
3) Twice, once by December 2014 (to summarize eNOI data) and another before EPA transmits the next draft VGP/sVGP to OMB for interagency review (to evaluate annual

reports in time to make changes for the next permit cycle, if need be), EPA will evaluate vessel data and identify ports/locations where the metric has been exceeded. If the metric has been exceeded, EPA will, as appropriate, identify and implement additional actions to reduce any unacceptable risks.

Potential Selenium Discharges in Exhaust Gas Scrubber (EGS)

Within 1 year of finalization of the VGPs, EPA will develop, in cooperation with the Services, and implement a monitoring and reporting plan for gas-scrubber discharges. The plan will contain the following elements:

1) EPA, with the technical assistance of the Services, will identify metrics for assessing selenium loadings to waters inhabited by ESA-listed species (i.e., using the number of ships reporting exhaust gas scrubbers in eNOIs for particular areas combined with expected and/or reported volume and concentration values as proxies).
2) EPA will state that vessel operators should analyze selenium in saline EGS discharges using methods capable of accurately measuring selenium in saline samples (e.g., Octopole Reaction Cell ICP-MS or Hydride generation AA methods or other methods which reduce matrix interference for selenium).
3) EPA will compile NOI and annual report information regarding the number and type of vessels that use exhaust gas scrubbers, discharge monitoring data for pollutants including selenium and copper, their home port/most frequented US port, and US ports anticipated visiting during the permit term.
4) Twice, once by December 2014 (to summarize eNOI data) and another before EPA transmits the next draft VGP/sVGP to OMB for interagency review (to evaluate annual reports in time to make changes for the next permit cycle, if need be), EPA will evaluate vessel data and identify ports/locations where the metric has been exceeded. If the metric has been exceeded, EPA will, as appropriate, identify and implement additional actions to reduce any unacceptable risk.

ANS

Within 1 year of finalization of the VGPs, EPA will develop, in cooperation with the Services, and implement a monitoring and reporting plan for vessel-mediated ANS invasions. The plan will contain the following elements:

1) EPA, with the technical assistance of the Services, will identify metrics to evaluate whether vessel mediated ANS invasion rates are being reduced over time.
2) EPA will track information from ANS invasions that might be potentially tied to vessel vectors. Information tracked may include:
    a. Dr. Greg Ruiz (Smithsonian Institution) conducts systematic surveys for ANS invasions caused by hull-fouling and ballast water in San Francisco, Chesapeake, and Tampa Bays to track changes in the rate of ANS invasions.
    b. Nationwide NEMESIS database for all newly documented, marine, ballast water and hull fouling ANS invasions. http://invasions.si.edu/nemesis/browseDB/searchQuery.jsp

c. National USGS NAS database for all newly documented, freshwater, shipping-related ANS invasions. http://nas.er.usgs.gov/queries/SpSearch.aspx

EPA and the Services acknowledge that new invasions observed from any of these sources are not necessarily linked to discharges authorized by the VGPs. Nonetheless, over a multi-permit lifespan, tracking invasions, and improving EPA's methodologies and understanding of the risk-release relationship, could provide a long-term perspective about ballast water discharge standards, antifouling technologies, and other practices to reduce invasions.

3) EPA will report to the Services once per year that they have reviewed this information and whether notable new invasions have been recorded in the previous year that could have been previously caused by vessel discharges. The Services note that they are requesting a short review (approximately 1 hour per year) to ensure that EPA continues to track aquatic bioinvasions. As discussed above, EPA and the Services agree that these observations are being used to inform EPA's general understanding of continued and ongoing invasion rates

If needed, EPA will identify and implement, as appropriate, additional actions (e.g., require anti-fouling coatings; require debris containment systems; require more stringent ballast water standards) as soon as practicable (including in future permit iterations) to ensure the protection of listed species.

AFCs and Debris Containment Technology

Before EPA transmits the next draft of the VGPs to OMB for interagency review:

1) EPA will compile information on new developments in AFCs (to minimize exposure to hull-fouling ANS and copper loading) and debris containment systems (to minimize the release of ANS and biocides during underwater husbandry).
2) EPA, with the technical assistance of the Services, will identify whether there are more efficacious BMPs or technologies for underwater husbandry and for anti-fouling coatings with fewer secondary environmental impacts that could be required of vessels.

Approach to the Assessment

Overview of NMFS' Assessment Framework

Section 7(a)(2) of the Endangered Species Act of 1973, as amended, requires federal agencies, in consultation with and with the assistance of the Services, to insure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of endangered or threatened species, and is not likely to destroy or adversely modify critical habitat that has been designated for these species (16 U.S.C. 1539). During consultations on specific actions, NMFS fulfills its obligations using an assessment framework that begins by identifying the physical, chemical, or biotic components of proposed actions that are likely to have individual, interactive, or cumulative direct and indirect effect on the environment (we use the term "potential stressors"

for these components of an action); we then determine whether listed species or designated critical habitat are likely to be exposed to those potential stressors; we estimate how listed species or designated critical habitat are likely to respond to any exposure; then we conclude by estimating the risks those responses pose to the individuals, populations, and species or designated critical habitat that are likely to be exposed.

General permits authorized by Federal agencies apply to activities over large geographic areas over long periods of time, with substantial uncertainty about the number, location, timing, frequency, and intensity of specific activities those programs authorize, fund, or carry out. Our traditional approaches to section 7 consultations, which focus on the specific effects of a specific proposal, are not designed to deal with the spatial and temporal scales and level of uncertainty that is typical of consultations on general permits.

Instead of trying to adapt traditional consultation approaches to programmatic consultations, we have developed an assessment framework that specifically allows us to help Federal agencies insure that their programs comply with the requirements of section 7(a)(2) of the ESA as described in the Interagency Endangered Species Consultation Handbook (U.S. Fish and Wildlife Service and NMFS 1998; Chapter 5). Our assessment framework for general permits first assesses whether the general permits are likely to adversely affect listed species or critical habitats by estimating exposure and response, as described above for traditional consultations. If listed species and critical habitats are likely to be adversely affected we then assess how the permit is structured to comply with section 7(a)(2).

Specifically, our programmatic consultations on general permits examine the decision-making processes that are integrated into Federal Agency programs to determine whether those decision-making processes are likely to insure that specific actions the agency authorizes, funds, or carries out through the program comply with the requirements of section 7(a)(2). That is, during programmatic consultations we ask whether or to what degree the Federal action agency (in this case, EPA) has structured its proposed general permit so that the agency (1) collects the information necessary to allow it to know or reliably estimate the probable individual and cumulative consequences of its actions on listed resources; (2) evaluates the information it collects to assess how its actions have affected endangered species, threatened species, and designated critical habitat specifically; and (3) use its authorities to bring those activities into compliance with the requirements of section 7(a)(2) of the ESA, if the information it collects suggests that it is necessary. Here, "permit structure" refers to the decision-making processes, applications of standards and criteria (including standards of information and treatment of uncertainty), feedback loops and internal audits, and controls (including permit conditions) that agencies rely upon to fulfill their obligations under the ESA.

Figure 1 displays a normative model of a decision-making process that includes these program elements and feedback loops. The process trigger on the left-hand side of the figure (Box D1) might represent an application from a prospective permittee or licensee, a request for prospective funding, or a prospective proposal to be undertaken by a Federal Agency. These process triggers are typically subjected to two screening processes (Box D2):

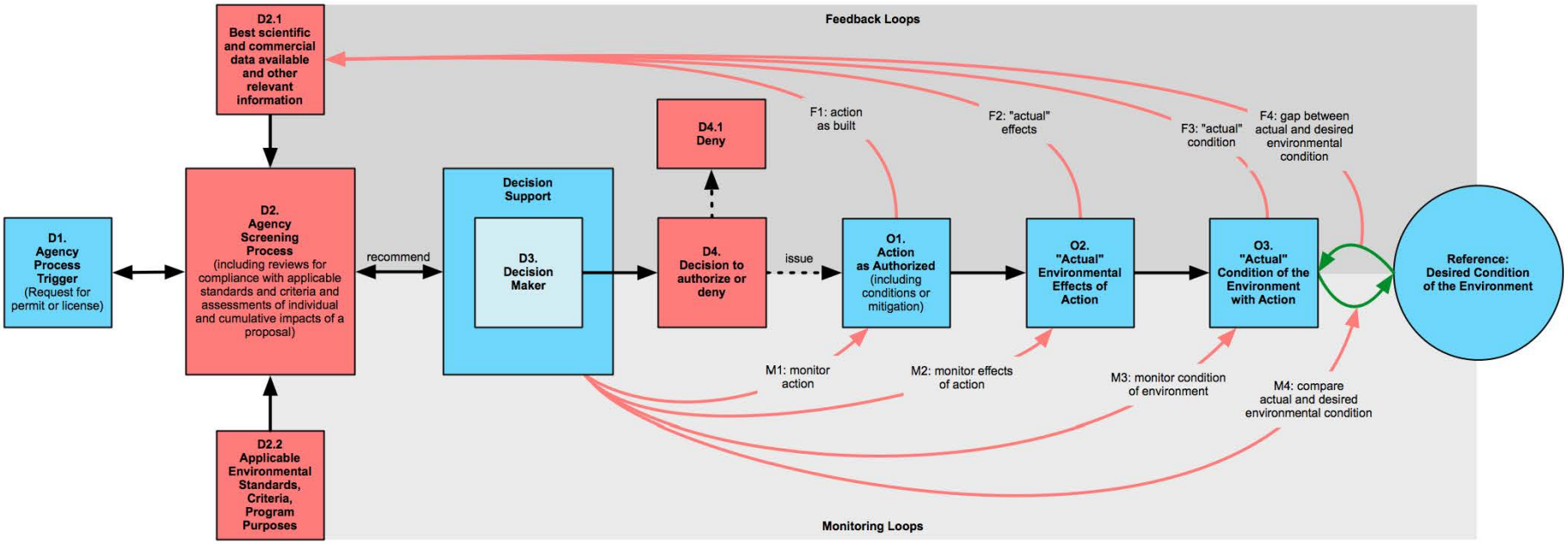
- An initial screening process that are designed to insure that proposals minimally comply with statutory, regulatory, or policy requirements that are applicable to requests for permits, licenses, or funding and
- Secondary screening processes that are designed to insure that an agency satisfies the statutory, regulatory, or policy requirements or criteria that must be met before an agency can issue a permit, license, or funding.

For example, the screening process the EPA applies to general permits includes reviews for completeness; analyses for compliance with Clean Water Act (CWA) guidelines; compliance with State water quality standards; compliance with toxic effluent standards; and compliance with the requirements of section 7(a)(2) of the ESA.

Agency screening processes typically produce recommendations to agency decision-makers, who have the authority to make final decisions on Agency Actions (see Figure 1, Box D3). Following those decisions, the Action Agency, permittee, licensee, or funding recipient undertakes the action, including any terms or conditions the Action Agency has attached (see Figure 1, Box O1). The action produces a set of direct, indirect, and cumulative effects on the environment and any living organisms that occur in or rely on the environment that is affected by the action (see Figure 1, Box O2) and the condition of the environment changes in response to those effects (see Figure 1, Box O3).

Figure 1 also displays a program that contains an audit function represented by a monitoring component (Lines M1 to M4), a feedback component (Lines F1 to F4), and an information gathering and evaluation component that informs a screening process (Box D2.1). The monitoring component would collect empirical information on individual actions or a sample of individual actions to (a) identify what action actually occurred for comparison with the action that had been proposed and approved (implementation monitoring; see Figure 1, Line M1); (b) identify which terms and conditions, if any, were satisfied, including any mitigation measures that were required (implementation monitoring; see Figure 1, Line M1), (c) gather empirical information on the action's direct and indirect effects on the environment, including the effectiveness of any mitigation measures that had been required (validation and compliance monitoring; see Figure 1, Line M2), (d) gather the empirical evidence to determine whether or to what degree the environment changed in response to those effects (see Figure 1, Line M3); and (e) gather the empirical evidence sufficient to determine whether a proposal contributed to environmental conditions that fail to meet program purposes and standards (compliance monitoring; see Figure 1, Line M4). The feedback component evaluates empirical data collected by monitoring and incorporates those data into agency decisions about prior or subsequent actions (see Figure 1, Lines F1 through F4).

Figure 1. A normative model of a decision-making process that includes these program elements and feedback loops. See text for further explanation



Regardless of whether an agency's decision-making processes corresponds to the model presented in Figure 1, five components of an agency's decision-making process are critical to our assessment of whether a general permit would be expected to insure that individual actions authorized, funded, or carried out by the program comply with the requirements of section 7(a)(2) of the ESA. The first critical component is the screening process an agency applies to specific actions authorized, funded, or carried out by a program. When we examine this component of an Agency's decision-making process, questions we ask include: What standards apply to the screening process? How rigorous are those standards? How rigorously does the Action Agency apply those standards? Are proposals assumed to comply with an Agency's statute barring evidence of non-compliance or vice versa? Which party (prospective permittees or the Action Agency) bears the responsibility for presenting the evidence that supports their position? Does an Agency's record of performance allow us to conclude that the screening process works as designed by filtering out proposals that do not satisfy applicable environmental mandates, standards, criteria, and program purposes?

The second critical component is the information that forms the foundation for the agency's screening process (see Figure 1, Boxes D1.1, D2.1, and D2.2). When we examine this component of an agency's decision-making process, questions we ask include: Does the agency assess the individual and cumulative impacts of specific proposals? Does the agency's methodology consider all of the variables that would have to be considered to determine whether a specific proposal is likely to have adverse consequence for endangered or threatened species and designated critical habitat? Do assessments employ data acquisition procedures that are likely to identify, gather, and analyze all of the information that would be relevant to identify the presence or absence of consequence for endangered or threatened species and designated critical habitat? Does the assessment process incorporate quality assurance and quality control procedures? Are those procedures designed to prevent Type I decision error (falsely concluding that a proposal had an adverse impact), Type II decision error (falsely concluding that a proposal had no adverse impact), or both?

The third critical component is an Action Agency's decision-making process, which includes the information and variables that inform the Agency's decision on whether or not to authorize, fund, or implement an action, the decisions the Agency makes, and any conditions or terms the Agency attaches to its decision.

The fourth critical component is an audit function. Does the Action Agency regularly or continuously audit the results of its actions? Are the monitoring and feedback loops (see Figure 1, lines M1 to M4 and F1 to F4) designed to allow the Agency to (a) collect empirical information that allows them to insure that specific actions they authorize, fund, or carry out are undertaken as designed (including any terms, conditions, or mitigation measures associated with the proposal), (b) assess the actual effects of those actions, and (c) determine whether the program is fulfilling its mandate, purposes, and goals.

The final critical component is the agency's authority to modify its prior and subsequent decisions when new information (particularly information provided by the audit function) reveals that particular authorizations have not satisfied applicable environmental mandates, standards, criteria, and program purposes (the applicable environmental mandates includes compliance with

section 7(a)(2) of the ESA).

We organize our programmatic consultations on general permits using a sequence of questions that focus on the agency's decision-making process, in general, and the five critical components we just described. Those questions focus on whether and to what degree an Agency has structured a program so that the Agency is in a position to know or reliably estimate whether endangered or threatened species or designated critical habitat are likely to be (a) exposed to stressors associated with specific actions a program would authorize, fund, or carry out; (b) respond to that exposure; and (c) experience individual-level, population-level, or species-level risks as a result of those responses. Further, we ask whether or to what degree an agency actively gathers that information, whether or to what degree an agency incorporates that information into its decision-making processes about specific actions, and whether or to what degree an agency changes the decisions it makes about specific actions based on that information.

Application of this Approach in this Consultation

As we have already discussed, we treat the issuance of the proposed VGPs as "programs" that would authorize discharges of pollutants over a five-year period. As we described earlier, during general permit consultations we ask whether or to what degree the programs:

- Collect or will have access to the information necessary to allow it to know how the actions it permits affect listed resources;
- Evaluate that information to assess how its actions have affected endangered species, threatened species, and designated critical habitat; and
- Uses its authorities to bring those activities into compliance.

Specific questions we ask about the proposed VGPs are:

1. Scope

Has the general permit been structured to reliably estimate the probable number, location and timing of the discharges that would be authorized by the program?

2. Stressors

Has the general permit been structured to reliably estimate the physical, chemical, or biotic stressors that are likely to be produced as a direct or indirect result of the discharges that would be authorized (that is, the stressors produced by the actual discharges to waters of the U.S.)?

3. Overlap

Has the general permit been structured to reliably estimate whether or to what degree specific endangered or threatened species or designated critical habitat are likely to be exposed to potentially harmful impacts that the proposed permit would authorize?

4. Monitoring/Feedback

Has the general permit been structured to identify, collect, and analyze information about authorized actions that may have exposed endangered or threatened species or designated critical habitat to stressors at concentrations, intensities, durations, or frequencies that are known or

suspected to produce physical, physiological, behavioral, or ecological responses that have potential individual or cumulative adverse consequences for individual organisms or constituent elements of critical habitat?

5. Responses of Listed Resources

Does the general permit have an analytical methodology that considers:

a) the status and trends of endangered or threatened species or designated critical habitat;
b) the demographic and ecological status of populations and individuals of those species given their exposure to pre-existing stressors in different drainages and watersheds;
c) the direct and indirect pathways by which endangered or threatened species or designated critical habitat might be exposed to the discharges to waters of the United States; and
d) the physical, physiological, behavior, sociobiological, and ecological consequences of exposing endangered or threatened species or designated critical habitat to stressors from discharges at concentrations, intensities, durations, or frequencies that could produce physical, physiological, behavioral, or ecological responses, given their pre-existing demographic and ecological condition?

6. Compliance

Does the general permit have a mechanism to reliably determine whether or to what degree operators have complied with the conditions, restrictions or mitigation measures the proposed permit requires when they discharge to waters of the United States?

7. Adequacy of Controls

Does the general permit have a mechanism to prevent or minimize endangered or threatened species or designated critical habitat from being exposed to stressors from discharges:

a) at concentrations, durations, or frequencies that are potentially harmful to individual listed organisms, populations, or the species, or;
b) to ecological consequences that are potentially harmful to individual listed organisms, populations, the species or Primary Constituent Elements of designated critical habitat?

As with all of our consultations, we conduct our section 7 analyses through a series of steps. These include: examination of the action; identification of the action area; exposure analyses; response analyses; analyses of aggregate impacts; and risk analyses. These steps are described below.

Proposed Action

In reviewing the proposed action and biological evaluation, the Services examine the activities that would be authorized by the proposed VGPs. This step of our analyses identifies spatial and temporal patterns associated with each category of activity; specifically (a) the geographic distribution of the different activities; (b) the number activities; (c) the amounts of pollutants that are likely to be discharged; and (d) the rate of discharges.

Our analysis evaluates the effects of all discharges of biological and chemical pollutants to waters of the US incidental to the normal operation of all eligible vessels that would be authorized by the VGPs. In addition, we consider the effects of interrelated and interdependent

actions of the proposed action. Interdependent actions are actions having no independent utility apart from the proposed action (50 CFR 402-02[1]). They are typically a consequence of the proposed action. For example, if our consultation were evaluating the effects of building a road, an interdependent action would be the planned construction of homes and other structures that would not be accessible without the presence of that road. Interrelated actions are actions that are part of a larger action and depend on the larger action for their justification (50 CFR 402-02[3]). They are actions that are typically associated with the proposed action.

In this consultation, NMFS addresses the potential effects of the proposed action for marine, coastal, estuarine, or anadromous endangered species, threatened species, and critical habitat that has been designated for those species in those ecosystems under its jurisdiction.

Action Area

We determine the degree of geographic and temporal overlap between the activities that would be authorized by the proposed VGPs and endangered and threatened species and designated critical habitat.

Exposure Analyses

Before assessing an agency's decision-making process (as described above), the NMFS first establishes whether the proposed action could expose listed species and critical habitat to potentially harmful stressors and whether listed species and critical habitats are likely to respond adversely. If exposure to stressors and adverse responses are not likely to occur, we do not assess the agency's decision-making process.

Our assessment focuses on whether and to what degree the EPA structured the VGPs in ways that would prevent or minimize endangered or threatened species or critical habitat that has been designated for those species from being exposed to harmful discharges of pollutants into waters of the United States and other harmful activities because such exposures commonly trigger responses that are difficult to prevent. For example, once individual plants and animals are exposed to a discharge containing pollutants, their responses to the exposure is controlled by the concentration, duration, and frequency associated with the exposure, their sensitivity to the discharged materials, other physical, chemical, or biotic stressors they are exposed to in the same time interval, their pre-existing physiological state, and their constitutional endowment. Because it is so difficult to prevent free-ranging organisms from responding to anthropogenic stressors once they have been exposed, the most effective management measures are designed to influence the exposure itself. For that reason, our assessment focuses on whether and to what degree the VGPs prevent or minimize endangered and threatened species and designated critical habitat from being exposed to harmful discharges and other harmful activities that would be authorized by the proposed VGPs. Rather than discuss the literature for each species, it would be only be necessary to discuss the risks of exposing species groups (for example, Pacific Salmon; Sturgeon; Sea Turtles; etc.).

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[1] See: http://cfr.vlex.com/vid/402-02-definitions-19895209

Response Analyses

We conduct a detailed review of the literature available on the physical, physiological, behavioral, social, and ecological responses of endangered or threatened species or constituent elements of critical habitat given exposure to harmful discharges into waters of the U.S. or to the effects of those discharges and the activities associated with those activities on the ecology of the watersheds in which they occur (that is, effects resulting from changes in populations of prey, predators, competitors, symbionts, etc.). Rather than discuss the literature for each species, we organize the data using species groups (e.g., Pacific salmon, sturgeon, sea turtles, etc.). We summarize the probable consequences of the identified responses for populations of endangered and threatened species and designated critical habitat.

Treatment of Aggregate Impacts

To address the question of whether the activities that would be authorized by the proposed VGPs have direct and indirect effects on listed resources that are sufficiently small both individually and in aggregate, we explicitly consider the aggregate impacts of the proposed permits in the Effects of the Action chapter of our Opinion. This usage of "aggregate impacts" is distinct from the term "cumulative effects" which the section 7 regulations defines as "those effects of future State or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation" (50 CFR 402.02). Aggregate impacts include: (1) time-crowded perturbations or perturbations that are so close in time that the effects of one perturbation do not dissipate before a subsequent perturbation occurs; (2) space-crowded perturbations or perturbations that are so close in space that their effects overlap; (3) interactions or perturbations that have qualitatively and quantitatively different consequences for the ecosystems, ecological communities, populations, or individuals exposed to them because of synergism (when stressors produce fundamentally different effects in combination than they do individually), additivity, magnification (when a combination of stressors have effects that are more than additive), or antagonism (when two or more stressors have less effect in combination than they do individually); and (4) nibbling or incremental and decremental effects are often, but not always, involved in each of the preceding three categories (NRC 1986).

Risk Analyses

We help Action Agencies determine whether or to what degree they have complied with the requirements of section 7(a)(2) of the ESA by assessing whether and to what degree an Agency has structured a program so that the Agency is in a position to know or reliably estimate: (a) whether endangered or threatened species are likely to be placed at increased risk of extinction, or (b) if those species avoid extinction, whether they are likely to experience increased risk of failing to recover from having been endangered or threatened because of the actions the program authorizes, funds, or carries out.

Our consideration of how well an Agency's program manages risks to endangered and threatened species reflects the relationship between a listed species, the populations it comprises, and the individuals those populations comprise. The continued existence of a species is determined by the fate of the populations it comprises and the continued existence of a population is determined by the fate of the individuals they comprise. Populations grow or

decline as the individuals they comprise live, die, grow, mature, migrate, and reproduce, or fail to do so. When we assess whether or to what degree an Agency's program is likely to eliminate or avoid risks to endangered or threatened species, we are mindful of the distinction between species, the populations they comprise, and the individuals comprised by those populations.

When we assess whether or to what degree an Agency's program is likely to eliminate or avoid risks to individual members of endangered or threatened species, we think in terms of the individuals' fitness, which integrates an individuals' longevity with its current and future reproductive success. In particular, we examine the scientific and commercial data available to determine if an individual's probable response to stressors produced by an Action would reasonably be expected to reduce the individual's current or expected future reproductive success by increasing an individual's likelihood of dying prematurely, increasing the age at which it becomes reproductively mature, reducing the age at which it stops reproducing, reducing the number of live births it produces during any reproductive bout, reducing the number of reproductive bouts it engages in over its reproductive lifespan (in animals that reproduce multiple times), or causing the individual's progeny to experience any of these phenomena (Brommer 2000, Brommer *et al*. 1998, 2002; Clutton-Brock 1998, Coulson *et al*. 2006, Kotiaho *et al*. 2005, McGraw and Caswell 1996, Newton and Rothery 1997, Oli and Dobson 2003, Roff 2002, Stearns 1992, Turchin 2003).

We evaluate whether individual members of an endangered or threatened species experience reductions in their current or expected future reproductive success or experience reductions in the rates at which they grow, mature, or become reproductively active. Then we determine if those reductions also reduce the abundance, reproduction rates, and growth rates (or increase variance in one or more of these rates) of the populations those individuals represent (see Stearns 1992). Similarly, we determine if those population-level reductions reduce the viability of the species those population(s) comprise. Our general permit assessments focus on whether or to what degree an Agency's program is likely to insure that the direct or indirect effects of actions the program would authorize are not likely to reduce the fitness of listed individuals to a degree that would be sufficient to reduce the viability of the population(s) those individuals represent and jeopardize the survival and recovery of the species.

Our consideration of how well an Agency's program manages risks to designated critical habitat focuses on the value of the physical, chemical, or biotic phenomena of the critical habitat for the conservation of the endangered and threatened species for which the critical habitat was designated. In this step of our assessment, we consider information about the contribution of constituent elements of critical habitat (or of the physical, chemical, or biotic features that give the designated area value for the conservation of listed species, particularly for older critical habitat designations that have no constituent elements) to the conservation value of those areas of critical habitat that occur in the action area. Then we consider the contribution of the conservation value of those areas to the conservation value of the entire critical habitat designation. Our general permit assessments focus on whether or to what degree an Agency's program is likely to insure that the direct or indirect effects of actions the program would authorize are not likely to reduce the value of the physical, chemical, or biotic phenomena of the critical habitat areas affected or are not likely to reduce that value of the physical, chemical, or

biotic phenomena to a degree that would be sufficient to reduce the value of the entire critical habitat for the conservation of the endangered and threatened species for which the critical habitat was designated.

Evidence Available for the Consultation

For all of the above analyses, we rely on two bodies of evidence for this consultation. We use the first body of evidence to determine whether or to what degree the proposed general permits can insure that vessel discharges on, over, or near waters of the U.S. are not likely to jeopardize the continued existence of endangered or threatened species or result in the destruction or adverse modification of critical habitat that has been designated for those species. To build this body of evidence, we search for, gather, and analyze published and unpublished sources that examine the effectiveness previous programs, and whether or to what degree the program has had: (1) adverse consequences for natural flora and fauna that have some dependency on the quality of waters of the United States, or (2) adverse consequences for species that have been listed as endangered or threatened. In particular, we consider information contained in EPA's BE, program reviews of effectiveness, compliance monitoring databases, risk assessments and ESA Section 7 consultations on similar actions, species status reviews, listing documents, recovery plans, reports on the status and trends of water quality, past and current research and population dynamics modeling. We supplement this information by searching for compliance data from available databases that track the number of inspections and enforcement actions over time.

We use a second body of evidence to assess the probable consequences of authorizing discharges of pollutants on, over, or near waters of the United States to endangered or threatened species or critical habitat that has been designated for those species. To assemble this body of information, we search peer-reviewed scientific literature, master's theses and doctoral dissertations, government reports and studies and reports from commercial vendors. For example, the National Library of Medicine's Hazardous Substance Data Bank, TOXNET, Toxline, EXTOXNET pesticide information profiles provide information on the toxicity of the pollutants. These searches will focus on identifying recent information on the biology, ecology, distribution, status, and trends of the threatened and endangered species considered in this consultation. We consider the results of these searches based on the quality of their study design, sample sizes and study results. Studies that relied on large sample sizes with small variances are generally ranked higher than studies that relied on small sample sizes or large variances.

Treatment of "Cumulative Impacts" (in the sense of NEPA)

To address the question of whether the activities that would be authorized by VGPs have direct and indirect effects on the environment that are small both individually and cumulatively, we explicitly consider the cumulative impacts of the proposed permits in a "cumulative impact" section of the Effects of the Action chapter of this Opinion. Here, we use the term "cumulative impact" in the NEPA sense of the term (the U.S. Council on Environmental Quality defines them under the term "cumulative effects" but we refer to them as "cumulative impacts" to distinguish between NEPA and ESA uses of "cumulative effects"). That is, we mean "cumulative impacts" as "the impact on the environment which results from the incremental impact of the action when

added to other past, present, and reasonably foreseeable future actions regardless of what agency (Federal or non-federal) or person undertakes such other actions" (40 CFR 1508.7).

The effects analyses of biological opinions considered the "impacts" on listed species and designated critical habitat that result from the incremental impact of an action by identifying natural and anthropogenic stressors that affect endangered and threatened species throughout their range (the *Status of the Species*) and within an Action Area (the *Environmental Baseline*, which articulate the pre-existing *impacts* of activities that occur in an Action Area, including the past, contemporaneous, and future *impacts* of those activities). We assess the effects of a proposed action by adding their direct and indirect effects to the *impacts* of the activities we identify in an *Environmental Baseline* (50 CFR 402.02), in light of the impacts of the status of the listed species and designated critical habitat throughout their range; as a result, the results of our effects analyses are equivalent to those contained in the "cumulative impact" sections of NEPA documents.

This usage of "cumulative impacts" is distinct from the term "cumulative effect." Section 7 regulations defines cumulative effects as "those effects of future State or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation" (50 CFR 402.02).

As we discussed previously, cumulative impacts includes (1) time-crowded perturbations or perturbations that are so close in time that the effects of one perturbation do not dissipate before a subsequent perturbation occurs; (2) space-crowded perturbations or perturbations that are so close in space that their effects overlap; (3) interactions or perturbations that have qualitatively and quantitatively different consequences for the ecosystems, ecological communities, populations, or individuals exposed to them because of synergism (when stressors produce fundamentally different effects in combination than they do individually), additivity, magnification (when a combination of stressors have effects that are more than additive), or antagonism (when two or more stressors have less effect in combination than they do individually); and (4) nibbling or incremental and decremental effects are often, but not always, involved in each of the preceding three categories (NRC 1986).

Action Area

The Action Area for this consultation consists of all waters of the U.S. in the United States, its territories, and its possessions (which includes American Samoa, Baker Island, Guam, Howland Island, Jarvis Island, Johnston Atoll, Midway Atoll, Navassa Island, the Commonwealth of the Northern Mariana Islands, Palmyra Atoll, Puerto Rico, the U.S. Virgin Islands, and Wake Island) where discharges incidental to the normal operation of all non-recreational and non-military vessels are to be authorized by the EPA's VGPs. Waters of the U.S. extend to the outer reach of the three mile territorial sea, defined in section 502(8) of the CWA as the belt of the seas measured from the line of ordinary low water along that portion of the coast which is in direct contact with the open sea and the line marking the seaward limit of inland waters, and extending

seaward a distance of three miles.

Waters of the U.S. (as defined in 40 CFR 122.2) are:

- All waters which are currently used, were used in the past, or may be susceptible to use in interstate or foreign commerce, including all waters which are subject to the ebb and flow of the tide;
- All interstate waters, including interstate "wetlands;"
- All other waters such as intrastate lakes, rivers, streams (including intermittent streams), mudflats, sandflats, "wetlands," sloughs, prairie potholes, wet meadows, playa lakes, or natural ponds the use, degradation, or destruction of which would affect or could affect interstate or foreign commerce including any such waters:
    - Which are or could be used by interstate or foreign travelers for recreational or other purposes;
    - From which fish or shellfish are or could be taken and sold in interstate or foreign commerce; or
    - Which are used or could be used for industrial purposes by industries in interstate commerce;
- All impoundments of waters otherwise defined as waters of the United States under this definition;
- Tributaries of those waters described above;
- The territorial sea; and
- "Wetlands" adjacent to waters (other than waters that are themselves wetlands).

The action area also includes other wetland or aquatic sites that do not meet the definition of "Waters of the U.S." and adjacent upland areas that may be affected by the introduction and spread of aquatic nuisance species.

Because NMFS only has jurisdiction over marine, coastal, estuarine, or anadromous endangered species, threatened species, and critical habitat that has been designated for those species in those ecosystems, this consultation addresses the potential effects of the proposed VGPs in a portion of this Action Area.

Status of the Species and Critical Habitat

In this section of this Opinion we describe the threatened and endangered species[2] and their designated critical habitat that occur in the action area and may be exposed to the direct or

[2] We use the word "species" as it has been defined in section 3 of the ESA, which include "species, subspecies, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature (16 U.S.C. 1533)". Pacific salmon that have been listed as endangered or threatened were listed as "evolutionarily significant units (ESU)" which NMFS uses to identify distinct population segments (DPS) of Pacific salmon. Any ESU or DPS is a "species" for the purposes of the ESA.

indirect effects of discharges authorized by the proposed VGPs. All listed species within NMFS' jurisdiction are "aquatic" or "aquatic dependent" and may occur within portions of the action area. NMFS has determined that the following species and critical habitat "may be affected" by EPA's proposed PGP (Table 1).

Table 1. Species Listed as Threatened, Endangered, and Proposed for Listing and their Designated Critical Habitat (Denoted by Asterisk) in the Action Area. Double Asterisks Denote Proposed Critical Habitat.

Common Name	Scientific Name	Status
Cetaceans		
Beluga whale (Cook Inlet*)	*Delphinapterus leucas*	Endangered
Blue whale	*Balaenoptera musculus*	Endangered
Bowhead whale	*Balaena mysticetus*	Endangered
False killer whale (Insular Hawaiian)	*Pseudorca crassidens*	Proposed Endangered
Fin whale	*Balaenoptera physalus*	Endangered
Humpback whale	*Megaptera novaeangliae*	Endangered
Killer Whale (Southern Resident *)	*Orcinus orca*	Endangered
North Atlantic right whale*	*Eubalaena glacialis*	Endangered
North Pacific right whale*	*Eubalaena japonica*	Endangered
Sei whale	*Balaenoptera borealis*	Endangered
Sperm whale	*Physeter macrocephalus*	Endangered
Pinnipeds		
Bearded seal (Beringia)	*Erignathus barbatus*	Proposed Threatened
Guadalupe fur seal	*Arctocephalus townsendi*	Threatened
Hawaiian monk seal* (also **)	*Monachus schauinslandi*	Endangered
Ringed seal (Arctic)	*Phoca hispida*	Proposed Threatened
Steller sea lion (Eastern population*)	*Eumetopias jubatus*	Threatened
Steller sea lion (Western population*)		Endangered
Marine Turtles		
Green sea turtle (Florida & Mexico's Pacific coast colonies*)	*Chelonia mydas*	Endangered
Green sea turtle (All other areas*)		Threatened
Hawksbill sea turtle*	*Eretmochelys imbricate*	Endangered
Kemp's ridley sea turtle	*Lepidochelys kempii*	Endangered
Leatherback sea turtle*	*Dermochelys coriacea*	Endangered
Loggerhead sea turtle (North Pacific)	*Caretta caretta*	Endangered
Loggerhead sea turtle (Northwest Atlantic Ocean)		Threatened
Olive ridley sea turtle (Mexico's Pacific coast breeding colonies)	*Lepidochelys olivacea*	Endangered
Olive ridley sea turtle (All other areas)		Threatened
Marine and Anadromous Fish		

Table 1. Species Listed as Threatened, Endangered, and Proposed for Listing and their Designated Critical Habitat (Denoted by Asterisk) in the Action Area. Double Asterisks Denote Proposed Critical Habitat.

Common Name	Scientific Name	Status
Atlantic salmon (Gulf of Maine)*	*Salmo salar*	Endangered
Bocaccio	*Sebastes paucispinis*	Endangered
Canary rockfish	*Sebastes pinniger*	Threatened
Pacific eulachon/smelt (Southern*)	*Thaleichthys pacificus*	Threatened
Yelloweye rockfish	*Sebastes ruberrimus*	Threatened
Chinook salmon (California coastal *)	*Oncorhynchus tshawytscha*	Threatened
Chinook salmon (Central Valley spring-run*)		Threatened
Chinook salmon (Lower Columbia River*)		Threatened
Chinook salmon (Upper Columbia River spring-run*)		Threatened
Chinook salmon (Puget Sound*)		Threatened
Chinook salmon (Sacramento River winter-run*)		Endangered
Chinook salmon (Snake River fall-run*)		Threatened
Chinook salmon (Snake River spring/summer-run*)		Threatened
Chinook salmon (Upper Willamette River*)		Threatened
Chum salmon (Columbia River*)	*Oncorhynchus keta*	Threatened
Chum salmon (Hood Canal summer-run*)		Threatened
Coho salmon (Central California coast*)	*Oncorhynchus kisutch*	Endangered
Coho salmon (Lower Columbia River)		Threatened
Coho salmon (Southern Oregon & Northern California coast*)		Threatened
Coho salmon (Oregon coast*)		Threatened
Sockeye salmon (Ozette Lake*)	*Oncorhynchus nerka*	Threatened
Sockeye salmon (Snake River*)		Endangered
Steelhead (Central California coast*)	*Oncorhynchus mykiss*	Threatened
Steelhead (California Central Valley*)		Threatened
Steelhead (Lower Columbia River*)		Threatened
Steelhead (Middle Columbia River*)		Threatened
Steelhead (Northern California*)		Threatened
Steelhead (Puget Sound)		Threatened
Steelhead (Snake River Basin*)		Threatened
Steelhead (South-Central California Coast*)		Threatened
Steelhead (Southern California*)		Endangered
Steelhead (Upper Columbia River*)		Threatened
Steelhead (Upper Willamette River*)		Threatened
Atlantic sturgeon (Gulf of Maine)	*Acipenser oxyrinchus oxyrinchus*	Threatened
Atlantic sturgeon (New York Bight)		Endangered
Atlantic sturgeon (Chesapeake Bay)		Endangered

Table 1. Species Listed as Threatened, Endangered, and Proposed for Listing and their Designated Critical Habitat (Denoted by Asterisk) in the Action Area. Double Asterisks Denote Proposed Critical Habitat.

Common Name	Scientific Name	Status
Atlantic sturgeon (Carolina)		Endangered
Atlantic sturgeon (South Atlantic)		Endangered
Green sturgeon (Southern*)	*Acipenser medirostris*	Threatened
Gulf sturgeon*	*Acipenser oxyrinchus desotoi*	Threatened
Shortnose sturgeon	*Acipenser brevirostrum*	Endangered
Smalltooth sawfish*	*Pristis pectinata*	Endangered
Marine Invertebrates		
Black abalone*	*Haliotis cracherodii*	Endangered
White abalone	*Haliotis sorenseni*	Endangered
Elkhorn coral*	*Acropora palmata*	Threatened
Staghorn coral*	*Acropora cervicornis*	Threatened
Marine Plant		
Johnson's seagrass*	*Halophila johnsonii*	Threatened

NMFS and the U.S. Fish and Wildlife Service have joint jurisdiction over sea turtles, gulf sturgeon and Atlantic salmon. To avoid redundancy, the U.S. Fish and Wildlife Service is generally responsible for endangered and threatened sea turtles above mean higher high water (when they are on their nesting beaches as opposed to when they are in or beyond the surf zone) and for Gulf sturgeon and Atlantic salmon when they are in fresh water (as opposed to when they are in estuarine or marine water).

Species and Critical Habitat Not Likely to be Adversely Affected by the Proposed Action

As described in the *Approach to the Assessment*, NMFS uses two criteria to identify those endangered or threatened species or critical habitat that are not likely to be adversely affected by the discharges that would be authorized by the proposed VGPs. The first criterion is *exposure* or some reasonable expectation of a co-occurrence between direct and indirect effects of the discharges authorized by the proposed permits and a particular listed species or designated critical habitat: if we conclude that a listed species or designated critical habitat is not likely to be exposed to those discharges or the resultant stressors, we must also conclude that the critical habitat is not likely to be adversely affected by those activities. The second criterion is the probability of a *response* given exposure, which considers *susceptibility*: species that may be exposed to a discharge, but are likely to be unaffected by the discharge (at levels they are likely to be exposed to) are also not likely to be adversely affected by that discharge. We applied these criteria to the species listed in Table 2; this subsection summarizes the results of those evaluations.

Based upon our analyses, we concluded that many of the endangered or threatened species listed in Table 2 are either not likely to be exposed to vessel discharges or are not likely to respond upon being exposed to those discharges. Specifically, we would not expect the following threatened or endangered species to respond physically, physiologically, behaviorally, or ecologically given exposure to discharges associated with the proposed VGPs:

Blue whales, bowhead whales, fin whales, North Pacific right whales, sei whales, sperm whales, false killer whales (insular Hawaiian), Guadalupe fur seals, Hawaiian monk seals, Steller sea lions, bearded seals (Beringia), and ringed seals (Arctic) all occur or have been reported to occur within the Action Area of the proposed VGPs. Nevertheless, these species are either not likely to be exposed to the direct or indirect effects of the discharges that would be authorized by the proposed VGPs, or they are not likely to respond given exposure.

Marine mammals would be primarily exposed to the aquatic nuisance species (ANS) and/or pollutants associated with discharges authorized by the VGPs through their diets or because of changes in the distribution or abundance of their prey (trophic exposure). Blue whales, bowhead whales, fin whales, North Pacific right whales, sei whales, sperm whales, false killer whales (insular Hawaiian), Guadalupe fur seals, Hawaiian monk seals, Steller sea lions, bearded seals (Beringia), and ringed seals (Arctic) are trophic generalists. The diet of baleen whales consists of zooplankton (e.g., krill, copepods, and other invertebrates), small schooling fish, and squid. Insular Hawaiian false killer whales eat fish and cephalopods. Sperm whales and Guadalupe fur seals forage on squid and other deep-water organisms. Steller sea lions, monk seals, Arctic ringed seals, and Beringia bearded seals feed on a variety of fish and invertebrates (including cephalopods and crustaceans). The abundance, variety, and location of the prey items identified above provide trophic resilience to these ESA-listed marine mammal.

Based upon review of available literature, we discount the possibility that pollutants of ANS from vessel discharges into waters of the U.S. would spread to offshore habitats. Inshore, estuarine, and freshwater habitats are fundamentally different in ecological terms and we discount the ability of an invasive species to establish in offshore regions. Therefore, pollutants and ANS are unlikely to result in the decline of prey species individually or in aggregate, and the above listed marine mammals are unlikely to be exposed to the indirect effects of the action. Based upon this, we discount the exposure of blue, fin, sei, sperm, North Pacific right, and Hawaiian insular false killer whales. For the same reasons, we also discount exposure to North Pacific right whale critical habitat.

Several listed or proposed resources also occur exclusively (except for extralimital occurrences) in areas where ballast water discharge is so limited that the probability of establishment of non-native species and subsequent exposure of listed resource is discountable. For some species, the isolated and/or offshore nature of the species was also a consideration. These include bowhead whales, Beringia bearded seals, and Arctic ringed seals. As a result, these species are not likely to be adversely affected by the proposed action, and we will not discuss them further in this Opinion.

Species and Critical Habitat Likely to be Adversely Affected by the Proposed Action

The species' narratives that follow focus on attributes of a species' life history and distribution that influence the manner and likelihood that a particular species may be exposed to the proposed action, as well as the species potential response and risk when exposure occurs. Subsequent narratives summarize a larger body of information on worldwide distribution, as well as localized movements within fresh water, estuarine, intertidal, and ocean waters, population structure, feeding, diving and social behaviors. Each species' narrative is followed by a description of its critical habitat (if applicable) with particular emphasis on any essential features of the habitat that may be exposed to the proposed action and may warrant special attention.

Anadromous Fish Species

Protective Regulations for Threatened Salmonid Species

Since 1997 NMFS promulgated a total of 29 limits to the ESA section 9(a) take prohibitions for 21 threatened Pacific salmon and steelhead Evolutionarily Significant Units (ESUs) or Distinct Populations Segments (DPSs) (62 FR 38479, July 18, 1997; 65 FR 42422, July 10, 2000; 65 FR 42485, July 10, 2000; 67 FR 1116, January 9, 2002; 73 FR 7816, February 11, 2008). On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160). NMFS took this action to provide appropriate flexibility to ensure that fisheries and artificial propagation programs are managed consistently with the conservation needs of threatened salmon and steelhead. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild.

Additionally, NMFS made several simplifying and clarifying changes to the 4(d) protective regulations including updating an expired limit (§ 223.203(b)(2)), providing a temporary exemption for ongoing research and enhancement activities, and applying the same set of 14 limits to all threatened Pacific salmon and steelhead ESUs or DPSs. These limits are:

1. Activities conducted in accordance with ESA section 10 incidental take authorization (50 CFR § 223.203(b)(1))
2. Ongoing scientific and conservation activities for which a permit application has been timely submitted, and treaty and non-treaty fisheries for which a comanager's management plan has been timely submitted (§ 223.203(b)(2))
3. Emergency actions related to injured, stranded, or dead salmonids (§ 223.203(b)(3))
4. Fishery management activities (§ 223.203(b)(4))
5. Hatchery and genetic management programs (§ 223.203(b)(5))
6. Activities in compliance with joint Tribal/State plans (§ 223.203(b)(6))
7. Scientific research activities conducted or permitted by the States (§ 223.203(b)(7))
8. State, local, and private habitat restoration activities (§ 223.203(b)(8))
9. Properly screened water diversion devices (§ 223.203(b)(9))
10. Routine road maintenance activities (§ 223.203(b)(10))

11. Certain park pest management activities (§ 223.203(b)(11))
12. Certain municipal, residential, commercial, and industrial development and redevelopment activities (§ 223.203(b)(12))
13. Forest management activities on State and private lands within the State of Washington (§ 223.203(b)(13))
14. Activities undertaken consistent with an approved Tribal resource management plan (§ 223.204).

Chinook Salmon

Description of the Species
Chinook salmon are the largest of the Pacific salmon and historically ranged from the Ventura River in California to Point Hope, Alaska in North America, and in northeastern Asia from Hokkaido, Japan to the Anadyr River in Russia (Healey 1991). We discuss the distribution, status, and critical habitats of the nine species[3] of endangered and threatened Chinook salmon separately, and summarize their common dependence on Waters of the U.S. However, because listed Chinook salmon species are virtually indistinguishable in the wild and comprise the same biological species, we begin this section describing those characteristics common across ESUs.

Chinook salmon exhibit one of the most varied and complex life history strategies. The "stream-type" of Chinook salmon resides in freshwater for a year or more following emergence and the "ocean-type" migrates to the ocean within their first year. The ocean-type typifies populations north of 56ºN (Healey 1991). The life cycle of all Chinook salmon spans fresh and marine waters. Spawning migrations generally occur in the spring and fall; spawning typically occurs earlier in the spring/summer at northern latitudes and later in southern latitudes (Healey 1991). Temperature and stream flow can significantly influence the timing of migrations and spawning, as well as the selection of spawning habitat (Geist et al. 2009, Hatten and Tiffan. 2009). Chinook salmon are semelparous (i.e. they die after spawning).

While in fresh water, juvenile Chinook salmon are often found in the lower reaches of a river near its estuary in areas of low water velocity. As they grow, they tend to move to deeper waters where the velocity is higher (Healey 1991). Generally, Chinook salmon outmigrants (smolts) are about 2 to 5 inches long when they enter saline (often brackish) waters. The process of smoltification enables salmon to adapt to the ocean environment (Wedemeyer et al. 1980). Several factors can affect smoltification process, not only at the interface between fresh water and salt water, but higher in the watershed as the process of transformation begins long before fish enter salt waters. These factors include exposure to chemicals such as heavy metals and elevated water temperatures (Wedemeyer et al. 1980).

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[3] We use the word "species" as it has been defined in section 3 of the ESA, which include "species, subspecies, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature (16 U.S.C. 1533)". Pacific salmon that have been listed as endangered or threatened were listed as "evolutionarily significant units (ESU)" which NMFS uses to identify distinct population segments (DPS) of Pacific salmon. Any ESU or DPS is a "species" for the purposes of the ESA.

Chinook salmon feed on a variety of prey organisms depending upon life stage. Adult oceanic Chinook salmon eat small fish, amphipods, and crab megalops (Healey 1991). Fish, in particular herring, make up the largest portion of an adult Chinook salmon's diet. In estuaries, Chinook salmon smolts tend to feed on chironomid larvae and pupae, *Daphnia*, *Eogammarus*, *Corphium* and *Neomysis,* as well as juvenile herring, sticklebacks and other small fish. In fresh water, Chinook salmon juveniles feed on terrestrial and aquatic insects and their larvae, including: dipterans, beetles, stoneflies, chironomids, and plecopterans (Healey 1991).

Threats

Natural Threats. Chinook salmon are prey for pelagic fishes, birds, and marine mammals, including harbor seals, sea lions, and killer whales. There have been recent concerns that the increasing size of tern, seal, and sea lion populations in the Pacific Northwest may have reduced the survival of some salmon species.

Anthropogenic Threats. Salmon survive only in aquatic ecosystems and, therefore, depend on the quantity and quality of those ecosystems. Salmon along the west coast of the U.S. share many of the same threats. Therefore, anthropogenic threats for all species and populations are summarized here. Salmon have declined under the combined effects of multiple anthropogenic stressors. The main drivers of the decline are known as the four "H"s: habitat loss, hatcheries, hydropower, and harvest. Examples of these include fishery over-harvest, competition from hatchery fish and non-native species, the effects of dams, water diversions, destruction or degradation of riparian habitat, and land use practices that destroy or degrade wetland and riparian ecosystems (Buhle et al. 2009).

Population declines have resulted from several human-mediated causes, but the greatest negative influence has likely been the establishment of waterway obstructions such as dams, power plants and sluiceways for hydropower, agriculture, flood control and water storage. These structures have blocked salmon migration to spawning habitat or resulted in direct mortality and have eliminated entire salmon runs as a result. While some of these barriers remain, others have been reengineered, renovated or removed to allow for surviving runs to access former habitat, but success has been limited. These types of barriers alter the natural hydrograph of basins, both upstream and downstream of the structure, and significantly reduce the availability and quality of spawning and rearing habitat (Hatten and Tiffan. 2009). Many streams and rivers, particularly in urban or suburban areas, suffer from streamside development, which contributes sediment, chemical pollutants from pesticide applications and automobile or industrial activities, altered stream flows, loss of streamside vegetation and allochthonous materials to name a few. These factors can directly cause mortality, reduce reproductive success or affect the health and fitness of all salmon life stages.

Changes in hydrological regimes are closely linked to salmon abundance (Hicks et al. 1991). From studies that have examined the effects of changes in land use patterns, we know that changes in hydrology can profoundly affect salmon abundance and the amount and availability of quality habitat. Hydrology is strongly correlated to early survival and can lead to the displacement of young fish as well as altering immigration and emigration timing which impacts the relative abundance of salmon within a watershed, as well as the relative abundance of age-classes (Hicks et al. 1991, Gregory and Bisson 1997). Such ecosystem changes are also likely to

alter macroinvertebrate communities and habitats, affecting the forage base for salmon and trout (McCarthy et al. 2009, Williams et al. 2009).

Fishing pressure has also negatively impacted salmon populations. Fishing reduces the number of individuals within a population and can lead to uneven exploitation of certain populations and size classes (Reisenbichler 1997). Targeted fishing of larger individuals results in excluding the most fecund individuals from spawning (Reisenbichler 1997). Genetic changes that promote smaller body sizes have occurred in heavily exploited populations in response to size-selective harvest pressures (Reisenbichler 1997). Fishing pressure can reduce age at maturity in fished populations as the fished populations compensate for the reductions in the numbers of spawning adults (Reisenbichler 1997).

Each year hatcheries along the west coast of the United States release nearly 1.2 billion juvenile salmon (Mahnken et al. 1998), with 200 million salmon released annually into the Columbia River alone. Hatcheries have the potential to reduce the viability of natural salmon populations through behavioral or reproductive incompatibility, introgression, and the alteration of run times (Ruckelshaus et al. 2002). These potential risks are not trivial; in chinook populations where hatchery fish are marked, escaped hatchery salmon can constitute up to 60% of the spawning population in areas without planned supplementation programs (Ruckelshaus et al. 2002).

Aquatic nuisance species (ANS), also described as non-native or invasive species, adversely affect listed salmon species through several mechanisms, including: predation, competition, trophic structure alteration, introgression, and transfer of pathogens. ANS pose as great or greater threat to the continued existence of salmonid species as the four Hs (Sanderson et al. 2009). 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 Chinook salmon (Haskell et al. 2006). The presence of brook trout in the Columbia River Basin is associated with a 12% reduction in the survival of juvenile salmon (Levin et al. 2002). Non-native crustaceans, mollusks, and plants pose significant risks to salmonids and the function of their ecosystems. For example, the invasive New Zealand mud snail has been detected in the diet of juvenile Columbia River Chinook salmon, indicating the potential for a shift in estuarine food web structure (Bersine et al. 2008). Non-native quagga and zebra mussel invasions in the eastern U.S. have resulted in competition with native mussels, disruption of food webs, and bioaccumulation of toxins; similar threats are expected if these species invade western waterways (Sanderson et al. 2009). Aquatic plants, such as purple loosestrife and Eurasian water milfoil, have been introduced to the Pacific Northwest through ballast water. These rapidly decomposing plants have the potential to alter ecosystem function through changes in seasonal nutrient availability and depressed dissolved oxygen concentrations (Unmuth et al. 2000, Blossey et al. 2001, Cronin et al. 2006). Climate change is likely to facilitate the establishment and expansion of ANS. In summary, non-native species have adversely affected salmonid species, primarily through predation and competition; however, they also have the potential, through the mechanisms listed above, to equal or exceed the impacts caused by overharvest, habitat loss, hatcheries, and threats to the hydrosystem (Ruckelshaus et al. 2002, Sanderson et al. 2009).

Pacific salmon species are exposed to a number of contaminants throughout their range and life

history cycle. Exposure to pollution is also of significant concern for all life stages, but is likely particularly significant for freshwater life stages. Organic pollutants, particularly PCBs, DDT and its congeners, pesticides, and endocrine disruptors are of particular concern. These chemicals can inhibit smell, disrupt reproductive behavior and physiology, impair immune function, and lead to mortality through impairment of water balance when traveling between fresh and salt water systems (Varanasi et al. 1993). Diffuse and extensive population centers contribute increase contaminant volumes and variety from such sources as wastewater treatment plants and sprawling development. Urban runoff from impervious surfaces and roadways often contains oil, copper, pesticides, PAHs, and other chemical pollutants and flow into surface waters. Point and nonpoint pollution sources entering rivers and their tributaries affect water quality in available spawning and rearing habitat for salmon. Juvenile salmonids that inhabit urban watersheds often carry high contaminant burdens, which is partly attributable to the biological transfer of contaminants through the food web (Varanasi et al. 1993).

California Coastal Chinook Salmon

Distribution and Description of the Listed Species
The California Coastal Chinook salmon ESU includes all naturally spawned populations of Chinook salmon from rivers and streams south of the Klamath River to the Russian River, California. Seven artificial propagation programs are part of this ESU. California Coastal Chinook salmon are a fall-run, ocean-type fish. A spring-run (river-type) component existed historically, but is now considered extinct (Bjorkstedt et al. 2005).

Status and Trends
NMFS listed California Coastal Chinook salmon as threatened on September 16, 1999 (64 FR 50393), and they retained their threatened status on June 28, 2005 (70 FR 37160). California Coastal Chinook salmon were listed due to the combined effect of dams that prevent them from reaching spawning habitat, logging, agricultural activities, urbanization, and water withdrawals in the river drainages that support them. Historical estimates of escapement, based on professional opinion and evaluation of habitat conditions, suggest abundance was roughly 73,000 in the early 1960s with the majority of fish spawning in the Eel River (Good et al. 2005). Since its original listing and status review, little new data are available or suitable for analyzing trends or estimating changes in this population's growth rate (Good et al. 2005). Long-term trends in Sprowl and Tomki creeks (tributaries of the Eel River), however, are negative. Good *et al.,* (2005) caution making inferences on the basin-wide status of these populations as they may be weak because the data likely include unquantified variability due to flow-related changes in spawners' use of mainstem and tributary habitats.

Critical Habitat
NMFS designated critical habitat for California Coastal Chinook salmon on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Redwood Creek, Trinidad, Mad River, Eureka Plain, Eel River, Cape Mendocino, Mendocino Coast and the Russian River. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to

support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat 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. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, California Coastal Chinook salmon occupy 45 watersheds (freshwater and estuarine). The total area of habitat designated as critical includes about 1,500 miles of stream habitat and about 25 square miles of estuarine habitat, mostly within Humboldt Bay. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 45 watershed reviewed in NMFS' assessment of critical habitat for California Coastal Chinook salmon, eight watersheds received a low rating of conservation value, 10 received a medium rating, and 27 received a high rating of conservation value for the species.

Critical habitat in this ESU consists of limited quantity and quality summer and winter rearing habitat, as well as marginal spawning habitat. Compared to historical conditions, there are fewer pools, limited cover, and reduced habitat complexity. The limited instream cover that does exist is provided mainly by large cobble and overhanging vegetation. Instream large woody debris, needed for foraging sites, cover and velocity refuges is especially lacking in most of the streams throughout the basin. NMFS has determined that these degraded habitat conditions are, in part, the result of many human-induced factors including dam construction, agricultural and mining activities, urbanization, stream channelization, water diversion and logging.

Central Valley Spring-Run Chinook Salmon

Distribution and Description of the Listed Species
The Central Valley spring-run Chinook salmon ESU includes all naturally spawned populations of spring-run Chinook salmon in the Sacramento River and its tributaries in California. This ESU includes one artificial propagation program. Central Valley spring-run Chinook salmon ESU includes Chinook salmon entering the Sacramento River from March to July and spawning from late August through early October, with a peak in September. Spring-run fish in the Sacramento River exhibit an ocean-type life history, emigrating as fry, sub-yearlings and yearlings. Central Valley spring-run Chinook salmon require cool freshwater while they mature over the summer.

Status and Trends
NMFS originally listed Central Valley spring-run Chinook salmon as threatened on September 16, 1999 (64 FR 50393), a classification this species retained on June 28, 2005 (70 FR 37160).

This species was listed because dams isolate them from most of their historic spawning habitat and the habitat remaining to them is degraded. Historically, spring-run Chinook salmon were predominant throughout the Central Valley occupying the upper and middle reaches (1,000 to 6,000 feet) of the San Joaquin, American, Yuba, Feather, Sacramento, McCloud and Pit Rivers, with smaller populations in most tributaries with sufficient habitat for over-summering adults (Stone 1874, Clark 1929).

The Central Valley drainage as a whole is estimated to have supported spring-run Chinook salmon runs as large as 700,000 fish between the late 1880s and the 1940s (Fisher 1994), although these estimates may reflect an already declining population, in part from the commercial gillnet fishery that occurred for this ESU (Good et al. 2005). Before construction of Friant Dam, nearly 50,000 adults were counted in the San Joaquin River alone (Fry 1961). Following the completion of Friant Dam, the native population from the San Joaquin River and its tributaries (i.e., the Stanislaus and Mokelumne Rivers) was extirpated. Spring-run Chinook salmon no longer exist in the American River due to the operation of Folsom Dam. Naturally spawning populations of Central Valley spring-run Chinook salmon currently are restricted to accessible reaches of the upper Sacramento River, Antelope Creek, Battle Creek, Beegum Creek, Big Chico Creek, Butte Creek, Clear Creek, Deer Creek, Feather River, Mill Creek, and Yuba River (CDFG 1998). Since 1969, the Central Valley spring-run Chinook salmon ESU (excluding Feather River fish) has displayed broad fluctuations in abundance ranging from 25,890 in 1982 to 1,403 in 1993 (Good et al. 2005).

As noted by Good *et al.,* (2005), the average abundance for the ESU was 12,499 for the period of 1969 to 1979, 12,981 for the period of 1980 to 1990, and 6,542 for the period of 1991 to 2001. In 2003 and 2004, total run size for the ESU was 8,775 and 9,872 adults respectively, well above the 1991 to 2001 average. Evaluating the ESU as a whole, however, masks significant changes that are occurring among populations that comprise the ESU (metapopulation). For example, the mainstem Sacramento River population has undergone a significant decline while the abundance of many tributary populations increased. Average abundance of Sacramento River mainstem spring-run Chinook salmon recently declined from a high of 12,107 for the period 1980 to 1990, to a low of 609 for the period 1991 to 2001, while the average abundance of Sacramento River tributary populations increased from a low of 1,227 to a high of 5,925 over the same periods.

Abundance time series data for Mill, Deer, Butte, and Big Chico creeks spring-run Chinook salmon confirm that population increases seen in the 1990s have continued through 2001(Good et al. 2005). Habitat improvements, including the removal of several small dams and increases in summer flows in the watersheds, reduced ocean fisheries, and a favorable terrestrial and marine climate, have likely contributed to this. All three spring-run Chinook salmon populations in the Central Valley have long-and short-term positive population growth.

Critical Habitat

NMFS designated critical habitat for Central Valley spring-run Chinook salmon on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Tehama, Whitmore, Redding, Eastern Tehama, Sacramento Delta, Valley-Putah-Cache, Marysville, Yuba, Valley-American, Colusa Basin, Butte Creek and Shasta Bally hydrological units. These areas are important for the species' overall conservation by protecting

quality growth, reproduction, and feeding. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat 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. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, Central Valley spring-run Chinook salmon occupy 37 watersheds (freshwater and estuarine). The total area of habitat designated as critical includes about 1,100 miles of stream habitat and about 250 square miles of estuarine habitat in the San Francisco-San Pablo-Suisun Bay complex. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 37 watersheds reviewed in NMFS' assessment of critical habitat for Central Valley spring-run Chinook salmon, seven watersheds received a low rating of conservation value, three received a medium rating, and 27 received a high rating of conservation value for the species.

Factors contributing to the downward trends in this ESU include: reduced access to spawning/rearing habitat behind impassable dams, climatic variation, water management activities, hybridization with fall-run Chinook salmon, predation, and harvest (CDFG 1998). Several actions have been taken to improve and increase the primary constituent elements of critical habitat for spring-run Chinook salmon. These include improved management of Central Valley water, implementing new and improved screen and ladder designs at major water diversions along the mainstem Sacramento River and tributaries, removal of several small dams on important spring-run Chinook salmon spawning streams, and changes in ocean and inland fishing regulations to minimize harvest. Although protective measures and critical habitat restoration likely have contributed to recent increases in spring-run Chinook salmon abundance, the ESU is still below levels observed from the 1960s through 1990. Many threats still exist.

Final Protective Regulations

On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the Central Valley spring-run Chinook salmon ESU.

Lower Columbia River Chinook Salmon

Distribution and Description of the Listed Species
The Lower Columbia River Chinook salmon ESU includes all naturally spawned populations of Chinook salmon from the Columbia River and its tributaries from its mouth at the Pacific Ocean upstream to a transitional point between Washington and Oregon, east of the Hood River and the White Salmon River, and includes the Willamette River to Willamette Falls, Oregon, exclusive of spring-run Chinook salmon in the Clackamas River.

Lower Columbia River Chinook salmon have three life history types, including early fall runs (tules), late fall runs (brights), and spring-runs. Spring and fall runs have been designated as part of a Lower Columbia River Chinook salmon ESU. The Cowlitz, Kalama, Lewis, White Salmon and Klickitat Rivers are the major river systems on the Washington side, and the lower Willamette and Sandy Rivers are foremost on the Oregon side. The eastern boundary for this species occurs at Celilo Falls, which corresponds to the edge of the drier Columbia Basin Ecosystem and historically may have been a barrier to salmon migration at certain times of the year. Fall Chinook salmon typically enter the Columbia River in August through October to spawn in the mainstem of the large rivers (Kostow 1995). Spring Chinook salmon enter freshwater in March through June to spawn in upstream tributaries and generally emigrate from fresh water as yearlings.

Status and Trends
NMFS originally listed Lower Columbia River Chinook salmon as threatened on March 24, 1999 (64 FR 14308); NMFS reaffirmed the threatened status of Lower Columbia River Chinook salmon on June 28, 2005 (70 FR 37160). Historical records of Chinook salmon abundance are sparse, but cannery records suggest a peak run of 4.6 million fish (43 million pounds) in 1883 (Lichatowich 1999). Although fall-run Chinook salmon are still present throughout much of their historical range, they are still subject to introgression by large-scale hatchery production, relatively high harvest, and extensive habitat degradation. The Lewis River late-fall-run Chinook salmon population is the healthiest and has a reasonable probability of being self-sustaining. Abundances largely declined during 1998 to 2000 and trend indicators for most populations are negative, especially if hatchery fish are assumed to have a reproductive success equivalent to that of natural-origin fish (Good et al. 2005). Most populations for which data are available have a long-term declining population trend (Good et al. 2005).

Critical Habitat
NMFS designated critical habitat for Lower Columbia River Chinook salmon on September 2, 2005 (70 FR 52630). Designated critical habitat includes all Columbia River estuarine areas and river reaches proceeding upstream to the confluence with the Hood Rivers as well as specific stream reaches in a number of tributary subbasins. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat 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. Of 52 subbasins reviewed in NMFS' assessment of critical habitat for the Lower Columbia River Chinook salmon ESU, 13 subbasins were rated as having a medium conservation value, four were rated as low, and the remaining subbasins (35), were rated as having a high conservation value to Lower Columbia River Chinook salmon. Factors contributing to the downward trends in this ESU are hydromorphological changes resulting from hydropower development, loss of tidal marsh and swamp habitat, and degraded freshwater and marine habitat from industrial harbor and port development, and urban development. Limiting factors identified for this species include reduced access to spawning/rearing habitat in tributaries, hatchery impacts, loss of habitat diversity and channel stability in tributaries, excessive fine sediment in spawning gravels, elevated water temperature in tributaries, and harvest impacts.

Final Protective Regulations
On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the Lower Columbia River Chinook salmon ESU.

Upper Columbia River Spring-run Chinook Salmon

Distribution and Description of the Listed Species
The Upper Columbia River spring-run Chinook salmon ESU includes all naturally spawned populations of Chinook salmon in all river reaches accessible to Chinook salmon in Columbia River tributaries upstream of Rock Island Dam and downstream of Chief Joseph Dam in Washington, excluding the Okanogan River. Six artificial propagation programs are part of this ESU. Spring-run Chinook salmon currently spawn in only three river basins above Rock Island Dam: the Wenatchee, Entiat, and Methow Rivers (Good et al. 2005).

Status and Trends
NMFS listed Upper Columbia River spring-run Chinook salmon as endangered on March 24, 1999 (64 FR 14308), and reaffirmed their status as endangered on June 28, 2005 (70 FR 37160), because they had been reduced to small populations in three watersheds. Based on redd count data series, spawning escapements for the Wenatchee, Entiat, and Methow rivers have declined an average of 5.6%, 4.8%, and 6.3% per year, respectively, since 1958. In the most recent 5-year geometric mean (1997 to 2001), spawning escapement for naturally produced fish was 273 for the Wenatchee population, 65 for the Entiat population, and 282 for the Methow population, only 8% to 15% of the minimum abundance thresholds, although escapement increased substantially in 2000 and 2001 in all three river systems. Based on 1980-2004 returns, the average annual growth rate for this ESU is estimated at 0.93 (meaning the population is not replacing itself; Fisher and Hinrichsen 2006). Assuming that population growth rates were to continue at 1980 to 2004 levels, Upper Columbia River spring-run Chinook salmon populations are projected to

have very high probabilities of decline within 50 years. Population viability analyses for this species (using the Dennis Model) suggest that these Chinook salmon face a significant risk of extinction: a 75 to 100% probability of extinction within 100 years (given return rates for 1980 to present).

Hatchery influence and genetic diversity are significant issues for the continued survival of Upper Columbia River Chinook salmon. This is a result of reduced genetic diversity from homogenization of populations that occurred under the Grand Coulee Fish Maintenance Project from 1939 to 1943. Stray hatchery fish and a high proportion of hatchery fish during spawning have contributed to the loss of genetic diversity.

Critical Habitat
NMFS designated critical habitat for Upper Columbia River spring-run Chinook salmon on September 2, 2005 (70 FR 52630). The designation includes all Columbia River estuaries and river reaches upstream to Chief Joseph Dam and several tributary subbasins. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat 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. The Upper Columbia River spring-run Chinook salmon ESU has 31 watersheds within its range. Five watersheds received a medium rating and 26 received a high rating of conservation value to the ESU. The Columbia River rearing/migration corridor downstream of the spawning range was rated as a high conservation value. Factors contributing to the downward trends in this ESU include mainstem Columbia River hydropower system mortality, tributary riparian degradation and loss of in-river wood, altered tributary floodplain and channel morphology, reduced tributary stream flow and impaired passage, and harvest impacts.

Puget Sound Chinook Salmon

Distribution and Description of the Listed Species
The Puget Sound Chinook salmon ESU includes all naturally spawned populations of Chinook salmon from rivers and streams flowing into Puget Sound including the Straits of Juan De Fuca from the Elwha River, eastward, including rivers and streams flowing into Hood Canal, South Sound, North Sound and the Strait of Georgia in Washington. Twenty-six artificial propagation programs are part of the ESU. The Puget Sound ESU is comprised of 31 historical populations, of which 22 or more are believed to be extant and nine are considered extinct.

Chinook salmon in this area generally have an "ocean-type" life history. Puget Sound populations include both early-returning and late-returning Chinook salmon spawners described by Healey (1991). However, within these generalized behavioral forms, significant variation

occurs in residence time in fresh water and estuarine environments. For example, Hayman *et al.,* (1996) described three juvenile Chinook salmon life histories with varying residency times in the Skagit River system in northern Puget Sound. Chinook salmon utilize nearshore Puget Sound habitats year-round, although they can be far from their natal river systems (Brennan et al. 2004).

Status and Trends

NMFS listed Puget Sound Chinook salmon as threatened in 1999 (64 FR 14308); that status was reaffirmed on June 28, 2005 (70 FR 37160). This ESU has lost 15 spawning aggregations (nine from the early-run type) that were either independent historical populations or major components of the remaining 22 existing independent historical populations identified (Good et al. 2005). The disproportionate loss of early-run life history diversity represents a significant loss of the evolutionary legacy of the historical ESU.

Data reported by Good *et al.,* (2005) indicate that long term trends in abundance for this ESU are split with about half of the populations declining, and the other half increasing. In contrast, the short-term trend for four populations is declining. The overall long-term trend in abundance indicates that, on average, populations are just replacing themselves. Estimates of the short-term median population growth rate (λ) (data years 1990-2002) indicate an even split between populations that are growing and those that are declining, although estimates would be lower for several populations if the fraction of naturally spawning hatchery fish were available for all populations within the ESU. For available data, when λ is calculated assuming that hatchery fish have the equivalent success of natural spawners then the largest estimated decline occurs in the Green River. Populations with the largest positive short and long-term trends include the White River and the North Fork Nooksack River (Good et al. 2005). Lambda for the Skagit River, which produces the most Chinook salmon in this ESU, has increased slightly. Overall, the recent analysis by Good *et al.,* (2005) illustrated that there has not be much change in this ESU since NMFS' first status review (Busby et al. 1996). Individual populations have improved, while others have declined. However, the lack of information on the fraction of naturally spawning, hatchery-origin fish for 10 of the 22 populations within this ESU limits our understanding of the trends in naturally spawning fish for a large portion of the ESU.

The estimated total run size of Chinook salmon in Puget Sound in the early 1990s was 240,000 fish, representing a loss of nearly 450,000 fish from historic numbers. During a recent 5-year period, the geometric mean of natural spawners in populations of Puget Sound Chinook salmon ranged from 222 to just over 9,489 fish. Most populations had natural spawners numbering in the hundreds (median recent natural escapement is 766), and of the six populations with greater than 1,000 natural spawners, only two have a low fraction of hatchery fish. The populations with the greatest estimated component of hatchery fish tend to be in mid- to southern Puget Sound, Hood Canal, and the Strait of Juan de Fuca regions. Estimates of the historical equilibrium abundance, based on pre-European settlement habitat conditions, range from 1,700 to 51,000 potential Puget Sound Chinook salmon spawners per population. The historical estimates of spawner capacity are several orders of magnitude higher than spawner abundances currently observed throughout the ESU (Good et al. 2005).

Critical Habitat
NMFS designated critical habitat for Puget Sound Chinook salmon on September 2, 2005 (70 FR 52630). The specific geographic area includes portions of the Nooksack River, Skagit River, Sauk River, Stillaguamish River, Skykomish River, Snoqualmie River, Lake Washington, Green River, Puyallup River, White River, Nisqually River, Hamma Hamma River and other Hood Canal watersheds, the Dungeness/Elwha Watersheds, and nearshore marine areas of the Strait of Georgia, Puget Sound, Hood Canal and the Strait of Juan de Fuca. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high water line is not defined the lateral extent is defined as the bankfull elevation.

The designation for this ESU includes sites necessary to support one or more Chinook salmon life stages. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. Specific primary constituent elements include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat 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. Of 49 subbasins (5th field Hydrological Units) reviewed in NMFS' assessment of critical habitat for the Puget Sound ESUs, nine subbasins were rated as having a medium conservation value, 12 were rated as low, and the remaining subbasins (40), where the bulk of Federal lands occur for this ESU, were rated as having a high conservation value to Puget Sound Chinook salmon. Factors contributing to the downward trends in this ESU are hydromorphological changes (such as diking, revetments, loss of secondary channels in floodplains, widespread blockages of streams, and changes in peak flows), degraded freshwater and marine habitat affected by agricultural activities and urbanization, and upper river tributaries widely affected by poor forest practices. Changes in habitat quantity, availability, diversity, flow, temperature, sediment load and channel stability are common limiting factors in areas of critical habitat.

Final Protective Regulations
On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the Puget Sound Chinook salmon ESU.

Sacramento River Winter-Run Chinook Salmon

Distribution and Description of the Listed Species
The Sacramento River winter-run Chinook salmon ESU includes all naturally spawned populations of winter-run Chinook salmon in the Sacramento River and its tributaries in California. Two artificial propagation programs are included in this ESU.

This ESU consists of a single spawning population that enters the Sacramento River and its tributaries in California from November to June and spawns from late April to mid-August, with a peak from May to June (Good et al. 2005). Sacramento River winter-run Chinook salmon historically occupied cold, headwater streams, such as the upper reaches of the Little Sacramento, McCloud and lower Pit Rivers. Young winter-run Chinook salmon venture to sea in November and December, after only four to seven months in fresh water (Groot and Margolis. 1991).

Status and Trends

NMFS listed Sacramento River winter-run Chinook salmon as endangered on January 4, 1994 (59 FR 440), and reaffirmed their status as endangered on June 28, 2005 (70 FR 37160), because dams restrict access to a small fraction of their historic spawning habitat and the habitat remaining to them is degraded. Sacramento River winter-run Chinook salmon consist of a single self-sustaining population which is entirely dependent upon the provision of suitably cool water from Shasta Reservoir during periods of spawning, incubation and rearing.

Construction of Shasta Dams in the 1940s eliminated access to historic spawning habitat for winter-run Chinook salmon in the basin. Winter-run Chinook salmon were not expected to survive this habitat alteration (Moffett 1949). However, cold water releases from Shasta Dam have created conditions suitable for winter Chinook salmon for roughly 60 miles downstream from the dam. As a result the ESU has been reduced to a single spawning population confined to the mainstem Sacramento River below Keswick Dam, although some adult winter-run Chinook salmon were recently observed in Battle Creek, a tributary to the upper Sacramento River. Quantitative estimates of run-size are not available for the period before 1996, the completion of Red Bluff Diversion Dam. However, winter-runs may have been as large as 200,000 fish based upon commercial fishery records from the 1870s (Fisher 1994). The California Department of Fish and Game estimated spawning escapement of Sacramento River winter-run Chinook salmon at 61,300 (60,000 in the mainstem, 1,000 in Battle Creek, and 300 in Mill Creek) in the early 1960s. During the first 3 years of operation of the county facility at the Red Bluff Diversion Dam (1967 to 1969), the spawning run of winter-run Chinook salmon averaged 86,500 fish. From 1967 through the mid-1990s, the population declined at an average rate of 18% per year, or roughly 50% per generation. The population reached critically low levels during the drought of 1987 to 1992; the 3-year average run size for the period of 1989 to 1991 was 388 fish. Based on the Red Bluff Diversion Dam counts, the population has been growing rapidly since the 1990s. Mean run size from 1995-2000 has been 2,191, but have ranged from 364 to 65,683 (Good et al. 2005). Most recent estimates indicate that the short term trend is 0.26, while the population growth rate is still less than 1(Good et al. 2005). The draft recovery goal for the ESU is an average of 10,000 female spawners per year and a population growth rate >1.0, calculated over 13 years of data (Good et al. 2005).

Critical Habitat

NMFS designated critical habitat for Sacramento River winter-run Chinook salmon on June 16, 1993 (58 FR 33212). The following areas consisting of the water, waterway bottom and adjacent riparian zones: the Sacramento River from Keswick Dam, Shasta County (river mile 302) to Chipps Island (river mile 0) at the westward margin of the Sacramento-San Joaquin Delta, and

other specified estuarine waters. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. Factors contributing to the downward trends in this ESU include reduced access to spawning/rearing habitat, possible loss of genetic integrity through population bottlenecks, inadequately screened diversions, predation at artificial structures and by nonnative species, pollution from Iron Mountain Mine and other sources, adverse flow conditions, high summer water temperatures, unsustainable harvest rates, passage problems at various structures, and vulnerability to drought (Good et al. 2005).

Final Protective Regulations
On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the Snake River fall-run Chinook salmon ESU.

Snake River Fall-Run Chinook Salmon

Distribution and Description of the Listed Species
The Snake River fall-run Chinook salmon ESU includes all naturally spawned populations of fall-run Chinook salmon in the mainstem Snake River below Hells Canyon Dam, and in the Tucannon River, Grande Ronde River, Imnaha River, Salmon River and Clearwater River subbasins. Four artificial propagation programs are part of this ESU.

Historically, the primary fall-run Chinook salmon spawning areas occurred on the upper mainstem Snake River (Connor et al. 2005). A series of Snake River dams blocked access to the upper reaches, which significantly reduced spawning and rearing habitat. Consequently, salmon now reside in waters that are generally cooler than pre-dam habitats. Currently, natural spawning occurs at the upper end of Lower Granite Reservoir to Hells Canyon Dam, the lower reaches of the Imnaha, Grande Ronde, Clearwater and Tucannon rivers and small mainstem sections in the tailraces of the lower Snake River hydroelectric dams.

Adult Snake River fall-run Chinook salmon enter the Columbia River in July and August, and spawning occurs from October through November. Juveniles emerge from the gravels in March and April of the following year, moving downstream from natal spawning and early rearing areas from June through early autumn. Prior to dam construction, fall Chinook salmon were primarily ocean-type (migrated downstream and reared in the mainstem Snake River during their first year). However, today both an ocean-type and reservoir-type occur (Connor et al. 2005). The reservoir-type juveniles overwinter in pools created by dams before migrating to sea; this response is likely due to early development in cooler temperatures which prevents rapid growth. Phenotypic characteristics have shifted in apparent response to environmental changes from hydroelectric dams (Connor et al. 2005). Migration downstream appears to be influenced by flow velocity within both river and reservoir systems (Tiffan et al. 2009).

Status and Trends
NMFS originally listed Snake River fall-run Chinook salmon as endangered in 1992 (57 FR 14653) but reclassified their status as threatened on June 28, 2005 (70 FR 37160). Estimated annual returns for the period 1938 to 1949 was 72,000 fish, and by the 1950s, numbers had declined to an annual average of 29,000 fish (Bjornn and Horner 1980). Numbers of Snake River fall-run Chinook salmon continued to decline during the 1960s and 1970s as approximately 80% of their historic habitat was eliminated or severely degraded by the construction of the Hells Canyon complex (1958 to 1967) and the lower Snake River dams (1961 to 1975). Counts of natural-origin adult Snake River fall-run Chinook salmon at Lower Granite Dam were 1,000 fish in 1975, and ranged from 78 to 905 fish (with an average of 489 fish) over the ensuing 25-year period (Good et al. 2005). Numbers of natural-origin Snake River fall-run Chinook salmon have increased over the last few years, with estimates at Lower Granite Dam of 2,652 fish in 2001, 2,095 fish in 2002, and 3,895 fish in 2003.

Snake River fall-run Chinook salmon have exhibited an upward trend in returns over Lower Granite Dam since the mid 1990s. Returns classified as natural-origin spawners exceeded 2,600 fish in 2001, compared to a 1997 to 2001 geometric mean natural-origin count of 871 (35% of the proposed delisting abundance criteria of 2,500 natural spawners averaged over 8 years). Both the long- and short-term trends in natural returns are positive. Harvest impacts on Snake River fall Chinook salmon declined after listing and have remained relatively constant in recent years. Mainstem conditions for sub-yearling Chinook migrants from the Snake River have generally improved since the early 1990s. The hatchery component, derived from outside the basin, has decreased as a percentage of the run at Lower Granite Dam from the 1998/99 status reviews (5-year average of 26.2%) to 2001 (8%). This reflects an increase in the Lyons Ferry component, systematic removal of marked hatchery fish at the Lower Granite trap, and modifications to the Umatilla supplementation program to increase homing of fall Chinook salmon release groups. Hatcheries stocking fish to the Snake River fall run produce genetic affects in the population due to three major components: natural-origin fish (which may be progeny of hatchery fish), returns of Snake River fish from the Lyons Ferry Hatchery program, and strays from hatchery programs outside the Snake River.

Critical Habitat
NMFS designated critical habitat for Snake River fall-run Chinook salmon on December 28, 1993 (58 FR 68543). This critical habitat encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed Snake River salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. Adjacent riparian zones are defined as those areas within a horizontal distance of 300 feet from the normal line of high water of a stream channel or from the shoreline of a standing body of water. Designated critical habitat includes the Columbia River from a straight line connecting the west end of the Clatsop jetty (Oregon side) and the west end of the Peacock jetty (Washington side) and including all river reaches from the estuary upstream to the confluence of the Snake River, and all Snake River reaches upstream to Hells Canyon Dam. Critical habitat also includes several river reaches

presently or historically accessible to Snake River fall-run Chinook salmon. Limiting factors identified for Snake River fall-run Chinook salmon include: mainstem lower Snake and Columbia hydrosystem mortality, degraded water quality, reduced spawning and rearing habitat due to mainstem lower Snake River hydropower system, harvest impacts, impaired stream flows, barriers to fish passage in tributaries, excessive sediment, and altered floodplain and channel morphology (NMFS 2005b).

Snake River Spring/Summer-Run Chinook Salmon

Distribution and Description of the Listed Species
The Snake River spring/summer-run Chinook salmon ESU includes all naturally spawned populations of spring/summer-run Chinook salmon in the mainstem Snake River and the Tucannon River, Grande Ronde River, Imnaha River and Salmon River subbasins. Fifteen artificial propagation programs are part of the ESU. The Interior Columbia Basin Technical Recovery Team has identified 32 populations in five major population groups (Upper Salmon River, South Fork Salmon River, Middle Fork Salmon River, Grande Ronde/Imnaha, Lower Snake Mainstem Tributaries) for this species. Historic populations above Hells Canyon Dam are considered extinct (ICBTRT 2003).

Snake River spring/summer-run Chinook salmon have a stream-type life history. Spawning occurs in late summer and early fall and eggs incubate over the following winter and hatch in late winter and early spring of the following year. Juveniles mature in the river for one year before migrating to the ocean in the spring of their second year. Larger outmigrants have a higher survival rate during outmigration (Zabel and Williams 2002, Zabel and Achord 2004). Depending on tributary and the specific habitat conditions, juveniles may migrate widely from natal reaches into alternative summer-rearing or overwintering areas. Spawners return to spawn primarily as 4- and 5-year-olds after 2 to 3 years in the ocean.

Status and Trends
NMFS originally listed Snake River spring/summer-run Chinook salmon as threatened on April 22, 1992 (57 FR 14653), and reaffirmed their status as threatened on June 28, 2005 (70 FR 37160). Although direct estimates of historical annual Snake River spring/summer Chinook salmon returns are not available, returns may have declined by as much as 97% between the late 1800s and 2000. According to Matthews and Waples (1991), total annual Snake River spring/summer Chinook salmon production may have exceeded 1.5 million adult fish in the late 1800s. Total (natural plus hatchery origin) returns fell to roughly 100,000 spawners by the late 1960s (Fulton 1968). The 1997 to 2001 geometric mean total return for the summer run component at Lower Granite Dam was slightly more than 6,000 fish, compared to the geometric mean of 3,076 fish for the years 1987 to 1996 (Good et al. 2005). Good *et al.,* (2005) reported that risks to individual populations within the ESU may be greater than the extinction risk for the entire ESU due to low levels of annual abundance and the extensive production areas within the Snake River basin. Although the average abundance in the most recent decade is more abundant than the previous decade, there is no obvious long-term trend.

Critical Habitat
NMFS designated critical habitat for Snake River spring/summer-run Chinook salmon on October 25, 1999 (64 FR 57399). This critical habitat encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed Snake River salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). Adjacent riparian zones are defined as those areas within a horizontal distance of 300 feet from the normal line of high water of a stream channel or from the shoreline of a standing body of water. Designated critical habitat includes the Columbia River from a straight line connecting the west end of the Clatsop jetty (Oregon side) and the west end of the Peacock jetty (Washington side) and including all river reaches from the estuary upstream to the confluence of the Snake River, and all Snake River reaches upstream to Hells Canyon Dam; the Palouse River from its confluence with the Snake River upstream to Palouse Falls, the Clearwater River from its confluence with the Snake River upstream to its confluence with Lolo Creek; the North Fork Clearwater River from its confluence with the Clearwater river upstream to Dworshak Dam. Critical habitat also includes several river reaches presently or historically accessible to Snake River spring/summer Chinook salmon. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. Limiting factors identified for this species include hydrosystem mortality, reduced stream flow, altered channel morphology and floodplain, excessive fine sediment, and degraded water quality (NMFS 2006d).

Final Protective Regulations
On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the Snake River spring/summer-run Chinook salmon ESU.

Upper Willamette River Chinook Salmon

Distribution and Description of the Listed Species
The Upper Willamette River Chinook salmon ESU includes all naturally spawned populations of spring-run Chinook salmon in the Clackamas River and in the Willamette River, and its tributaries, above Willamette Falls, Oregon. Seven artificial propagation programs are part of the ESU.

Upper Willamette River Chinook salmon occupy the Willamette River and its tributaries. All spring-run Chinook salmon in the ESU, except those entering the Clackamas River, must pass Willamette Falls. In the past, this ESU included sizable numbers of spawning salmon in the Santiam River, the middle fork of the Willamette River, and the McKenzie River, as well as smaller numbers in the Molalla River, Calapooia River and Albiqua Creek. Historically, access above Willamette Falls was restricted to the spring when flows were high. In autumn, low flows

prevented fish from ascending past the falls. The Upper Willamette spring-run Chinook salmon are one of the most genetically distinct Chinook salmon groups in the Columbia River Basin. Upper Willamette River Chinook salmon enter the Columbia River and estuary earlier than other spring Chinook salmon ESUs (Meyers et al. 1998). Fall-run Chinook salmon spawn in the Upper Willamette but are not considered part of the ESU because they are not native.

Status and Trends

NMFS originally listed Upper Willamette River Chinook salmon as threatened on March 24, 1999 (64 FR 14308), and reaffirmed their status as threatened on June 28, 2005 (70 FR 37160). The total abundance of adult spring-run Chinook salmon (hatchery-origin plus natural-origin fish) passing Willamette Falls has remained relatively steady over the past 50 years (ranging from approximately 20,000 to 70,000 fish), but it is an order of magnitude below the peak abundance levels observed in the 1920s (approximately 300,000 adults). Until recent years, interpretation of abundance levels has been confounded by a high but uncertain fraction of hatchery-produced fish. Although the number of adult spring-run Chinook salmon crossing Willamette Falls is in the same range (about 20,000 to 70,000 adults) it has been for the last 50 years, a large fraction of these are hatchery produced. Estimates of the percentage of hatchery fish range according to tributary, several of which exceed 70 percent (Good et al. 2005). The Calapooia River is estimated to contain 100 percent hatchery fish. Insufficient information on hatchery production in the past prevents a meaningful analysis of the population trend; therefore no formal trend analysis is available.

Most natural spring Chinook salmon populations of the Upper Willamette River are likely extirpated or nearly so, with only one remaining naturally reproducing population identified in this ESU: the spring Chinook salmon in the McKenzie River. Unfortunately, recently short-term declines in abundance suggest that this population may not be self-sustaining (Meyers et al. 1998). Abundance in this population has been relatively low (low thousands) with a substantial number of these fish being of hatchery origin. The population increased substantially from 2000 to 2003, probably due to increased survival in the ocean. Future survival rates in the ocean are unpredictable, and the likelihood of long-term sustainability for this population has not been determined. Of concern is that a majority of the spawning habitat and approximately 30 to 40% of total historical habitat are no longer accessible because of dams (Good et al. 2005). Individuals from the ESU migrate far north and are caught incidentally in ocean fisheries, particularly off southeast Alaska and northern Canada, and in the mainstem Columbia and Willamette rivers during spring.

Critical Habitat

NMFS designated critical habitat for Upper Willamette River Chinook salmon on September 2, 2005 (70 FR 52630). Critical habitat for upper Willamette River Chinook salmon includes defined areas within subbasins of the middle fork Willamette River, upper Willamette River, McKenzie River, Santiam River, Crabtree Creek, Molalla River and Clackamas River. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to

support one or more Chinook salmon life stages. Specific sites include freshwater spawning and rearing sites, freshwater migration corridors. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Of 65 subbasins reviewed in NMFS' assessment of critical habitat for the Upper Willamette River Chinook salmon ESU, 19 subbasins were rated as having a medium conservation value, 19 were rated as low, and the 27 remaining subbasins were rated as having a high conservation value to Upper Willamette River Chinook salmon. Federal lands were generally rated as having high conservation value to the species' spawning and rearing. Factors contributing to the downward trends in this ESU include reduced access to spawning/rearing habitat in tributaries, hatchery impacts, altered water quality and temperature in tributaries, altered stream flow in tributaries, and lost or degraded floodplain connectivity and lowland stream habitat.

Final Protective Regulations
On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the Upper Willamette River Chinook salmon ESU.

Chum Salmon

Description of the Species
Chum salmon are more widely distributed than other salmon, and may have at one time made up nearly 50% of the Pacific salmon biomass in the Pacific Ocean (Salo 1991). Historically, chum salmon were distributed throughout the coastal regions of western Canada and the U.S., as far south as Monterey Bay, California, to the Arctic coast and east to the Mackenzie River, in the Beaufort Sea. They also ranged in Asia from Korea to the Arctic coast of the Soviet Union and west to the Lena River. Presently, major spawning populations on the west coast of the U.S. are found only as far south as Tillamook Bay on the northern Oregon coast. In this section of our Opinion, we discuss the distribution, status and critical habitats of the two listed species of threatened chum salmon separately; however, because chum salmon in the wild are virtually indistinguishable between listed ESUs, and are the same biological species sharing the same generalized life history, we begin this section describing those characteristics common across ESUs.

There are no known landlocked or naturalized freshwater populations of chum salmon. Like Chinook salmon, chum salmon are semelparous (Randall et al., 1987 as cited in Johnson et al. 1997). Their general life cycle spans fresh and marine waters, although chum salmon are more marine oriented than the other Pacific salmon. Chum salmon spend 2 to 5 years in feeding areas in the northeast Pacific Ocean (Johnson et al. 1997). Chum salmon distribute throughout the North Pacific Ocean and Bering Sea (Neave et al., 1976 as cited in Johnson et al. 1997).

Spawning migrations generally occur in the summer and fall; the precise spawn timing and migration varies across populations. Generally, spawning runs consist of fish between 2 and 5 years of age. Fecundity is highly variable, and is correlated with body size and region (Salo 1991). Once they emerge from their gravel nests, chum salmon fry outmigrate to seawater almost immediately (Salo 1991). This ocean-type migratory behavior contrasts with the stream-type behavior of other species in its genus. Because of their small size chum salmon will form loosely aggregated schools, presumably to reduce predation by swamping predators (Miller and Brannon 1982; Pitcher 1986).

Generally, chum fry emigrate to estuaries between March through May where they forage on epibenthic and neritic food resources. The timing of juvenile entry into sea water is commonly correlated with nearshore warming and associated plankton blooms (Groot and Margolis. 1991). As food resources decline and the fish grow, they move further out to forage on pelagic and nektonic organisms (Simenstad and Salo 1982, Salo 1991). Migratory studies indicate that chum salmon in their first year of life will typically maintain a coastal migratory pattern although the pattern is variable as they mature at sea. At sea chum salmon feed on pteropods, euphausiids, amphipods, fish and squid larvae (Salo 1991).

Threats

Natural Threats. Chum salmon are exposed to high rates of natural predation each stage of their life stage and in particular during migration. Mortality at emergence or prior to emergence is significant because eggs develop in the interstitial spaces in the stream gravel, and storm surges that redeposit gravels and wash out eggs or introduce silt to the interstitial spaces can reduce egg survival. Other factors that reduce egg survival and larvae development include low dissolved oxygen, poor percolation and extreme cold or warm temperatures.

Anthropogenic Threats. Chum salmon, like the other listed salmon, have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the dynamics (hydrogeomorphology) of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile chum salmon; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems while introducing sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the fresh water, estuarine, and coastal ecosystems throughout the Pacific Northwest. These threats for are summarized in detail under the Chinook salmon of this section.

Columbia River Chum Salmon

Distribution and Description of the Listed Species

The Columbia River chum ESU includes all naturally spawned populations of chum salmon in the Columbia River and its tributaries in Washington and Oregon. Three artificial propagation programs are part of the ESU.

Most of the chum within this ESU return to northern tributaries of the Columbia River (in Washington State), primarily the Grays River, in areas immediately below Bonneville Dam, and in smaller numbers under the I-205 bridge near Vancouver. Chum populations that formerly occupied tributaries on the south bank of the Columbia (in Oregon) are considered extirpated or nearly so. Observers have documented spawning over multiple years in the mainstem Columbia River, near McCord Creek and Multnomah Falls in Oregon, although the number of spawners in these areas are generally quite low (McElhany et al. 2007).

Status and Trends

NMFS listed Columbia River chum salmon as threatened on March 25, 1999, and reaffirmed their threatened status on June 28, 2005 (71 FR 37160). Chum salmon in the Columbia River once numbered in the hundreds of thousands of adults and were reported in almost every river in the Lower Columbia River basin, but by the 1950s most runs disappeared (Rich 1942, Marr 1943, Fulton 1970). The total number of chum salmon returning to the Columbia River in the last 50 years has averaged a few thousand per year, with returns limited to a very restricted portion of the historical range. Significant spawning occurs in only two of the 16 historical populations, meaning that 88% of the historical populations are extirpated, or nearly so (Good *et al.,* 2005*)*. The two remaining populations are the Grays River and the lower Columbia Gorge tributaries (Good et al. 2005). Both long- and short-term trends for Grays River abundance are negative, but the current trend in abundance for the lower Columbia Gorge tributaries is slightly positive. Chum salmon appear to be extirpated from the Oregon portion of this ESU. In 2000, ODFW conducted surveys to determine the abundance and distribution of chum salmon in the Columbia River, and out of 30 sites surveyed, only one chum salmon was observed.
Few Columbia River chum salmon have been observed in tributaries between The Dalles and Bonneville dams. Surveys of the White Salmon River in 2002 found one male and one female carcass, with no evidence of spawning (Ehlke and Keller 2003). Chum salmon were not observed in any upper Columbia Gorge tributaries during the 2003 and 2004 spawning ground surveys. Finally, most Columbia River chum populations have been functionally extirpated or are presently at very low abundance levels.

Historically, the Columbia River chum salmon supported a large commercial fishery in the first half of this century which landed more than 500,000 fish per year as recently as 1942. Commercial catches declined beginning in the mid-1950s, and in later years rarely exceeded 2,000 per year (Good et al. 2005). During the 1980s and 1990s, the combined abundance of natural spawners for the lower Columbia Gorge, Washougal and Grays River populations was below 4,000 adults. In 2002, however, the abundance of natural spawners exhibited a substantial increase at several locations (estimate of natural spawners is approximately 20,000 adults) (Good et al. 2005). The cause of this dramatic increase in abundance is unknown. However, long- and short-term productivity trends for populations are at or below replacement. The loss of off-channel habitat and the extirpation of approximately 17 historical populations increase this species' vulnerability to environmental variability and catastrophic events. Overall, the populations that remain have low abundance, limited distribution, and poor connectivity (Good et al. 2005).

Critical Habitat
NMFS designated critical habitat for Columbia River chum salmon on September 2, 2005 (70 FR 52630). The designated includes defined areas in the following subbasins: Middle Columbia/Hood, Lower Columbia/Sandy, Lewis, Lower Columbia/Clatskanie, Lower Cowlitz, Lower Columbia subbasin and river corridor. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation.

The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more chum salmon life stages. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding and are rated as having high conservation value to the species. Columbia River chum salmon have primary constituent elements of 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 physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Of 21 subbasins reviewed in NMFS' assessment of critical habitat for the Columbia River chum salmon ESU, three subbasins were rated as having a medium conservation value, no subbasins were rated as low, and the majority of subbasins (18), were rated as having a high conservation value to Columbia River chum salmon. The major factors limiting recovery for Columbia River chum salmon are altered channel form and stability in tributaries, excessive sediment in tributary spawning gravels, altered stream flow in tributaries and the mainstem Columbia River, loss of some tributary habitat types, and harassment of spawners in the tributaries and mainstem.

Final Protective Regulations
On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the Columbia River chum salmon ESU.

Hood Canal Summer-Run Chum Salmon

Distribution and Description of the Listed Species
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 (64 FR 14508) from mid-September to mid-October (WDF 1993), but may enter natal rivers in late August. Eight artificial propagation programs are considered to be part of the ESU.

On average Hood Canal chum salmon reside in estuaries for 23 days; daily tidal migrations have not been observed, but prey availability does influence movement patterns (Bax 1983). Upon leaving their natal estuaries summer-run chum salmon generally migrate through Hood Canal and into the main body of Puget Sound.

Status and Trends

NMFS listed Hood Canal summer-run chum salmon as threatened on March 25, 1999 (64 FR 14508), and reaffirmed as threatened on June 28, 2005 (70 FR 37160). Historically, Hood Canal summer-run chum salmon comprised an estimated 16 populations. Only eight extant populations remain within this ESU (Good et al. 2005). Most of the extirpated populations historically occurred on the eastern side of Hood Canal, which is cause for concern over the current spatial structure of this ESU. The widespread loss of estuary and lower floodplain habitat is a continuing threat to ESU spatial structure and connectivity.

Adult returns for some populations showed modest improvements in 2000, with upward trends continuing in 2001 and 2002 (Good et al. 2005). The recent 5-year mean abundance is variable among populations in the species, ranging from one fish to nearly 4,500 fish in the Big/Little Quilcene rivers. Hood Canal summer-run chum are the focus of an extensive rebuilding program developed and implemented since 1992 by the State and Tribal comanagers. Two populations (the combined Quilcene and Union River populations) are above the conservation thresholds established by the rebuilding plan. However, most populations remain depressed. Estimates of the fraction of naturally spawning hatchery fish exceed 60% for some populations, indicating that reintroduction programs are supplementing the numbers of total fish spawning naturally in streams (Good et al. 2005). Long-term trends in productivity are above replacement for only the Quilcene and Union River populations (Good et al. 2005). Buoyed by recent increases, seven populations are exhibiting short-term productivity trends above replacement.

Of the eight programs releasing summer-run chum salmon that are considered to be part of the Hood Canal summer chum ESU, six of the programs are supplementation programs implemented to preserve and increase the abundance of native populations in their natal watersheds. NMFS' assessment of the effects of artificial propagation on ESU extinction risk concluded that these hatchery programs collectively do not substantially reduce the extinction risk of the ESU. The hatchery programs are reducing risks to ESU abundance by increasing total ESU abundance as well as the number of naturally spawning summer-run chum salmon.

Critical Habitat

NMFS designated critical habitat for Hood Canal summer-run chum salmon on September 2, 2005 (70 FR 52630). The specific geographic area includes the Skokomish River, Hood Canal subbasin, which includes the Hamma Hamma and Dosewallips rivers and others, the Puget Sound subbasin, Dungeness/Elwha 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 meters. This includes a narrow nearshore zone from the extreme high-tide to mean lower low tide within several Navy security/restricted zones. This also includes about 8 miles of habitat that was unoccupied at the time of the designation Finch, Anderson and Chimacum creeks (69 FR 74572; 70 FR 52630), but has recently been re-seeded. The designation for Hood Canal summer-run chum, like others

made at this time, includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined, the lateral extent is defined as the bankfull elevation.

The specific primary constituent elements identified for Hood Canal summer-run chum salmon are areas for spawning, freshwater rearing and migration, estuarine areas free of obstruction, nearshore marine areas free of obstructions, and offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Of 17 subbasins reviewed in NMFS' assessment of critical habitat for the Hood Canal chum salmon ESU, 14 subbasins were rated as having a high conservation value, while only three were rated as having a medium value to conservation. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. Limiting factors identified for this species include degraded floodplain and mainstem river channel structure, degraded estuarine conditions and loss of estuarine habitat, riparian area degradation and loss of in-river wood in mainstem, excessive sediment in spawning gravels, and reduced stream flow in migration areas.

Final Protective Regulations
On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the Hood Canal summer-run chum salmon ESU.

Coho Salmon

Description of the Species
Coho salmon occur naturally in most major river basins around the North Pacific Ocean from central California to northern Japan (Laufle et al. 1986). They spawn in the fall and winter and the young emerge in the spring. Adult coho salmon may remain in fresh water three or more months before spawning, with early migrants often moving farther upstream than later migrants (Sandercock 1991). Spawning occurs in a few third-order streams, but most spawning activity occurs in fourth- and fifth-order streams. As with other Pacific salmon, coho salmon fecundity varies with the size of the fish and latitudinally with coho salmon in northern climes generally exhibiting higher rates of fecundity (Sandercock 1991). Most coho salmon mature and spawn at age 3, although there are exceptions (Sandercock 1991).

The outmigration of coho smolts begins as early as February and may continue through the summer and fall, with peak outmigration often between March and June, although this varies among basins and environmental conditions (Sandercock 1991). While at sea, coho salmon tend to eat fish including herring, sand lance, sticklebacks, sardines, shrimp and surf smelt. While in

estuaries and in fresh water coho salmon are significant predators of Chinook, pink, and chum salmon, as well as aquatic and terrestrial insects. Smaller fish, such as fry, eat chironomids, plecoptera and other larval insects, and typically use visual cues to find their prey.

Threats

Natural Threats. Coho salmon, like other salmon, are exposed to high rates of natural predation at each life stage. Most mortality, however, occurs in the freshwater life stages. Winter mortality may be significant for coho salmon because they overwinter in fresh water, where they can be swept downstream from freshets or eaten by raccoon, cutthroat trout or other small animals. Once coho reach the ocean, survival is high (Sandercock 1991).

Anthropogenic Threats. Coho salmon have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the dynamics (hydrogeomorphology) of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile coho salmon; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems while introducing sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the fresh water, estuarine, and coastal ecosystems throughout the species' range. These threats for are summarized in detail under Chinook salmon.

Central California Coast Coho Salmon

Distribution and Description of the Listed Species

The Central California Coast coho salmon ESU extends from Punta Gorda in northern California south to and including the San Lorenzo River in central California (Sandercock 1991). The ESU includes all naturally spawned populations of coho salmon from Punta Gorda in northern California south to and including the San Lorenzo River in central California, as well as populations in tributaries to San Francisco Bay, excluding the Sacramento-San Joaquin River system. Four artificial propagation programs are part of the Central California Coast coho salmon ESU.

Coho salmon in this ESU enter rivers to spawn very late (peaking in January), with little time spent in fresh water between river entry and spawning. This compressed adult freshwater residency appears to coincide with the single, brief peak of river flow characteristic of this region.

Status and Trends

NMFS originally listed the central California coast coho salmon ESU as threatened on October 31, 1996 (61 FR 56138) and later reclassified their status to endangered June 28, 2005 (70 FR 37160). Information on the abundance and productivity trends for the naturally spawning component of the central California coast coho ESU is extremely limited. There are no long-term time series of spawner abundance for individual river systems. Historical estimated

escapement for this ESU is 56,100 for 1963, and more recent estimates suggest the ESU dropped to about one-fourth that size by the late 1980s and early 1990s (Good et al. 2005).

Where data are available, analyses of juvenile coho presence-absence information, juvenile density surveys, and irregular adult counts for the South Fork Noyo River indicate low abundance and long-term downward trends for the naturally spawning populations throughout the ESU (Good et al. 2005). Improved ocean conditions coupled with favorable stream flows and harvest restrictions have contributed to increased returns in 2001 in streams in the northern portion of the ESU, as indicated by an increase in the observed presence of fish in historically occupied streams (Good et al. 2005). Data are particularly lacking for many river basins in the southern two-thirds of the ESU where naturally spawning populations are considered to be at the greatest risk. The extirpation or near extirpation of natural coho salmon populations in several major river basins, and across most of the southern historical range of the ESU, represents a significant risk to ESU spatial structure and diversity (Good et al. 2005).

Artificial propagation of coho salmon within the Central California Coast ESU has declined since the ESU was listed in 1996 though it continues at the Noyo River and Scott Creek facilities, and two captive broodstock populations have recently been established. Genetic diversity risk associated with out-of-basin transfers appears to be minimal, but diversity risk from domestication selection and low effective population sizes in the remaining hatchery programs remains a concern. An out-of-ESU artificial propagation program for coho was operated at the Don Clausen hatchery on the Russian River through the mid 1990s, but was terminated in 1996. Termination of this program was considered by the biological review team as a positive development for naturally produced coho in this ESU.

For the naturally spawning component of the ESU, the biological review team found very high risk of extinction for the abundance, productivity and spatial structure of the Viable Salmon Population (VSP) parameters and comparatively moderate risk with respect to the diversity VSP parameter. The lack of direct estimates of the performance of the naturally spawned populations in this ESU, and the associated uncertainty this generates, was of specific concern to the biological review team. Informed by the VSP risk assessment and the associated uncertainty, the strong majority opinion of the biological review team was that the naturally spawned component of the Central California Coast coho ESU was "in danger of extinction." The minority opinion was that this ESU is "likely to become endangered within the foreseeable future." (70 FR 37160) Accordingly, NMFS upgraded the status of central California coast coho ESU to endangered on June 28, 2005 (70 FR 37160).

Central California Coast coho salmon populations continue to be depressed relative to historical numbers. Strong indications show that breeding groups have been lost from a significant percentage of historical stream range. A number of coho populations in the southern portion of the range appear to be either extinct or nearly so, including those in Gualala, Garcia, and Russian rivers, as well as smaller coastal streams in and south of San Francisco Bay (Good et al. 2005).

Critical Habitat

NMFS designated critical habitat for central California coast coho salmon on May 5, 1999 (64

FR 24049). The designation encompasses accessible reaches of all rivers (including estuarine areas and riverine reaches) between Punta Gorda and the San Lorenzo River (inclusive) in California, including two streams entering San Francisco Bay: Arroyo Corte Madera Del Presidio and Corte Madera Creek. This critical habitat designation includes all waterways, substrate and adjacent riparian zones of estuarine and riverine reaches (including off-channel habitats) below longstanding naturally impassable barriers (i.e. natural waterfalls in existence for at least several hundred years). These areas are important for the species' overall conservation by protecting growth, reproduction, and feeding.

Lower Columbia River Coho Salmon

Distribution and Description of the Listed Species
The lower Columbia River coho salmon ESU includes all naturally spawned populations of coho salmon in the Columbia River and its tributaries in Washington and Oregon, from the mouth of the Columbia up to and including the Big White Salmon and Hood Rivers, and includes the Willamette River to Willamette Falls, Oregon, Twenty-five artificial propagation programs are part of this ESU.

Two distinct runs distinguished by the timing of adult returns to fresh water (early returners and later returners) occur within the ESU. Early returning adults generally migrate south of the Columbia River once they reach the ocean, returning to fresh water in mid-August and to spawning tributaries in early September. Peak spawning of early returning adults occurs from mid-October to early November. Late returning adult coho salmon exhibit a northern oceanic distribution, return to the Columbia River from late September through December and enter tributaries from October through January. Most late return adults spawn between November through January, although some spawn in February and as late as March (Sandercock 1991). Almost all Lower Columbia River ESU coho salmon females and most males spawn at 3 years of age.

Status and Trends
NMFS listed Lower Columbia River coho salmon as threatened on June 28, 2005 (70 FR 37160). The vast majority (over 90%) of the historic population in the Lower Columbia River coho salmon ESU appear to be either extirpated or nearly so.
Only two populations of coho salmon within this ESU produce a sizeable number of naturally spawned fish, the upper Sandy River population above Marmot Dam and the Clackamas River population above the North Fork Dam. Most of the other populations are believed to have very little, if any, natural production. The long-term and short-term trends for Marmot Dam counts are both negative. The long-term median growth rate is slightly positive for both the Sandy and Clackamas rivers, but the confidence intervals for each are very wide indicating there is a large amount of uncertainty. Both populations within the Sandy and Clackamas rivers have suffered from recruitment failure a number of times over the past 15 years, despite the reductions in harvest. The most serious threat facing this ESU is the scarcity of naturally-produced spawners, with attendant risks associated with small populations, loss of diversity, and fragmentation and isolation of the remaining naturally-produced fish. Spatial structure has been substantially

reduced by the loss of access to upper basins from tributary hydro development (i.e., Condit Dam on the Big White Salmon River and Powerdale Dam on the Hood River). The diversity of populations in all three areas has been eroded by large hatchery influences and periodically, low effective population sizes.

Critical Habitat
NMFS has not designated critical habitat for Lower Columbia River coho salmon.

Southern Oregon/Northern California Coast Coho Salmon

Distribution and Description of the Listed Species
Southern Oregon/Northern California coast coho salmon consists of all naturally spawning populations of coho salmon that reside below long-term, naturally impassible barriers in streams between Punta Gorda, California and Cape Blanco, Oregon, as well as three artificial propagation programs: the Cole Rivers Hatchery, Trinity River Hatchery and Iron Gate Hatchery coho hatchery programs. The three major river systems supporting Southern Oregon – Northern Coastal California coast coho are the Rogue, Klamath (including the Trinity), and Eel rivers. Southern Oregon and Northern California coast coho immigrate to natal rivers in September or October. River entry is much later south of the Klamath River Basin, occurring in November and December, as well as in basins south of the Klamath River to the Mattole River, California. River entry occurs from mid-December to mid-February in rivers farther south. Because individuals enter rivers late, they spend much less time in the river. Coho salmon adults spawn at age 3, spending just over 1 year in fresh water and a year and a half in the ocean.

Status and Trends
Southern Oregon/Northern California coast coho salmon were listed as threatened on May 7, 1997 (62 FR 24588); they retained that classification when their status was reviewed on June 28, 2005 (70 FR 37160). Southern Oregon/Northern California Coast coho salmon extend from Cape Blanco in southern Oregon to Punta Gorda in northern California (Sandercock 1991). The status of coho salmon coast-wide, including the Southern Oregon/Northern California Coast coho salmon ESU, was formally assessed in 1995 (Sandercock 1991). Two subsequent status review updates have been published by NMFS, one addressing all West Coast coho salmon ESUs and a second specifically addressing the Oregon Coast Southern Oregon/Northern California Coast coho salmon ESUs (NMFS 1996, 1997a). In the 1997 status update, estimates of natural population abundance were based on very limited information. New data on presence/absence in northern California streams that historically supported coho salmon were even more disturbing than earlier results, indicating that a smaller percentage of streams contained coho salmon compared to the percentage presence in an earlier study (Good et al. 2005). However, it was unclear whether these new data represented actual trends in local extinctions or were biased by sampling effort.

Data on population abundance and trends are limited for the California portion of this ESU. No regular estimates of natural spawner escapement are available. Historical point estimates of coho salmon abundance for the early 1960s and mid-1980s suggest that statewide coho spawning escapement in the 1940s ranged between 200,000 and 500,000 fish. Numbers declined to about

100,000 fish by the mid-1960s with about 43% originating from this ESU. Brown *et al.,* (1994) estimated that the California portion of this ESU was represented by about 7,000 wild and naturalized coho salmon (Good et al. 2005). In the Klamath River, the estimated escapement has dropped from approximately 15,400 in the mid-1960s to about 3,000 in the mid 1980s, and more recently to about 2,000 (Good et al. 2005). The second largest producing river in this ESU, the Eel River, dropped from 14,000, to 4,000 to about 2,000 during the same period. Historical estimates are considered "best guesses" made using a combination of limited catch statistics, hatchery records and the personal observations of biologists and managers.

Most recently, Williams *et al.,* (2006) described the structure of historic populations of Southern Oregon/Northern California Coast coho salmon. They described three categories of populations: functionally independent populations, potentially independent populations and dependent populations. Functionally independent populations are populations capable of existing in isolation with a minimal risk of extinction. Potentially independent populations are similar but rely on some interchange with adjacent populations to maintain a low probability of extinction. Dependent populations have a high risk of extinction in isolation over a 100-year timeframe and rely on exchange of individuals from adjacent populations to maintain themselves.

Critical Habitat

NMFS designated critical habitat for Southern Oregon/Northern California Coast coho salmon on May 5, 1999 (64 FR 24049). Critical habitat for this species encompasses all accessible river reaches between Cape Blanco, Oregon, and Punta Gorda, California. Critical habitat consists of the water, substrate and river reaches (including off-channel habitats) in specified areas. Accessible reaches are those within the historical range of the ESU that can still be occupied by any life stage of coho salmon. Of 155 historical streams for which data are available, 63% likely still support coho salmon. These river habitats are important for a variety of reasons, such as supporting the feeding and growth of juveniles and serving as spawning habitat for adults. Limiting factors identified for this species include: loss of channel complexity, connectivity and sinuosity, loss of floodplain and estuarine habitats, loss of riparian habitats and large in-river wood, reduced stream flow, poor water quality, temperature and excessive sedimentation, and unscreened diversions and fish passage structures.

Final Protective Regulations

On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the Southern Oregon/Northern California coast coho salmon.

Oregon Coast Coho Salmon

Distribution and Description of the Listed Species

The Oregon Coast coho salmon ESU includes all naturally spawned populations of coho salmon

in Oregon coastal streams south of the Columbia River and north of Cape Blanco (63 FR 42587; August 1998). One hatchery population, the Cow Creek hatchery coho salmon, is considered part of the ESU.

Status and Trends

The Oregon coast coho salmon ESU was listed as a threatened species under the ESA on February 11, 2008 (73 FR 7816), the conclusion to a 13-year history of court cases. The most recent NMFS status review for the Oregon Coast coho ESU was conducted by the biological review team in 2003, which assessed data through 2002. The abundance and productivity of Oregon Coast coho since the previous status review represented some of the best and worst years on record (Sandercock 1991). Yearly adult returns for the Oregon Coast coho ESU were over 160,000 natural spawners in 2001 and over 260,000 in 2002, far exceeding the abundance observed for the past several decades (Good et al. 2005). These increases in spawner abundance in 2000 to 2002 followed three consecutive brood years (the 1994 to 1996 brood years returning in 1997 to 1999, respectively) exhibiting recruitment failure (recruitment failure is when a given year class of natural spawners fails to replace itself when its offspring return to the spawning grounds 3 years later). These 3 years of recruitment failure were the only such instances observed thus far in the entire 55-year abundance time series for Oregon Coast coho salmon (although comprehensive population-level survey data have only been available since 1980). The 2000 to 2002 increases in natural spawner abundance occurred in many populations in the northern portion of the ESU, which were the most depressed at the time of the last review (Sandercock 1991). Although encouraged by the increase in spawner abundance in 2000 to 2002, the biological review team noted that the long-term trends in ESU productivity were still negative due to the low abundances observed during the 1990s.

Since the biological review team convened, the total abundance of natural spawners in the Oregon Coast coho ESU has declined each year (i.e., 2003 to 2006). The abundance of total natural spawners in 2006 (111,025 spawners) was approximately 43 % of the recent peak abundance in 2002 (255,372 spawners). In 2003, ESU-level productivity (evaluated in terms of the number of spawning recruits resulting from spawners 3 years earlier) was above replacement, and in 2004, productivity was approximately at replacement level. However, productivity was below replacement in 2005 and 2006, and dropped to the lowest level since 1991 in 2006 (73 FR 7816).

Preliminary spawner survey data for 2007 (the average peak number of spawners per mile observed during random coho spawning surveys in 41 streams) suggest that the 2007 to 2008 return of Oregon Coast coho is either (1) much reduced from abundance levels in 2006, or (2) exhibiting delayed run timing from previous years. As of December 13, 2007, the average peak number of spawners per mile was below 2006 levels in 38 of 41 surveyed streams (see ODFW 2007 *in* 73 FR 7816). It is possible that the timing of peak spawner abundance is delayed relative to previous years, and that increased spawner abundance in late December and January 2008 will compensate for the low levels observed thus far.

The recent 5-year geometric mean abundance (2002 to 2006) of approximately 152,960 total natural spawners remains well above that of a decade ago (approximately 52,845 from 1992 to

1996). However, the decline in productivity from 2003 to 2006, despite generally favorable marine survival conditions and low harvest rates, is of concern (73 FR 7816).

Critical Habitat
NMFS designated critical habitat for Oregon Coast coho on February 11, 2008 (73 FR 7816). The designation includes 72 of 80 watersheds occupied by Oregon Coast coho salmon, and totals about 6,600 stream miles including all or portions of the Nehalem, Nestucca/Trask, Yaguina, Alsea, Umpqua and Coquille basins. These areas are essential for feeding, migration, spawning, and rearing. The specific primary constituent elements include: spawning sites with water and substrate quantity to support spawning, incubation, and larval development; freshwater rearing sites with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth, foraging, behavioral development (e.g., predator avoidance, competition), and mobility; freshwater migratory corridors free of obstruction with adequate water quantity and quality conditions; and estuarine, nearshore and offshore areas free of obstruction with adequate water quantity, quality and salinity conditions that support physiological transitions between fresh- and saltwater, predator avoidance, foraging and other life history behaviors.

Final Protective Regulations
ESA section 4(d) regulations for Oregon Coast coho were originally proposed on December 30, 1999 (64 FR 73479). These regulations for Oregon Coast coho were invalidated when the underlying listing was vacated in 2001. In 2004 NMFS proposed to again list Oregon Coast coho and to reinstate the 4(d) regulations. In the final rule, published February 11, 2008 (73 FR 7816), NMFS applied the 4(d) protective regulations adopted for other Pacific salmonids, as amended in June 2005 (70 FR 37160), to the Oregon Coast coho ESU. The final rule is substantially the same as that proposed in 1999.

Sockeye Salmon

Description of the Species
Sockeye salmon are the second most abundant of the seven Pacific salmon species, and occur in the North Pacific and Arctic oceans and associated freshwater systems. This species ranges south as far as the Sacramento River in California and northern Hokkaido in Japan, to as far north as far as Bathurst Inlet in the Canadian Arctic and the Anadyr River in Siberia (Burgner 1991). The largest populations, and hence the most important commercial populations, occur north of the Columbia River.

The majority of sockeye salmon are anadromous fish that make use of lacustrine habitat for juvenile rearing. Sockeye salmon also have a wholly freshwater life history form, called kokanee (Burgner 1991). In some cases a single population will give rise to both the anadromous and freshwater life history form. While in fresh water juveniles of both life history types prey primarily upon insects. In coastal lakes, where the migration to sea is relatively short and energetic costs are minimal, kokanee populations are rare.

Once smolts enter the Pacific Ocean, they distribute widely across the North Pacific, generally above 40ºN where a current boundary is located. Season, temperature, salinity, life stage, age, size, availability of prey and population-of-origin are all factors that influence offshore movements (Burgner 1991). They may migrate several thousand miles in search of prey and are considered to travel continuously (Royce et al. 1968). While at sea, sockeye prey upon a variety of organisms, including small fish (capelin, lantern fish, cod, sand lance, herring and pollock), squid, crustacean larvae, krill and other invertebrates (Foerster 1968, French et al. 1976, Wing 1977).

Spawning generally occurs in late summer and autumn, but the precise time can vary greatly among populations. Age at maturity varies by region from 2 to 5 years, but is generally 2 to 4 years in Washington State (Burgner 1991). Males often arrive earlier than females on the spawning grounds, and will persist longer during the spawning period.
Incubation is a function of water temperatures, but generally lasts between 100 and roughly 200 days (Burgner 1991). After emergence, fry move rapidly downstream or upstream along the banks to the lake rearing area. Fry emerging from lakeshore or island spawning grounds may simply move along the shoreline of the lake (Burgner 1991).

Ozette Lake Sockeye Salmon

Distribution and Description of the Listed Species
This ESU includes all naturally spawned sockeye salmon in Ozette Lake, Ozette River, Coal Creek, and other tributaries flowing into Ozette Lake, Washington. Composed of only one population, the Ozette Lake sockeye salmon ESU consists of five spawning aggregations or subpopulations which are grouped according to their spawning locations. The five spawning locations are Umbrella and Crooked creeks, Big River, and Olsen's and Allen's beaches (NMFS 2009).

Adult Ozette Lake sockeye salmon enter Ozette Lake through the Ozette River from mid-April to mid-August, holding three to nine months in Ozette Lake prior to spawning in late October through January. Sockeye salmon spawn primarily in lakeshore upwelling areas in Ozette Lake (particularly at Allen's Bay and Olsen's Beach), and in two tributaries Umbrella Creek and Big River. Minor spawning may occur below Ozette Lake in the Ozette River or in Coal Creek, a tributary of the Ozette River. Beach spawners are almost all age-4 adults, while tributary spawners are ages 3 and 5 (Haggerty *et al.,* 2009 in NMFS 2009). Spawning occurs in the fall through early winter, with peak spawning in tributaries in November and December. Eggs and alevins remain in the gravel until the fish emerge as fry in spring. Fry then migrate immediately to the limnetic zone in Ozette Lake, where the fish rear. After one year of rearing, in late spring, Ozette Lake sockeye salmon emigrate seaward as age-1+ smolts, where they spend between 1 and 3 years in ocean before returning to fresh water.

Status and Trends
NMFS originally listed Ozette Lake sockeye salmon ESU as a threatened species in 1999 (64 FR 14528). This classification was retained on June 28, 2005 (70 FR 37160). This ESU includes all naturally spawned populations of sockeye salmon in Ozette Lake, Ozette River, Coal Creek, and

other tributaries flowing into Ozette Lake, Washington. Two artificial propagation programs are considered part of this ESU: The Umbrella Creek and Big River sockeye salmon hatchery programs. NMFS considers these artificially propagated populations no more divergent relative to the local natural population than would be expected between closely related natural populations (70 FR 37160).

The historical abundance of Ozette Lake sockeye salmon is poorly documented, but may have been as high as 50,000 individuals (Blum 1988). The overall abundance of naturally–produced Ozette Lake sockeye salmon is believed to have declined substantially from historical levels. In the first study of lake escapement of Ozette Lake sockeye salmon (Kemmerich 1945), the run size entering the lake was estimated at a level of several thousand fish. These counts appear to be roughly double the current mean lake abundance, considering that they were likely conducted upstream from fisheries in or near to the Ozette River. Makah Fisheries Management (as cited in Good *et al.,*, 2005*)* concluded that there appears to be a substantial decline in the Tribal catch of Ozette Lake sockeye salmon beginning in the 1950s and a similar decline in the run size since the 1920s weir counts reported by Kemmerich (1945).

An analysis of total annual Ozette Lake sockeye salmon abundance (based on adult run size data presented in Jacobs *et al.,* 1996) indicates a trend in abundance averaging -2% per year over the period 1977 through 1998 (NMFS 1998d). The current tributary-based hatchery program was planned and initiated in response to the declining population trend identified for the Ozette Lake sockeye salmon population. The most recent (1996 to 2003) run-size estimates range from a low of 1,609 in 1997 to a high of 5,075 in 2003, averaging approximately 3,600 sockeye per year (NMFS 2009). For return years 2000 to 2003, the 4-year average abundance estimate was slightly over 4,600 sockeye. Because run-size estimates before 1998 are likely to be even more unreliable than recent counts, and new counting technology has resulted in an increase in estimated run sizes, no statistical estimation of trends is reported. The current trends in abundance are unknown for the beach spawning aggregations. Although overall abundance appears to have declined from historical levels, whether this resulted in fewer spawning aggregations, lower abundances at each aggregation, or both, is not known (Good et al. 2005). Based on estimates of habitat carrying capacity, a viable sockeye salmon population in Lake Ozette watershed would range between 35,500 to 121,000 spawners (Rawson et al. 2009 as cited in NMFS 2009).

There has been no harvest of Ozette Lake sockeye salmon for the past four brood-cycle years (since 1982). Prior to that time, ceremonial and subsistence harvests by the Makah Tribe were low, ranging from 0 to 84 fish per year. Harvest has not been an important mortality factor for the population in over 35 years. In addition, due to the early river entry timing of returning Ozette Lake sockeye salmon (beginning in late April, with the peak returns prior to late-May to mid-June), the fish are not intercepted in Canadian and U.S. marine area fisheries directed at Fraser River sockeye salmon. There are currently no known marine area harvest impacts on Ozette Lake sockeye salmon.

Overall abundance is substantially below historical levels (Good et al. 2005). Declines in abundance have been attributed to a combination of introduced species, predation, loss of

tributary populations, a loss of quality of beach spawning habitat, temporarily unfavorable ocean conditions, habitat degradation, and excessive historical harvests (Jacobs et al. 1996). In the last few years the number of returning adults has increased, although some of these individuals are of hatchery origin. This produces uncertainty regarding natural growth rate and productivity of the ESU's natural component. In addition, genetic integrity has perhaps been compromised due to the artificial supplementation that has occurred in this population, since approximately one million sockeye have been released into the Ozette watershed from the late 1930s to present (Kemmerich 1945).

Critical Habitat

On September 2, 2005, NMFS designated critical habitat for the Ozette Lake sockeye salmon ESU (70 FR 52630). The specific geographic areas designated as critical are the Hoh/Quillayute Subbasin, Ozette Lake and the Ozette Lake watershed, and include: the Ozette River upstream to endpoints in Big River, Coal Creek, East Branch Umbrella Creek, the North and South Fork of Crooked Creek and several other tributaries. The specific primary constituent elements identified for Lake Ozette sockeye salmon are areas for spawning, freshwater rearing and migration, estuarine areas free of obstruction, nearshore marine areas free of obstructions, and offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, and adequate passage conditions. Only one watershed supports this ESU, and it is rated as having a high conservation value. This watershed is essential to the species' overall conservation by protecting quality growth, reproduction, and feeding.

Final Protective Regulations

On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the Ozette Lake sockeye salmon ESU.

Snake River Sockeye Salmon

Distribution and Description of the Listed Species

Snake River sockeye salmon are unique compared to other sockeye salmon populations: it spawns at a higher elevation (6,500 feet) and a longer freshwater migration (approximately 900 miles) than any other sockeye salmon population in the world. Sockeye salmon in this ESU spawn in Redfish Lake in Idaho's Stanley Basin (Bjornn et al. 1968, Foerster 1968). Stanley Basin sockeye salmon are separated by 700 or more river miles from two other extant upper Columbia River populations in the Wenatchee River and Okanogan River drainages. These latter populations return to lakes at substantially lower elevations (Wenatchee at 1,870 feet and Okanagon at 912 feet) and occupy different ecoregions. The Snake River sockeye salmon ESU includes all anadromous and residual sockeye salmon from the Snake River basin of Idaho, as well as hatchery individuals from the Redfish Lake Captive Broodstock Program.

Status and Trends

Snake River sockeye salmon were originally listed as endangered in 1991 and retained that classification when their status was reviewed on June 28, 2005 (70 FR 37160). The only extant sockeye salmon population in the Snake River basin at the time of listing was that in Redfish Lake, in the Stanley Basin (upper Salmon River drainage) of Idaho. Other lakes in the Snake River basin historically supported sockeye salmon populations, including Wallowa Lake (Grande Ronde River drainage, Oregon), Payette Lake (Payette River drainage, Idaho) and Warm Lake (South Fork Salmon River drainage, Idaho; (Waples et al. 1997). These populations are now considered extinct. Although kokanee, a resident form of *O. nerka*, occur in numerous lakes in the Snake River basin, other lakes in the Stanley Basin, and sympatrically with sockeye in Redfish Lake, resident *O. nerka* were not considered part of the species at the time of listing (1991). Subsequent to the 1991 listing, a residual form of sockeye residing in Redfish Lake was identified. The residuals are non-anadromous, completing their entire life cycle in fresh water, but spawn at the same time and in the same location as anadromous sockeye salmon. In 1993, NMFS determined that residual sockeye salmon in Redfish Lake were part of the Snake River sockeye salmon. Also, artificially propagated sockeye salmon from the Redfish Lake Captive Propagation program are considered part of this species (70 FR 37160; June 28, 2005).

Five lakes in the Stanley Basin historically contained sockeye salmon: Alturas, Pettit, Redfish, Stanley and Yellowbelly (Bjornn et al. 1968). It is generally believed that adults were prevented from returning to the Sawtooth Valley from 1910 to 1934 by Sunbeam Dam. Sunbeam Dam was constructed on the Salmon River approximately 20 miles downstream of Redfish Lake. Whether Sunbeam Dam was a complete barrier to adult migration remains unknown. It has been hypothesized that some passage occurred while the dam was in place, allowing the Stanley Basin population or populations to persist (Bjornn et al. 1968, Waples et al. 1991).

Adult returns to Redfish Lake during the period 1954 through 1966 ranged from 11 to 4,361 fish (Bjornn et al. 1968). Sockeye salmon in Alturas Lake were extirpated in the early 1900s as a result of irrigation diversions, although residual sockeye may still exist in the lake (Chapman and Witty 1993). From 1955 to 1965, the Idaho Department of Fish and Game eradicated sockeye salmon from Pettit, Stanley, and Yellowbelly lakes, and built permanent structures on each of the lake outlets that prevented re-entry of anadromous sockeye salmon (Chapman and Witty 1993). In 1985, 1986, and 1987, 11, 29, and 16 sockeye, respectively, were counted at the Redfish Lake weir (Good et al. 2005). Only 18 natural origin sockeye salmon have returned to the Stanley Basin since 1987. During the fall of 1990, during the course of NMFS' first status review on the species, no fish were observed at Lower Granit Dam or entering the lake and only one fish was observed in each of the two previous years. The first adult returns from the captive broodstock program returned to the Stanley Basin in 1999. From 1999 through 2005, a total of 345 captive brood program adults that had migrated to the ocean returned to the Stanley Basin.

Recent annual abundances of natural origin sockeye salmon in the Stanley Basin have been extremely low. No natural origin anadromous adults have returned since 1998 and the abundance of residual sockeye salmon in Redfish Lake is unknown. This species is entirely supported by adults produced through the captive propagation program at the present time. Current smolt-to-adult survival of sockeye originating from the Stanley Basin lakes is rarely

greater than 0.3% (Hebdon et al. 2004). The status of this ESU is extremely precarious, such that there was unanimous consent among the biological review team members that the species remains in danger of extinction (Good et al. 2005).

Critical Habitat
Critical habitat for these salmon was designated on December 28, 1993 (58 FR 68543), and encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed Snake River salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). Adjacent riparian zones are defined as those areas within a horizontal distance of 300 feet from the normal line of high water of a stream channel or from the shoreline of a standing body of water. Designated critical habitat includes the Columbia River from a straight line connecting the west end of the Clatsop jetty (Oregon side) and the west end of the Peacock jetty (Washington side) and including all river reaches from the estuary upstream to the confluence of the Snake River, and all Snake River reaches upstream to the confluence of the Salmon River; all Salmon River reaches to Alturas Lake Creek; Stanley, Redfish, yellow Belly, Pettit, and Alturas Lakes (including their inlet and outlet creeks); Alturas Lake Creek and that portion of Valley Creek between Stanley Lake Creek and the Salmon River. Critical habitat also includes all river lakes and reaches presently or historically accessible to Snake River sockeye salmon. These habitats are critical for the conservation of the species because it provides spawning and juvenile rearing habitat, areas for juvenile growth and development, and migration corridors for smolts to the ocean and adults to spawning habitat from the Pacific Ocean. Limiting factors identified for Snake River sockeye include: reduced tributary stream flow, impaired tributary passage and blocks to migration, and mainstem Columbia River hydropower system mortality.

Steelhead

Description of the Species
Steelhead, the common name of the anadromous form of *O. mykiss*, are native to Pacific Coast streams extending from Alaska south to northwestern Mexico (Moyle 1976a, NMFS 1997b). The life history of this species varies considerably throughout its range. Generally, steelhead comprise into two races: the stream-maturing type; and the ocean-maturing type.
Summer steelhead enter fresh water between May and October in the Pacific Northwest (Nickelson et al. 1992, Busby et al. 1996). They require cool, deep holding pools during summer and fall, prior to spawning (Nickelson et al. 1992). Summer steelhead migrate inland toward spawning areas, overwinter in the larger rivers, resume migration in early spring to natal streams, and then spawn in January and February (Barnhart 1986, Meehan and Bjornn 1991, Nickelson et al. 1992). Winter steelhead enter fresh water between November and April in the Pacific Northwest (Nickelson et al. 1992), migrate to spawning areas, and then spawn, generally in April and May (Barnhart 1986). Some adults, however, do not enter some coastal streams until spring, just before spawning (Meehan and Bjornn 1991).

In late spring, after emerging from the gravel, fry usually inhabit shallow water along banks of perennial streams (Nickelson et al. 1992). Summer rearing takes place primarily in the faster

parts of pools, while winter rearing occurs more uniformly at lower densities across a wide range of fast and slow habitat types. Some older juveniles move downstream to rear in larger tributaries and mainstem rivers (Nickelson et al. 1992).

There is a high degree of overlap in spawn timing between populations regardless of run type (Busby et al. 1996). Difficult field conditions at that time of year and the remoteness of spawning grounds contribute to the relative lack of specific information on steelhead spawning. Unlike Pacific salmon, steelhead are capable of spawning more than once before death, although steelhead rarely spawn more than twice before dying; most that do spawn more than twice tend to be female (Nickelson et al. 1992, Busby et al. 1996).

Juvenile steelhead migrate little during their first summer and occupy a range of habitats featuring moderate to high water velocity and variable depths (Bisson et al. 1988). Steelhead hold territories close to the substratum where flows are lower and sometimes counter to the main stream; from these, they can make forays up into surface currents to take drifting food (Kalleberg 1958). Juveniles rear in fresh water from 1 to 4 years, then smolt and migrate to the ocean in March and April (Barnhart 1986). Winter steelhead juveniles generally smolt after 2 years in fresh water (Busby et al. 1996). Juveniles feed primarily on insects (chironomids, baetid mayflies, and hydropsychid caddisflies (Merz 1994) while adults feed on aquatic and terrestrial insects, mollusks, crustaceans, fish eggs, minnows, and other small fishes (including greenling and other trout; (Chapman and Bjornn 1969) .

Threats

Natural Threats. Steelhead, like other salmon, are exposed to high rates of natural predation each stage of their life stage. Mortality is high early in life and decreases with age. For example, Puget Sound steelhead leaving freshwater and estuarine habitats experience 55-86% survival to the point of reaching Hood Canal and 0-49% from Hood Canal to the Strait of Juan de Fuca, with survival increasing greatly upon entering the Pacific Ocean (Moore et al. 2010). In fresh water, fry fall prey to older steelhead and other trout, as well as birds, sculpin, and various mammals. In the ocean, marine mammals, and other fish prey on steelhead but the extent of such predation is not well known.

Anthropogenic Threats. Steelhead have declined under the combined effects of overharvests in fisheries, competition from hatchery fish and exotic species, dams that block their migrations and alter river hydrology, hydrogeomorphological changes, destruction or degradation of riparian habitat and land use practices that destroy or degrade fresh water, estuarine, and coastal ecosystems throughout the species' range (These threats for are summarized in detail under the Chinook salmon section).

Central California Coast Steelhead

Distribution and Description of the Listed Species

The Central California Coast steelhead DPS includes all naturally spawned anadromous steelhead populations below natural and manmade impassable barriers in California streams from the Russian River (inclusive) to Aptos Creek (inclusive), and the drainages of San

Francisco, San Pablo, and Suisun Bays eastward to Chipps Island at the confluence of the Sacramento and San Joaquin Rivers. Tributary streams to Suisun Marsh including Suisun Creek, Green Valley Creek, and an unnamed tributary to Cordelia Slough (commonly referred to as Red Top Creek), excluding the Sacramento-San Joaquin River Basin, as well as two artificial propagation programs: the Don Clausen Fish Hatchery, and Kingfisher Flat Hatchery/ Scott Creek (Monterey Bay Salmon and Trout Project) steelhead hatchery programs.

The DPS is entirely composed of winter run fish, as are those DPSs to the south. As winter-run fish, adults migrate upstream December-April, and smolts emigrate between March and May (Shapovalov and Taft 1954, Hayes et al. 2008). At the time of the 1996 status review and 1997 listing, little information was available on the specific demographics and life history characteristics of steelhead in this DPS. While age at smoltification typically ranges from 1 to 4 years, recent studies by Sogard and Williams (2009) indicate that growth rates in Soquel Creek likely prevent juveniles from undergoing smoltification until age 2. Survival in freshwater reaches tends to be higher in summer and lower from winter through spring for year classes 0 and 1 (Sogard et al. 2009). Larger individuals also survive more readily than do smaller fish within year classes (Sogard et al. 2009). Greater movement of juveniles in fresh water has been observed in winter and spring versus summer and fall time periods, with smaller individuals more likely to move between stream areas (Sogard et al. 2009). Growth rates during this time have rarely been observed to exceed 0.3 mm per day and are highest in winter through spring, potentially due to higher water flow rates and greater food availability (Boughton et al. 2007, Hayes et al. 2008, Sogard et al. 2009).

Status and Trends

The Central California Coast steelhead DPS was listed as a threatened species on August 18, 1997 (62 FR 43937); threatened status was reaffirmed on January 5, 2006 (71 FR 834). Estimates of historical abundance are provided here only for background, as the accuracy of the estimates is unclear. An estimate of historical abundance for the total DPS is provided by CDFG at 94,000 fish. This estimate is based on a partial data set and "best professional judgment" (see Good et al. 2005). Other estimates of historical abundance are on a per river basis: Shapovalov and Taft (1954) (as cited in Busby et al. 1996) described an average of about 500 adults in Waddell Creek (Santa Cruz County) for the 1930s and early 1940s.

No current estimates of total population size are available for this DPS, and consequently there is no time series data available to evaluate the central California coast steelhead population trends. Rather, a general dearth of data on adult steelhead within the DPS, led the biological review team to examine data collected on juvenile steelhead (see Good et al. 2005). In general, juvenile data is considered a poor indicator of the reproductive portion of the population as juvenile age classes exhibit greater mortality rates, which are closely tied to stochastic events, and may move widely within a basin (which may include intermixing with other populations). There is no simple relationship between juvenile and adult numbers (Shea and Mangel 2001). Nonetheless, there was not enough adult data upon which the biological review team could base an assessment of the population trends within the DPS. Therefore, the biological review team log-transformed and normalized juvenile survey data from a number of watersheds (presumed populations). As a result, the team derived trend estimates for five populations: the San Lorenzo River, Scott

Creek, Waddell Creek, Gazos Creek, and Redwood Creek in Marin County (see Good et al. 2005). All populations exhibited downward trends in abundance. Accordingly, provided the juvenile data is representative of the true trend, this data suggests that there is an overall downward trend in abundance in the DPS.

In the most recent review of the status of this DPS, most members of the biological review team (69 %) considered this DPS "likely to become endangered" thus supporting the renewal of the threatened status for central California coast steelhead. Notably, 25 % of the team voted that the DPS be upgraded to endangered status (see Good et al. 2005). Abundance and productivity were of relatively high concern (as a contributing factor to risk of extinction), and spatial structure was also of concern.

Since the original status review, fishing regulations have changed in a way that probably reduces extinction risk for Central California Coast steelhead. Ocean sport harvest is prohibited, and ocean harvest is considered rare. Although freshwater streams are closed to fishing year round, CDFG has identified certain streams as exceptions where they allow catch-and-release angling or summer trout fishing. In catch-and-release streams, all wild steelhead must be released unharmed.

Critical Habitat

Critical habitat was designated for the Central California Coast steelhead DPS on September 2, 2005 (70 FR 52488), and includes areas within the following hydrologic units: Russian River, Bodega, Marin Coastal, San Mateo, Bay Bridge, Santa Clara, San Pablo, and Big Basin. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat 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. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, Central California Coast steelhead occupy 46 watersheds (fresh water and estuarine). The total area of habitat designated as critical includes about 1,500 miles of stream habitat and about 400 square miles of estuarine habitat (principally Humboldt Bay). This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 46 occupied watersheds reviewed in NMFS' assessment of critical habitat for Central California Coast steelhead, 14 watersheds received a low rating of conservation value, 13 received a medium rating, and 19 received a high

rating of conservation value for the species.

Final Protective Regulations
On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the Central California Coast steelhead DPS.

California Central Valley Steelhead

Distribution and Description of the Listed Species
California Central Valley steelhead salmon occupy the Sacramento and San Joaquin Rivers and their tributaries, although they were once widespread throughout the Central Valley (Busby et al. 1996, Zimmerman et al. 2009). Steelhead were found from the upper Sacramento and Pit River systems (now inaccessible due to Shasta and Keswick Dams), south to the Kings and possibly the Kern River systems (now inaccessible due to extensive alteration from water diversion projects), and in both east- and west-side Sacramento River tributaries (Yoshiyama et al. 1996). The present distribution has been greatly reduced (McEwan and Jackson 1996). The CACSS (1988) reported a reduction of steelhead habitat from 6,000 miles historically to 300 miles today. Historically, steelhead probably ascended Clear Creek past the French Gulch area, but access to the upper basin was blocked by Whiskeytown Dam in 1964 (Yoshiyama et al. 1996). Steelhead also occurred in the upper drainages of the Feather, American, Yuba and Stanislaus Rivers which are now inaccessible (McEwan and Jackson 1996, Yoshiyama et al. 1996).

Existing wild steelhead populations in the Central Valley are mostly confined to the upper Sacramento River and its tributaries, including Antelope, Deer, and Mill Creeks and the Yuba River. Populations may exist in Big Chico and Butte Creeks and a few wild steelhead are produced in the American and Feather Rivers (McEwan and Jackson 1996). Recent snorkel surveys (1999 to 2002) indicate that steelhead are present in Clear Creek (Good et al. 2005). Because of the large resident *O. mykiss* population in Clear Creek, steelhead spawner abundance has not been estimated. Until recently, steelhead were thought to be extirpated from the San Joaquin River system. Recent monitoring has detected small self-sustaining populations of steelhead in the Stanislaus, Mokelumne, Calaveras, and other streams previously thought to be void of steelhead (McEwan 2001). On the Stanislaus River, steelhead smolts have been captured in rotary screw traps at Caswell State Park and Oakdale each year since 1995 (Demko and Cramer 2000). It is possible that naturally spawning populations exist in many other streams but are undetected due to lack of monitoring programs.

The Sacramento and San Joaquin Rivers offer the only migration route to the drainages of the Sierra Nevada and southern Cascade mountain ranges for anadromous fish. The CDFG considers all steelhead in the Central Valley as winter steelhead, although “three distinct runs,” including summer steelhead, may have occurred there as recently as 1947 (McEwan and Jackson

1996). Steelhead in these basins travel extensive distances in fresh water (some exceed 300 km to their natal streams), making these the longest freshwater migrations of any population of winter steelhead. The upper Sacramento River essentially receives a single continuous run of steelhead in from July through May, with peaks in September and February. Spawning begins in late December and can extend into April (McEwan and Jackson 1996).

Status and Trends

NMFS originally listed California Central Valley steelhead as threatened in 1998; this status was reviewed and retained on January 5, 2006 (71 FR 834). Historic Central Valley steelhead run size is difficult to estimate given the paucity of data, but 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. Hallock *et al.,* (1961) estimated an average of 20,540 adult steelhead occurred in the Sacramento River (upstream of the Feather River). Steelhead counts at Red Bluff Diversion Dam declined from an average of 11,187 for the period of 1967 to 1977, to an average of approximately 2,000 through the early 1990s, with an estimated total annual run size for the entire Sacramento-San Joaquin system at no more than 10,000 adults (McEwan and Jackson 1996, McEwan 2001). The five-year geometric mean, however, is just under 2,000 steelhead and the long-term trend suggests that the population is declining (Good *et al.,* 2005).

The only consistent data available on steelhead numbers in the San Joaquin River basin come from CDFG mid-water trawling samples collected on the lower San Joaquin River at Mossdale. These data indicate a decline in steelhead numbers in the early 1990s, which have remained low through 2002 (Good et al. 2005). In 2004, a total of 12 steelhead smolts were collected at Mossdale (Good et al. 2005).

Reynolds *et al.,* (1993) reported that 95% of salmonid habitat in California's Central Valley has been lost, largely due to mining and water development activities. They also noted that declines in Central Valley steelhead populations are "due mostly to water development, inadequate instream flows, rapid flow fluctuations, high summer water temperatures in streams immediately below reservoirs, diversion dams which block access, and entrainment of juveniles into unscreened or poorly screened diversions." Thus, overall habitat problems in this ESU relate primarily to water development resulting in inadequate flows, flow fluctuations, blockages, and entrainment into diversions (McEwan and Jackson 1996). Other problems related to land use practices (agriculture and forestry) and urbanization have also contributed to population declines. It is unclear how harvest has affected California's Central Valley steelhead, although it is likely a continuing threat. A CDFG creel census in 2000 indicated that most fish are caught and released, but due to the size of the catch and release fishery (more than 14,000 steelhead were caught and released according to the survey) even a small amount of mortality in this fishery could cause declines in the populations.

Critical Habitat

NMFS designated critical habitat for California Central Valley steelhead on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER

hydrological units: Tehama, Whitmore, Redding, Eastern Tehama, Sacramento Delta, Valley-Putach-Cache, American River, Marysville, Yuba, Valley American, Colusa Basin, Butte Creek, Ball Mountain, Shata Bally, North Valley Floor, Upper Calaveras, Stanislaus River, San Joaquin Valley, Delta-Mendota Canal, North Diablo Range and the San Joaquin Delta. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat 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. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, California Central Valley steelhead occupy 67 watersheds (freshwater and estuarine). The total area of habitat designated as critical includes about 2,300 miles of stream habitat and about 250 square miles of estuarine habitat in the San Francisco-San Pablo-Suisan Bay estuarine complex. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 67 watersheds reviewed in NMFS' assessment of critical habitat for California Central Valley steelhead, seven watersheds received a low rating of conservation value, three received a medium rating, and 27 received a high rating of conservation value for the species.

Final Protective Regulations
On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the California Central Valley steelhead DPS.

Lower Columbia River Steelhead

Distribution and Description of the Listed Species
Lower Columbia River steelhead include naturally produced steelhead returning to Columbia River tributaries on the Washington side between the Cowlitz and Wind rivers in Washington and on the Oregon side between the Willamette and Hood rivers, inclusive. In the Willamette River, the upstream boundary of this species is at Willamette Falls. This species includes both winter and summer steelhead. Two hatchery populations are included in this species, the

Cowlitz Trout Hatchery winter-run population and the Clackamas River population but neither was listed as threatened

Sexually immature summer steelhead return to the Columbia River from May to November and spend several months in fresh water prior to spawning. When winter steelhead enter fresh water from November to April, they are close to sexual maturity and spawn shortly after arrival in their natal streams. Where both races spawn in the same stream, summer steelhead tend to spawn at higher elevations than the winter forms (see Good *et al.,* 2005).

Status and Trends

NMFS listed Lower Columbia River steelhead as threatened on March 19, 1998 (63 FR 13347), and reaffirmed their status as threatened on January 5, 2006 (71 FR 834). The 1998 status review noted that this ESU is characterized by populations at low abundance relative to historical levels, significant population declines since the mid-1980s, and widespread occurrence of hatchery fish in naturally spawning steelhead populations. During this review NMFS was unable to identify any natural populations that would be considered at low risk.

All populations declined between 1980 and 2000, with sharp declines beginning in 1995. Those with adequate data for modeling are estimated to have a high extinction risk (Good et al. 2005). Abundance trends are generally negative, showing that most populations are in decline, although some populations, particularly summer run, have shown higher return in the last 2 to 3 years (Good et al. 2005). Historical counts in some of the larger tributaries (Cowlitz, Kalama and Sandy Rivers) suggest the population probably exceeded 20,000 fish while in the 1990s fish abundance dropped to 1,000 to 2,000. Recent abundance estimates of natural-origin spawners range from completely extirpated for some populations above impassable barriers to over 700 for the Kalama and Sandy winter-run populations (Good et al. 2005). A number of the populations have a substantial fraction of hatchery-origin spawners in spawning areas, and are hypothesized to be sustained largely by hatchery production. Exceptions are the Kalama, the Toutle, and East Fork Lewis winter-run populations (Good et al. 2005).

Critical Habitat

NMFS designated critical habitat for Lower Columbia River steelhead on September 2, 2005 (70 FR 52630). Designated critical habitat includes the following subbasins: Middle Columbia/Hood subbasin, Lower Columbia/Sandy subbasin, Lewis subbasin, Lower Columbia/Clatskanie subbasin, Upper Cowlitz subbasin, Cowlitz subbasin, Clackamas subbasin, Lower Willamette subbasin and the Lower Columbia River corridor. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat 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. The critical habitat designation (70 FR 52630) contains additional description of the watersheds that are included as part of this designation, and any areas specifically excluded from the designation.

In total, Lower Columbia River steelhead occupy 32 watersheds. The total area of habitat designated as critical includes about 2,340 miles of stream habitat. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. Of the 32 watersheds reviewed in NMFS' assessment of critical habitat for Lower Columbia River steelhead, two watersheds received a low rating of conservation value, 11 received a medium rating, and 26 received a high rating of conservation value for the species. Limiting factors identified for Lower Columbia River steelhead include: degraded floodplain and steam channel structure and function, reduced access to spawning/rearing habitat, altered stream flow in tributaries, excessive sediment and elevated water temperatures in tributaries, and hatchery impacts.

Final Protective Regulations
On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the Lower Columbia River steelhead DPS.

Middle Columbia River Steelhead

Distribution and Description of the Listed Species
The Middle Columbia River steelhead DPS includes all naturally spawned anadromous steelhead populations below natural and manmade impassible barriers in Oregon and Washington drainages upstream of the Hood and Wind River systems, up to and including the Yakima River (61 FR 41541). Steelhead from the Snake River Basin are excluded from this DPS. Seven artificial propagation programs are part of this DPS.

Middle Columbia River steelhead occupy the intermountain region of the Pacific Northwest, which includes some of the driest areas in the region generally receiving less than 15.7 inches of rainfall annually. Major drainages in this ESU are the Deschutes, John Day, Umatilla, Walla Walla, Yakima and Klickitat river systems. The area is generally characterized by its dry climate and harsh temperature extremes. Almost all steelhead populations within this DPS are summer-run fish; the only exceptions are the only populations of inland winter steelhead, which occur in the Klickitat River and Fifteen-mile Creek (Busby et al. 1996). According to Interior Columbia Basin Technical Recovery Team (ICTRT 2003) this DPS is comprised of 16 putative populations in four major population groups (Cascades Eastern Slopes Tributaries, John Day River, Walla Walla and Umatilla Rivers, and Yakima River) and one unaffiliated independent population (Rock Creek).

There are two extinct populations in the Cascades Eastern Slope major population group, the

White Salmon River and Deschutes Crooked River above the Pelton/Round Butte Dam complex. Present population structure is delineated largely on the basis of geographical proximity, topography, distance, ecological similarities or differences. Additional genetic studies are needed to describe the DPS substructure, as well as the fine-scale genetic structure of the populations within a particular basin (e.g., John Day River).

Most Middle Columbia River steelhead smolt at 2 years of age and spend 1 to 2 years at sea prior to re-entering natal river systems. They may remain in such rivers for up to a year prior to spawning (Howell et al. 1985b). Factors contributing to the decline of Middle Columbia river steelhead include hydropower development and agriculture; these land uses impede or prevent migrations, alter water availability, and alter water chemistry and temperatures.

Status and Trends

Middle Columbia River steelhead were listed as threatened in 1999 (64 FR 14517), and their status was reaffirmed on January 5, 2006 (71 FR 834). The precise pre-1960 abundance of this species is unknown. Based upon the Washington Department of Fish and Wildlife's estimates of the historic run size for the Yakima River at 100,000 steelhead, Busby *et al.,* (1996) surmised that total DPS abundance likely exceeded 300,000 returning adults. By 1993, the estimated 5-year average size (ending in 1993) of the Middle Columbia steelhead DPS was 142,000 fish (Busby et al. 1996). Survey data collected between 1997 and 2001 indicates that several populations within the DPS have increased since the last status review (Good et al. 2005). However, long-term annual population growth rate (λ) is negative for most populations.
In contrast, short term trends in major areas were positive for 7 of the 12 areas with available data (see Good *et al.,* 2005). Spawner numbers in the Yakima River, the Deschutes River and sections of the John Day River system were substantially higher compared to numbers surveyed between 1992 and 1997 (Good et al. 2005). Similarly, spawner numbers substantially increased in the Umatilla River and Fifteen-mile Creek relative to annual levels in the early 1990s. Nonetheless, most populations remain below interim target levels. For instance, the Yakima River returns are still substantially below interim target levels of 8,900 (the current 5-year average is 1,747 fish) and estimated historical return levels. In fact, the majority of spawning occurs in only one tributary, Satus Creek (Berg 2001as cited in Good et al., 2005). Based on recent 5-year geometric means, only the Deschutes River exceeded interim target levels (Good et al. 2005). While increases in short-term trends could suggest improvements within the DPS, given that the average population growth rate across all streams is negative (0.98 assuming hatchery spawners do not contribute to production, and 0.97 assuming that both hatchery and natural-origin fish contribute equally) and evidence of large fluctuation in marine survival for the species, recent increases in population sizes must be viewed cautiously (Good et al. 2005).

Critical Habitat

NMFS designated critical habitat for Middle Columbia River steelhead on September 2, 2005 (70 FR 52630). Designated critical habitat includes the following subbasins: Upper Yakima, Naches, Lower Yakima, Middle Columbia/Lake Wallula, Walla Walla, Umatilla, Middle Columbia/Hood, Klickitat, Upper John Day, North Fork John Day, Middle Fork John Day, Lower John Day, Lower Deschutes, Trout, and the Upper Columbia/Priest Rapids subbasins, and the Columbia River corridor. These areas are important for the species' overall conservation by

protecting quality growth, reproduction, and feeding. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat 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. The final rule (70 FR 52630) lists the watersheds that comprise the designated subbasins and any areas that are specifically excluded from the designation.

In total, there are 114 watersheds within the range of Middle Columbia River steelhead. The total area of habitat designated as critical includes about 5,800 miles of stream habitat. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. Of the 114 watersheds reviewed in NMFS' assessment of critical habitat for Middle Columbia River steelhead, nine watersheds received a low rating of conservation value, 24 received a medium rating, and 81 received a high rating of conservation value for the species. Although pristine habitat conditions are still present in some wilderness, roadless, and undeveloped areas, habitat complexity has been greatly reduced in many areas of designated critical habitat for Middle Columbia River steelhead. Limiting factors identified for Middle Columbia River steelhead include: hydropower system mortality, reduced stream flow, impaired passage, excessive sediment, degraded water quality, and altered channel morphology and floodplain.

Final Protective Regulations
On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the Lower Columbia River steelhead DPS.

Northern California Steelhead

Distribution and Description of the Listed Species
The Northern California DPS of steelhead includes all naturally spawned steelhead populations below natural and manmade impassible barriers in California coastal river basins from Redwood Creek south to, but not including the Russian river, and two artificial propagation programs (Yager Creek Hatchery, and North Fork Gualala River Hatchery). In the recent update on the status of this DPS, the southern boundary of the DPS was redefined to include the small coastal streams south of the Gualala River (between the Gualala River and the Russian River) that support steelhead. This DPS consists of winter and summer-run fish, as well as “half-pounders” – a steelhead that returns from the sea after spending less than a year in the ocean.

Status and Trends

NMFS listed Northern California steelhead as threatened on June 7, 2000 (65 FR 36074), and reaffirmed their status as threatened on January 5, 2006 (71 FR 834). Long-term data sets are limited for Northern California steelhead. Before 1960, estimates of abundance specific to this DPS were available from dam counts in the upper Eel River (Cape Horn Dam; annual average number of adults was 4,400 in the 1940s), the South Fork Eel River (Benbow Dam; annual average number of adults was 18,000 in the 1940s), and the Mad River (Sweasey Dam; annual average number of adults was 3,800 in the 1940s). According to California Department of Fish & Game nearly 200,000 spawning steelhead may have comprised this DPS in the early 1960s (Good et al. 2005). At the time of the first status review on this population, adult escapement trends could be calculated for seven populations. Five of the seven populations exhibited declines, while two exhibited increases with a range of almost 6% annual decline to a 3.5% increase. At the time, little information was available on the actual contribution of hatchery fish to natural spawning, there was and continues to be insufficient information to calculate an overall abundance estimate for Northern California steelhead (Busby et al. 1996).

Recent time series data is also limited for this DPS, with recent abundance estimates available for only four populations, three summer-run and one winter-run. Similarly, Good *et al.,* (2005) could only calculate the population growth rate for three populations. Population growth rates are negative for two of the three populations, the South Fork Eel River winter-run and the Middle Fork Eel River summer-run. Based on time series data for the Middle Fork Eel River, both the long-term and short-term trends are downward. Due to the lack of adult data on which to base their risk assessment, Good *et al.,* (2005) also examined data on juvenile steelhead, and found both upward and downward trends. The lack of data for the populations within this DPS, particular winter-run fish is of continuing concern.

Critical Habitat

NMFS designated critical habitat for Northern California steelhead on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Redwood Creek, Trinidad, Mad River, Eureka Plain, Eel River, Cape Mendocino and the Mendocino Coast. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat 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. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, Northern California steelhead occupy 50 watersheds (fresh water and estuarine). The total area of habitat designated as critical includes about 3,000 miles of stream habitat and about 25 square miles of estuarine habitat, mostly within Humboldt Bay. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by

the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 50 watersheds reviewed in NMFS' assessment of critical habitat for Northern California steelhead, nine watersheds received a low rating of conservation value, 14 received a medium rating, and 27 received a high rating of conservation value for the species. Two estuarine areas used for rearing and migration (Humboldt Bay and the Eel River estuary) also received a rating of high conservation value.

Final Protective Regulations
On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the Lower Columbia River steelhead DPS.

Puget Sound Steelhead

Distribution and Description of the Listed Species
The Puget Sound DPS for steelhead includes all naturally spawned anadromous winter-run and summer-run steelhead populations in watersheds of the Strait of Juan de Fuca, Puget Sound and Hood Canal, Washington. Boundaries of this DPS extend to and include the Elwha River to the west, and the Nooksack River and Dakota Creek to the north. Hatchery production of steelhead is widespread throughout this DPS, but only two artificial propagation programs are part of this DPS: the Green River natural and Hamma Hamma winter-run steelhead hatchery populations. The remaining hatchery programs are not considered part of the Puget Sound steelhead DPS because they are more than moderately diverged from the local native populations (Hard et al. 2007).

The oceanic distribution of Puget Sound steelhead is not well understood. Winter and summer runs from multiple DPS' comingle in the North Pacific Ocean and some may undergo extensive migrations as a result of the location of their natal streams and oceanic "centers of abundance" (Light et al. 1989). Tagging and genetic studies indicate that Puget Sound steelhead migrate to the central North Pacific ocean (see French et al. 1975, Hartt and Dell 1986, and Burgner et al. 1992 in Hard et al. 2007). Oceanic residence times varies among populations within the DPS, with some populations spending only one season in the ocean and others spending three years in marine waters before returning to their natal stream for spawning. Generally, winter-run steelhead enter their natal freshwater systems later (November to April) in the year than summer-run steelhead (May to October), and thus have a shorter freshwater residence time just prior to spawning. The result is that winter-run steelhead have a lower pre-spawn mortality rate than summer-run steelhead (Hard et al. 2007). Winter-run steelhead are also more prevalent than

summer-run fish, comprising 37 of the 53 populations within this DPS.

Status and Trends

NMFS listed Puget Sound steelhead as a threatened species on May 11, 2007 (72 FR 26722). At the time of the listing, the biological review team concluded that: the viability of Puget Sound steelhead is at a high risk due to declining productivity and abundance; Puget Sound steelhead are at moderate risk due to reduced spatial complexity and connectivity among populations within the DPS, and reduction in life-history diversity within populations and from the threats posed by artificial propagation and harvest. The Puget Sound steelhead DPS includes 53 putative populations; most of which are composed of winter-run fish. Summer-run populations within Puget Sound are small, with most averaging less than 200 spawners, and most lack sufficient data to estimate population abundance (Hard et al. 2007).

In general, steelhead are most abundant in the northern Puget Sound streams (Hard et al. 2007). The largest populations in this DPS are in the Skagit River and Snohomish River winter-run steelhead populations. The recent geometric mean escapement is 5,608 winter-run steelhead in the Skagit, and 3,230 winter-run steelhead in the Snohomish River (Hard et al. 2007). The Green River and Puyallup River populations, in central Puget Sound, are the next largest populations and average approximately 1,500 (Green) and 1,000 (Puyallup) winter-run steelhead spawners annually (Hard et al. 2007).

Estimates of historical abundance for this DPS are largely based on catch data. The earliest catch records from commercial fisheries in the late 1880s indicate that the catch peaked at 163,796 steelhead in Puget Sound in 1895 (Hard et al. 2007). Based on this catch data, estimates for the peak run size for Puget Sound steelhead ranges between 300,000 and 550,000 fish (Hard et al. 2007). Given that most fish were harvested in terminal fisheries (nets set at the mouth of rivers) NMFS expects that this estimate is a fair estimate of the Puget Sound DPS as it is unlikely to include fish from neighboring rivers outside of the Puget Sound DPS. As early as 1898, Washington officials expressed concerns that the run had declined by half of its size in only three years (Hard et al. 2007). Since 1925, Washington has managed steelhead as a game fish, and in 1932 the State prohibited the commercial catch, possession or sale of steelhead.

Run size for this DPS was calculated in the early 1980s at about 100,000 winter-run fish and 20,000 summer-run fish. It is not clear what portion were hatchery fish, but a combined estimate with coastal steelhead suggested that roughly 70% of steelhead in ocean runs were of hatchery origin. Escapement of wild fish to spawning grounds would be much lower without the influx of hatchery fish (Busby et al. 1996).

NMFS first status review for Puget Sound steelhead demonstrated that 80 % of the runs for which there was data had declining trends in abundance. Busby *et al.,* (1996) noted that the largest decline, an 18% annual decline, occurred in the Lake Washington population. On the contrary, the largest increase in abundance occurred in the Skykomish River winter-run steelhead (the Skykomish River is a tributary to the Snohomish River) at a 7% annual increase. Estimates of spawner abundance in the Skagit and Snohomish rivers, the two largest steelhead producing basins in the DPS, were about 8,000 naturally spawning adult steelhead each. These two basins exhibited modest overall upward trends at the time of the first status review. Recent data demonstrates significant declines in the natural escapement of steelhead throughout the DPS,

especially in the southern Puget Sound populations. Significant positive trends have occurred in the Samish and the Hamma Hamma winter-run populations. The increasing trend in the Hamma Hamma River appears to be the result of a captive rearing program, rather than due to natural escapement. The predominant downward trends in escapement and run size of natural steelhead in the Puget Sound DPS, both over the long-term and short-term, is of concern particularly given that despite widespread reductions in direct harvest since the mid 1990s (Hard et al. 2007).

Critical Habitat
NMFS has not designated critical habitat for Puget Sound steelhead.

Final Protective Regulations
On March 29, 2006, NMFS proposed to list the Puget Sound steelhead DPS as a threatened species (71 FR 15666) and on February 7, 2007 (72 FR 5648), NMFS proposed protective regulations for Puget Sound steelhead under section 4(d) of the ESA. On May 11, 2007, NMFS issued a final determination listing the Puget Sound steelhead DPS as threatened, and announced that it would finalize protective regulations in a subsequent Federal Register notice (72 FR 26722). In this final rule NMFS applied the 4(d) protective regulations adopted for other Pacific salmonids, as amended in June 2005 (70 FR 37160) described in *the Protective Regulations for Threatened Salmonid Species* section of this document, to Puget Sound steelhead. These section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild.

Snake River Steelhead

Distribution and Description of the Listed Species
The Snake River Basin steelhead DPS includes all naturally spawned populations of steelhead in streams in the Snake River basins of southeast Washington, northeast Oregon and Idaho. Six artificial propagation programs are considered part of this DPS: The Tucannon River, Dworshak National Fish Hatchery, Lolo Creek, North Fork Clearwater, East Fork Salmon River and the Little Sheep Creek/Imnaha river hatchery programs.

Snake River Basin steelhead are distributed throughout the Snake River drainage basin, migrating a considerable distance from the ocean to use high-elevation tributaries (typically 1,000-2,000 m above sea level) (Good et al. 2005). Generally, classified as summer-run fish, Snake River steelhead enter the Columbia River from late June to October. (Good et al. 2005) After remaining in the river through the winter, Snake River steelhead spawn the following spring (March to May) (Good et al. 2005). Managers recognize two life history patterns within Snake River steelhead primarily based on ocean age and adult size upon return: A-run steelhead are typically smaller, have a shorter fresh water and ocean residence (generally 1 year in the ocean) , and begin their up-river migration earlier in the year; whereas B-run steelhead are larger, spend more time in fresh water and the ocean (generally 2-years in ocean), and appear to start their upstream migration later in the year (Good et al. 2005).

Status and Trends
NMFS listed Snake River steelhead as threatened in 1997 (62 FR 43937), and reaffirmed their

status as threatened on January 5, 2006 (71 FR 834). NMFS 1997 status review identified sharp declines in the returns of naturally produced steelhead, beginning in the mid-1980s. At the time nine of 13 trend indicators were in decline and the average abundance (geometric mean, 1992-1996) for the DPS was 75,000 adult steelhead (8,900 naturally produced). Of this, about 7,000 were A-run adults, and about 1,400 were B-run adults (Busby et al. 1996).

The lack of data on adult spawning escapement for specific tributaries of the Snake River Basin DPS continues to make a quantitative assessment of viability difficult. Available data indicate that the overall long-term estimates of population trends have remained negative (Good et al. 2005). Annual return estimates are limited to counts of the aggregate return over Lower Granite Dam, and spawner estimates for the Tucannon, Asotin, Grande Ronde and Imnaha Rivers. The 2001 return over Lower Granite Dam was substantially higher relative to the low levels seen in the 1990s; the recent geometric 5-year mean abundance (Total escapement 106,175 with 14,768 natural returns) was approximately 28% of the interim recovery target level (52,000 natural spawners) (Good et al. 2005). The 10-year average for natural-origin steelhead passing Lower Granite Dam between 1996 and 2005 is 28,303 adults. Long-term trend estimates of the population growth rate (λ) across the available data set was 0.998 assuming that natural returns are produced only from natural-origin spawners, and 0.733 if both hatchery and wild spawners are contributing to production equally (Good et al. 2005). The Snake River supports approximately 63% of the total natural-origin production of steelhead in the Columbia River Basin (Good et al. 2005).

Critical Habitat

NMFS designated critical habitat for Snake River steelhead on September 2, 2005 (70 FR 52630). Designated critical habitat includes the following subbasins: Hells Canyon, Imnaha River, Lower Snake/Asotin, Upper Grand Ronde River, Wallowa River, Lower Grand Ronde, Lower Snake/Tucannon, Upper Salmon, Pahsimeroi, Middle Salmon-Panther, Lemhi, Upper Middle Fork Salmon, Lower Middle Fork Salmon, Middle Salmon, South Fork Salmon, Lower Salmon, Little Salmon, Upper and Lower Selway, Lochsa, Middle and South Fork Clearwater, and the Clearwater subbasins, and the Lower Snake/Columbia River corridor. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat 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. The final rule (70 FR 52630) lists the watersheds that comprise the designated subbasins and any areas that are specifically excluded from the designation.
There are 289 watersheds within the range of Snake River steelhead. The total area of habitat designated as critical includes about 8,000 miles of stream habitat. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. Of the 289 fifth order streams reviewed in this DPS, 231 received a high conservation value rating, 44 received a medium rating and 14 received a rating of low conservation value for the species. The lower Snake/Columbia

rearing/migration corridor downstream of the spawning range has a high conservation value. Limiting factors identified for Snake River Basin steelhead include: hydrosystem mortality, reduced stream flow, altered channel morphology and floodplain, excessive sediment, degraded water quality, harvest impacts, and hatchery impacts.

Final Protective Regulations
On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) (70 FR 37160) rule applies to the Snake River Basin steelhead DPS.

South-Central California Coast Steelhead

Distribution and Description of the Listed Species
The South-Central California Coast steelhead DPS includes all naturally spawned populations of steelhead (and their progeny) in streams from the Pajaro River (inclusive) to, but not including the Santa Maria River, California. No artificially propagated steelhead populations that reside within the historical geographic range of this DPS are included in this designation. The two largest basins within this DPS are the inland basins of the Pajaro River and the Salinas River. Both of these watersheds drain intercoastal mountain ranges and have long alluvial lower stretches. Principle sub-basins in the Pajaro River that support steelhead include: Corralitos Creek, Pescadero Creek, Uvas Creek and Pacheco Creek. Principle sub-basins in the Salinas River that support steelhead include the Arroyo Seco River, Gabilan Creek, Paso Robles Creek, Atascadero Creek and Santa Margarita Creek. Other important watersheds include the smaller coastal basins of the Carmel River, and St. Rosa and San Luis Obispos creeks.

Status and Trends
NMFS listed South-Central California Coast steelhead as threatened in 1997, and reaffirmed their status as threatened on January 5, 2006 (71 FR 834). Historical data on the South-Central California Coast steelhead DPS are sparse and no credible historic or recent estimates of total DPS size are available. Steelhead are present in a large portion of the historically occupied basins within this DPS (estimated 86-95 %) but observed and inferred abundance suggest many of this basins support a small fragment of their historic run size. Present population trends within individual watersheds continuing to support runs is generally unknown, but may vary widely between watersheds. No data are available to estimate the steelhead abundance or trends in the two largest watersheds in the DPS, the Pajaro and Salinas basins. These basins are highly degraded and expected to support runs much reduced in size from historical levels.

Steelhead in the Carmel Basin have been monitored at San Clemente Dam since 1964, representing one of the longest data sets available for steelhead in this DPS. However, this data is also limited because a nine year gap exists in the series, a large portion of the run spawns

below the dam and the older dam counts may be incomplete. Between NMFS' 1997 status review and 2005 status update, continuous data from San Clement dam suggests that the abundance of adult spawners in the Carmel River has increased. Carmel River time series data indicate that the population declined by about 22% per year between 1963 and 1993, and between 1991 and 1997 the population increased from one adult to 775 adults at San Clemente Dam. Good *et al.,* (2005) deemed this increase too great to attribute simply to improved reproduction and survival of the local steelhead population. Other possibilities were considered, including that the substantial immigration or transplantation occurred or that resident trout production increased as a result of improved environmental conditions within the basin. The five-year geometric mean calculated by Good *et al.,* (2005) for the Carmel River population (1998-2002) was 611 steelhead (range 1-881).

Critical Habitat

NMFS designated critical habitat for South-Central California Coast steelhead on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Pajaro River, Carmel River, Santa Lucia, Salinas River and Estero Bay. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat 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. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, South-Central California Coast steelhead occupy 30 watersheds (fresh water and estuarine). The total area of habitat designated as critical includes about 1,250 miles of stream habitat and about 3 square miles of estuarine habitat (e.g., Morro Bay). This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 30 watersheds reviewed in NMFS' assessment of critical habitat for South-Central California Coast steelhead, six watersheds received a low rating of conservation value, 11 received a medium rating, and 13 received a high rating of conservation value for the species.

Final Protective Regulations

On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to

natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the South-Central California steelhead DPS.

Southern California Steelhead

Distribution and Description of the Listed Species
The Southern California steelhead DPS includes all naturally spawned populations of steelhead in streams from the Santa Maria River, San Luis Obispo County, California (inclusive) to the U.S.-Mexico border. Artificially propagated steelhead that reside within the historical geographic range of this DPS are not included in the listing.

A comprehensive assessment of the distribution of steelhead within the Southern California DPS indicates that steelhead occur in most of the coastal basins (Boughton and Fish 2003 in Good *et al.,* 2005). Major watersheds occupied by steelhead in this DPS include the Santa Maria, Santa Ynez, Ventura, Santa Clara rivers. Smaller watersheds that support steelhead include the Los Angeles, San Gabriel, San Luis Rey and Sweetwater rivers, and San Juan and San Mateo creeks. Significant portions of several upper watersheds are contained with four national forests (Los Padres, Angeles, Cleveland, and San Bernardino National Forests), whereas coastal and inland valleys are dominated by urban development, with the Los Angeles basin being the most expansive and densest urban area in the DPS. Populations within the southernmost portion of the DPS (San Juan Creek, San Luis Rey River and San Mateo Creek) are separated from the northernmost populations by about 80 miles.

Status and Trends
NMFS listed Southern California steelhead as endangered in 1997 (62 FR 43937), and reaffirmed their status as endangered on January 5, 2006 (71 FR 834). Historical and recent data is generally lacking for Southern California steelhead, making a general assessment of their status difficult. The historical run size estimate for the entire DPS was between 32,000-46,000 steelhead, but this estimate omits the Santa Maria system and basins south of Malibu Creek (Busby et al. 1996). Estimates for the Santa Ynez River Basin, probably the largest run historically, range from 13,000 to 30,000 spawners, although this number may underestimate the steelhead abundance in the basin prior to the construction of Juncal and Gibraltar dams (Busby *et al.,* 1996; Good *et al.,* 2005). No recent data are available for steelhead in the Santa Ynez basin, and most of the historical spawning habitat was blocked by Bradbury and Gibraltar dams.

Steelhead and rainbow trout are known to occur in streams downstream of Bradbury Dam, but no estimates of abundance or trends are available. Similarly, Twitchell Dam in the Santa Maria River, and Casitas Dam on Coyote Creek and Matilija Dam on Matilija Creek block access to significant portions of historical spawning and rearing habitat, and alter the hydrology of the basins. A fish ladder and counting trap at the Vern Freeman Diversion Dam on the Santa Clara River is thought to be dysfunctional (Good et al. 2005). In general run sizes in river systems within the DPS are believed to range between less than five anadromous adults per year, to less than 100 anadromous adults per year. An estimated 26-52% of historically occupied basins are believed to still contain some steelhead, and about 30% are believed vacant, extirpated or nearly

extirpated due to dewatering or barriers that block spawning habitat.

Critical Habitat
NMFS designated critical habitat for Southern California steelhead on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Santa Maria River, Santa Ynez, South Coast, Ventura River, Santa Clara Calleguas, Santa Monica Bay, Callequas and San Juan hydrological units. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat 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. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, Southern California steelhead occupy 32 watersheds (fresh water and estuarine). The total area of habitat designated as critical includes about 700 miles of stream habitat and about 22 square miles of estuarine habitat, mostly within Humboldt Bay. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 32 watersheds reviewed in NMFS' assessment of critical habitat for Southern California steelhead, five watersheds received a low rating of conservation value, six received a medium rating, and 21 received a high rating of conservation value for the species.

Upper Columbia River Steelhead

Distribution and Description of the Listed Species
The Upper Columbia River steelhead DPS includes all naturally spawned populations of steelhead in streams in the Columbia River Basin upstream from the Yakima River, Washington, to the U.S.-Canada border. Six artificial propagation programs are part of this DPS.
Rivers in this DPS primarily drain the east slope of the northern Cascade Mountains and include the Wenatchee, Entiat, Methow and Okanogan River Basins. Some of these upper Columbia River subbasins, including the Okanogan River and the upper Columbia River proper, extend into British Columbia although steelhead do not occur in significant numbers in British Columbia and thus were not included in the DPS. Identified largely on the basis of spawning distributions, this DPS is composed of four putative populations defined by the Wenatchee, Entiat, Methow and Okanogan rivers. Historically (before the construction of Grand Coulee Dam blocked 50% of the river to Upper Columbia steelhead) major watershed that may have

supported steelhead within this DPS were the Sanpoil, Spokane, Colville, Kettle, Pend Oreille and Kootenai rivers (ICBTRT 2003).

All upper Columbia River steelhead are summer-run steelhead. Adults return in the late summer and early fall, with most migrating relatively quickly to their natal tributaries. A portion of the returning adult steelhead overwinters in mainstem reservoirs, passing over upper-mid-Columbia dams in April and May of the following year. Spawning occurs in the late spring of the year following river entry. Juvenile steelhead spend 1 to 7 years rearing in fresh water before migrating to sea. Smolt outmigrations are predominantly year class two and three (juveniles), although some of the oldest smolts are reported from this DPS (7 years). Most adult steelhead return to fresh water from sea after 1 or 2 years.

Status and Trends

NMFS originally listed Upper Columbia River steelhead as endangered in 1997 (62 FR 43937). On January 5, 2006, after reviewing the status of Upper Columbia River steelhead and noting an increase in abundance and more widespread spawning, NMFS reclassified the status of Upper Columbia River threatened (71 FR 834). In accordance with a U.S. District Court decision, NMFS reinstated the endangered status of Upper Columbia River steelhead in June 2007 (62 FR 43937). NMFS appealed the Court's decision, and on June 18, 2009, the District Court revised its ruling, effectively reinstating threatened status for Upper Columbia River steelhead (74 FR 42605). Thus, consistent with the court's rulings and the NMFS' listing determination of January 5, 2006, Upper Columbia River steelhead are listed as threatened under the ESA.

Since the 1940s, artificially propagated steelhead have seeded this DPS to supplement the numbers lost with the construction Grand Coulee Dam. Abundance estimates of returning naturally produced Upper Columbia River steelhead have been based on extrapolations from mainstem dam counts and associated sampling information (e.g., hatchery/wild fraction, age composition). Early estimates of steelhead in this DPS may be based on runs that were already depressed due to dams and steelhead fisheries. Nevertheless, these early dam counts are the best source of available data on the former size of the populations within this DPS. From 1933-1959 counts at Rock Island Dam averaged between 2,600 and 3,700 steelhead adults, which suggested the pre-fishery run size likely exceeded 5,000 adults destined for tributaries above Rock Island Dam (Chapman et al. 1994 as cited in Busby et al. 1996). Using counts at Priest Rapids Dam (located below the production areas for this DPS) as an indicator of DPS size and trends suggests that the total number of spawners has increased since NMFS' 1996 status review. The 1992-1996 average annual total returns (hatchery plus natural) of steelhead spawners was 7,800, and the 1997-2001 average is 12,900 steelhead (hatchery plus natural). The natural component increased in these same periods from 1,040 to 2,200, respectively (Good et al. 2005).

While the total number of naturally produced fish in this DPS increased between status reviews, the proportion of naturally produced steelhead to hatchery-origin fish has declined. Total escapement increased in the combined estimate for the Wenatchee and Entiat rivers to a geometric mean of 3,279 spawners (900 natural spawners) over NMFS' previous estimate of 2,500 hatchery and natural steelhead spawners (1989 to 1993, natural component 800 steelhead) (Good et al. 2005). Estimates of the hatchery contribution to this population increased from 65% to 71% of total escapement. (Good et al. 2005) A comparison of estimates for the Methow and Okanogan rivers during the same periods indicate that the total escapement increased from 2,400

to 3,714 while naturally produced steelhead declined from 450 to 358. Thus, the contribution of naturally produced steelhead declined from 19% to only 9% of total escapement between the 1993 and 2001 estimates (Good et al. 2005).

The assumptions of the role that hatchery fish play in the overall productivity and health of the DPS strongly influence estimates of population growth rates. Estimates based on the assumption that hatchery fish contribute to natural production at the same rate as natural-origin spawners consistently result in long-term population growth rates (expressed as λ) that are consistently below 1 (Good et al. 2005). Under the assumption that hatchery fish do not contribute to natural production, estimates of long term population growth rate suggest the population is growing. Determining the actual contribution of hatchery fish to natural production is important for understanding the true status of this DPS, particularly given that the proportion of naturally produced steelhead to hatchery-origin steelhead continues to decline. The extremely low replacement rate of naturally produced steelhead in this DPS is of concern.

The majority of the biological review team (54%) felt that this DPS warranted an "endangered" listing due to the growth rate and productivity, and uncertainty over the contribution of hatchery fish to natural production. NMFS, after convening a review of the artificial propagation programs of the six hatcheries in the DPS concluded that the programs collectively mitigate the immediacy of extinction risk in the DPS. Thus, NMFS listed the DPS as threatened rather than threatened (71 FR 834). NMFS concluded that the hatchery programs have increased total escapement and the distribution of spawning areas, and minimize the potential risks associated with artificial propagation. However, the abundance and productivity of naturally spawned steelhead remains a concern.

Critical Habitat

NMFS designated critical habitat for Upper Columbia River steelhead on September 2, 2005 (70 FR 52630). Designated critical habitat includes the following subbasins: Chief Joseph, Okanogan, Similkameen, Methow, Upper Columbia/Entiat, Wenatchee, Lower Crab, and the Upper Columbia/Priest Rapids subbasins, and the Columbia River corridor. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat 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. The final rule (70 FR 52630) lists the watersheds that comprise the designated subbasins and any areas that are specifically excluded from the designation.
There are 42 watersheds within the range of Upper Columbia River steelhead. The total area of habitat designated as critical includes about 1,250 miles of stream habitat. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. Of the 42 watersheds reviewed in NMFS' assessment of critical habitat for Upper Columbia River steelhead, three watersheds received a low rating of conservation value, eight received a medium rating, and 31 received a

high rating of conservation value for the species. In addition, the Columbia River rearing/migration corridor downstream of the spawning range was rated as a high conservation value. Limiting factors identified for the Upper Columbia River steelhead include: mainstem Columbia River hydropower system mortality, reduced tributary stream flow, tributary riparian degradation and loss of in-river wood, altered tributary floodplain and channel morphology, and excessive fine sediment and degraded tributary water quality.

Final Protective Regulations
NMFS proposed in June 2004 to list Upper Columbia River steelhead as threatened (69 FR 33102). As part of the proposed listing determination NMFS proposed applying the amended 4(d) protective regulations to Upper Columbia River steelhead. On January 5, 2006, NMFS issued a final determination listing the Upper Columbia River steelhead Distinct Population Segment (DPS) as threatened, and announced that it would finalize the protective regulations in a subsequent Federal Register notice (71 FR 834). In the final rule NMFS applied the 4(d) protective regulations, as amended in June 2005, to the Upper Columbia River steelhead DPS (71 FR 5178). These regulations are described in the *Protective Regulations for Threatened Salmonid Species* section of this document. These protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild.

Upper Willamette River Steelhead

Distribution and Description of the Listed Species
The Upper Willamette River steelhead DPS includes all naturally spawned populations of winter-run steelhead in the Willamette River, Oregon, and its tributaries upstream from Willamette Falls to the Calapooia River (inclusive). No artificially propagated populations that reside within the historical geographic range of this DPS are included in this listing. Hatchery summer-run steelhead occur in the Willamette Basin but are an out-of-basin population that is not included in this DPS.

The native (late) winter-run steelhead, with spring Chinook salmon, are the only two populations of salmon believed to historically occur above Willametter Falls. The construction of a fish ladder at the falls in the late 1880s, allowed for the passage of summer steelhead from Skamania Creek and winter-run steelhead from Big Creek (i.e., Gnat Creek). The two groups of winter-run steelhead exhibit different return times. The later run exhibits the historical phenotype adapted to passing the seasonal barrier that existed at Willamette Falls prior to construction of the fish ladder. The early run of winter-run steelhead are considered non-native, and were derived from Columbia River steelhead outside the Willamette River (Good et al. 2005). While the release of these hatchery winter-run fish was recently discontinued, some fish from earlier releases now reproduce naturally within the upper Willamette River Basin. Nonnative summer-run hatchery steelhead continue to be released within the upper basin (Good et al. 2005).

Native steelhead in the Upper Willamette are a late-migrating winter group that enters fresh water in January and February (Howell et al. 1985a). They do not ascend to their spawning areas until late March or April (Dimick and Merryfield 1945) and spawning occurs from April to June.

The smolt migration past Willamette Falls also begins in early April and proceeds into early June, peaking in early- to mid-May (Howell *et al.,* 1985). Smolts generally migrate through the Columbia via Multnomah Channel rather than the mouth of the Willamette River. Most spend 2 years in the ocean before re-entering natal rivers to spawn (Busby et al. 1996). Steelhead in the Upper Willamette River DPS generally spawn once or twice, although some may spawn three times. Repeat spawners are predominantly female and generally account for less than 10% of the total run size (Busby et al. 1996).

Status and Trends

NMFS originally listed Upper Willamette River steelhead as threatened in 1999 (64 FR 14517), and reaffirmed their status as threatened on January 5, 2006 (71 FR 834). The Upper Willamette steelhead DPS consists of four demographically independent populations, each of which remains extant although depressed from historical levels. Available data for this DPS comes from a combination of dam counts, redd count index surveys and hatchery trap counts. Estimates of abundance from NMFS 1996 status review on this DPS, demonstrate a mix of trends with the largest populations, Mollala and North Santiam Rivers, declining over the period of analysis. The 2005 review of the status of the Upper Willamette steelhead DPS indicated that each population showed a declining trend over the data series that extended to 2000 and 2001, while one population, the Calapooia River, increased over the short-term (Good *et al.,* 2005).

More recently, data reported in McElhany *et al.,* (2007) indicate that currently the two largest populations within the DPS are the Santiam River populations. Mean spawner abundance in both the North Santiam River and the South Santiam River is about 2,100 native winter-run steelhead. Long-term growth is negative for three of the populations within the DPS, with Calapooia River demonstrating a lambda of >1 indicating long-term growth in this population (McElhany *et al.,* 2007). Spatial structure for the North and South Santiam populations has been substantially reduced by the loss of access to the upper North Santiam basin and the Quartzville Creek watershed in the South Santiam subbasin due dam construction lacking passage facilities (McElhany et al. 2007). Additionally, habitat in the Molalla subbasin has been reduced significantly by habitat degradation and in the Calapooia by habitat degradation as well as passage barriers. Finally, the diversity of some populations has been eroded by small population size, the loss of access to historical habitat, legacy effects of past winter-run hatchery releases, and the ongoing release of summer steelhead (McElhany et al. 2007).

Critical Habitat

NMFS designated critical habitat for Upper Willamette River steelhead on September 2, 2005 (70 FR 52488). Designated critical habitat includes the following subbasins: Upper Willamette, North Santiam, South Santiam, Middle Willamette, Molalla/Pudding, Yamhill, Tualatin, and the Lower Willamette subbasins, and the lower Willamette/Columbia River corridor. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat 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. The final rule (70 FR 52630) lists the watersheds that comprise the designated subbasins and any areas that are specifically excluded from the designation.
There are 38 watersheds within the range of Upper Willamette River steelhead. The total area of habitat designated as critical includes about 1,250 miles of stream habitat. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull elevation. Of the 38 watersheds reviewed in NMFS' assessment of critical habitat for Upper Willamette River steelhead, 17 watersheds received a low rating of conservation value, six received a medium rating, and 15 received a high rating of conservation value for the species. In addition, the lower Willamette/Columbia River rearing/migration corridor downstream of the spawning range was rated as a high conservation value.

Final Protective Regulations
On June 28, 2005, as part of the final listing determinations for 16 ESUs of West Coast salmon, NMFS amended and streamlined the 4(d) protective regulations for threatened salmon and steelhead (70 FR 37160) as described in the *Protective Regulations for Threatened Salmonid Species* section of this document. Under this change, the section 4(d) protections apply to natural and hatchery fish with an intact adipose fin, but not to listed hatchery fish that have had their adipose fin removed prior to release into the wild. The amended June 2005 4(d) rule applies to the Upper Willamette River steelhead DPS.

Atlantic Salmon

Gulf of Maine Atlantic salmon

Description and distribution of the species
Gulf of Maine (GOM) DPS Atlantic salmon occur along the Atlantic coast from the Androscoggin River (Maine) in the south to the St. Croix River on the US-Canadian border. The lower Penobscot River has three primary tributaries that contain Atlantic salmon: Cove Brook, Kenduskeag Stream, Kennebec and Ducktrap rivers. The estimated population of Atlantic salmon in the lower Penobscot River and its tributaries is less than 20 adult Atlantic salmon. Atlantic salmon are also listed in the Dennys River, East Machias River, Machias River, Pleasant River, Narraguagus River, and Sheepscot River.

Habitat
The salmon's preferred spawning habitat is coarse gravel or rubble substrate (up to 3.5 inches in diameter) with adequate water circulation to keep the buried eggs well oxygenated (Peterson 1978). Water depth at spawning sites is typically between one and 2 feet deep, and water velocity averages 2 feet per second (Beland 1984). Spawning sites, or redds, average 8 feet long and 4.5 feet wide and are often located at the downstream end of riffles where water percolates through the gravel or where upwellings of groundwater occur (Moir et al. 1998). The annual egg production is approximately 240 eggs per 1,075 feet^2 of fluvial habitat (Chaput et al. 1998).

Movement, growth, and reproduction

Adult Atlantic salmon ascend the rivers of New England beginning in the spring and continuing into the fall, with peak numbers occurring in June. Although spawning does not occur until late fall, the majority of Atlantic salmon in Maine enter freshwater between May and mid-July (Meister 1958, Baum 1997). Salmon that return in early spring spend nearly 5 months in the river before spawning, often seeking cool water refuge (e.g., deep pools, springs, and mouths of smaller tributaries) during the summer months. Once an adult salmon enters a river, rising river temperatures and water flows stimulate upstream migration. Approximately 80% of salmon return to their home river after two years at sea, measuring approximately 2.5 feet long and weighing approximately 10 pounds (USFWS 2005b). A minority (10 to 20%) of Maine salmon return as smaller fish, or grilse, after only one winter at sea and still fewer return after three years at sea. A spawning run of salmon with representation of several age groups ensures some level of genetic exchange among generations. Once in freshwater, adult salmon cease feeding during their up-river migration. Spawning occurs in late October through November. Spawning sites are positioned within flowing water, particularly where upwelling of groundwater occurs, allowing for percolation of water through the gravel (Danie et al. 1984). These sites are most often positioned at the head of a riffle (Beland et al. 1982), the tail of a pool; or the upstream edge of a gravel bar where water depth is decreasing, water velocity is increasing (White 1942, McLaughlin and Knight 1987), and hydraulic head allows for permeation of water through the redd (a gravel depression where eggs are deposited).

A single female may create several redds before depositing all of her eggs. Female anadromous Atlantic salmon produce a total of 1,500-1,800 eggs per kilogram of body weight, yielding an average of 7,500 eggs per two sea-winter (SW) female (an adult female that has spent two winters at sea before returning to spawn) (Baum and Meister 1971).

After spawning, most Atlantic salmon move immediately downstream to backwater habitats near the head of tide (Cunjak et al. 1998, Fay et al. 2006). Upon returning to salt water, the spawned salmon or kelt resume feeding. If the salmon survives another one or two years at sea, it will return to its home river as a repeat spawner. From 1967 to 2003, approximately 3% of the wild and naturally reared adults that returned to rivers where adult returns are monitored--mainly the Penobscot River--were repeat spawners (USASAC 2004). Hatchery fish also return to the rivers into which they are stocked (Gorsky et al. 2009).

In late March or April, the eggs hatch into alevins. Alevins remain in the redd for about six weeks and are nourished by their yolk sac. Alevins emerge from the gravel about mid-May, generally at night, and begin actively feeding (Gustafson-Greenwood and Moring 1991). Survival from the egg to fry stage in Maine is estimated to range from 15-35% (Jordan and Beland 1981). Survival rates of eggs and larvae are a function of stream gradient, overwinter temperatures, interstitial flow, predation, disease, and competition (Bley and Moring 1988b). Once larval fry emerge from the gravel and begin active feeding they are referred to as fry. The majority of fry (>95%) emerge from redds at night (Gustafson-Marjenan and Dowse 1983). The survival rate of fry is affected by stream gradient, overwintering temperatures and water flows, and the level of predation and competition (Bley and Moring 1988a).

Within days, the free-swimming fry enter the parr stage, moving downstream to areas with adequate cover (rocks, vegetation, overhanging banks, and woody debris), water depths ranging from approximately four to 24 inches, velocities between 1foot and 3 feet per second, and temperatures near 61ºF (Beland 1984). When they finally reach their desired habitats, parr will actively defend territories that vary in size depending on the amount of food available and the density of other parr in the area (Symons 1971, McCormick et al. 1998, Armstrong et al. 1999). Some male parr become sexually mature and can successfully spawn with sea-run adult females. Water temperature, appetite, parr density, photoperiod, the level of competition and predation, and food supply may all influence the growth rate of parr (Lundqvist 1980, Randall 1982, Hearn 1987, Fausch 1988, Metcalfe et al. 1988, Elliot 1991, Nicieza and Metcalfe 1997). Maine Atlantic salmon parr densities are typically between three and nine parr per 1,075 feet^2, with years up to 16 parr per 1,075 feet^2 not uncommon (Beland 1996). There is no evidence of density-dependent limitations at densities of 13 parr per 1,075 feet^2 (Whalen et al. 2000). Parr feed on larvae of mayflies, stoneflies, caddisflies, chironomids, blackflies, annelids, and mollusks, as well as numerous terrestrial insects that fall into the river (Scott and Crossman 1973a).

In a parr's second or third spring, when it has grown 5 to 6 inches long, physiological, morphological, and behavioral changes occur (Schaffer and Elson 1975). This process, called smoltification, prepares parr for the dramatic change in osmoregulatory needs that comes with the transition from a freshwater to a saltwater habitat (Hoar 1976, McLeese et al. 1994, McCormick et al. 1998). In southern latitudes, including New England, most parr smolt after one year, but in cooler areas, they may take two to four years in freshwater before smolting (McCormick et al. 1998). Most smolts in New England rivers enter the sea during May and June to begin their ocean migration. Maine rivers produce approximately three smolts per 1,075 feet^2 of habitat.

Atlantic salmon of US origin are highly migratory, undertaking long marine migrations from the mouths of US rivers into the northwest Atlantic Ocean, where they are distributed seasonally over much of the region (Reddin 1985). The marine phase starts with smoltification and subsequent migration through the natal river and estuary. Upon completion of the physiological transition to saltwater, the post-smolt stage grows rapidly and has been documented moving in small, loosely aggregated schools near the surface (Dutil and Coutu 1988). After entering the nearshore waters of Canada, the post-smolts become part of a mixture of stocks of Atlantic salmon from various North American streams. Post-smolts appear to feed opportunistically on macroinvertebrates, amphipods, euphausiids, and fish (Hansen and Pethon 1985, Hansen and Quinn 1998, Andreassen et al. 2001). Once they mature to adult salmon, they travel individually and primarily eat capelin, herring, and sand lance (Hansen and Pethon 1985, Reddin 1985, Hansen and Quinn 1998).

Status and trends

The GOM DPS of anadromous Atlantic salmon was listed by the USFWS and NMFS as an endangered species on November 17, 2000 (65 FR 69495). The GOM DPS encompasses all naturally reproducing remnant populations of Atlantic salmon downstream of the former Edwards Dam site on the Kennebec River northward to the mouth of the St. Croix River. To

date, Atlantic salmon are listed in the Dennys, East Maccias, Machias, Pleasant, Narraguagus, Ducktrap, and Sheepscot Rivers, Kenduskeag Stream, and Cove Brook. Naturally reproducing Atlantic salmon in the Penobscot River and its tributaries downstream of the former Bangor Dam are listed as endangered. The USFWS's GOM DPS river-specific hatchery-reared fish are also included as part of the listed entity (73 FR 51415).

Anadromous Atlantic salmon were native to nearly every major coastal river north of the Hudson River, New York (USFWS 2005b). The annual historic Atlantic salmon adult population returning to US rivers has been estimated to be between 300,000 and 500,000 (Stolte 1981, Beland 1984). The largest historical salmon runs in New England were likely in the Connecticut, Merrimack, Androscoggin, Kennebec, and Penobscot Rivers.

By the early 1800s, Atlantic salmon runs in New England had been severely depleted, reducing the distribution in the southern half of its range. Restoration efforts were initiated in the mid-1800s, but there was little success (Stolte 1981). There was a brief period of success in the late 19th century when limited runs were reestablished in the Merrimack and Connecticut Rivers by artificial propagation, but these runs were extirpated by the end of the century. By the end of the 19th century, three of the five largest salmon populations in New England (Connecticut, Merrimack, and Androscoggin Rivers) had been eliminated.

Recently, Fay *et al.* (2006) used Population Viability Analysis (PVA) techniques to determine the conservation status of Atlantic salmon in the GOM DPS. Composite spawner data used to populate the model included adult return and rod kill estimates from the Penobscot River, adult spawner and rod kill estimates for the Narraguagus River, and adult spawner estimates for the GOM DPS. Using two time series, 1984 to 2004 and 1991 to 2004, Fay *et al.* (2006) calculated the negative population growth rates (for 1980-2004, lambda = 0.9690, variance = 0.0261; for 1991-2004, lambda = 0.9471, variance = 0.0142). From this, the estimated risk of extinction (defined herein as the number of spawners that falls below 100 individuals) within 100 years is 61% and 75% (or 28% and 45% in 40 years), for each respective data set.

Natural threats

Geographic features, such as waterfalls, pose natural barriers to salmon migration to spawning habitat. A variety of diseases affect Atlantic salmon, but are exacerbated by the presence of farming pens near river mouths. Atlantic salmon are prey for a variety of predators, including seals, porpoises, dolphins, otters, minks, birds, sharks, and a variety of other fishes at various salmon life stages.

Anthropogenic threats

Humans pose numerous threats to Atlantic salmon survival and recovery (see USFWS 2005b for a review). Water quality in both marine, estuarine, and aquatic habitats suffers from both point and non-point source pollution, both biological (bacteria) and chemical. Riverine environments are becoming acidified, which can cause physiological stress in adults and altered developmental biology in eggs or hatchlings. In association with acidification, aluminum toxicity can lead to osmoregulation failure. This is because Atlantic salmon are highly sensitive to pH changes and many runs of Atlantic salmon in Sweden, Norway, and Canada have been severely depleted or

extirpated due to acidity changes in river systems resulting from industrial activity (Watt 1981, Watt et al. 1983, Watt et al. 2000, Sandøy and Langåker 2001). Pesticide use and its immigration into Maine waterways is also of concern. For example, atrazine can significantly impair water balance in salmon even at low concentrations, resulting in a reduced ability for salmon to move between fresh and salt water (Jagoe and Haines 1990, Staurnes et al. 1993, WWF 2001). At levels that presently occur in stream environments, male salmon also experience impaired olfactory reception in being able to detect female pheromones (Waring and Moore 1998). Thus, male reproduction activity is not cued to that of females and has the potential to severely reduce recruitment. Nonylphenols are also severely detrimental to juvenile salmon. These chemicals also reduce the ability of smolts to transition between fresh and salt water, leading to mortality, as well as imitate female hormones leading to eggs that do not hatch (Fairchild et al. 1999, WWF 2001). Sedimentation due to erosion and development in and around aquatic waterways can degrade salmon habitat and the habitat of their invertebrate prey. Excessive nutrient load, as in marine systems, can lead to a bloom of plant growth and subsequent death, which reduces oxygen levels to anoxic conditions. This can lead to extensive habitat loss and salmon mortality.

Although changes overtly seem minor, increases in Maine's river temperatures can have broad impacts on salmon recovery, including changes in fish physiology, prey abundance and distribution, loss of spawning activity, and other effects (USFWS 2005b, Holbrook et al. 2009). As in Pacific salmon species, Atlantic salmon decline originated largely from manmade barriers across rivers preventing movement to and from spawning and marine habitats. Although many of these barriers have since been modified or removed, modern construction (bridges, culverts, etc.) that do not consider Atlantic salmon needs can hinder recovery efforts (Holbrook et al. 2009). When water temperatures exceed 22º C during spawning runs, Atlantic salmon tend to have poorer success in passing obstacles than (Holbrook et al. 2009).

Atlantic salmon fisheries have been discontinued in the US, Canada, and Greenland. A high threat is posed by farm-raised salmon due to the potential for these fish to escape (instances of thousands of fish escaping are known) and interbreed with wild salmon, thereby affecting the genetics of Atlantic salmon as a species. Recent evidence shows that supportive breeding programs, where wild Atlantic salmon are captured and bred in captivity and young are released early in life, produce fish that are genetically, morphologically, and behaviorally different from truly wild progeny (Blanchet et al. 2008). The presence of disease and parasites in farm-raised salmon pens can also have a deleterious effect on wild Atlantic salmon.

Climate change has the potential to be a strong negative influence on Altantic salmon. Remaining occupied habitat is at the southern edge of the ESU's range. To survive, populations have adapted to distinct physical and environmental conditions here (Saunders 1981). Climate models predict significant, extended warming (IPCC 2001a). Although periods of North Atlantic warming and cooling have occurred, changes have not been uniform as global warming is, changing sea temperatures, wind currents, fresh water input, and mixing of the ocean's surface layer. Small thermal changes can critically affect biological functions, such as protein metabolism, response to aquatic contaminants, reproductive performance, smolt development, and species distribution limits (Keleher and Rahel 1996, McCormick et al. 1997, Reid et al.

1997, Somero and Hofmann 1997, Van der Kraak and Pankhurst 1997, Welch et al. 1998). Atlantic salmon smolt growth is known to change with temperature, with a temperature increase from 57º to 64ºF resulting in a greater than 10% decrease in growth rate (Handeland et al. 2008).

It should be noted that positive effects may also be realized by climate change and specifically warmer water temperature. Increased opportunities for growth in spring and summer could increase the percentage of fish that enter the upper size distribution of a population and smolt the following spring (Thorpe 1977, Thorpe et al. 1980, Thorpe 1994). In addition, warmer rearing temperatures during the late winter and spring have been shown to advance the timing of the parr-smolt transformation in Atlantic salmon (Solbakken et al. 1994). There is, however, an optimal temperature range and a limit for growth after which salmon parr will stop feeding due to thermal stress. During this time, protein degradation and weight loss will increase with rising water temperature (McCarthy and Houlihan 1997). The NRC (2004) concluded that some degree of climate warming or change in hydrologic regime could be tolerated if other problems affecting Atlantic salmon are reduced.

Critical habitat

On June 19, 2009, 45 specific areas occupied by Atlantic salmon at the time of listing (approximately 19,571 km of perennial river, stream, and estuary habitat and 799 square kilometers of lake habitat within the range of the GOM DPS) were established for Atlantic salmon critical habitat (74 FR 29300). The PCEs for this critical habitat include:

- Deep, oxygenated pools and cover (e.g., boulders, woody debris, vegetation, etc.), near freshwater spawning sites, necessary to support adult migrants during the summer while they await spawning in the fall.
- Freshwater spawning sites that contain clean, permeable gravel and cobble substrate with oxygenated water and cool water temperatures to support spawning activity, egg incubation, and larval development as well as support emergence, territorial development and feeding activities of Atlantic salmon fry.
- Freshwater rearing sites with space to accommodate growth and survival of Atlantic salmon parr.
- Freshwater rearing sites with a combination of river, stream, and lake habitats that accommodate parr's ability to occupy many niches and maximize parr production.
- Freshwater rearing sites with cool, oxygenated (6 mg/L) water and diverse food resources (mayflies, stoneflies, chironomids, caddisflies, blackflies, aquatic annelids, and mollusks, as well as numerous terrestrial invertebrates, alewives, dace, or minnows) to support growth and survival of Atlantic salmon parr.
- Freshwater and estuary migratory sites free from physical and biological barriers that delay or prevent access of adult salmon seeking spawning grounds needed to support recovered populations or prevent emigration of smolts to the marine environment.
- Freshwater and estuary migration sites with pool, lake, and instream habitat that provide cool, oxygenated water and cover items (e.g., boulders, woody debris, and vegetation) to serve as temporary holding and resting areas during upstream migration of adult salmon.
- Freshwater and estuary migration sites with abundant, diverse native fish communities to

serve as a protective buffer against predation.

- Freshwater and estuary migration sites with sufficiently cool water temperatures and water flows that coincide with diurnal cues to stimulate smolt migration.
- Freshwater migration sites with water chemistry (particularly pH) needed to support sea water adaptation of smolts.

These PCEs have undergone significant degradation over in the recent past. Dams, along with degraded substrate and cover, water quality, water temperature, and biological communities, have reduced the quality and quantity of habitat available to Atlantic salmon populations within the DPS. A combined total of twenty FERC-licensed hydropower dams on the Penobscot River significantly impede the migration of Atlantic salmon and other diadromous fish to historically accessible spawning and rearing habitat. Agriculture and urban development largely affect the lower third of the Penobscot River below the Piscataquis River sub-basin by reducing substrate and cover, reducing water quality, and elevating water temperatures. Introductions of smallmouth bass and other non-indigenous species significantly degrade habitat quality throughout the mainstem Penobscot and portions of the Mattawamkeag, Piscataquis, and lower Penobscot sub-basins by altering predator/prey relationships. Similar to smallmouth bass, recent Northern pike introductions threaten habitat in the lower Penobscot River below the Great Works Dam.

Today, dams are the greatest impediment, outside of marine survival, to the recovery of salmon in the Penobscot, Kennebec and Androscoggin river basins (Fay et al. 2006). Hydropower dams significantly impede the migration of Atlantic salmon and other diadromous fish and either reduce or eliminate access to historically accessible spawning and rearing habitat. In addition to hydropower dams, agriculture and urban development largely affect the lower third of the Merrymeeting Bay recovery unit by reducing substrate and cover, reducing water quality, and elevating water temperatures. Additionally, smallmouth bass and brown trout introductions, along with other non-indigenous species, significantly degrade habitat quality throughout the Merrymeeting Bay recovery unit by altering natural predator/prey relationships.

Impacts to substrate and cover, water quality, water temperature, biological communities, and migratory corridors, among a host of other factors, have impacted the quality and quantity of habitat available to Atlantic salmon populations within the Downeast Coastal recovery unit. Two hydropower dams on the Union river, and to a lesser extent the small ice dam on the lower Narraguagus River, limit access to spawning and rearing habitat within these two watersheds. In the Union River, physical and biological features have been most notably limited by high water temperatures and abundant smallmouth bass populations associated with impoundments. In the Pleasant River and Tunk Stream, which collectively contain over 4,300 units of spawning and rearing habitat, pH has been identified as possibly being the predominate limiting factor. The Machias, Narraguagus, and East Machias rivers contain the highest quality habitat and collectively account for approximately 40 percent of the spawning and rearing habitat in the Downeast Coastal recovery unit.

Southern Pacific Eulachon

Eulachon are small smelt native to eastern North Pacific waters from the Bering Sea to Monterey Bay, California, or from 61º N to 31º N (1944, Eschmeyer et al. 1983b, Minckley et al. 1986, Hay and McCarter 2000). Eulachon that spawn in rivers south of the Nass River of British Columbia to the Mad River of California comprise the southern population of Pacific eulachon. This species is designated based upon timing of runs and genetic distinctions (Hart and McHugh 1944, McLean et al. 1999, Hay and McCarter 2000, McLean and Taylor 2001, Beacham et al. 2005).

Adult eulachon are found in coastal and offshore marine habitats (Allen and Smith 1988, Hay and McCarter 2000, Willson et al. 2006). Larval and post larval eulachon prey upon phytoplankton, copepods, copepod eggs, mysids, barnacle larvae, worm larvae, and other eulachon larvae until they reach adult size (WDFW and ODFW 2001). The primary prey of adult eulachon are copepods and euphausiids, malacostracans and cumaceans (Smith and Saalfeld 1955, Barraclough 1964, Drake and Wilson 1991, Sturdevant et al. 1999, Hay and McCarter 2000)

Although primarily marine, eulachon return to freshwater to spawn. Adult eulachon have been observed in several rivers along the west coast (Odemar 1964, Moyle 1976b, Minckley et al. 1986, Emmett et al. 1991, Jennings 1996, Wright 1999, Larson and Belchik 2000, Musick et al. 2000, WDFW and ODFW 2001). For the southern population of Pacific eulachon, most spawning is believed to occur in the Columbia River and its tributaries as well as in other Oregonian and Washingtonian rivers (Emmett et al. 1991, Musick et al. 2000, WDFW and ODFW 2001). Eulachon take less time to mature and generally spawn earlier in southern portions of their range than do eulachon from more northerly rivers (Clarke et al. 2007). Spawning is strongly influenced by water temperatures, so the timing of spawning depends upon the river system involved (Willson et al. 2006). In the Columbia River and further south, spawning occurs from late January to March, although river entry occurs as early as December (Hay and McCarter 2000). Further north, the peak of eulachon runs in Washington State is from February through March while Alaskan runs occur in May and river entry may extend into June (Hay and McCarter 2000). Females lay eggs over sand, course gravel or detritial substrate. Eggs attach to gravel or sand and incubate for 30 to 40 days after which larvae drift to estuaries and coastal marine waters (Wydoski and Whitney 1979a).

Eulachon generally die following spawning (Scott and Crossman 1973b). The maximum known lifespan is 9 years of age, but 20 to 30% of individuals live to 4 years and most individuals survive to 3 years of age, although spawning has been noted as early as 2 years of age (Wydoski and Whitney 1979b, Barrett et al. 1984, Hugg 1996, Hay and McCarter 2000, WDFW and ODFW 2001). The age distribution of spawners varies between river and from year-to-year (Willson et al. 2006).

Status and Trends

The southern population of Pacific eulachon was listed as threatened on March 18, 2010 (75 FR 13012). It is considered to be at moderate risk of extinction throughout its range because of a variety of factors, including predation, commercial and recreational fishing pressure (directed and bycatch), and loss of habitat. Further population decline is anticipated to continue as a result of climate change and bycatch in commercial fisheries. However, because of their fecundity, eulachon are assumed to have the ability to recover quickly if given the opportunity (Bailey and Houde 1989).

Eulachon formerly experienced widespread, abundant runs and have been a staple of Native American diets for centuries along the northwest coast. However, such runs that were formerly present in several California rivers as late as the 1960s and 1970s (i.e., Klamath River, Mad River and Redwood Creek) no longer occur (Larson and Belchik 2000). This decline likely began in the 1970s and continued until, in 1988 and 1989, the last reported sizeable run occurred in the Klamath River and no fish were found in 1996, although a moderate run was noted in 1999 (Larson and Belchik 2000, Moyle 2002). Eulachon have not been identified in the Mad River and Redwood Creek since the mid-1990s (Moyle 2002).

Critical Habitat

Critical habitat has been proposed for the southern population of Pacific eulachon (76 FR 515).

Threats

Natural threats. Numerous predatory fishes, marine mammals, birds and terrestrial mammals prey on eulachon (Clemens et al. 1936, Hart 1973a, Scott and Crossman 1973b, Jeffries 1984, Drake and Wilson 1991, Yang and Nelson 1999, Willson et al. 2006). The high fat content of eulachon make them a valuable prey for white sturgeon in the Columbia and Fraser rivers during winter (Willson et al. 2006).

Anthropogenic threats. Fisheries harvests are likely a major contributor to eulachon decline. The best available information for catches comes from the Columbia River, where catches have been as high as 5.7 million pounds per year, but averaged near 2 million pounds from 1938 to 1993 (Wydoski and Whitney 1979a). Since 1993, catches have not exceeded 1 million pounds annually and the median catch has been 43,000 pounds (97.7% reduction in catch), even when effort is accounted for (WDFW and ODFW 2001). Bycatch from fishing along U.S. and Canadian coasts has also been high, composing up to 28% of the total catch by weight (Hay and McCarter 2000, DFO 2008).

Construction projects have also had a negative impact on eulachon stocks. Dams, such as the Bonneville Dam on the Hood River, have blocked eulachon from moving into former spawning habitat (Smith and Saalfeld 1955). Such damming projects also alter sedimentation and flow dynamics that eulachon have developed around in their evolution. River substrate composition, likely critical to successful spawning, is also altered by dams. The impoundment of water tends to raise water temperatures; a factor that spawning eulachon are particularly sensitive to (NMFS 2008c). Eulachon ecotoxicological studies show high contaminant burdens, particularly of arsenic and lead (Futer and Nassichuk 1983, Rogers et al. 1990, EPA 2002).

Sturgeon

Sturgeon (*Acipenseridae*) are one of the oldest *Osteichthyes* (bony fish) families in existence. They are native to rivers and coastal areas of North America. The two listed sturgeon, discussed below, are part of the genus *Acipenser* and share some common characteristics. Members of the genus have a characteristic external morphology distinguished by the inferior mouth typical of bottom-feeders. Most species are anadromous, although a few species are entirely fresh water and many species can survive if they become land-locked. Both listed species (discussed below) are anadromous and tend to remain in coastal waters. As an anadromous fish, sturgeon spawn in fresh water and feed and rear in marine or estuarine waters. Sturgeon are capable of many reproductive cycles and tend to be very long-lived.

Threats
Natural Threats. Birds and larger freshwater fish feed on eggs and larvae, while sharks, pinnipeds and other large predators prey on marine adult and subadult fish.

Anthropogenic Threats. In general sturgeon have declined from the combined effects from the construction of dam and water diversion projects, dredging and blasting, water pollution and fisheries. The longevity, slow rate of growth, delayed maturation, and bottom-feeding habits of sturgeon makes them susceptible to over-harvest and exposure to (and the accumulation of) contaminants. Many sturgeon also do not spawn on an annual basis, but may spawn every other year or even more infrequently. Thus even small increases in mortality can affect population productivity (Heppell 2007). While sturgeon will be exposed to dissolved contaminants in the water column their body form, feeding habits and affinity with bottom sediments may expose them to a different suite of contaminants than pelagic fish. The sediment exposure pathway may be more significant. Benthic dwelling fish such as sturgeon may be exposed through the direct contact with sediment and its boundary layer and commonly have a higher rate of exposure through incidental ingestion.

Southern Green Sturgeon

Green sturgeon occur along the west coast of North America from Mexico to the Bering Sea (Adams et al. 2002, Colway and Stevenson 2007). Distinguished primarily according to genetic differences and spawning locations, NMFS recognizes two species of green sturgeon: a northern species whose populations are relatively healthy, and a Southern species that has undergone significant decline (Adams et al. 2007). NMFS listed the Southern species of green sturgeon as threatened in 2006 (71 FR 17757).

Green sturgeon are considered one of the most marine-oriented sturgeon species, spending much of their lives in coastal marine waters, estuaries and bays. Early life stages rear in fresh water, and adults return to fresh water when they are 15 years old or older to spawn. Across the species' range only three rivers contain documented spawning (Moyle et al. 1992, CDFG 2002). Outside of natal rivers, the distribution of southern green sturgeon and northern green sturgeon overlap. Both the northern species and southern species of green sturgeon occupy coastal estuaries and coastal marine waters from southern California to Alaska, including Humbolt Bay, the lower Columbia River estuary, Willapa Bay, Grays Harbor and southeast Alaska. In general, green sturgeon are more common north of Point Conception, California.

Green sturgeon are spring spawners and initiate spawning migrations as early as March. Fish in the Klamath River have been observed initiating migrations between April and June, Rogue River fish between May and July, whereas Heubein *et al.,* (2009) observed Sacramento River fish making their upstream migrations between March and April. Spawning generally occurs in deep pools of large rivers or off-channel coves (Moyle et al. 1992, Moyle et al. 1995, Rien et al. 2001, Heublein et al. 2009). Fish then tend to aggregate in deep pools, where they will over-summer before outmigrating in the fall, although some fish have been observed outmigrating relatively soon after presumed spawning events (Heublein et al. 2009). In the Sacramento River

adult green sturgeon spawn in late spring and early summer (Heublein et al. 2009). It appears that specific habitat for spawning includes large cobblestones (where eggs can settle between), although spawning is known to occur over clean sand or bedrock.

Green sturgeon are a long-lived fish and likely live for 60 to 70 years (Moyle 2002). Age at first maturation for green sturgeon is at least 15 years old, after which adults likely return every 2 to 5 years to spawn (Adams et al. 2002, Van Eenennaam et al. 2006). Most male spawners are young (17 to 18 years) while females on the spawning grounds are often older (27 to 28 years). Green sturgeon spend their first 1 to 4 years in their natal streams and rivers (Nakamoto et al. 1995, Beamesderfer and Webb 2002), although they are believed to be physiologically adapted to sea water survival at 6 months of age (Allen and Cech 2007, Allen and J.J. Cech 2007, Allen et al. 2009). Larvae are active at night, a behavior that likely reduces predation and avoids being moved downstream more than necessary (Cech Jr. et al. 2000). Green sturgeon larvae grow very rapidly, reaching about 300 mm by age one (Deng 2000). While in fresh water, juveniles feed on a variety of fishes and invertebrates (Moyle et al. 1992). One juvenile from the Sacramento-San Joaquin estuary was found to have preyed most commonly upon opisthobranch mollusks (*Philline* sp.), with bay shrimp (*Crangon* sp.) and overbite clams (*Potamocorbula amurensis*) as secondary prey. Other juveniles in the Sacramento River delta feed on opossum shrimp (*Neomysis mercedis*) and *Corophium* amphipods (Radtke 1966).

Upon outmigration from fresh water, subadult green sturgeon disperse widely along through continental shelf waters of the west coast within the 110 meter contour (Moyle et al. 1992, Erickson and Hightower 2007). It appears that green sturgeon generally distribute north of the river mouth from whence they emerge as juveniles during fall and move into bays and estuaries during summer and fall (Moser and Lindley 2007, Israel et al. 2009). The limited feeding data available for subadult and adult green sturgeon show that they consume benthic invertebrates including shrimp, clams, chironomids, copepods, mollusks, amphipods and small fish ((Houston 1988, Moyle et al. 1992). Sturgeon use electroreception to locate prey. Olfaction and taste may also be important to foraging, while vision is thought to play a minor role in prey capture (Miller 2004).

Status and Trends

NMFS listed the southern population of the North American green sturgeon as threatened on April 7, 2006 (71 FR 17757). Trend data for green sturgeon is severely limited. Available information comes from two predominant sources, fisheries and tagging. Only three data sets were considered useful for the population time series analyses by NMFS' biological review team: the Klamath Yurok Tribal fishery catch, San Pablo sport fishery tag returns and Columbia River commercial landings (NMFS 2005c). Using San Pablo sport fishery tag recovery data, the California Department of Fish and Game produced a population time series estimate for the southern species. San Pablo data suggest that green sturgeon abundance may be increasing, but the data showed no significant trend. The data set is not particularly convincing, however, as it suffers from inconsistent effort and since it is unclear whether summer concentrations of green sturgeon provide a strong indicator of population performance (NMFS 2005c). Although there is not sufficient information available to estimate the current population size of southern green sturgeon, catch of juveniles during State and Federal salvage operations in the Sacramento delta

are low in comparison to catch levels before the mid-1980s.

Critical Habitat

On October 9, 2009, NMFS designated critical habitat for southern green sturgeon (74 FR 52300). The geographical area identified as critical habitat is based upon the overlapping distribution of the southern and northern species, and encompasses all areas where the presence of southern green sturgeon have been confirmed or where their presence is likely. Therefore the geographical area defined as critical habitat is the entire range of the biological species, green sturgeon, from the Bering Sea, AK, to Ensenada, Mexico. Specific fresh water areas include the Sacramento River, Feather River, Yuba River and the Sacramento-San Joaquin Delta. Specific coastal bays and estuaries include estuaries from Elkhorn Slough, California, to Puget Sound, Washington. Coastal marine areas include waters along the entire biological species' range within a depth of 60 fathoms. The principal biological or physical constituent elements essential for the conservation of southern green sturgeon in fresh water include: food resources; substrate of sufficient type and size to support viable egg and larval development; water flow, water quality such that the chemical characteristics support normal behavior, growth and viability; migratory corridors; water depth; and sediment quality. Primary constituent elements of estuarine habitat include food resources, water flow, water quality, migratory corridors, water depth and sediment quality. The specific primary constituent elements of marine habitat include food resources, water quality and migratory corridors.

Critical habitat of the Southern species of green sturgeon is threatened by several anthropogenic factors. Four dams and several other structures currently are impassible for green sturgeon to pass on the Sacramento, Feather and San Joaquin rivers, preventing movement into spawning habitat. Threats to these riverine habitats also include increasing temperature, insufficient flow that may impair recruitment, the introduction of striped bass that may eat young sturgeon and compete for prey, and the presence of heavy metals and contaminants in the river.

Final Protective Regulations

The final 4(d) rule for southern green sturgeon was issued June 2, 2010, and became effective July 2, 2010 (75 FR 30714). Under this rule, the prohibitions listed under ESA sections 9(a)(1)(A) through 9(a)(1)(G) are applied for the Southern species, including all the ESA section 9(a)(1)(B) and 9(a)(1)(C) prohibitions except for: 1) Certain Federal, State or private-sponsored research or monitoring activities; 2) Emergency fish rescue and salvage activities; 3) Habitat restoration activities; 4) Commercial and recreational fisheries activities, if conducted under approved Fisheries Management and Evaluation Plans; and 5) Certain Tribal fishery management activities.

Threats

Natural Threats. Green sturgeon eggs and larvae are likely preyed upon by a variety of larger fish and animals, while sub-adult and adult sturgeon may occasionally be preyed upon by shark sea lions, or other large body predators (NMFS 2005c).

Anthropogenic Threats. The principle threat to southern green sturgeon comes from a drastic reduction in available spawning area from impassible barriers (e.g., Oroville, Shasta and

Keswick dams). Other threats include potentially lethal temperature limits, harvest, entrainment by water projects and toxins and invasive species (Adams et al. 2007, Erickson and Webb 2007, Lackey 2009). Since this species is composed of a single spawning population within the Sacramento River, stochastic variation in environmental conditions and significant fluctuations in demographic rates increases the risk of extinction for this species.

Studies from other sturgeon species indicate that sturgeon readily bioaccumulate contaminants. White sturgeon from the Kootenai River have been found to contain aluminum, arsenic, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, nickel, selenium, zinc, DDE, DDT, PCBs and other organochlorines (Kruse and Scarnecchia 2001). Mercury has also been identified from white sturgeon of the lower Columbia River (Webb et al. 2006). Numerous organochlorines, including DDT, DDD, DDE, chlordane and dieldrin have also been identified in these fish (Foster et al. 2001). Observed concentrations are likely sufficient to influence reproductive physiology.

Shortnose Sturgeon

Shortnose sturgeon occur along the Atlantic Coast of North America, from the St. John River in Canada, south to the St. Johns River in Florida. NMFS' recovery plan (1998b) recognized 19 wild populations based on their strong fidelity to their natal streams and several captive populations (from a Savannah River broodstock) that are maintained for educational and research purposes (NMFS 1998b).

Shortnose sturgeon are generally anadromous, but may migrate between fresh and salt water for reasons other than spawning. They can also maintain freshwater resident populations. In general, shortnose sturgeon are benthic fish that occupy the deep channel sections of large rivers or estuarine waters of their natal rivers and will migrate considerable distances. Dadswell (1979 in Dadswell et al. 1984)(1979 in Dadswell et al. 1984)(1979 in Dadswell et al. 1984) (1984a) observed shortnose sturgeon traveling up 160 km between tagging and recapture in the St. John estuary and it is not uncommon for adults to migrate 200 km or more to reach spawning areas (Kynard 1997). After spawning in the spring, adults tend to migrate rapidly downstream to feeding areas in the estuary or to tidally influenced fresh water (see Dadswell et al. 1984a).

Young-of-the year shortnose sturgeon move downstream after hatching, remaining in fresh water for about 1 year (Kynard 1997). Initially, young shortnose sturgeon will reside short distances from spawning areas and as they grow will tend to move further downstream (see Dadswell et al. 1984a). By age 3 or older juvenile sturgeon will spend a large portion of their year at the salt- and fresh water interface of coastal rivers (NMFS 1998c).

Habitat use in fresh water during summer and winter months overlaps between adult and age-1 shortnose sturgeon (O'Herron II et al. 1993, Kynard et al. 2000, Moser et al. 2000c). Kynard *et al.,* (2000) found that both age classes preferred deep-water curves with sand and cobble to higher velocity runs, particularly during winter months and shifted to channel habitat as water temperatures rose in summer months. In the Connecticut River and the Merrimack, Kynard *et al.,* (2000) found shortnose generally used water about 3 meters deep, ranging from less than a meter to about 15 meters deep.

Female shortnose sturgeon spawn every three to five years. Males spawn every other year, although some may spawn in consecutive years (Dovel et al. 1992, Collins and Smith 1993, Kieffer and Kynard 1993, NMFS 1998a). Spawning typically occurs during the spring, between mid-March and late May. Spawning areas are often located just below the fall line at the farthest accessible upstream reach of the river (NMFS 1998a).

Male shortnose sturgeon in southern rivers will first spawn between ages 2 and 5, while fish as far north as the St. John River, Canada first spawn at about 10 to 11 years of age (Dadswell et al. 1984a, NMFS 1998a). Age at first spawning for female shortnose sturgeon varies from about age 6 to 18 years, like males, varying on a latitudinal cline (Dadswell et al. 1984a, NMFS 1998a). In general, fish in the northern portion of the species' range live longer than individuals in the southern portion of the species' range (Gilbert 1989). The maximum age reported for a shortnose sturgeon in the St. John River in New Brunswick is 67 years (for a female), 40 years for the Kennebec River, 37 years for the Hudson River, 34 years in the Connecticut River, 20 years in the Pee Dee River and 10 years in the Altamaha River (Gilbert 1989). Male shortnose sturgeon appear to have shorter life spans than females (Gilbert 1989).

Like all sturgeon, shortnose have ventrally located, sucker-like mouths, structured for feeding on benthos. Foraging generally occurs in areas with abundant macrophytes, where juvenile and adult shortnose sturgeon feed on amphipods, polychaetes and gasteropods (Dadswell et al. 1984b, Moser and Ross 1995, NMFS 1998a). Sturgeon use electroreception to identify prey. Olfaction and taste are also likely important to foraging, while vision is thought to play a minor role (Miller 2004). As adults, a significant portion of a shortnose sturgeon's diet may consist of freshwater mollusks (Dadswell et al. 1984b). Based on observations by Kynard *et al.,* (2000), shortnose sturgeon will consume the entire mollusk, excreting the shell after ingestion.

Status and Trends

Shortnose sturgeon were listed as endangered on March 11, 1967, under the Endangered Species Preservation Act (32 FR 4001) and remained on the endangered species list with enactment of the ESA of 1973, as amended. Pollution and overfishing, including bycatch in the shad fishery, were listed as principal reasons for the species' decline. Shortnose sturgeon are listed as an endangered species throughout all of its range

Northern shortnose sturgeon population abundances are generally larger than southern populations (Kynard 1997). Updated population estimates also suggest that three of the largest populations (Kennebec, Hudson and Delaware River) may be increasing or stable, although data is limited. The New York (Hudson River) shortnose sturgeon population is the largest extant population of this species and, based on available data, appears to have increased (Bain et al. 2000). The most recent population estimate indicates this population consists of about 61,000-shortnose sturgeon (95% confidence interval [CI] was between 52,898 and 72,191 fish (Bain et al. 2000). A comparison of the Bain estimate to the 1979/1980 population estimate of spawning adults by Dovel *et al.,* (1992); about 13,000 fish) led Bain *et al.,* (2000) to conclude that the population had made a dramatic increase (about 400% increase) between 1979 and 1997. While still evidence of an increasing population, a comparison of total population estimates (30,000:60,000) would suggest the population has only doubled in size during the study years. Similarly, the Kennebec River population appears to be increasing. The most recent estimate for

this population is about 9,500 fish (Squiers 2003), suggesting the population has increased by about 30% in about a twenty year period.

Data from the Delaware River suggest that the population may be stable. Brundage and O'Herron (2006) estimate that the current population for the Delaware River is 12,047 adult fish (1999-2003; 95% CI: 10,757-13,589), which is similar to the 1981/84 estimate by Hastings *et al.,* (1987) of 12,796 fish (95% CI: 10,288-16367). The recent capture of several fish that were tagged as adults by Hastings *et al.,* (1987) suggests that older fish may comprise a substantial portion of the Delaware River population. Based on studies from other sturgeon species we know of no evidence of senescence in sturgeon and we would expect that these fish are reproductively active (Paragamian et al. 2005). Despite their longevity, the viability of sturgeon populations is sensitive to variability in juvenile recruitment and survival (Anders et al. 2002, Gross et al. 2002, Secor et al. 2002). Although interannual variation in juvenile recruitment would be expected as a result of stochastic factors that influence spawning and egg/larval survival, if the mean population size does not change over the long-term it then it would appear there is sufficient juvenile survival to provide at least periodic recruitment into the adult age classes. Data on juvenile recruitment or age-1+ survival would, however, establish whether this population is at a stable equilibrium.

South of Chesapeake Bay, populations are relatively small compared to their northern counterparts. The largest of the southern populations of shortnose sturgeon is the Altamaha River population. Population estimates have been calculated several times for sturgeon in the Altamaha since 1993. Total population estimates shown pretty sizeable interannual variation is occurring; estimates have ranged from as low as 468 fish in 1993 to over 6,300 fish in 2006 (NMFS 1998a, DeVries 2006). The Ogeechee River is the next most studied river south of Chesapeake Bay and abundance estimates indicate that the shortnose sturgeon population in this river is considerably smaller than that in the Altamaha River. The highest point estimate in 1993 using a modified Schnabel technique resulted in a total population estimate of 361 shortnose sturgeon (95% CI: 326-400). In contrast the most recent survey resulted in an estimate of 147 shortnose sturgeon (95% CI: 104-249), suggesting that the population may be declining.

Throughout the species range there are other extant populations, or at least evidence that several other basins are used periodically. Shortnose sturgeon have been documented in the St. Johns River (FL), the St. Mary's River, Chesapeake Bay, Potomac River, Piscataqua River, the Housatonic River and others. Some basins probably previously contained shortnose populations, but recent sampling has been largely unsuccessful. Despite the occasional observations of shortnose sturgeon, populations may be extinct in several basins (e.g., St. John's (FL), St. Mary's, Potomac, Housatonic and Neuse rivers). Those few fish that have been observed in these basins are generally presumed to be immigrants from neighboring basins. In some cases, (e.g. Chesapeake Bay) migratory information collected from tagged fish and genetic evidence confirms that fish captured in Chesapeake Bay were part of the Delaware River population (Grunwald et al. 2002, Wirgin et al. 2005).

Threats

Natural Threats. Yellow perch, sharks and seals are predators of shortnose sturgeon juveniles (NMFS 1998a). The effects of disease and parasites are generally unknown.

Anthropogenic Threats. Historic fishery harvests, as well as the incidental harvest in current fisheries, have had lasting effects on shortnose sturgeon. In the late nineteenth and early twentieth centuries shortnose sturgeon were harvested incidental to Atlantic sturgeon (NMFS 1998a). The effects of these harvests may have long-lasting impacts on some populations. At present there is no legal directed fishing effort for shortnose sturgeon in the U.S., although some illegal poaching is suspected. Additionally, shortnose sturgeon are often caught incidental to other fisheries. For instance, shortnose are caught incidentally by bass anglers, and incidentally to alewife/gaspereau and shad fisheries in the St. John River in Canada, shad fisheries in the Altamaha River, Hudson River, and others (COSEWIC 2005).

Habitat alterations from discharges, dredging or disposal of material into waterways and other developmental activities along riverine and estuarine systems threaten shortnose sturgeon habitat. Periodic maintenance of harbors and rivers likely results in the direct take of some sturgeon, but perhaps of greater impact is the manner in which dredging alters benthic topography and community structure and water quality (increase in suspended sediments). Shoreline development may increase the potential of ship strikes. In the Bay of Fundy, a tidal turbine killed at least three Atlantic salmon in the 1980s and may be a threat to shortnose sturgeon as well (Dadswell and Rulifson 1994). Although currently the only example of this type of turbine in North America, increasing interests in finding alternative energy sources is expected to result in an increase in the number of marine turbines along the coast.

Shortnose sturgeon and other benthic organisms are regularly in direct contact with legacy pollutants, as well as a suite of common contaminants added from more current industrial and agricultural practices. Studies demonstrate that shortnose sturgeon carry a wide number of potentially hazardous contaminants. Individuals from the Delaware River contain numerous metals (mercury, aluminum, antimony, barium, cadmium, calcium, chromium, copper, iron, magnesium, manganese, nickel, potassium, sodium, vanadium and zinc), PCDDs, PCDFs, PCBs, DDE, DDD, bis (2-ethylhexyl) phthalate, di-n-butylphthalate and chlordane (ERC 2002). Most of these metals, PCDDs, PCDFs and PCBs were also found in shortnose sturgeon in the Kennebec River (ERC 2003).

Critical Habitat
NMFS has not designated critical habitat for shortnose sturgeon.

Gulf sturgeon

Distribution
Gulf sturgeon are native to the Gulf of Mexico, from the Suwannee River in Florida to the Pearl River in Louisiana.

Status
Gulf sturgeon were listed as threatened on September 30, 1991, because of population declines caused by nearly a century of fishing pressure for meat and caviar, and habitat modifications caused by the disposal of dredged material, de-snagging (removal of trees and their roots), and other navigation maintenance activities; incidental take by commercial fishermen; poor water

quality associated with contamination by pesticides, heavy metals, and industrial contaminants; aquaculture and incidental or accidental introductions; and the Gulf sturgeon's slow growth and late maturation (56 *Federal Register* 49653).

Dependence on Waters of the United States

Gulf sturgeon are anadromous and spend the major part of a year in freshwater, migrating to saltwater in the fall. The best river habitat for gulf sturgeon are long, spring-fed free-flowing rivers. Steep banks and a hard bottom with an average water temperature of 60 to 72°F are also characteristic of rivers where sturgeon inhabit. Sturgeon occupy the river bottom downstream of springs where they seek thermal refuge during hot summer days.

Empirical studies of relationships between sturgeon and water quality have demonstrated that sturgeon populations are limited by low levels of dissolved oxygen levels and high temperatures in the rivers, streams, and estuaries they occupy; juvenile anadromous sturgeon also depend on the freshwater-brackish interface in the tidal portion of rivers for nursery areas. Siberian sturgeon (*Acipenser baeri*), for example, appear to have a preferred temperature range between 17.2 and 21.5° C and preferred dissolved oxygen levels between 5.9 and 13.2 mg/l (Khakimullin 1987). White sturgeon in the Bliss Reach of the Snake River (upstream of Brownlee Reservoir) were caught in water with temperatures between 10 and 22°C and dissolved oxygen levels between 8 and 16 mg/l (Lepla and Chandler 1995). Temperatures of 26°C and dissolved oxygen levels of 3 mg/l killed all juvenile Atlantic sturgeon (*Acipenser oxyrinchus*) in five out of six replicates (Secor and Gunderson 1997) and dissolved oxygen levels of 2.5 mg/l killed all 25-day old shortnose sturgeon (*Acipenser brevirostrum*), 96 percent of 32-day old shortnose sturgeon, 86 percent of all 64-day old sturgeon, and 12 percent of 104- to 310-day old shortnose sturgeon (Jenkins *et al*. 1993).

Critical Habitat

Critical habitat was designed for Gulf sturgeon in 2003 (68 *Federal Register* 13370). The designation encompasses 14 sites in Louisiana, Mississippi, Alabama, and Florida. The primary constituent elements essential for the conservation of Gulf sturgeon are those habitat components that support feeding, resting, and sheltering, reproduction, migration, and physical features necessary for maintaining the natural processes that support these habitat components. The primary constituent elements include:

1. Abundant prey items within riverine habitats for larval and juvenile life stages, and within estuarine and marine habitats and substrates for juvenile, subadult, and adult life stages;
2. Riverine spawning sites with substrates suitable for egg deposition and development, such as limestone outcrops and cut limestone banks, bedrock, large gravel or cobble beds, marl, soapstone or hard clay;
3. Riverine aggregation areas, also referred to as resting, holding, and staging areas, used by adult, subadult, and/or juveniles, generally, but not always, located in holes below normal riverbed depths, believed necessary for minimizing energy expenditures during fresh water residency and possibly for osmoregulatory functions;
4. A flow regime (*i.e,*. the magnitude, frequency, duration, seasonality, and rate-of-change of fresh water discharge over time) necessary for normal behavior, growth, and survival

of all life stages in the riverine environment, including migration, breeding site selection, courtship, egg fertilization, resting, and staging; and necessary for maintaining spawning sites in suitable condition for egg attachment, egg sheltering, resting, and larvae staging;

5. Water quality, including temperature, salinity, pH, hardness, turbidity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages;
6. Sediment quality, including texture and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages; and
7. Safe and unobstructed migratory pathways necessary for passage within and between riverine, estuarine, and marine habitats (*e.g.* a river unobstructed by any permanent structure, or a dammed river that still allows for passage).

Atlantic Sturgeon (5 DPSs)

Species description and distribution
Atlantic sturgeon are large fish with a long protruding snout. They lack scales and instead have armor-like plates along the dorsal, lateral, and ventral sides. Their historical range included major estuarine and riverine systems that spanned from Hamilton Inlet on the coast of Labrador to the Saint Johns River in Florida (Smith and Clugston 1997, NMFS 2007). Atlantic sturgeon have been documented as far south as Bermuda and Venezuela (Lee *et al.* 1980).

Listing status
Five DPSs of Atlantic sturgeon have been listed under the ESA. The Gulf of Maine DPS was listed as threatened while the New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs were listed as endangered (77 FR 5880, 77 FR 5914). NMFS has not designated critical habitat for Atlantic sturgeon.

Population designations, abundance, and trends
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 7 and 10 populations that can be statistically differentiated (King *et al.* 2001, Waldman *et al.* 2002, Wirgin *et al.* 2002, Wirgin *et al.* 2005, Grunwald *et al.* 2008). However, there is some disagreement among studies, and results do not include samples from all rivers inhabited by Atlantic sturgeon. NMFS (77 FR 5880, 77 FR 5914) identified five DPSs in the United States: the Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic. Overall, the genetic markers used in this analysis resulted in an average accuracy of 88% for determining a sturgeon's natal river origin, but an average accuracy of 94% for correctly classifying it to one of five DPSs.

Prior to 1890, Atlantic sturgeon populations were at or near carrying capacity. Between 1890 and 1905, Atlantic sturgeon (and shortnose sturgeon) populations were drastically reduced for sale of meat and caviar. Between 1920 and 1998, the harvest level remained very low due to small remnant populations. Prompted by research on juvenile production between 1985 and

1995 (Peterson *et al.* 2000), the Atlantic sturgeon fishery was closed by the Atlantic States Marine Fisheries Commission in 1998, when a coastwide fishing moratorium was imposed for 20-40 years, or at least until 20 year classes of mature female Atlantic sturgeon were present (ASMFC 1998).

Since the closure of the Atlantic sturgeon fishery, the only assessments of adult spawning populations have been made in the Hudson and Altamaha Rivers. While Atlantic sturgeon have been captured, tagged, and tracked through estuaries and rivers along the East Coast, no other estimates of spawning run size or juvenile population sizes have been made. Estimating the number of spawning adults relies on the assumptions that 1) all adults that migrate into the freshwater portion of a river are native to that river and 2) all adults are making that upstream migration with the intention of spawning. Kahnle *et al. (*2007) reported that approximately 870 adults per year returned to the Hudson River between 1985 and 1995. Peterson *et al.* (2008) reported that approximately 324 and 386 adults per year returned to the Altamaha River in 2004 and 2005, respectively.

Juvenile Atlantic sturgeon abundance may be a more precise way to measure the status of Atlantic sturgeon populations because it is believed that all age-1 and age-2 juveniles are restricted to their natal rivers (Dovel and Berggren 1983, Bain *et al.* 1999), avoiding the assumptions noted above. Peterson *et al.* (2000) reported that there were approximately 4,300 age-1 and -2 Atlantic sturgeon in the Hudson River between 1985 and 1995. Schueller and Peterson (2010) reported that age-1 and -2 Atlantic sturgeon population densities ranged from 1,000 to 2,000 individuals over a 4 year period from 2004 to 2007.

Threats

Sturgeon are often caught as bycatch in other fisheries, and this poses a major threat to the species. Most Atlantic sturgeon managers and researchers consider water quality as a moderate risk to every DPS in the United States (ASSRT 2007). During all stages of development, Atlantic sturgeon are sensitive to temperatures above 28°C (Niklitschek and Secor 2005, Kahn and Mohead 2010, Niklitschek and Secor 2010) and dissolved oxygen levels below 4.3 to 4.7 parts per million (Secor and Niklitschek 2002, Niklitschek and Secor 2009a). Juvenile sturgeon are also stressed by high salinities until they mature and out migrate. Additionally, sturgeons generally and Atlantic sturgeon specifically are sensitive to pesticides, heavy metals, and other toxins in the aquatic environment.

Life history

While intensely studied since the 1970s, many important aspects of Atlantic sturgeon life history are still unknown. The general life history pattern of Atlantic sturgeon is that of a long lived, late maturing, iteroparous, anadromous species. The species' historic range included major estuarine and riverine systems that spanned from Hamilton Inlet on the coast of Labrador to the Saint Johns River in Florida (reviewed in Murawski and Pacheco 1977, Smith and Clugston 1997). Atlantic sturgeon spawn in freshwater, but spend most of their sub-adult and adult life in the marine environment. While few specific spawning locations have been identified in the United States, through genetic analysis, many rivers are known to support reproducing populations. Early life stage Atlantic sturgeon coupled with upstream movements of adults suggest spawning

adults generally migrate upriver in the spring/early summer; February-March in southern systems, April-May in mid-Atlantic systems, and May-July in Canadian systems (Smith 1985, Bain 1997, Smith and Clugston 1997, Kahnle *et al.* 1998). Atlantic sturgeon spawn in freshwater, but spend most of their adult life in the marine environment. Some rivers may also support a fall spawning migration.

Sub-adult and adult Atlantic sturgeon undertake long marine migrations and utilize habitat up and down the East Coast for rearing, feeding, and migrating (Dovel and Berggren 1983, Bain 1997, Stevenson 1997). These migratory sub-adults, as well as adults, are normally located in shallow (10-50m) near shore areas dominated by gravel and sand substrate (Stein *et al.* 2004). Tagging and genetic data indicate that sub-adult and adult Atlantic sturgeon may travel widely once they emigrate from rivers. Once in marine waters, sub-adults undergo rapid growth (Dovel and Berggren 1983, Stevenson 1997). Despite extensive mixing in coastal waters, Atlantic sturgeon display high site fidelity to their natal streams. Straying between rivers within a proposed DPS would sometimes exceed five migrants per generation, but between DPSs was usually less than one migrant per generation, with the exception of fish from the Delaware River straying more frequently to southern rivers (Grundwald *et al.* 2008).

Atlantic sturgeon have been aged to 60 years (Mangin 1964); however, this should be taken as an approximation because the age validation studies conducted to date show ages cannot be reliably estimated after 15-20 years (Stevenson and Secor 1999). 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, Schueller and Peterson 2010) and three to five years for females (Vladykov and Greely 1963, Stevenson and Secor 1999, Schueller and Peterson 2010). Fecundity of Atlantic sturgeon has been correlated with age and body size (ranging from 400,000 – 8 million eggs) (Smith *et al.* 1982, Van Eenennaam and Doroshov 1998, Dadswell 2006). The average age at which 50 percent of maximum lifetime egg production is achieved estimated to be 29 years, approximately 3-10 times longer than for other bony fish species examined (Boreman 1997).

Sturgeon eggs are highly adhesive and are deposited on the bottom substrate, usually on hard surfaces (e.g., cobble) (Gilbert 1989; Smith and Clugston 1997). Hatching occurs approximately 94-140 hrs after egg deposition, and larvae assume a demersal existence (Smith *et al.* 1980). The yolksac larval stage is completed in about 8-12 days, during which time the larvae move downstream to rearing grounds over a 6 – 12 day period (Kynard and Horgan 2002). During the first half of their migration downstream, movement is limited to night. 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. Juvenile sturgeon continue to move further downstream into brackish waters, and eventually become residents in estuarine waters for months or years.

Rockfish

Bocaccio

Description of the species
The bocaccio is a rockfish species that genetic analyses suggest is composed of two distinct populations (Wishard et al. 1980, Matala et al. 2004). A southern population exists along the Pacific coasts of Mexican and California and is separated from a northern population by a region of apparent scarcity from northern California to southern Oregon (MacCall and He 2002b). It has been proposed that oceanographic features, such as current patterns restricting larval movement, are responsible for population discreteness (Matala et al. 2004, NMFS 2008e). The northern population is the entity that is proposed for listing. However, the presence of a third population has also been suggested (Queen Charlotte Island, Vancouver Island to Point Conception, California, and south of Point Conception)(Matala et al. 2004). For stock management purposes, the NMFS and Pacific Fisheries Management Council recognize these populations as separate stocks.

Distribution
Bocaccio occur from the central Baja peninsula of Mexico north along the continental shelf and slope as far as Stepovac Bay, Alaska (Love et al. 2002).

Habitat and movement
Preferred bocaccio habitat is largely dependent upon the life stage of an individual. Larvae and young juveniles tend to be found in deeper offshore regions (1-148 km offshore), but associated with the surface and occasionally with floating kelp mats (Hartmann 1987, Love et al. 2002, Emery et al. 2006). As individuals mature into older juveniles and adults, they transition into shallow waters and settle to the bottom, preferring algae-covered rocky, eelgrass, or sand habitats and aggregating into schools (Eschmeyer et al. 1983a, Love et al. 1991). After a few weeks, fish move into slightly deeper waters of 18-30 m and occupy rocky reefs (Feder et al. 1974, Carr 1983, Eschmeyer et al. 1983a, Johnson 2006, Love and Yoklavich 2008). As adults, bocaccio may be found in depths of 12-478 m, but tend to remain in shallow waters on the continental shelf (20-250 m), still associating mostly with reefs or other hard substrate, but may move over mud flats (Feder et al. 1974, Kramer and O'Connell 1995, Love et al. 2002, Love et al. 2005, Love and York 2005, Love et al. 2006). Artificial habitats, such as platform structures, also appear to be suitable habitat for bocaccio (Love and York 2006). Adults may occupy territories of 200-400 hectares, but can venture outside of this territory (Hartmann 1987). Adults tend to occupy deeper waters in the southern population compared to the northern population (Love et al. 2002). Adults are not as benthic as juveniles and may occur as much as 30 m above the bottom and move 100 m vertically during the course of a day as they move between different areas (Love et al. 2002, Starr et al. 2002). Prior to severe population reductions, bocaccio appeared to frequent the Tacoma Narrows in Washington State (DeLacy et al. 1964, Haw and Buckley 1971, Miller and Borton 1980).

Reproduction

Bocaccio are live-bearers with internal fertilization. Once females become mature (at 54-61 cm total length), they produce 20,000-2.3 million eggs annually, with the number increasing as females age and grow larger (Hart 1973b, Echeverria 1987, Love et al. 2002). However, either sex has been known to attain sexual maturity as small as 35 cm or 3 years of age and, in recent years as populations have declined, average age at sexual maturity may have declined as well (Hart 1973b, Echeverria 1987, Love et al. 2002, MacCall 2002b). Mating occurs between August and November, with larvae born between January and April (Lyubimova 1965, Moser 1967, Westrheim 1975, Wyllie Echeverria 1987, Love et al. 2002, MacCall and He 2002b).

Growth

Upon birth, bocaccio larvae measure 4-5 mm in length. These larvae move into pelagic waters as juveniles when they are 1.5-3 cm and remain in oceanic waters from 3.5-5.5 months after birth (usually until early June), where they grow at ~0.5-1 mm per day (Moser 1967, Matarese et al. 1989, Woodbury and Ralston 1991, Love et al. 2002, MacCall and He 2002b, MacCall 2003). However, growth can vary from year-to-year (Woodbury and Ralston 1991). Once individuals are 3-4 cm in length, they return to nearshore waters, where they settle into bottom habitats. Females tend to grow faster than males, but fish may take 5 years to reach sexual maturity (MacCall 2003). Individuals continue to grow until they reach maximum sizes of 91 cm, or 9.6 kg, at an estimated maximum age of 50 years (Eschmeyer et al. 1983a, Halstead et al. 1990, Ralston and Ianelli 1998, Love et al. 2002, Andrews et al. 2005, Piner et al. 2006). However, individuals tend to grow larger in more northerly regions (Dark et al. 1983).

Foraging

Prey of bocaccio vary with fish age, with bocaccio larvae starting with larval krill, diatoms, and dinoflagellates (Love et al. 2002). 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).

Status and trends

Bocaccio were proposed for listing on April 23, 2009 (74 FR 18516). Bocaccio as a species has undergone severe decline in the past several decades, with the species currently estimated to be 3.6% of its abundance in 1970 (MacCall and He 2002b). Prior to World War II, commercial landings of rockfish species generally remained under 20,000 lbs, but sky-rocketed during the war to 375,000 lbs annually and fluctuated between 50,000 and 220,000 lbs until 1970, when landings increased linearly with fishing effort to a peak of 900,000 lbs by 1980 (Palsson et al. 2008). Levels fluctuated after this between 48,000 and 300,000 lbs for the next decade and clearly crashed in the 1990's, with landings below 30,000 lbs annually. At the cessation of commercial fishing in 2003, 2,600 lbs of rockfish were harvested. Similar trends are seen in recreational landings from Puget Sound (WDF 1975-1986).

Among rockfish of the Puget Sound, bocaccio appear to have undergone a particular decline (MacCall and He 2002b). This has 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 2002b).

Bocaccio resistance to depletion and recovery is also hindered by demographic features (Love et al. 1998a). 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. 1998b). 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. 1998b, MacCall and He 2002b). Over the past 30 years, 1977, 1984, and 1988 are the only years in which recruitment appears to have been significant successes (it should be noted that 1999 and 2002 also appear to have been strong, but survivorship into maturity is still pending). Recruitment success appears to be linked to oceanographic/climactic patterns and may be related to cyclic warm/cool ocean periods, with cool periods having greater success (Sakuma and Ralston 1995, MacCall 1996, Love et al. 1998b, Moser et al. 2000b). Harvey *et al.* (2006a) suggested that bocaccio may have recently diverted resources from reproduction, potentially resulting in additional impairment to recovery. Overall, bocaccio have the highest variability of recruitment of any rockfish studied to date, with recruitment exhibiting a random walk and high temporal variability (MacCall and He 2002b, Tolimieri and Levin 2005).

Although population estimates are not available for the northern population, the southern population has been estimated to number 1.6 million fish of 1 year of age or older in 2002 (MacCall 2002a). Of these, 1.0 million were estimated to occur south of Pt. Conception, where recruitment has been stronger. However, individuals north of Pt. Conception tend to be larger and, hence, more fecund. In 2002, the southern population was estimated to produce 720 billion eggs annually (243 billion south of Pt. Conception). North of Pt. Conception, bocaccio are most abundant in the Monterey Bay area, where prime habitat seems to be over the continental slope and, secondarily, over the shelf (Dark et al. 1983).

The rate of decline for rockfish in Puget Sound has been estimated at ~3% annually for the period 1965-2007. Various rebuilding estimates for bocaccio populations have predicted recovery, but require long periods (98-170 years) and assume no mortality from fishing (intentional harvests are closed, but bycatch still occurs)(MacCall and He 2002a, MacCall 2008, NMFS 2008e).

Natural threats

Interspecies competition, predators, and climactic regimes are the primary natural factors that depress bocaccio numbers. Copper and quillback rockfish may compete with bocaccio in Puget Sound for available resources (NMFS 2008e). King salmon, lingcod, terns and other seabirds, harbor seals, and Steller sea lions are known predators of bocaccio and other rockfish species (Love et al. 2002, Beaudreau and Essington 2007, Lance and Jeffries 2007). Bocaccio and other rockfish appear to be negatively influenced by El Niño conditions, possibly reducing available prey supply (Moser et al. 2000a, Harvey 2005).

Anthropogenic threats

Although overfishing is the primary reason for bocaccio being proposed as a listed species,

bycatch and habitat loss are also human-related factors that have likely led to bocaccio decline. Although a frequent species captured in fisheries during the late 1970's, bocaccio were not recorded from any recreational surveys from 1996-2007 (WDF 1975-1986, Palsson et al. 2008). Apart from commercial fishing, recreational fishing (even catch-and-release) appears to incur significant mortality on bocaccio and other rockfishes (Schroeder and Love 2002). The species is considered overfished by the Pacific Fisheries Management Council and is not presently harvested intentionally. However, bycatch is still considered to be a high impact stressor to rockfish populations of Washington State waters (Palsson et al. 2008).

Habitat loss is also a factor in bocaccio decline, with rocky habitats (reportedly, there are only 217 km^2 in Puget Sound) being threatened by construction of bridges, sewer lines, cable and pipeline deployment, and dredge spoil (Palsson et al. 2008). Loss of kelp, which is valuable to juvenile fish recruitment, as well as anoxic conditions, exacerbate habitat loss (NMFS 2008e).

Critical habitat
Critical habitat has not been proposed or designated for the bocaccio.

Yelloweye rockfish

Description of the species
Yelloweye rockfish are likely composed of at least two populations and possibly more. Yamanaka *et al.* (2006) found that those individuals found within the Georgia Basin and Queen Charlotte Strait were genetically distinct from other samples from Oregon to Alaska. The Georgia Basin/Queen Charlotte Sound population is the one which has been proposed for listing in US waters.

Distribution
Yelloweye rockfish occur from Baja California to the Aleutian Islands, but are most common from central California to Alaska (Love et al. 2002).

Habitat
As with other rockfishes, yelloweye habitat varies based upon life stage. Larvae maintain a pelagic existence but as juveniles, move into shallow high relief rocky or sponge garden habitats (Eschmeyer et al. 1983a, Richards et al. 1985, Love et al. 1991). Juveniles may also associate with floating debris or pilings (Lamb and Edgell 1986). As adults, yelloweye rockfish move in to deeper habitats. Individuals have been found in waters as deep as 549 m, but are generally found in waters of less than 180 m (Eschmeyer et al. 1983a, Love et al. 2002). However, adults continue to associate with rocky, high relief habitats, particularly with caves and crevices, pinnacles, and boulder fields (Carlson and Straty 1981, Richards 1986, Love et al. 1991, O'Connell and Carlisle 1993, Yoklavich et al. 2000). Yelloweyes generally occur as individuals, with loose, residential aggregations infrequently found (Coombs 1979, DeMott 1983, Love et al. 2002). In the Puget Sound region, sport catch records from the 1970's indicate that Sucia Island and other islands of the San Juans as well as Bellingham Bay had the highest concentrations of catches (Delacy et al. 1972, Miller and Borton 1980).

Reproduction
Yelloweye rockfish are live bearers with internal fertilization. Copulation occurs between September and April, with fertilization taking place later as latitude increases (Hitz 1962, DeLacy et al. 1964, Westrheim 1975, O'Connell 1987, Wyllie Echeverria 1987, Lea et al. 1999). Puget Sound yelloweyes mate between winter and summer, giving birth from spring to late summer (Washington et al. 1978). Gestation lasts roughly 30 days (Eldridge et al. 2002). Although yelloweye rockfish were once believed to reproduce annually, evidence exists that indicate the potential for multiple births per year (MacGregor 1970, Washington et al. 1978). Females produce more eggs as they grow older and larger, with each individual producing roughly 300 eggs per year per gram of body weight (1.2-2.7 million eggs per year)(MacGregor 1970, Hart 1973b). In addition, older females of several rockfish species may be capable of provisioning their offspring better than their younger counterparts, meaning that they may be more a more influential component in a given year's recruitment success (Sogard et al. 2008).

Growth and development
Larvae are born at 4-5 mm in length and maintain a pelagic existence for the first 2 months of life, before moving to nearshore habitats and settling into rocky reef habitat at about 25 mm in length (DeLacy et al. 1964, Matarese et al. 1989, Moser 1996a, Love et al. 2002). Yelloweye growth is thought to vary by latitudinal gradient, with individuals in more northerly regions growing faster and larger. Year class strength appears to be most strongly linked to survival of the larval stage (Laidig et al. 2007). In general, sexual maturity appears to be reached by 50% of individuals by 15-20 years of age and 40-50 cm in length (Yamanaka and Kronlund 1997). As with other rockfish, yelloweyes can be long-lived (reported oldest age is 118 years)(Munk 2001). Maximum size has been reported as 910 cm, but assymptotic size in Alaskan waters for both males and females was estimated to be 690 cm and 659-676 mm along British Columbia (Clemens and Wilby 1961, Westrheim and Harling 1975, Rosenthal et al. 1982, Love et al. 2005, Yamanaka et al. 2006).

Movement
Individuals shift to deeper habitats as they age. Juveniles tend to begin life in shallow rocky reefs and graduate to deeper rocky habitats as adults. Once adult habitat is established, individuals tend to remain at a particular site (Love 1978, Coombs 1979, DeMott 1983).

Foraging
As with other rockfish species, yelloweye rockfish prey upon different species and size classes throughout their development. Larval and juvenile rockfish prey upon phyto- and zooplankton (Lee and Sampson 2009). Adult yelloweyes eat other rockfish (including members of their own species), sand lance, gadids, flatfishes, shrimp, crabs, and gastropods (Love et al. 2002, Yamanaka et al. 2006).

Status and trends
Yelloweye rockfish were proposed for listing on April 23, 2009 (73 FR 18516). Yelloweye rockfish abundance has been variable in the Puget Sound region over the past 60 years, ranging from less than 1% to greater than 3% of samples, although Wallace (2001) documented large

historical population in the Strait of Georgia. The latest samples have been historic lows in abundance. Perhaps more importantly, age classes appear to have been truncated to younger, smaller fish, severely hampering the ability of the species to recover from its primary cause of decline: overfishing (Berkeley et al. 2004).

Prior to World War II, commercial landings of rockfish species generally remained under 20,000 lbs, but sky-rocketed during the war to 375,000 lbs annually and fluctuated between 50,000 and 220,000 lbs until 1970, when landings increased linearly with fishing effort to a peak of 900,000 lbs by 1980 (Palsson et al. 2008). Levels fluctuated after this between 48,000 and 300,000 lbs for the next decade and clearly crashed in the 1990's, with landings below 30,000 lbs annually. At the cessation of commercial fishing in 2003, 2,600 lbs of rockfish were harvested. Over the period of 1965-2007, it is estimated that rockfish species has declined by 3% per year.

The most recent estimate of yelloweye rockfish abundance in the Puget Sound region was 3,000 individuals, with low abundance through spawning areas (Palsson et al. 2008).

Natural threats

Interspecies competition, predators, and climactic regimes are the primary natural factors that depress yelloweye rockfish numbers. Copper and quillback rockfish may compete with yelloweye rockfish in Puget Sound for available resources (NMFS 2008e). Lingcod, killer whales, and Steller sea lions are likely predators of yelloweye and other rockfish species (Love et al. 2002, Beaudreau and Essington 2007, Lance and Jeffries 2007). Yelloweye and other rockfish appear to be negatively influenced by El Niño conditions, possibly reducing available prey supply (Moser et al. 2000a, Harvey 2005, Black 2009). Oceanographic conditions (such as sea level anomalies and nearshore temperature conditions) appear to strongly influence the strength of each year's recruitment (Laidig et al. 2007). Rates of natural mortality have been reported to range from 2-4.6% annually (Yamanaka and Kronlund 1997, Wallace 2007).

Anthropogenic threats

Overfishing is considered the primary cause of yelloweye rockfish decline throughout their range, including in Washington State and British Columbian waters (Wallace 2007, NMFS 2008e). Although commercial harvesting of the species has ended, bycatch is still considered to be a high impact stressor to rockfish populations of Washington State waters (Palsson et al. 2008). It has been estimated that yelloweye rockfish have fallen 30% in abundance within 1/3 of a generation in the past few decades, an astonishing rate of decline.

Habitat loss is also a factor in yelloweye decline, with rocky habitats (reportedly, there are only 217 km^2 in Puget Sound) being threatened by construction of bridges, sewer lines, cable and pipeline deployment, and dredge spoil (Palsson et al. 2008). Anoxic conditions and chemical contamination are also considered threats to yelloweye rockfish recovery (NMFS 2008e).

Canary rockfish

Description of the species
It is unclear how many populations compose canary rockfish as a species. Genetic analysis have found that individuals south of Cape Blanco in southern Oregon lack an allele that individuals north of this point have (Wishard et al. 1980). This has been used to support the proposal of a northern DPS. In addition, canary rockfish are managed as two stocks in Canadian waters (COSEWIC in press). However, clear evidence of genetically or morphologically distinct populations is still lacking.

Distribution
Canary rockfish are found from the northern Baja peninsula north to the western Gulf of Alaska, and with the greatest abundance along British Columbia to central California (Miller and Lea 1972, Hart 1973b, Cailliet et al. 2000, Love et al. 2002).

Habitat
Canary rockfish occupy a variety of habitats based upon their life stage. Larvae and younger juveniles tend to occupy shallow waters at the beginning of their lives, but generally remain in the upper 100 m of the water column (Love et al. 2002). Juveniles initially settle into tide pools and rocky reefs (Miller and Geibel 1973, Love et al. 1991, Cailliet et al. 2000, Love et al. 2002). Juveniles have also been observed in diurnal movements, occurring near sand-rock interfaces in groups by day and moving over sandy areas at night (Love et al. 2002). After as much as 3 years, juveniles move into deeper rocky reefs, forming loose schools, rarely on but generally near the bottom (Phillips 1960, Boehlert 1980, Lamb and Edgell 1986, Rosenthal et al. 1998, Starr 1998, Cailliet et al. 2000, Johnson et al. 2003, Methot and Stewart 2005, Tissot et al. 2007). Adults may be found in waters of up to 400 m, but tend to be most common in the 80-200 m range, or even shallower (Moser 1996b, Methot and Stewart 2005, Tissot et al. 2007). Mid shelf locations seem to have the highest concentrations of canary rockfish off Washington and Oregon (Weinberg 1994). Adults tend to occur in shallow areas in higher latitudes than their southern counterparts, although adults do appear to move into progressively deeper waters as they age (Vetter and Lynn 1997, Methot and Stewart 2005). It is believed that, within Puget Sound, canary rockfish were most common in the 1960's and 1970's in Tacoma Narrows, Hood Canal, San Juan Islands, Bellingham, and Appletree Cove (Delacy et al. 1972, Miller and Borton 1980). A latitudinal gradient may be present by age class, with older and larger individuals preferably occupying more northerly habitat (Dark et al. 1983).

Movement
Individual canary rockfish can range widely (up to 700 km over several years), although patterns of residency have been observed (Gascon and Miller 1981, DeMott 1983, Casillas et al. 1998, Lea et al. 1999, Love et al. 2002). In addition, seasonal movements have been found, with individuals moving from 160-210 m depths in late winter to 100-170 m in late summer (COSEWIC in press).

Reproduction
Canary rockfish develop their young internally before giving birth to live young as larvae.

During each annual spawning event, a female can produce 260,000 to 1.9 million eggs, depending upon her size and age (Guillemot et al. 1985, NMFS 2008e). Unlike some other rockfish, there does not appear to be a latitudinal or geographic gradient associated with number of eggs produced (Gunderson et al. 1980, Love et al. 2002). Birth takes place in Oregonian and Washingtonian waters between September through March, with a peak in December and January. The peak in British Columbian waters is slightly later (February)(Hart 1973b, Westrheim and Harling 1975, Wyllie Echeverria 1987, Barss 1989).

Growth and development

When born, larvae are 3.6-4.0 mm in length and take from 1-4 months to develop into juveniles (Waldron 1968, Richardson and Laroche 1979, Stahl-Johnson 1985, Moser 1996a, Krigsman 2000, Love et al. 2002). As with other rockfish, females seem grow more quickly than do males, with females reaching sexual maturity at 7-9 years of age (35-45 cm in length) versus males at 7-12 years (~41 cm in length) off Oregon (Westrheim and Harling 1975, Boehlert and Kappenman 1980, Lenarz and Echeverria 1991, STAT 1999). Mean length at sexual maturity off Vancouver Island is 41 cm for females and 48 cm for males (Westrheim and Harling 1975). Canary rockfish are known to frequently reach 60-75 years of age and have been found to be as old as 84 years (Cailliet et al. 2000, Cailliet et al. 2001, Andrews et al. 2007). Maximum reported sizes are 76 cm and 4.5 kg (Boehlert 1980, IGFA 1991, Williams et al. 1999, Love et al. 2002, Methot and Stewart 2005).

Foraging

Canary rockfish prey upon different species as they age. Larvae are planktivores, consuming invertebrate eggs, copepods, and nauplii (Moser and Boehlert 1991, Love et al. 2002). Juveniles feed upon zooplankton, including crustaceans, juvenile polychaetes barnacle cyprids, and euphasiid eggs and larvae (Gaines and Roughgarden 1987, Love et al. 1991). However, adults move into a carnivorous lifestyle as well as eating euphasiids and other crustaceans. Adults consume other fishes such as shortbelly rockfish, mytophids and stomiatiods (Cailliet et al. 2000, Love et al. 2002). However, oceanographic and climactic shifts can alter foraging such that canary rockfish feed on other available species (Lee and Sampson 2009).

Status and trends

Canary rockfish were proposed for listing on April 23, 2009 (74 FR 18516). Canary rockfish were once considered common in Puget Sound, but has declined at a faster rate than any other rockfish species in the region (Holmberg et al. 1967, NMFS 2008e). Prior to World War II, commercial landings of rockfish species generally remained under 20,000 lbs, but sky-rocketed during the war to 375,000 lbs annually and fluctuated between 50,000 and 220,000 lbs until 1970, when landings increased linearly with fishing effort to a peak of 900,000 lbs by 1980 (Palsson et al. 2008). Levels fluctuated after this between 48,000 and 300,000 lbs for the next decade and clearly crashed in the 1990's, with landings below 30,000 lbs annually. At the cessation of commercial fishing in 2003, 2,600 lbs of rockfish were harvested. Canary rockfish have been noted for being much less frequently caught in the Puget Sound and Georgia Basin region since 1965 (NMFS 2008e). The rate of decline for rockfish in Puget Sound has been estimated at ~3% annually for the period 1965-2007.

Declines have been noted in both numbers as well as frequencies. This likely due to the targeted removal of larger, older, and more fecund individuals by commercial fisheries, reducing the ability of canary rockfish to rebound from excessive mortality (NMFS 2008e). For example, recreational fishing data have not reported any individuals caught greater than 55 cm since 2000, whereas a variety of large size classes had formerly been caught. There are concerns that even now some populations have been lost entirely, primarily due to over harvesting, but also due to low dissolved oxygen levels in some areas of Puget Sound (NMFS 2008e).

Natural threats
Interspecies competition, predators, and climactic regimes are the primary natural factors that depress canary rockfish numbers. Copper and quillback rockfish may compete with canary rockfish in Puget Sound for available resources (NMFS 2008e). Predators of canary rockfish include other rockfishes, lingcod (for which rockfish is a particularly important dietary component), cabezon, seabirds, salmon, sharks, dolphins, seals, Steller sea lions, and perhaps river otters (Merkel 1957, Miller and Geibel 1973, Morejohn et al. 1978, Roberts 1979, Antonelis Jr. and Fiscus 1980, Ainley et al. 1981, Rosenthal et al. 1982, Stevens and Miller 1983, Love et al. 1991, Beaudreau and Essington 2007, Lance and Jeffries 2007). Canary and other rockfishes appear to be negatively influenced by El Niño conditions, possibly reducing available prey supply (Moser et al. 2000a, Harvey 2005).

Anthropogenic threats
Overharvesting the primary cause of canary rockfish declines, but habitat loss is also important. Canary rockfish are considered overfished by the Pacific Fisheries Management Council and are not presently harvested intentionally. However, bycatch is still considered to be a high impact stressor to rockfish populations of Washington State waters (Palsson et al. 2008). Habitat loss is also a factor in canary rockfish decline, with rocky habitats (reportedly, there are only 217 km^2 in Puget Sound) being threatened by construction of bridges, sewer lines, cable and pipeline deployment, and dredge spoil (Palsson et al. 2008). Low oxygen levels as well as pollutant, chemical, and nutrient loading are also considered significant threats to canary rockfish recovery (NMFS 2008e).

Critical habitat
Critical habitat has not been designated or proposed for canary rockfish.

Smalltooth Sawfish

Description of the species
The smalltooth sawfish is a tropical marine and estuarine elasmobranch fish (sharks and rays) that has been reported to have a circumtropical distribution. Although they are rays, sawfish physically more resemble sharks, with only the trunk and especially the head ventrally flattened. Smalltooth sawfish are characterized by their “saw,” a long, narrow, flattened rostral blade with a series of transverse teeth along either edge.

Distribution
In the western Atlantic, the smalltooth sawfish has been reported from Brazil through the

Caribbean and Central America, the Gulf of Mexico, and the Atlantic coast of the US. The smalltooth sawfish has also been recorded from Bermuda (Bigelow and Schroeder 1953). Forms of smalltooth sawfish have been reported from the eastern Atlantic in Europe and West Africa; the Mediterranean; South Africa; and the Indo-West Pacific, including the Red Sea, India, Burma, and the Philippines (Bigelow and Schroeder 1953, Van der Elst 1981, Compagno and Cook 1995). Whether populations outside of the Atlantic are truly smalltooth sawfish or closely related species is unknown (Adams and Wilson 1995). Pacific coast records of smalltooth sawfish off Central America need confirmation (Bigelow and Schroeder 1953, Compagno and Cook 1995).

The range of the smalltooth sawfish in the Atlantic has contracted markedly over the past century. The northwestern terminus of their Atlantic range is located in the waters of the eastern US. Historic capture records within the US range from Texas to New York. Water temperatures no lower than 61°F to 64.4°F and the availability of appropriate coastal habitat serve as the major environmental constraints limiting the northern movements of smalltooth sawfish in the western North Atlantic (Simpfendorfer 2001). As a result, most records of this species from areas north of Florida occur during spring and summer periods (May to August) when inshore waters reach appropriately high temperatures. The data also suggest that smalltooth sawfish may utilize warm water outflows of power stations as thermal refuges during colder months to enhance their survival or become trapped by surrounding cold water from which they would normally migrate. Almost all occurrences of smalltooth sawfish in warm-water outflows were during the coldest part of the year, when water temperatures in these outfalls are typically well above ambient temperatures.

Movement

Historic records of smalltooth sawfish indicate that some large mature individuals migrated north along the US Atlantic coast as temperatures warmed in the summer and then south as temperatures cooled (Bigelow and Schroeder 1953). Recent Florida encounter data, however, do not suggest such migration. Only two smalltooth sawfish have been recorded north of Florida since 1963 (the first was captured off of North Carolina in 1999 and the other off Georgia 2002) but it is unknown whether these individuals resided in Georgia and North Carolina waters annually or if they had migrated north from Florida (Schwartz 2003, Burgess unpublished data). Given the very limited number of encounter reports from the east coast of Florida, Simpfendorfer and Wiley (2004) hypothesize the population previously undertaking the summer migration has declined to a point where the migration is undetectable or does not occur.

Most specimens captured along the Atlantic coast north of Florida have also been large (>2.7 m) adults and likely represent seasonal migrators, wanderers, or colonizers from a core population(s) to the south rather than being members of a continuous, even-density population (Bigelow and Schroeder 1953). It is likely that these individuals migrated southward toward Florida as water temperatures declined in the fall, as there is only one winter record from the Atlantic coast north of Florida. Based on smalltooth sawfish encounter data, the current core range for the smalltooth sawfish is from the Caloosahatchee River, Florida, to Florida Bay (NMFS 2000, Simpfendorfer

and Wiley 2004).[4]

Habitat

Smalltooth sawfish are euryhaline, occurring in waters with a broad range of salinities from freshwater to full seawater (Simpfendorfer 2001). Younger, smaller individuals tend to inhabit very shallow mud banks and tides are a major factor in their movement (Simpfendorfer et al. 2010). At this size smalltooth sawfish spend the vast majority of their time on shallow mud or sand banks that are less than 1 foot (30 cm) deep. As they grow, juveniles tend to occupy deeper habitat, but shallow areas (<1 m depth) remain preferred habitat; juveniles also expand their ranges, whereas small individuals have very restricted ranges (Simpfendorfer et al. 2010). Acoustic tracking studies have shown that at this size sawfish will remain associated with the same mud bank over periods of several days. These banks are often very small and daily home range sizes can be of the magnitude of 100–1,000 m^2 (Simpfendorfer 2003b). Acoustic monitoring studies have shown that juveniles have high levels of site fidelity for specific nursery areas for periods up to almost 3 months (Wiley and Simpfendorfer 2007). Their occurrence in freshwater is suspected to be only in estuarine areas temporarily freshwater from receiving high levels of freshwater input. Many encounters are reported at the mouths of rivers or other sources of freshwater inflows, suggesting estuarine areas may be an important factor in the species distribution (Simpfendorfer and Wiley 2004).

Information on juvenile smalltooth sawfish indicates that they prefer shallow euryhaline habitats adjacent to red mangroves (NMFS 2006b). They do still have a preference for shallow water, remaining in depths mostly less than 90 cm. Several sawfish approximately 150 cm in length fitted with acoustic tags have been relocated in the same general areas over periods of several months, suggesting a high level of site fidelity (Simpfendorfer 2003). The daily home ranges of these animals are considerably larger (1–5 km^2) than for the very small sawfish and there is less overlap in home ranges between days.

Smalltooth sawfish are most common in shallow coastal waters less than 25 m (Bigelow and Schroeder 1953, Adams and Wilson 1995). Indeed, the distribution of the smallest size classes of smalltooth sawfish indicate that nursery areas occur throughout Florida in areas of shallow water, close to shore and typically associated with mangroves (Simpfendorfer and Wiley 2004). However, encounter data indicate there is a tendency for smalltooth sawfish to move offshore and into deeper water as they grow. Larger animals are more likely to be found in deeper waters. Poulakis and Seitz (2004b) reported that almost all of the sawfish <3 m in length were found in water less than 10 m deep and 46% of encounters individuals >3 m in Florida Bay and the Florida Keys were reported at depths between 70 to 122 m. Since large animals are also observed in very shallow waters, it is believed that smaller (younger) animals are restricted to shallow waters, while large animals roam over a much larger depth range (Simpfendorfer 2001). Recent data from sawfish encounter reports and from satellite tagging indicate mature animals

[4] See the 2006 Draft Recovery Plan for more detailed information on the historic and current distribution of smalltooth sawfish in four regions of the eastern U.S. This information is based on the Status Review Team's analysis and the more recent encounter database research (NMFS 2006b).

occur regularly in waters in excess of 164 feet (Poulakis and Seitz 2004a, Simpfendorfer and Wiley 2004).

Growth and reproduction

As in all elasmobranchs, fertilization is internal. Bigelow and Schroeder (1953) report the litter size as 15 to 20. Simpfendorfer and Wiley (2004), however, caution this may be an overestimate, with recent anecdotal information suggesting smaller litter sizes (about ten). Smalltooth sawfish mating and pupping seasons, gestation, and reproductive periodicity are all unknown. Gestation and reproductive periodicity, however, may be inferred based on that of the largetooth sawfish, sharing the same genus and having similarities in size and habitat. Thorson (1976) reported the gestation period for largetooth sawfish was approximately five months and concluded that females probably produce litters every second year.

Bigelow and Schroeder (1953) describe smalltooth sawfish as generally about 61 cm long at birth and growing to a length of 5.5 m or greater. Recent data from smalltooth sawfish caught off Florida, however, demonstrate young are born at 76 to 87 cm (Simpfendorfer and Wiley 2004), with males reaching maturity at approximately 2.7 m and females at approximately 3.6 m (Simpfendorfer 2002). A recent study by Simpfendorfer suggests rapid juvenile growth occurs during the first two years after birth (Simpfendorfer 2008). First year growth is 65-85 cm and second year growth is 48-68 cm. Growth rates beyond two years are uncertain; however, the average growth rate of captive smalltooth sawfish has been reported between 13.9 and 19.6 cm per year. The maximum reported size of a smalltooth sawfish is 24.9 feet (Last and Stevens 1994), but the maximum size normally observed is 7.6 m (Adams and Wilson 1995). No formal studies on the age and growth of the smalltooth sawfish have been conducted to date, but growth studies of largetooth sawfish suggest slow growth, late maturity (10 years) and long lifespan (25 to 30 years Thorson 1982, Simpfendorfer 2000b). These characteristics suggest a very low intrinsic rate of increase (Simpfendorfer 2000b).

Simpfendorfer estimated intrinsic rates of natural population increase at 0.08 to 0.13 per year and population doubling times from 5.4 to 8.5 years (Simpfendorfer 2000a).

Feeding

Smalltooth sawfish feed primarily on fish, with mullet, jacks, and ladyfish believed to be their primary food resources (Simpfendorfer 2001). In addition to fish, smalltooth sawfish also prey on crustaceans (mostly shrimp and crabs), which are located by disturbing bottom sediment with their saw (Norman and Fraser 1937, Bigelow and Schroeder 1953).

Status and trends

The US smalltooth sawfish distinct population segment (DPS) was listed as endangered under the ESA on April 1, 2003 (68 FR 15674). The smalltooth sawfish is the first marine fish to be listed in the US. Despite being widely recognized as common throughout their historic range up until the middle of the 20th century, the smalltooth sawfish population declined dramatically during the middle and later parts of the century. The decline in the population of smalltooth sawfish is attributed to fishing (both commercial and recreational), habitat modification, and sawfish life history. Large numbers of smalltooth sawfish were caught as bycatch in the early

part of this century. Smalltooth sawfish were historically caught as bycatch in various fishing gears throughout their historic range, including gillnet, otter trawl, trammel net, seine, and to a lesser degree, handline. Frequent accounts in earlier literature document smalltooth sawfish being entangled in fishing nets from areas where smalltooth sawfish were once common but are now rare (Evermann and Bean 1898). Loss and/or degradation of habitat contributed to the decline of many marine species and continue to impact the distribution and abundance of smalltooth sawfish. Simpfendorfer (2001) estimated that the US population size is currently less than 5% of its size at the time of European settlement. One dataset from shrimp trawlers off Louisiana from the late 1940s through the 1970s suggests a rapid decline in the species from the period 1950-1964 (NMFS 2006b).

Seitz and Poulakis (2002) and Poulakis and Seitz (2004a) documented recent (1990 to 2002) occurrences of sawfish along the southwest coast of Florida, and in Florida Bay and the Florida Keys, respectively and includes a total of 2,969 smalltooth sawfish encounters. Mote Marine Laboratory also maintains a smalltooth sawfish public encounter database, established in 2000 to compile information on the distribution and abundance of sawfish. A total of 434 sawfish encounters have been validated since 1998, most from recreational fishers (Simpfendorfer and Wiley 2004). Dr. Simpfendorfer reluctantly gives an estimate of 2,000 individuals based on his four years of field experience and data collected from the public, but cautions that actual numbers may be plus or minus at least 50%.

The majority of smalltooth sawfish encounters today are from the southwest coast of Florida between the Caloosahatchee River and Florida Bay. Outside of this core area, the smalltooth sawfish appears more common on the west coast of Florida and in the Florida Keys than on the east coast, and occurrences decrease the greater the distance from the core area (Simpfendorfer and Wiley 2004). The capture of a smalltooth sawfish off Georgia in 2002 is the first record north of Florida since 1963. New reports during 2004 extend the current range of the species to Panama City, offshore Louisiana (south of Timbalier Island in 100 feet of water), southern Texas (unconfirmed), and the northern coast of Cuba.

The abundance of juveniles encountered, including very small individuals, suggests that the population remains reproductively active and viable (Seitz and Poulakis 2002, Simpfendorfer 2003a, Simpfendorfer and Wiley 2004). The declining numbers of individuals with increasing size is consistent with the historic size composition data (G. Burgess, pers. comm. in Simpfendorfer and Wiley 2004). This information and recent encounters in new areas beyond the core abundance area suggest that the population may be increasing. From 1989-2004, smalltooth sawfish relative abundance has increased by about 5 percent per year (Carlson et al. 2007). However, recovery of the species expected to be slow on the basis of the species' life history and other threats to the species remaining (see below), the population's future remains tenuous. Based on genetic sampling, the estimates of current effective population size are between 269.6 and 504.9 individuals (95% CI 139.3 – 1,515). (NMFS 2011). This number is usually 25-50% of census population size (breeding adults) in elasmobranchs, so it is likely that the breeding population consists of high hundreds to low thousands of individuals (NMFS 2011).

Natural threats
The primary natural threat to smalltooth sawfish survival is the species low reproductive rate. In the face of reduced population sizes, this biological parameter means that recovery, at best, will be slow, and that catastrophic perturbations can have severer consequences to recovery.

Anthropogenic threats
Smalltooth sawfish decline has been largely due to fisheries interaction (see NMFS 2006c for a review). The distinctive "saw" can easily become entangled in a variety of commercial and recreational fishing gear, resulting in drowning or injury. Even when individuals that have been entangled are retrieved alive, individuals may be killed for curio collection of the saw, fear of injury from fisherman, or injured from the gear or handling during gear removal. However, additional anthropogenic impacts result from habitat loss. Destruction of mangrove habitat, dredging, trawling and filling, and loss of reef habitat have negative impacts on all life stages of smalltooth sawfish. Although a concern, pollution impacts on particularly reproductive biology are unknown. However, habitat degradation due to runoff containing pesticides, eutrophying agents, and other contaminants can also have a negative impact on smalltooth sawfish habitat.

Critical habitat
On September 2, 2009, critical habitat was designated for smalltooth sawfish along the central and southwest coast of Florida (74 FR 45353). Although PCEs were not identified, the mangrove and adjacent shallow euryhaline habitat are important nursery habitat for smalltooth sawfish.

Sea Turtles

Loggerhead sea turtle, (All DPS)

Species Description and Distribution
The loggerhead sea turtle has a large head and powerful jaws, which enables it to feed on hard-shelled prey, such as whelks and conch. It is the most abundant sea turtle in U.S. coastal waters. It occurs throughout tropical and temperate oceans. Loggerhead nesting is confined to lower latitude temperate and subtropic zones but does not occur in tropical areas (NRC 1990b, NMFS and USFWS 1991b, Witherington et al. 2006).

Listing Status
In 1978, the loggerhead sea turtle was listed as threatened under the ESA (43 FR 32800). In 2011, the listing was modified to include nine distinct population segments (76 FR 58868). The Northwest Atlantic Ocean DPS has a large population size (>60,000 adults). Though the nesting trend (between 1989 and 2010) is slightly negative, it is not significantly different from zero (76 FR 58868). Therefore, the Northwest Atlantic Ocean DPS was listed as threatened. Critical habitat for this DPS has yet to be designated.

Threats
Incidental take in commercial fisheries has been the greatest source of mortality for loggerhead

sea turtles over the last half century (TEWG 2009). Shrimp trawls were the major source of mortality until 2003 when turtle excluder devices were mandated. Pelagic longline, finfish trawl, gill net, drift net, and pound net fisheries are additional sources of mortality. Estimated mean annual bycatch in the Atlantic is 26,500 loggerheads, of which an estimated 1,400 die (Finkbeiner et al. 2011b). The TEWG (2009) concludes that incidental capture of loggerheads in these commercial fisheries "may certainly be playing a role in the recent apparent decrease in the numbers of adult female loggerheads in the Western North Atlantic." Commercial harvest of turtles and eggs also plays a role. Threats to nesting habitat include: coastal development and construction, placement of erosion control structures, beachfront lighting, vehicular and pedestrian traffic, sand extraction, beach erosion, beach nourishment, beach pollution, removal of native vegetation, and planting of non-native vegetation (Baldwin 1992, USFWS 1998, Margaritoulis et al. 2003, Mazaris et al. 2009). Loggerhead sea turtles face numerous threats in the marine environment as well, including oil and gas exploration, marine pollution, trawl, purse seine, hook and line, gill net, pound net, longline, and trap fisheries, underwater explosions, dredging, offshore artificial lighting, power plant entrapment, entanglement in debris, ingestion of marine debris, marina and dock construction and operation, boat collisions, and poaching. Climate change may also have significant implications on loggerhead populations worldwide. In addition to potential loss of nesting habitat due to sea level rise, loggerhead sea turtles are very sensitive to temperature as a determinant of sex while incubating. Ambient temperature increase by just 1º-2º C can potentially change hatchling sex ratios to all or nearly all female in tropical and subtropical areas (Hawkes et al. 2007a). Increasing ocean temperatures may also lead to reduced primary productivity and food availability. Natural threats include predation by sharks, seabirds, raccoons, and crabs (Barton and Roth 2008).

Life History

Loggerhead females mature at 15 – 38 years of age (Frazer and Ehrhart 1985, NMFS 2001, Witherington et al. 2006, Casale et al. 2009). They breed approximately every 2-3 years (Richardson et al. 1978, Dodd 1988), and lay 4 – 5 nests per season, which generally lasts from March until August (Murphy and Hopkins 1984, Tucker 2010). The average clutch size is 100-130 eggs (Dodd 1988). There are five life stages for the loggerhead: year one (terrestrial to oceanic hatchlings); juvenile one (exclusively oceanic), juvenile two (oceanic or neritic, 41 – 82 cm standard carapace length), juvenile three (oceanic or neritic, 63 – 100 cm standard carapace length), and adult (TEWG 2009).

Diet

Loggerhead sea turtles are omnivorous and opportunistic feeders through their lifetimes (Parker et al. 2005). Hatchling loggerheads feed on macroplankton associated with *Sargassum* spp. communities (NMFS and USFWS 1991b). Pelagic and benthic juveniles forage on crabs, mollusks, jellyfish, and vegetation at or near the surface (Dodd 1988, Wallace et al. 2009). Loggerheads in the deep, offshore waters of the western North Pacific feed on jellyfish, salps, and other gelatinous animals (Dodd 1988, Hatase et al. 2002). Sub-adult and adult loggerheads prey on benthic invertebrates such as gastropods, mollusks, and decapod crustaceans in hard-bottom habitats, although fish and plants are also occasionally eaten (NMFS and USFWS 1998d). Stable isotope analysis and study of organisms on turtle shells has recently shown that although a loggerhead population may feed on a variety of prey, individuals have specialized

diets (Reich et al. 2010, Vander Zanden et al. 2010).

Diving
Loggerhead diving behavior varies based upon habitat, with longer surface stays in deeper habitats than in coastal ones. Routine dives can last 4–172 min (Byles 1988, Sakamoto et al. 1990, Renaud and Carpenter 1994). The maximum-recorded dive depth for a post-nesting female was over 230 m, although most dives are far shallower (9-21 m (Sakamoto et al. 1990).

Kemp's ridley turtle (*Lepidochelys kempii*)

Species description and distribution.
The Kemp's ridley turtle is the smallest of all sea turtles. It was formerly known only from the Gulf of Mexico and along the Atlantic coast of the U.S. (TEWG 2000); however, there have been recent records of Kemp's ridley turtles in the Mediterranean Sea (Tomas and Raga 2008).

Status
In 1970, the Kemp's ridley turtle was listed as endangered under the ESA (35 FR 18319). The International Union for Conservation of Nature (IUCN) has classified the species as critically endangered. It was once considered the most endangered sea turtle (NRC 1990a, USFWS 1999). NMFS has not designated critical habitat for Kemp's ridley sea turtle.

Population designations, abundance, and trends
Kemp's ridley females lay their eggs in synchronized nesting events called aribadas. The species was once abundant in the Gulf of Mexico, with arribadas of 40,000 turtles at Rancho Nuevo, Mexico (Hildebrand 1963). By 1978, arribadas had declined to ~200 turtles (USFWS and NMFS 1992, TEWG 2000). Protection led to increases in abundance in the early 1990s. By 2006, there were approximately 7,866 nests at Rancho Nuevo and ~12,000 nests throughout Mexico which translates to an estimated 4,000 females nesting annually (Rostal et al. 1997, USFWS 2006, Rostal 2007) and a total population size of 7,000 – 8,000 adult females (Marquez et al. 1989, TEWG 2000, Rostal 2007). In Mexico, there were 17,882 nests in 2008 (Gladys Porter Zoo 2008) and 21,144 nests in 2009 (Gladys Porter Zoo 2010). In 2010, nesting declined to 13,302 nests (Gladys Porter Zoo 2010) but recovered to 20,570 nests in 2011 (http://www.fws.gov/northflorida/SeaTurtles/Turtle%20Factsheets/kemps-ridley-sea-turtle.htm). A successful head-start program has resulted in the reestablishment of nesting at South Padre Island: in 2011, 199 turtles nested on Texan beaches (http://www.nps.gov/pais/naturescience/current-season.htm).

Threats
The primary cause for the mid-century decline in Kemp's ridley turtles was overharvesting of turtles and eggs. This threat has subsided in recent years. Habitat destruction remains a concern in the form of bottom trawling and shoreline development. Finkbeiner et al. (2011a) estimated that at least 98,300 individuals are caught annually in U.S. Atlantic fisheries (resulting in at least 2,700 mortalities). Kemp's ridley turtles have among the highest levels of PCB and DDT (Pugh and Becker 2001). Oil can also be hazardous to Kemp's ridley turtles, causing significant

mortality and morphological changes in hatchlings (Fritts and McGehee 1981). Natural threats include predation by sharks and "cold stunning," mortality induced by cold water temperatures (Innis et al. 2009).

Life History

Kemp's ridley females mature at 10 – 17 years of age (Snover et al. 2007). They nest multiple times in a single nesting season, producing an average of 3.075 nests per female per season (Rostal 2007). The annual average clutch size is 94 – 100 eggs per nest (Marquez-M. 1994, USFWS 2000, 2001, 2002, 2003, 2004, 2005a, 2006, Rostal 2007). The inter-nesting interval is approximately 1.8 to 2.0 years (Marquez et al. 1989, TEWG 2000, Rostal 2007).
Nesting occurs in large arribadas, primarily at Rancho Nuevo, Mexico (NMFS et al. 2011). Nesting at this location may be particularly important because hatchlings can more easily migrate to foraging grounds (Putman et al. 2010). Kemp's ridleys require approximately 1.5 – 2 years to grow from a hatchling to a size of approximately 7.9 inches long, at which size they are capable of making a transition to a benthic coastal immature stage, but can range from one to four years or more (Ogren 1989, Caillouet et al. 1995, Schmid and Witzell 1997, Zug et al. 1997, Schmid 1998, TEWG 2000, Snover et al. 2007). Based on the size of nesting females, it is assumed that turtles must attain a size of approximately 23.6 inches long prior to maturing (Marquez-M. 1994). Growth models based on mark-recapture data suggest that a time period of seven to nine years would be required for this growth from benthic immature to mature size (Schmid and Witzell 1997, Snover et al. 2007).

Migration and movement

Tracking of post-nesting females from Rancho Nuevo and Texas beaches indicates that turtles move along coastal migratory corridors either to the north or south from the nesting beach (Byles 1989b, Byles and Plotkin 1994, Renaud 1995, Renaud et al. 1996, Shaver 1999, 2002). These migratory corridors appear to extend throughout the coastal areas of the Gulf of Mexico and most turtles appear to travel in waters less than roughly 164 feet in depth. Turtles that headed north and east traveled as far as southwest Florida, whereas those that headed south and east traveled as far as the Yucatan Peninsula, Mexico (Morreale et al. 2007). Following migration, Kemp's ridley sea turtles settle into resident feeding areas for several months (Byles and Plotkin 1994, Morreale et al. 2007). Females may begin returning along relatively shallow migratory corridors toward the nesting beach in the winter in order to arrive at the nesting beach by early spring.

Habitat

Juvenile turtles occur in coastal habitats throughout the Gulf of Mexico and U.S. Atlantic coast (TEWG 2000, Morreale et al. 2007). Near-shore waters of 120 feet or less provide the primary marine habitat for adults, although it is not uncommon for adults to venture into deeper waters (Byles 1989a, Mysing and Vanselous 1989, Renaud et al. 1996, Shaver et al. 2005, Shaver and Wibbels 2007). The majority of nesting occurs on beaches at Rancho Nuevo; nesting is increasing throughout all of Mexico and Texas.

Diet

The diet of the Kemp's ridley turtle consists mainly of swimming crabs, but may also include

fish, jellyfish, mollusks, and tunicates (Witzell and Schmid 2005).

Diving
Kemp's ridley sea turtles can dive from a few seconds in duration to well over two and a half hours, although most dives range from 16 to 34 minutes (Mendonca and Pritchard 1986, Renaud 1995). Individuals spend the vast majority of their time underwater; over 12-hour periods, 89% to 96% of their time is spent below the surface (Byles 1989b, Gitschlag 1996).

Green sea turtle (*Chelonia mydas*)

Species description and distribution
The green sea turtle is a large, herbivorous, hard-shelled turtle. It has a circumglobal distribution, occurring throughout tropical, subtropical, and, to a lesser extent, temperate waters.

Status
In 1978, the green sea turtle was listed as threatened for all populations with the exception of the Florida and Pacific coast of Mexico breeding populations, which were listed as endangered (43 FR 32800). The International Union for Conservation of Nature (IUCN) has classified the species as endangered.

Population designations, abundance and trends
The species can be divided into two distinct populations: Atlantic and Indo-Pacific. Nesting beaches within each population are connected genetically via male mediated gene flow (Karl et al. 1992, Roberts et al. 2004), but females generally return to their natal beaches to nest, resulting in maternally isolated subpopulations (Bowen et al. 1992). An estimated 100,000 – 150,000 females nest each year at 46 locations. Based historical trends at some of these locations, there appears to an increase in the global population size, but trend data are available for just over half of all sites examined and very few datasets span a full generation (Seminoff 2004). Nesting is stable or increasing at most sites throughout the Atlantic (NMFS & USFWS 1991). Nesting has declined throughout the Pacific, with the exception of Hawaii, where the nesting population has steadily increased in abundance over the past 30 years (Balazs and Chaloupka 2004).

Threats
Habitat loss threatens the species globally. Nesting habitat is destroyed by human development, including: construction of buildings and pilings, beach armoring and renourishment, and sand extraction (Lutcavage et al. 1997, Bouchard et al. 1998). Lights on or adjacent to nesting beaches is often fatal to emerging hatchlings (Witherington and Bjorndal 1991) and alters the behavior of nesting adults (Witherington 1992). Anthropogenic disturbances also threaten coastal marine habitats, particularly areas rich in seagrass and marine algae. These impacts include contamination from herbicides, pesticides, oil spills, and other chemicals, as well as structural degradation from excessive boat anchoring and dredging (Francour et al. 1999, Lee Long et al. 2000, Waycott et al. 2005). Further, the introduction of alien algae species threatens the stability of some coastal ecosystems and may lead to the elimination of preferred dietary species of green sea turtles (De Weede 1996). In the Pacific, where populations are declining, direct take of

turtles and eggs poses the most serious threat (NMFS & USFWS 1998). Many green sea turtles exhibit fibropapillomatosis, a tumor forming disease, which may interfere with movement and foraging (Aguirre and Lutz 2004). Very few green sea turtles are bycaught in U.S. fisheries (Finkbeiner et al. 2011a), but low-level bycatch has been documented in longline fisheries (Petersen et al. 2009). Hatchlings are preyed upon by seabirds and sharks; eggs are preyed upon by dogs, pigs, rats, and crabs (Witzell 1981, Bell et al. 1994). Anthropogenic climate change may further destroy nesting habitat via sea level rise and may cause the widespread failure of nests via increased temperatures (Fuentes et al. 2010).

Life history

Female green sea turtles reach sexual maturity at 20 years of age or greater and reproduce for 17 – 23 years (Chaloupka and Musick 1997, Hirth 1997, Limpus and Chaloupka 1997, Zug and Glor 1998, Seminoff et al. 2002b, Zug et al. 2002, Chaloupka et al. 2004)). They return to their natal beaches to nest every 2 – 5 years (Hirth 1997) to deposit 1 – 7 clutches (usually 2-3) at 12-14 day intervals. Mean clutch size is highly variable among populations, but averages 110-115 eggs/nest (Balazs 1983). Hatchlings emerge from the nest and orient towards a light source, such as light shining off the ocean. Juvenile green sea turtles occupy pelagic, benthic, and coastal foraging habitats until they mature and return to their natal beach for parturition (Carr et al. 1978, Meylan et al. 1990). Green sea turtles exhibit slow growth rates, as a result of their herbivorous diet (Bjorndal 1982). They reach a maximum size just under 100 cm in carapace length (Tanaka 2009).

Migration and movement

Green sea turtles are highly mobile and undertake complex movements through geographically disparate habitats during their lifetimes (Musick and Limpus 1997, Plotkin 2003). Green sea turtles spend the majority of their lives in coastal foraging grounds, including open coastline and protected bays and lagoons. Adult females migrate to their natal beaches to nest.

Diet

Green sea turtles are not obligate herbivores; they also forage on invertebrates especially as juveniles (Godley et al. 1998, Heithaus et al. 2002, Seminoff et al. 2002a, Hatase et al. 2006, Parker and Balazs in press). A shift to a more herbivorous diet occurs when individuals move into neritic habitats, usually at 30 – 62 cm in carapace length (Cardona et al. 2010).

Diving

Based on the behavior of post-hatchlings and juvenile green turtles raised in captivity, we presume that those in pelagic habitats live and feed at or near the ocean surface, and that their dives do not normally exceed several meters in depth (NMFS and USFWS 1998, Hazel et al. 2009). The maximum recorded dive depth for an adult green turtle was just over 106 m (Berkson 1967), while subadults routinely dive to 20 m for 9-23 min, with a maximum recorded dive of over 1 h (Brill et al. 1995, I-Jiunn 2009).

Critical habitat

In 1998, critical habitat for green sea turtles was designated in coastal waters surrounding Culebra Island, Puerto Rico (63 FR 46693). Aspects of these areas that are important for green

sea turtle survival and recovery include important natal development habitat, refuge from predation, shelter between foraging periods, and food for green sea turtle prey.

Hawksbill sea turtle (*Eretmochelys imbricata*)

Species description and distribution
The hawksbill sea turtle has a sharp, curved beak-like mouth, which it uses to grasp sponges, its primary prey. It has a circumglobal distribution throughout tropical and, to a lesser extent, subtropical oceans.

Status
In 1970, the hawksbill sea turtle received protection under the Endangered Species Conservation Act (35 FR 8495). In 1973, it was listed as endangered under the ESA (35 FR 8491). The International Union for Conservation of Nature (IUCN) has classified the species as critically endangered (Mortimer and Donnelly 2008).

Population designations, abundance and trends
Surveys at 83 nesting sites worldwide indicate 21,212 – 28,138 females nest annually (NMFS and USFWS 2007). Long-term data exists for 58 sites for with historic trends, all of which indicate a decline over the past 20 to 100 years. Among 42 sites for which recent trend data are available, 10 (24 percent) are increasing, three (seven percent) are stable, and 29 (69 percent) are decreasing. Populations are distinguished generally by ocean basin (i.e., Atlantic, Pacific, and Indian) and more specifically by nesting location.

Threats
The greatest threats to hawksbill sea turtles are overharvesting of turtles and eggs, degradation of habitat, and fisheries interactions. Hawksbills are harvested for their meat and carapace, which is sold as tortoiseshell (Mortimer and Donnelly 2008). Eggs are taken at high levels, especially in Southeast Asia where collection approaches 100 percent in some areas (Mortimer and Donnelly 2008). Because hawksbills prefer to nest under vegetation (Mortimer 1982, Horrocks and Scott 1991), they are particularly impacted by beachfront development and clearing of dune vegetation (Mortimer and Donnelly in review). Lights on or adjacent to nesting beaches is often fatal to emerging hatchlings (Witherington and Bjorndal 1991) and alters the behavior of nesting adults (Witherington 1992). In addition to impacting the terrestrial zone, anthropogenic disturbances also threaten coastal marine habitats. These impacts include contamination from herbicides, pesticides, oil spills, and other chemicals, as well as structural degradation from excessive boat anchoring and dredging (Francour et al. 1999, Lee Long et al. 2000, Waycott et al. 2005). Finkbeiner et al. (2011a) estimated that annual bycatch interactions total at least 20 individuals annually for U.S. Atlantic fisheries (resulting in less than ten mortalities) and no or very few interactions in U.S. Pacific fisheries. Hatchlings are preyed upon by seabirds and sharks. Eggs are preyed upon by dogs, pigs, rats, and crabs (Ficetola 2008). Anthropogenic climate change may further destroy nesting habitat via sea level rise, but increased temperatures may help the species by skewing the hatchling sex ratio toward female (Wibbels 2003).

Life history
The best estimate of age at sexual maturity for hawksbill sea turtles is about 20-40 years (Chaloupka and Limpus 1997, Crouse 1999a). Reproductive females undertake periodic (usually non-annual) migrations to their natal beach to nest. Movements of reproductive males are less well known, but are presumed to involve migrations to their nesting beach or to courtship stations along the migratory corridor (Meylan 1999). Females nest an average of 3 – 5 times per season (Meylan and Donnelly 1999, Richardson et al. 1999). Clutch size is larger on average (up to 250 eggs) than that of other sea turtles (Hirth 1980). Hatchlings migrate to and remain in pelagic habitats until they reach approximately 22-25 cm in straight carapace length (Meylan 1988, Meylan and Donnelly 1999). As juveniles, they take up residency in coastal waters to forage and grow.

Habitat
Hawksbill sea turtles are highly migratory and use a wide range habitats during their lifetimes (Musick and Limpus 1997, Plotkin 2003). Small juvenile hawksbills (5-21 cm straight carapace length) have been found in association with *Sargassum* spp. in both the Atlantic and Pacific Oceans (Musick and Limpus 1997) and observations of newly hatched hawksbills attracted to floating weed have been made (Hornell 1927, Mellgren et al. 1994, Mellgren and Mann 1996). Juvenile hawksbills may occupy a range of habitats that include coral reefs or other hard-bottom habitats, sea grass, algal beds, mangrove bays and creeks (Musick and Limpus 1997, Bjorndal and Bolten 2010), and mud flats (R. von Brandis, unpublished data in NMFS and USFWS 2007). As adults, hawksbills are typically associated with coral reefs, which are among the world's most endangered marine ecosystems (Wilkinson 2000). Individuals of multiple breeding locations can occupy the same foraging habitat (Bowen et al. 1996, Bass 1999, Diaz-Fernandez et al. 1999, Bowen et al. 2007, Velez-Zuazo et al. 2008). Nesting sites appear to be related to beaches with relatively high exposure to wind or wind-generated waves (Santana Garcon et al. 2010).

Diet
As adults, hawksbills feed on sponges and corals (Meylan 1988, Leon and Bjorndal 2002). Data from oceanic stage hawksbills are limited, but indicate a diet of plant and animal material (Bjorndal 1997).

Diving
Hawksbill diving ability increases with age and body size (Blumenthal et al. 2009). Hawksbills have long dive durations, although dive depths are not particularly deep. Adult females along St. Croix reportedly have average dive times of 56 min, with a maximum time of 73.5 min (Starbird et al. 1999). Average day and night dive times were 34–65 and 42–74 min, respectively. Immature individuals have much shorter dives of 8.6–14 min to a mean depth of 4.7 m while foraging (Van Dam and Diez 1997).

Critical habitat
On September 2, 1998, NMFS established critical habitat for hawksbill sea turtles around Mona and Monito Islands, Puerto Rico (63 FR 46693). Aspects of these areas that are important for hawksbill sea turtle survival and recovery include important natal development habitat, refuge from predation, shelter between foraging periods, and food for hawksbill sea turtle prey.

Leatherback sea turtle (*Dermochelys coriacea*)

Species description and distribution
The leatherback is the largest of all sea turtles (up to 916 kg; Eckert & Luginbuhl 1988) and the only species lacking a hard, bony carapace. Its slightly flexible carapace is made primarily of tough, oil saturated connective tissue. The species has an extensive distribution, ranging from tropical to subpolar latitudes. This unique reptile is able to withstand broad temperature extremes because of its large body size (Paladino et al. 1990), thick peripheral insulation (Goff and Stenson 1988), counter-current heat exchange (Greer et al. 1973), and thermoregulatory behavior (Bostrom et al. 2010).
Status
The leatherback sea turtle has been listed as endangered under the ESA since 1973 and, prior to that, under the Endangered Species Conservation Act (35 FR 8491). The species is considered to be critically endangered worldwide (Sarti Martinez 2000).

Population designations, abundance, and trends
The global population of adult females has declined over 70 percent in less than one generation, from an estimated 115,000 adult females in 1980 to 34,500 adult females in 1995 (Pritchard 1982, Spotila et al. 1996), driven by dramatic reductions in several Pacific populations (Sarti Martinez 2000). Nesting aggregations occur in six broad geographic regions: eastern Atlantic, western Atlantic, eastern Pacific, western Pacific, and Indian. Genetic studies indicate the reproductive isolation of these designations, which are distinguished by the presence of unique mitochondrial DNA haplotypes or significant differences in haplotype frequencies (Dutton et al. 1999, Dutton et al. 2007).

Eastern Atlantic. In 2007, the Turtle Expert Working Group provided a population estimate of 34,000-94,000 adult leatherbacks in the North Atlantic Ocean (TEWG 2007), including both eastern and western Atlantic stocks. Based on genetic and tagging data, there are at least two stocks (eastern and western Atlantic) and possibly as many as seven (TEWG 2007). Nesting occurs in the eastern Atlantic, from Mauritania to Angola (Fretey et al. 2007). Gabon hosts the world's largest population of leatherbacks, estimated at 15,730- 41,373 females (Witt et al. 2009). Population dynamics are relatively stable in the Atlantic, but estimates fluctuate considerably due to individual variance in remigration intervals, clutch number, and inconsistent nest site fidelity (TEWG 2007).

Western Atlantic. In 2007, the Turtle Expert Working Group provided a population estimate of 34,000-94,000 adult leatherbacks in the North Atlantic Ocean (TEWG 2007), including both eastern and western Atlantic stocks. In the western Atlantic and Caribbean Sea, nesting occurs on beaches in Puerto Rico, St. Croix, Costa Rica, Panama, Colombia, Trinidad and Tobago, Guyana, Sao Tome and Principe, French Guiana, Suriname, and Florida (Márquez 1990, Spotila et al. 1996, Bräutigam and Eckert 2006). Population dynamics are relatively stable in the Atlantic, but estimates fluctuate considerably due to individual variance in remigration intervals, clutch number, and inconsistent nest site fidelity (TEWG 2007).

Eastern Pacific. In the eastern Pacific, the estimated number of adult females declined by from

4638 to 1690 (64 percent) between the years of 1995 and 2000 (Spotila et al. 1996, Spotila et al. 2000). The largest nesting aggregations of this region occur on the beaches of Mexico and Costa Rica. There has been a steady decline in the number of females observed at the four major nesting sites in Mexico: Mexiquillo, Tierra Colorada, Cahuitan, and Barra de la Cruz (Sarti Martinez et al. 2007). At Las Baulas National Park, the largest leatherback rookery in Costa Rica, the nesting population declined 83 percent (from 1367 to 231 adult females) from 1988 to 1999 (Fig.7; Reina et al. 2002).

Western Pacific. The western Pacific region supports an estimated 2100-5700 breeding females (Dutton et al. 2007). The largest leatherback rookeries occur at the Jamursba-Medi and Wermon beaches of Papua, Indonesia (Hitipeuw et al. 2007). There are also significant nesting aggregations on beaches throughout Papua New Guinea, Vanuatu, and the Solomon Islands (Benson et al. 2007, Dutton et al. 2007, Hitipeuw et al. 2007). It is difficult to describe leatherback abundance trends in the western Pacific because new nesting aggregations are continually being found, and historical data is limited to one location. At Jamursba-Medi, an estimated 300-900 females nest annually (Hitipeuw et al. 2007). Though these estimates are similar to those reported in the 1990s (Spotila et al. 1996, Dutton et al. 2007), they are lower than those from the early 1980s (Fig. 7; Hitipeuw et al. 2007). We do not include the Terengganu, Malaysia rookery here because it is genetically differentiated from western Pacific populations (Dutton et al. 2007), suggesting long-term reproductive isolation possibly as a result of previous barriers to gene flow (e.g., the Sunda Shelf).

Indian. The once large rookery at Terengganu, Malaysia is now functionally extinct. Beaches that once supported over 3000 females nesting annually, now host 2 or 3 females per year, representing a 99 percent decline since 1950 (Chan et al. 1988, Chan and Liew 1989). At present, the largest nesting aggregations in the Indian Ocean occur on Andaman and Nicobar Islands (India), where 400-500 females nest annually (Andrews and Shanker 2002). Significant nesting aggregations also occur in Sri Lanka, South Africa, and Mozambique (Hamann et al. 2006). We tentatively group all Indian Ocean rookeries, but no genetic or tagging data are available to assess the stock structure of these populations.

Threats

Leatherbacks face a multitude of threats. Bycatch, particularly by longline fisheries, is a major source of mortality for leatherback sea turtles (Crognale et al. 2008, Gless et al. 2008, Fossette et al. 2009, Petersen et al. 2009). Harvest of females along nesting beaches is of concern worldwide. Egg collection is widespread and has contributed to catastrophic declines, such as in Malaysia. There is increasing development and tourism along nesting beaches (Maison 2006, Hernandez et al. 2007, Santidrián Tomillo et al. 2007). Structural impacts to beaches include: building and piling construction, beach armoring and renourishment, and sand extraction (Lutcavage et al. 1997, Bouchard et al. 1998). In some areas, timber and marine debris accumulation as well as sand mining reduce available nesting habitat (Chacón Chaverri 1999, Formia et al. 2003, Laurance et al. 2008, Bourgeois et al. 2009). Lights on or adjacent to nesting beaches alter nesting adult behavior and are often fatal to emerging hatchlings as they are drawn to light sources and away from the sea (Witherington and Bjorndal 1991, Witherington 1992, Cowan et al. 2002, Deem et al. 2007, Bourgeois et al. 2009). Plastic ingestion is very common

in leatherbacks and can block gastrointestinal tracts leading to death (Mrosovsky et al. 2009). Although global climate change may expand foraging habitats into higher latitude waters, increasing temperatures may increase feminization of nests (Mrosovsky et al. 1984, James et al. 2006, McMahon and Hays 2006, Hawkes et al. 2007b). It may also result in rising sea levels, which may inundate nests on some beaches. Natural threats include: predation on adults by sharks and killer whales (Pitman and Dutton 2004); predation on hatchlings by seabirds, land predators, and sharks; and tidal inundation of nests (Caut et al. 2009).

Life history

Both males and females exhibit some degree of philopatry, returning to their natal beaches for breeding and nesting (James et al. 2005b). Age to maturity remains elusive, with estimates ranging from 5 to 29 years (Spotila et al. 1996, Avens et al. 2009). Females lay up to seven clutches per season, with more than 65 eggs per clutch and eggs weighing >80 g (Reina et al. 2002, Wallace et al. 2007). Females remigrate every 1-7 years, with most turtles returning every two years to nest in French Guiana (Rivalan et al. 2005) and every three years in Las Baulas, Costa Rica (Reina et al. 2002). The remigration interval for western Pacific leatherbacks is unknown but estimated as "several years" (Benson et al. 2011). According to Wallace (2007), high seasonal and lifetime fecundity likely reflect compensation for high and unpredictable mortality during early life history stages.

Migration and movement

Leatherback sea turtles migrate long distances between tropical nesting beaches temperate forage areas (i.e., remigration). Leatherbacks weigh ~33 percent more on their foraging grounds than at nesting, indicating that they probably catabolize fat reserves to fuel migration and subsequent reproduction (James et al. 2005a, Wallace et al. 2006). Sea turtles must meet an energy threshold before returning to nesting beaches (Rivalan et al. 2005, Sherrill-Mix and James 2008, Casey et al. 2010). Therefore, their remigration intervals (the time between breeding seasons) are dependent upon foraging success and duration (Hays 2000, Price et al. 2004). Eastern Pacific leatherbacks use a consistent migration corridor to forage in the South Pacific Gyre, a low-productivity region (Shillinger et al. 2008). Western Pacific turtles that nest during the boreal summer migrate to forage in temperate waters of the North Pacific or tropical waters of the South China Sea; those that nest during the boreal winter forage in temperate and tropical waters of the southern hemisphere (Benson et al. 2011). Post-nesting, western Atlantic leatherbacks migrate to foraging areas in the North Atlantic or the equatorial eastern Atlantic (Ferraroli et al. 2004, Hays et al. 2004, Eckert et al. 2006, Eckert 2006). Eastern Atlantic leatherbacks migrate to foraging areas in the equatorial Atlantic, temperate waters off South America, or temperate waters off southern Africa (Witt et al. 2011).

Diet

Leatherbacks feed primarily on jellyfish, such as Stomolophus, Chryaora, and Aurelia (Rebel 1974), and tunicates (salps, pyrosomas). These gelatinous zooplankton are relatively nutrient-poor (Doyle et al. 2007), such that leatherbacks must consume large quantities to support their body weight.

Diving
Leatherbacks are deep divers with a maximum-recorded depth of over 4000 m (Eckert et al. 1989, López-Mendilaharsu et al. 2009). Dives are typically 50-84 m (Standora et al. 1984) and last 1-14 min (Eckert et al. 1989, Eckert et al. 1996, Harvey et al. 2006b, López-Mendilaharsu et al. 2009). The depth of their dives often corresponds with the vertical distribution of their prey (Hodge and Wing 2000, Harvey et al. 2006b).
Habitat
Juveniles leatherbacks are restricted to tropical waters (≥26 °C); after they exceed 100 cm in carapace length, they are able to move into the temperate waters that comprise their primary foraging habitat (Eckert 2002). Adults migrate to areas of high prey density, often concentrated by oceanographic features such as frontal systems, eddy features, current boundaries, and coastal retention areas (Collard 1990, Davenport and Balazs 1991, Frazier 2001, HDLNR 2002, Benson et al. 2011). Leatherbacks off the western United States are more likely to occur in continental slope waters than shelf waters (Green et al. 1992, Carretta and Forney 1993, Green et al. 1993, Bowlby et al. 1994). Leatherbacks also require sandy beach habitat for nesting. In the southwest Pacific, females appear to favor nesting sites with higher wind and wave exposure, possibly as a means to aid hatchling dispersal (Garcon et al. 2010).

Critical Habitat
In 1979, leatherback critical habitat was identified adjacent to Sandy Point, St. Croix, Virgin Islands from the 183 m isobath to mean high tide level between 17° 42’12” N and 65°50’00” W (44 FR 17710). This habitat is essential for nesting, which has been increasingly threatened since 1979, when tourism increased significantly, bringing nesting habitat and people into close and frequent proximity. However, studies do not support significant critical habitat deterioration. On January 20, 2012, NMFS issued a final rule to designate additional critical habitat for the leatherback sea turtle (50 CFR Part 226). This designation includes approximately 43,798 km^2 stretching along the California coast from Point Arena to Point Arguello east of the 3,000 m depth contour; and 64,760 km^2 stretching from Cape Flattery, Washington to Cape Blanco, Oregon east of the 2,000 m depth contour (Fig. 5). The designated areas comprise approximately 108,558 km^2 of marine habitat and include waters from the ocean surface down to a maximum depth of 80 m.

Marine Mammals

Cook Inlet Beluga Whale
Beluga whales are widely distributed in Arctic and subarctic waters. In Alaska, five putative populations exist (Beaufort Sea, eastern Chukchi Sea, Bristol Bay, eastern Bering Sea and Cook Inlet) (Angliss et al. 2001). Cook Inlet beluga whales are the only population that is listed under the ESA. Mitochondrial and nuclear DNA distinguish Alaskan beluga whales from those that occur in Hudson Strait, Baffin Bay and the St. Lawrence River, with the Cook Inlet population demonstrating strong evidence of genetic isolation from the other Alaskan populations (O'Corry-Crowe et al. 2007, O'Corry-Crowe 2008).

Based on past studies of the summer distribution of beluga whales in Cook Inlet, it appears that

the population has experienced a contraction in its overall distribution (Speckman and Piatt. 2000, Hobbs et al. 2008, Rugh et al. 2010). According to Hobbs *et al.,* (2008) 90% of the whales in the 1970s were observed within 70 nmi of the western tip of Anchorage (Point Woronzof), whereas more recently (1998-2007) 90% were detected within 20 nmi. Although the precise reason for the range contraction is not known, the shrinking summer distribution likely reflects the reduction in the population size over the same intervals and the beluga whale's preference for dense aggregations of preferred prey species.

Analyses of beluga whale stomach contents indicate that beluga whales are opportunistic feeders, but specific species form the bulk of the prey when they are seasonally abundant (Hobbs et al. 2008). For instance, eulachon (*Thaleichthys pacificus*) also known as smelt or candlefish, are a small anadromous fish return that their natal rivers in spring for spawning. The high fat content of this species confers a significant source of energy for beluga whales, including calving whales that occur in the upper inlet during the same period (Calkins 1989). Based on stomach sample analyses from 2002-2007 fish compose the majority of the prey species, with gadids (cod and walleye pollock) and salmonids composing the majority of the fish eaten (Hobbs et al. 2008). Anadromous salmonids begin concentrating at the river mouths and intertidal flats in upper Cook Inlet in late spring and early summer as emigrating smolts and immigrating adult spawners. Like eulachon, salmon are another source of lipid-rich prey for the beluga whale and represent the greatest percent frequency of occurrence of the prey species found in Cook Inlet beluga whale stomachs (Hobbs et al. 2008). As salmonid numbers dwindle in the fall and winter, beluga whales return to feed on nearshore or deeper water species including cod, sculpin, flounder, sole, shrimp, crab and others (Hobbs et al. 2008).

Beluga whale calving is not well documented but the presence of cow/calf pairs in large river estuaries in the upper inlet, and accounts of Alaskan Natives, suggests that calving and nursery areas are located near the mouths of the Beluga and Susitna Rivers, Chickaloon Bay and Turnagain Arm (see NMFS 2008b). Recent surveys suggest that cow/calf pairs also make extensive use of Knik Arm in the summer and fall (Funk et al. 2005 as cited in NMFS 2008b). Neonates are often not seen until June in Cook Inlet (Burns and Seaman 1986). Some researchers have suggested that the shallow waters of Cook Inlet may be important for reproduction and calving, as the shallower water is warmer which may confer an important thermal advantage for calf survival as they have relatively limited fat deposits at birth (see NMFS 2008b). A beluga female's first parturition is at age five or six. Breeding is presumed to occur shortly after calving, in the late summer after about 14-15 months of gestation (Calkins 1989). Lactation lasts about two years, with breeding occurring during lactation (Calkins 1989).

Status and Trends

On October 22, 2008, NMFS listed the Cook Inlet beluga whale as endangered (73 FR 62919). Historic numbers of beluga whales in Cook Inlet are unknown. Dedicated surveys began in earnest in the 1990s when NMFS began conducting aerial surveys for beluga whales in Cook Inlet. Prior to then, survey efforts were inconsistent, part of larger sea bird and marine mammal surveys, made by vessel, or estimated following interviews with fishermen (Klinkhart 1966). In many cases the survey methodology or confidence intervals were not described. For instance, Klinkhart (1966) conducted aerial surveys in 1964 and 1965, where he describes having

estimated the populations at 300-400 whales, but the methodology was not described nor did he report the variance around these estimates. Other estimates were incomplete due to the small area the survey focused upon (e.g. river mouth estimates; e.g., Hazard 1988). The most comprehensive survey effort prior to the 1990s occurred in 1979 and included transects from Anchorage to Homer, and covered the upper, middle and lower portions of Cook Inlet. From this effort, and using a correction factor of 2.7 to account for submerged whales Calkins (1989) estimated the 1979 abundance at about 1,293 whales.

In 1993, NMFS began systematic aerial surveys of beluga whales in Cook Inlet and like the 1979 survey cover the upper, middle and lower portions of Cook Inlet. The survey protocol involves using paired observers who make independent counts at the same time a video of the whale grouping is recorded. Each group size estimate is corrected for subsurface and missed animals, or if video counts are not available then additional corrections are made (Allen and Angliss 2010). Between 1979 and 1994, according to above noted population estimates, Cook Inlet beluga whales declined by 50%, with another 50% decline observed between 1994 and 1998. Using a growth fitted model Hobbs *et al.,* (2008) observed an average annual rate of decline of -2.91% (SE = 0.010) from 1994 to 2008, and a -15.1% (SE = 0.047) between 1994 and 1998. A comparison with the 1999-2008 data suggests the rate of decline at -1.45% (SE=0.014) per year (Hobbs *et al.,* 2008). Given that harvest was curtailed significantly between 1999 and 2008, NMFS had expected the population would begin to recover at a rate of 2-6% per year. However, abundance estimates demonstrate that this is not the case (Hobbs and Shelden 2008).
In conducting its status review, NMFS ran a number of population viability analyses (PVAs) to estimate the time to extinction for Cook Inlet beluga whales. The models were sensitive to a variety of parameters such as killer whale predation, allee effects and unusual mortality events. The best approximation of the current population incorporated killer whale predation at only one beluga whale per year and allowed for an unusual mortality event occurring on average every 20 years. According to this model, there is an 80% probability that the population is declining, a 26% probability that the population will be extinct in 100 years (by 2108) and a 70% probability that the population will be extinct within 300 years (by 2308).

Threats
Natural Threats. Natural threats to Cook Inlet beluga whales include stranding, predation, parasitism and disease, environmental change and genetic risks associated with small populations (e.g., inbreeding, loss of genetic variability). As noted in NMFS' Cook Inlet beluga whale conservation plan (NMFS 2008b),beluga whales may strand accidentally as they occupy shallow water areas or escape predators, or as a result of diseases, illness or injury. Given the extreme tidal fluctuations in Cook Inlet, beluga whale strandings are not uncommon. According to NMFS (2008b) killer whales have been observed in Cook Inlet concurrent to beluga whale strandings and evidence of killer whale attacks is apparent in some beluga whale strandings.

Over 700 beluga whales have stranded in Cook Inlet since 1988, many of which occurred in Turnagain Arm and often coincided with extreme tidal fluctuations (NMFS 2008b). Where stranding occurs from extreme tidal fluctuations and animals are out of the water for extended periods the risk of mortality increases from cardiovascular collapse. Ten hours may be the upper limit for out of the water for beluga whales before serious injury or death occurs (NMFS 2008b).

Strandings may represent a significant threat to the conservation and recovery of the Cook Inlet beluga whale population.

Gaydos *et al.,* (2004a) identified 16 infectious agents in free-ranging and captive southern resident killer whales, but concluded that none of these pathogens were known to have high potential to cause epizootics. Many of these same infectious agents could pose a problem for Cook Inlet beluga whales. At this time little information is available to date to suggest bacterial or viral agents are actively contributing to the decline in the Cook Inlet population. About 80% of Cook Inlet beluga whales examined, however, have evidence of the parasite *Crassicauda giliakiana* in the kidneys, although it is presently unclear whether the parasite is affecting the status of the population (NMFS 2008b). Necropsies have also revealed infestations of the common nematode anasakids, or whaleworm in the stomach of adult Cook Inlet beluga whales. While the parasite tends to favor the stomach and can cause gastritis or ulcerations, the infestations in beluga whales has not been considered severe enough to have caused clinical responses (NMFS 2008b). Liver trematodes have also been identified in at least one beluga whale. At present, NMFS has no information to suggest that parasites are having a measureable impact on the survival and health of the Cook Inlet whale population (NMFS 2008b).

Anthropogenic Threats. Human induced threats to Cook Inlet beluga whales include subsistence harvest, poaching and illegal harvest, incidental take during commercial fishing and reduction of prey through fishing harvests, pollution, oil and gas development, urban development, vessel traffic including from tourism and whale watching, noise, as well as research activities directed at beluga whales. During the early 1900s there was a short-lived commercial whaling company, The Beluga Whaling Company, which operated at the Beluga River in upper Cook Inlet. The Company during its 5 years of operation harvest 151 belugas from 1917-1921 (Mahoney and Shelden 2000). Another commercial hunt of beluga whales in 1930s is recollected by residents, but no record of the hunt exists in Alaska fishery and fur seal documents (as cited in Mahoney and Shelden 2000). In 1999 and 2000 there was a voluntary moratorium on subsistence harvest. Thereafter, subsistence harvests have been conducted under co-management agreements. Since 2000, no more than 2 beluga whales have been taken in subsistence harvests in any one year (NMFS 2008b).

Commercial fisheries likely have varying levels of interactions with Cook Inlet beluga whales, according to the timing, gear types, targeted species and location of activities (NMFS 2008b). Reports of fatal interactions with commercial fisheries have been noted in the literature (Hobbs et al. 2008). Direct interactions with fishing vessels and nets are considered unusual, based on observer data and unlikely to inhibit the recovery of Cook Inlet beluga whales. The reduction of prey species, however, is of more concern for the species. In 2000 NMFS recommended the closing of the eulachon fishery due to a lack of understanding of how this fishery interfered with beluga whale feeding, but in 2005 this fishery was reopened with a harvest limited at 100 tons of eulachon. Currently, it is unclear if fishery harvest of prey species is having a significant impact on the beluga whale population. Impacts from recreational fisheries, which are very popular in the region, likely include the reduction of fish prey species particularly salmonid species and also the harassment from noise and risk of injury from vessel strikes from the operation of small watercraft in the estuarine/river mouths (NMFS 2008b).

Contaminants in beluga whales are of concern for both whale health and the health of subsistence users. Tissue samples are regularly collected from subsistence harvested and stranded beluga whales and archived. Tissues and organs commonly collected include blubber, liver and kidneys, as well as muscle, heart, bone, skin and brain. Blubber is the most commonly collected; due to the lipid content it typically contains the most lipophilic substances (Becker 2000). The kidney and liver are used to analyze heavy metal compounds. Relatively high levels of PCBs, chlorinated pesticides and mercury are evident in beluga whales, although the more contaminated belugas are from the St. Lawrence River, Canada (Becker 2000). Concentrations of chlorinated hydrocarbons in Cook Inlet beluga whales range from 0.1-2.4 µg/g, w.w. DDT, 0.6-4.7 µg/g, w.w. PCB, 0.1-0.6 µg/g, w.w. chlordane, <0.1-4.3 µg/g, w.w. toxaphene. Studies indicate that PCBs and chlorinated pesticide concentrations are higher in male beluga whales than females, reflecting the transference of body loads to the offspring that occurs during gestation and lactation (Becker et al. 2000). Other contaminants detected in Cook Inlet beluga whales include heavy metals such as cadmium, mercury, selenium, copper and zinc. Comparative studies suggest that Cook Inlet beluga whales generally carry less contaminant body burdens than beluga whales from other areas. An exception is copper, which is two to three times higher in Cook Inlet beluga whales than beluga whales from the eastern Beaufort Sea and the eastern Chukchi Sea, but is similar concentrations found in Hudson Bay beluga whales (Becker et al. 2000).

Critical Habitat

On April 11, 2011 NMFS designated critical habitat for the Cook Inlet beluga whale 76 FR 20180. Two specific areas are designated comprising 7,800 square kilometers of marine habitat. Area one encompasses all marine waters of Cook Inlet north of a line from the mouth of Threemile Creek (61°08.5′ N., 151°04.4′ W.) connecting to Point Possession (61°02.1′ N., 150°24.3′ W.), including waters of the Susitna River south of 61°20.0′ N., the Little Susitna River south of 61°18.0′ N. and the Chickaloon River north of 60°53.0′ N. (2) Area two encompasses all marine waters of Cook Inlet south of a line from the mouth of Threemile Creek (61°08.5′ N., 151°04.4′ W.) to Point Possession (61°02.1′ N., 150°24.3′ W.) and north of 60°15.0′N., including waters within two nautical miles seaward of the mean high water boundary along the western shoreline of Cook Inlet between 60°15.0′ N. and the mouth of the Douglas River (59°04.0′ N., 153°46.0′ W.); all waters of Kachemak Bay east of 151°40.0′ W.; and waters of the Kenai River below the Warren Ames bridge at Kenai, Alaska.

Area 1 has the highest concentration of beluga whales in the spring through fall as well as the greatest potential for adverse impact from anthropogenic threats. It contains many rivers with large eulachon and salmon runs, including two rivers in Turnagain Arm (Twenty-mile River and Placer River) which are visited by beluga whales in the early spring. Use declines in the summer and increases again in August through the fall, coinciding with coho salmon returns. Also included in Area 1 are Knik Arm and the Susitna delta. Area 2 is located south of Area 1 and is used by Cook Inlet beluga whales for fall and winter feeding and as transit waters.

The primary constituent elements essential to the conservation of Cook Inlet beluga whales are: (1) Intertidal and subtidal waters of Cook Inlet with depths <30 ft. (MLLW) and within 5 miles of high and medium flow accumulation anadromous fish streams; (2) Primary prey species consisting of four species of Pacific salmon (Chinook, coho, sockeye and chum salmon), Pacific

eulachon, Pacific cod, walleye pollock, saffron cod and yellowfin sole; (3) Waters free of toxins or other harmful agents; (4) Unrestricted passage within or between the critical habitat areas; and (5) An absence of in-water noise levels that result in the abandonment of habitat by Cook Inlet beluga whales.

Southern Resident Killer Whale

The SRKW has been listed as endangered under the ESA since November 18, 2005 (70 FR 69903); critical habitat for this species was designated on November 29, 2006 (71 FR 69054). In April 2004, the Washington Department of Fish and Wildlife (WDFW) designated killer whales in Washington State as a "State endangered species" (WAC 232-12-297). SRKWs are also protected by the MMPA and Canada's Species at Risk Act (SARA).
The Southern Resident killer whale (*Orcinus orca*) is a toothed whale and is the largest member of the dolphin family. Based on genetic research, it is believed that multiple subspecies of killer whales exist worldwide (Krahn et al. 2002, Waples and Clapham 2004, Jefferson et al. 2008). Resident killer whales in the Northeast Pacific are distributed from Alaska to California, with four distinct communities recognized: southern, northern, southern and western Alaska (Krahn et al. 2002, Krahn et al. 2004). The SRKW occurs in the northeastern Pacific Ocean along the west coasts of the U.S. and Canada. Resident whales exhibit advanced vocal communication and live in highly stable social matriarchal groupings called pods. They frequent a variety of marine habitats and their range does not appear to be limited by depth, temperature or salinity (Baird 2000).

The SRKW species consists of three pods, designated J, K and L, that reside for part of the year in the inland waterways of Washington State and British Columbia (Strait of Georgia, Strait of Juan de Fuca and Puget Sound), principally during the late spring, summer and fall (Bigg 1982, Ford et al. 2000, Krahn et al. 2002). Pods have visited coastal sites off Washington and Vancouver Island (Ford et al. 2000) and are known to travel as far south as central California and as far north as the Queen Charlotte Islands off British Columbia. The locations of SRKWs in the late fall, winter and early spring are less well known.

Parsons (2009) noted that members of different pods interact, but members generally remain within their matrilinear group. Interaction between pods has increased over the past two decades and may be the result of a common response among pods to the stress of a declining population (Parsons et al. 2009). The rate of intrapod interaction was lowest within L pod, which is the largest of the SRKW pods (Parsons et al. 2009).

Male SRKWs become sexually mature at a mean age of approximately 15 years and are thought to remain sexually active throughout their adult lives (Christensen et al. 1984, Perrin and Reilly 1984, Duffield and Miller 1988, Olesiuk et al. 1990). Females first give birth at a mean age of approximately 14.9 years and produce an average of approximately 5.4 surviving calves over a reproductive life span of about 25 years (Olesiuk et al. 1990, Matkin et al. 2003). Gestation periods, as observed in captive killer whales, average around 17 months (Asper et al. 1988, Walker et al. 1988, Duffield et al. 1995). The mean interval between viable calve births is four years (Bain 1990). Older mothers tend to have greater calving success, and they appear to be

assisted in calf rearing by grandmothers (Ward et al. 2009). Some females may reach 90 years of age (Olesiuk et al. 1990). Mothers and offspring maintain highly-stable, lifelong social bonds and this relationship appears to be the basis for their matrilinear social structure (Bigg et al. 1990, Baird 2000, Ford et al. 2000).

Although mating can occur year-round, most killer whale reproduction in the North Pacific has been observed to occur primarily from April to October (Olesiuk et al. 1990, Matkin et al. 1997), with a peak in calving occurring between September and March (Olesiuk et al. 2005, Jefferson et al. 2008). Killer whales are polygamous (Dahlheim and Heyning 1999) and genetic data indicate that resident males mate with females outside of their own pods almost exclusively. This reduces the chances of inbreeding (Barrett-Lennard 2000, Barrett-Lennard and Ellis 2001).
Killer whales are apex predators and consume a varied diet but fish are their preferred prey (Scheffer and Slipp 1948, Ford et al. 1998, Ford et al. 2000, Saulitis et al. 2000). Although the record is incomplete, data suggest that SRKWs have a strong preference for Chinook salmon during late spring to fall (Hanson et al. 2005, Ford and Ellis 2006). Their winter and early spring diet is largely unknown. SRKWs spend about half of their time hunting prey. Approximately 95% of their time spent underwater is at depths of less than 30 m (Baird 2000, Baird et al. 2003, Baird et al. 2005). They detect prey via echolocation and passive listening and likely hunt through a combination of vision and echolocation (Barrett-Lennard et al. 1996, Baird 2000). Maximum observed dive depths average 141 m (Baird et al. 2003). Baird *et al.,* (2005) reported that although the deepest recorded dive for a SRKW is 264 m, they are probably capable of diving to at least 330 m. No significant differences in the diving behavior of the three Southern Resident pods has been observed (Baird et al. 2005).

Status and Trends
The only pre-1974 account of Southern Resident abundance is from Sheffer and Slipp (1948) and merely notes that the species was “frequently seen” during the 1940s in the Strait of Juan de Fuca, northern Puget Sound, and off the coast of the Olympic Peninsula, with smaller numbers along Washington’s outer coast. Little information exists on the historic abundance of SRKWs. Until the mid- to late-1800s, the SRKW community may have numbered more than 200 animals (Krahn et al. 2002). Using the estimated abundance of SRKWs in 1971 of 67 whales and factoring in various sources of mortality, NMFS estimated a minimum historical abundance of about 140 SRKWs (Olesiuk et al. 1990). The SRKW population had grown to 90 whales by September 2006, but declined in 2007 with the loss of five individuals and the gain of two new calves leaving the total number at 87, with 25 whales in J pod, 19 whales in K pod and 43 whales in L pod (Center for Whale Research, unpublished data cited in NMFS 2008f). At present, the Southern Resident population has declined to essentially the same size that was estimated during the early 1960s, when it was considered to be depleted (Olesiuk et al. 1990).
Photo-identification catalogs for SRKWs provide information on recent abundance and trends of these pods (see Dahlheim 1997, Dahlheim et al. 1997, Ford and Ellis 1999, Matkin et al. 1999). From 1974–2007, the SRKWs as a whole have gone through several periods of growth and decline. For example, the species appeared to experience a period of recovery by increasing to 99 whales in 1995, but then declined by 20% to 79 whales in 2001 before another slight increase to 83 whales in 2003 (Ford et al. 2000, Carretta et al. 2005). This abrupt decline and unstable population status continue to be cause for concern, particularly given the small size of the species

which makes it potentially vulnerable to Allee effects (e.g., inbreeding depression) that could cause further population decline or preclude a substantial increase in abundance (see NMFS 2008f). The intensity of factors affecting the species is increased by stochastic events such as the small number of reproductive age males and high mortality rates for this group and is a major reason that the SRKW was listed as endangered rather than threatened (NMFS 2008f).

Using data from 1974–2003, Krahn *et al.,* (2002, 2004) further analyzed the population dynamics of the species to identify demographic factors contributing to the latest decline in abundance. Changes in survival were not related to stochastic variation caused by the SRKW community's small size, such as random patterns in births or deaths or to annual fluctuations in survival. Rather, the survival patterns were more likely influenced by external causes, such as changes in prey availability etc.

Threats

Natural Threats. The population of SRKWs has declined recently. The recent decline and unstable population structure make it difficult for SRKWs to recover from natural spikes in mortality (NMFS 2008f). Although disease outbreaks have not been identified in this population, increased contaminant loading may increase the susceptibility of individuals to disease.

Anthropogenic Threats. Salmon is the primary prey of killer whales and has been severely reduced due to habitat loss (NRC 1996, Slaney et al. 1996, Gregory and Bisson 1997, Lichatowich 1999, Lackey 2003, Pess et al. 2003, Schoonmaker et al. 2003). A 50% reduction in killer whale calving has been correlated with years of low Chinook salmon abundance (Ward et al. 2009).

Contaminants entering SRKW habitat in Puget Sound and its surrounding waters accumulate in water, benthic sediments and in prey organisms (Krahn et al. 2009). SRKWs bioaccumulate these toxins in their tissues which may lead to numerous adverse physiological changes (Krahn et al. 2009). The greatest contaminant threats are from organochlorines (e.g. PCBs, pesticides, dioxins, furans and DDT) (Ross et al. 2000, CBD 2001, Krahn et al. 2002, Cullon et al. 2009, Krahn et al. 2009). These chemicals bioaccumulate in fatty tissues, persist and can be transmitted from mother to offspring (Haraguchi et al. 2009, Krahn et al. 2009).
Vessel activity has also been identified as a threat to SRKWs. In 2005, a U.S. vessel participating in sonar exercises apparently caused significant behavior changes in killer whale activity , such that the whales vacated the area (NMFS 2005a). Additionally, the increase in "background noise" resulting from vessel traffic has the potential to influence or disrupt the ability of SRKWs to navigate, communicate and forage (Bain and Dahlheim 1994, Gordon and Moscrop 1996, Erbe 2002, Williams et al. 2002a, Williams et al. 2002b, Holt et al. 2009).

Critical Habitat

NMFS designated critical habitat for Southern Resident killer whales on November 29, 2006 (71 FR 69054). Three specific areas were designated; (1) the Summer Core Area in Haro Strait and waters around the San Juan Islands; (2) Puget Sound; and (3) the Strait of Juan de Fuca, which comprise approximately 6,630 square kilometers of marine habitat. Three primary constituent elements exist in these areas: 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 passage conditions to allow for migration, resting and foraging. Water quality has declined in recent years due to agricultural run-off, urban development resulting in additional treated water discharge, industrial development and oil spills. The primary prey of southern residents, salmon, has also declined due to overfishing and reproductive impairment associated with loss of spawning habitat. The constant presence of whale-watching vessels and growing anthropogenic noise background has raised concerns about the health of areas of growth and reproduction as well.

North Atlantic right whale

Description of the species
All North Atlantic right whales compose a single population. Although not all individuals undergo the same migratory pattern, no subpopulation structuring has been identified.

Distribution
Right whales occur in sub-polar to temperate waters in all major ocean basins in the world, with a clear migratory pattern of high latitudes in summer and lower latitudes in winter (Cummings 1985, Rice 1998, Perry et al. 1999). The historical range of North Atlantic right whales extended as far south as Florida and northwestern Africa, and as far north as Labrador, southern Greenland, Iceland, and Norway (Reeves et al. 1978, Cummings 1985, Rice 1998). Most sightings in the western North Atlantic are concentrated within five primary habitats or high-use areas: coastal waters of the southeastern US, Cape Cod and Massachusetts Bays, the Great South Channel, the Bay of Fundy, and the Scotian Shelf (Winn et al. 1986). In 1994, the first three of these areas were designated as critical habitat for the North Atlantic right whale.

North Atlantic right whales have been observed from the mid-Atlantic Bight northward through the Gulf of Maine year-round, but are primarily found along the northeast US during summer and Florida during winter, with migratory routes in between. In New England, peak abundance of North Atlantic right whales in feeding areas occurs in Cape Cod Bay beginning in late winter. In early spring (Late February to April), peak North Atlantic right whale abundance occurs in Jordan and Wilkinson basins to the Great South Channel (Kenney et al. 1995, Nichols et al. 2008, Pace III and Merrick 2008). In late June and July, North Atlantic right whale distribution gradually shifts to the northern edge of Georges Bank. In late summer (August) and fall, much of the population is found in waters in the Bay of Fundy, the western Gulf of Maine and around Roseway Basin (Winn et al. 1986, Kenney et al. 1995, Kenney et al. 2001, Pace III and Merrick 2008). However, year-to-year variation in space and time are known and likely result from patchy prey distribution (Nichols et al. 2008). Variation in the abundance and development of suitable food patches appears to modify the general patterns of movement by reducing peak numbers, stay durations and specific locales (Brown et al. 2001, Kenney 2001). In particular, large changes in the typical pattern of food abundance will dramatically change the general pattern of North Atlantic right whale habitat use (Kenney 2001).

Migration and movement

North Atlantic right whales exhibit extensive migratory patterns, traveling along the eastern seaboard of the US and Canada between calving grounds off Georgia and Florida to northern feeding areas off of the northeast US and Canada in March/April and the reverse direction in November/December. The longest tracking of a North Atlantic right whale was a migration of 1,200 miles in 23 days the Bay of Fundy to Georgia (Mate and Baumgartner 2001). Migrations are typically within 30 nautical miles of the coastline and in waters less than 160 feet deep. Although this pattern is well-known, most of the population, particularly the males and non-pregnant females, is not found in the calving area and may not follow this pattern. It is unknown where the majority of the non-calving population spends the winter.

There have been a few recent sightings of North Atlantic right whales far offshore, including those from Dutch ships indicating some individuals occur between 40° and 50°N, in waters influenced by the North Atlantic Current (the broad, eastward-flowing extension of the Gulf Stream). Right whales have been sighted offshore (greater than 30 miles) during surveys flown off the coast of northeastern Florida and southeastern Georgia from 1996 to 2001. These include three sightings in 1996, one in 1997, 13 in 1998, six in 1999, 11 in 2000, and six in 2001 (within each year, some were repeat sightings). Mate *et al.* (1997) recorded radio-tagged animals making extensive movements from the Gulf of Maine into deeper waters off the continental shelf (Mate et al. 1997). The frequency with which North Atlantic right whales occur in offshore waters in the southeastern US remains unclear. Occasionally, individuals are observed in distant locations, including the Gulf of Mexico, Bermuda, the Gulf of St. Lawrence, Newfoundland, Greenland, Iceland, and northern Norway (an area known as a historical North Atlantic right whale feeding area Smith et al. 2006). The Norwegian sighting (September 1992) represents one of only two sightings this century of a right whale in Norwegian waters, and the first since 1926. Together, these long-range matches indicate an extended range for at least some individuals and perhaps the existence of important habitat areas not presently well described.

Reproduction and demography

Data through the 1990s suggests that mean calving interval increased since 1992 from 3.67 years to more than five years, a significant trend that hampers North Atlantic right whale recovery (Best et al. 2001a, Kraus et al. 2007). This reproductive rate was approximately half that reported from studied populations of southern right whales (Best et al. 2001b). This has been attributed to several possible causes, including higher abortion or perinatal losses (Browning et al. 2009). An analysis of the age structure of North Atlantic right whales suggests that the population contains a smaller proportion of juvenile whales than expected, which may reflect lowered recruitment and/or high juvenile mortality (Hamilton et al. 1998, Best et al. 2001a). In addition, it is possible that the apparently low reproductive rate is due in part to unstable age structure or to reproductive senescence on the part of some females. However, knowledge on either factor is poor. Even though investment in calves is high for North Atlantic right whales, an incident of calf exchange (probably accidentally and soon after birth) and subsequent adoption through weaning has been found (Frasier et al. 2010). Although North Atlantic right whales historically separated from their calves within one year, a shift appears to have taken place around 2001 where mothers (particularly less experienced mothers) return to wintering grounds with their yearling at a much greater frequency (71% overall)(Hamilton and Cooper. 2010). The significance of this change is unknown.

Habitat
Available evidence from North Atlantic right whale foraging and habitat studies shows that North Atlantic right whales focus foraging activities where physical oceanographic features such as water depth, current, and mixing fronts combine to concentrate copepods (Wishner et al. 1988, Murison and Gaskin 1989, Mayo and Marx 1990, Baumgartner et al. 2003).

Feeding
North Atlantic right whales fast during the winter and feed during the summer, although some may opportunistically feed during migration. North Atlantic right whales use their baleen to sieve copepods from dense patches, found in highly variable and spatially unpredictable locations in the Bay of Fundy, Roseway Basin, Cape Cod Bay, the Great South Channel, and other areas off northern US and Canada (Pendleton et al. 2009). The primary prey of North Atlantic right whales is zooplankton, especially shrimp-like copepods such as *Calanus* (Kenney et al. 1985, Beardsley et al. 1996). North Atlantic right whales feed largely by skimming these prey from the ocean surface (Pivorunas 1979, Mayo and Marx 1990), but may feed anywhere in the water column (Watkins and Schevill 1976, 1979, Goodyear 1993, Winn et al. 1995). Feeding behavior has only been observed in northern areas and not on calving grounds or during migration (Kraus et al. 1993).

Status and trends
The Northern right whale was originally listed as endangered in 1970 (35 FR 18319), and this status remained since the inception of the ESA in 1973. The early listing included both the North Atlantic and the North Pacific populations, although subsequent genetic studies conducted by Rosenbaum (2000) resulted in strong evidence that North Atlantic and North Pacific right whales are separate species. Following a comprehensive status review, NMFS concluded that North Atlantic and North Pacific right whales are separate species. In March 2008, NMFS published a final rule listing North Pacific and North Atlantic right whales as separate species (73 FR 12024).

North Atlantic right whales were formerly abundant, with an estimated 5,500 individuals present in the 16th century throughout the North Atlantic (Reeves 2001, Reeves et al. 2007). A review of the photo-id recapture database in June 2006, indicated that only 313 individually recognized North Atlantic right whales were observed during 2001. This represents a nearly complete census, and the estimated minimum population size. However, no estimate of abundance with an associated coefficient of variation has been calculated for the population. The population growth rate reported for the period 1986 to 1992 by Knowlton *et al.* (1994) was 2.5%, suggesting the stock was showing signs of slow recovery. However, work by Caswell *et al.* (1999) suggested that crude survival probability declined from about 0.99 in the early 1980's to about 0.94 in the late 1990s. Additional work conducted in 1999 showed that survival had indeed declined in the 1990s, particularly for adult females (Best et al. 2001a). Another workshop in September 2002 further confirmed the decline in this population (Clapham 2002).

Natural threats
Several researchers have suggested that the recovery of North Atlantic right whales has been

impeded by competition with other whales for food (Rice 1974, Scarff 1986). Mitchell (1975) analyzed trophic interactions among baleen whales in the western North Atlantic and noted that the foraging grounds of North Atlantic right whales overlapped with the foraging grounds of sei whales. Both species feed preferentially on copepods. Reeves *et al.* (1978) noted that several species of whales feed on copepods in the eastern North Pacific, so that the foraging pattern and success of right whales would be affected by other whales as well. Mitchell (1975) argued that the North Atlantic right whale population had been depleted by several centuries of whaling before steam-driven boats allowed whalers to hunt sei whales; from this, he hypothesized that the decline of the right whale population made more food available to sei whales and helped their population to grow. He then suggested that competition with the sei whale population impedes or prevents the recovery of the right whale population.

Other natural factors influencing right whale recovery are possible, but unquantified. Right whales have been subjects of killer whale attacks and, because of their robust size and slow swimming speed, tend to fight killer whales when confronted (Ford and Reeves 2008). Similarly, mortality or debilitation from disease and red tide events are not known, but have the potential to be significant problems in the recovery of right whales because of their small population size.

Anthropogenic threats

Several human activities are known to threaten North Atlantic right whales: whaling, commercial fishing, shipping, and environmental contaminants. Historically, whaling represented the greatest threat to every population of right whales and was ultimately responsible for listing right whales as an endangered species. As its legacy, whaling reduced North Atlantic right whales to about 300 individuals in the western North Atlantic Ocean; the number of North Atlantic right whales in the eastern North Atlantic Ocean is probably much smaller, although we cannot estimate the size of that population from the data available.

Of the current threats to North Atlantic right whales, entanglement in commercial fishing gear and ship strikes pose the greatest threats. Along the Atlantic coast of the US and the Maritime Provinces of Canada, there were 43 reports of North Atlantic right whales entangled in fishing gear between 1999 and 2005 (Cole et al. 2005a). Of the 39 reports that NMFS could confirm, North Atlantic right whales were injured in five of the entanglements and killed in four entanglements. Recent efforts to disentangle right whales have met with success (Anonmyous. 2009b). In the same region, there were 18 reports of North Atlantic right whales being struck by vessels between 1999 and 2005 (Cole et al. 2005, Nelson et al. 2007). Of the 17 reports that NMFS could confirm, right whales were injured in two of the ship strikes and killed in nine. Present recommendations for slower vessel speeds in the Bay of Fundy appear to be largely ignored (Vanderlaan et al. 2008). Proposed rules for seasonal (June through December) slowing of vessel traffic to 10 knots or changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are predicted to be capable of reducing ship strike mortality by 62% in the Bay of Fundy region.

Concern also exists over climate change and its effect on the ability of North Atlantic right whales to recover (Greene et al. 2003). Specifically, the variations in oceanography resulting

from current shifts and water temperatures can significantly affect the occurrence of the North Atlantic right whale's primary food, copepod crustaceans. If climate changes such that current feeding areas cannot sustain North Atlantic right whales, the population may have to shift to reflect changes in prey distribution, pursue other prey types, or face prey shortage. Changes in calving intervals with sea surface temperature have already been documented for southern right whales (Leaper et al. 2006).

North Atlantic right whales, as with many marine mammals, are exposed to numerous toxins in their environment, many of which are introduced by humans. 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 North Atlantic right whale recovery (Wise et al. 2008, Chen et al. 2009). The organochlorines DDT, DDE, PCBs, dieldrin, chlordane, HCB, 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). Organochlorines and pesticides, although variable in concentration by season, do not appear to currently threaten North Atlantic right whale health and recovery (Weisbrod et al. 2000). Flame retardants such as PBDEs (known to be carcinogenic) have also been measured in North Atlantic right whales (Montie et al. 2010).

Critical habitat

Critical habitat is designated for right whales in the North Atlantic. NMFS designated three areas in June 1994 as critical habitat for *Eubalaena glacialis* for feeding and calving (59 FR 28805). The critical habitats for feeding cover portions of the Great South Channel (east of Cape Cod), Massachusetts Bay and Cape Cod Bay, and Stellwagen Bank. Northern critical habitat was designated because of the concentration of right whales that feed in the area, apparently associated with complex oceanographic features that drive prey density and distribution. This area has come under considerable scrutiny within the past few years because of the concern over ship strikes in this area. Boston serves as a major port facility and vessels transiting to and from the port cross critical habitat where North Atlantic right whale mortality occurs. Shipping traffic has generally increased in the recent past and could be considered to degrade the habitat due to the additional mortality and injury risk now present in the area. Although voluntary regulations are in place, these are frequently ignored and mandatory regulations are under consideration. The southern critical habitats are along Georgia and northeastern Florida coasts (waters from the coast out 15 nautical miles between the latitudes of 31°15' N and 30°15' N and from the coast out five nautical miles between 30°15' N and 28°00' N). Southern critical habitat is designated to protected calving and breeding grounds for North Atlantic right whales, which generally calve and breed in shallow coastal waters. This critical habitat has generally fared better than northern critical habitat and significant degradation has not been clearly identified.

Humpback whale

Species Description and Distribution

The humpback whale is a cosmopolitan species that occurs in the Pacific, Atlantic, Indian, and

Southern Oceans. Most populations migrate between breeding areas in tropical waters, usually near continental coastlines or island groups, and productive colder waters in temperate and high latitudes (Reilly et al. 2008).

Listing Status

The humpback whale was listed as endangered under the ESA in 1973. It is considered depleted by the MMPA. In 2008, the IUCN Red List of Threatened Species downgraded the species from Vulnerable to Least Concern, reflecting its low risk of extinction; however, the Arabian Sea and Oceania populations remain listed as Endangered (Reilly et al. 2008). It is also protected by the International Whaling Commission (IWC) and the Convention on International Trade in Endangered Species of wild flora and fauna (CITES). Critical habitat has not been designated for the species.

Population Designations, Abundance and Trends

Prior to commercial whaling, hundreds of thousands of humpback whales existed worldwide (Winn and Reichley. 1985, Roman and Palumbi 2003). Global abundance declined to the low thousands by 1968, the last year of substantial catches (Reilly et al. 2008). Since then, the total population size has grown to over 60,000 individuals and continues to increase (Reilly et al. 2008). Humpback whales are broadly divided into four broad geographic regions based on tagging and genetic data (Baker et al. 1990, Palsboll et al. 1995): North Pacific, North Atlantic, Arabian Sea, and Southern Hemisphere.

North Pacific. In the winter, humpback whales breed and calf in the coastal waters of Southeast Asia, Hawaii, Mexico, and Central America. In the summer, they move to foraging areas in the Bering Sea, the Gulf of Alaska, and the temperate eastern Pacific (Calambokidis 2010). All breeding areas and most foraging areas are genetically differentiated, as indicated by maternally inherited genetic markers (i.e., mitochondrial haplotypes; Baker 2008). In addition, most breeding and foraging areas exhibit significantly different haplotype frequencies, indicating that there is not a simple one-to-one relationship among feeding and breeding areas (Baker 2008). An estimated 15,000 humpback whales resided in the North Pacific in 1905 (Rice 1978). Commercial whaling depleted the population to the low thousands by 1965 (Perry et al. 1999). Current estimates indicate approximately 20,000 humpback whales in the North Pacific, with an annual growth rate of 4.9 percent (Calambokidis 2010).

North Atlantic. In the summer, North Atlantic humpback whales range from the Gulf of Maine in the west and Ireland in the east. The northern extent of their range includes the Barents Sea, Greenland Sea, and Davis Strait. In the winter, the majority migrate to breeding grounds in the West Indies, though a small number migrate to the Cape Verde Islands (Reilly et al. 2008). Limited genetic exchange among summer feeding areas but mixing in the winter breeding areas is indicative of a single panmictic (i.e., interbreeding) population in the North Atlantic (Palsboll et al. 1997). Whaling nearly extirpated humpback whales from the eastern North Atlantic by 1910 and the Canadian Atlantic by 1920 (Stevick et al. 2003). Protection against whaling began in 1955, and the population has since rebounded. As of 1993, there was an estimated 11,570 humpback whales in the North Atlantic, growing at a rate of three percent annually (Stevick et al. 2003).

Southern Hemisphere. Humpback whales are abundant throughout the Antarctic during the summer; they occur south to the ice edge but not within the pack ice zone (Reilly et al. 2008). In the winter, Southern Hemisphere whales migrate to coastal areas within the South Pacific, South Atlantic, and Indian Oceans. There is genetic differentiation within and among all southern ocean basins (Baker et al. 1998, Rosenbaum et al. 2009). Over 200,000 humpback whales were killed in the Southern Hemisphere during the early 20th century. The area now supports more than 36,000 humpback whales and is growing at a minimum annual rate of 4.6 percent (Reilly et al. 2008)

Arabian Sea. A small, genetically and demographically distinct population of humpback whales resides year-round in the Arabian Sea (Mikhalev 1997, Reilly et al. 2008). Though historical estimates are not available, 242 whales were killed in 1965 and 1966 (Reilly et al. 2008). The minimum population size, based on photo-identification data, is 56 whales (Mikhalev 1997); the maximum estimate is 400 (Reilly et al. 2008).

Threats

Three human activities are known to threaten humpback whales: directed harvest, fisheries interactions, and vessel collisions. Historically, whaling represented the greatest threat to every population of humpback whales. It was ultimately responsible for the global decline in humpback whales prior to their listing as an endangered species. Hundreds of thousands of whales were removed from the world's oceans prior to bans on commercial whaling in the mid-20th century (Reilly et al. 2008). Humpback whales are often killed or injured during interactions with commercial fishing gear. They are vulnerable to ship strikes, which are often fatal. Their natural predators my include orcas, false killer whales, and sharks.

Life History Information

Humpback whale reproductive activities occur primarily in winter. Gestation takes about 11 months (Winn and Reichley. 1985), followed by a nursing period of up to one year (Baraff and Weinrich 1993). Calving occurs in the shallow coastal waters of continental shelves and some oceanic islands (Perry et al. 1999). The calving interval is likely two to three years (Clapham and Mayo 1987), although some evidence exists of calving in consecutive years (Glockner-Ferrari and Ferrari 1985, Clapham and Mayo 1987, 1990, Weinrich et al. 1993). Mother/calf groups are found in relatively stable pairs (Ersts and Rosenbaum 2003). Sexual maturity in humpback whales is reached between five and 11 years of age (Clapham 1992, Gabriele et al. 2007). During the breeding season, humpback whales form small unstable groups (Clapham 1996). Males sing long, complex songs, compete for mates, and are polygamous (Clapham 1996).

Humpback whales migrate long distances from breeding areas to foraging areas. Although largely solitary, humpback whales often cooperate during feeding activities (Elena et al. 2002). Feeding groups are sometimes stable for long periods of times, and there is good evidence of some territoriality on both feeding (Clapham 1996) and wintering grounds (Tyack 1981). Humpbacks exhibit a wide range of foraging behaviors and feed on a range of prey types, including: small schooling fishes, euphausiids, and other large zooplankton (Nemoto 1957,

Nemoto 1959, Nemoto 1970, Krieger and Wing. 1984, Krieger and Wing. 1986). Because most humpback prey are likely found above 300 m depths, most dives are probably relatively shallow, with typical diving depths of approximately 60-170 m (Hamilton et al. 1997). Dives usually range between two and five minutes but can last to around 20 minutes (Dolphin 1987).

Invertebrates

Elkhorn coral

Description of the species

Although they resemble plants, elkhorn coral is a colony of invertebrates that collaboratively form frond-like branches radiating from a central trunk that is firmly attached to the sea floor. The largest species of its genus, colonies can reach at least 6.6 feet high and 13 feet in diameter (Veron 2000). Corallites (branches of radial arms of calcium carbonate) are tube-like and porous, 0.08 inch to 0.16 inch long, about 0.08 inch in diameter, white near the growing tip, and brown to tan away proximally.

Distribution

Elkhorn coral is found widely in the Caribbean, including in the Florida Keys, Abaco Island (The Bahamas), Alacran Reef, Mexico, Belize, Colombia, Costa Rica, Guatemala, Honduras, Nicaragua, Panama, Venezuela, Bonaire, Cayman Islands, Jamaica, Puerto Rico, US Virgin Islands, Navassa, and throughout the West Indies (Goreau 1959, Kornicker and Boyd 1962, Storr 1964, Scatterday 1974, Jaap 1984, Dustan and Halas 1987, NMFS 2006a). However, abundance within the distribution is reduced, largely due to water temperature and quality issues.

Growth and reproduction

Elkhorn corals employ both sexual and asexual reproduction. Sexual reproduction is accomplished by releasing sperm and egg during spawning events. Colonies are referred to as simultaneous hermaphrodites, meaning that a given colony contains both female and male reproductive sex organs (Szmant 1986). Spawning events are relatively short, with gametes released only a few nights during July, August, and/or September. In some populations, spawning is synchronous after a full moon. Annual egg production in Puerto Rico was estimated to be 3,870 to 5,100 eggs per square inch of living coral tissue (Szmant 1986). Once fertilization occurs, planktonic larvae form before settling and metamorphosizing on appropriate substrates, preferably coralline algae (Bak 1977, Sammarco 1980, Rylaarsdam 1983). Initial calcification ensues and develop into daughter corallites.

Studies indicate that larger colonies (as measured by surface area of the live colony) have higher fertility and fecundity rates; over 80% of the colonies larger than 620 inches^2 were fertile. Estimated colony size at sexual maturity was 248 inches^2 and the smallest reproductive colony observed was 6.3 inches^2 by 3.15 inches^2 (Soong and Lang 1992).

Biological and physical factors affect spatial and temporal patterns of recruitment. These include substrate availability and community structure, grazing pressure, fecundity, mode and timing of reproduction, behavior of larvae, hurricane disturbance, physical oceanography, the structure of

established coral assemblages, and chemical cues (Lewis 1974, Birkeland 1977, Goreau et al. 1981, Rogers et al. 1984, Baggett and Bright 1985, Harriott 1985, Hughes and Jackson 1985, Sammarco 1985, Morse et al. 1988, Fisk and Harriott 1990, Richmond and Hunter 1990).

Growth rates are relatively rapid, expressed as the linear extension of branches, ranging from 1.57 to 4.33 inches annually, and have enabled elkhorn coral to construct significant reefs in several locations throughout the Caribbean (Vaughan 1915, Jaap 1974, Adey 1978). Branching species, such as acroporid corals, grow differentially in response to light such that coral polyp growth maximizes exposure to available light (Kaniewska et al. 2009). Growth can also occur from fragmentation and dispersal (Tunnicliffe 1981, Bak and Criens 1982). A broken branch may be carried by waves and currents to another location and, if favorable, branches grow into a new colony. Rapid growth and fragment dispersal facilitate a competitive advantage for elkhorn coral relative to other coral and benthic species (Shinn 1976, Neigel and Avise 1983, Jaap et al. 1989).

Elkhorn coral require relatively clear water and depend almost entirely upon symbiotic photosynthesizers (zoozanthelle) for nourishment (Porter 1976, Lewis 1977, Jaap et al. 1989, Mieog et al. 2009) and is much more susceptible to increases in water turbidity than are some other corals. Different strains of symbiotic zoozanthelle (*Symbiodinium* spp.) can confer different thermal and light tolerances to acroporiids (Abrego et al. 2009, Ainsworth and Hoegh-Guldberg 2009, Abrego et al. 2010). The type of *Symbiodinium* spp. may change during ontogeny or remain the same, depending upon acroporiid species, and may be the same as parent colonies or not (Baird et al. 2007, Gómez-Cabrera et al. 2008, Abrego et al. 2009).

Habitat

Colonies of elkhorn coral often grow in dense stands and form interlocking framework known as thickets in fringing and barrier reefs, ranging in depth from 3.3 to 49 feet (Jaap 1984, Dustan 1985, Dustan and Halas 1987, Tomascik and Sander 1987, Wheaton and Jaap 1988). However, optimal depth range is considered to be 3.3 to 16.4 feet in depth, with possible exposure at low tide (Goreau and Wells 1967). Colonies generally do not form thickets below 16.4 feet, with maximum water depths of framework construction ranging from 10 to 39.4 feet (Lighty et al. 1982). Elkhorn coral thrive in shallow reef zones where wave energy is a significant factor. In areas with strong wave energy conditions only isolated colonies occur, while denser thickets may develop in intermediate wave energy conditions (Geister 1977). The preferred habitat of elkhorn coral is the seaward face of a reef (Shinn 1963, Cairns 1982, Rogers et al. 1982).

Status and trends

Elkhorn coral was listed as threatened under the ESA on May 9, 2006 (71 FR 26852). Elkhorn coral underwent precipitous declines in the early 1980s throughout its range and this decline has continued. Although quantitative data on historical distribution and abundance are scarce, best available data indicate declines in abundance (coverage and colony numbers) by greater than 97%. Recovery from a bleaching event in 2005 is expected to take 10-12 years; this is after a previous event in 1997. In all, roughly one-third of the *Acropora palamata* genotypes have been lost as a result of these events (Miller and Williamson 2010).

Natural threats
The overriding threats are disease, temperature-induced "bleaching" (loss of zoozanthelle), and physical damage from hurricanes (Carpenter et al. 2008, Mallela and Crabbe 2009, Baskett et al. 2010). Disease is widespread, episodic, and unpredictable in its occurrence and results in high mortality. This is primarily due to a disassociation of zoozanthelle from coral tissue. Just prior to this, coral epithelium and gastrodermis tissue begins to decay and die, likely as a result of stress to the individual coral (Ainsworth et al. 2008). Optimal water temperatures range from 77º to 84ºF, with mortality observed at 61º and 96ºF (Jaap 1979, Roberts et al. 1982). High light levels can also induce mortality. Synergistic analyses have found that high temperature increases the risk of colony mortality under a variety of sediment loading conditions, but excessive sediment appears to reduce mortality risk under high light and temperature regimes, possibly by reducing exposure to these stressors (Anthony et al. 2007, Boyett et al. 2007). High sediment with otherwise good light and temperature conditions appears to increase colony mortality (Anthony et al. 2007).

Elkhorn coral require near oceanic salinities (34 to 37 parts per thousand). High temperature or rapid heating can result in heat shock and alter cellular metabolism within the coral as well as possibly hinder immune response or the ability of zoozanthelle to thrive (Rodriguez-Lanetty et al. 2009, Middlebrook et al. 2010). Bleaching can occur due to adverse environmental conditions (Ghiold and Smith 1990, Williams and Bunkley-Williams 1990) and is currently a significant factor in deteriorating coral reef health. In 2005, wide-scale bleaching occurred throughout the Caribbean with wide-scale mortality, with some areas reaching 95% of coral colonies affected (Wilkinson and Souter 2008).

The US Virgin Islands, a location of *Acropora* critical habitat, experienced greater than 50% mortality of corals, the greatest level ever recorded. Puerto Rico and Florida (additional areas of *Acropora* critical habitat) also experienced disease rates of 50% of coral colonies or greater. Bleaching was associated with unusually warm waters in the region. Encouragingly, bleaching events can lead to increased thermal tolerance in affected reefs, meaning that subsequent bleaching events are not as severe (Maynard et al. 2008). A record number of hurricanes also caused extensive damage to coral reefs; the prevalence of hurricanes and subsequent coral reef damage has been linked to climate change (Wilkinson and Souter 2008). Ocean acidification is also a threat due to the increased solubility of calcium carbonate in even slightly more acidic sea water (thereby eroding the shells which form coral hard parts)(Anthony et al. 2008, De'ath et al. 2009, Wei et al. 2009, Crawley et al. 2010). Acidification also reduces the thermal tolerance of corals, meaning that bleaching can occur at lower temperatures (Anthony et al. 2008). Hurricanes can cause wide-scale inhibition of recruitment in years following storm passage as well as physical damage to coral colonies themselves (Mallela and Crabbe 2009).

Anthropogenic threats
Threats to elkhorn coral also include eutrophication, sedimentation, anchoring, which degrade coral condition and increase synergistic stress effects (e.g. bleaching).

Critical habitat
NMFS published a final rule to designate critical habitat for elkhorn and staghorn corals on

November 26, 2008 (73 FR 72210). This habitat serves as substrate of suitable quality and availability, in water depths from the mean high water line to 98 feet (except along some areas of Florida, where 6 foot contour is the shoreward limit), to support successful larval settlement, recruitment, and reattachment of fragments. Four specific areas are proposed for designation: the Florida unit, which comprises approximately 1,329 square miles of marine habitat; the Puerto Rico unit, which comprises approximately 1,383 square miles of marine habitat; the St. John/St. Thomas unit, which comprises approximately 121 square miles of marine habitat; and the St. Croix unit, which comprises approximately 126 square miles of marine habitat. NMFS proposes to exclude one military site, comprising approximately 47 square miles, because of national security impacts. The lone PCE identified thus far is natural consolidated hard substrate or dead coral skeleton that are free from fleshy or turf macroalgae cover and sediment cover. This feature is essential to the conservation of these two species because of the extremely limited recruitment currently being observed and the need for this species to have habitat to recruit into.

Staghorn coral

Description of the species

Although they resemble plants, staghorn coral is a colony of small shelled animals that collaboratively form staghorn-antler-like colonies, with cylindrical, straight or slightly curved diverging branches. Branches are 0.1 inch to 0.6 inch in diameter and rarely may grow back together. Colonies in turbulent water are smaller than in calm water, with greater branch density. Branching is irregular and secondary branches form at 60 to 90 degree angles relative to a primary branch. Prominent axial corallites (branches of radial arms of calcium carbonate) form at branch tips; bract-like corallites radiate symmetrically around branches. Individual colonies are up to 5 feet across and typically form monospecific thickets. Tissue color ranges from golden yellow to medium brown, with little or no color near the growing branch tips. The colony may or may not be firmly attached to the sea floor.

Distribution

Staghorn coral is found widely in the Caribbean, including in the Florida Keys, Abaco Island (The Bahamas), Alacran Reef, Mexico, Belize, Colombia, Costa Rica, Guatemala, Honduras, Nicaragua, Panama, Venezuela, Bonaire, Cayman Islands, Jamaica, Puerto Rico, US Virgin Islands, Navassa, and throughout the West Indies (Goreau 1959, Kornicker and Boyd 1962, Storr 1964, Scatterday 1974, Jaap 1984, Dustan and Halas 1987, NMFS 2006a). However, abundance within the distribution is reduced, largely due to water temperature and quality issues.

Growth and reproduction

Staghorn corals employ both sexual and asexual reproduction. Sexual reproduction is accomplished by releasing sperm and egg during spawning events (Szmant 1986). Colonies are referred to as simultaneous hermaphrodites, meaning that a given colony contains both female and male reproductive sex organs. Spawning events are relatively short, with gametes released only a few nights during July, August, and/or September. In some populations, spawning is synchronous after a full moon. Annual egg production in Puerto Rico was estimated to be 3,870 to 5,100 eggs per square inch of living coral tissue (Szmant 1986). Once fertilization occurs,

planktonic larvae form before settling and metamorphosizing on appropriate substrates, preferably coralline algae (Bak 1977, Sammarco 1980, Rylaarsdam 1983). Initial calcification ensues and develop into daughter corallites.

Studies indicate that larger colonies (as measured by surface area of the live colony) have higher fertility and fecundity rates; colonies with a branch length longer than 3.5 inches were fertile and over 80% of colonies with branches longer than 6.7 inches were fertile (Soong and Lang 1992). Estimated size at sexual maturity is 6.7 inches in branch length and the smallest known reproductive colony was 3.5 inches in branch length (Soong and Lang 1992).

Biological and physical factors affect spatial and temporal patterns of recruitment. These include substrate availability and community structure, grazing pressure, fecundity, mode and timing of reproduction, behavior of larvae, hurricane disturbance, physical oceanography, the structure of established coral assemblages, and chemical cues (Lewis 1974, Birkeland 1977, Goreau et al. 1981, Rogers et al. 1984, Baggett and Bright 1985, Harriott 1985, Hughes and Jackson 1985, Sammarco 1985, Morse et al. 1988, Fisk and Harriott 1990, Richmond and Hunter 1990).

Growth rates are relatively rapid, expressed as the linear extension of branches, ranging from 1.2 to 4.3 inches annually, and have enabled staghorn coral to construct significant reefs in several locations throughout the Caribbean (Vaughan 1915, Jaap 1974, Adey 1978). Branching species, such as acroporid corals, grow differentially in response to light such that coral polyp growth maximizes exposure to available light (Kaniewska et al. 2009). During the 1970s there were vast fields, or thickets, of staghorn coral on many reefs. The nominal situation in 2004 was isolated branches and small thickets, 1.6 to 3.3 feet across. Growth can also occur from fragmentation and dispersal (Tunnicliffe 1981, Bak and Criens 1982). A broken branch may be carried by waves and currents to another location and, if favorable, branches grow into a new colony. Rapid growth and fragment dispersal facilitate a competitive advantage for staghorn coral relative to other coral and benthic species (Shinn 1976, Neigel and Avise 1983, Jaap et al. 1989). Larval recruitment is influenced by the type and availability of benthic substrate, with certain types of coral or rock substrates resulting in greater or lesser recruitment success (Ritson-Williams et al. 2009).

Habitat

Historically, staghorn coral so dominated reef systems within the 23 to 49 feet depth that the area became known as the staghorn zone. In other reef systems (Jamaica, Cayman Islands, Belize, and eastern Yucatan), staghorn coral was a major mid-depth (33 to 82 feet) reef-builder (Adey 1977, 1978). Historically, staghorn coral was reported from depths ranging from surface to 200 feet, although it is considered rare below 66 feet (Goreau and Goreau 1973). In southeastern Florida, this species historically occurred on the outer reef (52 to 66 feet), on spur, groove bank, and transitional reefs, and on octocoral-dominated hard-bottom (Goldberg 1973, Davis 1982, Jaap 1984, Wheaton and Jaap 1988). Colonies were common in back- and patch-reef habitats (Gilmore and Hall 1976, Cairns 1982). Although staghorn coral colonies are sometimes found interspersed among colonies of elkhorn coral, they are generally in deeper water or seaward of the elkhorn zone and more protected from wave action.

Staghorn coral require relatively clear water and depend almost entirely upon symbiotic photosynthesizers (zoozanthelle) for nourishment (Porter 1976, Lewis 1977, Jaap et al. 1989, Mieog et al. 2009) and is much more susceptible to increases in water turbidity than are some other corals. Different strains of symbiotic zoozanthelle (*Symbiodinium* spp.) can confer different thermal and light tolerances to acroporiids (Abrego et al. 2009, Ainsworth and Hoegh-Guldberg 2009, Abrego et al. 2010). The type of *Symbiodinium* spp. may change during ontogeny or remain the same, depending upon acroporiid species, and may be the same as parent colonies or not (Baird et al. 2007, Gómez-Cabrera et al. 2008, Abrego et al. 2009).

Status and trends

Staghorn coral was listed as threatened under the ESA on May 9, 2006 (71 FR 26852). Staghorn coral underwent precipitous declines in the early 1980s throughout its range and this decline has continued. Although quantitative data on historical distribution and abundance are scarce, best available data indicate declines in abundance (coverage and colony numbers) by greater than 97%.

Staghorn corals still occupy their historic range, but localized range reductions and extirpations have occurred with most populations experiencing losses from 80-98% of their 1970s baseline (Bruckner 2002). Monitoring data from around the USVI indicates that staghorn corals have virtually disappeared from the north side of Buck Island, St. Croix, and only a few localized areas off the southern reef contain staghorn corals, representing 2-3% of the coral cover in these areas (Rogers et al. 2002). Surveys of fragments of staghorn from nearshore areas of St. Thomas and outlaying cays indicate that colonies of these corals were once much more abundant than the numbers recorded in the 2003 survey (Rogers et al. 2008).

In Puerto Rico, well-developed and dense thickets of staghorn coral were present through the late 1970s at many reefs surrounding the main island, and also the offshore islands of Mona, Vieques and Culebra (Almy and Carrión-Torres 1963, McKenzie and Benton 1972, Goenaga and Cintrón 1979, Boulon Jr. 1980). Later, in 1978-79 during an island-wide survey, staghorn coral was found on only 20% of those reefs (Bruckner 2002). Prior to Hurricane David in 1979, 20 random 0.6 m^2 photoquadrats were selected from each of 10, 40-m-long transects parallel to the depth contours across the reef (16.7 to 19.2 m depth). Based on analysis of point count data, staghorn coral had a mean total cover of 31.1% (range of 9.9 to 56.9%); after the storm, total cover of staghorn coral dropped to a mean of 0.90% (range of 0.02 to 2.7%)(NMFS 2008a). In the summer of 2004, there was an epidemic outbreak of white pox disease at Los Corchos Reef in Culebra, Puerto Rico. Prior to the outbreak, coral cover on the reef reached values of 80%. However, three weeks after Tropical Storm Jeanne, 80 to 90% of the staghorn coral colonies at permanent monitoring sites at Los Corchos were already dead or dying; likely as a result of impacts from both disease and storm damage (NMFS 2008a). During the 2005 bleaching event, near Culebra Island, almost 100% of staghorn colonies suffered partial to complete mortality due to bleaching (García-Saís et al. 2008). Similar to the situation in USVI, the bleaching event was followed by a white plague-like massive outbreak that caused mass mortality and resulted in a net 20-60% decline in living coral cover at surveyed reefs of the east coast within a period of approximately six months.

Following the 2005 bleaching event, monitoring data indicate that total coral cover is now less

than 12% on many reefs (Rogers et al. 2008). Coral mortality due to the 2005 bleaching event was more severe than at any time in the last 40 years of monitoring in USVI (Woody et al. 2008). Staghorn corals suffered widespread mortality associated with the 2005 bleaching event and current monitoring data does not indicate significant recovery (Rothenberger et al. 2008, Woody et al. 2008). Overall, colonies of Atlantic *Acropora* have declined by up to 98% and live colonies were no longer present at many study sites in the USVI following the 2005-2006 bleaching event.

Natural threats

The overriding threats are disease, temperature-induced "bleaching" (loss of zoozanthelle), and physical damage from hurricanes (Carpenter et al. 2008, Mallela and Crabbe 2009, Baskett et al. 2010). Disease is widespread, episodic, and unpredictable in its occurrence and results in high mortality. This is primarily due to a disassociation of zoozanthelle from coral tissue. Just prior to this, coral epithelium and gastrodermis tissue begins to decay and die, likely as a result of stress to the individual coral (Ainsworth et al. 2008). Optimal water temperatures range from 77º to 84ºF, with mortality observed at 61º and 96ºF (Jaap 1979, Roberts et al. 1982). High light levels can also induce mortality. Synergistic analyses have found that high temperature increases the risk of colony mortality under a variety of sediment loading conditions, but excessive sediment appears to reduce mortality risk under high light and temperature regimes, possibly by reducing exposure to these stressors (Anthony et al. 2007, Boyett et al. 2007). High sediment with otherwise good light and temperature conditions appears to increase colony mortality (Anthony et al. 2007). Elkhorn coral require near oceanic salinities (34 to 37 parts per thousand). High temperature or rapid heating can result in heat shock and alter cellular metabolism within the coral as well as possibly hinder immune response or the ability of zoozanthelle to thrive (Rodriguez-Lanetty et al. 2009, Middlebrook et al. 2010). Bleaching can occur due to adverse environmental conditions (Ghiold and Smith 1990, Williams and Bunkley-Williams 1990) and is currently a significant factor in deteriorating coral reef health. The major El Niño/La Niña Southern Oscillation cycle in 1997-1998 resulted in a large bleaching event in the Caribbean and the Atlantic, as well as massive losses of corals in the Indian Ocean and Western Pacific (Wilkinson and Souter 2008). However, the most significant bleaching event to date in the USVI and other areas of the Caribbean occurred in 2005 when sea surface temperatures exceeded the 29.5°C coral bleaching threshold for twelve weeks, and maximum temperatures exceeded 30°C (Woody et al. 2008). Bleaching occurred in twenty-two species, including *Acropora*, over a wide range of depths and affected more than 90% of the coral cover, on average, between July and November in the USVI (Woody et al. 2008). Wide-scale mortality, with some areas reaching 95% of coral colonies affected, resulted from this event (Wilkinson and Souter 2008). The US Virgin Islands, a location of *Acropora* critical habitat, experienced greater than 50% mortality of corals, the greatest level ever recorded. Puerto Rico and Florida (additional areas of *Acropora* critical habitat) also experienced disease rates of 50% of coral colonies or greater. Bleaching was associated with unusually warm waters in the region. Encouragingly, bleaching events can lead to increased thermal tolerance in affected reefs, meaning that subsequent bleaching events are not as severe (Maynard et al. 2008). A record number of hurricanes also caused extensive damage to coral reefs; the prevalence of hurricanes and subsequent coral reef damage has been linked to climate change (Wilkinson and Souter 2008). Ocean acidification is also a threat due to the increased solubility of calcium carbonate in

even slightly more acidic sea water (thereby eroding the shells which form coral hard parts)(Anthony et al. 2008, De'ath et al. 2009, Wei et al. 2009, Crawley et al. 2010). Acidification also reduces the thermal tolerance of corals, meaning that bleaching can occur at lower temperatures (Anthony et al. 2008). Hurricanes can cause wide-scale inhibition of recruitment in years following storm passage as well as physical damage to coral colonies themselves (Mallela and Crabbe 2009).

White band disease is thought to be the major factor responsible for the rapid loss of Atlantic *Acropora* due to mass mortalities. White band disease is the only coral disease to date that has been documented to cause major changes in the composition and structure of reefs (Humann and Deloach 2003). In 2011, Sutherland *et al.* (2011) were able to definitively identify human waste as a cause for white pox disease in elkhorn corals.

Reductions in long-term water clarity can also reduce the coral photosynthesis to respiration ratio (P/R ratio). Telescnicki and Goldberg (1995) and Yentsch *et al.* (2002) found that elevated turbidity levels did not affect gross photosynthetic oxygen production, but did lead to increased respiration that consumed the products of photosynthesis with little remaining for coral growth.

Unfortunately, since staghorn corals are broadcast spawners once colonies become rare, the distance between colonies may limit fertilization success and there is substantial evidence to suggest that sexual recruitment of staghorn corals is currently compromised. Reduced colony density in some areas is compounded by low genotypic diversity, indicating that fertilization success and consequently, larval availability, is likely reduced. This can have long-term implications for genetic variability of remaining colonies due to the reduced potential for exchange of genetic material between populations that are spatially further apart (Bruckner 2002).

Data on levels of genetic diversity and population structure suggest that there is a population structure among islands, and even over spatial scales of no more than 20 km, as well as varying degrees of genetic diversity within local populations (Lirman 2002, Vollmer 2002). For instance, one clone of staghorn coral may dominate areas up to 10 m^2 in size and the clones are generally spatially discrete with larval exchange between staghorn populations as close as 2 to 15 km being extremely limited, suggesting that larval sources need to be conserved on a very small spatial scale (Baums et al. 2005, Vollmer and Palumbi 2007).

Anthropogenic threats

Threats to staghorn coral are exacerbated further by eutrophication, sedimentation, and anchoring, which degrade coral condition and increase synergistic stress effects (e.g. bleaching). Excessive sedimentation can smother corals and increased nutrient availability promotes algal growth on corals, leading to light blockage to zoozanthelle and death of corals (*Acropora* Biological Review Team 2005). Although reefs in the Florida Keys currently experience about 10% macroalgal cover or less, much of the wider Caribbean Sea may exceed 20% cover (Bruno 2008), inhibiting and reducing coral survival. Global warming is also projected to have negative impacts on coral survival through coral bleaching, increased storm intensity, and reduced calcification (*Acropora* Biological Review Team 2005).

Critical habitat
NMFS published a final rule to designate critical habitat for elkhorn and staghorn corals on November 26, 2008 (73 FR 72210). This habitat serves as substrate of suitable quality and availability, in water depths from the mean high water line to 98 feet (except along some areas of Florida, where 6 foot contour is the shoreward limit), to support successful larval settlement, recruitment, and reattachment of fragments. Four specific areas are proposed for designation: the Florida unit, which comprises approximately 1,329 square miles of marine habitat; the Puerto Rico unit, which comprises approximately 1,383 square miles of marine habitat; the St. John/St. Thomas unit, which comprises approximately 121 square miles of marine habitat; and the St. Croix unit, which comprises approximately 126 square miles of marine habitat. NMFS proposes to exclude one military site, comprising approximately 47 square miles, because of national security impacts. The lone PCE identified thus far is natural consolidated hard substrate or dead coral skeleton that are free from fleshy or turf macroalgae cover and sediment cover. This feature is essential to the conservation of these two species because of the extremely limited recruitment currently being observed and the need for this species to have habitat to recruit into.

White abalone

Distribution
White abalone occur along the US west coast among offshore islands and banks (particularly Santa Catalina and San Clemente islands) and mainland inshore waters from Point Conception, California south to Punta Abreojos, Baja California, Mexico (Bartsch 1940, Cox 1960, 1962). White abalone occur primarily along the mainland coast in their northern and southern range, but are more frequently at the offshore islands (especially San Clemente and Santa Catalina islands) in the middle portion of the California range (Cox 1962, Leighton 1972). However, individuals have also been found around several Mexican islands including Isla Cedros and Isla Natividad (Guzmán Del Proó 1992). There are no recognized subspecies of white abalone although there is one possible subspecies of white abalone inhabiting Guadalupe Island, Mexico (Hobday and Tegner 2000).

Habitat
White abalone occupy kelp forests in relatively exposed areas of low-relief rocky habitat surrounded by sand. Large juvenile and small adult abalone (3 to 4 inches) are cryptic and seek shelter in crevices and under rocks before, as adults, moving to more open habitats on the tops and sides of rocks where food is more plentiful (Haaker et al. 1986, Hobday and Tegner 2000). Adult white abalone are found most abundantly at depths of 82 to 99 feet, but may occur in waters from 66 to 197 feet (Hobday and Tegner 2000).

Feeding
White abalone eat algae, feeding as postlarvae and early juveniles on bacteria, sessile pennate diatoms, and other benthic microflora. As advanced juveniles and adults, drifting brown algae and microalgal films provide the primary source of nutrition (Tutschulte 1976).

Growth and reproduction
Abalone aggregate for spawning, but low numbers and physical barriers can prevent large spawning aggregations from forming (Babcock and Keesing 1999, Leet et al. 2001). A brief annual spawning event occurs *en mass* generally between February and April (Tutschulte 1976). Although an average female is capable of producing over 20 million larvae over her lifetime, larval survival to adulthood is estimated at <1% (Leighton 2000).

Twenty four hours after fertilization, a free-swimming larva emerges from the fertilized egg and joins the plankton (Leighton 1989, 2000). After 2 to 3 weeks in the plankton the larvae settle to the bottom. One to 3 months after settlement juveniles are fully formed and resemble adults. After 2 to 4 years, white abalone are mature and inhabit the tops and sides of rocky substrates. However, low food resource availability has been shown to stunt growth in blacklip abalone (Saunders et al. 2009a, Saunders et al. 2009b).

Movement
Abalone movements tend to vary by individual, with some staying at a particular home site and others undertaking extensive movements (Momma and Sato 1969, Ault and Martini 1987). This has been further characterized in green lip abalone by Cennie *et al.* (2009a), where individuals belonged to either a sedentary or a wandering clade (which moved over larger areas and covered greater distances, occupied inner portions of hides more frequently, responded more rapidly to food odor, foraged for longer periods, and displaced conspecifics from food patches).

Life span
Researchers believe that abalone are long-lived, but the average life span is unclear (Hobday and Tegner 2000). Models of growth predict maximum size is reached in 34 to 35 years and a maximum life span of 35 to 40 years has been estimated (Tutschulte 1976, Tutschulte and Connell 1988).

Status and trends
On May 29, 2001, the white abalone was listed as an endangered species throughout its range under the ESA (66 FR 29046). In the white abalone status review, Hobday and Tegner (2000) estimated pre-exploitation abundance at 2,221,800 abalone, but the population in 1996 to 1997 was estimated at 1,613 individuals, representing a 99.9% decline, and was estimated to disappear by 2010. White abalone recovery is hindered by low spawning densities, resulting in recruitment failure. White abalone along Mexico are believed to be depleted, but their status is generally unknown. Based upon survey data, Hobday *et al.* (2001) updated the 1996 to 1997 white abalone abundance estimate to 2,540.

White abalone recruitment is highly variable (Tutschulte 1976). However, estimates of population size have been difficult to calculate because estimates are only based upon adults, as juveniles are infrequently observed. White abalone observed during surveys were of large size which corresponds to predicted ages near the end of the predicted life span (Davis 1996, Davis et al. 1998, Hobday and Tegner 2000, Hobday et al. 2001). Because no white abalone were observed in the smaller age/size classes during the surveys there appears to be a lack of successful recruitment since the 1960s (Hobday and Tegner 2000).

Natural threats
Natural pressures exist from sea otter predation (Johnson et al. 2009) and the low density of individuals during spawning events. Interspecific competition has not been studied in white abalone, but blacklip abalone face significant competition for algal resources by sea urchins (Strain and Johnson 2009).

Anthropogenic threats
White abalone numbers were severely reduced due to excessive harvest. This has led to below-threshold spawning densities in many areas that are blamed for the inability of the species to recover. Although small-scale aquaculture takes have occurred to attempt captive breeding and recovery, these takes are small in number. Otherwise, substantial human harvesting is not known. No commercial or recreational takes are permitted under ESA protection.

Although toxicology of abalone is poorly known, red abalone have demonstrated metabolic breakdown when exposed to the pesticide 3-trifluoromethyl-4-nitrophenol (Viant et al. 2001). Silver, cadmium, and mercury are also known to bioaccumulate in abalone, likely from ingested algae (Huang et al. 2008). Update rates have been measured as 1.78 L g^{-1} d^{-1} for silver, 0.056 L g^{-1} d^{-1} for cadmium and 0.32 L g^{-1} d^{-1} for mercury, of which 58 to 83%, 33 to 59%, and 65 to 78%, respectively, is assimilated. Abalone are known to bioaccumulate high levels of heavy metals in the presence of high environmental concentrations (Wang et al. 2009).

Changes in sea surface temperatures have been suggested as a driving force in altering red abalone distribution in the past (Braje et al. 2009); it is unknown what affect, if any climate change may have on white abalone.

Critical habitat
Critical habitat has not been designated for white abalone.

Black abalone

Distribution
Black abalone historically occurred between Coos Bay, Oregon, and Cape San Lucas, Baja California (Cox 1962), but were rare north of San Francisco (Morris et al. 1980). Distribution extended to the Channel Islands and from Cedros to Punta Asuncion along Baja, Mexico (Guzmán Del Proó 1992). Present occurrence remains throughout much of this range, although greatly reduced.

Growth and reproduction
Spawning occurs during spring and summer (Cox 1960). Synchronicity of gamete release is vital, as likelihood of fertilization is reliant upon dense adult aggregation and subsequent high egg and sperm density (Davis 1996). Fecundity increases exponentially with size, with small mature females producing a few hundred thousand eggs each year, but older individuals producing 10 to 15 million eggs (Hahn 1989). Fertilized eggs sink and hatch into free-swimming

larvae within 72 hours. After one or two weeks, larvae settle (CDFG 1993). Larval mortality is assumed to be high (Leighton 1972). At 1.5 inches (roughly 3 years of age), black abalone are considered sexually mature (Blecha et al. 1992). Size at sexual maturity appears to decrease with latitude, suggesting precocity in southern portions of the range (Munoz and Camacho 1976, Guzman del Proo et al. 1980).

Feeding

Black abalone eat algae, with larvae eating pelagic plankton and postlarvae feeding upon on bacterial films, benthic diatoms (Cox 1962, Ault 1985), and coralline algae. Juveniles and adults feed upon large-bodied algae, such as giant and feather-boa kelp (Cox 1962, Howorth 1978). However, low food resource availability has been shown to stunt growth in blacklip abalone (Saunders et al. 2009a, Saunders et al. 2009b).

Habitat

Black abalone occupy rocky intertidal and shallow subtidal zones (Haaker et al. 1986). Depth distribution is usually from the shore to about 10 feet of water depth. Mobility patterns are size dependent (Blecha et al. 1992). Individuals above 4.1 inches undergo limited movement in exposed rocky locations. Juveniles (smaller than 1.6 inches) remain within rocky crevices during daylight hours, but become more active at night (Cox 1960). At 3 to 4 inches in length, when they emerge into more open rocky habitats where food may be more abundant (Haaker et al. 1986). Movement patterns have been further characterized in green lip abalone by Cennie *et al.* (2009a), where individuals belonged to either a sedentary or a wandering clade (which moved over larger areas and covered greater distances, occupied inner portions of hides more frequently, responded more rapidly to food odor, foraged for longer periods, and displaced conspecifics from food patches).

Life span

Black abalone lifespan is unknown, but abalone are believed to survive for 30 years or more (Blecha et al. 1992).

Status and trends

On January 11, 2008, NMFS published a proposed rule to list black abalone as endangered under the ESA (73 FR 1986). Species decline is reflected by the decrease in commercial catches until 1993, when commercial harvests were halted. Historic levels approached 2,200 tons in California in 1879 and declined to around 1000 tons in the 1970's. Commercial landings then decreased to 19.1 tons in the last year of harvests, when mortality from withering syndrome devastated remaining black abalone stocks throughout southern California (Haaker 1994). Over 20 years, densities of more than 100 individuals per cubic yard disappeared from most of their former range south of Point Conception (Davis 1993). A similar mass mortality was reported at Palos Verdes Peninsula in the late 1950's, where average density decreased from more than 2.8 individuals per square yard from 1975 to 1979 down to about 0.03 individuals per square yard from 1987 to 1991 (Cox 1962). Island habitats experienced more severe trends; 99% of black abalone vanished from Anacapa, Santa Barbara, and Santa Rosa Islands in less than 5 years (Haaker et al. 1989, Richards and Davis 1993).

Black abalone have also experienced severe declines due to a temperature-related disease called withering syndrome. This bacteria-based disease prevents assimilation of nutrients in the digestive system and results in abalone that "wither" as individuals consume body tissues. The disease was first identified west of Santa Cruz and Anacapa islands in 1985 and 1986 before spreading to Santa Rosa Island and Santa Barbara Island by 1988. The disease made its appearance along the mainland in 1988 in San Luis Obispo county, where 85% of the resident black abalone died in Diablo Cove. This die-off was attributed to the presence of warm-water effluent from a nuclear power facility. From 1988 to the early 1990's, withering syndrome continued to spread throughout the Channel islands to 2000, when it was estimated that only 1% of the original population remained (Richards 2000).

Natural threats

Along with depletion from commercial harvests, withering syndrome has also been significant in black abalone decline. Withering syndrome is a chronic, degenerative disease responsible for mass mortalities (Moore et al. 2000). Warm temperature, although not associated with the initiation of withering syndrome, is associated with increased mortality rates (Lafferty and Kuris 1993, Harvell et al. 2009). This has been observed from power plant effluent and incursions of warm water into traditionally temperate regions. In red abalone, higher rates of infection and more prominent signs of infection are associated with El Niño events (Moore et al. 2009). Interspecific competition has not been studied in black abalone, but blacklip abalone face significant competition for algal resources by sea urchins (Strain and Johnson 2009).

Compounding these factors are reproductive factors that further hamper species recovery. At low densities, individuals aggregate for spawning are not close enough for fertilization to occur. As a result, annual recruitment of juvenile black abalone has declined steeply since adult populations dropped below half of initial densities (Richards and Davis 1993).

Black abalone at various life stages experience predation from several species. Juvenile abalone hiding amongst rocks are food for crabs, lobsters, octopi, starfish, fish, and predatory snails (Haaker et al. 1986). Abalone of intermediate sizes are vulnerable to octopus and fish predation, particularly sheepshead and cabezon. As adults, black abalone are primarily preyed upon by sea otters, which can be major regulators of black abalone populations, but are not know to extirpate communities as other threats are known to (Braje et al. 2009, Johnson et al. 2009). Interactions with other species can hinder species recovery in other ways, namely competition for space and food resources. Purple and red sea urchins tend to feed on the same kelp and brown algae food as black abalone and, when in high abundance and food is plentiful, have the potential to out-compete abalone for food (Leighton 1968, Paine 1974, Tegner and Levin 1982, Tegner 1989, Miller and Lawrenz-Miller 1993). However, abalone tend to inhabit different habitats than these urchin species (CDFG 1993). Space competition may also occur between black abalone and sand castle worms. This species cements itself to the underside of rocks, the same habitat that black abalone seek for refuge (Connell et al. 1988). This could limit the habitat available for black abalone to recruit into during recovery. Other factors that can threaten black abalone include storms (crushing abalone between rocks and sedimentation in rocky habitat), fresh water input, sedimentation on gills leading to asphyxiation, and temperature impacts on reproduction and growth (Cox 1960, 1962).

Anthropogenic threats
Although commercial harvests historically lead to black abalone depletion, current harvesting is a small fraction of those levels. However, small removals are still significant in small populations.

Although toxicology of abalone is poorly known, red abalone have demonstrated metabolic breakdown when exposed to the pesticide 3-trifluoromethyl-4-nitrophenol (Viant et al. 2001). Silver, cadmium, and mercury are also known to bioaccumulate in abalone, likely from ingested algae (Huang et al. 2008). Update rates have been measured as 1.78 L g^{-1} d^{-1} for silver, 0.056 L g^{-1} d^{-1} for cadmium and 0.32 L g^{-1} d^{-1} for mercury, of which 58 to 83%, 33 to 59%, and 65 to 78%, respectively, is assimilated. Abalone are known to bioaccumulate high levels of heavy metals in the presence of high environmental concentrations (Wang et al. 2009).

Changes in sea surface temperatures have been suggested as a driving force in altering red abalone distribution in the past (Braje et al. 2009); it is unknown what affect, if any climate change may have on black abalone.

Critical habitat
On October 27, 2011, the NMFS designated critical habitat for black abalone This includes rocky areas from mean high water to six meters 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. These areas include primary constituent elements required by black abalone, such as rocky substrates to cling to, food resources (bacterial and diatom films, crustose coralline algae, and a source of detrital macroalgae), juvenile settlement habitat (rocky intertidal habitat containing crustose coralline algae and crevices or cryptic biogenic structures (e.g., urchins, mussels, chiton holes, conspecifics, anemones)), suitable water quality (temperature, salinity, pH, and other chemical characteristics necessary for normal settlement, growth, behavior, and viability of black abalone), and suitable nearshore circulation patterns (where sperm, eggs, and larvae are retained in the nearshore environment).

Johnson's seagrass

Description of the species
Johnson's seagrass has only relatively recently been identified as a distinct species and therefore no historical distribution information is available (Eiseman and McMillan 1980).

Distribution
Current distribution includes lagoons along approximately 125 miles of southeastern Florida between Sebastian Inlet and north Biscayne Bay which means that Johnson's seagrass has the most limited geographic distribution of any seagrass in the world (Kenworthy 1997). However, northern range extentions (likely temporary) have recently been observed (Virnstein and Hall

2009). The largest known groups of patches are located near Sebastian Inlet and Lake Worth.

Habitat

Patches of Johnson's seagrass have been observed to grow from the intertidal zone down to 3.3 feet water depth and in waters with variable temperatures and salinities (15 to 43 parts per thousand) and temperatures (Dawes et al. 1989, Kenworthy 1993, Virnstein et al. 1997, Kahn and Durako 2009). Patches near freshwater discharges have been observed (Gallegos and Kenworthy 1996), although Torquemada et al (2005) noted that highly hypo- or hypersaline conditions can negatively impact growth. Intertidal patches may be completely exposed at low tides, suggesting tolerance to desiccation and wide temperature ranges (Kahn and Durako 2009).

Growth and reproduction

Only female flowers have been observed; no fruit or seeds have been found to date (Eiseman and McMillan 1980, Heidelbaugh et al. 2000). Meiosis does occur however, meaning that if male pollen ware even rarely present, sexual reproduction could take place (York 2005). However, there is no evidence of male flowers, meaning Johnson's seagrass may only reproduce by cloning or asexual branching and fragmentation (Jewitt-Smith et al. 1997, Hammerstom and Kenworthy 2003). Consequently, genetic diversity is low (Freshwater and York 1999).

Clonal reproduction in plants occurs when plants form new leaf-pair, root and rhizome segments that arise from terminal buds (Posluszny and Tomlinson 1990). On average, new buds are formed on rhizomes every two to four days and rhizomes can grow at 0.2 inch per day (Bolen 1997, Kenworthy 1997). However, these clones can expand rapidly (1 to 3 feet per month) during periods of prolific branching (Kenworthy 1997, Greening and Holland 2003, Kenworthy 2003). As clones expand, high density "patches" are formed ranging from three to 66 feet^2 in size (Kenworthy 1997, Virnstein et al. 1997, Kenworthy 2000, 2003, Virnstein and Morris 2007). Patches can expand rapidly (nine feet^2 per month)(Kenworthy 2003) leading to coalescence with adjacent patches and large meadows of up to 30 acres (Kenworthy 1997).

Fragments or entire plants can be uprooted and drift extensively, providing a mechanism for dispersal and colonization of new areas (Hall et al. 2006). Virnstein *et al.* (2009) recently proposed that Johnson's seagrass occurs in "pulsating patch," with two to three consecutive summers of growth followed by a rapid decline. Johnson's seagrass populations may undergo whole patch mortality followed by recolonization (Virnstein et al. 1997, Heidelbaugh et al. 2000, Greening and Holland 2003, Kenworthy 2003, Virnstein and Morris 2007).

Johnson's seagrass appears to be physiologically adapted to exploit unstable environments and unvegetated patches, with minimal resources allocated to the holding of space (Dean and Durako 2007). This characteristic may allow for more rapid overall patch growth and the exploitation of areas in which Johnson's seagrass could not otherwise compete (Dean and Durako 2007). These growth characteristics also help explain its patchy distribution (Kenworthy 1993, Virnstein et al. 1997). Although successful in unstable areas, Johnson's seagrass may be out-competed by more stable-selected plants in areas not subject to regular disturbance (Durako 2003). Johnson's seagrass thrive in unstable or newly-created unvegetated environments, but have little capacity for holding occupied space. As a result, Johnson's seagrass can be highly variable in its

occurrence over relatively short time frames (Virnstein et al. 2009). Due to this species' physiology, low capacity for storage, and shallow root system, growth over large unsuitable patches may be unlikely, and its ability to recover from widespread habitat loss may be limited.

Status and trends
On September 14, 1998, Johnson's seagrass was listed as threatened under the ESA (69 FR 49035).

Historical abundance estimates of Johnson's seagrass are not available due to the species having only recently been differentiated. Limited data indicate no large distributional gaps or changes in abundance over much of Johnson's seagrass distribution from 1994 to 1999. However, recent increases in reported occurrence could be an artifact of recent increases in search efforts.

Natural threats
Storms can easily uproot or rip apart individuals and seagrass beds. Although this can serve to disperse individuals into new habitats, it can also disturb established meadows. Subsequent siltation following high turbidity events can also bury individuals or parts of plants.

Anthropogenic threats
Due to its delicate morphology, endemic range, lack of genetic diversity and a physiology ill equipped to hold space and compete with other seagrasses, Johnson's seagrass is vulnerable to prolonged widespread human-induced disturbance and habitat loss and its potential for recovery may be limited. Johnson's seagrass and its habitat are threatened by several natural and anthropogenic factors, including (1) dredging and filling, (2) construction and shading from in- and overwater structures, (3) boat propeller and anchor scarring, (4) trampling, (5) altered water quality (such as stormwater runoff and turbidity), and (6) siltation, as well as multiple deterious effects of climate change (Waycott et al. 2009).

Critical habitat
Critical habitat for Johnson's seagrass was designated on April 5, 2000 (65 FR 17786) and includes (1) locations with populations that have persisted for 10 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; and (5) locations with a documented high abundance of Johnson's seagrass compared to other areas in the species' range. These PCEs are critical to the conservation of the species because they protect persistently reproductive and genetically diverse populations, allow for protective buffers along the distribution limits (i.e., edges of survival), and protect regions of high density that without further knowledge of species biology, appear to serve the needs of Johnson's seagrass. Ten regions of sheltered bay and inlet waters are designated, including north and south of Sebastian Inlet, near Fort Pierce Inlet, north of St. Lucie Inlet, a portion of Hobe Sound, the southern side of Jupiter Inlet, Lake Worth Lagoon (north of Bingham Island and Boynton Inlet), waters of Lake Wyman, and wide areas of northern Biscayne Bay. These regions occupy approximately 22,574 acres or 9,139 hectares. Simply the nature of Johnson's seagrass critical habitat makes it variable and prone to change. The growth of boating in Florida and development of coastal areas has resulted in trampling, propeller scarring, dredging, filling, shading, and altered water quality that has degraded these

areas compared to historical conditions. Although many of the factors that can negatively affect Johnson's seagrass and their habitat are generally well regulated and enforced, the species is still under threat from high development pressure and subsequent habitat degradation throughout its range.

Environmental Baseline

By regulation, environmental baselines for Biological Opinions include the past and present impacts of all state, Federal or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early Section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process (50 CFR 402.02). The environmental baseline for this Biological Opinion includes the effects of several activities that affect the survival and recovery of endangered and threatened in the Action Area.

A number of human activities have contributed to the current status of populations of endangered and threatened species in the action area. Some of those activities, most notably commercial whaling, occurred extensively in the past, ended, and no longer appear to affect populations of these species, although the effects of these reductions likely persist today. Other human activities are ongoing and appear to continue to affect populations of endangered and threatened species in the Action Area. The following discussion summarizes the principal phenomena in the Action Area that are known to affect the likelihood that these endangered and threatened species will survive and recover in the wild.

Because this is a programmatic consultation on a continuing action with a geographic scope that encompasses all waters of the United States, its territories and possessions, this *Environmental Baseline* serves a slightly different purpose. First, because this is both a programmatic and national consultation that does not assess the consequences of EPA's proposed VGPs for specific sites or listed resources that occur at those sites, this *Environmental Baseline* focuses primarily on the status and trends of the aquatic ecosystems in the United States and the consequences of that status for listed resources. As such, this *Environmental Baseline* elaborates on some of the narratives we have already presented in our treatments of the *Status of Listed* resources by focusing on the status and trends of waters of the United States at a national scale.

We begin this *Environmental Baseline* with a discussion of the status and trends of waters of the United States. Because the status and trends of those waters, and the endangered and threatened species that depend on them, has been strongly influenced by wetland ecosystems in the United States, we summarize the status and trends of wetlands in the United States as a second step. We summarize large-scale impacts to coastal and wetland habitat, including aquatic nuisance species, pollutants, and the impacts of global climate change. Then we summarize the effect of Federal programs designed to protect and restore waters of the United States. We conclude by integrating and synthesizing this information to assess the effect of these programs on

endangered species, threatened species, and designated critical habitat.

The Changing Landscapes of the United States

The continental United States has a land area of about 2.3 billion acres. Since colonial times, the landscapes of the United States reflect the abundance, distribution and economics of the human population. In 1790, the United States had a resident population that was slightly less than 4 million and a population density of 4.5 people per square mile. By 2010, that population had grown to slightly more than 308 million people and the population density had increased to 85 per square mile. Most of the population growth in the United States occurred in urban areas, first in central cities and later suburbs. In 1910, three times more Americans lived in central cities than in suburban areas; by 1970, slightly more Americans lived in suburban areas than in either cities or rural areas. From 1950 to 1996, the urban population increased by 63 percent, the rural population decreased by 19 percent, and the greatest relative change occurred in the suburban population, an increase of 274 percent.

Other human activities that have altered the landscapes of the United States include agricultural practices that include land conversion, sod busting, and applications of pesticides; forest practices that include timber harvests, silviculture, and the construction of logging roads; mining practices that include open-pit mining, mountain-top mining, placer mining, heap-leach mining, and removal of overburden materials; road construction practices that include alteration of land in the right of way, spraying herbicides to maintain the right of way, and construction of quarries for source materials; civil works projects that include canals, drainage ditches, projects to deliver water to arid lands in the western States, projects to drain wetlands in southeastern States, projects to control flooding in mid-western and eastern States, port construction, projects to maintain shipping channels, and the construction of more than 8,100 major dams on rivers and streams in the United States, Puerto Rico, and the U.S. Virgin Islands.

The direct and indirect effects of these changes in land-use and land-cover change have had a lasting effect on the quantity, quality and distribution of every major terrestrial, aquatic, and coastal ecosystem in the United States, its territories and possessions. By the mid-1990s, at least 27 types of ecosystems had declined by more than 98 percent (Noss and Murphy 1995). Aquatic and semi-aquatic ecosystems have not fared much better than terrestrial ecosystems. Between the 1780s and 1980s, 30 percent of the nation's wetlands had been destroyed, including 74 percent of the wetlands in Connecticut, 73 percent of the wetlands in Maryland, 52 percent of the wetlands in Texas, 91 percent of all wetlands in California, including 94 percent of all inland wetlands (Dahl 1990). From 1982 to 1987, the wetland area throughout the conterminous United States declined by 1.1 percent and the expansion of urban – suburban metropolitan areas accounted for 48 percent of this decline (Brady and Flather 1994).

Because of these changes in land use, many of the native plant and animal species that inhabited those native ecosystems over the past have become extinct or extinct in the wild over the past 200 years. The last passenger pigeon, a species that once numbered in the billions and covered most of the eastern and mid-western United States, became extinct in 1912. In the same year, the

Louisiana parakeet (*Conuropsis carolinensis ludoviciana*) became extinct followed two years later by the extinction of its relative, the Carolina parakeet (*C. c. carolinensis*). The heath hen became extinct in the mid-1920s, the June sucker (*Chasmistes liorus liorus*) in the mid-1930s, Tecopa pupfish (*Cyprinodon nevadensis calidae*) in the early 1940s, and Ash Meadows killifish (*Empetrichthys merriami*) and Thicktail chub (*Gila crassicauda*) in the 1950s. Over the past 200 years, a substantial portion of the bird fauna of the Hawaiian islands -including the Oahu akepa, Kona finch, Lanai creeper, black mamo, and Hawai'i o'o- became extinct combined with the extinction of substantial portions of the freshwater mussel fauna of the Mississippi, Ohio, and Tennessee Rivers and regional extirpations of the flora and fauna of California, Florida, Oregon, Puerto Rico, and the desert states.

Status and Trends of Waters of the United States

All of the endangered and threatened species and designated critical habitat under the jurisdiction of NMFS depend on the health of aquatic ecosystems for their survival. These species were listed as endangered or threatened, at least in part, because of the consequences of human activities on the aquatic ecosystems -the estuaries, rivers, lakes, streams, and associated wetlands, floodplains, and riparian ecosystems- of the United States, its Territories and possessions. The status and trends of those aquatic ecosystems determines the status and trends of these species and the critical habitat that has been designated for them.

Over the past 30 to 40 years, the nation's aquatic ecosystems have improved substantially. In particular, pollution from point sources has been significantly reduced over the past 35 years. Sewage and industrial discharges into aquatic ecosystems have been controlled and some agricultural pesticides have been restricted or banned. Programs like the Conservation Reserve Program have taken highly erodible lands out of production. Despite this progress, however, many aquatic ecosystems remain highly polluted. Of the waters bodies they assessed -39 percent of the river and stream miles, 46 percent of the lake area and 51 percent of the estuarine area- one or more designated uses are impaired. Non-point pollution from urban and agricultural land (e.g. siltation, nutrients, bacteria, metals and oxygen depleting substances) that is transported by precipitation and runoff was the primary cause of the impairment.

These water quality problems, particularly the problem of non-point sources of pollution, have resulted from the changes humans have imposed on the landscapes of the United States over the past 100 – 200 years. One way of relating these changes in water quality to land uses relies on the surface area of a watershed that is covered by porous versus impervious surfaces. Most land areas that are covered by natural vegetation are highly porous and have very little sheet flow; precipitation falling on these landscapes infiltrates the soil, is transpired by the vegetative cover or evaporates. The increased transformation of the landscapes of the United States into a mosaic of urban and suburban land uses has increased the area of impervious surfaces -roads, rooftops, parking lots, driveways and sidewalks- in those landscapes.

The amount of impervious surface in a watershed is a reliable indicator of a suite of phenomena that influence a watershed's hydrology (Center for Watershed Protection, 2003). Above certain thresholds, landscapes with impervious surfaces respond to precipitation differently than other

land-uses: rain that would normally infiltrate in forest, grassland and wetland soils falls on and flows over impervious surfaces. That runoff is then channeled into storm sewers and released directly into surface waters (rivers and streams), which changes the magnitude and variability of water velocity and volume in those receiving waters.

Aquatic Nuisance Species

Aquatic nuisance species (ANS) are aquatic and terrestrial organisms, introduced into new habitats throughout the United States and other areas of the world, that produce harmful impacts on aquatic natural resources in these ecosystems and on the human use of these resources (http://www.anstaskforce.gov). They are also referred to as invasive, alien, or nonindigenous species. Introduction of these species is cited as a major threat to biodiversity, second only to habitat loss (Wilcove et al. 1998). They have been implicated in the endangerment of 48% of the species listed under ESA (Czech and Krausman 1997). In Hawai'i, 343 marine aquatic invasive species have been documented. Over 250 nonindigenous species of invertebrates, algae, and microorganisms have established themselves in the coastal marine ecosystems of California, whose waters have been the subject of most in-depth analyses of aquatic invasions in the U.S. More than 180 invaders have been detected and described in the Great Lakes, which are among the best-studied freshwater systems in the world.

Climate Change

All species discussed in this Opinion are or will be threatened by the direct and indirect effects of global climatic change. There is now widespread consensus within the scientific community that atmospheric temperatures on earth are increasing (warming) and that these increases will continue for at least the next several decades (IPCC 2001b). The Intergovernmental Panel on Climate Change (IPCC) estimated that average global land and sea surface temperature has increased by 0.6°C (± 0.2) since the mid-1800s, with most of the change occurring since 1976. This temperature increase is greater than what would be expected given the range of natural climatic variability recorded over the past 1,000 years (Crowley and Berner 2001). The IPCC reviewed computer simulations of the effect of greenhouse gas emissions on observed climate variations that have been recorded in the past and evaluated the influence of natural phenomena such as solar and volcanic activity. Based on their review, the IPCC concluded that natural phenomena are insufficient to explain the increasing trend in land and sea surface temperature, and that atmospheric warming observed over the last 50 years is probably attributable to human activities (IPCC 2001b). Climatic models estimate that global temperatures would increase between 1.4 to 5.8°C from 1990 to 2100 if humans do nothing to reduce greenhouse gas emissions from current levels (IPCC 2001b).

In the Northeast, annual average temperatures have increased by 2°F since 1970, with winter temperatures increasing by up to 4°F (Karl et al. 2009). Over the same time interval, the Northeast has experienced more days with temperatures greater than 90°F, a longer growing season, increased heavy precipitation, more winter precipitation falling as rain than as snow, reduced snowpack, earlier breakup of winter ice on lakes and rivers, earlier spring snowmelt resulting in earlier peak river flows, rising sea surface temperatures and sea level.

Over the next several decades, the Northeast is expected to experience temperatures increases of

another 2.5 to 4°F during the winter season and 1.5 to 3.5°F during the summer season as a result of carbon emissions that have already occurred (Burakowski et al. 2008, Karl et al. 2009). Forecasts beyond the middle of this century are sensitive to the level of carbon emissions produced today. If carbon emissions are not reduced, the length of the winter snow season would be cut in half across northern New York, Vermont, New Hampshire, and Maine, and reduced to a week or two in southern parts of the region; the Northeast would have fewer cold days during the winter and experience more precipitation (Hayhoe et al. 2007, Karl et al. 2009).

Cities in the Northeast that currently experience temperatures greater than 100°F for a few days each summer would experience an average of 20 days of such temperatures each summer; some cities in the Northeast -- Hartford, Connecticut, and Philadelphia, Pennsylvania, for example -- would experience an average of 30 days of such temperatures each summer (Karl et al. 2009). Hot summer conditions would arrive three weeks earlier and last three weeks longer into the fall. Droughts lasting from one- to three-months are projected to occur as frequently as once each summer in the Catskill and Adirondack Mountains, and across the New England states. Finally, sea levels in this region are projected to rise more than the global average, which would increase coastal flooding and coastal erosion (Kirshen et al. 2008, Karl et al. 2009).

In the Pacific Northwest, annual average temperatures have increased by about 1.5°F over the past century with some areas experiencing increases of up to 4°F (Karl et al. 2009, Littell et al. 2009, Elsner and Hamlet 2010). Higher temperatures during the cool season (October through March) have caused more precipitation to fall as rain rather than snow and contribute to earlier snowmelt. The amount of snowpack remaining on April 1, which is a key indicator of natural water storage available for the warm season, has declined substantially throughout the Northwest region. In the Cascade Mountains, for example, the snowpack remaining on April 1 declined by an average of 25 percent over the past 40 to 70 years; most of this decline is attributed to the 2.5°F increase in temperatures during the winter season over the time interval (Payne et al. 2004, Christensen et al. 2007).

Over the next century, average temperatures in the Northwest Region are projected to increase by another 3 to 10°F, with higher emissions scenarios resulting in warming in the upper end of this range (Christensen et al. 2007, Karl et al. 2009). Increases in winter precipitation and decreases in summer precipitation are projected by many climate models, though these projections are less certain than those are for temperature.

There is consensus within the scientific community that warming trends will continue to alter current weather patterns and patterns of natural phenomena that are influenced by climate, including the timing and intensity of extreme events such as heat-waves, floods, storms, and wet-dry cycles. Oceanographic models project a weakening of the thermohaline circulation resulting in a reduction of heat transport into high latitudes of Europe, an increase in the mass of the Antarctic ice sheet, and a decrease in the Greenland ice sheet, although the magnitude of these changes remain unknown (Schmittner et al. 2005, Levermann et al. 2007). As ice melts in the Earth's polar regions in response to increases in temperature, increases in the distribution and abundance of cold water are projected to influence oceanic currents, which would further alter weather patterns. In addition to influencing atmospheric temperatures and weather patterns,

increases in greenhouse gases in the Earth's atmosphere have begun to increase rates of carbon capture and storage in the oceans: as carbon dioxide levels in the oceans increase, the waters will become more acidic, which would affect the physiology of large marine animals and cause structures made of calcium carbonate (for example, corals) to dissolve (IPCC, 2001; Royal Society of London, 2005).

Climate change is projected to have substantial direct and indirect effects on individuals, populations, species, and the structure and function of marine, coastal, and terrestrial ecosystems in the foreseeable future (McCarthy et al. 2001, Parry et al. 2007). Climate-mediated changes in the global distribution and abundance are expected to reduce the productivity of the oceans by affecting keystone prey species in marine ecosystems such as phytoplankton, krill, and cephalopods.

Increasing atmospheric temperatures have already contributed to changes in the quality of the freshwater, coastal and marine ecosystems that are essential to the survival and recovery of salmon populations and have contributed to the decline of populations of endangered and threatened species (Mantua et al. 1997, Karl et al. 2009, Littell et al. 2009). Since the late 1970s, sea surface temperatures have increased and coastal upwelling -which is recognized as an important mechanism governing the production of both phytoplankton and zooplankton- has decreased resulting in reduced prey availability and poorer marine survival of Pacific salmon. Changes in the number of Chinook salmon escaping into the Klamath River between 1978 and 2005 corresponded with changes in coastal upwelling and marine productivity and the survival of Snake River spring/summer Chinook salmon and Oregon coho salmon has been predicted using indices of coastal ocean upwelling (Karl et al. 2009, Littell et al. 2009, Elsner and Hamlet 2010). The majority (90%) of year-to-year variability in marine survival of hatchery reared coho salmon between 1985 and 1996 can be explained by coastal oceanographic conditions.

Changes in temperature and precipitation projected over the next few decades are projected to decrease snow pack, affect stream flow and water quality throughout the Pacific Northwest region (Stewart et al., 2004, Knowles et al. 2006, Mote et al. 2008, Rauscher et al. 2008). Warmer temperatures are expected to reduce snow accumulation and increase stream flows during the winter, cause spring snowmelt to occur earlier in the year causing spring stream flows to peak earlier in the year, and reduced summer stream flows in rivers that depend on snow melt (most rivers in the Pacific Northwest depend on snow melt). As a result, seasonal stream flow timing will likely shift significantly in sensitive watersheds (Littell et al. 2009).

The States of Idaho, Oregon, and Washington, are likely to experience increased forest growth over the next few decades followed by decreased forest growth as temperature increases overwhelm the ability of trees to make use of higher winter precipitation and higher carbon dioxide. In coastal areas, climate change is forecast to increase coastal erosion and beach loss (caused by rising sea levels), increase the number of landslides caused by higher winter rainfall, inundate areas in southern Puget Sound around the city of Olympia, Washington (Littell et al. 2009).

Table 2. Phenomena associated with projections of global climate change (adapted from IPCC 2001 and Campbell-Lendrum Woodruff 2007).

Phenomenon	Confidence in Observed Changes (observed in the latter 20th Century)	Confidence in Projected Changes (during the 21st Century)
Higher maximum temperatures and a greater number of hot days over almost all land areas	Likely	Very likely
Higher minimum temperatures with fewer cold days and frost days over almost all land areas	Very likely	Very likely
Reduced diurnal temperature range over most land areas	Very likely	Very likely
Increased heat index over most land areas	Likely over many areas	Very likely over most areas
More intense precipitation events	Likely over many mid- to high-latitude areas in Northern Hemisphere	Very likely over many areas
Increased summer continental drying and associated probability of drought	Likely in a few areas	Likely over most mid-latitude continental interiors (projections are inconsistent for other areas)
Increase in peak wind intensities in tropical cyclones	Not observed	Likely over some areas
Increase in mean and peak precipitation intensities in tropical cyclones	Insufficient data	Likely over some areas

Rising stream temperatures will likely reduce the quality and extent of freshwater salmon habitat. The duration of periods that cause thermal stress and migration barriers to salmon is projected to at least double by the 2080s for most analyzed streams and lakes (Littell et al. 2009). The greatest increases in thermal stress (including diseases and parasites which thrive in warmer waters) would occur in the Interior Columbia River Basin and the Lake Washington Ship Canal. The combined effects of warming stream temperatures and altered stream flows will very likely reduce the reproductive success of many salmon populations in Washington watersheds, but impacts will vary according to different life-history types and watershed-types. As more winter precipitation falls as rain rather than snow, higher winter stream flows scour streambeds, damaging spawning nests and washing away incubating eggs for Pacific Northwest salmon. Earlier peak stream flows flush young salmon from rivers to estuaries before they are physically mature enough for transition, increasing a variety of stressors including the risk of being eaten by predators.

As a result of these changes, about one third of the current habitat for either the endangered or threatened Northwest salmon species will no longer be suitable for them by the end of this century as key temperature thresholds are exceeded (Littell et al. 2009). As summer temperatures increase, juvenile salmon are expected to experience reduced growth rates, impaired smoltification and greater vulnerability to predators.

Ocean acidification caused by increasing amounts of carbon dioxide (CO2) in the Earth's atmosphere poses a more wide-spread threat because virtually every major biological function has been shown to respond to acidification changes in seawater, including photosynthesis, respiration rate, growth rates, calcification rates, reproduction, and recruitment (London 2005, Smith 2008).

At the same time as these changes in regional weather patterns and ocean productivity are expected to occur, the oceans are expected to being increasingly acidic. Over the past 200 years, the oceans have absorbed about half of the CO2 produced by fossil fuel burning and other human activities. This increase in carbon dioxide has led to a reduction of the pH of surface seawater of 0.1 units, equivalent to a 30 percent increase in the concentration of hydrogen ions in the ocean. If global emissions of carbon dioxide from human activities continue to increase, the average pH of the oceans is projected to fall by 0.5 units by the year 2100 (Royal Society of London, 2005).

Although the scale of these changes would vary regionally, this resulting pH would be lower than the oceans have experienced over at least the past 420,000 years and the rate of change is probably one hundred times greater than the oceans have experienced at any time over that time interval. More importantly, it will take tens of thousands of years for ocean chemistry to return to a condition similar to that occurring at pre-industrial times (Royal Society of London, 2005).

Marine species such as fish, larger invertebrates, and some zooplankton take up oxygen and lose respired carbon dioxide through their gills. Increased carbon dioxide levels and decreased pH would have a major effect on this respiratory gas exchange system because oxygen is much harder to obtain from surface seawater than it is from air (primarily because concentrations of oxygen are lower in water). The processes involved in supplying oxygen to the gills means that more carbon dioxide is removed from these aquatic animals than is removed from air breathing animals of a similar size. This more ready removal of carbon dioxide from body fluids means that the level and range of CO_2 concentration in the bodies of water-breathing animals are much lower than is the case for air-breathing animals. As a result, large water breathing marine animals are more sensitive to changes in the carbon dioxide concentration in the surrounding seawater than are large air-breathing animals.

This has important implications because higher ambient levels of carbon dioxide would acidify the body tissues and fluids of these species and affect the ability of their blood to carry oxygen. Experimental studies have demonstrated that acidosis of tissues decrease cellular energy use, lower respiratory activity, and lower rates of protein synthesis (Pörtner et al. 2000, Pörtner et al. 2004)(Pörtner *et al* 2000, 2004). These changes would reduce the performance of almost every physiological process of larger animals including their growth and reproduction (Langenbuch and Pörtner 2002, 2003). By itself, this effect of climate change poses severe risks for endangered and threatened anadromous and marine species. In combination with changes in seasonal temperatures, formation of snow pack in terrestrial ecosystems, upwelling phenomena, and ocean productivity, ocean acidification would lead us to expect the status of endangered and threatened anadromous, coastal, and marine species to trend toward increasing decline over the next three or four decades.

In addition to these changes, climate change may affect agriculture and other land development as rainfall and temperature patterns shift. Aquatic nuisance species invasions are also likely to change over time, as oceans warm and ecosystems become less resilient to disturbances.

Clean Water Act

The Federal Water Pollution Control Act, or Clean Water Act (33 U.S.C §1251 et seq.), is the principal law concerned with polluting activity in streams, lakes and estuaries in the United States. The original 1948 statute was totally rewritten in 1972 (P. L. 92-500) to produce its current purpose: "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters." Congress made substantial amendment to the Clean Water Act in the Water Quality Act of 1987 (P. L. 100-4) in response to the significant and persistent water quality problems.

The Clean Water Act uses two primary approaches to achieve its goal. The first approach uses regulations to achieve a goal of zero discharge of pollutants into waters of the United States. The second approach provides Federal technical assistance for municipal wastewater treatment construction. Both approaches are supported by research activities, permits and provisions for enforcement.

To achieve its objectives, the Clean Water Act prohibits all discharges into the nation's waters, unless they are specifically authorized by a permit. For example, the National Pollutant Discharge Elimination System or NPDES program regulates discharges of pollutants like bacteria, oxygen-consuming materials, and toxic pollutants like heavy metals, pesticides, and other organic chemicals.

Most of these Federal programs are administered by EPA, while state and local governments have the principal day-to-day responsibility for implementing the law. Nonpoint sources of water pollution, which are believed to be responsible for the majority of modern water quality problems in the United States, are not subject to Clean Water Act permits or the regulatory requirements. Instead, non-point sources of pollution are regulated by State programs.

Puget Sound as an Example of the Impact

Puget Sound provides and illustrative example of the impacts of the environmental baseline on endangered and threatened species under NMFS' jurisdiction. Between 2000 and 2006, counties in Puget Sound increased by 315,965 people or by more than 50,000 people per year, with associated increases in the area of impervious surface and population density per square mile of impervious surface in the Puget Sound region (Puget Sound Action Team 2007). Between 1991 and 2001, the area of impervious surface in the Puget Sound basin increased 10.4 percent (Puget Sound Action Team, 2007). By 2001, impervious surface covered 7.3 percent of the Puget Sound region below 1,000 feet elevation; in some counties and watersheds in the region, this area was substantially higher.

Over the same time interval, about 190 square miles of forest (about 2.3 percent of the total forested area of the Puget Sound basin) was converted to other uses. In areas below 1,000 feet

elevation, the change was more dramatic: 3.9 percent of total forest area was converted to other uses. By 2004, about 1,474 fresh and marine waters in Puget Sound were listed as "impaired waters" in Puget Sound. Fifty-nine percent of these waters tested were impaired because of toxic contamination, pathogens, low dissolved oxygen or high temperatures. Less than one-third of these impaired waters have cleanup plans in place. Chinook salmon from Puget Sound have 2-to-6 times the concentrations of PCBs in their bodies as Chinook salmon from other populations on the Pacific Coast. Because of this contamination, the Washington State Department of Health issued consumption advisories for Puget Sound chinook (Puget Sound Action Team, 2007). The quality of water in the Puget Sound Basin and aquatic biota those water support have been affected by a range of forestry, agricultural, and urban development practices. The chemical quality of surface water in the foothills and mountains is generally suitable for most uses. However, the physical hydrology, water temperature and biologic integrity of streams have been influenced to varying degrees by logging (Black and Silkey 1998).

Because of development, many streams in the Puget Lowlands have undergone changes in structure and function with a trend toward simplification of stream channels and loss of habitat (Black and Silkey, 1998). Sources of contaminants to lowland streams and lower reaches of large rivers are largely nonpoint because most major point sources discharge directly to Puget Sound. Compared with that in small streams in the Puget Lowlands, the quality of water in the lower reaches of large rivers is better because much of the flow is derived from the forested headwaters.

More than half of the agricultural acreage in the basin is located in Whatcom, Skagit and Snohomish Counties. Agricultural land use consists of about 60 percent cropland and 40 percent pasture. Livestock produce a large amount of manure that is applied as fertilizer to cropland, some- times in excess amounts, resulting in runoff of nitrogen and phosphorus to surface water and leaching of nitrate to ground water. Runoff from agricultural areas also carries sediment, pesticides, and bacteria to streams (Staubitz et al. 1997). Pesticides and fumigant-related compounds are present, usually at low concentrations, in shallow ground water in agricultural areas.

Heavy industry is generally located on the shores of the urban bays and along the lower reaches of their influent tributaries, such as Commencement Bay and the Puyallup River in Tacoma and Elliott Bay and the Duwamish Waterway in Seattle. High-density commercial and residential development occurs primarily within and adjacent to the major cities. Development in recent years has continued around the periphery of these urban areas but has trended toward lower density. This trend has resulted in increasing urban sprawl in the central Puget Sound Basin. Urban land-use activities have significantly reduced the quality of streams in the Puget Sound Basin (Staubitz and others, 1997). Water-quality concerns related to urban development include providing adequate sewage treatment and disposal, transport of contaminants to streams by storm runoff, and preservation of stream corridors. Water availability has been and will continue to be a major, long- term issue in the Puget Sound Basin. It is now widely recognized that ground-water withdrawals can deplete streamflows (Morgan and Jones 1999), and one of the increasing demands for surface water is the need to maintain instream flows for fish and other aquatic biota. Chinook salmon from Puget Sound have 2-to-6 times the concentrations of PCBs in their bodies

as other Chinook salmon populations on the Pacific Coast. Because of this contamination, the Washington State Department of Health has issued consumption advisories for Puget Sound Chinook (Puget Sound Action Team, 2007). Nevertheless, between 2000 and 2006, counties in Puget Sound counties increased by 315,965 people or by more than 50,000 people per year, with associated increases in impervious surfaces and population density per square mile of impervious surface (Puget Sound Action Team, 2007).

Pollutants founds in Puget Sound Chinook salmon have found their way into the food chain of the Sound. Harbor seals in southern Puget Sound, which feed on Chinook salmon, have PCB levels that are seven times greater than those found in harbor seals from the Georgia Basin. Concentrations of polybrominated diphenyl ether (also known as PBDE, a product of flame retardants that are used in household products like fabrics, furniture, and electronics) in seals have increased from less than 50 parts per billion in fatty tissue to more than 1,000 ppb over the past 20 years (Puget Sound Action Team, 2007).

Water quality appears poised to have larger-scale effects on the marine ecosystem of the Puget Sound Georgia Basin as evidenced by the intensity and persistence of water stratification in the basin. Historically, Puget Sound was thought to have an unlimited ability to assimilate waste from cities, farms and industries in the region and decisions about human occupation of the landscape were based on that belief. More recent data suggests that the marine ecosystems of the basin have a much more limited ability to assimilate pollution, particularly in areas such as Hood Canal, south Puget Sound, inner Whidbey basin and the central Georgia Basin. In these areas, as strong stratification has developed and persisted, the respective water quality has steadily decreased. As waters become more stratified, through weather, climate or circulation changes, they become even more limited in their ability to assimilate pollution.

The presence of high levels of persistent organic pollutants, such as PCB, DDT, and flame–retardants have also been documented in southern resident killer whales (Herman et al. 2005, Ross 2006, Ylitalo et al. 2008). Although the consequences of these pollutants on the fitness of individual killer whales and the population itself remain unknown, in other species these pollutants have been reported to suppress immune responses (Kakuschke and Prange 2007), impair reproduction and exacerbate the energetic consequences of physiological stress responses when they interact with other compounds in an animal's tissues (Martineau 2007). Because of their long life span, position at the top of the food chain, and their blubber stores, killer whales would be capable of accumulating high concentrations of contaminants.

Ambient noise is background noise in the environment. Several authors have reported that ambient noise levels in the northeast Pacific Ocean increased between the mid-1960s, the mid-1990s and the early 2000s. Andrew *et al*. (2002) reported that ambient sound levels increased by about 10 dB in the frequency ranges between 20 and 80 Hz and 200 and 300 Hz between the period from 1963 to 1965 and 1994 to 2001. In the frequency range between 200 and 300 Hz, ambient sound levels increased by about 3 dB. Since the 1960s, ambient noise in the 30–50 Hz band has increased by 10–12 dB, with most of this increase resulting from changes in commercial shipping (McDonald et al. 2006) and increases in whale song (Andrew *et al*. 2002). Many researchers have described behavioral responses of marine mammals to the sounds

produced by helicopters and fixed-wing aircraft, boats and ships, as well as dredging, construction, geological explorations, etc. (Richardson *et al,* 1995).

Integration and Synthesis of the Environmental Baseline

The direct and indirect effects of changes in land-use and land-cover of the United States have had lasting effect on the quantity, quality, and distribution of every major terrestrial, aquatic, and coastal ecosystems. Many native ecosystems exist as small isolated fragments surrounded by expanses of urban and suburban landscapes or "natural" areas that are dominated by non-native species. As a result, many of the native plant and animal species that inhabited those native ecosystems over the past have become extinct, extinct in the wild, endangered, or threatened over the past 200 years.

Beginning in the 1960s, a wide variety of programs undertaken by Federal, State, and local governments, non-governmental organizations, and private individuals have been established to protect or restore our nation's forests, grasslands, wetlands, estuaries, rivers, lakes, and streams. Those programs have helped slow and, for many ecosystems, reverse declining trends that began in the past. However, those efforts have benefited some ecosystems and their associated flora and fauna more than other ecosystems. Despite the efforts of agencies at every level of government, non-governmental organizations, and private individuals, non-point sources of pollution and ANS invasions still degraded our rivers, lakes, and streams; freshwater aquifers in coastal areas remain at risk from saltwater intrusion because of water withdrawals; nutrients transported down the Mississippi River remains sufficient to produce an hypoxic zone in the Gulf of Mexico that had more than doubled in size; and the acreage of wetland declined from slightly more than 274 million acres of wetlands to about 107.7 million acres between the 1980s and 2004 (Dahl 2006).

Water quality is important to all of the above listed resources, either directly, as for fishes and invertebrates, or indirectly, as for cetaceans that are affected through their diet. In some cases, detriments to water quality have led to the endangerment of species; in all cases, activities that threaten water quality also threaten these listed resources. Endangered and threatened species have experienced population declines that leave them vulnerable to a multitude of threats. Because of reduced abundance, low or highly variable growth capacity, and the loss of essential habitat, these species are less resilient to additional disturbances. For example, stressors that affect only a limited number of individuals were once tolerated with little impact to the species, whereas now, the same stressors are likely to reduce population viability. It is with this understanding of the environmental baseline that we consider the effects of the proposed action, including the likely effects that ANS and pollutants will have on endangered and threatened species and their designated critical habitat.

Effects of the Action

The *Effects of the Action* is the primary focus of an Opinion. In this section, we first describe the population of vessels covered under the VGPs. We then examine the consequences of exposing endangered and threatened species and designated critical habitat to the discharges authorized by EPA's proposed VGPs. We organize our analyses by general stressor type: ANS or pollutants. Under each section, we describe discharges authorized by the VGPs that are likely to produce either ANS or pollutant stressors. For example, ballast water, hull fouling, and chain locker effluent are grouped as discharges that carry ANS. We first characterize exposure of ESA-listed species to the discharges authorized by the VGPs; we also describe the actions EPA will take to prevent exposure of listed resources to authorized discharges. Next, we identify the physical, chemical, and biological stressors associated with vessel discharges. Then we review the literature describing the responses of threatened and endangered individuals, given exposure to these stressors. Finally, we summarize the risks posed to the continued existence of populations, species, or designated critical habitat. Our purpose is not to provide a comprehensive review of the probable responses of and consequences to endangered or threatened species; instead, our intention is to identify the range of representative responses and consequences we would expect given exposure to vessel discharges authorized by EPA's VGPs.

Vessel Populations Covered by the VGPs

The demographics of the vessels to be covered under the VGPs influence loading of stressors. Vessel size affects discharge volume. Vessel service class and age generally determine the type and performance of equipment used onboard vessels and therefore the characteristics of discharges from the equipment. Vessel age structure also indirectly indicates the implementation rate of build-date dependent protective measures over the 5-year permit period. Finally, distribution of vessel service classes among ports and harbors influences pollutant loading from VGP authorized discharges. The following paragraphs, describing the vessel service classes authorized to discharge to waters of the U.S. under the VGPs, are summarized in Table 3 and Table 4.

The types of vessels covered under the 2013 VGP include approximately 2,000 large commercial fishing vessels greater than 79 feet along with cruise ships, ferries, barges, mobile offshore drilling units, oil or petroleum tankers, bulk carriers, cargo ships, container ships, other cargo freighters, refrigerant ships, research vessels, emergency response vessels, including firefighting and police vessels, and any other vessels operating in a capacity as a means of transportation. Approximately half of the vessels covered under the sVGP are commercial fishing vessels involved in such activities as fish catching (e.g., longliner, shrimper, and trawler), fish processing, fishing tendering, and charter fishing. The other half are distributed among a variety of vessel types, including passenger vessels (e.g., water taxis, tour boats, harbor cruise ships, dive boats), utility vessels (e.g., tug/tow boats, research vessels, offshore supply boats), and freight barges (USEPA 2010b). These vessels and their relative abundance are described below.

VGP Vessels

EPA's Notices of Intent (NOI) database cannot be used alone to estimate the total number of vessels covered by the VGP Permit because only vessels greater than 300 gross tons or with the capacity to carry more than 8 cubic meters of ballast water were required to submit NOIs under the 2008 VGP. To estimate the population of domestically-flagged vessels subject to the VGP, EPA used two existing databases, the Marine Information for Safety and Law Enforcement (MISLE) and Waterborne Transportation Lines of the United States (WTLUS) databases, to create a master database identifying approximately 60,000 vessels eligible for coverage. Using the Foreign Traffic Vessel Entrances and Clearances (FTVEC) database[5], EPA estimated approximately 12,400 additional foreign-flagged vessels also subject to the VGP requirements, making a total estimated vessel population of 72,400 vessels. From these vessels, 43,431 active NOIs were submitted under the 2008 VGP as of August 2010. This represents about 60% of the estimated vessel population covered. The NOI form listed nine types of vessels under which the vessel owners/operators could classify their vessel along with an option of selecting vessel service class "Other," with space to designate vessel type in writing. In addition to the vessels identified in Table 4, about 10,000 were identified as "other." These include 9,839 vessels for which no vessel class was specified, while 2,044 others self-identified as some variety of cargo vessel, 143 as tug/tow boats, 115 as crew/supply/support vessels, 83 as chemical/oil tankers, 34 as barges, and 18 as passenger vessels. These vessels are integrated into the table of relative service class distribution and demographic properties for vessels submitting NOIs and with tonnage/ballast water capacity (Table 4).

The EPA-provided metrics in Table 3 include median[6] values for vessel gross tonnage, ballast capacity and maximum persons aboard along with the proportion of vessels with on board treatment and using antifouling coatings. In addition to these metrics, NMFS used the build dates reported in the NOI database to estimate the proportion of vessels within each service class that are 5 years of age or younger. This provides an index of the expected implementation rate of build date-dependent measures[7] within the permit.

Barges. Barges represent the greatest number of vessels permitted under the VGP, with 20,444 uniquely identified vessels. Barges are large, flat-bottomed boats typically used to move cargo in inland waterways. There are many types of barges, including hopper barges, chemical barges,

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[5] The FTVEC database provides information on foreign vessels entering or clearing U.S. Customs ports in calendar year 2008, the most recent year for which data are published (USACE, 2010).

[6] The median of a set of observations is the center value in the dataset where half of the observations are below and half are above. As a measure of central tendency, the median is robust against the influence of outliers and extreme values found in skewed data, that is, not distributed symmetrically around the mean. When the mean is greater than the median, it suggests the data are skewed such that it is influenced by a few larger values.

[7] The VGP requires biannual graywater monitoring for vessels with a maximum capacity to overnight 10 or more crewmembers and constructed on after December 19, 2013. In addition, all vessels constructed on or after December 19, 2013 must use an environmentally acceptable lubricant in all oil-to-sea interfaces. Older vessels technically able to use environmentally acceptable lubricants must do so and those which cannot, must report the use of unsafe lubricants in their Annual Report to EPA.

fuel barges, crane barges, dry bulk cargo barges, etc. Barges are usually not powered vessels, but are instead pushed or pulled by tugboats. Build dates for barges reported in the NOI indicate that 21% of barges are less than 5 years old. Because of their relatively small size (vessel tonnage), ballast capacity, and maximum persons aboard, most discharges incidental to the normal operation of these vessels are of limited concern. EPA included barges in its analysis due to the large number of barges in operation.

Oil and Gas Tankers. Oil and gas tankers represent the second greatest number of vessels permitted under the VGP, with 3,363 uniquely identified vessels. Oil tankers are designed to carry oil and other petroleum products in bulk tanks. Oil and gas tankers are large utility vessels, second only to large cruise ships in size (median tonnage equal to approximately 25,000 gross tons). Moreover, the median ballast water capacity of these vessels is very relatively large (24,000 m^3). Nearly all oil and gas tankers also use anti foulant coatings to reduce attachment of organisms to the vessel hull. Approximately half of these vessels (47%) are less than 5 years old. EPA included oil and gas tankers in their analysis because of their prevalence at certain ports, their overall size, their large ballast water capacity, likelihood they are engaged in long distance voyages, and their use of AFCs.

Large Commercial Fishing Vessels[8]. Large commercial fishing vessels are those greater than 79 feet in length that are engaged in the catching, taking, or harvesting of fish, crabs, lobster, shrimp, or other aquatic organisms for commercial sale. The MISLE[9] database (USCG 2009) indicates there are 2,326 commercial fishing vessels larger than 79 feet in length that are eligible for coverage under the VGP. The median ballast water capacity of these vessels is moderately high (1,388 m^3), as is the number of crew/passengers aboard. In addition, only 13 percent of the large commercial fishing vessels are less than 5 years old. Commercial fishing vessels generate fish hold and fish hold cleaning waste streams that are unique to commercial fishing vessels, and these waste streams have been shown to contain relatively high concentrations of conventional pollutants and nutrients (USEPA 2010b). Since most large commercial fishing vessels are concentrated in a limited number of home ports, and since fish holds are frequently discharged and cleaned by vessel crews between catches, EPA included large commercial fishing vessels in its analysis. (Note that industrial operations from vessels used as seafood processing facilities are not subject to the VGP because these operations are not incidental to the normal operation of a vessel engaged in a capacity of transportation).

Large Cruise Ships. Large cruise ships are the largest vessels (by size) covered under the VGP.

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[8] Public Law 110-299 subsequently extended by Public Law 111-215 provided for a temporary moratorium on NPDES permitting for all commercial fishing vessels. The moratorium on NPDES permitting for commercial fishing vessels expires on December 18, 2013 and therefore all commercial fishing vessels larger than 79 feet will be required to obtain permit coverage under the VGP in the event the moratorium is not extended.

[9] The U.S. Coast Guard's Marine Information for Safety and Law Enforcement (MISLE) database provides a wide range of information regarding vessel and facility characteristics, accidents, marine pollution incidents, and other pertinent information tracked by the U.S. Coast Guard. The USCG MISLE database provides information on a wide ensemble of vessels, including identification numbers, vessel category, size, area of operation, passenger and crew capacity, propulsion method, construction material and design, and year built or age.

Median vessel tonnage (53,526 gross tons) is approximately twice that of even the large oil and gas tankers. While the actual number of large cruise ships seeking coverage under the 2008 VGP is low (only 127 vessels), the number of crew and passengers aboard is relatively high - approximately 3,500 per vessel. EPA NOI data indicate that 20% of large cruise ships are less than 5 years old. Because of the large number of crew and passengers living on-board cruise ships, graywater generation can average nearly 600 m^3/day (158,500 gal/day). Median ballast water capacity of large cruise ships vessels is moderate (3,136 m^3). A high percentage of large cruise ships also use AFC. Moreover, since aesthetics are important to the industry, deck washdowns may be performed more frequently on cruise ships than other types of vessels. The above factors led EPA to include large cruise ships in its analysis.

Medium Cruise Ships. Medium cruise ships are those ships authorized to carry 100 to 499 passengers for hire and provide overnight accommodations to those passengers. Although only 22 medium cruise ships are included in EPA's NOI database, these vessels are relatively large, ranging in size from approximately 1,500 gross tons up to 5,000 gross tons, and have ballast capacities ranging from approximately 350 to 400 m^3. Only 14% of these vessels are less than 5 years old. With passenger and crew capacities up to 499 persons, medium cruise ships have the potential to generate more than 20,000 gallons/day of graywater. Untreated graywater from medium cruise ships has been shown to contain similar pollutants to those in untreated graywater discharges from large cruise ships (ADEC 2006, 2007). In addition, most medium cruise ships use AFC to prevent biofouling of the vessel hull.

Large Ferries. The NOI database includes 118 large ferries, 12 percent of which are less than 5 years old. Large ferries are vessels for hire that are designed to carry passengers and/or vehicles between two ports, usually in inland, coastal, or nearshore waters. For purposes of the VGP, large ferries are those ferries authorized to carry a) more than 100 tons of cars, trucks, trains, or other land-based transportation or b) 250 or more people. Large ferries usually travel the same route several times a day but do not provide overnight accommodations to their passengers. Because many ferries can travel the same route several times per day, they can discharge pollutants from a variety of sources (e.g., graywater, bilgewater, and seawater engine and exhaust cooling) to same water body. EPA included large ferries in its analysis due to these potential discharges.

Cargo [Bulk] and Other Carriers. Cargo carriers (e.g., bulk carrier, freighters, roll-on/roll-off, container ships) comprise the majority of the "Other" vessels classified in the NOI database. The attributes of this group of vessels are similar to those of the oil and gas tankers in terms of vessel size (median tonnage equal to approximately 16,000 gross tons) and median ballast water capacity (10,198 m^3). Approximately 61 percent of the owners/operators of these vessels reported using on-board treatment systems to treat the various waste streams aboard the vessels, and fewer use AFC. This is the youngest vessel population reporting in the NOI, with 56% of the vessels less than 5 years old. EPA included cargo carriers in its analysis due to their ballast water capacity and prevalence at certain ports.

Emergency and Research Vessels. The emergency and research vessels types include relatively few numbers of vessels and are relatively small in size, ballast capacity, and number of passengers/crew. Build date data indicate that 16% of research vessels are less than 5 years old

and only 3% of emergency vessels are less than 5 years old. Additionally, these vessels do not tend to engage on longer voyages where they can serve as a vector for invasive species (exceptions on voyage movements include large research vessels such as those operated by NOAA and the maritime services industry). Furthermore, the areas of operation for these vessels are not centrally located in any one home port or region. For these reasons, EPA considered discharges from these vessels to be discountable for purposes of the Endangered Species Act and does not consider that these vessel types warrant extensive analysis in the BE.

Table 3. Distribution of vessel service classes, equipment, and metrics among VGP covered vessels submitting NOIs and reporting vessel tonnage/ballast water capacity(USEPA 2012b).

Vessel Service Class	Estimated Number of Vessels	Median Gross Tonnage	Median Ballast Capacity (m^3)	Median Maximum Persons Aboard	% Vessels w/On-board Treatment	% Vessels Using Anti-fouling Hull Coatings	% Vessels 0-5 Years of Age
Barges	20,444[a]	705	206	4	1	4	21
Oil and Gas Tankers	3,363[b]	24,908	23,893	22	78	80	47
Large Commercial Fishing	2,326[c]	1,997	1,388	29	41	85	13
Large Cruise Ships	127	53,526	3,136	3,455	79	84	20
Large Ferries	118	1,451	404	559	45	86	12
Medium Cruise Ships	22	4,584	364	315	50	82	14
Bulk Cargo and Other Carriers	2,044[d]	15,939	10,198	25	61	65	56
Research Vessels	82	1,780	425	41	78	95	16
Emergency Vessels	56	563	332	16	43	96	3

[a] "Other" was selected for 34 NOIs and some variety of barge was designated.
[b] "Other" was selected for 83 NOIs and some variety of oil/chemical tanker was designated.
[c] Fishing vessels greater than 79 feet long, data from (USCG 2009)
[d] "Other" was selected for all 2,044 bulk cargo and other carriers with some variety of carrier specified. These were were dominated by: general cargo/not otherwise specified (91percent); car/truck carrier (3.6 percent); refrigerated cargo (2.3 percent); log/lumber/woodchip carrier (1.2 percent); and roll-on roll-off cargo, cement, grain/ore and ore/oil (cumulatively less than one percent) (USEPA 2012c).

sVGP Vessels

EPA estimates that between 115,000 and 138,000 vessels may be covered by the sVGP. The primary source of information used to characterize these vessels and their discharges is EPA's

Study Report to Congress: Study of Discharges Incidental to Normal Operation of Commercial Fishing Vessels and Other Non-recreational Vessels Less than 79 Feet (USEPA 2010b). EPA's BE and the Report to Congress do not provide the same level of detail on the distribution of ballast capacity, onboard treatment and the presence of antifouling coatings for sVGP vessels as was provided for VGP covered vessels. Details for sVGP eligible vessels are provided for hull material and horsepower (Table 4).

Approximately half of the sVGP eligible vessels are commercial fishing vessels involved in such activities as fish catching (111,094 vessels, e.g., longliner, shrimper, and trawler), fish processing, fishing tendering, and charter fishing. The other half are distributed among a variety of vessel types, including passenger vessels (34,609 vessels, e.g., water taxis, tour boats, harbor cruise ships, dive boats), utility vessels (18,743 vessels, e.g., tug/tow boats, research vessels, offshore supply boats), freight barges (11,596 vessels), freight ships (1,102 vessels) along with 35,578 vessels with unspecified service class (USCG 2009, USEPA 2010b). Recreational vessels are generally excluded because a separate act (the Clean Boating Act of 2008 (P.L. 110-288)) exempts discharges incidental to the normal operation of these vessels from NPDES permitting requirements. The Clean Boating Act defines recreational vessels as those that are either 1) manufactured or used primarily for pleasure or 2) leased, rented, or chartered to a person to use for pleasure. Furthermore, vessels that are subject to U.S. Coast Guard inspection and that are either engaged in commercial use or that carry paying passengers are not considered recreational vessels under the Clean Boating Act.

Hull Materials. According to the analysis in EPA's report to Congress, the three most common hull material types among sVGP eligible vessels are fiberglass, wood, and steel in order of most common usage. Commercial fishing vessels with wood hulls account for over three quarters of the total number of wood hulled vessels, although wood is also used in the hulls of a significant share of freight ships and passenger vessels less than 79 feet in length. The type of hull material affects the type of anti-foulant coatings that are applied and has implications on vessel discharges and receiving water quality. For example, steel hulls often have an anti-corrosive as well as anti-foulant hull coatings. The type of hull material may also affect the frequency with which certain maintenance procedures such as hull inspections are conducted(USEPA 2010b).

Vessels less than 5 years old. Vessel age, for the purposes of this Biological Opinion, is an indicator of the implementation rate for build date dependent measures specified by the VGPs. The sVGP requires that "All motorized vessels constructed on or after December 19, 2013 must have a functioning fuel-air separator or a fuel tank vent to prevent a fuel spill." In addition, for both VGPs, all vessels constructed on or after December 19, 2013 must use an environmentally acceptable lubricant in all oil-to-sea interfaces. Older vessels technically able to use environmentally acceptable lubricants must do so and those which cannot, must report the use of unsafe lubricants in their Annual Report to EPA.

Self-Propelled Vessels and Horsepower Ahead. Not all vessels covered by the sVGP are self-propelled; rather they are moved about by other vessels (e.g. tug boats, tow boats). Horsepower ahead represents the rated power of a vessel's engine in forward motion (as opposed to horsepower astern) and is expressed as the work accomplished per unit of time (e.g., 1 hp = 550 foot-pounds of work per second). This power is transferred to the propulsion mode (e.g., jet or

propeller) to create thrust and determines the vessel's speed at any given weight, or the weight that can be moved at any given speed. Vessel power rating may determine the amount and characteristics of discharges from operating vessels by affecting the size, type, and complexity of onboard propulsion equipment. Most sVGP eligible vessels have a horsepower rating between 100 and 500 hp, with fewer having horsepower ratings of 500 to 1,000 hp.

Table 4. Vessel metrics for sVGP eligible vessels with specified length less than 79 (USEPA 2012b).

Vessel Service Class	Number of Vessels	Distribution of 3 Most Common Hull Materials		Number Vessels 0-5 Years of Age[a,b]	Number of Self Propelled Vessels[a]	Horsepower Ahead for Vessels Reporting Horsepower[c]	
Commercial Fishing Vessel	111,094	fiberglass	19,433	3,125	54,258	5k or more	21
		wood	21,479			1 – 5k	443
		steel	7,805			500 – 1k	1732
						100 – 500	8478
						<100	1364
Passenger Vessel	34,609	fiberglass	8,022	2,502	17,061	5k or more	1
		wood	5,002			1 – 5k	642
		steel	1,633			500 – 1k	1,808
						100 – 500	4,473
						<100	1,059
Unspecified	37,578	fiberglass	6,595	2,648	16,392	1 – 5k	27
		wood	859			500 – 1k	142
		steel	1,998			100 – 500	1,570
						<100	606
Utility Vessel	18,743	fiberglass	408	920	8,698	5k or more	2
		wood	510			1 – 5k	711
		steel	7,082			500 – 1k	1,174
						100 – 500	1,159
						<100	111
Freight Barge	11,596	fiberglass	20	321	320	1 – 5k	1
		wood	87			500 – 1k	4
		steel	4,391			100 – 500	13
						<100	4
Freight Ship	1,102	fiberglass	38	19	610	5k or more	1
		wood	285			1 – 5k	5
		steel	224			500 – 1k	16
						100 – 500	54
						<100	9
Tank Barge	1,144	fiberglass	4	50	48	500 – 1k	6
		wood	2			100 – 500	1
		steel	323			<100	2
Public Vessel, unclassified	684	fiberglass	16	13	123	5k or more	2
		wood	1			1 – 5k	1
		steel	5			500 – 1k	2
						100 – 500	4
						<100	2
Tank Ship	220	fiberglass	0	5	85	1 – 5k	1
		wood	6			500 – 1k	2
		steel	43			100 – 500	19

Vessel Service Class	Number of Vessels	Distribution of 3 Most Common Hull Materials	Number Vessels 0-5 Years of Age[a,b]	Number of Self Propelled Vessels[a]	Horsepower Ahead for Vessels Reporting Horsepower[c]
					<100 10

[a] Based on operational, U.S. flagged commercial fishing vessels and other non-recreational vessel less than 79 feet (including vessels of unspecified length) identified in the 2009 MISLE database (USEPA 2010b)
[b] Vessel age was either not reported or an invalid age (i.e. less than zero) was reported for approximately 43,000 vessels.
[c] Horsepower ahead was not reported or an invalid value was reported for approximately 114,000 vessels.

Aquatic Nuisance Species

Exposure Analysis

The introduction of non-native species is one of the leading causes of species endangerment and extinction, second only to habitat loss (Wilcove et al. 1998). Shipping has led to more ANS introductions than any other source (Fofonoff et al. 2003). Within the broad category of shipping, ballast water and hull fouling are the primary vectors for ANS introduction to coastal and marine environments (Ruiz et al. 2000), which are among the most heavily invaded systems on earth (Grosholz 2002).

Globally, hull fouling and ballast water have been identified as the source of 228 (57%) marine ANS invasions (Molnar et al. 2008). In North America, these vectors were identified as the source for 164 of 316 (52 percent) marine invertebrate and algal invasions; these vectors were identified as a possible source for another 87 (27.5 percent) invasions, for a total of 251 (80 percent) invasions (Fofonoff et al. 2003). At least 40 introductions of 32 non-native fish species have been linked to ballast water transport (Wonham et al. 2000). As indicated in the BE, biofouling is likely a greater threat than ballast water. Globally, 39% of marine ANS invasions were linked to hull fouling, whereas 31% of marine ANS invasions were likely to have been transported via ballast water (Molnar 2008). Bax et al. (2003) identified hull fouling as the source of 36% of the nonnative coastal marine species established in continental North America (Bax et al. 2003), whereas ballast water only accounted for 20% (Carlton 2001).

Propagule pressure is the quantity, quality, and frequency with which propagules (ANS or their larvae, eggs, spores, etc.) are introduced to a given location (NAS 2011). The invasion process begins when an organism survives transport by a vector and is released into a novel environment. It may survive, reproduce, and spread throughout its environment; however, its successful establishment depends upon a myriad of factors, including: climate, food resources, its reproductive strategy, competition, predation, and parasitism (Carlton 1996). While species are most likely to be adversely affected by ANS after establishment, even limited exposure to an ANS organism can result in adverse effects, such as disease transmission.

The rate of ANS introductions has increased over time as vessel traffic has increased. For example, in the San Francisco Bay/Inland Delta, ANS were introduced an average of once every

36 weeks from 1850 to 1995, but in the last ten years (from 1985 to 1995), ANS were introduced once every 12 weeks (http://anstaskforce.gov/more_impacts.php). Invasion rates also differ by location. In the Lower Columbia River Basin, the ANS invasion rate is one every 20 weeks (Sytsma, 2004); in the Great Lakes the rate was one new invasion every 28 weeks, although no new invasions have been documented in the Great Lakes since 2006 (US EPA 2012b).

For the vectors described below, we estimate the number of vessels authorized by the VGPs and overall discharge volumes. We also attempt to provide a broad estimate of the number of ANS released to waters of the U.S., as authorized under the VGPs. We describe EPA's proposed actions that are likely to minimize the exposure of listed resources to ANS. Finally, we summarize the likelihood that ESA-listed resources would be exposed to ANS as a result of vessel discharges. When possible, we provide quantitative estimates, but in most instances, these data are not available. We therefore review the best available scientific data to provide a qualitative analysis of the risk of ANS invasion and the exposure of listed resources (NAS 2011; G. Ruiz, J. Carlton, pers. comm.).

Hull Fouling

Biofouling of hulls is a significant, if not the single greatest, source of ANS invasions. For example, 90 percent of the 343 marine aquatic invasive species in Hawai'i are thought to have arrived via hull fouling (Carlton 2001), while 36 percent of the nonnative coastal marine species established in continental North America are attributed to hull fouling (Bax et al. 2003).

Approximately 800 million square meters of hull area arrives in the U.S. annually (Miller et al. 2007). As described in the BE, Miller et al. (2007) investigated the extent of biofouling on 21commercial vessels and found that even the best-maintained commercial vessels had biofouling on up to 5% of their wetted surface, while 10 drydocked ships were covered over 5 to >90% of their wetted surface. Container ships are expected to have the lowest levels of biofouling due to their fast speeds and short port durations. In a study of 22 in-service container ships at the Port of Oakland, Davidson et al. (2009) found low levels of biofouling on 21 hulls; however, biofouling covered 90% of one hull. Taxonomic richness was high among all container ships. A study of underwater video and scrapings taken from 20 vessels shortly after their arrival to the Great Lakes estimated that total abundance averaged >170,000 invertebrates per vessel, belonging to 109 species (Sylvester and MacIsaac 2010). A study of 40 transoceanic vessels arriving in Vancouver and Halifax estimated that total abundance averaged up to 600,000 invertebrates per ship belonging to 156 species (Sylvester et al. 2011).

Hull fouling ANS are generally bottom-associated sessile and sedentary organisms (http://www.serc.si.edu/labs/marine_invasions/vector_ecology/fouling.aspx). They attach in succession: a slime layer of bacteria and algae colonizes first; barnacles, bryozoans, and tube worms follow; and with this base established, mollusks, sponges, sea squirts, and seedweed are able to colonize (US EPA 2012b). Highly developed fouling communities may provide microhabitats for mobile organisms, such as fish, crabs, and sea stars (EPA 2011). These organisms are able to release themselves, gametes, or fragments into aquatic environments encountered by the vessel, including novel harbors and waterways.

To remove biofouling organisms, and to reduce or slow the attachment of new organisms, vessels perform underwater ship husbandry or hull husbandry, i.e., the maintenance of the underwater portions of a vessel. Hull husbandry includes the application of antifouling coatings (AFCs) and removal of biofouling organisms during dry dock periods, as well in-water hull cleaning. The shipping industry performs hull husbandry primarily for economic reasons. Biofouling on a ship's hull increases the hydrodynamic drag of the vessel and leads to increased fuel consumption (Chambers et al. 2006). It has been estimated that fouling increases the annual fuel consumption of the world's commercial shipping fleet by 40 percent, or 120 million tons of fuel at a cost of about $ 7.5 billion per year, in year 2000 dollars (GISP 2008).

Underwater hull cleaning can be quite effective at removing marine fouling; however, the effluent stream from cleaning is difficult to control. The potential for ANS release resulting from the uncontained discharge of underwater cleaning effluent is widely recognized. For example, underwater hull cleaning has been banned by the ANZECC Code of Practice (ANZECC 1997). Underwater hull cleaning is also prohibited in California, Maine, and Massachusetts (EPA 2011).

As described in the BE, underwater hull husbandry has the potential to spread ANS by releasing them into the environment. To examine the ANS risks posed by various hull cleaning operations, researchers from the New Zealand National Institute of Water and Atmospheric Research (NIWA) visited hull cleaning facilities in four New Zealand ports where vessels were removed from the water for cleaning (drydock and haul-out facilities) or where fouling organisms were removed underwater by divers (Floerl et al. 2005). For the operations examined, physical removal of fouling assemblages from vessel hulls did not result in mortality of all organisms. Most fouling organisms (72%) survived and remained viable after removal using underwater hull cleaning methods, which did not involve exposure to air or high-pressure water blasting. The overall viability of organisms removed from hulls was lowest in haul-out facilities (16 %) and drydock facilities (43 %), where fouling organisms were often exposed to air, high-pressure freshwater blasting, and trampling.

Vessel hull fouling and underwater husbandry thus provide two pathways for ANS invasions. To address these concerns, EPA included the following requirements in the VGP (suggested measures are not included here or in the *Description of the Action* section; for example, the VGP does not require AFCs, but if owners/operators use AFCs, they are required to "give consideration" to selecting products that maximize efficacy but minimize adverse environmental effects):

- Minimize the transport of attached living organisms when traveling into U.S. waters from outside the U.S. economic zone or between Captain of the Port zones.
- For underwater hull cleaning, vessel owners/operators must, when feasible, use a vacuum or other control technologies to minimize the release of dispersion of fouling organisms into the water column.

To address ANS hull fouling concerns, EPA included the following requirements in the sVGP:

- Minimize the transport of any visible living aquatic organisms from one waterbody to another by regularly cleaning and maintaining the hull.
- Vessel hulls must be periodically inspected, and if necessary, cleaned to prevent the spread or dispersal of potentially invasive species.
- Prior to transporting the vessel from one waterbody to another overland, you must inspect the visible areas of the vessel for any attached or visible stowaway living organisms. If organisms are found, they must be removed and appropriately discarded onshore. Removed organisms may not be discharged into waters subject to this permit.

It is not clear, from reading the BE or the permits, the extent to which these requirements will minimize the risk of ANS invasions. For example, EPA requires vessel owners to minimize the transport of ANS, but EPA does not require anti-fouling coatings in either the VGP or the sVGP. If vessel owners/operators use AFCs, EPA requires that they "give consideration" to selecting the most effective product for their vessel type, but EPA does not provide guidance on which AFCs are the best available technology for each vessel type. The sVGP requires hull cleaning but does not require that debris containment systems be used to collect ANS. The VGP requires the use of such technology *when feasible*; however, the BE states that vacuum systems "are not widely commercially available; hence EPA has not required that they must be used in this permit," (BE). For these reasons, we conclude that the VGPs are not likely to substantially reduce the exposure of listed resources to ANS via hull fouling or hull husbandry.

The permit requirements to "minimize" ANS are not likely to result in major changes to vessel operations. Therefore, we will use the above estimates of hull area and hull fouling coverage to provide a rough estimate of fouling organisms. It is important to note that not all hull fouling organisms are non-native; not all are likely to become invasive; and not all are likely to affect listed species and critical habitat. We do not have data to tease out these variables; therefore, our estimate is the worst-case scenario of biofouling.

According to the U.S. Maritime Administration, there are ~60,000 vessel calls at U.S. ports annually (http://www.marad.dot.gov). As described above, Sylvester et al. (2011) estimated hull fouling propagule pressure at 600,000 invertebrates per vessel. This provides a rough estimate of 36 billion hull fouling organisms entering waters of the U.S. annually. (Note: this estimate does not differentiate between native and non-native species or the myriad of plant, vertebrate, or microbial organisms that foul hulls.)

While we are unable to quantitatively estimate how many of these organisms are likely to become established in waters of the U.S. (NAS 2011), we used the NEMESIS database to quantify the annual number of newly documented hull-fouling ANS. It should be noted that this database does not include all ANS that were introduced by hull fouling; rather, it represents the number of hull-fouling ANS *documented* nationwide. Therefore, in using this database, we expect considerable ascertainment bias, i.e., greater effort is likely to increase the number of ANS detected. Also, this number of documented ANS likely underestimates the total number of invasions because there are no monitoring programs throughout most of the waters of the U.S. Still, these data provide a qualitative measure of exposure of listed resources to documented hull-fouling ANS.

In querying the database (http://invasions.si.edu/nemesis/browseDB/searchQuery.jsp), we limited our search criteria to only include hull-fouling ANS documented in a single calendar year. We performed this search for each year between 2008 and 2011 (years covered under the previously issued VGP). The annual number of newly documented hull-fouling ANS in waters of the U.S. ranged from 10 to 20 species, with a mean of 13.5 (95% CI: 11.3 – 15.7).

As described above, the VGPs require few explicitly mandated anti-fouling and hull husbandry practices or technologies. Therefore, we do not expect to see substantial reductions of hull-fouling ANS under the proposed VGPs, and these invasions are likely to continue at a minimum rate of 13 species per year. We conclude that there is at least a small likelihood that listed resources would be exposed to ANS invasions as a result of hull fouling and hull husbandry as authorized by the VGPs.

Ballast Water

Ballast water is water taken onboard in large volumes to assist with vessel draft, buoyancy, and stability. It is taken from coastal port areas and transported with the ship to the next port of call, where the water may be discharged or exchanged. If a ship is receiving or delivering cargo to a number of ports, it may release or take on a portion of ballast water at each port. Large commercial vessels (e.g., container ships, bulk carriers, other cargo vessels, tankers, and passenger vessels) normally have ballast tanks dedicated to this purpose, and some vessels may also ballast empty cargo holds. The discharge volume varies by vessel type, ballast tank capacity, and type of deballasting equipment.

According to the BE, the VGP would authorize discharges from approximately 72,000 vessels; many of these vessels utilize ballast water. The sVGP would authorize discharges from 115,000 to 138,000 vessels; however these vessels (i.e., commercial fishing vessels less than 79 feet, passenger vessels, and utility vessels) do not have ballast water tanks greater than 8 cubic meters. Further, the BE notes that according to the U.S. Coast Guard: vessels less than 100 feet "typically operate in more sheltered environments and do not load and discharge ballast. Their stability characteristics generally accommodate the amount and type of cargo they carry, precluding the need to use ballast as a stability enhancer" (USCG 2008).

A National Academy of Science study estimates that ~52 billion gallons (196 million metric tons) of ballast water are discharged annually (NAS 2011). Ballast water that originated from outside of North America (i.e., that was taken on from a foreign source port) accounted for 28.5 percent of the total discharge volume, and the remainder came from other ports or locations within the U.S. and Canada (NAS 2011).

Vessel class estimates of ballast water discharge volumes reported in the BE are based on median ballast water capacity for those vessels that have ballast tanks (USEPA 2011c). The median is the middle observation in a dataset or 50th percentile. It a more robust measure of central tendency than the mean when the data are not symmetrically distributed around the mean (i.e., the data are skewed). However, reporting the median alone does not provide much useful

information about the distribution of the data such as the direction of skew or range of observations, nor does it represent a plausible worst-case scenario. NMFS used data reported to the NOI database to generate a more complete characterization of the distribution of ballast water capacity among VGP eligible vessels (Table 5). The distribution of ballast water capacity is strongly positively skewed for all vessel service classes, meaning that the majority of the observations are below the mathematical mean. The skew is particularly strong for barges, cargo carriers and large cruise ships.

Table 5. Vessel service class relative distribution of ballast water capacity reported in the NOI database (all values except skew are m^3).

Vessel Service Class	Number of Vessels Reporting Ballast Water Capacity	Mean	Standard Deviation	Minimum	Median (50th Percentile)	75th Percentile	Maximum	Skew
Barge	4692	2,477	57,302	8	447	1,030	3,543,700	54.9
Bulk Cargo and Other Carriers	6311	23,950	37,565	9	16,618	30,887	1,567,953	21.4
Commercial Fishing Vessel	211	6,691	12,765	12	594	6,775	78,007	2.9
Emergency Vessel	41	3,003	16,211	12	332	468	104,036	5.9
Large Cruise Ship	146	15,527	145,092	19	3,041	4,625	1,756,454	11.8
Large Ferry	44	2,806	8,022	17	387	1,625	51,224	5.0
Medium Cruise Ship	29	2,273	5,845	9	404	1,689	26,744	3.3
Oil or Gas Tanker	3955	32,522	29,951	17	23,473	42,018	662,128	3.6
Passenger	47	5,469	35,332	11	80	213	242,423	6.4
Utility	832	3,550	19,401	8	190	780	255,680	8.4

The uptake and discharge of ballast water potentially transports planktonic (i.e., occur in the water column) ANS to novel water bodies. Benthic ANS can also be transferred when sediments are taken in along with ballast water. Ballasting and deballasting occur through openings in vessel hulls, which are usually covered by protective gates that are permeable to viruses, bacteria, diatoms, algae, and invertebrate larvae (especially mollusks and crustaceans); however, corrosion or the loss of these grates permits larger organisms, including postlarval fish, to be ballasted (Wonham et al. 2000).

On any given day, approximately 7,000 individual species may be "in motion" in ballast tanks (Carlton 2001), and there is no evidence that ship age, seasonal timing, or age of ballast water affects the abundance of individuals or species in the ballast tanks (Drake and Lodge 2007). When species in ballast tanks are transported between water bodies and discharged, they have the potential for establishing new, non-indigenous populations that can cause severe economic and ecological impacts to waters of the U.S. It is not possible to quantitatively assess the number and type of ANS that have the ability to establish a viable population from the 7,000 individual species currently being carried in ballast tanks (NAS 2011).

It is possible, however, to link past ANS invasions to ballast water (NAS 2011). In the Laurentian Great Lakes, which are among the best-studied freshwater ecosystems in North America, 55 – 70% of the 180 ANS invasions are attributed to ballast water release (Ricciardi, 2006; Kelly et al., 2009; Holeck et al., 2004). For coastal marine ecosystems in western North America, 10-50% of over 250 invasions are attributed to ballast water release (Cohen and Carlton 1995).

Though the density of organisms in ballast water is just one of many variables that influence the probability that an invasive species will successfully establish a population in U.S. waters, there is evidence that significantly reducing the density of organisms in ballast water will reduce the probability of invasions, when controlling for all other variables (NAS 2011). Thus, establishing a benchmark discharge standard to reduce concentrations of coastal organisms below current levels is a logical first step.

Under current U.S. laws, regulations, and permitting requirements, existing best management practices are required to help reduce the potential impacts of ANS from ballast water discharges. These include ballast water exchange and saltwater flushing for certain vessels/voyages. These practices have been shown to offer some protection in mitigating the transfer and potential introduction of invasive species. While useful in reducing the presence of potentially invasive organisms in ballast water, ballast water exchange is not feasible for all vessels (e.g., vessels that cannot voyage offshore), can have variable effectiveness, and in some circumstances may not be feasible due to vessel safety concerns (Albert et al. 2010).

One way to address these limitations associated with ballast water exchange is to require treatment of ballast water prior to discharge to meet an established standard for the concentration of living organisms. On March 23, 2012, the US Coast Guard published final regulations to phase in ballast water treatment requirements, which became effective on June 21 of that year.

These regulations established a standard for the allowable concentration of living organisms contained in ships' ballast water discharges.

Similarly, to reduce the number of ANS in ballast water, EPA has included several requirements in the VGP. The following requirements are likely to reduce the exposure of listed species to ANS:

- Minimize or avoid ballast water discharge and update in critical habitat
- Ballast water numeric discharge limits (consistent with the Coast Guard's 2012 standards):
  - < 10 living organisms/m^3 ballast water, for organisms greater than or equal to 50 micrometers
  - < 10 living organisms/mL ballast water, for organisms less than 50 micrometers and greater than or equal to 10 micrometers
- Vessels may meet the above standards by one of the following management methods
  - Onshore treatment of ballast water
  - Use of public water supply for ballast water
  - No discharge of ballast water
  - Ballast water treatment system (with additional testing, calibration, and reporting requirements)

In the BE, EPA concluded that their VGP numeric effluent limits are likely to result in a "very small absolute risk of invasion." However, as described the BE, an estimated 196 million m^3 of ballast water is released into U.S. waters annually. Therefore, EPA requirements authorize an annual discharge of:

- < 1.96 billion living organisms, greater than or equal to 50 micrometers
- < 1.96 trillion living organisms, less than 50 micrometers and greater than or equal to 10 micrometers

For the above calculation, we assumed that all released ballast water in waters of the U.S. would be subjected to the numeric discharge limits of the VGP; however, this not the case. The majority of vessels in the U.S. (115,000 to 138,000 vessels) would be covered under the sVGP, which does not limit the number of living organisms that may be released into waters of the U.S. Above, we calculated the annual number of living organisms that EPA authorized to be released in ballast water using the VGP numeric effluent limits; however, EPA does not require numeric effluent limits for ballast water under the sVGP. Because the majority of vessels are covered under the sVGP and would not be required to meet numeric treatment limits, the total number of living organisms in ballast water that EPA authorizes to be released into waters of the U.S. is likely:

- > 1.96 billion living organisms, greater than or equal to 50 micrometers
- > 1.96 trillion living organisms, less than 50 micrometers and greater than or equal to 10 micrometers

While we are unable to quantitatively estimate how many of these organisms are likely to become established in waters of the U.S. (NAS 2011), we used the NEMESIS database to

quantify the annual number of newly documented ballast water ANS. It should be noted that this database does not include all ANS that were introduced by ballast water; rather, it represents the number of ballast-water-mediated ANS *documented* nationwide. Therefore, in using this database, we expect considerable ascertainment bias, i.e., greater effort is likely to increase the number of ANS detected. The number of documented ANS grossly underestimates the total number of invasions because there are no monitoring programs throughout most waters of the U.S. Still, these data provide a qualitative measure of exposure of listed resources to documented ballast-water-mediated ANS.

In querying the database (http://invasions.si.edu/nemesis/browseDB/searchQuery.jsp), we limited our search criteria to only include ballast water ANS documented in a single calendar year. We performed this search for each year between 2008 and 2011 (years covered under the previously issued VGP). The annual number of newly documented ballast-water-mediated ANS in waters of the U.S. ranged from 2 to 8 species, with a mean of 5.75 (95% CI: 4.5 – 7.0).

The numeric standards in the VGP are likely to reduce ballast-water-mediated ANS invasion rates; therefore, we expect that the number of documented ANS invasions is likely to be reduced over the life of the permit. Under the sVGP, however, equivalent standards are not required, and the only sVGP requirement that would substantially reduce exposure of listed species to ballast-water ANS is "minimize or avoid ballast water discharge and update in critical habitat." Thus, there remains a small likelihood that listed resources would be exposed to ANS invasions as a result of ballast water discharges authorized by the VGPs.

Other Vectors

Several other discharges incidental to the normal operation of a vessel serve as potential vectors for ANS. These include effluent from chain lockers, seawater cooling systems, and use and disposal of live bait. Less is known about these discharges and their potential to serve as ANS vectors; however, they are generally considered to be significantly less important than hull fouling and ballast water.

Chain locker effluent is water that collects in the below-deck storage area during anchor retrieval. A sump collects the liquids and materials that enter the chain locker and discharges it overboard or into the bilge. Chain locker effluent may contain marine organisms including ANS. The sVGP does not require any actions to minimize ANS transfer in chain locker effluent. To minimize the number of ANS in chain locker effluent, EPA has included the following requirements in the VGP:

- The anchor chain must be carefully and thoroughly washed down (i.e., more than a cursory rinse) as it is being hauled out of the water to remove sediment and marine organisms.
- Chain lockers must be thoroughly cleaned during dry-docking

The surfaces of seawater cooling systems, especially sea chest intakes and seawater piping, can support the growth of fouling organisms including ANS. In vessel sea chests, Coutts et al.

(2003) found typical fouling organisms, the introduced European clam (*Corbula gibba*), an invasive sea-star (*Asterias amurensis*), several species of small fish, and three introduced Green crabs (*Carcinus maenas*), two of which were ovigerous females capable of releasing viable propagules. Coutts and Dodgshun (2007) found that marine growth protection systems significantly reduce fouling by sessile and sedentary organisms, but have little influence on the mobile species within sea chests. The sVGP does not require any actions to minimize ANS transfer in seawater cooling systems. To minimize the spread of ANS through seawater piping, EPA has included the following requirements in the VGP:

- Vessel owners/operators must remove fouling organisms from seawater piping on a regular basis and dispose of removed substances in accordance with local, state, and federal regulations (regular basis is not defined)
- Removed fouling organisms shall not be discharged into waters of the U.S.

Fish hold effluent also has the potential to introduce ANS into receiving waters from fish waste or water transported from one port to another. To minimize the spread of ANS through fish hold effluent, EPA has included the following requirements in the VGP and the sVGP:

- Minimize the discharge of fish hold water and/or ice while in port and at the pier.
- If fish waste is contained in the fish hold effluent, it may not be discharged while in port unless a physical separation method is used.
- The discharge of fish hold effluent (including dirty ice) is prohibited if a shore-based discharge facility is available.
- The discharge of unused bait overboard is prohibited, unless bait was caught in the same water body or watershed.

The above requirements are included, as written, in the VGP and sVGP. In their comments (dated November 16, 2012) on our draft Opinion, EPA indicated that they have been making the following clarification for the sVGP (italics added by NMFS): Avoid discarding any unused *live* bait overboard, unless caught in that waterbody or watershed. Unused *live* bait purchased from a bait shop or dealer may not be discharged overboard.

Though we found the fish hold effluent requirements included in the VGP to be stringent, specific, and enforceable, the clarification on the sVGP is reason for concern. The discharge of dead bait from small vessels is likely to expose listed resources to non-indigenous pathogens. Therefore, while the VGP is likely to minimize the exposure of listed resources to ANS transported in fish hold effluent; discharges authorized by the sVGP are not likely to expose listed resources to ANS pathogens. Furthermore, the sVGP does not require any actions to minimize ANS transfer in seawater cooling systems or chain locker effluent. The VGP is vague on its requirements for the "regular" removal of organisms from seawater piping. Thus, it is not clear how or to what extent the VGPs will minimize the risk of ANS invasions from seawater cooling systems or chain lockers. Though these vectors are responsible for fewer ANS invasions than hull fouling and ballast water, they are regarding as ANS "hotspots" and cannot be discounted. Therefore, it is likely that listed resources will be exposed to ANS invasions as a result of EPA's authorization of discharges from seawater cooling systems, chain lockers, and fish hold effluent/dead bait.

Stressors

Established ANS are stressors that have the potential to jeopardize the future existence of threatened and endangered species through the following mechanisms:

- Predation
- Competition
- Food web alterations
- Alteration of the structure or function of ecosystems
- Introgressive hybidization
- Transfer of disease and parasites
- Destruction or modification of critical habitat

These stressors are the mechanisms by which ANS adversely affect native species; they are often multifold and interlinked. Here we highlight two examples of the multitude of stressors caused by ANS included in EPA's Biological Evaluation. Introduction of the green crab (*Carcinus maenus*) has been implicated in the destruction of soft-shell clam (*Mya arenaria*) fisheries in New England (Cohen and Carlton 1995) and the reduction of populations of other commercially important bivalves including the scallop, *Argopecten irradians*, and the northern quahog, *Mercenaria mercenaria* (Grosholz and Ruiz 2002). In Connecticut, weekly rates of crab predation on scallops were as high as 70% leading Tettlebach (1986) to observe that green crabs were responsible for most observed mortality in scallops and were a limiting factor in population size. MacPhail et al. (1955) concluded that the green crab was "one of the worst, if not the worst, clam predators we know." Their ability to out-compete native species for food resources, high reproductive capacity, and wide environmental tolerances enable them to fundamentally alter community structure in coastal ecosystems.

Many ANS are filter feeders, which at high densities can deplete the phytoplankton consumed by zooplankton, juvenile life stages, and planktivorous fish. The Asian clam (*Potamocorbula amurensis*) invasion of the San Francisco Bay is closely correlated to the shutdown of the spring phytoplankton bloom, which usually fuels much of the pelagic ecosystem (i.e., zooplankton and larval fish). Most of this primary production is transferred to the benthic food webs, which are dominated by benthic invertebrates and bottom-feeding fishes (Grosholz 2002). The overbite clam (*Corbula amurensis*) spread throughout the waterways surrounding San Francisco Bay within two years of being detected in 1986. The clam accounts for up to 95% of the biomass in some shallow portions of the bay floor. It is believed to be a major contributor to the decline of several pelagic fish species in the Sacramento-San Joaquin River Delta, including the threatened delta smelt, by reducing the planktonic food base of the ecosystem (Takata et al. 2011).

Response Analysis

Here we consider the probable responses of ESA-listed species to ANS invasions. We group species by four taxonomic groups (fishes, mollusks, corals, and cetaceans) because similar species are likely to respond similarly to the stressors described above. We focus on the individual responses to the following ANS-related stressors: predation, competition, trophic

alteration, ecosystem alteration, disease transmission, and habitat modification. We determine whether these responses are likely to reduce an individual's fitness (i.e., survival and reproductive).

Fishes

Several ESA-listed fish species are likely to be adversely affected by the introduction of ANS and the resulting stressors of competition, predations, food web alterations, changes in ecosystem structure and function, hybridization, transfer of diseases and parasites, and adverse modification or destruction of critical habitat. Though there are significant differences among fish species, their responses to ANS invasions are likely to be similar. Rather that discussing each species' responses individually, we group them taxonomically.

Salmonids. Here we evaluate the possible responses of listed salmonids (including Pacific salmon, Atlantic salmon, and steelhead trout) to the stressors of ANS invasions. The following discussion is extracted and summarized from a review of ANS risks to Pacific salmonids written by Sanderson et al. (2009), whose results indicate that the effect of nonindigenous species on salmon could equal or exceed that of four commonly addressed causes of adverse impacts (habitat alteration, harvest, hatcheries, and the hydrosystem). They suggest that managing nonindigenous species may be imperative for salmon recovery.

During their life cycle, salmonids traverse large geographic areas spanning freshwater, estuarine, and ocean habitats where they encounter numerous nonnative species, including introduced sportfish, invertebrate, and plant species. For Pacific salmonids in the Columbia River system alone, at least eight introduced fish species prey upon and compete with juvenile salmonids. Introduced channel catfish, large and smallmouth bass, and walleye prey upon juvenile salmon. In Columbia River reservoirs, large channel catfish (> 67 centimeters) consume thousands of juvenile salmon, which comprise 50% to 100% of their diets (Vigg et al. 1991). Although both smallmouth and largemouth bass prey on juvenile salmon, the impact is better documented for smallmouth bass, which consume 35% or more of juvenile salmon outmigrants in some regions (Fritts and Pearsons 2004). Walleye consume an estimated 250,000 to 2,000,000 Pacific salmon smolts annually in the Columbia River (Rieman et al. 1991, Tinus and Beamesderfer 1994). Juvenile American shad (*Alosa sapidissma*) compete with juvenile Chinook salmon for zooplankton prey (Haskell et al. 2006). Brook trout are associated with a 12% reduction of juvenile salmon in Snake River basin streams, though the mechanism for this reduction remains unknown (Levin et al. 2002). Many of the above-described invasive species were intentionally introduced for the purposes of recreational fishing; however, ballast water, hullfouling, and other vessel-related vectors may transport these species (as gametes, fry, or in their adult phase) into novel environments. For example, at least 40 introductions of 32 non-native fish species have been linked to ballast water transport (Wonham et al. 2000). As described above, the introduction of these species is highly likely to reduce the survival of individual salmonids through predation and competition for prey.

Invasive invertebrates species are also likely to adversely affect salmonids in the Pacific Northwest. The New Zealand mud snail (*Potamopyrgus antipodarum*), which represents more than 95% of the invertebrate biomass in some areas, has been reported to consume 75% of

autochthonous gross primary production (Hall Jr et al. 2003). They feed primarily on bottom-dwelling algae and detritus, competing with larval mayflies, stoneflies, and caddisflies (potential salmon prey) for food (Kerans et al. 2005). Mud snails have been identified in the stomachs of juvenile Chinook salmon sampled from the Columbia River estuary (Bersine et al. 2008). Whether these snails were intentionally ingested is not clear, but because of its thick shell and operculum, the snail is a poor nutritional source for salmon and other fish species, compared with insect larvae, fish, or other mollusks (Vinson and Baker 2008). The Siberian freshwater shrimp (*Exopalaemon modestus*), which was very likely introduced via ballast water, preys on native amphipods, such as *Corophium salmonis,* and directly competes with juvenile endangered salmon for important food resources (Emmett et al. 2002). Thus, the introduction of invertebrate ANS is likely to reduce survival and reproduction (i.e., fitness) of individual salmonids through reduction of their prey base and food web alteration.

Two introduced plants, purple loosestrife (*Lythrum salicaria*) and Eurasian water milfoil (*Myriophyllum spicatum*), have displaced native wetlands vegetation in the Pacific Northwest. The rapidly decomposing purple loosestrife produces a seasonal shift in local nutrient availability. Eurasian water milfoil forms dense mats of vegetation that reduce dissolved oxygen concentrations (Unmuth et al. 2000, Cronin et al. 2006). These invasive plants, which were likely introduced via shipping, are likely to change the structure and function of the ecosystems (Sanderson et al. 2009), reducing the survival of individual salmonids.

Parasites and disease have been transmitted to native salmonids via ANS. When salmon fry were imported to Sweden in the 1950s, a parasitic fluke worm infected native Norwegian wild salmon stocks, resulting in 95% mortality among the naïve natives (Josefsson and Andersson 2011). Viral hemorrhagic septicemia virus was initially introduced to North America from Western Europe via ballast water discharge in the Great Lakes. Following its introduction, it caused extensive morbidity and mortality to both farmed and wild fish populations including salmonids, smelts, and rockfish (McKenna 2007, Bain et al. 2010). Given EPA's recent clarifications to the sVGP (made in the November 16, 2013 comments on the draft Opinion), which authorize the discharge of dead bait, salmonids are likely to be adversely affected by novel pathogens if the bait was not obtained from the same waterbody in which it was used. As described above, such pathogens are often lethal to salmonids. Thus, ANS are likely to reduce the survival of individual salmonids through disease transmission.

In summary, the introduction of many vessel-related ANS species (including fish, invertebrates, plants, and parasitic organisms) is likely to result in reductions in fitness for individual salmonids.

<u>Pacific Eulachon.</u> Pacific euchalon are similar to Pacific salmon in that they are anadromous, occupy similar freshwater habitats (including the Columbia River), and exhibit diminishing abundance (Hay and McCarter 2000); however, they are far less studied than salmon. Their responses to ANS invasions are likely to be the same as described above for salmonids. Aquatic invasive species are likely to reduce the survival of individual Pacific euchalon via competition, predation, habitat modification, and disease transmission.

Sturgeon. Sturgeon species are also likely to be adversely affected by ANS species. Introduced predatory ANS would prey on the eggs and juveniles of sturgeon. Other ANS (e.g., Asian clams, zebra mussels, blue and flathead catfish) are likely to deplete sturgeon prey and/or compete for sturgeon resources (US EPA 2012b). Introduced ANS are likely to expose sturgeon species to viruses enzootic to the west coast of the U.S., spread diseases to these naïve populations (US EPA 2012b). Therefore, shipping-related introduction of ANS is likely to reduce the survival of individual sturgeon through predation, competition, and disease transmission.

Cetaceans

Cetaceans would be primarily exposed to ANS associated with vessel discharges through changes in the abundance and distribution of their prey (trophic exposure).

Southern Resident Killer Whale. Southern Resident killer whales prey primarily on fish; salmon are clearly the preferred prey, especially Chinook salmon during late spring to fall (Hanson et al. 2005, Ford and Ellis 2006). These whales consume salmon in large amounts to support individual growth to reach sexual maturity and reproduction, including lactation and successful rearing of calves (71 FR 69054).

As described above, ANS invasions are likely to reduce the fitness of individual threatened and endangered salmonids. Sanderson et al. (2009) report that the impact of a devastating ANS invasion is likely to equal or exceed that of all previously identified threats to salmon (i.e., habitat alteration, harvest, hatcheries, and the hydrosystem). In essence, ANS is likely to reduce the abundance of salmonids, and the loss of its salmonid prey base is likely to reduce the survival and reproduction of individual southern resident killer whales. In this manner, ANS invasions are likely to reduce the fitness of Southern Resident killer whales.

Beluga whale (Cook Inlet DPS). Cook Inlet beluga whales feed primarily on Pacific eulachon and salmonids. Pacific eulachon have a high fat content that confers a significant source of energy for beluga whales, including calving whales that occur in the upper inlet during the spring spawning (Calkins 1989). Like eulachon, salmon are another source of lipid-rich prey for the beluga whale and represent the greatest percent frequency of occurrence of the prey species found in Cook Inlet beluga whale stomachs (Hobbs et al. 2008). As described above, ANS invasions are likely to reduce the abundance of Pacific eulachon and salmonids, and the loss of the prey base of Cook Inlet beluga whales is likely to reduce the survival and reproduction of individuals. In this manner, ANS invasions are likely to reduce the fitness of Cook Inlet beluga whales.

Baleen whales. Microorganisms may constitute a majority of the species found in ballast water (USCG 2012). Phytoplankton, particularly diatoms and dinoflagellates, can be especially abundant and may cause the harmful algal bloom known as “red tides” (Carlton and Geller 1993) or produce toxic compounds. Ballast water discharge containing ANS has contributed to the increase in the frequency, intensity, and duration of harmful algal blooms in waters of the United States (Hallegraeff 1993). Harmful algal blooms can produce toxic compounds, including brevetoxin, saxitoxin and microcystins. Toxins can bioaccumulate in high trophic level marine animals (Bushaw-Newton and Sellner 1999), with adverse effects ranging from cell and tissue

damage to mortality.

Baleen whales are exposed to these neurotoxins by preying on highly contaminated zooplankton (Durbin et al. 2002). For example, the summertime habitat of the North Atlantic right whale overlaps with seasonal blooms of the toxic dinoflagellate *Alexandrium fundyense*. Foraging right whales would ingest large concentrations of contaminated copepods, which affect respiratory capabilities, feeding behavior, and ultimately the reproductive condition of the populations (Durbin et al. 2002).

Sea Turtles
Vessel mediated ANS can include the organisms that cause harmful algal blooms. As described above, harmful algal blooms can produce toxic compounds, including brevetoxin, saxitoxin and microcystins. Toxins can bioaccumulate in high trophic level marine animals (Bushaw-Newton and Sellner 1999), with adverse effects ranging from cell and tissue damage to mortality. Immune system responses have been affected by brevetoxin exposure in the endangered Loggerhead sea turtle (BE).

Mollusks
Invasive species are likely to adversely affect endangered abalone species via predation, competition, and by transmitting parasites. The invasive green crab, described earlier, preys on a variety of mollusks and is likely to prey upon black and white abalone. The invasive snail *Batillaria attramentaria* outcompetes a native mollusk species, *Cerithidea californica*, along northern California via a more efficient ability to convert food into body tissue and possibly greater dispersal potential (Savino and Kolar 1996, Carlton 1999, Pothaven et al. 2001). It, or other introduced sea snails, are likely to compete with reduced black and white abalaone populations for prey and habitat. In addition, mollusks are particularly sensitive to pathogens. Withering syndrome threatens the survival and recovery of the black abalone; it is caused by a Rickettsia-like bacterium, *XenoHaliotis californiensis,* that was likely introduced to southern California via ballast water discharge (Friedman et al. 2000, Smith et al. 2003, Bower 2009, Cohen 2010). Various species of the genus *Vibrio* have been identified in ballast water (Anguiano-Beltrán et al. 1998, Ben-Haim and Rosenberg 2002, Aguirremacedo et al. 2008) and are known to cause high rates of mortality in abalone (Chew 1990, Ciguarria and Elston 1997)Simon *et al.* 2006) (Nell 2002, Lleonart et al. 2003, Nehring 2006, Haupt et al. 2010). A parasitic worm, *Terebrasabella heterouncinata*, was introduced to red abalone via aquaculture but likely to be spread to endangered abalone species via ballast water (Kuris and Culver 1999, Cohen and Webb 2002, Culver and Kuris 2002, Bower 2006). Thus, the introduction of ANS is likely to reduce the survival of endangered mollusks through predation, competition, and disease transmission.

Corals
Vessel-mediated introductions of ANS adversely affect coral species through the mechanisms of competition and disease transmission. Invasive corals and algae overgrow native corals (Lesser et al. 2007). For example, introduced red algae competes with native Hawaiian coral species for space on the reef (Lafferty and Kuris 1996). The snowflake coral, *Carajoa riisei*, has overgrown up to 90% of surveyed black coral colonies in Hawai'i, where it was introduced via shipping

(Kahng and Grigg 2005).

The introduction of novel pathogens into a habitat can cause overwhelming infections, particularly when coupled with stresses of a disturbed environment which corals must already cope (Lesser et al. 2007). In the Caribbean, endangered elkhorn and staghorn coral (*Acropora palmate* and *A. cervicornis*) face severe declines in part due to diseases, including white pox and white plague type II. These coral diseases are caused by various species of the genus *Vibrio*, which have been identified in ballast water of vessels (Anguiano-Beltrán et al. 1998, Ben-Haim and Rosenberg 2002, Aguirremacedo et al. 2008). Thus ANS invasions are likely to reduce the survival of corals through competition and disease.

Johnson's Seagrass
The species is likely to be adversely affected by ANS through the mechanisms of competition and disease transmission. For example, the introduced seagrass, *Zostera japonica*, has led to the decline of native seagrasses and ecosystem function (Williams 2007). The "wasting disease" that devastated New England eelgrass populations in the 1930 has be transmitted to native seagrass communities via ANS (Minchin 2007). Johnson's seagrass has the most limited geographic distribution of any seagrass in the world (Kenworthy 1997). In its native Florida habitat, 50 ANS have become established (Courtenay 1997), 23 of which are aquatic plant species (McCann et al. 1996). Disease would likely spread quickly through the small population. An invasive seagrass is likely to overgrow the endangered seagrass beds throughout its small range. Therefore, we conclude that ANS are likely to reduce the survival of Johnson's seagrass through disease and competition.

Extinction and Endangerment as a Result of ANS Invasions

In our *Exposure Analysis*, we concluded that threatened and endangered species and their designated critical habitat are likely to be exposed to ANS in the following discharges: ballast water, hull fouling/underwater husbandry, sea cooling systems, and chain lockers. In our *Response Analysis*, we concluded that listed individuals exposed to ANS invasions are likely to experience reduced fitness as a result. Here we ask whether these reductions in fitness are likely to reduce the viability of populations and species. We start by reviewing the published scientific literature on species endangerment and extinction as a result of ANS invasions.

The introduction of non-native species poses one of the greatest threats to biodiversity, second only to habitat loss (Wilcove et al. 1998). In the United States, more than 40% of species listed under the ESA are threatened by invasive species (Wilcove et al. 1998). The ANS Task Force identifies ANS as a major cause of decline of species and a significant impediment to their recovery (http://www.anstaskforce.gov). The detrimental effects of ANS have contributed to 68% of North American fish extinctions over the past century, including two *Oncorhynchus* species (Miller et al. 1989). A study by USGS (2012) identified ANS as a contributing factor in the listing of more than 50% of fish species and 33% of invertebrate species listed under the ESA. Some argue whether invasive species are the *primary* cause of endangerment or extinction, but it is widely agreed that listed species become further imperiled as a result of ANS invasions (Gurevitch and Padilla 2004, Clavero and Garcia-Berthou 2005).

While there are many examples of fresh water species extinctions as a result of ANS, marine equivalents are less prevalent in the literature; however this may be a result of underreporting and low detection power (Roberts and Hawkins 1999). As terrestrial animals, humans are less likely to notice declines and extinctions in the marine environment; furthermore, many marine species have yet to be described, and their extinction would go unnoticed. Though we were unable to find any examples of marine extinctions caused exclusively by ANS, there are several examples where ANS invasions led to the decline of native species. For example, egg predation by an introduced starfish (*Asterias amurensis*) led to the decline of the critically endangered spotted handfish (*Brachionichthys hirsutus*) in Australia. In California, the introduced Japanese mud snail (*Batillaria attramentaria*) outcompetes the native snail (*Cerithidea californica*) and is predicted to eventually replace the native species (Byers 2000). Molecular analyses of museum specimens suggest that alien mussels (*Mytilus galloprovincialis*) displaced the native mussel (*M. trossulus*) in southern California, some time in the last century (Geller 2001).

We present statistics on ANS-related extinctions and endangerments to illustrate that ANS invasions have reduced the viability of numerous populations and species and are likely to do so in the future. As described above, ANS invasions affect species by predation, competition for habitat or prey, reduction of the prey base, habitat alteration, disease transmission, and ecosystem restructuring. These types of impacts rarely affect one or a few individuals; they are more likely to affect many individuals or an entire population. For example, we described several diseases that ANS are likely to transmit to listed fish, abalone, corals, and seagrass. These infectious diseases would first affect exposed individuals, but they would quickly spread through these small immunologically naïve populations, which often exhibit low genetic diversity a reduced ability to mount an effective immune response. Ultimately, such ANS-transmitted diseases would reduce the viability of the population, and the loss of populations would reduce the viability of species. Similarly, we described above how ANS are likely to reduce the survival of salmonids and Pacific eulachon through competition, predation, disease transmission, habitat loss, and ecosystem restructuring. Again, these mechanisms are likely to affect many individuals, reducing the viability of populations, and thus the species (or listed DPS). A significant loss in the abundance of salmonids and Pacific eulachon is likely to reduce the fitness of many Cook Inlet beluga whales and southern resident killer whales that depend on these species as their primary prey. These DPSs have already experienced severe population declines, in part as a result of reduction in their prey base. Additional reductions would reduce the viability of these species. In conclusion, ANS invasions, which are likely to occur as a result of the discharges authorized by EPA, are likely to result in reduced population and species viability of listed fish, marine mammal, invertebrate, and plant species.

Critical Habitat

Invasive species are likely to adversely affect water quality in marine, estuarine, and riverine freshwater environments by altering processes such as sedimentation rates and nutrient cycling. Introduced non-native algal species from ballast water (as well as other sources) combined with nutrient overloading may increase the intensity and frequency of algal blooms. An overabundance of algae can degrade water quality upon bacterial decomposition, which depletes

oxygen levels in an ecosystem. Oxygen depletion can result in "dead zones," murky water, seagrass and coral habitat degradation, and large-scale fish kills (Deegan and Buchsbaum 2005). Therefore, successful ANS invasions are likely to adversely affect the critical habitat of listed species, especially when ANS threaten the prey that has been designated as a primary constituent element, i.e., physical or biological features that are essential to the conservation of a given species, including: space for individual and population growth and for normal behavior; food, water, air, light, minerals, or other nutritional or physiological requirements; cover or shelter; sites for breeding, reproduction, and rearing of offspring; and habitats that are protected from disturbance or are representative of the historical geographical and ecological distribution of a species (50 CFR 424.12(b)).

Salmon critical habitat primary constituent elements include areas that contain prey, including "aquatic invertebrates and fishes" (e.g. 70 FR 52630). The primary constituent elements of Pacific eulachon critical habitat includes "prey resources to support larval eulachon survival…[and] prey items, in a concentration that supports foraging leading to adequate growth and reproductive development for juveniles and adults in the marine environment" (76 FR 65324). Sturgeon critical habitat primary constituent elements include: "abundant food items, such as detritus, aquatic insects, worms, and/or molluscs, within riverine habitats for larval and juvenile life stages; and abundant prey items, such as amphipods, lancelets, polychaetes, gastropods, ghost shrimp, isopods, molluscs and/or crustaceans, within estuarine and marine habitats and substrates for subadult and adult life stages" (68 FR 13370). As described above, ANS invasions are likely to reduce the availability of prey and thus adversely affect critical habitat for these fish species. In addition, invasive plant species are likely to adversely affect the critical habitat of endangered salmonids. Two introduced plants, purple loosestrife (*Lythrum salicaria*) and Eurasian water milfoil (*Myriophyllum spicatum*), have displaced native wetlands vegetation in the Pacific Northwest. The rapidly decomposing purple loosestrife produces a seasonal shift in local nutrient availability. Eurasian water milfoil forms dense mats of vegetation that can depress dissolved oxygen concentrations (Unmuth et al. 2000, Cronin et al. 2006). These invasive plants, which were likely introduced via shipping, are likely to change the structure and function of the ecosystem (Sanderson et al. 2009) and destroy native salmonid.

The primary constituent elements of critical habitat for the southern resident killer whale include "prey species of sufficient quantity, quality, and availability to support individual growth, reproduction, and development, as well as overall population growth" (71 FR 69054). Cook Inlet beluga whale critical habitat includes the primary constituent element of "primary prey species consisting of four species of Pacific salmon (Chinook, sockeye, chum, and coho), Pacific eulachon, Pacific cod, walleye pollock, saffron cod, and yellowfin sole" (76 FR 20180). As described above, ANS invasions are likely to reduce the availability of prey and thus adversely affect critical habitat for these cetacean species.

Black abalone critical habitat includes a primary constituent element of "abundant food resources including bacterial and diatom films, crustose coralline algae, and a source of detrital macroalgae, required for growth and survival of all stages of black abalone" (76 FR 66806). Elkhorn and staghorn coral critical habitat includes the primary constituent element of "substrate of suitable quality and availability" means natural consolidated hard substrate or dead coral

skeleton that is free from fleshy or turf macroalgae cover and sediment cover (73 FR 72210). As described above, ANS invasions are likely to reduce the availability of prey or introduce macroalgae and thus adversely affect critical habitat for these invertebrate species.

In conclusion, there is a small likelihood that listed resources will be exposed to ANS as a result of discharges authorized by EPA's VGPs. If exposed, individuals are likely to experience reductions in fitness. As a consequence, species and population viability is likely to be diminished. In addition, exposure to ANS is likely to adversely affect designated critical habitat.

Pollutants

Exposure Analysis

The constituents of the discharges authorized by the proposed VGPs are mixtures of pollutants capable of producing adverse effects to endangered or threatened species and their critical habitat. This section first reviews the vessel discharges authorized by the VGPs in terms of discharge volumes and constituent mixtures then summarizes pollutant loads and EPA's modeled estimates of final harbor concentrations. Pollutant load, expressed in terms of mass per day, is the net amount of pollutant discharged based on discharge volume and pollutant concentration. Many pollutants are found in more than one discharge source, with some sources being more significant pollutant load contributors than others by virtue of their greater volume or constituent concentrations.

The vessels discharge information used by EPA in its BE were taken from the Report to Congress and other sources such as the Uniform National Discharge Standards for Vessels of the Armed Services Technical Development document (USDOD/USEPA 1999) and several reports generated by EPA in support of the VGPs (USEPA 2008, 2011d, b). The BE relied heavily on discharge volume and constituents data in the *Report to Congress: Study of Discharges Incidental to Normal Operation of Commercial Fishing Vessels and Other Non--Recreational Vessels Less than 79 Feet* (EPA 2010). It is important to note this data represents opportunistic sampling from a total of 61 vessels intended to represent a population of the greater than 90,000 non-recreational small vessels[10] tracked in the MISLE database. Not all discharge types were sampled from each vessel. The same constituents were not analyzed for in all discharges due to sample volume constraints or decisions made regarding whether or not the discharge would contain a given analyte or analyte group (e.g. nonylphenols in engine exhaust). In a few cases, target analytes differed for the same discharge type within vessel classes, for example, additional analytes would be screened for when sample collectors observed an unexpected analyte source. These data properties limit systematic characterization of discharges within and among vessel service types. Given the very limited number of vessels examined, inferences made based on these data regarding discharges from other vessels, particularly the larger vessels covered by the VGP, are accompanied by high levels of uncertainty. After describing exposure data used in the

[10] Small recreational vessels far outnumber small non-recreational vessels, but are covered under the Clean Boating Act and are not subject to the sVGP.

BE we provide a summary of the expected overlap of these exposures with the critical habitat and range of our species. This *Exposure Analysis* section is followed by the *Response Analysis,* which describes the potential effects of discharge constituents and the consequences of exposing listed species to those constituents.

Discharges Authorized by the VGP and sVGP

EPA identified 27 specific effluent streams as discharges eligible for coverage under the VGP while the sVGP authorizes discharges or discharge creating activities conducted in the normal operation of vessels. The VGP and sVGP authorized discharges are cross-walked in Table 6. Several of discharges authorized by the VGP, such as sonar dome and elevator pit effluent, are specific to larger vessels and are therefore not included in the sVGP. Many sVGP discharge activities are either identical to VGP-authorized discharges (e.g., ballast water) or are broader discharge classifications. For example, the "Engine and Oil Control" discharge activity covered by the sVGP conceptually includes the "Bilgewater/Oily Water Separator Effluent" and "Controllable Pitch Propeller and Thruster Hydraulic Fluid and Other Oil-to-Sea Interfaces…" discharges authorized under the VGP.

Table 6. Discharge classes authorized under the Vessels General Permit and small vessels general permit.

VGP Authorized Discharge	sVGP Authorized Discharge
Anti-Fouling Hull Coatings/Hull Coating Leachate	Vessel Hull Maintenance
Underwater Ship Husbandry Discharges	
Cathodic Protection	
Ballast Water	Ballast Water
Bilgewater/Oily Water Separator Effluent	Engine and Oil Control
Boat Engine Wet Exhaust	
Controllable Pitch Propeller and Thruster Hydraulic Fluid and Other Oil-to-Sea Interfaces Including Lubrication Discharges from Paddle Wheel Propulsion, Stern Tubes, Thruster Bearings, Stabilizers, Rudder Bearings, Azimuth Thrusters, Propulsion Pod Lubrication, and Wire Rope and Mechanical Equipment Subject to Immersion.	
Motor Gasoline and Compensating Discharge	Fuel Management
Fish Hold Effluent	Fish Hold Effluent
Boiler/Economizer Blowdown	These discharges are not typically associated with sVGP eligible vessels
Welldeck Discharges	
Refrigeration and Air Condensate Discharge	
Exhaust Gas Scrubber Washwater Discharge	
Freshwater Layup	
Gas Turbine Washwater	

VGP Authorized Discharge	sVGP Authorized Discharge
Non-Oily Machinery Wastewater	
Seawater Piping Biofouling Prevention	
Seawater Cooling Overboard Discharge	
Sonar Dome Discharge	
Aqueous Film Forming Foam (AFFF)	
Elevator Pit Effluent	
Graywater Mixed with Sewage	
Distillation and Reverse Osmosis Brine	
Firemain Systems	
Graywater	Graywater
Not applicable to VGP	Solid and Liquid Waste Management
Deck Washdown and Runoff and Above Water Line Hull Cleaning	Deck Washdown and Runoff and Above Water Line Hull Cleaning

The decision process EPA followed in its selection of which discharges to analyze for risk in the BE was based on the relative number of vessels releasing the discharge, the volume of the discharge, and the potential risk posed by the concentration and constituents of that discharge. The following sections briefly describes each discharge selected for evaluation in the BE. This is followed with a table summarizing those discharges not evaluated in the BE and EPAs decision process for each.

Anti-Fouling Leachate from Anti-Fouling Hull Coatings. Anti-fouling coatings are applied to the vessel hull and seawater piping systems to limit attachment of aquatic species. Virtually all vessels that are permanently kept in saltwater use anti-fouling coatings. Biocides such as copper contained in anti-fouling coatings continuously leach into surrounding waters. EPA analyzed the risk of this discharge in its BE due to the high number of vessels discharging, the large volume and frequency of discharges, along with the potentially high risk posed by the concentration and types of pollutant discharged. In its supplement to the BE, EPA estimates pollutant loadings from antifouling coatings (Table 7).

Table 7. Copper loading AFC of various vessel service classes

Vessel Type	Average Vessel Length (ft)	Estimated Vessel Draft (ft)	Wetted Hull Area (ft^2)	Average Copper Leaching Rate ($\mu g/cm^2/day$)[j]	Estimated CopperLeaching Rate (lbs/day/vessel)
Barges	175a	10d	3,500	10.5	0.075n
Oil and Gas Tankers	797b	75e	120,257	10.5	2.6
Large Commercial Fishing Vessels	250a	17f	8,500	10.5	0.18
Large Cruise Ships	995c	28g	55,706	10.5	1.2

Large Ferries	200a	15.5h	6,200	10.5	0.13
Medium Cruise Ships	325a	15.5h	10,075	10.5	0.22
Bulk Cargo and Other Carriers	663b	38i	50,355	10.5	1.1
Small Commercial Fishing Vessels	33k	4l	264	10.5	0.0057
Passenger Vessels (tour boats, water taxis, etc.)	45k	4m	360	10.5	0.0077
Utility Vessels (tug boats, off-shore vessels, etc.)	50k	4m	400	10.5	0.0086

a) Average length estimated from Figure 1.3 of USEPA's Report to Congress (USEPA, 2010).
b) Average length for oil tankers and bulk cargo carriers from Chamber of Shipping of America, MSD Survey, 2010.
c) Average cruise ship length calculated from the following cruise ships: Carnival Freedom, Disy Magic, Golden Princess and Freedom of the Sea.
d) Estimated draft for barges based on Barge Number 4727, Ingram, Nashville, TN.
e) Estimated draft for oil and gas tankers based on a Suezmax tanker, http://people.hofstra.edu/geotrans/eng/ch5en/appl5en/tankers.html
f) Estimated draft for large commercial fishing boats based on trawlers, http://maritimesales.com/fishing%20vessels.htm
g) Estimated draft for large cruise ships based on Royal Caribbean Freedom of the Seas.
h) Estimated draft for large ferries and medium cruise ships based on Washington State Department of Transportation's M/V Sealth, passenger/auto ferry.
i) Estimated draft for bulk cargo vessels based on Golden Pegasus Navigation S.A High Glow vessel.
j) Average copper leaching rate from USEPA Underwater Ship Husbandry Discharges, (USEPA, 2011f).
k) Average length estimated from Figure 1.5 of USEPA's Report to Congress (USEPA, 2010).
l) Estimated draft for small commercial fishing boats 30 to 40 feet in length, http://pacificboatbrokers.com/fishing/licvess.asp.
m) Estimated draft for various passenger and utility vessels, http://www.taxi-boat.com/, http://commercial.apolloduck.com/, http://allmandboats.com/.
n) Per EPA's Notice of Intent database, very few barges use anti-fouling coatings. EPA assumes these barges are engaged in coastwise trade of bulk cargo.

Bilgewater. Bilgewater is generated by all vessels and consists of water and other residue that accumulates in a compartment of the vessel's hull. The source of bilgewater is typically drainage from interior machinery, engine rooms, and from deck drainage. EPA analyzed the risk of this discharge in its BE due to the high number of vessels discharging, the large volume and frequency of discharges, along with the potentially high risk posed by the concentration and types of pollutant discharged.

The amount of bilgewater generated by a vessel varies based on vessel size, age, and the amount below-deck mechanical systems; discharges typically occur several times per week (Table 8). Cruise ship bilgewater volumes are estimated at 25,000 gallons per week (3,570 gal/day) (USEPA 2008). Bilgewater discharges from fishing vessels are estimated to range from a few gallons per day up to 750 gallons/day (USEPA 2010b). Bilgewater water discharges rates for tour boats are approximately 11 gallons/day (0.04 m^3/day) and for water taxis are approximately 34 gallons/day (0.13 m^3/day). Tow and salvage boats discharge approximately 37 gallons/day (0.139 m^3/day) of bilgewater (USEPA 2010b). To estimate bilgewater flow rates for medium cruise ships, large ferries, barges, cargo/bulk carriers, and oil tankers, EPA compared the vessel gross tons (see Table 3-1) to large cruise ships and multiplied by the bilgewater generation rate

for large cruise ships. The example below shows how daily bilgewater discharge was estimated for cargo/bulk carriers using information for large cruise ships.

Example: 15,939 gross tons (cargo/bulk carrier)/53,526 gross tons (large cruise ship) x 3,570 g/day = 1,063 g/day

Table 8. Estimated Bilgewater Discharge Rates used in EPAs BE Analyses.

Vessel Type	Estimated Bilgewater Discharge Rates	
	(m^3/day)	(gallons/day)
Barges	0.18	47
Cargo/Bulk Carriers	4.0	1,063
Commercial Fishing	1.4	375
Large Cruise Ships	13.5	3,570
Medium Cruise Ships	2.0	305
Large Ferries	2.0	97
Oil and Gas Tankers	6.3	1,661
Passenger Vessels (e.g., water taxis, tour boats)	0.08	22
Utility Boats (tugs and tow boats)	0.14	37

EPA estimated concentrations of bilgewater pollutants based on data for vessels examined in the Report to Congress. A total of eight bilgewater samples were collected from seven vessels with a total of 128 analytes detected. The vessels sampled included two fishing vessels, two tow/salvage vessels, a tour boat and two water taxis. Contaminants of concern in bilgewater identified by EPA are listed in Table 9 and include conventional pollutants, nutrients, metals and volatile/semivolatile organic chemicals (sVOCs/VOCs). Table 9 includes additional information for these analytes from the Report to Congress, which provides details on the detection frequency and distribution of all analytes detected in bilge water.

Table 9. Estimated Bilgewater Pollutant Concentrations used in EPAs Biological Evaluation

	Pollutant	Point Estimate Used in BE	Detection frequency	Mean (range)	$50^{th}/75^{th}/90^{th}$ Percentiles[b]
Conventional Pollutants	Ammonia Nitrogen (mg/L)	1.7	4/5	1.7 (nd-7.6)	0.24/4/7.6
	Total Phosphorous (mg/L)	3	5/5	3 (0.084-13)	0.47/7.1/13
	Total Residual Chlorine (TRC, µg/L)	0.077	3/7	0.077 (nd[a]-0.16)	nc[c]/0.13/0.16
	Total Suspended Solids (TSS, mg/L)	39	7/7	39 (3.7-88)	38/71/88

	Pollutant	Point Estimate Used in BE	Detection frequency	Mean (range)	50th/75th/90th Percentiles[b]
	Biochemical Oxygen Demand (BOD, mg/L)	190	7/7	190 (2-770)	14/330/770
	Hexane Extractable Oil and Grease (HEM, mg/L)	9.3	7/7	9.3 (1.1-44)	5.2/7/44
	Silica Gel Treated-Hexane Extractable Oil and Grease (SGT-HEM, mg/L)	4.4	7/7	4.4 (1.1-18)	2.4/3.5/18
Metals	Aluminum (total, µg/L)	370	7/7	370 (26-940)	330/640/940
	Cadmium (dissolved, µg/L)	1.9	1/7	1.9 (nd-10)	nc/nc/10
	Chromium (dissolved, µg/L)	12	5/7	12 (nd-56)	1.6/17/56
	Copper (dissolved, µg/L)	100	7/7	100 (6.6-350)	56/120/350
	Nickel (dissolved, µg/L)	9.2	6/7	9.2 (nd-15)	8.8/14/15
	Zinc (dissolved, µg/L)	130	7/7	130 (53-250)	100/190/250
sVOCs and VOCs	2,4,6-Trichlorophenol (µg/L)	7	1/7	7 (nd-24)	nc/nc/24
	Benzene (µg/L)	61	4/7	61 (nd-410)	0.1/1.3/410
	Bis (2-ethylhexyl)-phthalate (µg/L)	15	4/7	15 (nd-71)	1.4/21/71
	Trichloroethene (µg/L)	2.6	1/7	2.6 (nd-0.3)	nc/nc/0.3

[a]Nondetect (censored) concentrations were replaced with ½ of the reporting limit for calculating average concentrations.
[b] Percentiles are the concentrations of each analyte below which at least that percentage of the values fall. So the 75th percentile is the concentration below which at least 75% of the observations were found.
[c] Distribution statistics were only calculated when analytes were detected at a sufficient frequency.

Deck Washdown. Deck washdown occurs from all vessels as a result of deck cleaning. Constituents in the discharge can include detergent, soap, deck surface components (e.g., rust, paint chips) and anything dropped, spilled, dripped, or scattered onto the deck surface. Deck washdown volumes vary widely and are highly dependent on a vessel's purpose, service, and practices. For example, since aesthetics are important to the industry, deck washdowns may be performed more frequently on cruise ships than other types of vessels. EPA analyzed the risk of this discharge in its BE due to the high number of vessels discharging, the large volume and frequency of discharges, along with the potentially high risk posed by the concentration and types of pollutant discharged.

EPA used information for commercial fishing vessels, passenger vessels and utility boats to estimate deck washdown volumes (USEPA 2010b) for barges, cargo/bulk carriers, and oil and gas tankers (Table 10).

EPA estimated the deck washdown volumes by comparing vessel gross tons for these vessels with the gross tons of a typical utility vessel. EPA estimated utility vessels are 80 gross tons based on 50% of the utility vessels being approximately 65 feet long (USEPA 2010b). For medium and large cruise ships and large ferries, EPA estimated deck washdown volumes by comparing vessel gross tons for these vessels with that of a typical passenger vessel. EPA estimated passenger vessels are 35 gross tons based on 50% of passenger vessels being 45 feet long (USEPA 2010b). The example below shows how EPA used the vessel gross tons and deck washdown discharge volume for passenger vessels to estimate the deck washdown flow for large ferries.

Example: 1,451 gross tons (large ferry)/35 gross tons (passenger vessel) x 27 g/day = 1,083 g/day

Table 10. Estimated Deck Washdown Discharge Rates used in EPAs BE Analysis.

Vessel Type	Estimated Deck Washdown Discharge Rate	
	m^3/day	gallons/day
Barges	0.88	232
Cargo/Bulk Carriers	19.9	5,259
Commercial Fishing (average shrimper, trawler and troller)	0.39	103
Large Cruise Ships	152.9	40,396
Medium Cruise Ships	13.1	3,461
Large Ferries	4.1	1,083
Oil and Gas Tankers	31.1	8,216
Passenger Vessels (average water taxi, tour boat)	0.10	27
Utility Boats (average fire boat, supply boat, tow boat and tug boat)	0.05	14

EPA estimated concentrations of deckwash pollutants based on data for the vessel types examined in the Report to Congress (Table 11). Deck washwater was collected from a total of 32 vessels: 11 fishing vessels, nine tugboats, six tow/salvage vessels, two tour boats, a water taxi, a fire boat, a supply boat, and a recreational boat. A total of 88 analytes were detected in deckwass samples. Table 11 includes the point estimates used in EPAs BE plus additional information for these analytes from the Report to Congress, which provides details on the detection frequency and distribution of all analytes detected.

Table 11. Estimated Deckwash Pollutant Concentrations used in EPAs Biological Evaluation.

Pollutant	Point Estimate Used in BE	Detection frequency	Mean (range)	50th/75th/90th Percentiles[b]
Total Phosphorus (mg/L)	1.7	31/31	1.7 (0.06-22)	0.79/1.6/2.9
Total Residual Chlorine (mg/L)	0.12	7/31	0.12 (nd[a]-0.8)	nc[c]/nc/0.37
Total Suspended Solids (TSS, mg/L)	170	32/32	170 (27-530)	120/250/470
Biochemical Oxygen Demand (BOD, mg/L)	110	30/32	110 (nd-830)	56/92/370
Hexane Extractable Material (HEM, mg/L)	14 (130)	26/29	14 (nd-130)	2.8/12/39
Silica Gel Treated HEM (SGT-HEM, mg/L)	7.0 (84)	22/29	7 (nd-84)	1.7/3.8/13

Aluminum, Total (µg/L)	3400	30/31	3400 (nd-13000)	1900/4700/8300
Cadmium, Dissolved (µg/L)	1.3	2/31	1.3 (nd-22)	nc/nc/nc
Copper, Dissolved (µg/L)	42	29/31	42 (nd-200)	23/59/120
Lead, Dissolved (µg/L)	6	15/31	6 (nd-54)	nc/4.7/19
Zinc (dissolved, µg/L) ()	260	31/31	260 (16-1200)	120/430/620
Bis (2-ethylhexyl) phthalate (µg/L)	13	1/2	4.7 (nd-6.7)	6.7/6.7/6.7
Bromodichloromethane (µg/L)	4.7	1/2	13 (nd-1.2)	1.2/1.2/1.2
Dibromochloromethane (µg/L)	13	1/2	13 (nd-0.7)	0.7/0.7/0.7

[a]Nondetect (censored) concentrations were replaced with ½ of the reporting limit for calculating average concentrations.
[b] Percentiles are the concentrations of each analyte below which at least that percentage of the values fall. So the 75th percentile is the concentration below which at least 75% of the observations were found.
[c] Distribution statistics were only calculated when analytes were detected at a sufficient frequency.

Engine Wet Exhaust Effluent. Engine wet exhaust effluent is generated when gasoline or diesel engine cooling water is injected into the engine exhaust. The engine cooling water decreases the exhaust temperature, reduces engine noise, and reduces exhaust emissions. Engine wet exhaust discharge rates can range from 5 to 10 gallons per minute for electrical generators to more than 100 gallons per minute on larger diesel engines operating at high inputs. Most passenger vessels, a large number of fishing vessels, and utility vessels use engines that generate wet engine exhaust. Large commercial vessels occasionally operate small auxiliary craft that discharge engine wet exhaust (e.g., life boats on cruise ships); however, discharge volumes for these vessels are negligible. EPA analyzed the risk of this discharge in its BE due to the high number of vessels discharging, the large volume and frequency of discharges, along with the potentially high risk posed by the concentration and types of pollutant discharged.

Large engines, like those found on VGP eligible vessels (e.g., large ferries, medium and large cruise ships, tankers and cargo/bulk container ships) use dry engine exhaust systems and therefore do not generate engine wet exhaust effluent. Barges do not have engines and therefore do not generate engine wet exhaust effluent. Estimated volume of wet engine exhaust from the EPA report to congress were directly applied to EPA's BE (USEPA 2010b).

Table 12. Estimated Propulsion and Generator Engine Wet Exhaust Discharge Rates used in EPAs Analysis.

Vessel Type	Estimated Propulsion Engine Wet Exhaust Discharge Rate		Estimated Generator Engine Wet Exhaust Discharge Rate	
	m3/day	gallons/day	m3/day	gallons/day
Commercial Fishing[a]	15.8	4,161	1.41	372
Passenger Vessels (e.g., water taxis, tour boats)[b]	41	10,832	6.45	1,704
Utility Boats (fire boat, tugs)[c]	36.3	9,590	1.80	475

EPA estimated concentrations of engine wet exhaust pollutants based on data from the Report to Congress (Table 13). A total of 90 analytes were detected. EPA collected engine cooling water discharge samples from a variety of vessel classes with different engine types. Two of the sampled vessels are recreational vessels and are not study vessels. These vessels and four of the six sampled tow/salvage vessels (those with outboard propulsion engines) were manufactured for pleasure. EPA sampled engine effluent from these vessels because all of the sampled engines can be installed on either recreational or nonrecreational vessels and are representative of engines on study vessels. Contaminants of concern include copper and volatile/semivolatile organic chemicals (sVOCs/VOCs). Table 13 includes additional information for these analytes from the Report to Congress, which provides details on the detection frequency and distribution of all analytes detected.

Table 13. Estimated Engine Wet Exhaust Pollutant Concentrations used in EPAs Biological Evaluation.

Pollutant	Point Estimate Used in BE	Detection frequency	Mean (range)	50th/75th/90th Percentiles[b]
Copper, Dissolved (µg/L)	16	12/13	16 (nd[a]-53)	6.6/23/51
Benzene (µg/L)	12	9/12	12 (nd-120)	2.3/5.4/84
Benzo (a)anthracene (µg/L)	3.3	1/12	3.3 (nd-18)	nc/nc/13
Benzo (a)pyrene (µg/L)	3.2	1/12	3.2 (nd-16)	nc/nc/11
Benzo (b)fluoranthene (µg/L)	2.8	1/12	2.8 (nd-11)	nc/nc/7.8
Benzo (k)fluoranthene (µg/L)	3.1	1/12	3.1 (nd-15)	nc/nc/11
Chrysene (µg/L)	1.7	1/12	3.3 (nd-18)	nc/nc/12
Indeno (1,2,3-cd) pyrene (µg/L)	3.3	1/12	2.5 (nd-8)	nc/nc/5.6

[a]Nondetect (censored) concentrations were replaced with ½ of the reporting limit for calculating average concentrations.
[b] Percentiles are the concentrations of each analyte below which at least that percentage of the values fall. So the 75th percentile is the concentration below which at least 75% of the observations were found.
[c] Distribution statistics were only calculated when analytes were detected at a sufficient frequency.

Exhaust Gas Scrubber Washwater Effluent. Exhaust gas scrubber washwater discharge occurs as a result of cleaning the exhaust gas system on marine diesel engines. The washwater discharge can be highly acidic, and can also contain traces of oil, polycyclic aromatic hydrocarbons (PAHs), heavy metals and nitrogen. Washwater volumes of 2.8 million gallons per day are estimated from a 10 MWh engine. While EPA anticipates very low numbers of vessels will potentially be releasing this discharge, the risk of this discharge was analyzed in the BE due to the very large volume and frequency of discharges, along with the potentially high risk posed by the concentration and types of pollutant discharged.

Exhaust gas scrubbers would be used on large vessels such as oil and gas tankers, cargo/bulk carriers, large ferries and medium and large cruise ships burning high-sulfur bunker oil (USEPA 2011b). The exhaust gas scrubbers remove sulfur dioxide generated by the vessel engines during combustion of the bunker oil. Exhaust gas scrubber systems using seawater typically generate 45 m^3 (11,900 gallons) of washwater per megawatt-hour (MWh) for scrubbing (MEPC 2008). Closed freshwater exhaust gas scrubber systems have much smaller discharge rates than seawater scrubbers. Freshwater closed loop scrubbers generate 0.1 to 0.3 m^3/MWh (26 to 79 gallons/MWh) of scrubber washwater (MEPC 2008). To estimate exhaust gas scrubber washwater flows for vessels like tankers, cargo/bulk carriers, ferries and cruise ships, EPA used information on the engine sizes for each type of vessel.

Table 14 shows the range of engine sizes by vessel type and the estimated seawater and freshwater exhaust gas scrubber washwater flow rates based on engine size. The size of the propulsion engines for large cruise ships and large ferries was obtained from EPA's Technical Development Document for Exhaust Gas Scrubber Washwater Effluent (USEPA 2011b). EPA assumed that the power requirements for propulsion engines on large ferries are similar to those for medium cruise ships. For oil and gas tankers and cargo/bulk carriers, EPA used information for the 80 MW (109,000 HP) engine onboard the *Emma Maersk* large container ship (Technology 2011). EPA assumed that for smaller vessels such as tour boats, water taxis, fishing boats and tug boats using low sulfur fuels will be a less expensive compliance alternative than installing and operating exhaust gas scrubbers; therefore, exhaust gas scrubber washwater flows for these vessel types were not estimated.

Table 14. Estimated Exhaust Gas Scrubber Washwater Discharge Rates used in EPAs Analysis

Vessel Type	Average Propulsion Power (MW)	Estimated Saltwater Exhaust Gas Scrubber Washwater Discharge Rate		Estimated Freshwater Exhaust Gas Scrubber Washwater Discharge Rate	
		m^3/hr	gallons/hr	m^3/hr	gallons/hr
Cargo/Bulk Carriersa	80	3,600	951,000	16	4,227
Large Cruise Ships	45	2,025	535,000	9	2,377
Medium Cruise Ships	4.8	216	57,000	1	264

Vessel Type	Average Propulsion Power (MW)	Estimated Saltwater Exhaust Gas Scrubber Washwater Discharge Rate		Estimated Freshwater Exhaust Gas Scrubber Washwater Discharge Rate	
		m^3/hr	gallons/hr	m^3/hr	gallons/hr
Large Ferries	4.8	216	57,000	1	264
Oil and Gas Tankers[a]	80	3,600	951,000	16	4,227

EPA estimated concentrations of exhaust gas scrubber washwater pollutants based on data from the Zaandam Scrubber Trial (USEPA 2011b, Holland America Line and Hamworthy – Krystallon. 2010), Table 15). Data provided in the trial report were point values and did not include details on analytical observations.

Table 15. Estimated Exhaust Gas Scrubber Washwater Pollutant Concentrations used in EPAs Biological Evaluation.

Pollutant	Point Estimate used in BE (µg/L)
Arsenic (dissolved)	81
Arsenic (total)	81
Copper (dissolved)	15
Nickel (dissolved)	12
Selenium (dissolved)	94
Selenium (total)	100
Benzo (a)-anthracene	0.262
Benzo (a)-pyrene	0.136
Chrysene	0.373

Fish Hold and Fish Hold Cleaning Effluent. All fishing vessels generate some form of fish hold effluent. Fish hold effluent is composed of seawater, ice-melt, or ice slurry collected inside fish hold tanks. Fish hold effluent contains pollutants which result from seafood catch and other onboard vessel sources. These pollutants can include biological wastes, metals, and nutrients, as well as wastewater resulting from fish hold cleaning activities.

Table 16 shows the volume of water discharged from various commercial fishing vessels during fish holds in operation and during tank cleaning. Fish hold effluent discharge rates can range from as low as 211 gal/day for a gillnetter to as high as 5,090 gallons per day for a tender vessel while cleaning volumes range from 283 gal/day for purse seiners to 58 gallons per day for trawlers (USEPA 2010b). The volume of fish hold effluent can vary from less than 100 gallons during off-loading to tens of thousands of gallons for large fish tenders. EPA analyzed the risk of this discharge in its BE due to the high number of vessels discharging, the large volume and frequency of discharges, along with the potentially high risk posed by the concentration and types of pollutant discharged.

Table 16. Fish Hold Effluent and Cleaning Effluent Discharge Rates (USEPA 2010).

Commercial Fishing Vessel Type	Estimated Fish Hold Effluent Discharge Rate		Estimated Fish Hold Cleaning Effluent Discharge Rate	
	m^3/day	gal/day	m^3/day	gal/day
Gillnetter	0.80	211		
Lobster Boat	2.83	748		
Longliner	2.83	748		
Purse Seiner	16.3	4,306	1.07	283
Shrimper	1.25	330		
Tender	19.3	5,099	0.66	174
Trawler	1.25	330	0.22	58
Troller	3.04	803	0.66	174

EPA estimated concentrations of fish hold and fish hold cleaning effluent pollutants based on data for the vessel types examined in the Report to Congress (Table 17). Contaminants identified in fish hold and fish hold cleaning effluent include metals and volatile/semivolatile organic chemicals (sVOCs/VOCs).

Table 17. Estimated Fish Hold and Fish Hold Cleaning Effluent Pollutant Concentrations used in EPAs Biological Evaluation.

Pollutant	Fish Hold Effluent				Fish Hold Cleaning Effluent			
	BE Point Estimate	Detection Frequency	Mean (range)	50th/75th/90th Percentile[b]	BE Point Estimate	Detection Frequency	Mean (range)	50th/75th/90th Percentile
Ammonia Nitrogen	12	25/26	12 (nd[a]-160)	2.1/6.7/160	16	7/9	16 (nd-97)	4.8/18/97
Total Phosphorous	13	25/26	13 (nd-76)	9.7/17/76	8.5	7/9	8.5 (nd-20)	11/17/20
Total Residual Chlorine	0.096	10/26	0.096 (nd-0.3)	nc[c]/0.13/0.3	0.29	6/9	0.29 (nd-1.5)	0.11/0.29/1.5
Total Suspended Solids (TSS)	210	26/26	210 (10-1100)	130/190/1100	190	9/9	190 (16-460)	84/400/460
Biochemical Oxygen Demand (BOD)	840	24/26	840 (nd-5100)	440/830/5100	470	6/9	470 (nd-1800)	300/770/1800
Hexane Extractable Oil and Grease (HEM)	3.2	18/26	3.2 (nd-16)	1.5/2.9/16	5.4	6/9	5.4 (nd-28)	1.4/4.2/28
Silica Gel Treated-Hexane Extractable Oil and Grease (SGT-HEM)	3.4	15/26	3.4 (nd-4.4)	0.98/2.2/4.4	4.9	4/9	4.9 (nd-12)	nc/2.8/12
Aluminum (total)	827	24/26	490 (nd-1000)	670/850/1000	1100	9/9	1100 (850-1700)	930/1500/1700
Arsenic (dissolved)	31	10/26	31 (nd-350)	nc/5.7/350	22	5/9	22 (nd-97)	5.3/38/97
Arsenic (total)	40	16/26	40 (nd-380)	4.8/13/380	35	5/9	35 (nd-150)	8.7/64/150
Cadmium (dissolved)	0.77	1/26	0.77 (nd-1.4)	nc/nc/1.4				
Copper (dissolved)	96	23/26	96 (nd-920)	15/38/920	34	8/9	34 (nd-180)	12/32/180
Lead (dissolved)	0	3/26	2.3 (nd-8)	nc/nc/8	2.7	1/9	2.7 (nd-8.7)	nc/nc/8.7
Selenium (dissolved)	9.2	6/26	9.2 (nd-61)	nc/2.5/61	6	1/9	6 (nd-14)	nc/nc/14
Zinc (dissolved)	180	26/26	180 (24-790)	120/240/790	190	8/9	190 (nd-640)	53/420/640

[a]Nondetect (censored) concentrations were replaced with ½ of the reporting limit for calculating average concentrations.
[b] Percentiles are the concentration below which that percentage of the values fall. So 75 percent of observations occur below the 75th percentile.
[c] Distribution statistics were only calculated when analytes were detected at a sufficient frequency.

Graywater and Graywater Mixed with Sewage. Graywater volumes depend primarily on the type of vessel and the number of passengers and crew onboard (Table 18). For example, cruise ships generate more graywater than cargo ships since they have many more passengers and crew. Further, cruise ships also generate higher graywater volumes per capita than cargo ships because passengers and crew on these vessels use galleys and accommodations (sinks and showers) to a greater extent than crew on cargo ships. Estimated graywater generation rates reported in response to EPA's 2004 cruise ship survey ranged from 36 to 119 gallons/day/person, with an average of 67 gallons/day/person (USEPA 2008) for large cruise ships. Finland's Ministry of the Environment estimated cruise ship grey water generation at 120 liters per day (32 gallons/day/person) (BMEPC 2007). During EPA's 2004 Alaskan cruise ship sampling program, graywater generation on one ship was 45 gallons/day/person (USEPA, 2006).

Table 18. Estimated Graywater Discharge Rates Used in EPAs BE.

Vessel Type	Number of Passengers and Crew[a]	Estimated Graywater Discharge Rate[d]	
		gal/day	m^3/day
Barges	4	61[e]	0.23
Cargo/Bulk Carriers	25	1,125	4.2
Commercial Fishing Vessels	29	1,305	4.9
Large Cruise Ships	3,455	155,475	588
Medium Cruise Ships	315	14,175	53.6
Large Ferries	559	25,155	95.2
Oil and Gas Tankers	22	990	3.7
Passenger Vessels (tour boats, water taxis, etc.)	282[b]	12,690	48
Utility Vessels (tug boats, off-shore vessels, etc.)	6[c]	1	270

a) Number of crew and passengers from EPA's NOI Report (USEPA, 2011e) unless otherwise specified.
b) Average number of passengers and crew calculated based on information gathered from both personnel communication with operators and web site marketing data for tour boats, dinner cruise vessels, wedding party boats, whale watching tours, and shuttle service vessels. Vessels do not provide overnight accommodations.
c) Based on personal contact with AEP River Operations.
d) Based on a graywater generation rate of 45 gallons per person per day (USEPA, Sampling Episode Report for Holland America Veendam, Sampling Episode 6503) Washington, DC (USEPA. 2006).
e) Average flow adjusted based on an estimated 33.8% of barges are manned and would therefore generate graywater (USEPA, 2011e, Section 2.2.1.5 and Figure 2.1-1)

Graywater can contain high levels of pathogens, nutrients, soaps and detergents, and organics. EPA analyzed the risk of this discharge in its BE due to the high number of vessels discharging, the large volume and frequency of discharges, along with the potentially high risk posed by the concentration and types of pollutant discharged. EPA estimated concentrations of graywater pollutants based on data for the vessel types examined in the Report to Congress (Table 19). Contaminants identified in graywater include metals oil and grease and nutrients.

Table 19. Estimated Graywater Pollutant Concentrations used in EPAs Biological Evaluation.

Pollutant	Point Estimate Used in BE	Detection Frequency	Mean (range)	50th/75th/90th Percentiles[c]
Ammonia as Nitrogen (NH3-N, mg/L)	1.3 (2.1)[a]	8/8	1.3 (0.19-4.5)	0.75/1.8/4.5
Total Phosphorus (mg/L)	1.4 (10)	8/8	1.4 (0.42-3.4)	1.2/2.2/3.4
Total Residual Chlorine (mg/L)	0.12	6/8	0.12 (nd[b]-0.11)	0.02/nc[d]/0.11
Total Suspended Solids (TSS) (mg/L)	52	8/8	52 (14-81)	58/69/81
Hexane Extractable Material (HEM) (mg/L)	39 (149)	8/8	39 (9.4-100)	29/68/100
Silica Gel Treated HEM (SGT-HEM) (mg/L)	8.1 (36.6)	6/8	8.1 (nd-35)	1.5/9.4/35
Aluminum (total, µg/L)	380	8/8	380 (50-910)	420/540/910
Arsenic (total, µg/L)	2	2/8	2 (-2.9)	/1.5/2.93
Copper (dissolved, µg/L)	55	8/8	55 (5.3-280)	17/60/280
Lead (dissolved, µg/L)	2.5	4/8	2.5 (nd-6)	1.1/4.2/6
Zinc (dissolved, µg/L)	400	8/8	400 (70-1500)	240/610/1500

[a] Values in parentheses are for large cruise ships.
[b] Nondetect (censored) concentrations were replaced with ½ of the reporting limit for calculating average concentrations.
[c] Percentiles are the concentrations of each analyte below which at least that percentage of the values fall. So the 75th percentile is the concentration below which at least 75% of the observations were found.
[d] Distribution statistics were only calculated when analytes were detected at a sufficient frequency.

Stern Tube Packing Gland Effluent and Other Oil/Water Interfaces. A stuffing box packed with greased flax rings is designed to leak a few drops per minute of ambient water (EPA estimate, 4 to 8 gallons per day (USEPA 2010b) to cool the gland when a vessel is underway. Stuffing boxes are also used to seal rudder stocks that penetrate the hull below the waterline. The packing gland effluent water is often collected in a segregated section of the bilge that contains an automatic bilge pump.

To estimate stern tube effluent discharge rates for the other vessel types requiring detailed analysis for the BE, EPA used the size of a utility vessel (measured as gross tons), compared this vessel size to the other vessel types, and applied an average stern tube packing gland discharge rate of 6 gallons per day as shown in the example calculation below.

Example: 25 gross tons (fishing vessel)/80 gross tons (utility vessel) x 6 g/day = 2 g/day

Pollutants in the stern tube packing gland effluent include metals, oil and grease, suspended solids, organics, and phthalates. Oil to sea interfaces include any mechanical or other equipment where seals or surfaces may release small quantities of oil and grease into the sea. Examples

include controllable pitch propellers, rudder bearings and wire ropes and cables that have lubricated (greased) surfaces which are submerged in seawater during use. EPA analyzed the risk of this discharge in its BE due to the high number of vessels discharging, the large volume and frequency of discharges, along with the potentially high risk posed by the concentration and types of pollutant discharged.

Table 20 shows the vessel sizes and the estimated stern tube packing gland discharge rates for the vessel types requiring detailed analysis. Flows range from as little as 2 gal/day for commercial fishing vessels to as great as 4,000 gal/day for large cruise ships.

Table 20. Estimated Stern Tube Packing Gland Effluent Discharge Rates used in EPAs BE.

Vessel Type	Vessel Size (gross tons)[a]	Estimated Stern Tube Packing Gland Effluent Discharge Rates	
		m^3/day	gallons/day
Barges	705	0[e]	0[e]
Cargo/Bulk Carriers	15,939	4.5	1,195
Commercial Fishing	25b	0.008	2
Large Cruise Ships	53,526	15.1	4,014
Medium Cruise Ships	4,584	1.3	343
Large Ferries	1,451	0.4	108
Oil and Gas Tankers	24,908	7.1	1,868
Passenger Vessels (e.g., water taxis, tour boats)	35[c]	0.01	2.6
Utility Boats (tugs and tow boats)	80[d]	0.02	6

a) Vessel size from EPA's NOI Report (USEPA, 2011e) unless otherwise specified.
b) Vessel gross tons estimated assuming 50% of commercial fishing vessels are 40 feet long (USEPA, 2010).
c) Vessel gross tons estimated assuming 50% of passenger vessels are 45 feet long (USEPA, 2010).
d) Vessel gross tons estimated assuming 50% of utility vessels are 65 feet long (USEPA, 2010).
e) Barges do not have propeller shafts and associated stern tube packing glands.

EPA estimated concentrations of stern tube packing gland effluent pollutants based on data for the vessel types examined in the Report to Congress (Table 21). Contaminants identified in stern tube packing gland effluent include metals and volatile/semivolatile organic chemicals (sVOCs/VOCs).

Table 21. Estimated Stern Tube Packing Gland Effluent Pollutant Concentrations used in EPAs Biological Evaluation.

Pollutant	Point Estimate Used in BE	Detection Frequency	Mean (range)	50th/75th/90th Percentiles[b]
Total Phosphorus (mg/L)	0.13	7/9	0.13 (nd[a]-0.25)	0.1/0.22/0.25
Total Suspended Solids (TSS, mg/l)	59	9/9	59 (5.6-270)	28/81/270
Hexane Extractable Material (HEM, mg/l)	14	5/9	14 (nd-67)	1.65/23/67
Chromium, (dissolved, µg/L)	13	5/9	19 (nd-110)	3.8/20/110
Silica Gel Treated HEM (SGT-HEM, mg/l)	19	5/9	13 (nd-56)	1.7/19/56
Copper (dissolved, µg/L)	22	4/9	22 (nd-92)	nc[c]/38/92
Nickel (dissolved, µg/L)	210	6/9	210 (nd-1000)	13/370/1000
Nickel (total, µg/L)	610	8/9	610 (nd-3200)	45/970/3200
Bis (2-ethylhexyl)phthalate (µg/L)	4.7	3/9	4.7 (nd-24)	nc/4.1/24
Tetrachloroethene (µg/L)	1.4	1/9	1.4 (nd-0.2)	nc/nc/0.2

[a]Nondetect (censored) concentrations were replaced with ½ of the reporting limit for calculating average concentrations.
[b] Percentiles are the concentrations of each analyte below which at least that percentage of the values fall. So the 75th percentile is the concentration below which at least 75% of the observations were found.
[c] Distribution statistics were only calculated when analytes were detected at a sufficient frequency.

Underwater Ship Husbandry Discharges. Although discharges from underwater ship husbandry is considered a low volume and low frequency discharge generated by a low number of vessels, EPA analyzed the risk of this discharge in its BE due to the high potential risk posed by the concentration and types of pollutant discharged. Underwater husbandry discharge volumes were not estimated because EPA was unable to find available data to accurately estimate volume of these discharges. Furthermore, EPA round approached ANS risk qualitatively, so volume estimates were not necessary. EPA's Report to Congress did provide copper release estimates for underwater ship husbandry events (Table 22).

Table 22. Dissolved Copper Release from Vessel AFSs During an Underwater Hull Cleaning "Event" (Schiff et al. 2003, after USEPA 2010).

AFS	Cleaning Method	Copper Release (µg/cm2/event)
Epoxy copper antifouling paint	Less abrasive management practices	8.6
	No management practices	17.4
Hard vinyl/Teflon copper antifouling paint	Less abrasive management practices	3.8
	No management practices	4.2
Biocide-free coating	Less abrasive management practices	0.03
	No management practices	0.05

Discharges not Evaluated in the BE. EPA's decision process identified a number of discharges that would not be evaluated in the BE dues to the relative number of vessels releasing the discharge, the volume of the discharge, and/or the potential risk posed by the concentration and constituents of that discharge. Table 23 summarizes each discharge not evaluated in the BE and EPAs decision process for each. Details on discharge estimates and other information EPA used in arriving at these decisions are provided in footnotes to table 3-3 in the BE.

Table 23. Decision Process for those Vessel Discharges Not Evaluated in EPA's BE.

Vessel Discharge	Description	Number of Vessels Discharging	Discharge Volume and Frequency	Amount and Types of Pollutants Present in the Discharge
Firemain Systems	Firemain systems are found on a large number of vessels and draw in water through the sea chest to supply water for fire hose stations and sprinkler systems. Systems are activated during testing or during an actual fire. Small amounts of metals may be added to the fire water from vessel piping system.	High	Low	Low
Well Deck Discharges	The welldeck is a floodable platform used for launching or loading small satellite vessels, vehicles, and cargo from select vessels. Welldeck discharges may include water from precipitation, welldeck and storage area washdowns, equipment and engine washdowns, and leaks and spills from stored machinery. Potential constituents of welldeck discharges include fresh water, distilled water, firemain water, graywater, air-conditioning condensate, sea-salt residues, paint chips, wood splinters, dirt, sand, organic debris and marine organisms, oil, grease, fuel, detergents, combustion by-products, and lumber treatment chemicals.	Low	High	Low
Aqueous Film Forming Foam (AFFF)	Firefighting agent added to fire suppression systems on some vessels to create foam. Used infrequently (annually or semi-annually) to test equipment for maintenance, certification, or training. Constituents include fluorosurfactants and/or fluoroproteins.	Low	Low	Potentially High
Freshwater Layup	Freshwater layup is generated when a vessel is pier side or in port for more than a few days, the main steam plant is shut down, and the condensers do not circulate. A freshwater layup includes replacing the seawater in the system with potable or surrounding freshwater (e.g., lake water). Freshwater layup discharges can be as large as 6,000 gallons per evolution and can contain residual saltwater, freshwater, tap water, and possibly metals leached from the pipes or machinery.	Low	Low	Low
Gas Turbine Water Wash	Gas turbines are used for propulsion and electricity generation on some vessels. Occasionally, they must be cleaned to remove byproducts that can	Low	Low	Potentially High

Table 23. Decision Process for those Vessel Discharges Not Evaluated in EPA's BE.

Vessel Discharge	Description	Number of Vessels Discharging	Discharge Volume and Frequency	Amount and Types of Pollutants Present in the Discharge
	accumulate and affect their operation. Although not subject to the VGP but similar in operation to VGP vessels regarding this discharge, large naval vessels can generate up to 244 gallons of washwater per day. Wash water can include salts, lubricants, and combustion residuals.			
Boiler/Economizer Blowdown	Boiler blowdown occurs on vessels with steam propulsion or a steam generator and is used to control the concentration of scaling constituents in boiler systems. Boiler blowdown are infrequent, of short duration (seconds), in small volumes, and at high pressure. The blowdown can contain water and steam or sludge-bearing water at elevated temperatures (above 325°F). The discharge can contain metals or boiler water treatment chemicals.	Low	Low	Low
Motor Gasoline and Compensating Discharge	Motor gasoline is transported on vessels to operate vehicles and other machinery. As the fuel is used, ambient water is added to the fuel tanks to replace the weight. This ambient water is discharged when the vessel refills the tanks with gasoline or when performing maintenance. Most vessels are designed not to have motor gasoline and compensating discharge. The volume of the compensating discharge is expected to range from less than 50 gallons to up a few hundred gallons. The discharge can contain small amounts of fuel and other fuel-related pollutants.	Low	Low	Potentially High
Cathodic Protection	Nearly all vessels having steel hulls or metal hull appendages use cathodic protection systems to prevent corrosion. Based on underwater hull inspections and maintenance records, one-half of anodes are consumed after three years. The primary pollutant released from cathodic protection is zinc. Average pierside and underway zinc generation rates are 1.3 x 10-6 and 5.1 x 10-6 (lb zinc/square foot of underwater surface area)/hr, respectively. Concentrations modeled for military vessels using a mixing zone envelope approach indicate that zinc, aluminum and mercury concentrations were below WQS within 0.5, 0.1 and 0.1 feet from the vessels surface, respectively (USDOD/USEPA 1999).	High	Low	Low
Non-oily	Some larger vessels are expected to have some non-oily machinery discharges,	Low	Low	Low

Table 23. Decision Process for those Vessel Discharges Not Evaluated in EPA's BE.

Vessel Discharge	Description	Number of Vessels Discharging	Discharge Volume and Frequency	Amount and Types of Pollutants Present in the Discharge
Machinery Wastewater	such as distilling plants start-up discharge, chilled water condensate drains, fresh-and saltwater pump drains, and potable water tank overflows. These flows are generally low in volume and are not expected to contain significant amounts of pollutants.			
Refrigeration and Air Condensate Discharge	Condensation from cold refrigeration or evaporator coils of air conditioning systems drips from the coils and collects in drip troughs which typically empty to a drainage system. Large numbers of vessels are equipped with refrigeration systems to keep food and other perishable items from spoiling. Air conditioning systems are also used for passenger and crew comfort. Condensates may contain very small amounts of pollutants such as metals derived from vessel piping systems.	High	Low	Low
Coal Ash from Coal Fired Propulsion Systems on Ferries	Combusting coal in boilers produces a residue of combustible flue constituents known as ash. Some of the ash consists of very fine particles that are entrained in the flue gas and carries out of the furnace (fly ash), while heavier ash settles at the bottom of the boiler (bottom ash). Bottom ash may be hydraulically conveyed (i.e., sluiced with water). The discharge of coal ash slurry from coal fired propulsion systems on a ferry is authorized by the 2008 VGP until December 19, 2012.	Low	High	Potentially High
Seawater Cooling Overboard Dischargenn	Seawater cooling systems use ambient water to absorb the heat from heat exchangers, propulsion systems, and mechanical auxiliary systems. The water is typically circulated through an enclosed system that does not come in direct contact with machinery, but still may contain sediment from water intake, traces of hydraulic or lubricating oils, and trace metals leached or eroded from the piping within the system. Additionally, because it is used for cooling, the effluent will have an increased temperature.	High	Low	Low
Distillation and Reverse Osmosis Brine	Discharges of brine can occur on vessels that do not bunker potable water but instead use onboard plants to distill seawater or desalinate seawater using reverse osmosis (RO) to generate fresh water. Distillation units generate brine	Low	High	Low

Table 23. Decision Process for those Vessel Discharges Not Evaluated in EPA's BE.

Vessel Discharge	Description	Number of Vessels Discharging	Discharge Volume and Frequency	Amount and Types of Pollutants Present in the Discharge
	at a rate of 17 gallons of brine for every gallon of fresh water produced. Reverse osmosis units generate approximately 4 gallons of brine for every gallon of fresh water produced. The three sources of the constituents of brine discharge are: 1) influent seawater; 2) anti-scaling treatment chemicals; and 3) the purification plant components, including heat exchangers, casings, pumps, piping and fittings. The primary constituents of the brine discharge are identical to those in seawater; however, they are more concentrated due to volume reduction.			
Seawater Piping Biofouling	Some vessels that use seawater cooling systems introduce anti-fouling compounds (e.g., sodium hypochlorite) in their interior piping and component surfaces to inhibit the growth of fouling organisms. These anti-fouling compounds are then typically discharged overboard. Most vessels that have seawater piping systems are expected to use piping materials such as copper to prevent biofouling rather than injecting high concentrations of anti-fouling compounds into their piping systems.	Low	Low	Potentially High
Elevator Pit Effluent	Large vessels with multiple decks are equipped with elevators to facilitate the transportation of maintenance equipment, people, and cargo between decks. A pit at the bottom of the elevator shaft collects small amounts of liquids and debris from elevator operations and deck washdown and runoff depending on the elevator configuration. Water entering the elevator pit can contain materials that were on the deck, including fuel, hydraulic fluid, lubricating oil, residual water, and AFFF. The runoff may also contain lubricant applied to the elevator doors, door tracks, and other moving elevator parts. Residue in the elevator car from the transport of materials may also be washed into the elevator pit. The cleaning solvent used during maintenance cleaning operations, as well as liquid wastes generated by the cleaning process, can also drain into the elevator pit sump.	Low	Low	Potentially High
Sonar Dome	Water is used to maintain the shape and pressure of domes that house sonar	Low	Low	Low

Table 23. Decision Process for those Vessel Discharges Not Evaluated in EPA's BE.

Vessel Discharge	Description	Number of Vessels Discharging	Discharge Volume and Frequency	Amount and Types of Pollutants Present in the Discharge
Discharge	detection, navigation, and ranging equipment on large vessels. Discharges occasionally occur when the water must be drained for maintenance or repair or from the exterior of the sonar dome. Although not subject to the VGP but similar in operation to VGP vessels regarding this discharge, sonar dome discharge volumes on naval vessels can range from 300 gallons per event up to 74,000 gallons per event. Pollutant levels are expected to be low due to the ban on the use of tributyl tin.			
Underwater Ship Husbandry Discharges	Underwater ship husbandry is grooming, maintenance, and repair activities of hulls or hull appendages performed while the vessel is located in the water. Underwater ship husbandry discharges can contain aquatic organisms and residue such as rust and biocide from AFC. Underwater ship husbandry is typically performed only when excessive biological growth is causing vessel drag and excessive fuel consumption outside of regular dry dock inspections.	Low	Low	Potentially High
Chain Locker Effluent	Chain locker effluent is water that drips from the anchor chain and anchor during anchor retrieval. Discharge volumes are small and chain locker effluent is expected to contain zinc, rust, paint, grease, and any constituents from the firemain water. The small volume of chain locker effluent results in small mass loadings and provides opportunity for the transfer of non-indigenous species.	High	Low	Low

Estimates of Exposure Resulting from Discharges Authorized by the VGP and sVGP

This section summarizes the modeling approach EPA used to estimate exposure of threatened and endangered species and their critical habitat to VGP-authorized discharges. Greater detail is provided in section 5 of EPA's BE. Representative Action Areas (RAAs) were selected by EPA after consultation with the Services to be used to evaluate the potential exposure and effects of permitted discharges on listed aquatic and aquatic-dependent species, as well as their designated critical habitats. The following selection criteria were used to identify the RAAs evaluated in the BE:

- High density of vessels covered under the VGP and sVGP to capture a reasonable "worst case" scenario;
- Varying percentages of vessel types present to capture the variability in pollutant loading rates;
- Selection of RAAs that provide broad coverage of diverse ecosystem types and broad geographical representation;
- Selection of "special interest" locations with known or expected sensitivities for specific taxa and their critical habitats (i.e., Pacific coast salmonid populations, sea turtles in Gulf of Mexico, etc.) and/or specific vulnerabilities (e.g., historic introduction and spread of ANS in the Great Lakes).

EPA's NOI Summary Report and EPA's Report to Congress on Vessel Discharges were used to identify the top Home/Hailing Ports for each vessel type. For vessels expected to be covered under the sVGPs (large and small), New York, NY; Houston, TX, and Seattle, WA were consistently among the top 10 port cities, as well as other ports such New Orleans, LA; Miami, FL; Los Angeles, CA; Norfolk, VA; Pittsburgh, PA; and St. Louis, MO (to name a few). Among these port cities, nearly all are representative of estuarine/marine ecosystems, except for inland port cities such as Jeffersonville (Indiana), Pittsburgh, St. Louis and Cincinnati (top port cities for barges on large inland rivers). Port cities on a large freshwater lake (Great Lakes) system do not appear to be among the most heavily trafficked. From this initial list of port cities, EPA selected seven RAAs for detailed evaluation in the BE that were both ecologically and geographically diverse and captured several special interest locations identified during preliminary informal consultation meetings with the Services (Table 24).

Table 24. RAAs Selected for Exposure Assessment Modeling to Represent Water Bodies and Ecosystems Inhabited by Listed Resources.

Reference Port/Harbor	Waterbody Represented	Ecosystem Represented	Justification
Duluth, MN	Lake Superior Harbor	Freshwater [Great] Lake	Major port in sensitive "cold-water" Great Lakes area with known ANS issues.
Houston, TX	Galveston Bay	Estuarine/Marine	RAA with large population of oil and gas tankers, barges, and cargo carriers. This area also encompasses nesting ground of listed sea turtles.
Miami, FL	Biscayne Bay	Estuarine/Marine	RAA with significant populations of cargo/bulk carriers and large population of cruise ships. Area encompasses extensive listed Johnson's seagrass beds and represents a portion of Florida's coral reef tract. In addition, Miami was requested to be an additional RAA during informal consultation with Services, in light of the presence of endangered species and their critical habitats with unique characteristics not found in other RAAs
New York, NY	Upper and Lower New York Bay	Estuarine/Marine	RAA with significant presence of all major vessel types, and in proximity to listed short nose sturgeon population along the Hudson River. RAA includes multiple critical habitat areas.
San Francisco, CA	San Francisco Bay Estuary	Estuarine/Marine	RAA with significant populations of cargo bulk carriers, utility boats, and commercial fishing vessels. Estuary is emblematic of one subjected to heavy ANS invasion. In addition, San Francisco was requested to be an additional RAA during informal consultation with Services due to the presence of a significant number of endangered and threatened species from a broad range of taxa and critical habitat areas.
St. Louis, MO	Upper Mississippi River	Large Freshwater River	RAA with large barge and supporting utility vessel presence. The upper Mississippi River near St. Louis also represents a large freshwater river ecosystem home to a sensitive listed freshwater unionid mussel species.
Seattle, WA	Puget Sound	Estuarine/marine	RAA with significant large and small commercial fishing vessel presence. Puget Sound near Seattle is home to several Pacific Northwest salmonid populations. Includes multiple critical habitats.

EPA selected simple screening-level models to estimate exposure concentrations in the receiving water from vessel discharges covered under the VGP and sVGP. EPA selected the "fraction of freshwater model" to calculate receiving water concentrations in an estuarine harbor and a dilution equation to estimate receiving water concentrations in a river harbor.

For the fraction of freshwater model, EPA identified two types of estuarine systems in commonly found in the port areas:

- Estuaries where tidal influence is the primary driver for mixing in the estuary as indicated by a lower river flow and higher salinity value; and
- Estuaries where the freshwater inflow is the primary driver for mixing in the estuary as indicated by a higher river flow and lower salinity value.

For the river harbor model, the dilution equation requires the average annual river flow rate to calculate a receiving water concentration. EPA selected the minimum and maximum river flow rates from the nine river harbors to represent the bounding conditions for the river harbors, resulting in two river harbor scenarios for the BE effects analysis.

A range of water body flow, salinity, vessel populations, and loading assumptions were applied to the model to explore different "likely" and "worst-case" scenarios pollutant exposure concentrations resulting from vessel discharges covered under the VGP and sVGP. Vessel population scenarios were developed for each RAA based on data provided by the RAA port authorities, available Department of Transportation port call data, and other vessel related literature to characterize the average daily vessel equivalents present in the RAA by vessel class. Vessel discharges were expressed as vessel equivalents where a vessel equivalent represents a single vessel discharging 24-hours per day in the harbor. For example, 1 vessel equivalent could represent 24 vessels that each operates (and discharges) for 1 hour in the modeled harbor each day. EPA also collected receiving water characteristic data from a variety of estuaries and rivers where vessels covered under VGP and sVGP are known to travel.

EPA identified 32 pollutants to include in the modeling analysis based on pre-permit frequency, vessel discharge flow rates and pollutant concentrations as described in *Discharges Authorized by the VGP and sVGP*. EPA then estimated reductions in vessel discharge pollutant concentrations and flow rates anticipated to occur as the result of issuance of the VGP and sVGP for the pollutants, discharges and vessel types evaluated. Harbor specific pollutant loads were estimated for each of the seven RAAs by multiplying vessel and discharge specific pollutant loads by the number of vessel equivalents present in the RAA harbors. EPA used this information to model 14 scenarios addressing harbor specific pollutant loads for 7 RAAs and receiving water characteristics representing minimum and maximum dilution conditions for estuary and river ports (Table 25).

Table 25. Receiving Water Model Scenarios Assessed by EPA.

Model Scenario Number	Harbor Scenario	RAA Loading Scenario
1	Estuary mixing driven by tidal influence	Seattle Pollutant Load
2	Estuary mixing driven by tidal influence	Houston Pollutant Load
3	Estuary mixing driven by tidal influence	New York Pollutant Load
4	Estuary mixing driven by tidal influence	San Francisco Pollutant Load
5	Estuary mixing driven by tidal influence	Miami Pollutant Load
6	Estuary mixing driven by river inflow	Seattle Pollutant Load
7	Estuary mixing driven by river inflow	Houston Pollutant Load
8	Estuary mixing driven by river inflow	New York Pollutant Load
9	Estuary mixing driven by river inflow	San Francisco Pollutant Load
10	Estuary mixing driven by river inflow	Miami Pollutant Load
11	Minimum river flow	St. Louis Pollutant Load
12	Minimum river flow	Duluth Pollutant Load
13	Maximum river flow	St. Louis Pollutant Load
14	Maximum river flow	Duluth Pollutant Load

EPA stated that it believed the scenarios modeled to be representative of the types of water bodies where VGP and sVGP vessels likely operate based on the wide geographical range of waterbodies used to develop the input values. By selecting the minimum water characteristic values for the harbor scenarios, and evaluating the maximum pollutant loading values from the seven RAAs, EPA also believed that the model scenarios reasonably capture a "worst case" scenario within the action area applicable to the BE. Table 26 lists the receiving water concentrations estimated from the 10 estuary harbor exposure scenarios modeled. Table 27 lists the receiving water concentrations estimated from the four river harbor exposure scenarios modeled.

Table 26. Estimated Receiving Water Concentrations Resulting from Vessel Discharge Pollutant Loadings Under Estuary Mixing Conditions.

Polutant Class	Analyte	Units	Pre VGP and sVGP Receiving Water Average (range)	Post VGP and sVGP Receiving Water Average (range)	Percent Reduction Average Scenario Loading (worst-best case scenarios)
Classicals	Biochemical Oxygen Demand (BOD)	mg/L	0.195 (0.00629-0.915)	0.162 (0.0053-0.676)	17% (16%-26%)
Classicals	Hexane Extractable Material (HEM)	mg/L	0.0264 (0.00146-0.0656)	0.013 (0.000701-0.0324)	Approximately 50%[a]
Classicals	Silica Gel Treated HEM (SGT-HEM)	mg/L	0.00684 (0.000438-0.0169)	0.00343 (0.000215-0.0089)	Approximately 50%
Classicals	Total Residual Chlorine	mg/L	0.000281 (0.0000032-0.00214)	0.000103 (0.00000238-0.00066)	63% (26%-69%)
Classicals	Total Suspended Solids (TSS)	mg/L	0.0435 (0.00335-0.202)	0.0281 (0.0015-0.13)	35% (36%-55%)
Metals	Aluminum, Total	µg/L	0.405 (0.0542-1.28)	0.293 (0.0357-1.02)	28% (20%-34%)
Metals	Arsenic, Dissolved	µg/L	0.681 (0.0483-1.94)	0.68 (0.0476-1.94)	Minimal change (<10%)
Metals	Arsenic, Total	µg/L	0.682 (0.0493-1.94)	0.681 (0.0484-1.94)	Minimal change (<10%)
Metals	Cadmium, Dissolved	µg/L	0.000189 (0.0000215-0.000867)	0.000132 (0.0000123-0.000697)	30% (20%-43%)
Metals	Chromium, Dissolved	µg/L	0.000463 (0.0000565-0.00191)	0.000418 (0.0000465-0.00185)	10% (3%-18%)
Metals	Copper, Dissolved	µg/L	0.999 (0.14-2.99)	0.999 (0.11-2.33)	0% (0%-22%)
Metals	Lead, Dissolved	µg/L	0.0011 (0.000116-0.00295)	0.000934 (0.00008-0.00262)	15% (11%-31%)
Metals	Nickel, Dissolved	µg/L	0.102 (0.00718-0.291)	0.102 (0.00701-0.291)	No change

Table 26. Estimated Receiving Water Concentrations Resulting from Vessel Discharge Pollutant Loadings Under Estuary Mixing Conditions.

Polutant Class	Analyte	Units	Pre VGP and sVGP Receiving Water Average (range)	Post VGP and sVGP Receiving Water Average (range)	Percent Reduction Average Scenario Loading (worst-best case scenarios)
Metals	Nickel, Total	µg/L	0.00507 (0.000654-0.0125)	0.00507 (0.000654-0.0125)	No change
Metals	Selenium, Dissolved	µg/L	0.788 (0.0538-2.25)	0.788 (0.0536-2.25)[11]	No change
Metals	Selenium, Total	µg/L	0.837 (0.0565-2.39)	0.837 (0.0565-2.39)	No change
Metals	Zinc, Dissolved	µg/L	0.145 (0.00813-0.493)	0.134 (0.00635-0.452)	Minimal change (<10%)
Nutrients	Ammonia As Nitrogen (NH3-N)	mg/L	0.00149 (0.0000232-0.00954)	0.00123 (0.0000209-0.00742)	17% (10%-22%)
Nutrients	Total Phosphorus	mg/L	0.00267 (0.000112-0.0119)	0.0000905 (0.00000632-0.000543)	Greater than 94%
SVOC	2,4,6-Trichlorophenol	µg/L	0.000178 (0.000012-0.00103)	0.0000927 (0.00000607-0.000514)	Approximately 50%
SVOC	Benzo (a)anthracene	µg/L	0.00306 (0.000393-0.00685)	0.00306 (0.000393-0.00685)	No change
SVOC	Benzo (a)pyrene	µg/L	0.00198 (0.000214-0.00614)	0.00198 (0.000214-0.00614)	No change
SVOC	Benzo (b)fluoranthene	µg/L	0.000733 (0.0000185-0.00486)	0.000733 (0.0000185-0.00486)	No change
SVOC	Benzo (k)fluoranthene	µg/L	0.000812 (0.0000205-0.00538)	0.000812 (0.0000205-0.00538)	No change
SVOC	Bis (2-ethylhexyl) phthalate	µg/L	0.00117 (0.000122-0.00569)	0.000754 (0.000048-0.00424)	36% (25%-61%)

[11] EPA recalculated selenium concentrations to reflect fewere vessels using this technology. We were not provided revised harbor model estimates, only revised total loadings and RQs.

Table 26. Estimated Receiving Water Concentrations Resulting from Vessel Discharge Pollutant Loadings Under Estuary Mixing Conditions.

Polutant Class	Analyte	Units	Pre VGP and sVGP Receiving Water Average (range)	Post VGP and sVGP Receiving Water Average (range)	Percent Reduction Average Scenario Loading (worst-best case scenarios)
SVOC	Chrysene	µg/L	0.00399 (0.00055-0.00943)	0.00399 (0.00055-0.00943)	No change
SVOC	Ideno (1,2,3-cd)pyrene	µg/L	0.000655 (0.0000165-0.00434)	0.000655 (0.0000165-0.00434)	No change
VOC	Benzene	µg/L	0.00554 (0.000393-0.0311)	0.0042 (0.000186-0.0258)	24% (17%-53%)
VOC	Bromodichloromethane	µg/L	0.00085 (0.000139-0.00195)	0.00025 (0.0000364-0.000573)	Approximately 70%
VOC	Dibromochloromethane	µg/L	0.00085 (0.000139-0.00195)	0.00025 (0.0000364-0.000573)	Approximately 70%
VOC	Tetrachloroethene	µg/L	0.0000116 (0.0000015-0.0000288)	0.00000582 (0.00000075-0.0000144)	50% reduction regardless of scenario
VOC	Trichloroethene	µg/L	0.159 (0.00122-1.14)	0.0000344 (0.00000225-0.000191)	Eliminated

[a] generalized for cases where the average, minimum and maximum reductions are within 5% of each other

Table 27. Estimated Receiving Water Concentrations Resulting from Vessel Discharge Pollutant Loadings Under River Mixing Conditions.

Polutant Class	Analyte	Units	Pre VGP and sVGP Receiving Water Average (range)	Post VGP and sVGP Receiving Water Average (range)	Percent Reduction Average Scenario Loading (worst-best case scenarios)
Classicals	Biochemical Oxygen Demand (BOD)	µg/L	4.38 (0.0506-9.2)	4.17 (0.0459-9.11)	5% (1%-9%)
Classicals	Hexane Extractable Material (HEM)	µg/L	0.396 (0.00445-0.855)	0.149 (0.00173-0.311)	62% (61%-64%)
Classicals	Silica Gel Treated HEM (SGT-HEM)	µg/L	0.102 (0.00117-0.214)	0.041 (0.00049-0.0837)	60% (58%-61%)
Classicals	Total Residual Chlorine	µg/L	0.00152 (0.0000169-0.00328)	0.00139 (0.0000144-0.00319)	9% (3%-15%)
Classicals	Total Suspended Solids (TSS)	µg/L	1.1 (0.0126-2.33)	0.72 (0.00764-1.62)	35% (30%-39%)
Metals	Aluminum, Total	µg/L	0.015 (0.000162-0.0334)	0.0119 (0.000102-0.0309)	21% (7%-37%)
Metals	Arsenic, Dissolved	µg/L	0.0413 (0-0.164)	0.0413 (0-0.164)	no change
Metals	Arsenic, Total	µg/L	0.0413 (0.000000231-0.164)	0.0413 (0.000000231-0.164)	no change
Metals	Cadmium, Dissolved	µg/L	0.0000059 (0.0000000653-0.0000129)	0.00000446 (0.0000000366-0.0000118)	24% (9%-44%)
Metals	Chromium, Dissolved	µg/L	0.0000158 (0.000000142-0.0000399)	0.0000141 (0.000000137-0.0000338)	11% (4%-15%)
Metals	Copper, Dissolved	µg/L	0.047 (0.000136-0.164)	0.037229 (0.000106-0.131)	21% (20%-22%)
Metals	Lead, Dissolved	µg/L	0.0000416 (0.00000045-0.0000922)	0.0000362 (0.000000345-0.0000878)	13% (5%-23%)
Metals	Nickel, Dissolved	µg/L	0.0062 (0.000000338-0.0246)	0.0062 (0.000000334-0.0246)	minimal change <2%
Metals	Nickel, Total	µg/L	0.000211 (0.000000716-0.000723)	0.000211 (0.000000716-0.000723)	no change
Metals	Selenium, Dissolved	µg/L	0.0479 (0-0.191)	0.0479 (0-0.191)	no change

Table 27. Estimated Receiving Water Concentrations Resulting from Vessel Discharge Pollutant Loadings Under River Mixing Conditions.

Polutant Class	Analyte	Units	Pre VGP and sVGP Receiving Water Average (range)	Post VGP and sVGP Receiving Water Average (range)	Percent Reduction Average Scenario Loading (worst-best case scenarios)
Metals	Selenium, Total	µg/L	0.051 (0-0.203)	0.051 (0-0.203)	no change
Metals	Zinc, Dissolved	µg/L	0.00446 (0.0000496-0.00968)	0.00418 (0.0000439-0.00948)	6% (2%-11%)
Nutrients	Ammonia As Nitrogen (NH3-N)	µg/L	0.0149 (0.000167-0.032)	0.014 (0.000167-0.0285)	6% (0%-11%)
Nutrients	Total Phosphorus	µg/L	0.0225 (0.000271-0.0453)	0.00248 (0.0000242-0.00594)	89% (87%-91%)
SVOC	2,4,6-Trichlorophenol	µg/L	0.00000539 (0.0000000626-0.0000113)	0.00000376 (0.0000000313-0.00000986)	30% (13%-50%)
SVOC	Benzo (a)anthracene	µg/L	0.000268 (0.00000164-0.0008)	0.000268 (0.00000164-0.0008)	no change
SVOC	Benzo (a)pyrene	µg/L	0.0002 (0.00000159-0.000536)	0.0002 (0.00000159-0.000536)	no change
SVOC	Benzo (b)fluoranthene	µg/L	0.000114 (0.00000139-0.000228)	0.000114 (0.00000139-0.000228)	no change
SVOC	Benzo (k)fluoranthene	µg/L	0.000126 (0.00000154-0.000252)	0.000126 (0.00000154-0.000252)	no change
SVOC	Bis (2-ethylhexyl) phthalate	µg/L	0.0000981 (0.00000118-0.000198)	0.0000839 (0.000000979-0.000175)	14% (12%-17%)
SVOC	Chrysene	µg/L	0.000325 (0.00000164-0.00102)	0.000325 (0.00000164-0.00102)	no change
SVOC	Ideno (1,2,3-cd)pyrene	µg/L	0.000102 (0.00000124-0.000203)	0.000102 (0.00000124-0.000203)	no change
VOC	Benzene	µg/L	0.000579 (0.00000701-0.00117)	0.000538 (0.00000641-0.0011)	<10%
VOC	Bromodichloromethane	µg/L	0.0000431 (0.000000451-0.0000981)	0.0000159 (0.000000116-0.0000443)	63% (55%-74%)
VOC	Dibromochloromethane	µg/L	0.0000431 (0.000000451-0.0000981)	0.0000159 (0.000000116-0.0000443)	63% (55%-74%)

Table 27. Estimated Receiving Water Concentrations Resulting from Vessel Discharge Pollutant Loadings Under River Mixing Conditions.

Polutant Class	Analyte	Units	Pre VGP and sVGP Receiving Water Average (range)	Post VGP and sVGP Receiving Water Average (range)	Percent Reduction Average Scenario Loading (worst-best case scenarios)
VOC	Tetrachloroethene	µg/L	0.000000484 (0.00000000164-0.00000166)	0.000000242 (0.000000000821-0.00000083)	50%
VOC	Trichloroethene	µg/L	0.0595 (0.0000000232-0.237)	0.0000014 (0.0000000116-0.00000366)	100% (50%-100%)

Geographical Distribution and Overlap of Discharges Authorized by the VGP and sVGP with Threatened and Endangered Species

In order to determine whether listed resources will be exposed to these discharges, it is necessary to characterize the overlap of vessel operation with species Critical Habitat and Range. Since some vessel discharges are essentially continuous during operation, and vessels covered by the VGPs travel widely both coast-wise and across the ocean, we expect most NMFS listed species to be exposed to vessel discharges to some degree. In its BE, EPA modeled exposures for several Representative Action Areas (RAAs) selected after consultation with the Services to evaluate the potential effects of the permit actions on listed aquatic and aquatic-dependent species, as well as their designated critical habitats.

The Services provided EPA with a list of over 1,900 federally-listed species, 612 of which were identified by EPA as aquatic and aquatic-dependent animal and plant species with more than limited exposure to "waters of the United States" and may be affected by ANS of pollutants from vessel discharges covered under the proposed actions. This list includes 44 mammals, 4 birds, 24 reptiles, 27 amphibians, 135 fish, 27 crustaceans, 34 snails, 93 bivalve mollusks (clams and abalone), 25 insects, 2 corals, 157 sea grasses and aquatic and wetland plants and 217 designated critical habitats that may be affected by the permits. The ranges for 454 of these aquatic and aquatic-dependent animal and plant species are concurrent with waters generally navigable by vessels covered under the VGP and sVGP. The list of 454 aquatic and aquatic-dependent animal and plant species identified in EPA's analysis constitute the full range of species and their designated critical habitats that may be affected by both ANS and traditional pollutants from the permit actions. The list of species addressed by EPA's RAA analysis includes 23 of the 87 NMFS listed threatened and endangered species (Table 28).

Table 28. NMFS listed Species in RAAs exposed to VGP discharges

Status[a]	Species (*Scientific Name*)	Biscayne Bay	Galveston Bay	Puget Sound	San Francisco Bay Estuary	Upper and Lower New York Bay
E	Shortnose Sturgeon (*Acipenser brevirostrum*)	x				x
T	North American Green Sturgeon (*Acipenser medirostris*)				x	
T	Gulf Sturgeon (*Acipenser oxyrinchus desotoi*)	x				
T	Staghorn Coral (*Acropora cervicornis*)	x				
T	Elkhorn Coral (*Acropora palmata*)	x				
E	Finback Whale (*Balaenoptera physalus*)	x	x			x

Status[a]	Species (*Scientific Name*)	Biscayne Bay	Galveston Bay	Puget Sound	San Francisco Bay Estuary	Upper and Lower New York Bay
T	Loggerhead Sea Turtle (*Caretta caretta*)	x	x		x	x
E,T	Green Sea Turtle (*Chelonia mydas*)	x	x	x	x	x
E	Leatherback Sea Turtle (*Dermochelys coriacea*)	x	x	x	x	x
E	Hawksbill Sea Turtle (*Eretmochelys imbricata*)	x	x			x
E	North Atlantic Right Whale (*Eubalaena glacialis*)	x				x
E,T	Steller Sea-Lion (*Eumetopias jubatus*)			x	x	
E	White Abalone (*Haliotis sorenseni*)				x	
T	Johnson's Seagrass (*Halophila johnsonii*)	x				
E	Kemps's Ridley Sea Turtle (*Lepidochelys kempii*)		x			x
T	Olive Ridley Sea Turtle, (*Lepidochelys olivacea*)				x	
E	Humpback Whale (*Megaptera novaeangliae*)	x	x	x		x
E,T	Steelhead (Central and South Central CA) (*Oncorhynchus (=Salmo) mykiss*)				x	
T	Chum Salmon (Hood Canal summer-run) (*Oncorhynchus keta*)			x		
E,T	Steelhead Salmon (Puget Sound) (*Oncorhynchus mykiss*)			x		
E,T	Chinook Salmon (Puget Sound) (*Oncorhynchus tshawytscha*)			x		
E	Killer Whale (Southern Resident DPS) (*Orcinus orca*)			x		
E	Smalltooth Sawfish (*Pristis pectinata*)	x	x			

[a] T =Threatened, E = Endangered

The remaining NMFS listed species are not located in the specific RAAs examined by EPA, but are likely to be exposed to vessel discharges by virtue of their being marine dwelling organisms. Additional listed marine mammals likely exposed to vessel discharges include the Cook Inlet Beluga whale (*Delphinapterus leucas).* Black Abalone *(Haliotis cracherodii)* is also likely to be exposed. In addition to the species identified in Table 28, 8 additional listed species may be adversely affected by the VGPs due to exposures to discharges in both the marine environment and portions of their freshwater habitat. These species include: Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus),* Coho Salmon (*Oncorhynchus kisutch),* Sockeye Salmon (*Oncorhynchus nerka),* Largetooth Sawfish (*Pristis perotteti),* Atlantic Salmon (*Salmo salar),*

Bocaccio (*Sebastes paucispinis),* Canary Rockfish (*Sebastes pinniger),* and the Pacific Eulachon/Smelt (*Thaleichthys pacificus)*. This list includes ESUs and DPSs not specifically associated with the RAAs evaluated by EPA.

The VGPs include limits on certain discharges in *Waters Federally Protected Wholly or in Part for Conservation*. This list includes marine sanctuaries, units of the National Park System, units of the National Wildlife Refuge System, National Wilderness areas, national wild and scenic rivers system components and any water body designated as an Outstanding National Resource Water (ONRW) by a State or Tribe.

These discharge limitations for *Waters Federally Protected Wholly or in Part for Conservation* include:

- Bilgewater may not be discharged into *Waters Federally Protected Wholly or in Part for Conservation* unless the discharge is necessary to maintain the safety and stability of the ship.
- Ballastwater discharge or uptake must be avoided in/into waters within, or that may directly affect, marine sanctuaries, marine preserves marine parks, or coral reefs or *Waters Federally Protected Wholly or in Part for Conservation.*
- Unexchanged or untreated ballast water or sediment may not be discharged to *Waters Federally Protected Wholly or in Part for Conservation* by:
    - Any vessels that carry ballast water that was taken on in areas less than 200 nautical miles from any shore that will subsequently operate beyond the Exclusive Economic Zone (EEZ) and more than 200 nm from any shore
    - Vessels Carrying Ballast Water Engaged in Pacific Nearshore Voyages
    - Vessels with any Ballast Water Tanks that are Empty or have Unpumpable Residual Water
    - Vessels Engaged in Pacific Nearshore Voyages with Unpumpable Ballast Water and Residual Sediment (including NOBOBs)
- AFFF discharges may not occur in or within 1 nm of *Waters Federally Protected Wholly or in Part for Conservation* unless:
    - For emergency purposes;
    - By rescue vessels such as fireboats for firefighting purposes; or
    - By vessels owned or under contract to do business exclusively in or within 1 nm of those protected areas by the United States government or state or local governments.
- Boiler/economizer blowdown may not be discharged In *Waters Federally Protected Wholly or in Part for Conservation* except for safety purposes.
- Firemain systems may not be discharged *Waters Federally Protected Wholly or in Part for Conservation* except for safety purposes.
- Graywater may not be discharged *Waters Federally Protected Wholly or in Part for Conservation* by vessels with the capacity to store graywater. Vessels which cannot store graywater must minimize graywater production.

Response Analysis

The previous *Exposure Analysis* section reviewed the discharges in terms of their volume and constituent mixtures, then described EPAs harbor modeling methodology and results. This section summarizes the modeled harbor discharge constituent concentrations for each pollutant class and the relative importance of discharge sources before reviewing the responses of listed, or surrogate, species to exposures to these pollutants.

Nutrients

In their Biological Evaluation EPA acknowledged that Nutrient pollution is one of the leading causes of water quality impairment in the nation, primarily because the quantity of nutrients reaching the nations waters has dramatically escalated over the past 50 years (USEPA 2009). Nutrient loadings in the form of 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 impairment of drinking water sources. In general, 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 phosphorus 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).

Nutrients in Vessel Discharges

EPA reported that nutrients, particularly ammonia nitrogen and total phosphorus, were constituents commonly found in bilgewater, deck washdown, fish hold and fish hold cleaning effluent, and graywater. Based on studies of pollutants in vessel discharges EPA reported that average ammonia nitrogen concentrations were above their screening level benchmark of 1.2 mg/L in both fish hold effluent and fish hold cleaning effluent (USEPA 2010b) (see Appendix D of the BE for more information regarding screening level benchmarks). Protein, free amino acids, and nucleotides from fish and fish by-products are all potential sources of nitrogen in the fish hold effluent and fish hold cleaning effluent. EPA also reported that ammonia nitrogen concentrations can also be above screening level benchmarks in graywater. EPA found the average ammonia nitrogen concentration from all graywater sources on cruise ships was 2.1

mg/L (USEPA 2008). Average phosphorous concentrations were above EPAs screening level benchmark of 0.1 mg/L in bilgewater, graywater, deck washdown discharges, fish hold effluent, and fish hold cleaning effluent. EPA suggested that the likely source of phosphorous in bilgewater and deck washdown water was liquid detergents used on board for deck washing. They also indicated that sources of phosphorous in graywater and fish hold effluent may reflect food or other wastes. Table 29 (3-14 from the BE) shows the average concentrations of ammonia nitrogen and total phosphorous in the various vessel discharges requiring detailed analysis.

Table 29. Average Nutrient Concentrations in Vessel Discharges at Levels Above Screening Level Benchmarks

Vessel Discharge Requiring Detailed Analysis	Average Ammonia Nitrogen (mg/L)	Average Total Phosphorous (mg/L)
Deck Washdown	NA	1.7
Fish Hold Effluent	12	13
Fish Hold Cleaning Effluent	16	8.5
Graywater	1.3 2.1a	1.4 10a
Stern Tube Packing Gland Effluent	NA	0.13

Source: USEPA, 2010, unless otherwise specified.
NA: Pollutant does not exceed the screening level benchmark for this vessel discharge (see Appendix D of the BE for more information regarding screening level benchmarks).
a) Average concentration in graywater from large cruise ships (USEPA, 2008, EPA sampling data).

Direct effects of Ammonia Nitrogen

As mentioned above, nitrogen is essential for primary production in aquatic ecosystems and over-enrichment can lead to eutrophication and water quality impairments. Ammonia nitrogen (NH3 and NH4-) can also be highly toxic. Recent studies have shown that freshwater mollusks (mussels and snails) are particularly sensitive to ammonia (Wang et al. 2007a, Wang et al. 2007b, Besser et al. 2009). In fact, they are more sensitive than previously tested species. As a result, EPA is revising their 304a aquatic life criteria for ammonia to include these new mollusk data so that freshwater mussels, snails and other mollusks are adequately protected. A draft of the updated ammonia criteria document was published in the federal register for public comment in 2009.

EPA used the freshwater mussel *Lampsilis siliquoidea* (Fatmucket) as a surrogate to represent freshwater invertebrates in their response analysis. The IC20 for Fatmucket (0.3027 mg N/L) was the lowest chronic effects thresholds (CTET) among all taxa (freshwater or marine) considered in the analysis (EPA BE Table 5-49). EPA predicted the concentrations of ammonia nitrogen in river and estuarine harbors, resulting from vessel discharges, to be 0.0000285 (mg N/L) and 0.00742 (mg N/L), respectively. These concentrations are orders of magnitude below the Fatmucket IC20 and the resulting risk quotients were extremely low (EPA BE Table 5-50). Therefore risks from chronic exposure to ammonia nitrogen in harbors where, vessel discharges are the sole source, should be very low.

Phosphorus

EPA noted in their BE that under natural conditions, freshwater ecosystems generally have low phosphorous concentrations (< 100 µg/L), but each waterbody is different, and there are numerous factors which 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 channelizing. Additionally, excessive phosphorus and the resulting problems associated with nutrient enrichment in general are frequently widespread and manifested at a location remote from the sources, and may not show themselves for some time after significant inputs to the system have occurred (WERF 2010). Because of the above, EPA chose to use the lowest statewide and/or site-specific water quality criteria for total phosphorus (10 µg/L) as the basis for $CTET_{A,W}$ in the analysis.

Since NMFS' assessment for Ammonia Nitrogen is based on EPA's exposure analysis, which assumed that vessel discharges would be completely mixed within the harbor area, acute or short term exposure of species to pollutants in close proximity to vessels (i.e. within areas of discharge mixing, is not considered). The uncertainty associate with this assumption is discussed below. Although, high ammonia concentrations are predicted for discharges (fish hold and fish hold cleaning effluent) from some vessels, the infrequent occurrence of those vessels (fishing vessels) in freshwater river RAA harbors (EPA BE) suggest that the likelihood that sensitive listed species will be exposed, short term, to acutely toxic concentrations of ammonia is low. In addition, effluent limits and BMPs in the VGP/sVGP for nutrient reduction should further reduce the chance of hazardous exposures.
EPA estimated that vessel discharges would result in phosphorus concentrations of 0.00594 ug/L in freshwater environments and 0.543 ug/L in Estuarine/Marine Environments. These values are well below the most protective phosphorus criteria (at this time). Low expected phosphorus concentrations along with effluent limits and BMPs required by the VGP/sVGP for nutrient reduction suggest that vessel discharges will not produce the adverse effects associated with nutrient enrichment.

Effluent limitations and BMPs in the 2013 VGP and sVGP are expected to reduce/minimize nutrient loadings from vessel discharges. The Services conclude that adverse effects of nutrient loading by vessel discharges under the VGPs should not adversely affect listed resources.

Pathogens

The sewage indicator species *E. coli*, fecal coliform, and enterococcus are typically nonpathogenic and are used as indicators for the possible presence of disease causing microbes and protozoans found in the feces of warm blooded animals (USEPA 1986). EPA did not estimate pathogen loads or select response thresholds for sewage pathogens in its BE. The risks posed by sewage indicator bacteria for listed resources cannot be quantitatively evaluated due to the absence of screening benchmarks for protection of fish and wildlife and the absence of exposure-response data in the literature on the ecological effects of Sewage Pathogen Indicators. Analysis of pathogen risk to listed resources is further complicated because pathogen exposure–

response follows a "one hit" model which accounts for the possibility, although low likelihood, that even a single pathogen could cause infection, with the likelihood of infection increasing with increasing exposure intensity (EPA/USDA 2012).

EPA's qualitative assessment described an investigation of graywater discharges from cruise ships which reported enterococci standards for marine water bathing were exceeded 66 percent of the time and fecal coliform standards for harvesting shellfish were exceeded over 80 percent of the time (USEPA 2008). Graywater pathogen data provided in the Report to Congress were several orders of magnitude greater than the screening benchmark for marine water bathing by humans (USEPA 2010b). In its BE, EPA noted that sewage-associated pathogens are generally not considered to be harmful to fish or reptiles, stating:

> "EPA is not aware of any outbreak of pathogenic disease among aquatic or aquatic-dependent species from vessel discharges. Fish and shellfish are known vectors of pathogens resulting in human impacts when consumed, but appear to be largely unaffected themselves. Since it is currently not possible to separate infections in aquatic and aquatic-dependent animals due to [human] or any other pathogens contributed from vessel discharges versus infection from existing natural background conditions within a waterbody, and since it is highly improbable that any infection could occur at the levels detected in vessel discharges, EPA discounts any risk to pathogens in the BE to the listed species for which the analysis applies"

NMFS is supplementing EPA's qualitative assessment of sewage indicator species because the BE did not acknowledge that human sewage has been identified as the likely source for the strain of *Serratia marcescens* which caused outbreaks of white pox disease in threatened elkhorn coral in the Caribbean (Sutherland et al. 2010) or that marine mammals can host and become infected by bacterial pathogens associated with sewage (Grillo 2001, Thompson et al. 2005, Venn-Watson et al. 2010). While EPA concluded that infection is highly improbable at levels detected in vessel discharges, we must note that EPAs modeling assumed complete mixing of discharges, and did not address more intense exposures within mixing zones created by potentially large volume discharges such as graywater. A report by the Alaska Department of Environmental Conservation evaluating the impact of cruise ship wastewater discharge on Alaska waters indicates that, although offshore wastewater sources would likely be much smaller in magnitude than on shore sources, they can represent a more direct and undiluted exposure (ADEC 2002).

We focus the discussion on sewage pathogen indicator levels in graywater because graywater accounts for greater than 96% of sewage pathogen indicator load in vessels authorized to discharge under the VGPs, with sewage pathogen indicator loadings through deckwash at 2%, bilgewater at 1% and, for fishing vessels, fish hold effluent at 2% and fish hold cleaning effluent at <1 % of total sewage pathogen loading. Graywater sewage pathogen levels in large cruise ships, utility vessels, and large ferries were much higher than that for other vessel classes by several orders of magnitude.

Human sewage has been identified as the likely source for the strain of *Serratia marcescens*, which caused outbreaks of white pox disease in threatened elkhorn coral in the Caribbean.

Diseases such as white pox are listed as a major contributor to the decline of Elkhorn coral. Along with white band disease, white pox is the most commonly reported disease in this species (Sutherland et al. 2010). Disease among elkhorn is widespread, episodic, and unpredictable in its occurrence and results in high mortality. The effects are primarily due to disassociation of zoozanthellae from coral tissue. Just prior to this, coral epithelium and gastrodermis tissue begins to decay and die, likely as a result of stress to individual corals (Ainsworth et al. 2008). The zoozanthellae are what gives coral its color. The widespread, episodic loss of zoozanthellae is what gives rise to coral bleaching. Elkhorn recovery from a bleaching event in 2005 is expected to take 10-12 years; this is after a previous event in 1997. Although quantitative data on historical distribution and abundance are scarce, best available data indicate declines in abundance (coverage and colony numbers) by greater than 97%. In all, roughly one-third of the *Acropora palamata* genotypes have been lost as a result of these events (Miller and Williamson 2010). Given that vessel graywater discharges can represent more direct and undiluted exposure "pulses", discharge of untreated graywater within coral critical habitat could expose elkhorn to *Serratia marcescens* and cause an outbreak of whitepox disease. Human sewage associated pathogens have not been implicated in causing adverse effects in other marine plants or invertebrate species.

At this time the impact of sewage-associated bacteria found in marine mammals is uncertain because the role of infectious disease has not been traditionally investigated in cetacean population health assessments. When pathogens are detected in tissues, it is often unknown if the pathogen was associated with a chronic or acute infection, underlying or immediate cause of death, or simply an incidental finding (Venn-Watson et al. 2010). In general, small, isolated cetacean populations with limited gene pools and populations with nutritional challenges are expected to be at higher risk of infectious disease host susceptibility and disease transmission (Venn-Watson et al. 2010). Such populations match this description and will likely encounter VGP-authorized discharges, include the Southern Resident Killer Whale and the Cook Inlet Beluga.

The NMFS Recovery plan for the Southern Resident Killer Whale evaluated the role of disease in this population. Among the pathogens identified from free-ranging animals, only *Edwardsiella tarda* is also found in the gut flora of humans (Gaydos et al. 2004b). While in 2000, a male Southern Resident died from a severe infection caused by *E. tarda* (Ford et al. 2000), disease epidemics have never been reported in killer whales in the northeastern Pacific (Gaydos et al. 2004b). A recently published report evaluating the exposure of Cook Inlet Beluga to vessel discharges (URS 2011) examined sanitary discharge from offshore ship, barge and recreational boat traffic as a pathogen source for this species. The report found no readily available information to characterize the nature and magnitude of potential sanitary waste discharge from vessel traffic into Cook Inlet. NMFS examined scientific literature, agency reports, industry and NGO publications, but was unable to find reported incidents or any other evidence related to sewage associated pathogen exposures or effects in Cook Inlet Beluga.

While marine turtles are known to harbor human pathogens such as *Pseudomonas, Mycobacterium*, *Salmonella*, *Vibrio*, and *Chlamydia*, these pathogens are typically opportunistic organisms which are found throughout the environment and in a wide range of vertebrate

species. *(Glazebrook and Campbell 1990, PAHO 2001)* While discharges containing sewage pathogens can enhance the risk of exposures resulting in infection, a causal link has not been established between exposure to human sewage pathogens and adverse effects in marine turtles. While several ubiquitous pathogens are found in both humans and fish, those associated with sewage are not known to adversely affect marine and anadromous fish.

At this time, NMFS does not have a good understanding of the potential impact of human sewage indicator pathogens on the population health of threatened and endangered vertebrate species. Understanding is complicated by our ability to discriminate between primary infections and opportunistic infections in wounded or weakened animals.

VGP Measures to Mitigate Potential Exposure of Elkhorn Coral to *Serratia marcescens*. Elkhorn coral critical habitat overlaps with areas restricted from graywater discharge by the VGP. These areas, termed *"Waters Federally Protected Wholly or in Part for Conservation Purposes"* include the Florida Keys National Marine Sanctuary, Virgin Islands National Park and Virgin Islands Coral Reef National Monument of St. John and Buck Island National Monument off the coast of St. Croix. The Florida Keys National Marine Sanctuary protects 2,900 square nautical miles of waters surrounding the Florida Keys, from south of Miami westward to encompass the Dry Tortugas. The sanctuary excludes Dry Tortugas National Park, but as a National Park, it is also included among the VGPs restricted waters. These protected areas encompass the Florida Unit of the designated critical habitat for both staghorn and elkhorn corals.

The Caribbean Unit for coral critical habitat includes Puerto Rico and the U.S. Virgin Islands. While the Virgin Islands National Park and Virgin Islands Coral Reef National Monument of St. John appear to encompass greater than 80% of the island's coral reefs and Buck Island National Monument provides some cover off of St. Croix, the remainder of the Caribbean Unit, including St. Thomas and Puerto Rico, is not protected from graywater discharges under the VGP.

However, elkhorn coral habitat and the paths vessels authorized to discharge under the VGP do not likely overlap. Elkhorn coral generally do not form colonies below 16.4 feet deep (Lighty et al. 1982) and usually occur between 3.3 and 16.4 feet in depth with occasional individual corals found at depths of up to 65 feet (Goreau and Wells 1967). This contrasts with estimates of draft for VGP-authorized vessels (from Table 3-18 of EPA's BE), which range from a shallowest draft of 10 feet for barges to 15.5 feet for large ferries and medium cruise ships, 17 feet for large commercial fishing ships and 18 feet for large cruise ships on up to 75 feet for oil and gas tankers. VGP-authorized vessels therefore will not likely discharge over elkhorn coral habitat. We therefor conclude that graywater discharges from VGP-authorized vessels are not likely to result in harmful exposures of elkhorn coral to the white pox pathogen in vessel graywater.

The sVGP, on the other hand, covers smaller vessels, which have shallower draft and may travel over and discharge untreated graywater (or graywater mixed with sewage) within elkhorn coral habitat. These vessels also discharge graywater at much lower rates than VGP vessels. Tugboats, with a crew of three to five persons, generate approximately 130 gallons per day of graywater. Water taxis generate less graywater per person because the discharge is limited to bathroom sinks, producing an estimated 75 gallons per day discharge. Graywater generation on

commercial fishing boats is highly variable, ranging from a few to hundreds of gallons per day, depending on the length of the trip and the size of the crew (USEPA 2010b). The sVGP does not restrict graywater discharges to specified *"Waters Federally Protected Wholly or in Part for Conservation Purposes,"* but effluent limits under the sVGP include:

> "Minimize graywater discharges in areas that have heavy vessel traffic or heavy recreational use and in marine sanctuaries, national wildlife refuges, national wild and scenic rivers, and national wilderness areas. If the vessel has the capacity to store graywater in these waters, it should be stored and later discharged in other waters or onshore."

The lower rate of discharge for individual vessels, taken with the sVGP requirement to minimize discharges in sensitive areas and discharge graywater while underway greatly reduces the likelihood of direct and undiluted exposures of elkhorn coral to human sewage pathogens. However, since pathogen infection is described by the "one hit" model (EPA/USDA 2012), VGP-authorized vessels in areas not restricting graywater discharges could result in an exposure.

Oil and Grease

Transportation activities are estimated to contribute 24% of the total oil input into the oceans (Tong et al. 1999). Oil and grease are expressed in the BE in terms of 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 substances, including toxic partially combusted hydrocarbons and petrochemical constituents such as polycyclic aromatic hydrocarbons, benzene, toluene, ethylbenzene and xylene (SVOCs and VOCs).

Both petroleum and non-petroleum oils have similar physical properties that can harm wildlife such as sticking to fur and feathers and reducing their insulative properties (USEPA, 1997). EPA examined the physical and toxic effects of oil and grease separately because a qualitative assessment was required for the HEM and quantitative assessments were required for the individual toxic constituents in the SGT-HEM in the section evaluating SVOCs and VOCs.

EPA's estuarine harbor model estimated post permit HEM concentrations to average 0.013 mg/L and range from 0.0007 to 0.03 mg/L. These values represent an approximately 50 percent reduction in hexane extractable material for all scenarios. For the river harbor model, EPA's average estimated post permit hexane extractable material concentration was 0.15 mg/L and ranged from 0.002 to 0.3 mg/L, suggesting approximately 60% reduction in discharged hexane extractable material. While HEM averaged below the 15-mg/L U.S. Coast Guard standard screening level benchmark in most discharges, all discharges had at least one observation that exceeded this benchmark. Graywater was the only discharge with an average concentration exceeding the benchmark, at 39 mg/L. Due to the highly variable graywater generation volumes possible within vessel classes, EPA was unable to fully define graywater generation rates.

Graywater accounts for greater than 90% of the oil and grease discharged from vessels. Loads range from as little as 4 grams per day from utility boats to as much as 600 kg per day (0.6 tons)

from large cruise vessels. The sources for oil and grease in graywater are expected to be personal care products and the oils, fats and grease used in cooking. These sources are considered to be biodegradable and less toxic to aquatic and aquatic-dependent wildlife than oil and grease discharges associated with engine oil (USEPA 2011a). The data support this conclusion. The toxic SGT-HEM fraction of oil and grease representing about 17% of the total oil and grease in graywater. Meanwhile the SGT-HEM fraction of engine associated discharges accounts for 32-57% of total oil and grease. The SGT-HEM fraction in fish hold and deckwash discharges is also high, at 33-51% total oil and grease. The source for the SGT-HEM is thought to be the lubricants and other petroleum products used on the vessel. Among other vessel sources of oil and grease, bilge water contributes an estimated 1-12 grams oil and grease per vessel per day and deckwash (an intermittent exposure) contributes 1-14 grams per day, depending on vessel class.

In conclusion, literature reviews conducted separately by EPA and NMFS found that the physical effects of oil and grease are typically associated with accidents involving large amounts of oil rather than discharges due to incidental vessel operation. Although pre-permit oil and grease discharges exceeded the USCG limits, even these are not at the intensities encountered during oil spill events and would not likely result in the physical effects associated with HEM.

Metals

EPA estimates that concentrations of aluminum, arsenic, cadmium, chromium, lead, nickel, and zinc associated with vessel discharges are, and will likely remain, below levels that may cause adverse effects to listed species. Depending on the estuarine harbor scenario modeled, overall post-permit metals concentrations declined by an average of 5% to 13%. Overall metals concentrations modeled for post-permit river harbor scenarios were reduced by 2% to 11%. However, the modeling predicted a broad range of reductions for individual metals. Selenium, nickel and arsenic concentrations were not reduced under “worst case” estuarine and river harbor scenarios. Under the best of conditions, the permit could result in reductions of cadmium concentrations of about 40%. Cumulative RQs for metals ranged up to 0.311 for exposures of estuarine/marine invertebrates, with copper accounting for about 95% of the cumulative RQ. Given the uncertainties in EPA’s exposure analysis, we believe that these concentrations may be higher than modeled. EPA also omitted discharges from zinc sacrificial anodes in its analysis. Our rough estimate of this metals source increases the cumulative RQ for estuarine/marine invertebrates to 0.422. In addition, vessel discharges are likely to combine with other sources of metals in waterways, resulting in exposure to species at concentrations greater than considered in this analysis.

Metals are a diverse group of pollutants, many of which can be toxic to aquatic and aquatic-dependent species. As described by EPA, vessel discharges can contain a variety of metal constituents from a variety of on-board sources. For example, EPA’s study of cruise ship graywater found a total of 13 different metals in at least 10% of samples, with copper, nickel and zinc detected in 100% of samples (USEPA 2008). Bilgewater has also been shown to contain numerous metals, the exact constituents of which vary dependent upon on-board activities on the vessel and the materials used in the construction of the vessel. Other metals, such as copper, are

known to leach from vessel hulls and can cause exceedances of water quality standards. For example, significant leaching of copper from the hulls of sailboats, powerboats, and cruise ships has been documented (Srinivasan and Swain 2007).

While some metals, including copper, nickel and zinc, are known to be essential to organism function, many others, including thallium and arsenic, are non-essential and/or are known to have only adverse impacts. 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 and aquatic-dependent organisms. In the aquatic environment, elevated concentrations of dissolved metals can be toxic to many species of algae, crustaceans, and fish because it is this dissolved form of the metal that is most readily available and taken up and internalized by an organism. Additionally, metals may not be fully eliminated from blood and tissues processes, and may bioaccumulate in predatory and scavenging organisms further up the food chain, including fish, birds, reptiles, and mammals.

One of the key factors in evaluating risk from exposure to metals is the bioavailability of the metal to an organism. As indicated above, 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, the protection of aquatic life for metals is typically expressed in the dissolved metal form.

In contrast, evaluation of total metals is more appropriate for aquatic-dependent animals where the primary route of exposure is assumed to be through the consumption of aquatic organisms, and where the digestive process is assumed to transform all forms of metals to the dissolved phase, thus increasing the amount of biologically available metals. This latter is also applicable in aquatic animals exposed to metalloids such as arsenic and selenium.

EPA describes effects of aluminum, arsenic, cadmium, chromium, lead, nickel, and zinc that result in measured changes to survival, growth, and reproduction. In addition, each of these elements may result in sublethal effects to aquatic and aquatic-dependent wildlife that may affect a species' fitness, either alone or in combination with other physiological or environmental stressors. Exposure to metals may result in a wide range of effects to endpoints such as behavior, enzyme activity, blood chemistry, osmoregulation, respiration, disease resistance, and other physiolological parameters. In many cases, these effects have been found to occur at concentrations lower than those used in established aquatic life criteria (Eisler 2000).
One example of a particular sensitive endpoint is avoidance behavior in fish, observed at concentrations as low as 23.9 ug/L nickel and 10 ug/L zinc in rainbow trout (Eisler 2000). Other sublethal effects of metals were recorded at concentrations generally consistent with those observed to effect survival, growth, and reproduction.

While abundant data are available for metal effects in most aquatic species, information on

metals in marine mammals is generally limited to tissue concentrations (Varanasi et al. 1994, Sanpera et al. 1996, Wagemann et al. 1996, Wood and VanVleet 1996, Sydeman and Jarman 1998, Meador et al. 1999, Welfinger-Smith et al. 2011). There are a few in vitro investigations demonstrating cytolotoxic, genotoxic and immunotoxic effects of metals in marine mammal tissues (DeGuise et al. 1996, Kakuschke et al. 2008, Frouin et al. 2010, Chen et al. 2012). While data from in vitro studies are difficult to interpret in ecological terms, they identify the mechanisms by which pollutants affect organisms.

Because EPA's modeled concentrations of the metals aluminum, arsenic, cadmium, chromium, lead, nickel, and zinc are well below levels documented to cause adverse effects in aquatic and aquatic-dependent species, we do not anticipate adverse effects to listed species from these metals discharged from vessels, even given the uncertainties associated with the assessment. We expect that monitoring of discharges from new builds will assist in the reduction of uncertainty regarding these concentrations and assist in the verification of assumptions made herein.

Copper

Copper concentrations estimated by EPA from anti-fouling hull coatings and other discharges range up to 2.99 ug/L pre-permit and 2.33 ug/L post-permit in hypothetical estuarine harbor scenarios, and up to 0.164 ug/L pre-permit and 0.131 ug/L post-permit in hypothetical river harbor scenarios, suggesting an approximately 20 percent reduction in discharges of dissolved copper. Post permit copper estimates fell below the response thresholds selected by EPA for evaluating risks of exposure to copper. These thresholds ranged from 3.5 to 119 ug/L for freshwater organisms and from 7.9 ug/L to 249 ug/L for estuarine/marine organisms. The threshold values EPA selected for freshwater and estuarine/marine invertebrates and for marine vertebrates were derived from NOECs and LOECs. The invertebrates NOECs were 2.5 ug/L and 6.1 for freshwater and estuarine/marine organisms, respectively. The NOEC for estuarine/marine vertebrates was 249 ug/L.

EPA predicts that high-end concentrations are likely to be overestimations of even worst-case scenarios. However, we believe that given the uncertainties in EPA's exposure analysis, species may be exposed to copper concentrations from vessel discharges that not only meet but exceed estimates calculated by EPA. In addition, vessel discharges are likely to combine with other sources of copper in waterways to produce even higher concentrations than those predicted in this analysis. At present, copper impairs 14,000 miles of rivers and streams (12% of impaired rivers and streams) and 964 square miles of bays and estuaries (43% of impaired bays and estuaries). Due to the high sensitivity of aquatic species to copper, adverse effects to listed species are likely to occur if copper discharges from vessels reach concentrations predicted in this analysis.

As described by EPA, copper is a nutritionally essential inorganic element for all species, both plant and animal, and thus has the potential to accumulate in tissues of aquatic 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. Toxicity of copper to aquatic organisms is dependent on pH, temperature, alkalinity, hardness, and concentrations of bicarbonate, sulfide, and organic ligands. Synergistic toxicity is suggested for mixtures of copper and aluminum, iron, zinc, mercury, anionic detergents, or various organophosphorus insecticides (Eisler 2000).

In the BE, EPA describes effects of copper from studies that directly measure changes to survival, growth, and reproduction. In addition to these endpoints, exposure to contaminants may result in sublethal effects to aquatic and aquatic-dependent wildlife that may affect a species' fitness, either alone or in combination with other physiological or environmental stressors. In many cases, these effects have been found to occur at concentrations lower than those that directly affect survival, growth, or reproduction. They may also be lower than criteria that have been deemed protective of aquatic life. In the case of copper, effects upon chemoreception and behavior have been particularly well-studied and documented in aquatic species. Though absent from the BE, EPA acknowledges and describes these effects in its report to Congress:

> "At relatively low concentrations, copper is toxic to a wide range of aquatic organisms, not just fouling organisms (CRWQCB, 2005). Concentrations as low as 5 to 25 µg/L can be lethal for marine invertebrates (Chambers et al., 2006). Elevated copper levels affect growth, development, feeding, reproduction, and survival at various life stages of fish, mussels, oysters, scallops, crustaceans, and sea urchins. High copper levels also change the types of phytoplankton that thrive in boat basins (Calabrese et al., 1984). Low levels of dissolved copper affect the olfactory capabilities in juvenile Coho salmon, which is critical for homing, foraging, and predator avoidance (Baldwin et al., 2004). The effect of copper on olfaction of juvenile salmonids suggests that copper might affect other fish species, too. Most effects on fish are sublethal (e.g., they may hinder metabolic processes, reproduction, development, activity levels and behavior). Thus, the damage is chronic and less noticeable than, for example, fish kills caused by sudden oxygen depletion (Evans et al., 1994)."

Hecht and colleagues compiled data on sensory effects to juvenile salmonids exposed to dissolved copper (Hecht et al. 2007). In their analysis, benchmark concentrations ranged from 0.18 – 2.1 µg/L, corresponding to reductions in predator avoidance behavior from approximately 8 – 57%. Recent work is consistent with previous observations (Baldwin et al. 2011, McIntyre et al. 2012). These effects can manifest over a period of minutes to hours and can persist for weeks. In aquatic systems, chemoreception is one of oldest and most important sensory systems used by animals to collect information on their environment and generate behaviors involved in growth, reproduction, and survival (Pyle and Mirza 2007). These behaviors include recognition of conspecifics, mates and predators, food search, defense, schooling, spawning and migration. Stimuli are perceived by sensory structures and converted to electrical signals that are conducted to the central nervous system where the information is integrated and appropriate behavioral responses are generated (Baatrup 1991). Detection of chemical signals involves not only recognition of a spectrum of unique compounds or mixtures but also their spatial and temporal distribution in the medium (Atema 1995). Sensory receptors are in direct contact with the environment, and therefore pollutants may disrupt normal chemosensory function by masking or counteracting biologically relevant chemical signals or by causing direct morphological and

physiological damage to the receptors (Baatrup 1991).

These concentration thresholds for juvenile salmonid sensory and behavioral responses fall within the range of other sublethal endpoints affected by dissolved copper, such as behavior, growth, and primary production (Hecht et al. 2007). In addition to sensory effects, impaired disease resistance, hyperactivity, impaired respiration, disrupted osmoregulation, pathology of kidneys, liver, and gills, impaired function of olfactory organs and brain, altered blood chemistry, and enzyme activity have been documented in fish, with effects thresholds as low as 0.1 ug/L (Eisler 2000). Like olfactory effects, other sublethal effects may manifest quickly and persist after exposure ceases. For example, rapid elevations of plasma cortisol, an indicator of stress, occurred in rainbow trout after a 1-hour exposure to approximately 0.2 µg/L of copper at a hardness of 12 mg/L (Munoz et al. 1991). The elevated plasma cortisol levels were maintained throughout the experiment's duration of 21 days. Elevated plasma cortisol levels are indicative of stress, and potentially represent a diversion of energy from normal physiological processes that may render salmonids more vulnerable to disease.

Information for effects on other anadromous species groups is limited to data for white sturgeon. This species group, particularly early life stages, is highly sensitive to copper, with LC20 response thresholds falling within the range of concentrations causing sensory effects in salmonids as described by Hecht et al. (2007) (Vardy et al. 2011, Little et al. 2012). Data on copper for marine mammals are limited to tissue concentrations (Varanasi et al. 1994, Sanpera et al. 1996, Wagemann et al. 1996, Wood and VanVleet 1996, Sydeman and Jarman 1998, Meador et al. 1999, Welfinger-Smith et al. 2011).

Sublethal effects from low levels of copper have also been documented in mussels and other mollusks. Exposure to copper at concentrations <10 ug/L has been demonstrated to affect valve closure rate, ability of glochidia to infect host fish, growth, swimming behavior, oxygen consumption, ammonia excretion, and spawning frequency (Eisler 2000). In freshwater mussels, mean 10% inhibition concentrations were as low as 3.1 ug/L for survival and 5.5 ug/L for growth following 28 days of exposure (Wang et al. 2009). Even lower concentrations produced effects to biomass and survival in rainbow mussels (*Villosa iris*) (Wang et al. 2011). Both metrics varied with concentration of dissolved organic content (DOC), and resulted in reductions to biomass at levels as low as 0.8 ug/L (biotic ligand model normalized EC20; 2.5 mg/L DOC) and to survival at levels as low as 1.7 ug/L (biotic ligand model normalized EC20; 10 mg/L DOC) (Wang et al. 2011). The 48 hour EC10 for development of black lip abalone was somewhat higher, with obvious morphological deformities in veliger occurring at concentrations above 3.7 ug/L (data serving as a surrogate for NMFS listed abalone (Gorski and Nugegoda 2006)).

As described above, effects of copper can manifest following very short exposures to copper (minutes to hours). Therefore, the potential for species to be exposed can result from both chronic loading and acute discharges. EPA's Report to Congress indicated that average concentrations of dissolved copper were found to exceed benchmark screening levels in every sampled discharge type except for outboard engine and generator engine effluents. Concentrations in these discharges ranged from 22 – 100 ug/L, far exceeding effect thresholds described above.

EPA's copper monitoring plan will identify areas of high concentrations of ships with copper-hull coatings. Where these areas overlap with sensitive listed species, additional actions will be taken to reduce exposure. Because copper toxicity can manifest quickly (minutes to hours) at very low concentrations, we expect adverse effects such as disrupted chemoreception to occur even with this monitoring in place. However, we expect that EPA's monitoring plan will prevent these effects from rising to the level of jeopardy.

Selenium

EPA reported that selenium was detected in several vessel discharges authorized by the VGPs (EPA BE and Report to Congress). Among those, discharges from Exhaust Gas Scrubbers have the potential to generate substantial selenium loadings. As described above, exhaust gas scrubbers are used to clean the exhaust gas system on marine diesel engines. These discharges can be highly acidic, and can contain traces of oil, polycyclic aromatic hydrocarbons (PAHs), nitrogen, selenium and other metals. Relatively few vessels currently use seawater scrubber technology, however, more restrictive air emissions standards, adopted in 2008, will become enforceable in 2012 (USEPA 2011b). To comply with these new standards, which require lower emissions of SOx, vessels can either burn low sulfur fuel oil or clean the exhaust gas. Cleaning, or scrubbing, the exhaust gas is generally accepted to be more economical (USEPA 2011b). Therefore, EPA anticipates that the number of vessels using exhaust gas scrubbers may increase significantly (EPA BE Table 3-3).

In their Biological Evaluation, EPA estimated selenium loadings to Reference Action Area (RAA) harbors from vessel discharges authorized under the VGPs, resulting concentrations of selenium in harbor waters, and risk to listed species from selenium exposure. EPA modeled several RAA harbor scenarios using different values for selected parameters, i.e. flushing time, percentage of vessels equipped with wet exhaust gas scrubbers, and concentration of selenium in scrubber effluent. Scenarios expected to produce the highest selenium concentrations in water also predicted the highest selenium loading. For some harbors, loads exceeded 30 lb Se/day and would represent a significant new source of selenium. For example, recent studies of the San Francisco Bay-Delta report selenium loading from oil refineries, which along with agricultural drainage comprise the two major sources of selenium to the system, of 1287 lb Se/year in 2009 or about 3.5 lb Se/day (Presser and Luoma 2010a). EPA predicted selenium loading to San Francisco Bay, from vessel discharges, of 30.3 lb Se/day (Post-permit: EPA BE Table G-9), which is nearly 9 times greater than loading from refineries and more than twice the combined load from refineries and agriculture.

Following discussions with the Services, EPA re-assessed their original loading estimates for selenium. They acknowledged that their initial predictions of the number of vessels that would be using exhaust gas scrubbers (10% of vessels in specific classes) were unrealistically high. They also re-examined the analytical data for selenium in exhaust gas scrubber washwater effluent and concluded that concentrations may be unexpectedly high because of matrix interferences found in seawater. EPA revised their loading estimates for selenium using two approaches to account for potential interferences (USEPA 2012a). The revised selenium loadings and predicted concentrations of selenium in RAA harbors and substantially lower than

the original estimates (USEPA 2012a).

It is worth noting that while EPA considers their original loading estimates to be unrealistically high they, nevertheless, concluded that risks to listed species from these high selenium discharges would be "negligible" (for anadromous salmonids/sturgeons and marine mammals) or "remote" (for all other taxa) (EPA BE Table 5-42). These conclusions were based on risk quotients which ranged from 0.002 to 0.136 for surrogate animal taxa. In addition, to account for any possible (although unexpected) difference in sensitivity between a listed species and its surrogate, EPA opted to apply its recently published safety adjustment factor of 2.0 (Mayer et al. 2008) to the Agency's criterion continuous concentration (i.e. chronic criterion) to further evaluate risk to listed species. Risk quotients using the safety adjustment factor approach were 0.06 for saltwater and 0.1 for freshwater (EPA BE Table 5-56). Considering the relative magnitude of these predicted loadings (based on original estimates) compared to known anthropogenic sources of selenium, such as described above for the San Francisco Bay-Delta, it is unwarranted that EPA's methodology predicted that risks to listed species would so low.

As we analyzed the potential effects, on listed species, of selenium-containing vessel discharges we became aware of information regarding the chemical/biological dynamics of selenium in aquatic environments and effects to aquatic and aquatic-dependent species that did not appear to be considered by EPA in their risk assessment. Much of this information was produced as the outcome of prior consultations with EPA on water quality criteria for selenium in California. The following excerpt from a 2010 USGS report by Presser and Luoma describes the history and context for the derivation of selenium criteria intended to protect listed fish and wildlife species in California which, we believe, will help inform the risk assessment for the VGPs:

> "Adjustments to the development of Se criteria specifically for California were called for by 1) the USEPA through the National Toxics Rule (NTR) and the California Toxics Rule (CTR) (USEPA, 1992; 2000); and 2) the USFWS and National Marine Fisheries Service (NMFS) through their Biological Opinion (USFWS and NMFS, 1998 and amended, 2000). In general, these adjustments were necessary to consider 1) the bioaccumulative nature of Se in aquatic systems; 2) Se's long-term persistence in aquatic sediments and food webs; 3) the importance of dietary pathways in determining toxicity; and 4) protection of threatened and endangered species." (Presser and Luoma 2010b)

Specifically, pursuant to section 7(a) of the U.S. Endangered Species Act (ESA) (1973), EPA consulted with the USFWS and NMFS concerning EPA's rulemaking action for California. They submitted a Biological Evaluation for their review as part of the consultation process in 1994. This evaluation found that the proposed CTR was not likely to jeopardize the continued existence of any federally listed species or result in the destruction or adverse modification of designated critical habitat. In April of 1998, the Services sent EPA a draft Biological Opinion that found that EPA's proposed rule would jeopardize federally listed species. After discussions with the USFWS and NMFS, the EPA agreed to several changes in the final rule and USFWS and NMFS, in turn, issued a final Biological Opinion finding that EPA's action would not likely jeopardize the continued existence of federally-listed species. The agencies agreed that federally-listed fish and wildlife species that are aquatic system foragers would be protected under future

criteria and procedures for site-specific adjustments.

To achieve these goals and as part of the remedy for these problems, the EPA initiated an interagency project with the USFWS and the U.S. Geological Survey (USGS) to address issues of 1) a methodology for translation of a tissue guidelines to protective site-specific dissolved Se concentrations (implementation of tissue criteria); 2) inclusion of protection of wildlife species (i.e., federally-listed species) in regulatory methodologies; and 3) site-specific criteria development for the Bay-Delta (USEPA 1999).

A methodology for ecosystem-scale modeling of Se is now available (Presser and Luoma 2010b, a). Analysis from this biodynamically-based methodology showed, in general, that:

- A crucial factor ultimately defining Se toxicity is the link between dissolved and particulate phases at the base of the food web (i.e., K_d);
- Collection of particulate material phases and analysis of their Se concentrations are key to representing the dynamics of the system;
- Bioaccumulation in invertebrates is a major source of variability in Se exposure of predators within an ecosystem, although that variability can be explained by invertebrate physiology (i.e., $TTF_{invertebrate}$);
- TTF_{fish} is relatively constant over the range of species considered here; and
- Se concentrations are at least conserved and usually magnified at every step in a food web."

The ecosystem-scale model developed by Presser and Luoma (Presser and Luoma 2010b, a) reveals the importance of understanding the biodynamics of selenium in aquatic environments to more accurately predict selenium toxicity. Conventional methods that relate water column selenium concentrations to tissue concentrations using bioaccumulation factors (BAFs), as was used in in the BE, have a high degree of uncertainty (Presser and Luoma 2010b). We reevaluated the risk assessment for selenium to examine how the analysis might benefit from a more informed, ecosystem-scale approach. We adapted the model used to support criteria development to estimate tissue concentrations (referred to as EC_T values in the BE) from concentrations of selenium in water (Table 31). For this evaluation we selected parameter estimates (Kd's and TTFs; Table 32) that were developed by Presser and Luoma (Presser and Luoma 2010a) for the San Francisco Bay-Delta system, as well as, estimates used to represent a hypothetical estuary (Presser and Luoma 2010b). We incorporated effects estimates for fish and bird reproduction (i.e. larval edema/skeletal defects and hatchability) and survival that were derived by the USFWS and used by Pressor and Luoma (Presser and Luoma 2010a) to predict water concentrations necessary for protecting sensitive listed fish and bird species in the San Francisco Bay-Delta.

Our revaluation indicates that EPAs, BAF-based, analysis likely underestimates risks of selenium exposure to listed species. As previously mentioned, EPA concluded (based on their original loading estimates) that risks to listed species from selenium discharged by vessels would be negligible for some taxa and remote for others. In contrast, risk quotients calculated using the ecosystem-scale approach ranged from 2.05 to 79.8 (Table 33), indicating that significant effects

on reproduction and survival would likely to occur. Risk quotients for the alternative exposure scenarios A and B (where fewer vessels are discharging EGS washwater and the Se effluent concentrations are lower) were substantially lower. However, even at the lower exposures predicted under scenarios A and B some of the risk quotients for the San Francisco Bay-Delta model approached or equaled 1.0 for fish and exceeded 1 for birds. These results suggest that selenium from vessel discharges could pose a risk to listed species in the future if the number of vessels utilizing exhaust gas scrubbers increase beyond what is anticipated.

EPA has agreed to develop and implement a monitoring plan for gas-scrubber discharges. With technical assistance from the Services, EPA will identify metrics for assessing selenium loadings in waters inhabited by ESA- listed species. EPA will also state which analytical methods should be used to accurately measure selenium in saline gas-scrubber discharges. They will also compile NOI and annual report information regarding the number and type of vessels using exhaust gas scrubbers and discharge monitoring data for selenium. And finally, EPA will evaluate vessel data, identify ports/locations where the metric has been exceeded and will, as appropriate, identify and implement additional actions to reduce any unacceptable risk.

Table 30. Generalized steps in ecosystem-scale methodology for translation of a tissue Se concentration to a water-column Se concentration for protection of fish and aquatic-dependent wildlife. [Adapted from Table 5, Presser and Luoma, 2010b].

Translation of Tissue Criterion to Water-Column Concentration

• Develop a conceptual model of food webs in watershed.
• Choose toxicity guideline for fish or aquatic bird species in estuary.
• Choose fish or bird species to be protected in watershed.
• For fish, choose species-specific TTFfish or use default TTFfish of 1.1; for birds, choose species-specific TTFibird or use default TTFbird of 2.0.
• Identify appropriate food web(s) for selected fish or bird species based on species-specific diet.
• Choose site-specific TTFinvertebrate for invertebrates in selected food web(s) or use default TTFinvertebrate for species of invertebrate (see list in Presser and Luoma, 2010).
• Choose site-specific Kd or use Kd indicative of a) generalized source of Se and receiving water conditions or b) site-specific hydrologic type and speciation; or a default Kd of 1000 (see list in Presser and Luoma, 2010).
• Solve equation(s) for allowable water-column concentration for protection of fish or birds (i.e., predator)
If assume single invertebrate diet, then
o Cwater = (Cpredator) ÷ (TTFpredator) Kd (TTFinvertebrate)
If assume a mixed diet of invertebrates, then
o Cwater = (Cpredator) ÷ (TTFpredator) (Kd) [(TTFinvertebrate a) (prey fraction)] + [(TTFinvertebrate b) (prey fraction)] + [(TTFinvertebrate c) (prey fraction)]
If assume sequential bioaccumulation in longer food webs, then
o Cwater = (Cpredator) ÷ (TTFpredator) Kd (TTFinvertebrate a) (TTFforage

fish)

- Cwater = (Cpredator) ÷ (TTFpredator) Kd (TTFTL2 invertebrate) (TTFTL3 invertebrate) (TTFTF3 fish)

  where TL = trophic level

Table 31. Equations used to calculate Fish and Bird ECT's for the BO; adapted from Presser and Luoma, 2010a.

Translation of Water-Column Concentration to Tissue Concentration
Fish EC_T = $C_{fish\text{-}clam\ pathway}$ = C_{water} * K_d * TTF_{clam} * TTF_{fish}
Fish EC_T = $C_{fish\text{-}invert\ pathway}$ = C_{water} * K_d * TTF_{invert} * TTF_{fish}
Bird EC_T = $C_{bird\text{-}clam\ pathway}$ = C_{water} * K_d * TTF_{clam} * TTF_{bird}

Table 32. Kd and TTF values used to calculate fish and bird ECT's.

Kd and TFF values		
	SanFrancisco Bay-Delta[1]	Hypothetical Estuary[2]
Kd	5784	3000
TTF_{invert}	2.8	1.3
TTF_{clam}	17.1	6.25
TTF_{fish}	1.1	1.1
TTF_{bird}	2.6	1.8

[1] from Presser and Luoma 2010a, Kd value is average of four time periods (Table 17)
2 from Presser and Luoma 2010b, Figure 6D&E.

Table 33. Application of the Ecosystem-scale approach a) estimate tissue concentrations of selenium from water concentrations predicted from vessel discharges and b) calculating risk quotients for sensitive fish and bird species (adapted from Presser and Luoma 2010a&b)

							Risk Quotients (EC_T/EC_{10} or NEC)			
	Percentage of vessels with EGS[1]	Se conc. in EGS discharge[1]	Max Se concentration in RAA Harbors (ug/L)[1]	Fish EC_T (Insect food web) (ug/g dw)	Fish EC_T (clam food web) (ug/g dw)	Bird EC_T (clam food web) (ug/g dw)	Adult Female sturgeon EC_{10} (larval edema and skeletal defects)=8.1ug/g dw	Juvenile salmon whole body EC_{10} (mortality)=1.8 ug/g dw	Generic insect-eating fish whole-body NEC=5.0 ug/g dw	Bird (scoter and scaup) egg $EC_{10(hatchability)}$=7.7 ug/g dw
				San Francisco Bay-Delta						
Original BE estimate	10	100	2.39	42.6	260.0	614.6	32.1	23.7	8.5	79.8
Revised estimate - scenario A	1	14.5	0.035	0.6	3.8	9.0	0.5	0.3	0.1	1.2
Revised estimate - scenario B	1	31.2	0.075	1.3	8.2	19.3	1.0	0.7	0.3	2.5
				Hypothetical Estuary						
Original BE estimate	10	100	2.39	10.3	49.3	80.7	6.09	5.70	2.05	10.48
Revised estimate - scenario A	1	14.5	0.035	0.2	0.7	1.2	0.09	0.08	0.03	0.15
Revised estimate- scenario B	1	31.2	0.075	0.3	1.5	2.5	0.19	0.18	0.06	0.33

[1] data from Addendum to EPAs BE

Classical Pollutants

Classical pollutants combine several standard water quality parameters such as conductivity, salinity, temperature, pH, Biochemical Oxygen Demand (BOD), etc. along with other parameters EPA defined as conventional pollutants such as total suspended solids (TSS) and total residual chlorine (TRC). In the BE, EPA limited its analysis to TRC and TSS. EPA also stated that only those pollutants with toxic effects (i.e., TRC) to aquatic and aquatic-dependent species lend themselves to analysis of quantitative effects.

Biological Oxygen Demand (BOD)

BOD is a measure of the oxygen used by microorganisms to oxidize organic matter present in water. Bacteria working to decompose organic matter consume oxygen in the process, exerting an oxygen demand in wastewater and receiving waters, thereby depressing dissolved oxygen (DO) concentrations. If there is a large quantity of organic waste in water, there will be substantial demand for oxygen, resulting in a high BOD and the associated potential to depress DO concentrations in receiving waters. As the waste is consumed or dispersed through the water, BOD levels will begin to decline.

Vessel discharges that exert relatively high oxygen demand include fish hold effluent, bilgewater, deckwash and graywater. BOD levels in these discharges exceed the screening level benchmark for wastewater discharges of 30 mg/L[12] by at least an order of magnitude. Although BOD is highest in fish hold effluent, averaging 840 mg/L, the net oxygen demand only accounts for less than 3% of a the total organic matter discharge from a fishing vessel. Due to its greater volume, graywater accounts for 90% or more of a vessel's organic matter discharge even though BOD levels are barely over half that of fish hold effluent, averaging 430 mg/L. Deck washdown and bilge water BOD levels are 430 and 190 mg/L, respectively and account for less than 0-6% of a vessels organic matter discharge, depending on service class. EPA estimated post permit BOD in the estuarine harbor model averaged 0.16 mg/L and ranged from 0.005 to 0.68 mg/L, suggesting a 16 to 26 percent reduction in organic material discharged. For the river harbor model, EPA's average estimated post permit BOD was 4.17 mg/L and ranged from 0.0459 to 9.11 mg/L, suggesting a 1 to 9 percent reduction in organic material discharged.

The implications of BOD loading on dissolved oxygen in receiving waters were not evaluated by EPA. However, organic enrichment and oxygen depletion is the fourth most common cause for impairment among 303(d) listed waters. Nearly 4,250 waters of the United States are listed with this impairment, with about 550 of them marine environments (388 miles of coastline and 3000 square miles of ocean, bays and estuaries). Low dissolved oxygen alters biological assemblages and the food webs of which they are a part. The oxygen status of water also influences biologically mediated environmental fate processes of pollutants like mercury methylation and

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[12] The screening level benchmark for BOD is based on the 1984 secondary treatment limit for publically owned treatment works (see 49 FR 37006, Sept. 20, 1984).

metabolism of organic chemicals. While organic loading from the many land-based sources are orders of magnitude larger than that from vessels, vessel discharges pose yet another incremental source to those land based loadings. Data are not available to place vessel organic loading in context of consequences for our species. We review the effect of low dissolved oxygen on listed resources and their surrogates to acknowledge the potential effects of this discharge constituent on listed resouces.

Intergravel dissolved oxygen concentrations for survival of embryo and larval salmonids need to be at least 8 mg/L dissolved oxygen, which requires dissolved oxygen levels of 11 mg/L in overlying water. Other salmonid lifestages require 8 mg/L in the water column, with impairments to productivity occurring when levels decline to 6 mg/L or lower (LD and CA 20056). Sturgeon basal metabolism, growth, consumption and survival are all very sensitive to changes in oxygen levels, which may indicate their relatively poor ability to oxyregulate. Based on bioenergetics and behavioral responses of young of the year juveniles aged 30 to 200 days, productivity losses occurred at oxygen saturation levels below 60 percent, which corresponds to 5 mg/liter at 25°C (Wirth et al. 2000). Accordingly, dissolved oxygen levels of 5 mg/L and above are also considered protective of sturgeon (Southworth et al. 2011).

Marine mammals and marine turtles are air-breathing organisms and are not expected to be directly affected by increased BOD or associated decline in dissolved oxygen. Johnson's seagrass is tolerant of low dissolved oxygen. Reported effects of low dissolved oxygen on abalone species include mortality, immune suppression (Sturgeon 1954), and reduced growth rate (Movahedinia et al. 2012). Holding conditions specified in the recovery plan for this species require dissolved oxygen levels to be maintained at or within 10% of saturation. However, intertidal species like black abalone are tolerant to extremes in environmental conditions such as fluctuations in dissolved oxygen (Alam et al. 2000). Searches of the open literature and the NMFS status review for coral species did not identify issues with depletion of dissolved oxygen or BOD.

Direct linkage of BOD with adverse effects on listed species is complicated by extrapolating the organic loading represented by BOD to actual levels of the proximate stressor, low dissolved oxygen. The VGPs limit discharges into impaired waters and discharges of high BOD waste streams in protected areas. The VGPs also require BOD monitoring of graywater in new build vessels. These restrictions lead NMFS to conclude that BOD contributions from vessels compliant with the VGPs are not likely tip the balance between neutral and harmful levels of organic loading to waters where our species occur.

Total Residual Chlorine (TRC)

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, 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 2010a). These products are referred to as chlorine produced oxidants. Total residual chlorine was detected in bilgewater, deck washdown, and graywater discharges. Eighty eight to nearly 100% of the total residual chlorine load from vessels is discharged in graywater at concentrations ranging from 13 mg/day from utility vessels to nearly 2 kg per day from large Cruise ships (USEPA 2010b).

The post permit TRC concentrations in EPA's estuarine harbor model averaged 0.1 ug/L and ranged from 0.002 ug/L to 0.0007 mg/L, suggesting an approximately 25 to 70 percent reduction in TRC discharged. For the river harbor model, EPA's average post permit TRC was 0.0014 mg/L and ranged from 0.0144 ug/Lto 0.0032 mg/L, suggesting a 3 to 15 percent reduction the amount of TRC discharged. These post permit TRC estimates fell below the response thresholds selected by EPA for evaluating risks of exposure to TRC. These thresholds ranged from 3.7 to 50 mg/L for freshwater organisms and from 7.5 ml/L to 46.5 ml/L for estuarine/marine organisms. The threshold values EPA selected for freshwater and estuarine/marine vertebrates and for freshwater vertebrates were derived from NOECs and LOECs. The vertebrate NOECs were 6 ml/L and 40 ml/L for freshwater and estuarine/marine organisms, respectively. The NOEC for freshwater invertebrates was 2 ug/L.

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 does not persist in the aquatic environment in the form in which it was discharged. As such, listed species and their critical habitat are not expected to be exposed to total residual chlorine at levels which could cause adverse effects.

Total Suspended Solids

EPA's estimated post permit TSS concentrations in the estuarine harbor model averaged 0.0281 mg/L and ranged from 0.0015 to 0.13 mg/L, suggesting a 35 to 55 percent reduction in TSS discharges. For the river harbor model, EPA's post permit TSS averaged 0.72 mg/L and ranged from 0.00764 to 1.62 mg/L, suggesting a 35 to 40 percent reduction in TSS discharges. Post permit TSS estimates fell below the response threshold of 3.33 mg/L selected by EPA for evaluating risks of exposure to TSS. This threshold is based on EPAs existing ambient water quality criteria converted into TSS equivalents.

Direct effects of suspended materials on invertebrates and fish are complex, ranging from behavioral to physiological to toxicological. Suspended sediments have been documented to have a negative effect on the survival of fish, freshwater mussels, and other benthic organisms. In a frequently cited review paper prepared by Newcombe and Jensen (1996), sublethal effects (e.g. increased respiration rate) were observed in eggs and larvae of salmonids and nonsalmonids, as well as in adult estuarine and freshwater non-salmonids, when exposed to Total Suspended Solids concentrations as low as 55 mg/L for one hour. Mussels compensate for

increased levels of suspended sediment by increasing filtration rates, increasing the proportion of filtered material that is rejected, and increasing the selection efficiency for organic matter. Excess sediment smothers benthic organisms and the surface layer of the benthos can be heavily impacted and altered. Increased turbidity associated with suspended sediments can reduce primary productivity of algae as well as growth and reproduction of submerged vegetation (Jha, 2003). In addition, once in the system, resuspension and deposition can "recycle" sediments so that they exert water column and benthic effects repeatedly over time and in multiple locations.

A review of effects thresholds described in studies published between 1969 and 2001 supports EPAs conclusion that TSS discharges authorized by the VGPs alone would not result in adverse effects (Rowe et al. 2003). While TSS loading from the many land based sources are orders of magnitude larger than that from vessels, vessel discharges pose yet another incremental source to those land based loadings. The RQ for vessel associated TSS loading in freshwater environments is 3 orders of magnitude lower than the threshold for a risk conclusion of "remote" (RQ=0.1). The RQ for eastuarine/marine environments is higher, but still ~ an order of magnitude below the threshold for a risk conclusion of "remote" (RQ=0.1). NMFS concludes that TSS contributions from vessels are therefore not expected to tip the balance between neutral and harmful levels of TSS in waters where our species occur.

VOCs and SVOCS

The toxicity of oil and grease measured as SGT-HEM to aquatic and aquatic-dependent animals is most often attributed to the water-soluble monocyclic aromatic hydrocarbons or MAHs (benzene, toluene, ethylbenzene, and xylene, otherwise known as BTEX), and the polycyclic aromatic hydrocarbons or PAHs (McGrath and DiToro, 2009). EPA's study of cruise ship graywater found a total of 17 different volatile and semi-volatile organic compounds in at least 10 percent of samples, for which the most significant rates and levels of detection were phthalates, phenol, and tetrachloroethylene. EPA also found elevated pththalates (bis(2-ethylhexyl)phthalate) in a few bilgewater and stern tube packing gland effluent samples, as well as benzene in bilgewater (USEPA 2010b).

EPA stated that for many of the VOCs and SVOCs, there are no definitive chronic toxicity test data available for aquatic organisms, as well as no data to support selection of response thresholds based on growth or reproduction. NMFS concurs with EPA regarding the availability of response thresholds for the many VOC and SVOC substances detected in vessel discharges. Those organics for which EPA conducted quantitative analysis include PAH and MAH compounds, bis(2-ethylhexyl) phthalate and 2,4,6-trichlorophenol. The MAH compounds like benzene are low-molecular-weight compounds that usually do not persist in the environment because they are volatile and highly biodegradable. Conversely, the PAH compounds (particularly those with four or more carbon rings) are high-molecular-weight compounds that are less volatile and degrade more slowly, and thus, persist in the aquatic environment. The services are supplementing EPAs assessment of VOCs and SVOCS to include nonylphenol and octylphenol ethoxylates due to the greater toxicity and environmental persistence the degradate nonylphenol (NP).

PAH and MAH

Because MAHs and PAHs are Type I narcotic compounds, the toxicity of these compounds in water (and sediments) is most easily assessed using what is called the target lipid model or TLM (DiToro and McGrath, 2000). Under the TLM, acute and chronic toxicity of MAHs and PAHs to aquatic animals is predicted as single pollutants and as mixtures by calculating the critical body burden in the target lipid of an organism. EPA used the critical body burden of benzo(a)pyene and benzene in lipid to develop its response thresholds against which modeled environmental concentrations would be compared. Under EPAs estuarine harbor model, average post permit PAH and MAH concentrations were unchanged for all analytes but benzene, which was reduced between 17 and 53 percent under worst case and best case conditions, respectively. For the river harbor model, average post permit PAH and MAH concentrations were unchanged for all but benzene, which was reduced between 6 and 9 percent, depending on scenario.

The TLM guidelines proposed by McGrath and DiToro (2009) are equivalent to a HC5, which is the hazard concentration considered to protect 95% of the tested species. Thresholds for individual analytes ranged from 0.286 ug/L for bezo(a)pyrene to 2,433 for benzene ug/L. Average RQs calculated by EPA for individual analytes in modeled estuarine/marine harbors ranged from <0.0001 to about 0.022 while modeled river harbors ranged from <0.0001 to approximately 0.002. Since these thresholds are all based on the same mechanisms of action, narcosis, EPA summed the individual RQs to estimate the aggregate toxic risk of MAH and PAH using the modeled harbor concentrations. RQs for PAH and MAH ranged from 0.002 to 0.08 in estuarine/marine scenarios and from <0.0001 to 0.0022 in river scenarios.

PAHs (and to a lesser extent, MAHs) accumulate in tissues of aquatic organisms as well as biomagnify up the food chain, through partitioning and subsequent bioaccumulation in the lipid fraction of organisms. TLM-calculated critical body burdens, when paired with tissue concentrations predicted by multiplying the maximum toxicant concentration in receiving water with the freely dissolved BAF in prey species, provides the critical body burden in the lipid of prey organisms which serves as a dietary response threshold. EPA chose to include on multiple routes of exposure in the BE, but limited its analysis to potential risks for bioaccumulations to just benzo(a)pyrene and benzene because: 1) benzo(a)pyrene is the most toxic of the PAHs being considered here, and thus, the most relevant for evaluating risk, and 2) benzene is the only MAH for which a critical lipid body burden exists to support risk evaluation from multiple routes of exposure (as well as exposure via oral ingestion by wildlife). In describing this decision, EPA noted that PAHs are biotransformed and detoxified in organisms, so they generally do not bioaccumulate to particularly high levels. RQs calculated for freshwater vertebrates and invertebrates based on estimated concentrations accumulated in tissues from continuous exposure to maximum concentrations of benzo(a)pyrene and benzene in ambient receiving waters under post-permit conditions were all below 0.01.

Information on the toxicity of hydrocarbons to coral species is extremely limited. A recent review compiled information on molecular mechanisms and effects of pollutants on coral (Rotchell and Ostrander 2011). The limited information on this species group does not indicate effects at concentrations suggested by EPA's modeling. Responses ranged from alterations in enzyme activity at 5 ug/L after 1 day exposure to benzo(a) pyrene, an order of magnitude higher

than EPA's HC5 TLM for benzo(a)pyrene 0.28 ug/L. The services were unable to identify information on the effects of PAH and MAH substances on aquatic vegetation.

In addition to survival and reproductive effects assessed by EPA, PAHs have been identified as having immunotoxic (Reynaud and Deschaux 2006, Bravo et al. 2011) and neurodevelopmental (Danion et al. 2011, He et al. 2011, He et al. 2012, Shi et al. 2012) effects in marine and anadromous fish. A recent investigation of PAH implications for threatened and endangered Chinook salmon reported mean PAH concentrations measured in some stomach content samples from wild caught fish were near the threshold concentrations associated with variability and immune dysfunction in juvenile salmonids (Yanagida et al. 2012). This study demonstrated that dietary PAH exposure, rather than water column concentrations (which ranged from 0.001 to 0.005 ug/L), was the primary source of exposure for juvenile salmonids. Dietary threshold concentrations calculated for pink salmon in EPAs BE were 38.7 and 71.7 mg/kg wet weight for benzo(a)pyrene and benzene, respectively. However, these thresholds were for PAH induced mortality alone and did not take into account other effects which may also lead to mortality, such as effects on immune function. Mortality was 60% after 14 days among Chinook salmon fed 16.5 +/- 0.1 mg/kg total PAH and challenged with *Listonella anguillarum*. Mortality among vaccinated PAH exposed fish was markedly lower (approximately 16%, read from figure). The PAH exposure mixture which only contained about 0.72% benzo(a)pyrene, was formulated to approximate the actual PAH profile observed in the stomach contents of juvenile wild Chinook (Palm et al. 2003). Increased spinal curvature rates in rockfish embryos occurred after 7 days exposure to ambient concentrations of benzo(a)pyrene as low as 0.126 ug/L. While statistically significant, the difference between controls and embryos exposed to 0.126 ug/ml B(a)P appeared to be less than 5% (read from figure). The incidence rate for spinal deformities was 80% at 1.26 ug/L B(a)P (He et al. 2011).

Information regarding PAH and MAH marine turtles is limited to tissue concentrations (Alam and Brim 2000, Camacho et al. 2012) and much of the information on PAH and MAH for marine mammals are tissue concentrations (Hellou et al. 1991, Fair et al. 2010, Godard-Codding et al. 2011). Tissue concentration information confirms exposure and uptake of PAHs by marine turtles occurs, the ecological implications of such exposures are difficult to interpret. There are also a few biomarker and in vitro studies for marine mammals. Two investigations establish point of effect exposures by identifying DNA adducts or biomarkers of exposure/toxicant metabolism for marine mammals (Wilson et al. 2005, Godard-Codding et al. 2011). Environmental PAH concentrations and the presence of PAH DNA adducts have been of implicated in the occurrence of cancer within the St. Lawrence population of beluga whale (Ray et al. 1991, Beland et al. 1993, Martineau et al. 1994). One in vitro investigation demonstrated cytolotoxic effects of benzo(a)pyrene in cell cultures from Northern Right Whale, with greatest toxicity found for testis. Another investigation exposing harbor seal lymphoma cell lines to benzo(a)pyrene identified genototic, but not immunotoxic effects (Frouin et al. 2010). While data from biomarker and in vitro studies are difficult to interpret in ecological terms, they identify the mechanisms by which pollutants can affect organisms.

Both pre- and post- permit receiving water concentrations are orders of magnitude below individual toxicant effects thresholds identified by EPA and those thresholds identified through our review of recent literature as was the net PAH-MAH toxicity for freshwater taxa. The aggregate PAH-MAH toxicity for saltwater organisms was somewhat higher, with an RQ of 0.08, approaching the threshold between "remote" and "no effect". NMFS concludes that PAH-MAH contributions from vessels are not expected to tip the balance between neutral and harmful levels in waters where our species occur.

2,4,6-Trichlorophenol

EPA post permit 2,4,6-trichlorophenol in the estuarine harbor model to average 0.0927 ng/L and ranged from 0.00607 to 0. 514 ng/L, suggesting a 48 to 50 percent reduction in 2,4,6-trichlorophenol discharged. For the river harbor model, EPA's average post permit 2,4,6-trichlorophenol was 0.00376 ng/L and ranged from 0.0000313 to 0. 00986 ng/L, suggesting a 30 to 67 percent reduction in 2,4,6-trichlorophenol discharged. Post permit 2,4,6-trichlorophenol estimates fell below the response thresholds selected by EPA for evaluating risks of exposure to 2,4,6-trichlorophenol. These thresholds ranged from 0.46 to 5,923 ug/L for freshwater organisms. The threshold values EPA selected for freshwater vertebrates were derived from a NOEC and LOEC, with the NOECs being 530 ug/L. EPA stated that threshold values were not available for estuarine/marine organisms. After review of the available literature, concurs with EPA's conclusion. Further, NMFS literature searches did not identify information of effects important to our species such as olfaction, immune function or behavior.

Both pre- and post- permit receiving water concentrations are greater than 3 orders of magnitude below toxicant effects thresholds identified by EPA and thresholds below those identified by EPA or for effects not considered by EPA were not found through NMFS literature reviews. NMFS concludes that 2,4,6-trichlorophenol contributions from vessels are not expected to tip the balance between neutral and harmful levels in waters where our species occur.

Bis(2-Ethylhexyl) Phthalate (DEHP)

EPA estimated post permit DEHP in the estuarine harbor model to average 0.754 ng/L and range from 0.048 to 4.24 ng/L, suggesting a 25 to 61 percent reduction in DEHP discharged. For the river harbor model, EPA's average post permit DEHP was 0.0839 ng/L and ranged from 0.000979 to 0.175 ng/L, suggesting a 14 to 58 percent reduction in DEHP discharged. EPA stated that it was unable to find threshold response data for DEHP useful in evaluating risk to aquatic organisms.

NMFS identified several studies evaluating DEHP as an endocrine disruptor. California EPA reviewed these and other studies evaluating more traditional effects (Carlisle et al. 2009). Response threshold data were reviewed for effects on development and maturation, growth, and reproductive tissue metrics. The concentrations at which responses were reported ranged from 0.01 to 10,000 ug/L. The only responses that occurred at concentrations at or below those modeled by EPA for its hypothetical harbors used the standard laboratory species medaka. Mortality rates were higher and male body weights were lower for adult individuals hatched from eggs exposed to as little as 0.01 ug/L DEHP. Exposure did not affect female body weights or the gonadosomatic index (GSI) of either sex (Chikae et al., 2004a – after Carlisle et al 2009).

A later study by the same workers reported declines in female body weights at 0.1, 1.0, and 10.0 µg/L and reduction of male GSI at 0.01 – 10 ug/L (Chikae et al., 2004b – after Carlisle et al 2009). Interpretation of the ecological implications of such data in the light of conflicting results reported by the same research group leads us to exclude this study from consideration when evaluating EPAs post permit DEHP estimates.

NMFS concludes that DEHP contributions from vessels are not expected to tip the balance between neutral and harmful levels in waters where our species occur because the bulk of the response thresholds reviewed in Carlisle et al (2009) occurred at concentrations orders of magnitude greater than ambient water concentrations modeled by EPA and conflicting data from the same research group were reported for those responses observed at or near EPAs modeled concentrations.

Nonylphenol and Octylphenol Ethoxylates

EPAs Report to Congress describes the environmental fate and transport of nonylphenol and octylphenol ethoxylates in vessel discharges (USEPA 2010b), yet the BE does not acknowledge this discharge constituent class. These substances are often collectively referred to as nonylphenols and represent two distinct subsets of the broader family of alkylphenols that are commonly used in many products such as liquid detergents and soaps. Longer chain nonylphenol and octylphenol ethoxylates degrade to shorter chained ethoxylates under aerobic conditions. In general, the shorter the chain, the more hydrophobic, persistent, and toxic the substance becomes. 4-Nonylphenol (NP) is a shorter-chain nonylphenol that has been found in surface water and is toxic to aquatic life. NP is formed from the longer chain nonylphenol and octylphenol ethoxylates as they break down. In general, the hydrophobicity, persistence, and toxicity of the substance all increase as the ethoxylate chain becomes shorter. NP accumulates in sediment, occurring at concentrations several orders of magnitude greater in sediment than in water. The short-chained isomers may be quite persistent once they are buried in the sediment, and bottom-feeding fish can be significantly exposed to these persistent and toxic compounds (USEPA 2010b).

EPA's Report to Congress indicated that there were no analytical results which could be directly compared with EPA's 1.7 ug/L marine acute benchmark for the degradate, NP. The Services note that Toxic Equivalency Factors are available for adjusting for NPE mixtures. An alternative to a Toxic Equivalency analysis would be to estimate NP degradate loading equivalents based on moles NP producing substances in vessel discharges.

Toxicity data indicate that some listed species are greater than the proposed correction factor of two more sensitive to these substances than standard test species (e.g., LC50s[13] for atlantic sturgeon is 50 ug/L, shortnose sturgeon and spotfin chub LC50s were reported at 80 ug/L (Dwyer et al 2005), Fathead minnow LC50s averaged 290 ug/L and sheepshead minnow averages 460 ug/L in EPAs ECOTOX database). NP is a common adjuvant used in pesticides

[13] LC50 refers to lethal concentration, 50% or the amount of a toxin required to kill half the members of a tested population after a specified test duration.

and was reviewed by NMFS in its Biological Opinion for the EPA-NMFS consultation on *Pesticides Containing Chlorpyrifos, Diazinon, and Malathion* (NMFS 2008d). In a review of ECOTOX data in this Biological Opinion, the lowest fish LOEC reported was 0.15 ug/L for fathead minnow reproduction. Numerous fish studies reported LOECs at or below 10 ug/L. Additionally, salmonid prey species are also sensitive to sublethal effects of nonylphenol. The amphipod, *Corophium volutator*, grew less and had disrupted sexual differentiation (Brown et al. 1999). Multiple studies with fish indicated that nonylphenol disrupts fish endocrine systems by mimicking the female hormone 17B-estradiol (Arsenault et al. 2004; Brown and Fairchild 2003; Hutchinson et al. 2006; Jardine et al. 2005; Lerner et al. 2007a; Lerner et al. 2007b; Luo et al. 2005; Madsen et al. 2004; McCormick et al. 2005; Segner 2005). NP induced the production of vitellogenin in fish at concentrations ranging from 5-100 ug/L (Arukwe and Roe 2008; Hemmer et al. 2002; Ishibashi et al. 2006; Schoenfuss et al. 2008). Vitellogenin is an egg yolk protein produced by mature females in response to 17-β estradiol, however immature male fish contain the capacity to produce vitellogenin if exposed to estrogenic compounds. As such, vitellogenin is a robust biomarker of exposure. A retrospective analysis of an Atlantic salmon population crash suggested the crash was due to NP applied as an adjuvant in a series of pesticide applications in Canada (Brown and Fairchild 2003; Fairchild et al. 1999). Additionally, processes involved in sea water adaptation of salmonid smolts are impaired by NP (Jardine et al. 2005; Lerner et al. 2007a; Lerner et al. 2007b; Luo et al. 2005; Madsen et al. 2004; McCormick et al. 2005).

Since graywater discharges primarily contain the NP precursors, there is little concern that direct acute exposures to NP discharge plumes would occur. Additionally EPA proposes to add the following sentence to the VGP and sVGP fact sheets definition of minimally toxic: "EPA expects that soaps that are "minimally toxic" will contain little to no nonylphenols." For these reasons, NMFS concludes that nonylphenols from vessel discharges are not expected to result in harmful levels in waters where our species occur.

Programmatic Summary

In this section, we summarize whether and to what degree EPA has structured the VGPs, as well as monitoring and compliance, to insure that vessel discharges into waters of the U.S. are not likely to jeopardize the continued existence of endangered species or threatened species or result in the destruction or adverse modification of critical habitat that has been designated for those species. In this evaluation, we ask whether or to what degree EPA has structured its proposed permits so that the Agency (1) understands the scope of its action; (2) reliably estimates the physical, chemical, or biotic stressors that are likely to be produced as a direct or indirect result of their action; (3) reliably estimates the exposure of ESA-listed resources (species and designated critical habitat) to these stressors; (4) collects and monitors information on authorized activities throughout the life of the permit; (5) evaluates the information to assess how its actions have affected listed resources; (6) monitors and enforces permit compliance; and (7) modifies its action if new information (including inadequate protection for species or low levels of compliance) becomes available.

(1) Scope

In this section, we ask whether EPA is aware of the scope of their Action. Section 7 regulations define an action as "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." It defines effects of the action as the "direct and indirect effects of an action on the species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action that will be added to the environmental baseline. Indirect effects are those that are caused by the proposed action and are later in time, but still are reasonably certain to occur. Interrelated actions are those that are part of a larger action and depend on the larger action for their justification. Interdependent actions are those that have no independent utility apart from the action under consideration."

As described earlier, the scope of the action includes all aspects of EPA's issuance and implementation of the VGPs, including the monitoring of discharges authorized by the permits and enforcement of the permits. The VGPs include several references to monitoring, compliance, and enforcement, all of which will be conducted by EPA as part of their action. In sections 1.4 of the VGP and 4.3 of the sVGP, EPA requires permit compliance and affirms its authority to determine enforcement responses to permit violations including fines, requirements, and schedules for taking corrective actions. Section 3.1 of the VGP discusses EPA's authority (either an EPA official or official agent acting on EPA's behalf) to determine that effluent limits/water quality standards/pollution control measures/BMPs have not been met and how an applicant can take corrective actions. Section 4.5 of the sVGP discusses EPA's authority to inspect any vessel, equipment, practices, or operations regulated under the permit; and to sample or monitor for the purpose of assuring permit compliance. It also discusses EPA's authorities under Section 308 of the Clean Water Act, including its authority to enter, access, inspect, sample, monitor, and obtain information to verify compliance with the CWA.

In addition to specific references to these activities, through the course of our consultation, EPA has committed to several monitoring, compliance, and enforcement activities, discussed in the relative sections below (see 4, 6, and 7). Therefore, we conclude that EPA is aware of the full scope of its action.

To reliably estimate the probable individual or cumulative effects to ESA-listed species or designated critical habitat, EPA would need to know or reliably estimate the probable number of discharges that it would authorize under the VGPs. Therefore, we also ask whether EPA has structured their general permits to reliably estimate the probable number, location, and timing of discharges that would be authorized by their VGPs. For the VGP, EPA requires a Notice of Intent (NOI) for all vessels greater than or equal to 300 gross tons or with the capacity for 8 cubic meters of ballast water. This NOI must be submitted electronically no later than December 12, 2013 for existing vessels, by date of transfer for resale vessels, and seven days prior to entering waters of the U.S. for new builds. The NOI asks for location information, including homeport, anticipated port visits during the term of the permit, and whether the vessel is engaged in nearshore voyages. The NOI also asks information on the types waste streams that would be discharged (e.g. exhaust gas scrubber, graywater, bilgewater, and ballast water). The EPA used the NOI data from the 2008 VGP to estimate the number of discharges that would be authorized

under the VGP. They did not estimate the location and timing of discharges, nor did the account for the vessels that do not submit NOIs; for this they should use the annual reports in the future.

The sVGP is a new permit; unlike the VGP, it will not require NOIs or annual reports. Therefore, even after the permit goes into effect, EPA will not know or be able to reliably estimate the number or location of the vessels authorized to discharge under the sVGP. Instead, EPA uses (and would continue to use) the U.S. Coast Guard's MISLE database to search for vessels that are less than 79 feet in length, except for recreational and military vessels. They estimate that 115,000 to 138,000 vessels would be *eligible* for coverage under the sVGP. They do not estimate how many vessels of these vessels are likely be covered under the sVGP, nor how many vessel owners/operators will consider themselves to be covered under the sVGP or self-identify (i.e., have the knowledge that they are discharging wastes under a Federal permit).

Accurate self-identification is an acknowledged problem among general permits. For example, self-identification by municipal and industrial storm water dischargers was among the metrics evaluated in a study of 136 facilities subject to stormwater General Permit requirements (Cross and Duke 2008). Among these facilities, approximately 21% were aware that their discharges were subject to the permit even though stormwater permits had been in place over 15 years. The authors suggested that outreach might improve compliance among dischargers. The rates of self-identification among municipal and industrial storm water dischargers do not necessarily apply to vessels authorized to discharge under the either of the VGPs. Cultural differences among these regulated communities and the well-defined vessel populations and discharges specified under the VGPs should lead to greater awareness of vessel operator obligations under these permits. An online query for information on the VGPs indicated information sharing activity during and after the VGPs comment period among professional associations, service providers, state and federal agencies within the US and US trading partners.

Although there is evidence that self-identification rates for the VGPs may be greater than that for other General Permits, EPA has not at this time reliably estimated the actual number, location, and timing of discharges authorized by the sVGP. One way in which EPA will address these estimates is through inspection of a random sample of small vessels and extrapolation to the larger population of small vessels. As described below (under the compliance section, number 6), one question that will be asked during small vessel inspections is whether vessel owners/operators consider their vessels to be authorized under the sVGP and perform actions (e.g., signing a PARI form and quarterly inspections) required of the permit.

We conclude that EPA is aware of the scope of their Action and has reliably estimated the probable number of discharges that it would authorize under the VGP. Though EPA has not reliably estimated the probable number of discharges that it would authorize under the sVGP, we expect that their proposed inspection and compliance plan will allow them to do so.

(2) Stressors

Here we ask whether the EPA has reliably estimated the physical, chemical or biotic stressors that are likely to be produced as a direct or indirect result of the vessel discharges that would be authorized by the VGPs. We also ask whether the EPA would know or be able to reliably

estimate whether those discharges have occurred in concentrations, frequencies, or for durations that violate the terms of the proposed VGPs.

ANS

In the BE, EPA identifies ANS as one of the greatest threats to biodiversity globally and one of the largest threats to coastal and freshwater ecosystems. They describe invasive species as the second largest threat to endangered species after habitat loss and indicate that some scientists believe the impact of invasive species is greater than that of habitat loss. They identify commercial shipping as the "most important means of introducing ANS to coastal environments" and the "predominant vector for the human transport of invasive species in marine environments."

The EPA identifies several mechanisms by which ANS are likely to adversely affect listed resources. These include: outcompeting native species, damaging habitat, changing food webs, and altering the chemical and physical aquatic environment. They also identify the taxonomic range of possible ANS, dividing the organisms by those likely to be transported via ballast water (planktonic, but also benthic organisms) and biofouling organisms.

The EPA does not estimate the concentrations, frequencies, or durations of discharges carrying ANS. In the BE, EPA suggests that the lack of species-specific chronic response data for ANS precludes calculation of a risk quotient to support a quantitative effects analysis. We found several datasets describing actual or potential responses to ANS. In our analyses, we used past documented ANS invasions from a nationwide database to determine that listed resources are likely to be exposed to ANS, but our analyses do not include all ANS invasions and do not represent invasion probabilities. Instead, we agree with the conclusion of the NAS study, which states that "the data are not sufficient in present form to characterize a biologically meaningful relationship, much less estimate the associated uncertainty, to be able to identify with confidence the invasion probabilities associated with particular discharge standards." For this reason, EPA cannot quantify ANS invasion probabilities associated with authorized ballast water discharge, hull fouling/hull husbandry, and other vessel discharges. Instead, they use a qualitative approach to estimate exposure (discussed below in section 3).

The EPA has identified ANS as potentially harmful stressors that are likely to be released into the waters of the U.S. as a direct or indirect result of the vessel discharges that would be authorized by the VGPs. To date, EPA has not reliably estimated whether those discharges have occurred in concentrations, frequencies, or for durations that violate the terms of the proposed VGPs; however, their monitoring and reporting plan, discussed below (section 4) will allow EPA to reliably estimate the volumes of discharges that are likely to carry ANS and the number of living organisms carried in ballast water.

Pollutants

In their BE, EPA used pollutant data collected for the Report to Congress to characterize the discharges the VGPs will authorize. This required extrapolation from the sampled 61 vessels and their discharges to the universe of vessels authorized to discharge under the VGPs. Given

that the sVGP authorizes discharges from between 115,000 and 139,000 vessels, discharge estimates are based on data from approximately 0.05% of the permitted population. The vessels sampled for the report were commercial fishing vessels and other non-recreational vessels less than 79 feet, that is to say, sVGP-authorized vessels. Vessels described in the Report to Congress volunteered to allow EPA to sample discharges. This "opportunity sampling" approach likely favored participation of operators who were confident that their vessel discharges were low impact.

In its report, EPA discussed whether these data could be extrapolated to other vessels, including larger vessels, such as those authorized to discharge under the VGP. EPA cautioned against extrapolating bilgewater data to vessels not sampled for the Report to Congress (e.g., cruise ships, ferries, barges, tankers) because these vessels differ considerably in design, construction, and operation. The BE also used information from a 2008 report on cruise ship discharges (USEPA 2008), but this report only provides data for bilgewater volume in these larger vessels, not data on the pollutants present in bilgewater Ultimately EPA characterized VGP bilgewater discharges using bilgewater pollutant data from the Report to Congress and bilgewater volume data from the cruise ship discharges report. EPA also cautioned against extrapolating deckwash and graywater volume and constituents among vessels because these would differ due to activities conducted on board. For example, fishing vessels and cruise ships would have very different graywater and deckwash characterisitcs. Nevertheless, voluntary discharge data from 61 small vessels were used to characterize discharges from VGP vessels.

EPA considered zinc discharges from cathodic protection sacrificial anodes to be of too low volume and toxicity to be considered in the BE. NMFS calculated zinc load using the discharge rate reported in BE Table 3-3 and the wetted hull area estimates reported in BE table 3-18. These calculations indicate that sacrificial anode discharges can actually make up a substantial portion of the zinc load discharged from some vessels. Sacrificial anodes contribute approximately 20% to 80% of the zinc discharged from barges, cargo ships and oil tankers at pierside and approximately 50 to 95% of zinc discharges from these vessels while underway. Under worst-case conditions (i.e. oil tanker anodes), the inclusion of anode discharges would result in a five-fold increase in zinc loading while pierside and 17 fold increase while underway. Fortunately, increasing the modeled zinc estimates by 17 fold does not appreciably change the risk conclusions of the BE analysis.

While the analyte list examined in the BE was extensive, there were a number of omissions that should be noted:

- EPA states that mercury was not evaluated because it was not possible to sample away from sources of metals or sources of airborne mercury such as engines or generators, which may contaminate the sample. In stating this, EPA indirectly acknowledges that mercury is a contaminant generated on vessels and that it does not know the relative sources or extent to which mercury is discharged from vessels.
- EPA acknowledges that pharmaceuticals and personal care products (PPCPs) would occur in graywater and sewage mixed with graywater but stated that PPCPs were not

analyzed due to a lack of resources. EPA therefore does not know the types of PPCPs and the extent to which they are discharged by vessels.

- While the Report to Congress included data for nonylphenols, EPA did not acknowledge this data in the body of the BE.

The breadth of the extrapolations and the omissions made by EPA when characterizing the discharges authorized by the VGPs indicate substantial gaps and uncertainty in EPA's knowledge of the pollutant discharges authorized by the VGPs. While the Services acknowledge that uncertainty exists in all risk analyses, the magnitude of uncertainty introduced into the BE analysis due to EPA's lack of knowledge regarding authorized discharges is substantial. This further emphasizes the need for EPA to monitor discharges to gather additional information to more accurately identify stressors and determine which may be problematic. The EPA will monitor discharges from the following vessel systems: ballast water treatment systems, exhaust gas scrubbers, oily water separator, cruise ship graywater, and copper-based anti-fouling hull coatings.

(3) Exposure to potentially harmful impacts

In this section, we ask whether EPA has reliably estimated whether and to what degree listed resources are likely to be exposed to potentially harmful impacts of discharges authorized by the permit. EPA identified uncertainties in their effects analysis that could potentially limit the interpretation of results and conclusions. In order to address whether EPA has demonstrated sufficient knowledge of the exposures resulting from discharges authorized by the VGP, we must first review these uncertainties acknowledged by EPA for the BE's assessment.

Uncertainties in Exposure Estimates Identified by EPA

The EPA used simple screening models to represent the estuarine and river water bodies and to estimate the exposure concentrations evaluated in the risk analysis of all U.S. waters navigable by vessels covered under the VGP and sVGP. They acknowledged that the use of such screening-level models is typically limited to identifying major water quality issues and to identify data gaps, not to perform risk analyses as they were used in the BE.

EPA acknowledged that uncertainty was added to the analysis by the extrapolations needed to fill data gaps for the various combinations of vessel types, discharges, pollutants, and vessel populations. These extrapolations were required to develop the pollutant loads used in the modeling analysis. EPA also assumed that the background concentration for all pollutants in the harbor scenarios was zero. Therefore, the model output used to inform the effects analysis does not directly evaluate the potential for vessels to contribute to pollutant impairments or quantify if vessel pollutant loads serve as the "tipping point" between exceeding and not exceeding a given effect threshold. Rather than assess pollutant load contribution to existing baseline levels in the RAAs it modeled, EPA relied on the magnitude of risk quotients to provide a sense as to whether incidental vessel discharges covered under the VGP and sVGP would make notable or sizable contributions to existing harbor pollutant levels.

Impacts on a local scale potentially occurring in stagnant side embayments, coves within the larger water bodies, or occurring as a result of large volume discharges are not addressed in EPAs assessment because screening models assume instantaneous and universal mixing within the modeled harbor. EPA concluded that by using model parameters to bound in the analysis and reflect a "worst case" scenario (e.g., minimum river flow from the nine rivers considered in the analysis), the screening-level models appropriately characterize the range of potential exposure concentrations and subsequent pollutant risk.

EPA believes that the analysis also represents vessel discharges to intracoastal waterways because several of the major water bodies (e.g., Elizabeth River, Charleston Harbor, Biscayne Bay) that are part of the Atlantic and Gulf intracoastal waterway were considered in developing the receiving characteristics for the estuary model. Even though a significant amount of vessel traffic travels these waters on a daily basis, the daily vessel equivalent population is assumed to be significantly lower in these waters than in a harbor environment. EPA acknowledged that listed species present in intracoastal waterways may be directly exposed to a vessel discharge plume, but they also stated that the exposure duration is expected to be brief such that the chronic effects thresholds considered in the BE are not of concern.

Although the environmental concentrations calculated for the effects analysis do not specifically account for localized effects, EPA considered the potential risk posed by vessel discharges in other unique environments and determined vessel discharges are unlikely a concern. For example, in the case of localized changes in buffering capacity due to climate change, exhaust gas scrubber discharges have the greatest potential to depress pH values in receiving waters as the discharge can have pH values as low as 3(USEPA 2011b). EPA originally relied on its economic and benefit analysis for the 2013 VGP and concluded that that a maximum of 5 to 10 percent of affected vessels would install this treatment by the end of the permit period. EPA has since revised this estimate to reflect far fewer vessels plying this technology. EPA assumed that the concurrence of site-specific changes in buffering capacity with a significant population of vessels discharging exhaust gas scrubber water is highly unlikely to occur. Similar to the assessment of intracoastal waterways, EPA assumed that any localized effects caused by direct exposure to an exhaust gas scrubber discharge plume would be brief such that risk effects to listed species considered in the BE are not of concern. However, the discharge volume and concentrations of selenium associated with this discharge was identified as a major contributor to existing selenium issues in San Francisco.

Lakes and reservoirs isolated from the major rivers navigable by vessels covered under the VGP and sVGP represent another unique environment that was not specifically modeled in EPA's analysis. EPA believes that the maximum exposure concentrations estimated in the hypothetical river harbor can be used to inform effects determinations for species identified in the lake environment. 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, EPA concluded that the permitted vessel equivalents in lake environments are very likely to be significantly lower than the vessel equivalents estimated in the two river RAAs included in the effects analysis, which would result in lower estimated pollutant concentrations in a lake environment than the maximum exposure

concentration calculated for the hypothetical river harbor. Given this assumption, EPA believes that the calculated river EC values and subsequent risk conclusions for freshwater species adequately capture the potential "worst case" scenario pollutant concentrations and risks in a lake environment.

Services Assessment of Uncertainty in the Exposure Analysis

In order to determine if EPA's decision process in evaluating its action has insured against jeopardizing listed species, the Services must evaluate EPA's acknowledged uncertainties and how they were addressed by EPA along with additional uncertainties identified by the Services. The Service took EPA's acknowledged uncertainties into consideration while identifying discharges and stressors that posed the greatest risk to listed species. Additional monitoring agreed to by EPA is expected to provide mechanisms for identifying unacceptable risks and implementing corrective measures.

Perhaps the greatest source of uncertainty in the effects analysis lies in EPA's ability to predict exposure to pollutants. EPA identified three major uncertainties in their exposure analysis, including: model pollutant load inputs reflect the data limitations in available vessel discharge data (discharge flow, discharge concentrations, and vessel populations); model inputs assume no background concentrations of pollutants are present in the receiving water; and the model is not designed to evaluate localized impacts or stressors.

Uncertainties regarding EPA's estimation of pollutant loadings from vessel discharges arise primarily for the limited amount of data for discharges, i.e., discharge volumes and concentrations of pollutants in those discharges for different discharge/vessel types. In terms of pollutant concentrations, where sample data were available EPA based their estimates on median values (EPA's report to Congress). Predictions based on medians may underestimate the full breath of potential exposures. Data collected as part of the monitoring requirements proposed in VGP and sVGP should improve the quality of information used for EPA's future analyses. Models used to predict concentrations of pollutants in water assumed no background concentrations in the receiving water. While these predictions may capture pollutant concentrations attributable to vessel discharges, they likely underestimate actual exposures because other sources of pollution are not considered. In addition, the models assume complete mixing throughout the harbor and are not designed to evaluate localized impacts or stressors. Organisms that reside in close proximity to vessels may be exposed to higher pollutant concentrations. Risks associated with these acute exposures will be dependent on the overlap between species and vessel locations, proximity to discharges, and duration of exposure. Sessile species will likely be most vulnerable to these types of exposures. Because of the scope of the VGP and sVGP, EPA evaluated exposure using Reference Action Area (RAA) harbors. The RAA harbors are expected to represent areas where the greatest numbers of vessels are located. EPA modeled several different harbor scenarios and based their risk assessment on the maximum predicted pollutant concentration (among all harbor scenarios). The RAA harbor approach and use of maximum predicted pollutant concentrations should help balance the uncertainties mentioned above and reduces the likelihood that pollutant exposures used in the effects analysis are underestimated.

ANS

For ballast water, EPA concludes that its proposed ballast water standards in the VGP are likely to result in a "very small absolute risk of invasion." Using the NEMESIS database (with the caveats listed above), we estimated an annual average of 5.75 newly documented ANS invasions nationwide that were attributed to ballast water discharge. The proposed VGP ballast water numeric discharge limitations are likely to reduce the number of ANS invasions over time; however, similar standards are not required of small vessels under the sVGP. We conclude that there is a small likelihood that listed resources will be exposed to ANS invasions as a result of ballast water discharges authorized by the VGPs.

EPA did not provide a qualitative or quantitative estimate of exposure of listed resources to hull fouling ANS. We used the NEMESIS database to estimate an annual average of 13.5 newly documented ANS invasions nationwide that were attributed to hull fouling. This number represents a minimum number of ANS invasions annually. The VGPs require few anti-fouling/hull husbandry practices or technologies that are likely to result in substantial changes to biofouling ANS invasion rates. Therefore, there is at least a small likelihood that listed resources would be exposed to ANS invasions as a result of hull fouling and hull husbandry as authorized by the VGPs.

Listed resources are also likely to be exposed to ANS as a result of authorized discharges of chain locker effluent, seawater piping effluent, and dead bait.

We conclude that EPA has not reliably estimated whether, and to what degree, listed resources are likely to be exposed to potentially harmful impacts of discharges authorized by the permit. We acknowledge that several, substantial uncertainties prevent or complicate the reliable estimation of exposure to several of these stressors. These uncertainties emphasize the need for additional monitoring and data collection, as described in the following section.

(4) Monitoring/Feedback

In this section, we ask whether EPA proposes to identify, collect, and analyze information about its authorized discharges that may expose listed resources to harmful stressors.

The EPA requires owners/operators to self-monitor their discharges as one means to identify, collect, and analyze information to determine whether vessel discharges into waters of the U.S. expose listed resources to ANS or pollutants at concentrations, durations, or frequencies that are known or suspected to produce adverse effects. Under requirements of the VGP, permittees are required to monitor and report any unavoidable or unauthorized discharges. This places the responsibility for oversight largely on the permittees who would have little incentive to do so, given that such observations could result in a violation of the permit and result in enforcement responses by the EPA. It is unlikely that the required self-monitoring and self-reporting will allow EPA to continually identify, collect, and analyze information to evaluate exposure or adverse effects. Therefore, EPA will review additional monitoring of discharges that have the greatest potential to adversely affect listed resources. These include: ballast water, bilge water, graywater, copper (from anti-fouling hull coatings and gas scrubber effluent), selenium (from gas scrubbers effluent), and vessel-mediated ANS.

Annually, EPA will collect and review information on the ballast water treatment system discharges through the electronic submittal of reports by all vessels using such systems. All ballast water treatment systems must meet the ballast water numeric discharge limitations, as described in the proposed VGP, which limits the number of biological organisms discharged to waters of the U.S. In addition, EPA will monitor the following data, submitted annually by vessel owners/operators:

- Monthly indicators of ballast water treatment system functionality.
- Annual (at minimum) calibration tests of all sensors and equipment.
- Results of analytical tests for biological indicators (i.e., total heterotrophic bacteria, *E. coli*, and *Enterococci*) to ascertain compliance with ballast water numeric discharge limitations.
- Results of analytical tests for residual biocides and derivatives used in the treatment process (e.g., chlorine dioxide, chlorine, ozone, peracetic acid, and hydrogen peroxide).

Annually, EPA will collect and review information on treated graywater through the electronic submittal of reports by large and medium-sized cruise vessels covered under the VGP.

Quarterly, cruise ship owners/operators must collect and analyze graywater samples for the following constituents: BOD, fecal coliform, suspended solids, pH, total residual chlorine, *E. coli*, TP ammonia, nitrate/nitrite, and TKN. They must submit these results to EPA on an annual basis.

For new build vessels (constructed on or after December 19, 2013) greater than 400 gross tons that elect to discharge bilgewater into waters subject to this permit, vessel owners/operators must collect and analyze a sample of the bilgewater effluent for oil content; they must submit these data electronically to EPA.

At least once annually, EPA will summarize data from Annual Reports (which include all instances of noncompliance and unavoidable discharges authorized by the VGP) and Discharge Monitoring Reports and provide summaries to the Services.

For copper, EPA will develop, in cooperation with the Services, and implement a monitoring and reporting plan, within 1 year of finalization of the VGPs. The plan will contain the following elements:

1) EPA, with the technical assistance of the Services, will identify metrics for assessing copper loadings to waters inhabited by ESA-listed species (i.e., using the number of ships reporting copper hull treatments in eNOIs for particular areas as a proxy).
2) EPA will compile eNOI and annual report information regarding the number and type of vessels that use copper-based anti-fouling hull coating and/or exhaust gas scrubbers, their home port/most frequented US port, and US ports anticipated visiting during the permit term.
3) Twice, once by December 2014 (to summarize eNOI data) and another before EPA transmits the next draft VGP/sVGP to OMB for interagency review (to evaluate annual

reports in time to make changes for the next permit cycle, if need be), EPA will evaluate vessel data and identify ports/locations where the metric has been exceeded. If the metric has been exceeded, EPA will, as appropriate, identify and implement additional actions to reduce any unacceptable risks.

For potential selenium discharges in exhaust gas scrubber effluent, EPA will develop, in cooperation with the Services, and implement a monitoring and reporting plan, within 1 year of finalization of the VGPs. The plan will contain the following elements:

1) EPA, with the technical assistance of the Services, will identify metrics for assessing selenium loadings to waters inhabited by ESA-listed species (i.e., using the number of ships reporting exhaust gas scrubbers in eNOIs for particular areas combined with expected and/or reported volume and concentration values as proxies).
2) EPA will state that vessel operators should analyze selenium in saline EGS discharges using methods capable of accurately measuring selenium in saline samples (e.g., Octopole Reaction Cell ICP-MS or Hydride generation AA methods or other methods which reduce matrix interference for selenium).
3) EPA will compile NOI and annual report information regarding the number and type of vessels that use exhaust gas scrubbers, discharge monitoring data for pollutants including selenium and copper, their home port/most frequented US port, and US ports anticipated visiting during the permit term.
4) Twice, once by December 2014 (to summarize eNOI data) and another before EPA transmits the next draft VGP/sVGP to OMB for interagency review (to evaluate annual reports in time to make changes for the next permit cycle, if need be), EPA will evaluate vessel data and identify ports/locations where the metric has been exceeded. If the metric has been exceeded, EPA will, as appropriate, identify and implement additional actions to reduce any unacceptable risk.

For vessel-mediated ANS invasions, EPA will develop, in cooperation with the Services, and implement a monitoring and reporting plan, within 1 year of finalization of the VGPs. The plan will contain the following elements:

1. EPA, with the technical assistance of the Services, will identify metrics to evaluate whether vessel mediated ANS invasion rates are being reduced over time.
2. EPA will track information from ANS invasions that might be potentially tied to vessel vectors. Information tracked may include:
    a. Dr. Greg Ruiz (Smithsonian Institution) conducts systematic surveys for ANS invasions caused by hull-fouling and ballast water in San Francisco, Chesapeake, and Tampa Bays to track changes in the rate of ANS invasions.
    b. Nationwide NEMESIS database for all newly documented, marine, ballast water and hull fouling ANS invasions. http://invasions.si.edu/nemesis/browseDB/searchQuery.jsp
    c. National USGS NAS database for all newly documented, freshwater, shipping-related ANS invasions. http://nas.er.usgs.gov/queries/SpSearch.aspx
    d. EPA and the Services acknowledge that new invasions observed from any of these sources are not necessarily linked to discharges authorized by the VGPs. Nonetheless, over a multi-permit lifespan, tracking invasions, and improving our

methodologies and understanding of the risk-release relationship, could provide a long-term perspective about ballast water discharge standards, antifouling technologies, and other practices to reduce invasions.

3. EPA will report to the Services once per year that they have reviewed this information and whether notable new invasions have been recorded in the previous year that could have been previously caused by vessel discharges. The Services note that they are requesting a short review (approximately one hour per year) to ensure EPA continues to track aquatic bioinvasions. As discussed above, EPA and the Services agree that these observations are being used to inform EPA's general understanding of continued and ongoing invasion rates.
4. EPA will identify and implement, as appropriate, additional actions (e.g., require anti-fouling coatings; require debris containment systems; require more stringent ballast water standards) as soon as practicable (including in future permit iterations) to ensure the protection of listed species if needed.

These ANS monitoring methods were chosen to provide an indicator of how invasion rates change over time. For example, researchers at the Smithsonian Environmental Research Center (SERC)[14] have conducted systematic surveys for non-native species in San Francisco Bay (past 12 years) as well as Tampa and Chesapeake Bays (past year). These studies were designed to provide an indicator of how ANS invasion rates change over time as regulations change. Data analyses include identifying the source location (e.g., South China Sea) of the ANS as well as the vector for the ANS (e.g., ballast water, biofouling, etc.). The databases provide a nationwide repository for newly reported ANS invasions in marine (NEMESIS) and freshwater (USGS NAS) environments; they do not account for changes in effort as above.

NMFS spoke with the Senior Scientist and Primary Investigator of the SERC Marine Invasions Research Lab and NOAA's Senior Liaison to the ANS Task Force, to determine whether these methods would be informative and whether they could be used as a gross measure of the efficacy of the VGPs. Both agreed that the systematic surveys were a good method to track the effectiveness of the permit. They acknowledged the lag time for ANS invasions (months to years after introduction) but were confident that the metric would be meaningful within the life of the permit (i.e., 5 years). They also saw the value in establishing base and trend lines in the early years of the permits. Finally, they explained that EPA could ask SERC to tailor their data analyses to link invasion rates directly to the VGPs. For example, they could restrict the dataset to ANS introduced by transoceanic commercial vessels, whose ballast water and anti-fouling/hull husbandry practices are authorized under the VGP.

As they pertain to ballast water, EPA's monitoring measures can be compared to USCG's Ballast Water Rulemaking. Under the VGP, permittees are responsible for self-inspections and monitoring; they are required to provide annual reports. EPA affirms "those data will be submitted to an electronic reporting tool and database currently in development. EPA plans to

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[14] See: http://www.serc.si.edu/labs/marine_invasions/index.aspx

make data submitted to the Agency in electronic form available to the public in electronic form." For vessels covered under the sVGP, self-monitoring reports are kept on board; though never sent to EPA, these reports could be checked during an inspection. Though plans for sVGP inspections are still under development, EPA assured the Services that a small subset of vessels covered under the sVGP would be inspected annually (Senior Manager's Meeting, dated October 19, 2012). In comparison, the U.S. Coast Guard program includes mandatory reporting and an existing system for receiving and archiving report information; they conduct inspections of nearly every vessel under their jurisdiction to corroborate report claims (NAS 2011). Thus, EPA's monitoring program is not as well-developed as that of the U.S. Coast Guard.

We conclude that EPA is likely to identify, collect, and analyze important information about its authorized discharges under its VGP. Their ability to collect monitoring data on vessel discharges authorized under the sVGP is dependent on the development of inspection plans, described below (under compliance, section 6). This monitoring information is essential because it will provide EPA with data needed to reliably estimate stressors and exposure throughout the life of the permit. It will also provide EPA with a means to determine the efficacy of its permit in protecting waters of the U.S. and the listed resources that depend upon them (as discussed further below, number 7).

(5) Effects on Listed Resources

Here we ask whether EPA has used analytical methodology that considers:

a) the status and trends of endangered or threatened species or designated critical habitat;
b) the demographic and ecological status of populations and individuals of those species given their exposure to pre-existing stressors in different drainages and watersheds;
c) the direct and indirect pathways by which endangered or threatened species or designated critical habitat might be exposed to the discharges to waters of the United States; and
d) the physical, physiological, behavior, sociobiological, and ecological consequences of exposing endangered or threatened species or designated critical habitat to stressors from discharges at concentrations, intensities, durations, or frequencies that could produce physical, physiological, behavioral, or ecological responses, given their pre-existing demographic and ecological condition.

As with uncertainties in its exposure analysis, EPA also identified uncertainties in the response analysis. In section 5.2.6.2 of the BE, EPA provided a comprehensive discussion of uncertainties associated with the selection of chronic toxicity values. In order to address whether EPA has demonstrated sufficient knowledge of the effects on listed resulting from exposures to discharges authorized by the VGP, we must first review these uncertainties for the BE's assessment and how EPA addressed them, then evaluate the additional uncertainties identified by the Services.

Uncertainty in Effects Analysis Identified by EPA

EPA acknowledged the uncertainty added to its analysis by the interspecies extrapolations necessary to fill data gaps in response toxicity threshold information for threatened and endangered species. The use of chronic effects data from tests with surrogate species

traditionally requires applying data for the closest taxonomically related species with appropriate data. EPA chose to analyze data for the most sensitive surrogate species rather than the closest taxonomically related species. This decision provides an added measure of conservatism in the risk analysis. EPA also assumed that listed species may be uniquely sensitive to pollutants compared to surrogate species. For example, comparative analysis of the extreme chronic sensitivity of the darter species, *Etheostoma fonticola*, and the generally-sensitive surrogate species rainbow trout, to copper (Besser et al., 2005a) along with comparisons made in a number of related studies by the same group of investigators, suggests that dividing water quality guidelines by a factor of 2.0 may be a reasonable safety factor (e.g., Dwyer et al. 2005; Gensemer et al. 2007; Mayer et al. 2008).

In addition to uncertainty associated with the use of effects data for standard lab species, there is uncertainty associated with response to the net toxicity of vessel discharges as mixtures. Organisms are simultaneously exposed to multiple pollutants in their natural environment, yet risk assessment, as a science, still focuses on effects based on exposures to single pollutants. Accordingly, the bulk of EPAs analysis evaluated toxicity on a single stressor basis. Mixture toxicity is extremely complicated to assess due to the vast number of combinations of pollutants that could be present in a mixture which differ in mechanism of action, biological availability, and propensity to interact with coocurring toxicants or water quality characteristics which influence toxicity and bioavailability. To address this uncertainty, EPA applied a concentration addition approach, summing the RQs for pollutants with the same toxicological mechanism of action: metals, which cause oxidative injury of biological tissues (Stohs and Bagchi 1995), and type 1 narcotics, such as PAHs which induce stupor through altering nerve transmission (USEPA, 2005c; McGrath and DiToro, 2009). In this analysis, EPA summed the RQs for water-column only exposure based on a mixture of metals or PAHs to give a total RQ representative of the risk of those mixtures to listed aquatic species.

EPA acknowledged issues with its use of NOECs (no observable effect concentration)/ LOECs (lowest observable effect concentration)/MATCs (maximum acceptable toxicant concentration), as sources of uncertainty that have the potential to influence the response analysis. EPA described the shortcomings of NOECs/ LOECs/MATCs as follows: "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 maximum acceptable toxicant concentration or 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 (see Mount et al., 2003 for more discussion)." EPA did acknowledge that use of point estimates of chronic toxicity, such as the EC_{10} or EC_{20}, overcome many of the shortcomings of the NOEC-LOEC approach, but the availability of point estimates associated with chronic effects is limited, 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.

EPA also asserted that it was correct in not considering sublethal and nontraditional effects when selecting its response thresholds. EPA stated that:

> "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."

And:

> "EPA relies on the *1985 Guidelines* data requirements and data selection process as the process by which information is fully-vetted for its utility and applicability in the BE analysis. "

Services Assessment of Uncertainty in the Repose Analysis

EPAs decisions to select the most sensitive surrogate species, assess risk of metals and PAH mixtures, and perform risk threshold analyses using water quality criteria adjusted by a factor of two to account for potential increased sensitivity of threatened and endangered species are, at this time, appropriate adjustments for the uncertainties introduced by interspecies extrapolation. However, EPA did not acknowledge that an adjustment of water quality criteria by a factor of two may also be necessary to account for the greater significance of effects on individuals to the survival of a population when protecting threatened and endangered species.

When selecting its response thresholds, EPA stated that it relies on the *1985 Guidelines* data requirements and data selection process. However, these guidelines also state:

> "For any threshold material, continuous exposure to any combination of concentrations below the threshold will not cause an unacceptable effect … However, it is important to note that this is a threshold of unacceptable effect, not a threshold of adverse effect. Some adverse effect, possibly even a small reduction in the survival, growth, or reproduction of a commercially or recreationally important species, will probably occur at, and possibly even below, the threshold. The Criterion Continuous Concentration (CCC) is intended to be a good estimate of this threshold of unacceptable effect."

What constitutes an unacceptable effect for threatened and endangered species was not addressed in EPA's guidelines. For some sublethal effects, such as olfaction, effects may occur at concentrations below which traditional endpoints (survival growth and reproduction) are affected (reference Copper section of the BO). If such responses are not considered when selecting chronic toxicity values, the resulting effects analyses may underestimate risk to susceptible species.

The services also have further comment on the ecological relevance of NOECs/LOECs and MATCs. Due to the hypothesis driven selection of response thresholds, the underlying "level of effect" associated with NOECs, LOECs, and MATCs may be high. For example, Suter et al. (1987) reported that MATCs for fish fecundity, on average, corresponded to a 42% level of adverse effect. These shortcomings can be addressed by using a regression approach to derive point estimates of chronic toxicity, such as the EC_{10}.

In conclusion, the Service evaluated EPA's BE and identified specific discharges (anti-fouling hull coatings and exhaust gas scrubbers) and pollutants (copper and selenium) that could pose substantial risks to listed species. Considering the uncertainties described above, EPA, with the technical assistance of the Services, will identify metrics for assessing selenium and copper loadings to waters inhabited by ESA-listed species. EPA will evaluate vessel data and identify ports/locations where the metric has been exceeded and, as appropriate, identify and implement additional actions to reduce any unacceptable risk.

ANS

According to the BE, EPA considered the individual life history for 612 threatened and endangered species (those identified by EPA to represent all listed species that occur in the action area) to assess exposure and the likelihood of adverse effects from ANS. They anticipate that both permits will "significantly benefit listed species due to the anticipated reduction in potential ANS release from ballast water due to the requirements of the VGP and sVGP." While we agree that ANS invasions are likely to be reduced as a result of the VGP ballast water numeric standards, we do not agree that there are similarly effective requirements in the sVGP. Furthermore, there are few requirements to prevent the introduction of ANS through hull fouling, underwater hull husbandry, and other discharges (i.e., chain lockers, seachests, and fish holds). In the BE, EPA describes a "relatively low historic potential for adverse impacts from ANS introductions," but their literature review identifies invasive species introductions as the second greatest threat to biodiversity, with shipping as the greatest source of ANS introductions to marine and coastal ecosystems. Our literature review similarly indicates that ANS introductions have led to the endangerment and extinction of species. Therefore, there is a high potential for adverse impacts from ANS introductions as a result of discharges authorized by the VGPs.

Despite anticipating a reduction in ANS introductions, EPA does not expect ANS invasions to cease as a result of their VGPs. In light of limited data and uncertainty quantifying the risk of ANS introduction, EPA concluded that ANS introductions as a result of their action "may affect" aquatic species and their critical habitat. They concluded that aquatic listed species, including all listed species under NMFS' authority, may be affected by new vessel-discharge-related ANS, and thus made a "may affect" determination for these species. We agree that the species considered by EPA may be affected by discharges containing ANS that are authorized by their VGPs.

(6) Compliance

Here we ask whether EPA has a mechanism to reliably determine whether and to what degree operators have complied with the conditions, restrictions, or mitigation measures required of the VGPs.

To monitor and insure compliance with its VGP, EPA will develop, with the technical assistance of the Services, and implement a monitoring and compliance plan by December 2013 to assess the performance of the permit in protecting listed resources. Currently, the U.S. Coast Guard conducts all inspections of vessels covered under the VGP using the EPA and USCG jointly developed Job Aid, i.e., "Guidelines for Coast Guard Evaluations of Compliance with the US

Environmental Protection Agency's (EPA) Vessel General Permit (VGP) for Discharges Incidental to the Normal Operation of Vessels." To insure that the inspections assess new requirements of the proposed VGP, EPA will:

- Work with the U.S. Coast Guard, with the input of the Services, to update the Job Aid to reflect requirements included in the new VGP.
- Create an online training and/or a webinar training regarding the VGP requirements/prohibitions, within one year of final permit issuance.
- Create a comprehensive checklist of requirements for inspector training in conjunction with the U.S. EPA National Enforcement Training Institute, within one year of final permit issuance.
- Provide and continue to provide the U.S. Coast Guard with the title, office name, current address and phone number for points of contact at headquarters offices and in each EPA Region and USCG District.

In addition, EPA's Office of Enforcement and Compliance Assurance will meet with the Services to discuss enforcement under the VGP. The EPA will gather, review and analyze inspection data submitted by the Coast Guard; the results will be summarized (by type of vessel, location when available, type of deficiencies, and number of deficiencies) and provided to the Services annually.

To monitor and insure compliance with its sVGP, EPA will develop and implement a monitoring and compliance plan by December 2013 to assess the performance of the permit in protecting listed resources. To insure inspection of a sample of small vessels, EPA will:

- Organize a meeting with their Office of Enforcement and Compliance Assurance for the Services to discuss and provide input to OECA's inspection and enforcement plans for the sVGP.
- Work with USCG, with the input of the Services, to update the inspection and enforcement document inspection checklist (i.e., Job Aide) to reflect sVGP requirements
- Conduct outreach to the states to raise awareness about small vessel requirements; EPA will encourage States to check records of quarterly self-inspection records and visible portions of the hull while inspecting vessels when implementing their own programs.
- Develop trainings covering the small vessel requirements and share these with the Coast Guard and States

The EPA will provide the Services with a yearly summary of the number of inspections (by EPA, the Coast Guard, States or other sources) and results (including number of inspections by type of vessel, location when available and number and type of deficiencies).

Unanticipated, low levels of compliance for either the VGP or sVGP constitute new information. In the VGPs, EPA includes a "Permit Reopener Clause." This clause allows EPA to modify its permits or require individual permits if new information becomes available. This information would also allow EPA to determine whether to request reinititation of formal ESA Section 7 consultation as provided in 50 CFR 402.16 and described in the Fact Sheets.

The VGP requires that vessel operators inform EPA of instances of noncompliance with the permit terms, including the results of monitoring, in annual reports submitted to EPA. The VGP also relies on the USCG to evaluate VGP compliance when conducting vessel inspections. Unlike other General Permits, VGP inspection and compliance data are entered into the USCG MISLE database, and EPA must rely on the Coast Guard to provide information regarding inspection results. The States may provide equivalent inspection and compliance data for the sVGP, though this mechanism has yet to be established. The EPA has assured the Services that at least a small subset of vessels covered under the sVGP would be inspected annually. These inspections direct bearing on EPA's ability to know whether discharges by individual permittees have violated the terms of the proposed VGPs.

In the sections on the Water Quality-Based Effluent Limitations, the VGPs state that if at any time the permittee becomes aware, or EPA determines, that the discharge causes or contributes to an exceedance of applicable water quality standards, the permittee must take corrective action. However, it is unlikely that EPA or an operator would become aware that its discharge has resulted in an exceedance of a water quality standard because the VGPs do not require either the operator or EPA to monitor for such exceedances for the vast majority of the discharges to be authorized. As such, the EPA cannot know or reliably estimate the physical, chemical or biotic stressors that are likely to be produced as a direct or indirect result of the activities to be authorized by the proposed permits.

It is important to note that the first stage of compliance under a general permit, accurate self-identification of discharges requiring coverage under the permit, is a recognized barrier to permit effectiveness. For the 2008 VGP, EPA received approximately 50,000 NOIs, indicating a relatively high level of self-identification for the VGP. In the course of consultation, the Services requested existing VGP inspection and compliance information from EPA. EPA provided an illustration identifying a total of 182 deficiencies found in the course of 22,500 inspections conducted by USCG between March and September 2011 under the VGP. EPA indicated that most deficiencies were corrected upon discovery and a total of 10 warning letters were sent out.

Levels of self-identification under the sVGP, however, are unknown because the permit is new. Under the sVGP, vessel owners/operators are not required to submit Annual Reports; the inspection plan for the sVGP is yet to be determined. For these reasons, self-identification and compliance under the sVGP may resemble that of other general permits, which is low.

In a study of 136 facilities subject to the stormwater permit requirements, researchers evaluated self-identification of dischargers under municipal and industrial storm water general permits (Cross and Duke 2008). Among these facilities, approximately 21% were aware that their discharges were subject to the permit and only 10% had completed the first stages of complying with the permit, even though stormwater permits had been in place over 15 years. While the authors suggested that outreach regarding the duty to comply might stimulate greater compliance among dischargers, they also noted that compliance would still likely be low because among those facilities whose managers were aware of their duty to comply, fewer than half pursued the required actions. The authors attributed the low rate of compliance to penalties that were not

sufficiently strict to motivate compliance.

To know or be able to determine reliably estimate compliance for dischargers participating in general permits, EPA must have an effective means of oversight. EPA collects information on NPDES permit compliance through its Online Tracking and Information System databases. We analyzed compliance and inspection data for EPA-issued general and individual permits (data downloaded September 2012). Our analysis indicated that inspected facilities had 3.5 to 4 times increased detection of permit violations as compared to uninspected facilities.

Issues identified with self-identification and compliance among EPA's other general permits do not necessarily apply to vessels authorized to discharge under the either of the VGPs due to cultural differences among these regulated communities and the well-defined vessel populations specified under the VGPs. Data from the USCG regarding compliance under the existing VGP indicate much higher rates of inspection and compliance. Data for the VGP indicated a violation rate of 182 deficiencies among 22,500 inspections (<1%). Given our findings regarding compliance rates among other general permittees, the newness of permit, and the lack of reporting requirments, we are concerned with noncompliance under the sVGP. Therefore, as described above, the development of an effective inspection plan is essential to ensuring compliance for vessels covered of the sVGP.

(7) Adequacy of Controls

Finally, we ask whether EPA has a mechanism to prevent or minimize listed resources' exposure to stressors in discharges if (1) EPA finds that these stressors occur at concentrations, durations, or frequencies that are potentially harmful to individual listed organisms, populations, or species; or (2) EPA identifies that the discharges lead directly or indirectly to ecological consequences that are potentially harmful to individual listed organisms, populations, species or primary constituent elements of designated critical habitat.

In the VGPs, EPA includes a "Permit Reopener Clause." This clause allows EPA to modify its permits or require individual permits if new information becomes available or if water quality standards are exceeded. The modification would be based on the receipt of new information that was not available at the time of permit issuance and would have justified the application of different permit conditions at the time of permit issuance. This new information may include data on ANS invasion rates, available technologies to minimize ANS invasions, discharge concentrations or volumes, pollutant risks, compliance or lack thereof, and adverse effects to listed resources. This information would also allow EPA to determine whether to request reinitiation of formal ESA Section 7 consultation as provided in 50 CFR 402.16 and described in the Fact Sheets.

For the VGP, with respect to ballast water discharges, new information that will be considered in determining whether to modify this permit includes, but is not limited to, data or information from permittees, the general public, states, academia, scientific or technical articles or studies, and results of monitoring conducted under this permit indicating that:

- Treatment technology has improved such that these improved technologies would have

justified the application of significantly more stringent effluent limitations or other permit conditions had they been known at the time of permit issuance;
• Treatment technologies known of at the time of permit issuance perform better than understood at the time of permit issuance such that this improved performance would have justified the application of significantly more stringent effluent limitations or other permit conditions had this been understood at the time of permit issuance;
• Scientific understanding of pollutant effects or of invasion biology has evolved such that this new information would have justified the application of significantly more stringent effluent limitations or other permit conditions had this been understood at the time of permit issuance; or
• The cumulative effects of any discharge authorized by the VGP on the environment are unacceptable.

As they pertain to ballast water, EPA's control measures can be compared to USCG's Ballast Water Rulemaking. Under the VGP, EPA will modify their permit and reinitiate if "new information" becomes available. In comparison, the U.S. Coast Guard program includes a mandatory review of ballast water numeric standards after 3 years. Though EPA's reopener clause provides flexibility in modifying the permit at any time, the Coast Guard's numeric standards review provides a systematic and concrete requirement for assessing the ability to require more stringent requirements.

As described in the Exposure Section, the VGPs require few anti-fouling/hull husbandry practices or technologies that are likely to result in substantial changes to biofouling ANS invasion rates. This is partially because developing technologies are not yet widely available. To ensure the protection of listed species, EPA will identify and implement, as appropriate, additional actions (e.g., require anti-fouling coatings (AFCs); require debris containment systems; require more stringent ballast water standards) as soon as practicable (including in future permit iterations). Before they transmit the next draft of the VGPs to OMB for interagency review, they will:

1) Compile information on new developments in AFCs (to minimize exposure to hull-fouling ANS and copper loading) and debris containment systems (to minimize the release of ANS and biocides during underwater husbandry).
2) Identify whether there are more efficacious BMPs or technologies for underwater husbandry and for anti-fouling coatings with fewer secondary environmental impacts that could be required of vessels.

With the technical assistance of the Services, EPA will identify metrics to evaluate whether vessel mediated ANS invasion rates are being reduced over time. As appropriate, EPA will identify and implement additional actions (e.g., require anti-fouling coatings; require debris containment systems; require more stringent ballast water standards) as soon as practicable (including in future permit iterations) to ensure the protection of listed species.
With the technical assistance of the Services, EPA will identify metrics for assessing vessel-related copper loadings to waters inhabited by ESA-listed species (i.e., using the number of ships reporting copper hull treatments in eNOIs for particular areas as a proxy). If the metric has been exceeded, EPA will, as appropriate, identify and implement additional actions to reduce any

unacceptable risks.

With the technical assistance of the Services, EPA will identify metrics for assessing vessel-related selenium loadings to waters inhabited by ESA-listed species (i.e., using the number of ships reporting exhaust gas scrubbers in eNOIs for particular areas combined with expected and/or reported volume and concentration values as proxies). If the metric has been exceeded, EPA will, as appropriate, identify and implement additional actions to reduce any unacceptable risk.

Aquatic nuisance species, copper, and selenium, are the stressors most likely to reduce individual fitness and population viability for listed species. In addition, ANS are likely to adversely affect the primary constituent elements of designated critical habitat. For vessel discharges associated with these stressors, EPA has committed to monitoring metrics, which if exceeded, will trigger additional protective measures. In addition, EPA will monitor, and implement when practicable, new technologies designed to minimize ANS invasions. Finally, if new information becomes available that would have justified different permit requirements, EPA will modify their permits and/or request reinitiation of Section 7 consultation. We conclude that EPA has adequate mechanisms to prevent or minimize listed resources' exposure to stressors in discharges.

Cumulative Effects

Cumulative effects include the effects of future state, Tribal, local or private actions that are reasonably certain to occur in the action area considered in this Biological Opinion. Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to Section 7 of the ESA.

The biological evaluation that EPA submitted to support its request for formal consultation and which is required to discuss cumulative effects (as they are defined for the purposes of section 7 of the ESA) did not identify future State, Tribal, local, or private actions that were reasonably certain to occur in the action area and that would not require Federal authorization, Federal funding, or the actions of a Federal agency. During this consultation, NMFS searched for information on future State, Tribal, local or private actions that were reasonably certain to occur in the action area. NMFS conducted electronic searches of business journals, trade journals and newspapers using *First Search*, Google and other electronic search engines. Those searches produced no evidence of future private action in the action area that would not require Federal authorization or funding and is reasonably certain to occur. As a result, NMFS is not aware of any actions of this kind that are likely to occur in the action area during the near future.

Integration and Synthesis

The EPA proposes to issue and implement the Vessel General Permit (VGP) and the Small Vessel General Permit (sVGP), which authorize discharges incidental to the normal operation of non-recreational and non-military vessels into waters of the U.S. The statutory authority for the proposed action is the National Pollution Discharge Elimination System (NPDES) of the Clean Water Act (33 USC 1342 et seq.; CWA). The VGP was first issued by EPA in 2008; the proposed permit, if issued, will become active in December 2013 and expire in December 2018. The sVGP is a new permit; if issued, it will become active in December 2013 and expire in December 2018.

The EPA expects that vessels greater than 79 feet in length or with 8 cubic meters or more of ballast water capacity will seek coverage under the VGP. Vessels less than 79 feet and with less than 8 cubic meters of ballast water capacity may be eligible for coverage under the sVGP, which has less stringent restrictions and fewer reporting requirements. The EPA estimates that approximately 72,000 vessels may be eligible for coverage under the VGP and approximately 138,000 vessels may be eligible for coverage under the sVGP.

The action will occur throughout the waters of the U.S., i.e., the action area. The proposed action is likely to adversely affect the following ESA-listed species: Cook Inlet beluga whale, southern resident killer whale, North Atlantic right whale, humpback whale, green sea turtle (all populations), hawksbill sea turtle, Kemp's ridley sea turtle, leatherback sea turtle, loggerhead sea turtle, olive ridley sea turtle, Gulf of Maine Atlantic salmon, bocaccio, canary rockfish, Pacific eulachon, yelloweye rockfish, Chinook salmon (all DPS), chum salmon (all DPS), coho salmon (all DPS), sockeye salmon (all DPS), steelhead (all DPS), Atlantic sturgeon (all DPS), green sturgeon, gulf sturgeon, shortnose sturgeon, smalltooth sawfish, black abalone, white abalone, elkhorn coral, staghorn coral, and Johnson's seagrass.

The proposed action is likely to adversely affect the following designated critical habitat: Cook Inlet beluga whale, southern resident killer whale, North Atlantic right whale, Gulf of Maine Atlantic salmon, Chinook salmon (all DPS), chum salmon (all DPS), coho salmon (all DPS), sockeye salmon (all DPS), steelhead (all DPS), southern green sturgeon, gulf sturgeon, smalltooth sawfish, black abalone, elkhorn coral, staghorn coral, and Johnson's seagrass.

In the Opinion, we described the status of each species and the environmental baseline. We also discussed the effects of the action, i.e., exposure, stressors, response, and risks. Here, we integrate this information to describe stressors of the greatest concern (i.e., ANS, copper, and selenium) and the likely consequences of exposing listed resources to these stressors.

Aquatic nuisance species (carried in ballast water, on hulls, and in chain lockers, seachests, and fish holds of vessels) may be discharged into waters of the U.S. While ANS are themselves stressors, the mechanisms by which they adversely affect listed resources include: predation, competition, trophic alteration, ecosystem alteration, disease transmission, and genetic introgression. Because of the magnitude of these mechanisms, ANS are likely to reduce the fitness (i.e., survival and/or reproduction) of exposed individuals.

Anadromous fishes are threatened by the four Hs: habitat loss, harvest, hydrology alterations, and for some species, hatcheries. These species are highly dependent upon waters of the U.S. and thus are sensitive to the introduction of pollutants and ANS. Invasive species are likely to prey on juveniles, compete for prey resources, alter essential habitat, and transmit infectious diseases. Scientists estimate that ANS invasions are as great a threat as the four Hs combined because ANS have the potential reduce the fitness of a large number of individuals through a multitude of mechanisms. Therefore, ANS invasions are likely to reduce population and species viability. Invasions are also likely to adversely affect the critical habitat of these species, which includes prey resources as a principle constituent element.

In addition, the discharges of chemical pollutants that would be authorized by the permit are known to adversely impact threatened and endangered species under NMFS' jurisdiction. Copper affects chemoreception of salmonids, reducing their ability to forage, migrate, avoid predators, and find mates. It also impairs their disease resistance, respiration, osmoregulation, blood chemistry, enzyme activity, and internal organ functions. Selenium reduces reproductive potential in anadromous fishes. Therefore, exposure to copper and/or selenium are likely to reduce population and species viability.

The southern resident killer whale and Cook Inlet beluga whale DPSs are isolated populations that rely on anadromous fish as their primary prey. Because of their small population sizes and specialized diets, these DPSs have little resilience to ecosystem perturbations, such as the introduction of pollutants or ANS. Invasive species that reduce salmon or Pacific eulachon abundance are likely to reduce the survival and reproduction of an individual. The reduction or loss of their primary forage base is likely to affect many individuals, reducing population and species viability. The North Atlantic right whale exhibits an extremely small population size and is also vulnerable to ecosystem perturbations, including ANS invasions that produce toxic algal blooms. Invasions would also adversely affect the critical habitat of these species, which include prey resources as a principle constituent element.

Threatened and endangered marine invertebrates have experienced dramatic declines as a result of ecosystem restructuring, overharvest, environmental changes, and disease. Their reproductive strategies (e.g., broadcast spawning) make them susceptible to detrimental Allee effects at small population sizes. They are especially vulnerable to the introduction ANS invasions which would lower the fitness of individuals through the introduction of novel pathogens and competition for space and prey. These mechanisms are likely to affect a large number of individuals through the spread of disease and niche replacement. Therefore, ANS invasions are likely to reduce population and species viability of listed invertebrates. Invasions are also likely to adversely affect the critical habitat of these species, which includes prey resources and suitable substrate as principle constituent elements. Copper has also been shown to reduce the survival and reproduction of mollusks such as the endangered black and white abalone species under NMFS' jurisdiction. Therefore, exposure to copper is likely to reduce population and species viability.

Listed plant species have declined primarily as a result of habitat modification and destruction and altered water quality. Their limited habitat range provides little resilience to ecosystem or environmental perturbations. Invasive species are likely to transmit infectious diseases and compete for resources (including space). Disease is likely to spread quickly through the small

population and a competitor could easily overgrow native plants with small habitat ranges. Therefore, ANS invasions are likely to reduce population and species viability of listed plants.

Our *Programmatic Summary* describes whether and to what degree EPA structured the VGPs to insure that vessel discharges into waters of the U.S. are not likely to jeopardize the continued existence of endangered species or threatened species under NMFS' jurisdiction or result in the destruction or adverse modification of critical habitat that has been designated for those species. We concluded:

1. EPA has structured the proposed VGPs so they will know or be able to reliably estimate the probable number of the discharges that they would be authorizing. The scope of their action includes issuance and implementation of the VGPs, including monitoring, compliance, and enforcement.

2. EPA will be in a position to reliably estimate the physical, chemical or biotic stressors that are likely to be produced as a direct or indirect result of the vessel discharges that would be authorized by the VGPs and would know or be able to reliably estimate whether those discharges have occurred in concentrations, frequencies, or for durations that violate the terms of the proposed VGPs. While there were gaps in EPA's original identification of stressors of the action, EPA will implement monitoring programs to fill these gaps in knowledge.

3. EPA has structured the VGPs so it will know or be able to reliably estimate whether or what degree specific endangered or threatened species or designated critical habitat are likely to be exposed to potentially harmful discharges that the proposed permit would authorize, and to the ecological consequences of these discharges. EPA will use monitoring programs to estimate exposure of ESA-listed resources to the stressors produced by the actions it authorizes by the permit throughout the five-year permit cycle.

4. EPA will adequately identify, collect, and analyze information about its authorized discharges that may expose listed resources to harmful stressors throughout the five-year permit cycle. For ANS and other pollutants, EPA will summarize data from Annual Reports (which include all instances of noncompliance and unavoidable discharges authorized by the VGP) and Discharge Monitoring Reports and provide this summary to the Services. In addition, EPA, in cooperation with NMFS, will develop and implement a monitoring and reporting plan specifically for ANS, copper, and selenium discharges.

5. EPA, with the technical assistance of NMFS, will gather data and identify metrics for assessing specific discharges and pollutants that could pose risks to listed species and undertake corrective actions as necessary. In doing so, EPA will employ an analytical methodology that considers (a) the status and trends of endangered or threatened species or designated critical habitat; (b) the demographic and ecological status of populations and individuals of those species given their exposure to pre-existing stressors in different drainages and watersheds; (c) the direct and indirect pathways by which endangered or threatened species or designated critical habitat might be exposed to the discharges to waters of the United States; and (d) the physical, physiological, behavior, sociobiological,

and ecological consequences of exposing endangered or threatened species or designated critical habitat to stressors from discharges at concentrations, intensities, durations, or frequencies that could produce physical, physiological, behavioral, or ecological responses, given their pre-existing demographic and ecological condition.

6. EPA has a mechanism to reliably determine whether and to what degree operators have complied with the conditions, restrictions, or mitigation measures required of the VGPs. EPA will develop and implement monitoring and compliance plans for the VGPs. They will gather, review, and analyze inspection data and provide a summary of these results to the Services annually. EPA will also provide training and resources for States and the US Coast Guard to facilitate their inspection and enforcement capabilities.

7. EPA has a mechanism to prevent or minimize listed resources' exposure to stressors in discharges if: (1) EPA finds that these stressors occur at concentrations, durations, or frequencies that are potentially harmful to individual listed organisms, populations, or species; or (2) EPA identifies that the discharges lead directly or indirectly to ecological consequences that are potentially harmful to individual listed organisms, populations, species or primary constituent elements of designated critical habitat. EPA has committed to modify its action as appropriate if new information (including inadequate protection for species or low levels of compliance) becomes available. Modifications may include additional actions or requirements, reopening of the permits, and reinitiation of Section 7 consultation.

This programmatic review describes EPA's ability to insure that vessel discharges into waters of the U.S. are not likely to jeopardize the continued existence of endangered species or threatened species or result in the destruction or adverse modification of critical habitat that has been designated for those species; however, we must address our final area of concern: exposure.

Given exposure to ANS invasions and pollutants, listed resources are likely to be adversely affected, resulting in reduced viability of populations and species. Therefore, it is imperative that EPA prevent the exposure of listed resources to vessel-mediated ANS invasions and pollutants. There is a small likelihood that listed resources would be exposed to ANS or harmful concentrations of pollutants. To reduce this likelihood of exposure even further, EPA will implement the following four-step feed-back mechanism: permit requirements, monitoring and collecting new information, additional actions to reduce any unacceptable risks, and reopening the permit and/or reinitiating consultation as appropriate to reassess permit requirements.

First, EPA includes several requirements in their VGPs to reduce the exposure of listed resources to ANS and pollutants, For ANS, EPA requires vessel owners/operators to "minimize the transport of attached living organisms" into waters of the U.S. For vessels with ballast water capacity greater than 8 cubic meters, EPA requires that discharges meet numeric ballast water standards. They require vessel owners/operators to minimize hull husbandry and avoid discharging ballast water in designated critical habitat, when feasible. For pollutants, ANS requires that vessel owners/operators meet technology-based and water-quality based effluent limits.

Second, EPA will monitor pollutants and ANS. To monitor pollutants in VGP discharges, EPA requires and reviews the annual analytical test results for treated ballast water, exhaust gas scrubber effluent, bilgewater (new builds), and graywater (cruise ships). For copper and selenium, EPA will develop and implement a monitoring and reporting plan within one year of the finalization of the VGPs. They will identify metrics for copper and selenium and compile data through NOIs and annual reports. They will review the data twice during the term of the permit to determine whether the metrics have been exceeded. For ANS invasions, they will annually track new vessel-mediated invasions using nationwide ANS databases; they will also review invasion rates as estimated by systematic surveys of ANS in San Francisco, Tampa, and Chesapeake Bays. To identify improvements in technology designed to minimize the transport of biofouling ANS, EPA will compile information on new developments in AFCs (to minimize exposure to hull-fouling ANS and copper loading) and debris containment systems (to minimize the release of ANS and biocides during underwater husbandry). They will identify whether there are more efficacious BMPs or technologies for underwater husbandry and for anti-fouling coatings with fewer secondary environmental impacts that could be required of vessels.

Third, EPA will, if necessary, identify and implement additional actions to reduce any unacceptable risks associated with vessel mediated ANS or pollutants. For example, EPA will require additional requirements to minimize the transport of ANS (e.g., anti-fouling coatings, debris containment systems, or more stringent ballast water standards) as soon as practicable. These additional actions would be designed to further reduce exposure of listed resources to ANS and pollutants.

Fourth, if appropriate, EPA will reopen and modify their permits if they receive new information (including that discussed above) that was not available at the time of permit issuance and would have justified the application of different permit conditions at the time of permit issuance. This new information may include data on ANS invasion rates, available technologies to minimize ANS invasions, discharge concentrations or volumes, pollutant risks, compliance or lack thereof, and adverse affects to listed resources. This information would also allow EPA to determine whether to request reinititation of formal ESA Section 7 consultation as provided in 50 CFR 402.16 and described in the Fact Sheets. Either of these actions (reopening the permit or requesting reinitiation of Section 7 consultation) under the fourth step provides a mechanism for EPA to revise the permit requirements, thus restarting the feedback loop at step one to prevent exposure of listed resources to ANS and pollutants.

To assess and encourage compliance, EPA authorizes and/or conducts vessel inspections. In this endeavor, EPA has created a Memmorandum of Understanding[15] (MoU) with the U.S. Coast Guard, which potentially inspects all vessels covered under the VGP on an annual basis. Prior to its effective date (December 2013), EPA will create and implement a plan to inspect (or authorize inspection of) a subset of vessels covered under the sVGP. The EPA will review annual reports and inspection data to insure that vessel owners/operators are complying with the requirements of the permits. Lower than anticipated compliance levels would constitute new

[15] See: http://www.epa.gov/compliance/resources/agreements/cwa/mou-coastguard-vesselpermitrequirements.pdf

information, as described in the fourth step, and provide a mechanism to reopen/modify the permit or request reinitiation of Section 7 consultation. In this manner, EPA has the ability to increase compliance through additional requirements to their permits or through additional actions (steps one and three of the feedback mechanism).

This four-step feedback mechanism provides continual monitoring of potential exposure of listed resources to potentially harmful stressors. If metrics indicate the possibility of exposure to potentially harmful levels of ANS and pollutants, EPA can implement measures to reduce these threats before individuals are exposed. This iterative process first minimizes the likelihood of exposure; if there is exposure, it minimizes the number of exposed individuals exposed; and if individuals are exposed, it minimizes the likelihood that an entire population or species will be affected. In this manner, EPA has insured that their action is not likely to jeopardize the continued existence of any listed species and is not likely to adversely modify or destroy designated critical habitat.

Conclusion

After reviewing the current status of the listed species, the environmental baseline for the action area, the potential direct and indirect effects of EPA's proposal to issue and implement the VGPs which would authorize discharges incidental to the normal operation of vessels into waters of the United States, an examination of the controls EPA proposes to implement to mitigate these effects, and cumulative effects, it is our biological and conference opinion that EPA has insured that its action is not likely to jeopardize any listed or proposed species under NMFS' jurisdiction.

After reviewing the current status of the critical habitat that has been designated for endangered and threatened species, the environmental baseline of the action area, the potential direct and indirect effects of the action, an examination of the controls EPA proposes to implement to mitigate these effects, and cumulative effects, it is our biological opinion that EPA has insured that its action is not likely to destroy or adversely modify designated critical habitat.

Incidental Take Statement

Section 9 of the ESA and Federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without 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 NMFS 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. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of Section 7(b)(4) and Section 7(o)(2), taking that

is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the Act provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

Amount or Extent of Take

This programmatic consultation focuses on whether the EPA has insured that their issuance of the general permit is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. It does not address the specific individual actions that the general permit would authorize (i.e., a particular vessel, in a single location, discharging specific quantities on a specified date). The controls and measures in the general permit are designed to reduce or in most cases prevent the exposure of endangered or threatened species under NMFS' jurisdiction to adverse impacts as a result of activities authorized by the general permit. However, it is possible that such exposures may still take place as a result of those activities, and it is possible that these exposures may cause incidental take. As a result, incidental take of endangered or threatened species is possible over the duration of the proposed general permit.

Because of the large scale and broad scope of the proposed action, even the best scientific and commercial data available are not sufficient to enable NMFS to estimate the specific amount of potential incidental take associated with the action. Therefore, NMFS identifies, as a surrogate for the allowable extent of take, the ability of this action to proceed without any adverse incident to any fish, whale, abalone, coral or marine plant of any species, which is attributed to any discharge in accordance with the general permit in the range of listed endangered or threatened species under NMFS' jurisdiction.

An adverse incident is considered attributable to a discharge in accordance with the general permit if a discharge of a pollutant authorized by the general permit is known to, or suspected to have occurred prior to, and near or upstream from the adverse incident, and if the suspected pollutant is detected in local water samples or in tissue samples of affected fish, or –in the case of aquatic nuisance species– that adverse incident is directly attributable to the introduction of any aquatic nuisance species known or suspected to have occurred from discharges authorized by the general permit.

Reasonable and Prudent Measures

The measures to avoid or minimize take described below are non-discretionary and must be undertaken by the EPA so that they become a binding condition of any applicant, as appropriate, for the exemption in section 7(o)(2) to apply. The EPA has a continuing duty to regulate the activity covered by this incidental take statement. The protective coverage of section 7(o)(2) may lapse if the EPA: (1) Fails to assume and implement the terms and conditions; or (2) Fails to require any applicant to adhere to the terms and conditions of the incidental take statement through enforceable terms that are added to the general permit. In order to monitor the impact of incidental take, the EPA must report the progress of the action and its impact on the species to NMFS OPR as specified in the incidental take statement (50 CFR§402.14(i)(3)). The reporting requirements are established in accordance with 50 CFR 220.45 and 228.5.

To satisfy its obligations pursuant to section the ESA as described above, the EPA must gather information on the direct, indirect, and cumulative impacts of the activities authorized by the issuance of the general permit; and evaluate the direct, indirect, or cumulative impacts of the activities authorized by the issuance of the general permit and the consequences of those effects on endangered and threatened species under NMFS' jurisdiction.

The purpose of the monitoring is to provide data for EPA to use to identify necessary modifications to the general permit in order to reduce exposures to endangered and threatened species under NMFS' jurisdiction. NMFS believes all measures described as part of the proposed action, together with the Reasonable and Prudent Measures described below, are necessary and appropriate to minimize the likelihood of incidental take of ESA listed species due to implementation of the proposed action.

The EPA shall:

1. Gather information on any incidental take or surrogate measure of take that occurs from the action; and
2. Report to NMFS OPR as soon as practicable if this information indicates that such take has occurred.

Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, the EPA must comply with the following condition. This condition implements the reasonable and prudent measures described above. This condition is non-discretionary.

The EPA shall include the following instructions requiring reporting of adverse incidents to listed species in the general permit fact sheet or online:
"NOTICE: Incidents where any fish, whale, abalone, coral or marine plant appear injured or killed as a result of discharges into Waters of the United States from discharges as authorized by this permit in the range of endangered or threatened species under the jurisdiction of the National Marine Fisheries Service shall be reported to the National Marine Fisheries Service, Office of Protected Resources at (301) 713.1401 and EPA at (202) 564.0748. The finder should leave the plant or animal alone, make note of any circumstances likely causing the death or injury, note the location and number of individuals involved and, if possible, take photographs. Adult animals should not be disturbed unless circumstances arise where it is obviously injured or killed by discharge exposure, or some unnatural cause. The finder may be asked to carry out instructions provided by NMFS OPR to collect specimens or take other measures to ensure that evidence intrinsic to the specimen is preserved."

Conservation Recommendations

Section 7(a)(1) of the Act directs Federal agencies to utilize their authorities to further the purposes of the Act by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to

minimize or avoid adverse effects of a proposed action on ESA-listed species or critical habitat, to help implement recovery plans, or to develop information.

The following conservation recommendations would provide information for future consultation involving EPA's issuance and implementation of VGPs:

1. Review more stringent ballast water, hull fouling, and hull husbandry requirements of States and other nations to evaluate their practicability for incorporation in future permits.

2. Begin information ESA Section 7 consultation ("preconsultation") well in advance of the publishing the draft permits in the Federal Register for public comment. This will allow EPA to incorporate recommended actions designed to protect listed resources into its permits at an early stage and receive public comment on these actions.

In order to keep NMFS' Endangered Species Division informed of actions minimizing or avoiding adverse effects, or benefiting ESA-listed species or their habitats, the U.S. Environmental Protection Agency should notify the Endangered Species Division of any conservation recommendations they implement in their final action.

Reinitiation Notice

This concludes formal consultation on the U.S. Environmental Protection Agency's issuance of the Vessel General Permit and the Small Vessel General Permit. As provided in 50 CFR 402.16, 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) New information reveals effects of the agency action that may affect endangered or threatened species under NMFS' jurisdiction or to designated critical habitat in a manner or to an extent not considered in this Opinion (e.g. if EPA were to fail to execute the monitoring and reporting or other commitments it made during ESA Section 7 consultation which are cited in this Biological Opinion as partial justification for NMFS' determination that the proposed action would not jeopardize any ESA-listed endangered or threatened species under NMFS' jurisdiction or destroy or adversely modify any critical habitat that has been designated for those species); (2) The agency action is subsequently modified in a manner that causes an effect to the ESA-listed species or critical habitat not considered in this Opinion; (3) A new species is listed or critical habitat designated that may be affected by the action; or (4) The amount or extent of take specified in the incidental take statement is exceeded. In instances where the amount or extent of take specified in the incidental take statement is exceeded, the U.S. Environmental Protection Agency must immediately request reinitiation of Section 7 consultation.

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