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

Title: Programmatic Biological and Conference Opinion on the Towing of Inactive U.S. Navy Ships from their Existing Berths to Dismantling Facilities or other Inactive Ship Sites

Consultation Conducted By: Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce

Action Agency: U.S. Navy

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1 INTRODUCTION

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

The Federal action agency shall confer with the NMFS for species under NMFS jurisdiction on any action which is likely to jeopardize the continued existence of any proposed species or result in the destruction or adverse modification of proposed critical habitat (50 C.F.R. §402.10). If requested by the Federal agency and deemed appropriate, the conference may be conducted in accordance with the procedures for formal consultation in §402.14.

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

The action agency for this consultation is the U.S. Navy Inactive Ships Office (hereafter referred to as "Navy"). The Navy proposes to tow inactive Navy ships from their existing berths to dismantling facilities, between inactive ship facilities, and from active sites to inactive sites.

This consultation, biological opinion (opinion), and ITS, were completed in accordance with section 7(a)(2) of the statute (16 U.S.C. 1536 (a)(2)), associated implementing regulations (50 C.F.R. §§401-16), and agency policy and guidance was conducted by NMFS Office of Protected Resources ESA Interagency Cooperation Division (hereafter referred to as "we"). This opinion and the ITS were prepared by NMFS Office of Protected Resources ESA Interagency Cooperation Division in accordance with section 7(b) of the ESA and implementing regulations at 50 C.F.R. §402.

This document represents NMFS opinion on the effects of these actions on ESA-listed species and designated critical habitat. A complete record of this consultation is on file at the NMFS Office of Protected Resources in Silver Spring, Maryland.

1.1 Background

The Navy Inactive Ships Office first requested consultation under section 7 of the ESA for the towing of inactive Navy ships in August 2012. Consultation was initially requested to address concerns regarding the potential for towed inactive ships to strike ESA-listed species while in transit. Between 2012 and 2016, NMFS completed five informal consultations on Navy ship tow events. Each of these consultations concluded that the Navy's proposed action to tow inactive ships was not likely to adversely affect ESA-listed species. These consultations also concluded that these proposed actions would not destroy or adversely modify designated critical habitat.

Starting in 2014, NMFS expressed concern to the Navy regarding the transport and establishment of potentially invasive species that may be attached to the underwater components of decommissioned ships. NMFS recommended the Navy make efforts to minimize the risk associated with potentially transferring invasive species by cleaning the underwater components of decommissioned ships prior to towing each ship. For the latest informal consultation, completed in August 2016, the Navy complied with NMFS' request to clean the hull of the ex-Independence prior to towing the ship from Bremerton, Washington to Brownsville, Texas.

This programmatic opinion establishes a set of project design criteria to be implemented by the Navy to minimize potential adverse effects to ESA-listed resources, and streamlines the environmental compliance process for the towing of inactive Navy ships.

1.2 Consultation History

- On June 5, 2017, the Navy requested initiation of formal consultation on their proposed action to tow inactive Navy ships from their existing berths to dismantling facilities, between inactive ship facilities, and from active sites to inactive sites. The initiation request was accompanied by a biological evaluation (BE) which considered the effects of the proposed action on ESA-listed species and designated critical habitat (NUWC 2017a).
- On July 10, 2017, the Navy and NMFS held an in-person meeting in Silver Spring, Maryland to discuss the Navy's proposed action and to explore possible mitigation measures that may be implemented in order to minimize adverse effects to ESA-listed resources.
- On July 18, 2017, NMFS responded to the Navy's June 2017 request to initiate formal consultation and stated that the BE did not provide sufficient information to begin formal consultation, as outlined in the regulations governing interagency consultation (50 CFR 402.14). In this response, NMFS requested additional information from the Navy

including how the Navy will minimize the risk of transferring invasive species from ship berthing locations to inactive ports or ports where dismantling facilities are located.

- Between July 2017 and January 2018, the Navy and NMFS communicated via email and telephone to clarify the proposed action. Below we include a summary of major decisions points throughout this period.
    - The Navy's June, 2017 BE stated that under the proposed action, inactive Navy ships could be towed to any U.S. port. On August 7, 2017, the Navy changed the scope of the proposed action such that ships could only be towed from the seven origination ports listed in the BE and that ships could only be towed to the seven destination ports listed in the BE.
    - The Navy's June 2017 BE did not include mitigation measures (e.g., in-water hull cleaning or dry-docking) as part of the proposed action. Between August 7 and August 31, 2017, the Navy provided written responses to NMFS' comments, including a decision-making framework and risk matrix for determining when mitigation would be necessary to minimize the risk of nonnative species transfer by the towing of inactive ships to sensitive ocean waters and destination ports.
    - On November 17, 2017, NMFS emailed the Navy requesting additional information including a proposed plan for monitoring the potential effects of the proposed action on ESA-listed species and designated critical habitat and a strategy for minimizing impacts to water quality during in-water hull cleaning, should it occur.
    - On January 19, 2018, the Navy sent NMFS a letter responding to the request for additional information. The Navy response included a plan for conducting biofouling surveys and water quality monitoring under specified scenarios and additional information regarding the effects of in-water hull cleaning on ESA-listed species and designated critical habitat.
- On February 1, 2018, NMFS initiated formal section 7 consultation with the Navy. NMFS transmitted a letter dated February 2, 2018 to the Navy to officially inform them of consultation initiation.
- On March 3, 2018, the Navy provided NMFS with an updated risk matrix with the following changes to the proposed action: (1) San Diego was removed as an origin port, and (2) the water type for Mayport, Florida was changed from saltwater to brackish.
- On June 4, 2018, NMFS provided the Navy with a draft opinion for review.
- On June 17, 2018, NMFS held a conference call with representatives of the Suquamish Tribe to respond to their inquiries regarding the inactive ship tow consultation with the Navy in furtherance of its responsibilities to engage the Tribe in government-to-government consultation.

- On June 25, 2018, the Navy returned the draft opinion to NMFS with comments. As part of their review, the Navy updated a table in the draft opinion with estimated towing frequencies for each of the potential port-to-port combinations. The Navy also indicated via email that they were adding San Diego as an origination port for the proposed action.
- On June 28, 2018, the Navy and NMFS met in Silver Spring, Maryland to discuss the draft opinion. Much of the discussion revolved around conservation measures for minimizing the risk of invasive species introductions as a result of the proposed action. NMFS interpreted the mitigation measures discussed with the Navy during pre-consultation as being "conservation measures" that the Navy had agreed to as part of the consultation process. This interpretation was largely based a document the Navy sent to NMFS on August 30, 2017 (and subsequently updated on March 3, 2018) titled "INACTSHIPS Programmatic ESA Threat Risk Matrix." This document identified tow route scenarios with an elevated risk of invasive species effects for which mitigation could be warranted (i.e., scenarios where both ports [origin and destination] have similar salinities and there are ESA-listed species at the destination port). NMFS' approach to the effects analysis in the draft opinion was to consider such mitigation measures, which were derived during the course of consultation, as being part of the Navy's action. NMFS revised the scenarios for which mitigation would be conducted based on more accurate information on ESA-listed species presence at destination ports and updated port salinity information. The Navy clarified that they did not intend for mitigation measures discussed during pre-consultation to be considered part of the proposed action for this consultation.
- On June 28, 2018, a member of the Suquamish Tribe emailed NMFS with supporting literature they wanted NMFS to consider for the section 7 inactive ship tow consultation with the Navy.
- On July 18, 2018, NMFS met in person with representatives of the Suquamish Tribe in response to their request for government-to-government consultation on the inactive Navy ship tow section 7 consultation.
- On August 2, 2018, the Navy and NMFS met in-person to discuss the draft opinion. The purpose of this meeting was to discuss outstanding technical issues regarding the occurrence of ESA-listed species and salinities at Navy ports, and risk factors that should be integrated into the invasive species effects analysis.
- From August 6 through August 30, 2018, the Navy sent NMFS supplemental information to be incorporated into the final opinion. This included additional salinity data for the port in Beaumont, Texas, a revised estimated towing frequency table, a description of the Navy's in-water hull cleaning process, and additional information on the frequency of ships being towed through the Panama Canal.

- On August 24, 2018, the NMFS West Coast Regional Office, Protected Resources Division provided comments on the draft opinion.
- On September 7, 2018, the Navy sent NMFS an updated invasive species risk matrix, which documented the decision to implement the mitigation measures derived through this consultation process. The Navy also sent NMFS a description of the Inactive Ships' process for determining dry dock availability.
- From October 15 to October 18, 2018, the Navy provided additional information, as requested by NMFS, regarding seasonal work windows for in-water hull cleaning, the Navy's use of ship hull antifouling paints containing tributyltin (TBT), and the estimated time lapse between completion of hull fouling mitigation and start of the ship tow.
- On October 19, 2018 NMFS emailed the Navy a proposed revised timeline for final review of the draft opinion and completion of the consultation.
- On November 7, 2018 NMFS sent the Navy a revised draft opinion for their review. As agreed to by both agencies, the Navy's review of the revised draft would be limited in scope to the following sections: Section 3 - Description of Proposed Action; Section 4 - Mitigation Measures Agreed to by Navy during the Course of Section 7 Consultation with NMFS; Section 14 Incidental Take Statement; and a review of the remainder of the opinion for accuracy of descriptions of the Navy's proposed action only.
- On December 13, 2018 the Navy provided NMFS with comments on the revised draft opinion. The Navy requested a follow-up meeting with NMFS to discuss some of their comments.
- Starting on December 26, 2018, the consultation was held in abeyance for 38 days due to a lapse in appropriations and resulting partial government shutdown. The consultation resumed on January 28, 2019.
- On February 4, 2019, the Navy and NMFS met via conference call to discuss the Navy's review of the revised draft opinion.

2 THE ASSESSMENT FRAMEWORK

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

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

"*Destruction or adverse modification*" means a direct or indirect alteration that appreciably diminishes the value of designated critical habitat for the conservation of an ESA-listed species. Such alterations may include, but are not limited to, those that alter the physical or biological features (PBFs) essential to the conservation of a species or that preclude or significantly delay development of such features (50 C.F.R. §402.02). An ESA section 7 assessment involves the following steps:

1) We identify the proposed action and those aspects (or stressors) of the proposed action that are likely to have direct or indirect effects on the physical, chemical, and biotic environment within the action area, including the spatial and temporal extent of those stressors.

2) We identify the ESA-listed species and designated critical habitat that are likely to co-occur with those stressors in space and time.

3) We describe the environmental baseline in the action area including: past and present impacts of Federal, state, or private actions and other human activities in the action area; anticipated impacts of proposed Federal projects that have already undergone formal or early section 7 consultation, impacts of state or private actions that are contemporaneous with the consultation in process.

4) We identify the number, age (or life stage), and gender of ESA-listed individuals that are likely to be exposed to the stressors and the populations or subpopulations to which those individuals belong. We also consider whether the action "may affect" designated critical habitat. This is our exposure analysis.

5) We evaluate the available evidence to determine how individuals of those ESA-listed species are likely to respond given their probable exposure. We also consider how the action may affect designated critical habitat. This is our response analyses.

6) We assess the consequences of these responses of individuals that are likely to be exposed to the populations those individuals represent, and the species those populations comprise. This is our risk analysis.

7) The adverse modification analysis considers the impacts of the proposed action on the essential habitat features and conservation value of designated critical habitat.
8) We describe any cumulative effects of the proposed action in the action area.
9) We integrate and synthesize the above factors by considering the effects of the action to the environmental baseline and the cumulative effects to determine whether the action could reasonably be expected to:
   a) Reduce appreciably the likelihood of both survival and recovery of the ESA-listed species in the wild by reducing its numbers, reproduction, or distribution; or
   b) Diminish appreciably the conservation value of designated or proposed critical habitat.
10) We state our conclusions regarding jeopardy and the destruction or adverse modification of designated critical habitat.

If, in completing the last step in the analysis, we determine that the action under consultation is likely to jeopardize the continued existence of ESA-listed species or destroy or adversely modify designated critical habitat, we must identify a reasonable and prudent alternative to the action. The reasonable and prudent alternative must not be likely to jeopardize the continued existence of ESA-listed species nor adversely modify their designated critical habitat and it must meet other regulatory requirements.

In addition, we include an ITS that specifies the impact of the take, reasonable and prudent measures to minimize the impact of the take, and terms and conditions to implement the reasonable and prudent measures. ESA section 7 (b)(4); 50 C.F.R. §402.14(i). We also provide discretionary conservation recommendations that may be implemented by the action agency. 50 C.F.R. §402.14(j). Finally, we identify the circumstances which would require reinitiation of consultation. 50 C.F.R. §402.16.

"Take" is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by regulation to include significant habitat modification or degradation that results in death or injury to ESA-listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. NMFS has not yet defined "harass" under the ESA in regulation. However, on December 21, 2016, NMFS issued interim guidance on the term "harass," defining it as an action that "creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering" (NMFS 2016c). For purposes of this consultation, we relied on NMFS' interim definition of harassment to evaluate when the proposed activities are likely to harass ESA-listed species. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity.

To comply with our obligation to use the best scientific and commercial data available, we conducted electronic literature searches throughout the consultation, including within NMFS Office of Protected Resource's electronic library. We examined the literature that was cited in the submittal documents and any articles we collected through our electronic searches. We also considered the documents provided to NMFS by the Navy, including their BE (NUWC 2017a) and supplemental information provided during the course of consultation. These resources were used to identify information relevant to the potential stressors to and responses of ESA-listed species and designated critical habitat under NMFS' jurisdiction that may be affected by the proposed action to draw conclusions on risks the action may pose to the continued existence of these species and the value of designated critical habitat for the conservation of ESA-listed species.

Programmatic Analysis

The section 7 regulatory definition of federal action includes federal agency programs (see 50 CFR §402.02). Such programs may include a collection of activities of a similar nature, a group of different actions proposed within a specified geographic area, or an action adopting a framework or policies for the development of future actions. NMFS uses programmatic consultations to evaluate the expected effects of groups of related agency actions expected to be implemented in the future.

The following elements were included in our programmatic consultation to ensure its consistency with ESA section 7 and its implementing regulations:

- Project design criteria (PDC) or standards that will be applicable to future projects implemented under the consultation document. The PDC serve to prevent adverse effects to listed species, or to limit adverse effects to predictable levels.
- A description of the manner in which projects to be implemented under the programmatic consultation may affect ESA-listed species and critical habitat and evaluation of expected effects from covered projects.
- A process for evaluating expected, and tracking actual aggregate or net additive effects of all projects expected to be implemented under the programmatic consultation. This programmatic opinion demonstrates that when the PDC are applied to each project, the aggregate effect of all projects will either not adversely affect ESA-listed species or their critical habitat, or will not jeopardize ESA-listed species or destroy or adversely modify their critical habitat, as applicable.
- The project-specific procedures provide contingencies for proposed projects that cannot be implemented in accordance with the PDC or if proposed projects are too dissimilar in nature or in expected effects from those projected in the programmatic consultation document.
- Procedures for monitoring projects and validating effects predictions; and

- A comprehensive review of the program.

This programmatic opinion includes, where sufficient information is available, an analysis of the likely effects of future actions implemented by the Navy Inactive Ships Office on ESA-listed resources. For these actions, this opinion identifies PDC and includes an ITS where appropriate (i.e., if take is anticipated). Actions implemented by the Navy Inactive Ships Office under the Inactive Ship Program that were not evaluated in this opinion, or actions too dissimilar in nature or effect, may require subsequent section 7 consultation if the action is likely to affect ESA-listed species or designated critical habitat.

3 Description of the Proposed Action

"Action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by federal agencies. The proposed action for this consultation is the towing of inactive Navy ships from their existing berths to dismantling facilities, between inactive ship facilities, and from active sites to inactive sites. The Navy awards contracts for dismantlement of its inactive ships to private companies. Control of the ship is turned over to the contractor when they commence tow with their tug. The Navy maintains title until the ship is completely dismantled, but the contractor controls the ship. Although the Navy does not consider dismantling as part of their proposed action, this activity is interrelated to the proposed action and, therefore, was evaluated during this consultation (see Section 5 below).

The potential origination and destination ports for inactive ships are listed in Table 1. In recent history, most ships appropriated for dismantling have been towed to dismantling facilities in Brownsville, Texas.

Table 1. Origination and destination ports for the towing of inactive Navy ships.

Origination Ports	Destination Ports
Beaumont, Texas	Baltimore, Maryland
Mayport, Florida	Beaumont, Texas
Norfolk, Virginia	Brownsville, Texas
Pearl Harbor, Hawaii	New Orleans, Louisiana
Philadelphia, Pennsylvania	Pearl Harbor, Hawaii
Puget Sound, Washington	Philadelphia, Pennsylvania
San Diego, California	Puget Sound, Washington

Inactive Ships Program Office near-term requirements (across the Future Years Defense Program) related to the towing and/or dismantling of inactive ships are detailed in Appendix 6 of Office of the Chief of Naval Operations (CNO) Reports to Congress on the Annual Long-Range Plan for Construction of Naval Vessels (OCNO 2016), submitted per Section 231 of Title 10, United States Code, and in compliance with the Senate Armed Service Committee request for additional information regarding decommissioning and disposal of naval vessels. These requirements are subject to change in future CNO Reports to Congress. The Navy has approximately 50 ships in the inactive fleet. There are currently seven inactive ships located in Puget Sound, Washington, 13 ships located in Pearl Harbor, Hawaii, and 31 ships located in Philadelphia, Pennsylvania. The ships range in size from ocean tugs approximately 200 feet in length to aircraft carriers over 1,000 feet in length. On average, the Navy estimates that nine inactive ship tows would occur annually and of those tows, approximately four would be towed for dismantling. The anticipated average number of ship tows for each origination/destination port combination over a five-year period are shown in Table 2.

Inactive ships are towed by tugs, with tow cables ranging in length between 1,500 and 2,500 feet, depending on the size of the towed ship. Tow cables generally consist of either 1.75 or 2.25 inch diameter wire rope, with the catenary of the tow cable reaching up to 100 feet below the surface of the water. Tugs maintain up to 75 tons of strain on the cable during towing. Once in the open ocean, the tug and tow typically travel at speeds less than 10 knots. Tow operators will ensure that every ship maintain a lookout to minimize the risk of collision.

Table 2. The anticipated average number of ship tows for each origination/destination port combination over a five-year period.

Origin Port	**Destination Port**						
	Baltimore	Beaumont	Brownsville	New Orleans	Pearl Harbor	Philadelphia	Puget Sound
Beaumont	0		1	0	0	0	0
Mayport	0	0	0	0	0	1	0
Norfolk	0	1	0	0	0	8	0
San Diego	0	0	0	0	7	0	2
Pearl Harbor	0	0	4	1		0	0
Philadelphia	1	0	9	5	0		0
Puget Sound	0	0	4	1	0	0	

Note: Grey shading indicates that any ship towed to one of these three ports will be dismantled. A "0" indicates that the towing of an inactive ship between these ports would be a very rare event, and would likely not exceed once every 25 years, on average.

The tow route depends on the origination and destination port and the size of the inactive ship. Ships traveling from the Pacific Ocean basin to the Atlantic Ocean basin, or vice versa, would travel either through the Panama Canal, or around Cape Horn through the Strait of Magellan depending on ship size (e.g., aircraft carriers are too large to travel through the Panama Canal). The Navy anticipates that about 90 percent of the inactive ships towed from the Pacific Ocean basin to the Atlantic Ocean basin, or vice versa, would travel through the Panama Canal (R. Johnson, Navy, pers. comm. to R. Salz, NMFS, August 28, 2018).

Figure 1 shows typical tow routes for ships destined for Brownsville, Texas that are small enough to travel through the Panama Canal; Figure 2 shows a typical tow route for a ship traveling to Brownsville around Cape Horn.

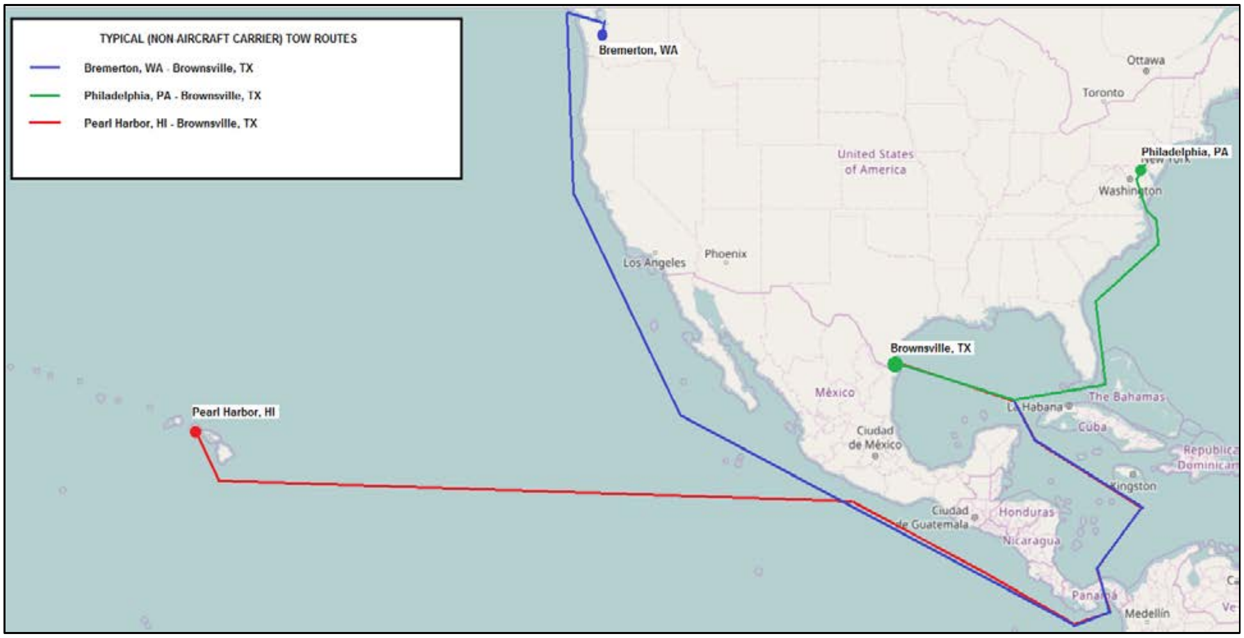


Figure 1. Typical tow routes for ships traveling to Brownsville, Texas for dismantling that are small enough to travel through the Panama Canal.

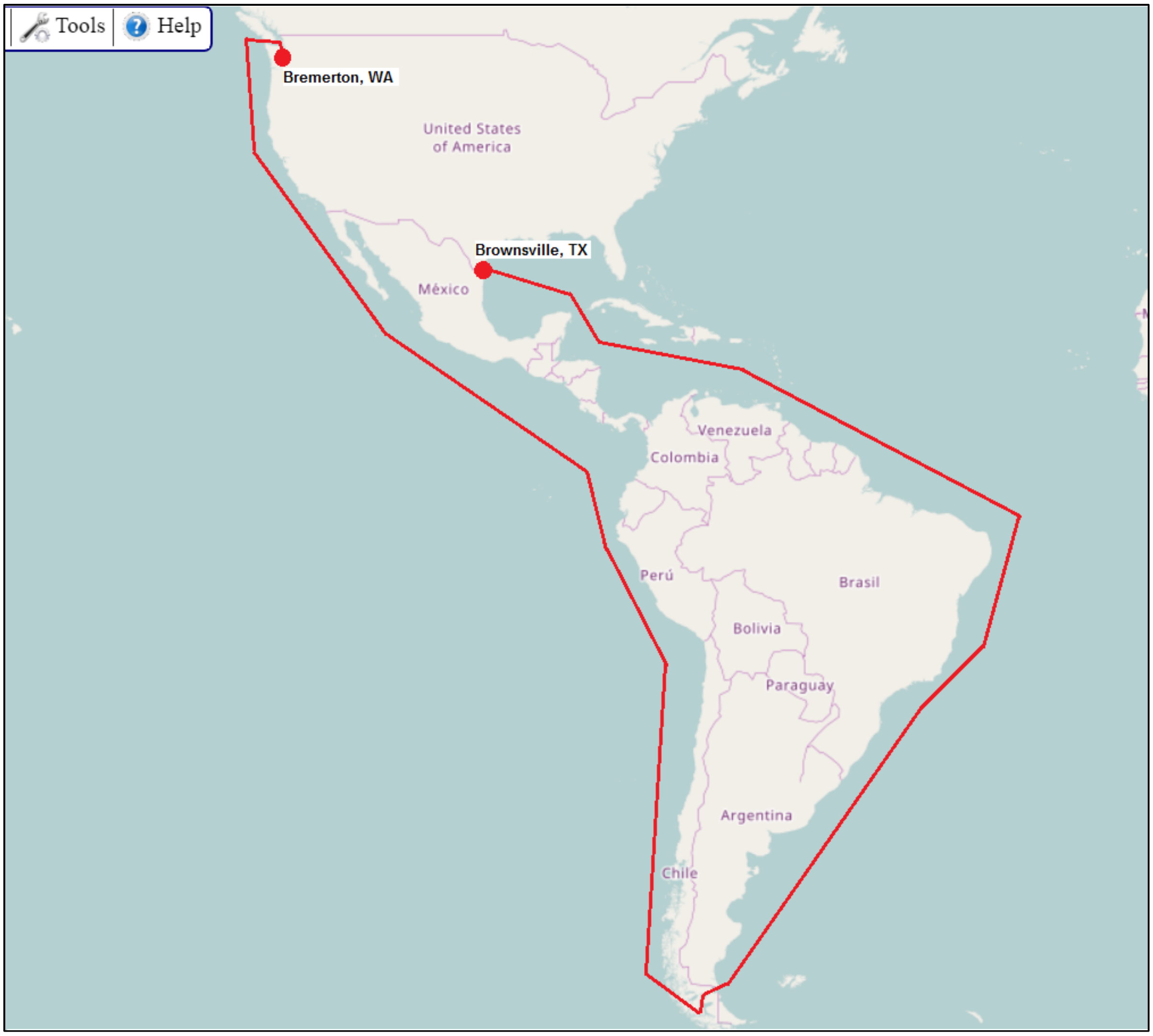


Figure 2. Typical tow route for aircraft carriers and ships of similar size traveling from Puget Sound, Washington to Brownsville, Texas for dismantling.

The towing of inactive ships may occur at any time of the year. To take advantage of calmer waters in the vicinity of Cape Horn during the Southern Hemisphere summer (Northern Hemisphere winter), aircraft carriers and ships of similar size from the U.S. West Coast that are transferred to Gulf Coast ship dismantling yards are typically towed during the Northern Hemisphere winter. The Navy's estimated ship tow durations for each origination and destination port combination are shown in Figure 3.

Ballast water is required for towing inactive ships for list, trim, and stability purposes. When inactive ships are towed for transfer purposes to ports consisting of inactive fleet sites, ballast water will not be removed when the ship arrives at the destination port. Prior to towing the ship in the future, ballast water may be added, transferred between tanks, or removed. If any ballast water is removed, it will be sent offsite for treatment and disposal in accordance with all federal, state, and local laws and regulations. For inactive ships being towed for dismantlement, ballast water will be removed as part of the dismantlement process and is under the cognizance of the Navy contractor. As part of the dismantling process, the contractor will pump the ballast water ashore and sends it for treatment and disposal in accordance with all federal, state, and local laws and regulations. The privately owned and operated tug boat will operate under all U.S. Coast Guard laws and regulations. When a ship is designated to be inactivated, the Navy identifies material, equipment, and supplies that can cost-effectively be removed and restored for use on other active Navy ships. These materials are removed prior to towing the ship or while the vessel is maintained in the inactive fleet.

Inactive Ship Towing Durations*

* All durations are estimates and can greatly vary based on time of year, weather during tow, size of towing vessel, etc…

Origination Port	Destination Port						
	Baltimore	Beaumont	Brownsville	New Orleans	Pearl Harbor	Philadelphia	Puget Sound
Beaumont	12-20 days	N/A	1-3 days	2-4 days	10-14 weeks	12-20 days	6-8 weeks
Mayport	10-14 days	2-5 days	2-4 days	1-3 days	10-14 weeks	10-14 days	6-8 weeks
Norfolk [2]	1-3 days	10-20 days	10-20 days	8-16 days	12-18 weeks 8-10 weeks	2-4 days	10-14 weeks 6-8 weeks
Pearl Harbor (Strait of Magellan)	12-16 weeks	10-14 weeks	10-14 weeks	10-14 weeks	N/A	12-18 weeks	1-2 weeks
Pearl Harbor (Panama Canal)	8-10 weeks	6-8 weeks	6-8 weeks	6-8 weeks	N/A	8-10 weeks	N/A
Philadelphia [2]	1-3 days	15-25 days	15-25 days[1]	14-24 days[1]	10-16 weeks 6-8 weeks	N/A	12-18 weeks 8-10 weeks
Puget Sound (Strait of Magellan)	12-18 weeks	10-14 weeks	10-14 weeks[1]	10-14 weeks	1-2 weeks	12-18 weeks	N/A
Puget Sound (Panama Canal)	8-10 weeks	6-8 weeks	6-8 weeks[1]	6-8 weeks	N/A	8-10 weeks	N/A

1 Tow duration based on historical data

2 Where two tow durations are shown the top duration is for vessels transiting around South America, and bottom duration is for vessels transiting through Panama Canal

Figure 3. Estimated inactive ship towing durations as provided by the Navy (NUWC 2017a).

"Dismantling" is the breaking down of a ship's structure into smaller pieces for recycling or disposal. The two methods of dismantling a ship are the afloat method and the dry-dock method. In the dry-dock method, the ship is docked in either a floating dry dock or graving dock that is de-flooded. Most ship dismantling is performed afloat, in slips, which are dredged openings in the bank of the ship channel. Slips are generally 400 to 1,100 feet long and 100 to 120 feet wide at the entrance. A large winch at the head of the slip is used to drag the hull farther into the slip as work progresses. Booms are placed around the ship as a precaution to contain any debris. Dismantling consists of harvesting any components that retain an above-scrap market value for resale or reuse, followed by physically cutting the remainder of the hull into smaller segments for scrap. All scrap segments are removed from the ship and taken onto land for further processing. At no time does the scrap material enter the water in the slip. Once a piece is removed from the ship, it is sized for the mill and demilitarized (i.e., combat systems are destroyed to their basic material content).

The Navy awards contracts for the towing and dismantling of conventionally powered inactive Navy ships so these activities are conducted by private contractors. The contractor is responsible for choosing the tow company and providing details of the planned tow in accordance with Navy towing requirements for approval, including tug(s) and tow routes. The tow date (usually within three months after award) and length of time required for transit depends upon several factors, such as the extent of shipboard preparations required to prepare the ship for tow, and the distance of the tow. When the contractor removes the ship from the pier for tow to their dismantling facility, the Navy transfers custody and control of the ship to the contractor for purposes of performing the contract requirements.

4 Mitigation Measures Agreed to by the Navy during the Course of Section 7 Consultation with NMFS

The Navy's proposed action, as described in their BE, is summarized above (NUWC 2017a). As discussed in greater detail in Section 10.1.3 of this opinion, the Navy's proposed action poses a risk of transporting non-indigenous biofouling species between origination and destination ports. If these species were to establish in destination ports (either dismantling locations or other inactive ship facilities), this has the potential to result in impacts to ESA-listed resources occurring in the destination ports and other connected waterways. Because of these concerns, through the ESA section 7 consultation process, NMFS recommended measures to minimize the risk of invasive species being transported and established in new locations. The mitigation measures[1] the Navy has agreed to implement are described below. We consider the effects of the Navy's implementation of these mitigation measures as effects of the action in this opinion.

4.1 Ship Hull and Underwater Component Cleaning Mitigation

In response to NMFS' comments on the June 2017 BE, the Navy developed a risk analysis using origination and destination port characteristics to determine the appropriate towing scenarios for which mitigation measures may be required to avoid or minimize the risk of transporting invasive species between origination and destination ports. (Navy 2018b). First, the Navy considered the salinity at each port. The Navy determined if a ship were to be towed from one type of port (e.g., saltwater) to another type (e.g., freshwater), this would greatly reduce the chance of a fouling species surviving at the new location. For scenarios where a ship is towed from one type of port (e.g., saltwater) to a similar type, the Navy determined that the likelihood of fouling species surviving would be higher (Navy 2018b). Table 3 lists the water types based on salinity at each origin and destination port for the proposed action. All water types in this table are based on information provided in the BE or supplemental documentation provided by the Navy to NMFS (Navy 2018b).

[1] NMFS considers mitigation measures agreed to by the Navy during the section 7 inter-agency cooperative process as "conservation measures" for purposes of the consultation. Conservation measures are actions to benefit or promote the recovery of ESA-listed species that are included by the Federal agency as an integral part of the proposed action USFWS, and NMFS. 1998. Endangered Species Act Consultation Handbook: Procedures for Conducting Section 7 Consultations and Conferences. U.S. Fish and Wildlife Service and National Marine Fisheries Service. These actions serve to minimize, or compensate for, project effects on the species under review. These may include actions taken prior to the initiation of consultation, or actions which the Federal agency or applicant have committed to complete in a biological assessment or similar document. The implementation of mitigation measures derived during the course of consultation is required under the terms of the consultation ibid..

Table 3. Navy determined water types at origin and destination ports.

Origin Port	Water Type	Destination Port	Water type
Beaumont, Texas	Freshwater	Baltimore, Maryland	Brackish
Mayport, Florida	Brackish	Beaumont, Texas	Freshwater
Norfolk, Virginia	Saltwater	Brownsville, Texas	Saltwater
Pearl Harbor, Hawaii	Saltwater	New Orleans, Louisiana	Freshwater
Philadelphia, Pennsylvania	Freshwater	Pearl Harbor, Hawaii	Saltwater
Puget Sound, Washington	Saltwater	Philadelphia, Pennsylvania	Freshwater
San Diego, California	Saltwater	Puget Sound, Washington	Saltwater

When determining the scenarios in which mitigation measures should be implemented, the Navy also considered whether or not ESA-listed species were present at the destination port. If ESA-listed species were not present, the Navy determined mitigation would not need to be implemented (Navy 2018b).

Table 4 lists all the possible tow scenarios, including information on whether salinity levels between the origination and destination ports are similar (i.e., "like-to-like") and whether ESA-listed species under NMFS jurisdiction are present. All scenarios where <u>either</u> the origination or destination port is "brackish" were considered "like-to-like" due to the higher likelihood of fouling species survival under such scenarios (Navy 2018b).

Table 4. Origination and destination ports included in the proposed action, salinity comparison between origination and destination ports, and the presence or absence of ESA-listed species under NMFS jurisdiction at each destination port (Navy 2018b).

Origin Port	Destination Port	Like-to-like salinities?	ESA-listed species at destination port?
Beaumont, TX	Baltimore, MD	Yes	Yes
	Brownsville, TX	No	Yes
	New Orleans, LA	Yes	No
	Pearl Harbor, HI	No	Yes
	Philadelphia, PA	Yes	Yes
	Puget Sound, WA	No	Yes
Mayport, FL	Baltimore, MD	Yes	Yes
	Beaumont, TX	Yes	No
	Brownsville, TX	Yes	Yes
	New Orleans, LA	Yes	No
	Pearl Harbor, HI	Yes	Yes
	Philadelphia, PA	Yes	Yes
	Puget Sound, WA	Yes	Yes
Norfolk, VA	Baltimore, MD	Yes	Yes
	Beaumont, TX	No	No
	Brownsville, TX	Yes	Yes
	New Orleans, LA	No	No
	Pearl Harbor, HI	Yes	Yes
	Philadelphia, PA	No	Yes
	Puget Sound, WA	Yes	Yes
Pearl Harbor, HI	Baltimore, MD	Yes	Yes
	Beaumont, TX	No	No
	Brownsville, TX	Yes	Yes
	New Orleans, LA	No	No
	Philadelphia, PA	No	Yes
	Puget Sound, WA	Yes	Yes

Origin Port	Destination Port	Like-to-like salinities?	ESA-listed species at destination port?
Philadelphia, PA	Baltimore, MD	Yes	Yes
	Beaumont, TX	Yes	No
	Brownsville, TX	No	Yes
	New Orleans, LA	Yes	No
	Pearl Harbor, HI	No	Yes
	Puget Sound, WA	No	Yes
Puget Sound, WA	Baltimore, MD	Yes	Yes
	Beaumont, TX	No	No
	Brownsville, TX	Yes	Yes
	New Orleans, LA	No	No
	Pearl Harbor, HI	Yes	Yes
	Philadelphia, PA	No	Yes
San Diego, CA	Baltimore, MD	Yes	Yes
	Beaumont, TX	No	No
	Brownsville, TX	Yes	Yes
	New Orleans, LA	No	No
	Pearl Harbor, HI	Yes	Yes
	Puget Sound, WA	Yes	Yes
Note: Shaded cells indicate the largest risk to ESA species if no mitigation is implemented.			

The ship tow routes in Table 5 represent the scenarios in which the Navy has agreed to implement mitigation measures to reduce the potential for species attached to the hull and other underwater components of inactive ships to become invasive and impact ESA-listed resources (Navy 2018b). The Navy will implement hull cleaning mitigation measures for <u>all</u> inactive ship tow scenarios shown in Table 5, <u>except</u> for ships that have had their hull painted within the past three years. That is, if the time between completion of hull painting and commencement of towing is less than three years, hull cleaning mitigation will not be conducted.

Table 5. Scenarios in which the Navy will implement mitigation measures (i.e., in-water hull cleaning or dry-docking) based on "like-to-like" port salinities and the presence of threatened or endangered species at the destination port (Navy 2018b).

Origin Port	Destination Port
Beaumont, TX	Baltimore, MD; Philadelphia, PA;
Mayport, FL	Baltimore, MD; Brownsville, TX; Pearl Harbor, HI; Philadelphia, PA; Puget Sound, WA
Norfolk, VA	Baltimore, MD; Brownsville, TX; Pearl Harbor, HI; Puget Sound, WA
San Diego, CA	Baltimore, MD; Brownsville, TX; Pearl Harbor, HI; Puget Sound, WA
Pearl Harbor, HI	Baltimore, MD; Brownsville, TX; Puget Sound, WA
Philadelphia, PA	Baltimore, MD
Puget Sound, Washington	Baltimore, MD; Brownsville, TX; Pearl Harbor, HI

For each of these scenarios the Navy would use one of three methods to remove fouling organisms from the underwater components (i.e., hull and accessible niche areas) of each inactive ship prior to its departure from the origination port. The three potential methods are: 1) dry-docking long enough to allow species on the hull to "die-off" through desiccation, 2) hull cleaning while the ship is in dry-dock, or 3) in-water hull cleaning (Navy 2018b). For purposes of this opinion, the term "hull cleaning" will be used to describe any of these three options. We will specify "in-water hull cleaning" mitigation, "dry dock hull cleaning," or "dry dock desiccation" as needed to distinguish between these proposed mitigation measures. For all ship tow scenarios involving hull cleaning mitigation the Navy contractor would tow the ship as soon as practicable upon completion of hull cleaning (R. Johnson, Navy, pers. comm. to R. Salz, NMFS, October 17, 2018). This is intended to minimize the attachment of hull fouling organisms during the period between completion of hull cleaning and initiation of the ship tow, thus maximizing the effectiveness of this mitigation measure in reducing the risk of invasive species spread.

For scenarios in which the Navy agrees to implement mitigation measures (see Table 5), the Navy, in accordance with the proposed Uniform National Discharge Standards (81 FR 69753), would first give consideration to one of the dry docking mitigation methods (i.e., dry docking long enough to allow species on the hull to "die-off" through desiccation or hull cleaning while the ship is in dry dock) (Navy 2018d). This regulatory framework states: "Vessel hulls must be inspected, maintained, and cleaned to minimize the removal and discharge of antifouling hull coatings and transport of fouling organisms. To the greatest extent practicable, rigorous vessel hull cleanings must take place in dry dock or at a land-based facility where the removed fouling organisms or spent antifouling hull coatings can be disposed of onshore in accordance with any applicable solid waste or hazardous substance management and disposal requirements" (81 FR 69753). Dry-docking would be the first method considered, though this method would only be implemented if there is a sufficiently sized dry-dock available during the required timeframe that is in close enough proximity to the origination port to preclude risk of invasive species transfer (Navy 2018d). Factors that could limit dry dock availability include: (1) Limited DoD/DoD contractor dry docks in existence, even more so for those of a sufficient size to accommodate larger vessels, particularly aircraft carriers, (2) Priority is always placed on active duty vessels for national defense and mission readiness; this includes both regularly scheduled maintenance and upgrades as well as emergent situations. (3) Priority is also placed on dry dock maintenance and certification over the inactive fleet, and (4) The cost of private dry dock usage could significantly exceed the cost of other mitigations available (e.g. in-water hull cleaning) (Navy 2018d).

In cases where the Navy determines dry docking is not practicable, the Navy will perform in-water hull cleaning (Navy 2018c). In such cases, organisms and/or biofouling communities attached to the hull would be removed using underwater hull cleaning methods and equipment as specified in the Naval Ships' Technical Manual (NSTM) chapter-081, "Waterborne Underwater Hull Cleaning of Navy Ships" (Command 2006). This manual provides a description of the various tools; such as diver-operated machines with rotating brushes, either multi-brush or single-brush fitted with different brush types depending on the machine and fouling conditions present. Professional divers will use hand-held or self-propelled rotary equipment (e.g., brushes or waterjets), hand-held water jets and hydro-lance equipment, manual hand tools (e.g., scrapers, pads, and brushes), and other similar industry-recognized equipment. Multi-brush units are used to clean large unobstructed, easily accessible, areas of the hull. These are fitted with rotary cleaning heads (e.g., brushes) that sweep the growth off the hull. The units are typically held against the hull with either a high volume impeller or the suction generated by brush rotation, and ride along the hull on a set of wheels. Single-brush units are held in place by the diver and the suction force generated from the rotating brush, and are used to clean appendages and hull areas that the large multi-brush unit cannot access. For areas that are harder to reach, divers employ high-pressure water jets, hydro-lances, and hand tools (e.g., scrapers, pads and brushes). The NSTM requires divers to inspect the hull and select the appropriate brush for the fouling condition. The primary goal is to safely and efficiently remove fouling with minimal impact to the underlying hull coating system. The in-water hull cleaning methods proposed by the Navy are not expected to result in the scraping of all hull areas down to the bare hull. Divers will attempt to minimize removal and release of paints and other coatings, or damage to the physical integrity of the hull, by using the least aggressive tools necessary to effectively remove biofouling. Given the poor condition of the hulls of some of the obsolete Navy ships and the need to minimize the release of paint or coating residues, it is recognized that the cleaning operation will not remove all of the "hard fouling" organisms. With the proper equipment and selection of the least aggressive cleaning equipment required to remove fouling, divers can effectively remove upper structure of fouling while preventing excess coating removal. The Navy will continually evaluate the various types of hull cleaning technologies available (e.g., capture systems), and determine whether those methods are effective and feasible to implement in the future (Navy 2018c).

Through the course of the section 7 consultation process, the Navy has also agreed, to the maximum extent practicable, to conduct in-water hull cleanings in the ports of Philadelphia, Pennsylvania, Norfolk, Virginia and Bremerton (Puget Sound), Washington during optimal time frames (i.e., seasonal work windows) to minimize the potential for adverse effects to ESA-listed resources (Navy 2017). Although juvenile ESA-listed fish species (i.e., sturgeon and/or salmon) could be present in these ports year-round, conducting in-water hull cleaning at these ports during the winter would avoid anadromous fish migrations to freshwater habitats, and the seasonal occurrences other ESA-listed species or sensitive life stages (Smith 1985) (Byles 1988; Lutcavage and Musick 1985; Mansfield 2006) (Healey 1991) (NMFS 2016i). In addition, potential adverse effects from in-water hull cleaning on dissolved oxygen (DO) levels and turbidity may be exacerbated during warmer months (Campbell and Goodman 2004; Kahn and Mohead 2010; Secor and Gunderson 1998; Secor and Niklitschek 2001), particularly if hull cleaning at these ports co-occurs with seasonal algal blooms. The effects of in-water hull cleaning on water quality should, therefore, be lessened by conducting this activity during colder months.

In Philadelphia, the Navy will, to the maximum extent practicable, schedule required inactive ship hull-cleaning activities for scenarios involving in-water hull cleaning from November 1 through March 15 (R. Johnson, Navy, pers. comm. to R. Salz, NMFS, October 18, 2018) to avoid impacts to adult Atlantic sturgeon during spawning migrations and early life stages of Atlantic sturgeon (Navy 2017; Smith 1985). In Norfolk, the Navy will, to the maximum extent practicable, schedule required inactive ship hull-cleaning activities for scenarios involving in-water hull cleaning from November 1 through February 28 (Navy 2017) (R. Johnson, Navy, pers. comm. to R. Salz, NMFS, October 15, 2018) to avoid impacts to adult Atlantic sturgeon during spawning migrations (Smith 1985) and the seasonal occurrence of sea turtles in Chesapeake Bay (Byles 1988; Lutcavage and Musick 1985; Mansfield 2006). In Puget Sound, the Navy will, to the maximum extent practicable, schedule required inactive ship hull-cleaning activities for scenarios involving in-water hull cleaning from December 1 through February 28 to avoid impacts to adult Chinook salmon during spawning migrations (Healey 1991) and sensitive life stages of ESA-listed rockfish species (NMFS 2016i). Factors that may preclude the Navy's ability to conduct in-water hull cleaning within these optimal work windows include, but are not limited to, contracting schedules, weather, and availability of personnel or ship berthing space (Navy 2017) (R. Johnson, Navy, pers. comm. to R. Salz, NMFS, October 15, 2018).

For inactive ships which are too large to travel through the Panama Canal (i.e., aircraft carriers or vessels of similar size), towing actions have a preferred seasonal window for going around Cape Horn through the Strait of Magellan. This window is during the Southern Hemisphere's summer months due to the likelihood of severe weather in other seasons. Considering this preferred window for towing, in-water hull cleaning of aircraft carriers (or vessels of similar size) in preparation for towing between ocean basins (e.g., Pacific to Atlantic/Gulf of Mexico) would likely be conducted during the winter months since all origination ports are located in the Northern Hemisphere. As discussed above, this would also likely reduce the potential for adverse effects to ESA-listed resources from in-water hull cleaning.

5 INTERRELATED AND INTERDEPENDENT ACTIONS

Interrelated actions are those that are part of a larger action and depend on that action for their justification. *Interdependent* actions are those that do not have independent utility apart from the action under consideration.

The Navy’s proposed action and analysis of effects, as described in their BE, did not include the process of dismantling inactive ships upon arrival at a dismantling facility. This dismantling process is interrelated to the towing of inactive ships from berthing locations to dismantling facilities, and we therefore consider the effects of the dismantling process on listed species in this opinion.

6 STRESSORS ASSOCIATED WITH THE PROPOSED ACTION AND INTERRELATED AND INTERDEPENDENT ACTIONS

The proposed action involves multiple activities, each of which can create stressors. Stressors are any physical, chemical, or biological entity that may directly or indirectly induce an adverse response either in an ESA-listed (or proposed) species or their designated (or proposed) critical habitat. During consultation, we deconstructed the proposed action to identify potential stressors that could result from the proposed activities. The potential stressors that may result from the proposed action (or from interrelated and interdependent actions) are:

1. Stressors associated with ships (tug and inactive ship) in transit from origination to destination ports including:
   - Ship strike;
   - Ship sound and behavioral disturbance.
2. Water and sediment quality and sound related stressors associated with shipbreaking/dismantling inactive ships at destination ports by ship dismantling contractors, including:
   - Release of heavy metals associated with hull coatings, paints and other ship parts;
   - Release of hazardous materials and other pollutants (e.g., polychlorinated biphenyls [PCBs], petroleum products, asbestos, waste water/sludge, and ozone depleting substances) commonly found in Navy ships;
   - Increased turbidity;
   - Sound produced by shipbreaking/dismantling activities.
3. Water and sediment quality and sound related stressors associated with in-water hull cleaning mitigation conducted at origination ports, including:
   - Release of copper, zinc and other contaminants associated with hull antifouling paints;
   - Depression of DO from the decay of organic matter removed from the hull;
   - Increased turbidity;
   - Release of nutrients (nitrates, nitrites, and ammonia) which are precursors to reduction in DO and algal blooms;
   - Resuspension of contaminants (e.g., PCBs and mercury) in bottom sediments where inactive Navy ship are docked;

   - Changes in organic matter load, dissolved organic carbon (DOC), and biochemical oxygen demand (BOD);
   - Sound produced by in-water hull cleaning activities.

4. Stressors associated with the introduction of non-native species via ballast water exchange or hull fouling, including:
   - Increased competition for food or available habitat;
   - Increased predation;
   - Altered prey availability due to ecosystem level trophic effects;
   - Changes in biological and physical environmental features including habitat structure and water quality;
   - Increased exposure to harmful pathogens and parasites.

The effects of these potential stressors are discussed in more detail in our *Effects of the Action* (Section 10) below.

7 Action Area

Action area means all areas affected directly, or indirectly, by the Federal action, and not just the immediate area involved in the action (50 C.F.R. §402.02). The action area for this consultation includes berthing locations of inactive ships, tow routes, destination ports, and areas where hull fouling species transported via towed inactive Navy ships may become established. The origination and destination ports for the Navy's proposed action are listed in Table 1 (above) and typical tow routes are presented in Figure 1 and Figure 2 (above).

To determine where hull fouling species may become established, we need to consider not only the initial introduction location but also the likelihood of dispersal to surrounding areas. The most likely invasive species introduction locations are the proposed destination ports: Baltimore, Maryland; Brownsville, Texas; Beaumont, Texas; New Orleans, Louisiana; Pearl Harbor, Hawaii; Philadelphia, Pennsylvania; and Puget Sound (Bremerton), Washington. Invasive species initially introduced at one of these destination ports could spread to surrounding areas with similar environmental conditions. As such, we include the connected estuaries and tidally influenced river reaches below as part of the action area for this consultation:

- Chesapeake Bay estuary;
- Neches River, Texas from the saltwater barrier (about eight miles upstream from the port of Beaumont) to where the river enters Sabine Lake;
- Tidal portion of the Mississippi River, Lake Pontchartrain, and Lake Borgne;
- Pearl Harbor, Hawaii entire harbor and waters leading out to Mamala Bay;
- Brownsville (Texas) shipping channel (entire length) and connected intercoastal waterway areas including San Martin Lake, Bahia Grande, Laguna Larga, South Bay, Laguna Madre and Baffin Bay;
- Tidal portion of the Delaware River and Delaware Bay estuary;
- Entire Puget Sound area including Strait of Juan de Fuca, Haro Strait, and the Salish Sea;

8 SPECIES AND CRITICAL HABITAT EVALUATED

The ESA-listed (or proposed) species and designated (or proposed) critical habitat potentially occurring within the action area that may be affected by the proposed action are shown in Table 6 and Table 7, respectively, along with their regulatory status.

NMFS uses two criteria to identify the ESA-listed species or critical habitat that are likely to be adversely affected by the proposed action, as well as the effects of activities that are interrelated to or interdependent with the Federal agency's proposed action.

The first criterion is exposure, or some reasonable expectation of a co-occurrence, between one or more potential stressors associated with the proposed activities and ESA-listed species or designated critical habitat. If we conclude that an ESA-listed species or designated critical habitat is not likely to be exposed to the proposed activities, we must also conclude that the species or critical habitat is not likely to be adversely affected by those activities.

The second criterion is the probability of a response given exposure. ESA-listed species or designated critical habitat that is exposed to a potential stressor but is likely to be unaffected by the exposure is also not likely to be adversely affected by the proposed action.

An action warrants a "may affect, not likely to be adversely affected" finding when its effects are wholly *beneficial, insignificant* or *discountable*. *Beneficial* effects have an immediate positive effect without any adverse effects to the species or habitat. Beneficial effects are usually discussed when the project has a clear link to the ESA-listed species or its specific habitat needs and consultation is required because the species may be affected.

Insignificant effects relate to the size or severity of the impact and include those effects that are undetectable, not measurable, or so minor that they cannot be meaningfully evaluated. Insignificant is the appropriate effect conclusion when plausible effects are going to happen, but will not rise to the level of constituting an adverse effect. That means the ESA-listed species may be expected to be affected, but not harmed or harassed.

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

During consultation we evaluated the spatial and temporal overlap of species in the action area and the stressors that are likely to cause a response. Our results are summarized below.

Table 6. Threatened and endangered species that may be affected by the proposed action.

Species	ESA Status	Recovery Plan
Marine Mammals – Cetaceans		
Blue Whale (*Balaenoptera musculus*)	E – 35 FR 18319	07/1998
False Killer Whale (*Pseudorca crassidens*) Main Hawaiian Islands Insular DPS	E – 77 FR 70915	-- --
Fin Whale (*Balaenoptera physalus*)	E – 35 FR 18319	75 FR 47538
Gulf of Mexico Bryde's Whale (*Balaenoptera edeni*)	E – 81 FR 88639 (proposed)	-- --
Humpback Whale (*Megaptera novaeangliae*) Central America DPS	E – 81 FR 62259	11/1991
Humpback Whale (*Megaptera novaeangliae*) Mexico DPS	T – 81 FR 62259	11/1991
Killer Whale (*Orcinus orca*) Southern Resident DPS	E – 70 FR 69903	73 FR 4176
North Atlantic Right Whale (*Eubalaena glacialis*)	E – 73 FR 12024	70 FR 32293
North Pacific Right Whale (*Eubalaena glacialis*)	E-73 FR 12024	78 FR 34347
Southern Right Whale **(*Eubalaena australis*)**	E – 35 FR 8491	
Sei Whale (*Balaenoptera borealis*)	E – 35 FR 18319	12/2011
Sperm Whale (*Physeter macrocephalus*)	E – 35 FR 18319	75 FR 81584
Marine Mammals – Pinnipeds		
Guadalupe Fur Seal (*Arctocephalus townsendi)*	T – 50 FR 51252	-- --
Hawaiian Monk Seal (*Neomonachus schauinslandi*)	E – 41 FR 51611	72 FR 46966
Marine Reptiles – Sea Turtles		
Green Turtle (*Chelonia mydas*) Central North Pacific DPS	T – 81 FR 20057	63 FR 28359
Green Turtle (*Chelonia mydas*) East Pacific DPS	T – 81 FR 20057	63 FR 28359
Green Turtle (*Chelonia mydas*) North Atlantic DPS	T – 81 FR 20057	FR Not Available 11/1991

Species	ESA Status	Recovery Plan
Green Turtle (*Chelonia mydas*) South Atlantic DPS	T – 81 FR 20057	-- --
Hawksbill Turtle (*Eretmochelys imbricata)*	E – 35 FR 8491	63 FR 28359 and 57 FR 38818
Kemp's Ridley Turtle (*Lepidochelys kempii*)	E – 35 FR 18319	9/2011
Leatherback Turtle (*Dermochelys coriacea*)	E – 35 FR 8491	63 FR 28359 and 10/1991
Loggerhead Turtle (*Caretta caretta*) Northwest Atlantic Ocean DPS	T – 76 FR 58868	74 FR 2995
Loggerhead Turtle (*Caretta caretta*) North Pacific Ocean DPS	E – 76 FR 58868	63 FR 28359
Loggerhead Turtle (*Caretta caretta*) South Atlantic Ocean DPS	T – 76 FR 58868	
Loggerhead Turtle (*Caretta caretta*) South Pacific Ocean DPS	E – 76 FR 58868	-- --
Olive Ridley Turtle (*Lepidochelys olivacea*) Mexico's Pacific Coast Breeding Colonies	E – 43 FR 32800	63 FR 28359
Olive Ridley Turtle (*Lepidochelys olivacea*) All Other Areas	T – 43 FR 32800	-- --
Fishes		
Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*) Carolina DPS	E – 77 FR 5913	-- --
Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*) Chesapeake DPS	E – 77 FR 5879	-- --
Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*) Gulf of Maine DPS	T – 77 FR 5879	-- --
Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*) New York Bight DPS	E – 77 FR 5879	-- --
Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*) South Atlantic DPS	E – 77 FR 5913	-- --
Bocaccio (*Sebastes paucispinis*) Puget Sound/Georgia Basin DPS	E – 75 FR 22276 and 82 FR 7711	10/2017
Chinook Salmon (*Oncorhynchus tshawytscha)* California Coastal ESU	T – 70 FR 37160	81 FR 70666

Species	ESA Status	Recovery Plan
Chinook Salmon (*Oncorhynchus tshawytscha)* Central Valley Spring-Run ESU	T – 70 FR 37160	79 FR 42504
Chinook Salmon (*Oncorhynchus tshawytscha)* Lower Columbia River ESU	T – 70 FR 37160	78 FR 41911
Chinook Salmon (*Oncorhynchus tshawytscha)* Puget Sound ESU	T – 70 FR 37160	72 FR 2493
Chinook Salmon (*Oncorhynchus tshawytscha)* Sacramento River Winter-Run ESU	E – 70 FR 37160	79 FR 42504
Chinook Salmon (*Oncorhynchus tshawytscha)* Snake River Fall-Run ESU	T – 70 FR 37160	80 FR 67386 (Draft)
Chinook Salmon (*Oncorhynchus tshawytscha)* Snake River Spring/Summer Run ESU	T – 70 FR 37160	81 FR 74770 (Draft)
Chinook Salmon (*Oncorhynchus tshawytscha)* Upper Columbia River Spring-Run ESU	E – 70 FR 37160	72 FR 57303
Chinook Salmon (*Oncorhynchus tshawytscha)* Upper Willamette River ESU	T – 70 FR 37160	76 FR 52317
Chum Salmon (*Oncorhynchus keta*) Columbia River ESU	T – 70 FR 37160	78 FR 41911
Chum Salmon (*Oncorhynchus keta*) Hood Canal Summer-Run ESU	T – 70 FR 37160	72 FR 29121
Coho Salmon (*Oncorhynchus kisutch*) Central California Coast ESU	E – 70 FR 37160	77 FR 54565
Coho Salmon (*Oncorhynchus kisutch*) Lower Columbia River ESU	T – 70 FR 37160	78 FR 41911
Coho Salmon (*Oncorhynchus kisutch*) Oregon Coast ESU	T – 73 FR 7816	81 FR 90780
Coho Salmon (*Oncorhynchus kisutch*) Southern Oregon and Northern California Coasts ESU	T – 70 FR 37160	79 FR 58750
Eulachon (*Thaleichthys pacificus*) Southern DPS	T – 75 FR 13012	9/2017
Giant Manta Ray (*Manta birostris*)	T – 83 FR 2916	-- --

Species	ESA Status	Recovery Plan
Green Sturgeon (*Acipenser medirostris*) Southern DPS	T – 71 FR 17757	2010 (Outline)
Gulf Sturgeon (*Acipenser oxyrinchus desotoi*)	T – 56 FR 49653	09/1995
Nassau Grouper (*Epinephelus striatus*)	T – 81 FR 42268	-- --
Oceanic Whitetip Shark (*Carcharhinus longimanus*)	T – 83 FR 4153	-- --
Scalloped Hammerhead Shark (*Sphyrna lewini*) Central and Southwest Atlantic DPS	T – 79 FR 38213	-- --
Scalloped Hammerhead Shark (*Sphyrna lewini*) Eastern Pacific DPS	E – 79 FR 38213	-- --
Shortnose Sturgeon (*Acipenser brevirostrum*)	E – 32 FR 4001	63 FR 69613
Smalltooth Sawfish (*Pristis pectinata*) U.S. portion of range DPS	E – 68 FR 15674	74 FR 3566
Largetooth Sawfish **(*Pristis pristis*)**	E – 76 FR 40822 and E - 79 FR 73977	
Sockeye Salmon (*Oncorhynchus nerka*) Ozette Lake ESU	T – 70 FR 37160	74 FR 25706
Sockeye Salmon (*Oncorhynchus nerka*) Snake River ESU	E – 70 FR 37160	80 FR 32365
Steelhead Trout (*Oncorhynchus mykiss*) California Central Valley DPS	T – 71 FR 834	79 FR 42504
Steelhead Trout (*Oncorhynchus mykiss*) Central California Coast DPS	T – 71 FR 834	81 FR 70666
Steelhead Trout (*Oncorhynchus mykiss*) Lower Columbia River DPS	T – 71 FR 834	78 FR 41911
Steelhead Trout (*Oncorhynchus mykiss*) Middle Columbia River DPS	T – 71 FR 834	74 FR 50165
Steelhead Trout (*Oncorhynchus mykiss*) Northern California DPS	T – 71 FR 834	81 FR 70666
Steelhead Trout (*Oncorhynchus mykiss*) Puget Sound DPS	T – 72 FR 26722	-- --
Steelhead Trout (*Oncorhynchus mykiss*) Snake River Basin DPS	T – 71 FR 834	81 FR 74770 (Draft)

Species	ESA Status	Recovery Plan
Steelhead Trout (*Oncorhynchus mykiss*) South-Central California Coast DPS	T – 71 FR 834	78 FR 77430
Steelhead Trout (*Oncorhynchus mykiss*) Southern California DPS	E – 71 FR 834	77 FR 1669
Steelhead Trout (*Oncorhynchus mykiss*) Upper Columbia River DPS	T – 71 FR 834	72 FR 57303
Steelhead Trout (*Oncorhynchus mykiss*) Upper Willamette River DPS	T – 71 FR 834	76 FR 52317
Yelloweye Rockfish (*Sebastes rubberimus*) Puget Sound/Georgia Basin DPS	E – 75 FR 22276 and 82 FR 7711	10/2017
Marine Invertebrates		
Black Abalone (*Haliotis cracherodii*)	E – 74 FR 1937	-- --
Boulder Star Coral (*Orbicella franksi*)	T – 79 FR 53851	-- --
Elkhorn Coral (*Acropora palmata*)	T – 79 FR 53851	80 FR 12146
Lobed Star Coral (*Orbicella annularis*)	T – 79 FR 53851	-- --
Mountainous Star Coral (*Orbicella faveolata*)	T – 79 FR 53851	-- --
Rough Cactus Coral (*Mycetophyllia ferox*)	T – 79 FR 53851	-- --
Pillar Coral (*Dendrogyra cylindrus*)	T – 79 FR 53851	-- --
Staghorn Coral (*Acropora cervicornis*)	T – 79 FR 53851	80 FR 12146
White Abalone (*Haliotis sorenseni*)	E – 66 FR 29046	73 FR 62257

Table 7. Designated critical habitat that may be affected by the proposed action.

Species	Designated Critical Habitat
Marine Mammals – Cetaceans	
North Atlantic Right Whale (*Eubalaena glacialis*)	59 FR 28805 and 81 FR 4837
False Killer Whale (*Pseudorca crassidens*) Main Hawaiian Islands Insular DPS	83 FR 35062
Killer Whale (*Orcinus orca*) Southern Resident DPS	71 FR 69054
Hawaiian Monk Seal (*Neomonachus schauinslandi*)	80 FR 50925
Marine Reptiles – Sea Turtles	
Leatherback Turtle (*Dermochelys coriacea*)	44 FR 17710 and 77 FR 4170
Loggerhead Turtle (*Caretta caretta*) Northwest Atlantic Ocean DPS	79 FR 39856
Fishes	
Bocaccio (*Sebastes paucispinis*) Puget Sound/Georgia Basin DPS	79 FR 68041
Chinook Salmon (*Oncorhynchus tshawytscha)* Puget Sound ESU	70 FR 52629
Chum Salmon (*Oncorhynchus keta*) Hood Canal Summer-Run ESU	70 FR 52629
Green Sturgeon (*Acipenser medirostris*) Southern DPS	74 FR 52300
Gulf Sturgeon (*Acipenser oxyrinchus desotoi*)	68 FR 13370
Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus)* New York Bight DPS	82 FR 39160
Steelhead Trout (*Oncorhynchus mykiss*) Puget Sound DPS	81 FR 9251
Yelloweye Rockfish (*Sebastes rubberimus*) Puget Sound/Georgia Basin DPS	79 FR 68041
Marine Invertebrates	
Black Abalone (*Haliotis cracherodii*)	76 FR 66805
Elkhorn Coral (*Acropora palmata*)	73 FR 72210
Staghorn Coral (*Acropora cervicornis*)	73 FR 72210

8.1 Species and Critical Habitat Not Carried Forward

The ESA-listed species and designated critical habitats deemed not likely to be adversely affected (NLAA) by the proposed action are listed below. The rationale for reaching the determination of NLAA for each of these species and/or their designated critical habitat is discussed in this section.

ESA-Listed Species Not Likely to be Adversely Affected

- Cetaceans:
  - Blue whale
  - Southern right whale
  - False killer whale: Main Hawaiian islands insular distinct population segment (DPS)
  - Fin whale
  - Humpback whale DPSs: Central America and Mexico
  - North Atlantic right whale
  - North Pacific right whale
  - Sei whale
  - Sperm whale
- Pinnipeds:
  - Guadalupe fur seal
- Sea Turtles:
  - Green DPSs: South Atlantic
  - Loggerhead DPSs: North Pacific, South Atlantic, and South Pacific
  - Olive ridley: Mexico's Pacific Coast Breeding Colonies and all other areas
- Fishes:
  - Chinook salmon ESUs (Evolutionary Significant Units): California Coastal, Central Valley spring-run, Lower Columbia River, Sacramento River winter-run, Snake River fall-run, Snake River spring/summer-run, Upper Columbia River spring-run, Upper Willamette River;
  - Chum salmon: Columbia River ESU
  - Coho salmon ESUs: Central California Coast, Lower Columbia River, Oregon Coast, Southern Oregon/Northern California Coasts
  - Giant manta ray
  - Nassau grouper
  - Oceanic whitetip shark
  - Scalloped hammerhead shark DPSs: Central and Southwest Atlantic and Eastern Pacific
  - Smalltooth sawfish: U.S. portion of range DPS

    - Largetooth sawfish
    - Sockeye salmon: Snake River ESU
    - Steelhead trout DPSs: California Central Valley, Central California Coast, Lower Columbia River, Middle Columbia River, Northern California, Snake River Basin, South-Central California Coast, Southern California, Upper Columbia River, Upper Willamette River
- Invertebrates:
    - Black abalone
    - Corals: boulder star coral, elkhorn coral, lobed star coral, mountainous star coral, pillar coral, rough cactus coral, staghorn coral,
    - White abalone

Proposed Species Not Likely to be Adversely Affected

- Gulf of Mexico Bryde's whale

Designated Critical Habitat Not Likely to be Adversely Affected

- Cetaceans:
    - North Atlantic right whale
    - False killer whale: Main Hawaiian islands insular DPS
- Hawaiian monk seal
- Invertebrates:
    - Black abalone
    - Elkhorn coral
    - Staghorn coral

8.1.1 Species and Critical Habitats Occurring in Deepwater, Pelagic Areas

Several of the ESA-listed (or proposed) species or designated (or proposed) critical habitats that may be present within the action area occur exclusively (or nearly so) in offshore or deepwater marine environments. It is unlikely that these species will be exposed to stressors associated the introduction of invasive species from inactive Navy ships. Based on our review of the available literature, detailed further in our effects analysis (Section 10.1.3 below), we discount the possibility that invasive species introduced from inactive Navy ships which are docked in nearshore or inland ports would spread to offshore habitats. Given the slow tow speeds, it is also highly unlikely that hull-fouling species would release from the hull in these offshore environments while the ship is in-tow, or that they would be able to survive and propagate in such a different environment from the origin port. 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. The mitigation measures (described in Section 4 above) agreed to by the Navy during the course of section 7 consultation further reduce this risk. Since none of these offshore species occur in (or in close proximity to) the ports considered in this opinion, there will be no effect to these species and habitats from in-water hull cleaning or ship dismantling activities.

Ship and tow line strike could potentially affect these offshore and pelagic ESA-listed marine mammals and fish species. Although the tug, tow cable, and towed ship may affect ESA-listed species encountered along the proposed tow routes, the chance of such an encounter is extremely remote because of the low probability that an ESA-listed species would overlap with the infrequent towing events (i.e., estimated less than ten per year). The relatively low speed (less than ten knots) of the tug and tow further reduces the chance of a ship striking a large whale (Conn and Silber 2013; Jensen and Silber 2004; Laist et al. 2001; Vanderlaan and Taggart 2007). The majority of ship strikes of large whales occur when ships are traveling at speeds greater than approximately ten knots, with faster ships, especially of large ships (80 meters or greater), being more likely to cause serious injury or death (Conn and Silber 2013; Jensen and Silber 2004; Laist et al. 2001; Vanderlaan and Taggart 2007). Ship strikes of elasmobranchs are very rare in general and their distribution in the water column further reduces the chance of a strike from the proposed action.

The temporary disturbance of ESA-listed marine mammals and fish species associated with the anticipated infrequent ship interactions as part of the proposed action are not expected to result in injury or reduced fitness as we do not anticipate any significant disruption of breeding, feeding, or sheltering to occur. Therefore, any potential effects from ship sound or ship avoidance behavior resulting from the proposed action would be undetectable, not measurable, or so minor that they cannot be meaningfully evaluated, and thus would be insignificant.

In summary, for the following offshore (or predominantly offshore) and pelagic species, we discount (i.e., extremely unlikely to occur) the exposure to invasive species and to ship strike, and find the potential effects from ship sound or ship avoidance behavior to be insignificant (i.e., undetectable, not measurable, or so minor that they cannot be meaningfully evaluated). As such, we determine that the proposed action is not likely to adversely affect the following ESA-listed resources:

- Cetaceans:
  - Blue whale
  - Southern right whale
  - False killer whale: Main Hawaiian islands insular DPS
  - Fin whale
  - Humpback whale DPSs: Central America and Mexico
  - North Atlantic right whale
  - North Pacific right whale
  - Sei whale

  - Sperm whale
  - Gulf of Mexico Bryde's whale
- Fishes:
  - Giant manta ray
  - Oceanic whitetip shark
  - Scalloped hammerhead shark DPSs: Central and Southwest Atlantic and Eastern Pacific
- Designated Critical Habitat:
  - North Atlantic right whale
  - False killer whale: Main Hawaiian islands insular DPS
  - Hawaiian monk seal

8.1.2 Species and Critical Habitats Occurring in Coastal, Inshore Areas

In this section we address the ESA-listed (or proposed) species or designated (or proposed) critical habitats that may be present within the action area and occur in habitats (i.e., inshore, estuarine, and freshwater) similar to one of the proposed origin/destination ports but are located far from any of the proposed ports (see list below). Since none of these species occur in (or in close proximity to) either the origin or destination ports considered in this opinion, there will be no effect to these species and their habitats from in-water hull cleaning or ship dismantling activities. The species and critical habitats listed below may be affected by the following stressors as a result of the proposed action: vessel strikes, vessel disturbance, and invasive species introduction. In our analysis below we discuss why the effects from these stressors would likely be either discountable (vessel strike and invasive species) or insignificant (vessel disturbance).

- Pinnipeds:
  - Guadalupe fur seal: Southern resident DPS
- Sea Turtles:
  - Green sea turtle South Atlantic DPS
  - Loggerhead DPSs: North Pacific, South Atlantic, and South Pacific
  - Olive ridley: Mexico's Pacific Coast Breeding Colonies and all other areas
- Fishes:
  - Chinook salmon ESUs: California Coastal, Central Valley spring-run, Lower Columbia River, Sacramento River winter-run, Snake River fall-run, Snake River spring/summer-run, Upper Columbia River spring-run, Upper Willamette River
  - Chum salmon: Columbia River ESU
  - Coho salmon ESUs: Central California Coast, Lower Columbia River, Oregon Coast, Southern Oregon/Northern California
  - Nassau grouper

  - Smalltooth sawfish: U.S. portion of range DPS
  - Largetooth sawfish
  - Sockeye salmon: Snake River ESU, Ozette Lake ESU
  - Steelhead trout DPSs: California Central Valley, Central California Coast, Lower Columbia River, Middle Columbia River, Northern California, Snake River Basin, South-Central California Coast, Southern California, Upper Columbia River, Upper Willamette River
- Invertebrates:
  - Black abalone
  - Corals: boulder star coral, elkhorn coral, lobed star coral, mountainous star coral, pillar coral, rough cactus coral, staghorn coral,
  - White abalone
- Designated Critical Habitat:
  - Black abalone
  - Elkhorn coral
  - Staghorn coral

Given the very large distance between these species' ranges and the inactive ship destination ports, we discount the possibility that invasive species introduced after the ship has reached the destination port will affect these species or their habitats. Although marine invasive species dispersal rates can be high once the species is established (Sorte et al. 2010), given the very large distances involved it is speculative to say that a particular invasive species introduced by an inactive Navy ship will spread naturally from the destination port to within the range of one of these ESA-listed species, and that the invasive species will adversely affect the ESA-listed species or critical habitat.

As discussed above, it is also unlikely that hull-fouling species would release from the hull while the ship is in-tow given the slow tow speeds, or that they would be able to survive and propagate post-release. To become established in areas along the tow route, biofouling organisms would need to dislodge from towed ships, land on hard substrate, be tolerant of the physiochemical properties of the habitat, be abundant enough and in high enough density to reproduce, and compete for resources with already established organisms. In the unlikely event of an invasive species introduction while the ship is being towed, it remains uncertain that the newly established species will adversely affect an ESA-listed species or critical habitat.

Ship and tow line strike along the tow route could potentially affect the pinniped, turtles and fish species above. Results from a study by Hazel et al. (2007) suggest that green turtles cannot consistently avoid being struck by ships moving at relatively moderate speeds (i.e., greater than four kilometers per hour). Although the tug, tow cable, and towed ship may affect ESA-listed species encountered along the proposed tow routes, the chance of such an encounter is extremely remote because of the low probability that an ESA-listed species would overlap with the infrequent towing events (i.e., estimated less than ten per year). The relatively low speed (less than ten knots) of the tug and tow further reduces the chance of a ship strike. Turtles may use auditory cues to react to approaching ships rather than visual cues, making them more susceptible to strike as ship speed increases (Hazel et al. 2007). Ship strikes of the fish species above are extremely rare in general and their distribution in the water column further reduces the chance of a strike from the proposed action.

The temporary disturbance of ESA-listed species associated with the anticipated infrequent ship interactions as part of the proposed action are not expected to result in injury or reduced fitness as we do not anticipate any significant disruption of breeding, feeding, or sheltering to occur. Therefore, any potential effects from ship sound or ship avoidance behavior resulting from the proposed action would be undetectable, not measurable, or so minor that they cannot be meaningfully evaluated, and thus would be insignificant..

In summary, for the ESA-listed species above, we discount (i.e., extremely unlikely to occur) the exposure to invasive species and to ship strike, and find the potential effects from ship sound or ship avoidance behavior to be insignificant (i.e., undetectable, not measurable, or so minor that they cannot be meaningfully evaluated). As such, we determine that the proposed action is not likely to adversely affect the ESA-listed resources above.

8.2 Status of Species and Critical Habitat Analyzed Further

In this section we summarize the biology and ecology of ESA-listed species analyzed further in this opinion, including a discussion of their life histories and designated critical habitats within the action area based on the best available information. The status is determined by the level of risk that the ESA-listed species and designated critical habitats face, based on parameters considered in documents such as listing decisions, status reviews, and recovery plans. This section helps to inform the description of the species' current "reproduction, numbers, or distribution" as described in 50 CFR 402.02. More detailed information on the status and trends of these ESA-listed species, and their biology and ecology can be found in the listing regulations and critical habitat designations published in the Federal Register, status reviews, recovery plans, and on the NMFS Web site: http://www.nmfs.noaa.gov/pr/species/esa/listed.htm. The ESA-listed species and designated critical habitats analyzed further in this opinion are listed below.

ESA-listed Species Analyzed Further in this Opinion

- Cetaceans:
    - Killer whale, Southern Resident DPS
- Pinnipeds:
    - Hawaiian monk seal

- Sea Turtles:
  - Green DPSs: Central North Pacific, and North Atlantic
  - Hawksbill
  - Kemp's ridley
  - Leatherback
  - Loggerhead: Northwest Atlantic Ocean DPS
- Fishes:
  - Atlantic sturgeon DPSs: Carolina, Chesapeake, Gulf of Maine, New York Bight, and South Atlantic
  - Bocaccio: Puget Sound/Georgia Basin DPS
  - Chinook salmon: Puget Sound ESU
  - Chum salmon: Hood Canal summer-run ESU
  - Eulachon: Southern DPS
  - Green sturgeon: Southern DPS
  - Gulf sturgeon
  - Shortnose sturgeon
  - Steelhead trout: Puget Sound DPS
  - Yelloweye rockfish: Puget Sound/Georgia Basin DPS

<u>Designated Critical Habitat Analyzed Further in this Opinion</u>

- Cetaceans:
  - Killer whale, Southern Resident DPS
- Sea Turtles:
  - Leatherback
  - Loggerhead: Northwest Atlantic Ocean DPS
- Fishes:
  - Bocaccio: Puget Sound/Georgia Basin DPS
  - Chinook salmon: Puget Sound ESU
  - Chum salmon: Hood Canal summer-run ESU
  - Green sturgeon: Southern DPS
  - Atlantic sturgeon, New York Bight DPS
  - Gulf sturgeon
  - Steelhead trout: Puget Sound DPS
  - Yelloweye rockfish: Puget Sound/Georgia Basin DPS

8.2.1 Killer Whale, Southern Resident DPS and Designated Critical Habitat

Killer whales are distributed worldwide, but populations are isolated by region and ecotype. Killer whales have been divided into DPSs on the basis of differences in genetics, ecology, morphology and behavior. The Southern Resident DPS of killer whale can be found along the Pacific Coast of the United States and Canada, and in the Salish Sea, Strait of Juan de Fuca, and Puget Sound (Figure 4).

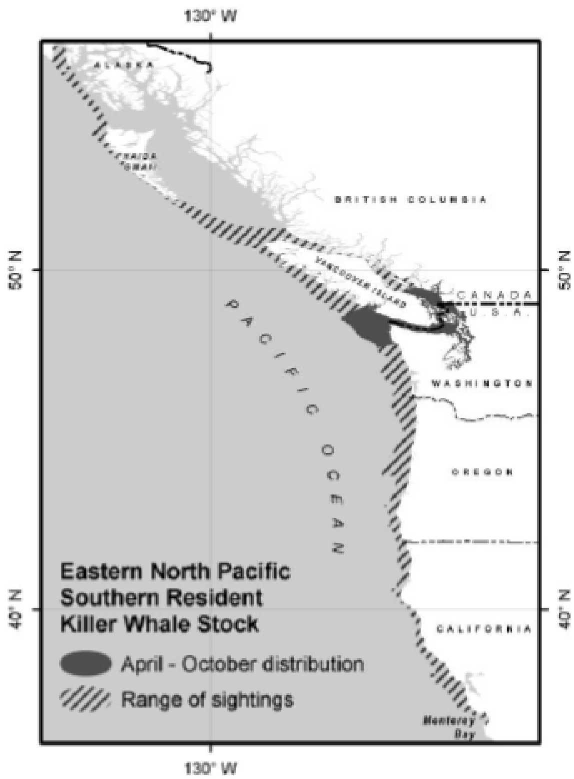


Figure 4. Map identifying the range of the endangered Southern Resident DPS of killer whale. Approximate April to October distribution of the Southern Resident DPS of killer whale (shaded area) and range of sightings (diagonal lines) (Carretta et al. 2016).

Killer whales are odontocetes and the largest delphinid species with black coloration on their dorsal side and white undersides and patches near the eyes. They also have a highly variable gray or white saddle behind the dorsal fin (Figure 5).

Figure 5. Southern Resident killer whales. Photo: NOAA.

The Southern Resident DPS of killer whales was listed as endangered under the ESA on November 18, 2005 (Table 8).

Table 8. Summary of Southern Resident killer whale listing and recovery plan information.

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Orcinus orca	Killer Whale	Southern Resident	Endangered	2016	70 FR 69903 11/18/2005	73 FR 4176	71 FR 69054 2006

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

Life History

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

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes population growth rate, genetic diversity, and distribution as it relates to the Southern Resident DPS of killer whale.

Population Growth Rate

The most recent abundance estimate for the Southern Resident DPS is 81 whales in 2015 (Carretta et al. 2017) (80 whales in 2016 [2]). This represents a decline from just a few years ago, when in 2012, there were 85 whales. Population abundance has fluctuated over time with a maximum of approximately 100 whales in 1995 (Carretta et al. 2017), with an increase between 1974 and 1993, from 76 to 93 individuals. As compared to stable or growing populations, the DPS reflects lower fecundity and has demonstrated little to no growth in recent decades (NMFS 2016h).

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

Genetic Diversity

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

Distribution

Southern Resident DPS of killer whales occur in the inland waterways of Puget Sound, Strait of Juan de Fuca, and Southern Georgia Strait during the spring, summer and fall. During the winter, they move to coastal waters primarily off Oregon, Washington, California, and British Columbia (Figure 4).

Vocalization and Hearing

Killer whales have advanced vocal communication and also use vocalizations to aid in navigation and foraging (NMFS 2008). Their vocalizations typically have both a low frequency component (250 Hertz to 1.5 kilohertz) and a high frequency component (five to 12 kilohertz) (NMFS 2008). Killer whale vocalizations consist of three main types, echolocation clicks, which are primarily used for navigation and foraging, and tonal whistles and pulse calls, which are

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

thought to be used for communication (NMFS 2008). Individual Southern Resident DPS of killer whale pods have distinct call repertoires, with each pod being recognizable by its acoustic dialect (NMFS 2008). Killer whale hearing is one of the most sensitive of any odontocete, with a hearing range of one to 120 kilohertz, with the most sensitive range being between 18 and 42 kilohertz range (Szymanski et al. 1999).

Status

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

Critical Habitat

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

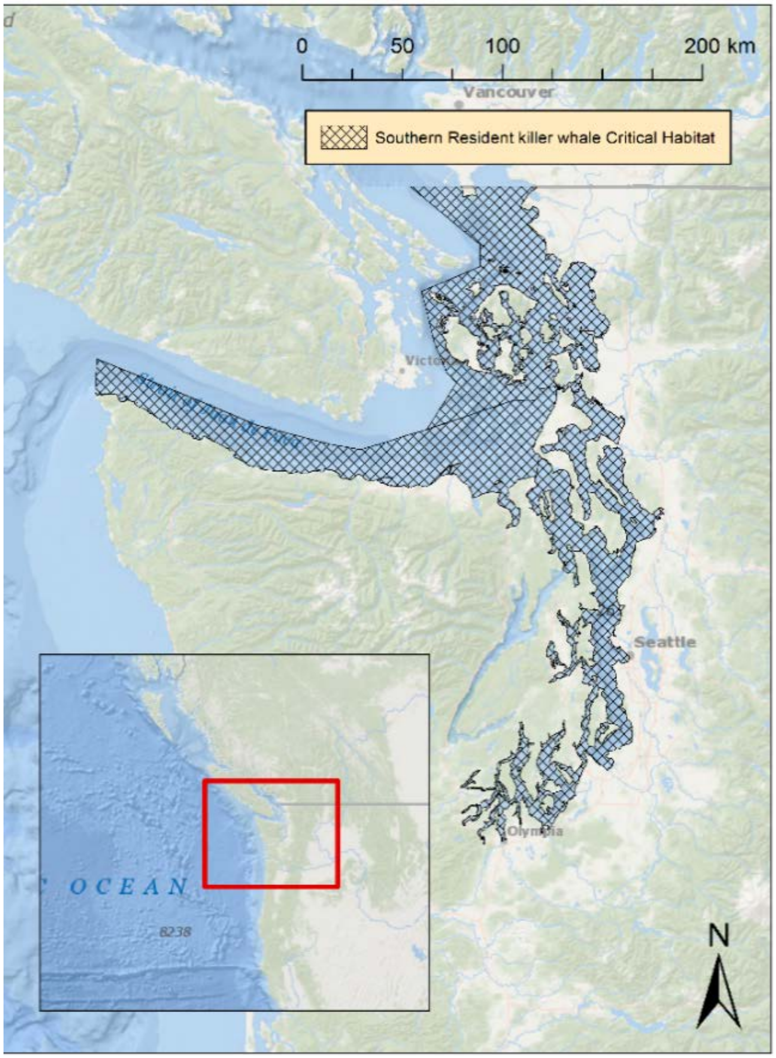


Figure 6. Map depicting designated critical habitat for the endangered Southern Resident DPS of killer whale.

Recovery Goals

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

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

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

8.2.2 Hawaiian Monk Seal

The Hawaiian monk seal is a large phocid ("true seal") that is one of the rarest marine mammals in the world. The Hawaiian monk seal inhabits the Northwestern Hawaiian Islands and Main Hawaiian Islands (Figure 7). Hawaiian monk seals are silvery-grey with a lighter creamy coloration on their underside (newborns are black), they may also have light patches of red or green tinged coloration from attached algae (Figure 8).

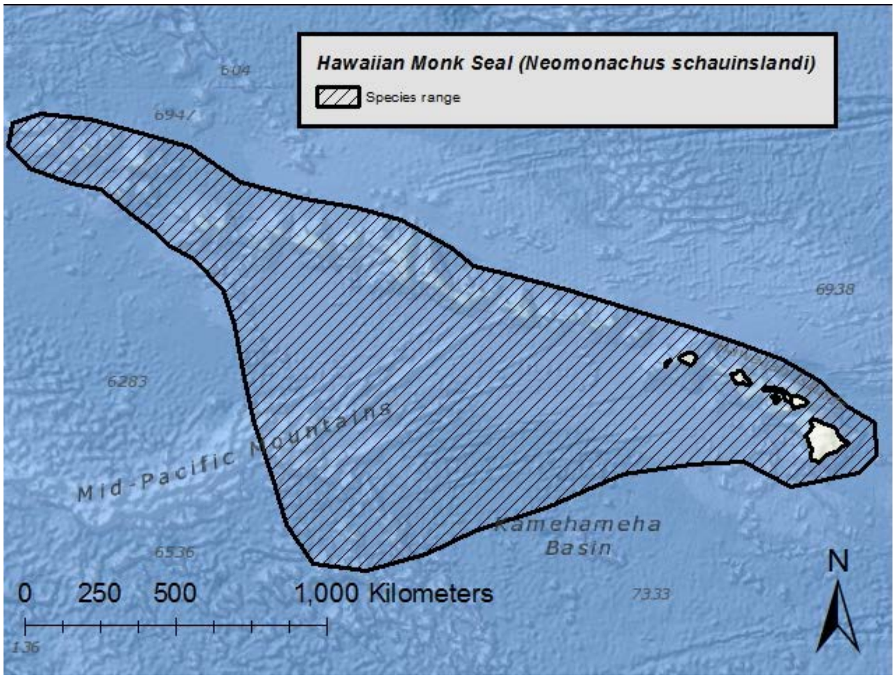


Figure 7. Map identifying the range of the endangered Hawaiian monk seal.

Figure 8. Hawaiian monk seal. Photo: NOAA.

The Hawaiian monk seal was originally listed as endangered on November 23, 1976 (Table 9).

Table 9. Summary of Hawaiian monk seal listing and recovery plan information.

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Neomonachus schauinslandi	Hawaiian monk seal	None	Endangered	2007	41 FR 51611 11/23/1976	2007	80 FR 50925 2015 53 FR 18988 1988

Information available from the recovery plan (NMFS 2007e), recent stock assessment report (Carretta et al. 2016), and status review (NMFS 2007c) were used to summarize the life history, population dynamics and status of the species as follows.

Life History

Hawaiian monk seals can live, on average, twenty-five to thirty years. Sexual maturity in females is reached around five years of age and it is thought to be similar for males but they do not gain access to females until they are older. They have a gestation period of ten to eleven months, and calves nurse for approximately one month while the mother fasts and remains on land. After nursing, the mother abandons her pup and returns to the sea for eight to ten weeks before returning to beaches to molt. Males compete in a dominance hierarchy to gain access to females (i.e., guarding them on shore). Mating occurs at sea, however, providing opportunity for female mate choice. Monk seals are considered foraging generalist that feed primarily on benthic and demersal prey such as fish, cephalopods, and crustaceans. They forage in subphotic zones either because those areas host favorable prey items or because these areas are less accessible by competitors (Parrish et al. 2000).

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes population growth rate, genetic diversity, and distribution as it relates to the Hawaiian monk seal.

Population Growth Rate

The entire range of the Hawaiian monk seal is located within U.S. waters. In addition to a small but growing population found on the main Hawaiian Islands there are six main breeding subpopulations in the northwestern Hawaiian Islands identified as: Kure Atoll, Midway Islands, Pearl and Hermes Reef, Lisianski Island, Laysan Island, and French Frigate Shoals. The best estimate of the total population of Hawaiian monk seals is 1,112. This estimate is the sum of estimated abundance at the six main northwestern Hawaiian islands subpopulations, an extrapolation of counts at Necker and Nihoa Islands (smaller breeding sub-populations), and an estimate of minimum abundance in the main Hawaiian islands. The minimum population size for the entire species is 1,088 (781 for the six main northwestern Hawaiian islands reproductive sites, 38.3 and 89.3 for Necker and Nihoa Islands respectively, and 179 individuals in the main Hawaiian islands).

The overall abundance of Hawaiian monk seals has declined by over sixty-eight percent since 1958. Current estimates indicate a growth rate of approximately 6.5 percent annually for the main Hawaiian islands subpopulation (Baker et al. 2011). Likewise, sporadic beach counts at Necker and Nihoa Islands suggest a positive growth rate. The six main northwestern Hawaiian Islands subpopulations continue to decline at approximately 3.4 percent annually.

Genetic Diversity

Genetic analysis indicates the species is a single panmictic population, thus warranting a single stock designation (Schultz et al. 2011). Genetic variation among monk seals is extremely low and may reflect a long-term history at low population levels and more recent human influences (Kretzmann et al. 2001; Schultz et al. 2009). In addition to low genetic variability, studies by Kretzmann et al. (1997) suggest the species is characterized by minimal genetic differentiation among sub-populations and, perhaps some naturally occurring local inbreeding. The potential for genetic drift should have increased when seal numbers were reduced by European harvest in the nineteenth century, but any tendency for genetic divergence among sub-populations is probably mitigated by the inter-island movements of seals. Since the population is so small there is concern about long-term maintenance of genetic diversity making it quite likely that this species will remain endangered for the foreseeable future.

Distribution

The Hawaiian monk seal inhabits the Northwestern Hawaiian Islands and Main Hawaiian Islands (Figure 7).

Status

Hawaiian monk seals were once harvested for their meat, oil, and skins, leading to extirpation in the main Hawaiian Islands and near-extinction of the species by the twentieth century (Hiruki and Ragen 1992; Ragen 1999). The species partially recovered by 1960, when hundreds of seals were counted on northwestern Hawaiian Islands beaches. Since then, however, the species has declined in abundance. Though the ultimate cause(s) for the decline remain unknown threats include: food limitations in northwestern Hawaiian Islands, entanglement in marine debris, human interactions, loss of haul-out and pupping beaches due to erosion in northwestern Hawaiian Islands, disease outbreaks, shark predation, male aggression towards females, and low genetic diversity. With only approximately 1,112 individuals remaining the species’ resilience to further perturbation is low.

Critical Habitat

Hawaiian monk seal critical habitat was originally designated on April 30, 1986 and was extended on May 26, 1988 (Table 9). It includes all beach areas, sand spits, and islets (including all beach crest vegetation to its deepest extent inland), lagoon waters, inner reef waters, and ocean waters out to a depth of twenty fathoms (thirty-seven meters) around the northwestern Hawaiian Islands breeding atolls and islands. The marine component of this habitat serves as foraging areas, while terrestrial habitat provides resting, pupping, and nursing habitat (Figure 9).

On September 21, 2015, NMFS published a final rule to revise critical habitat for Hawaiian monk seals, extending the current designation in the northwestern Hawaiian Islands out to the 200 meter depth contour (including Kure Atoll, Midway Islands, Pearl and Hermes Reef, Lisianski Island, Laysan Island, Maro Reef, Gardner Pinnacles, French Frigate Shoals, Necker Island, and Nihoa Island). It also designates six new areas in in the main Hawaiian Islands (i.e., terrestrial and marine habitat from five meters inland from the shoreline extending seaward to the 200 meter depth contour around Kaula, Niihau, Kauai, Oahu, Maui Nui, and Hawaii).

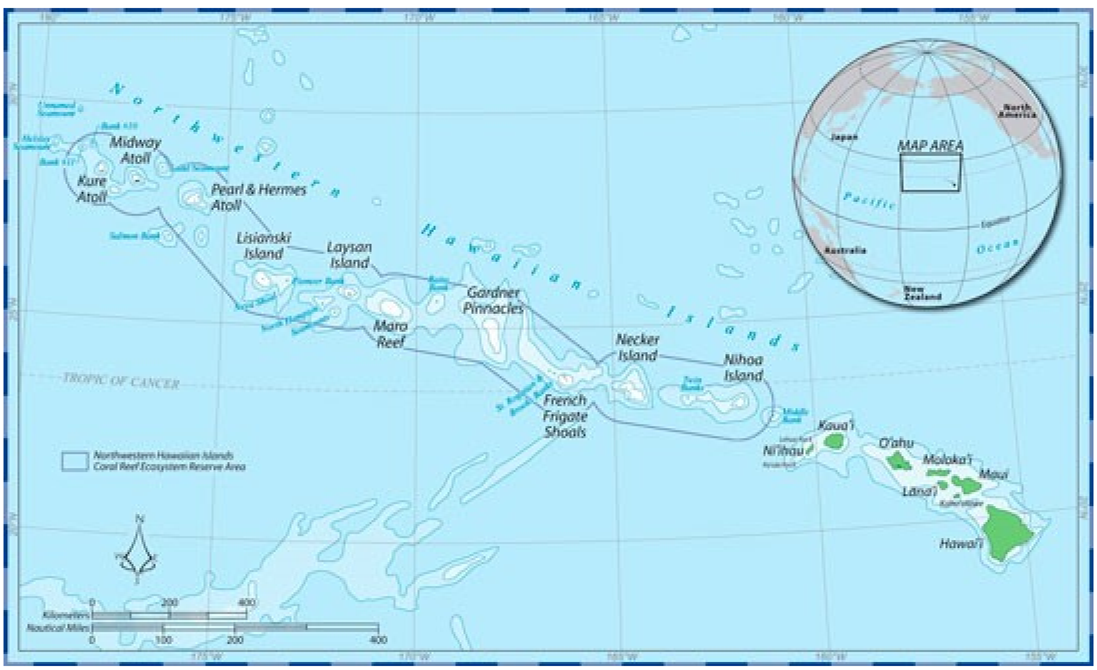


Figure 9. Map identifying designated critical habitat in the Northwest Hawaiian Islands and Main Hawaiian Islands for the endangered Hawaiian monk seal.

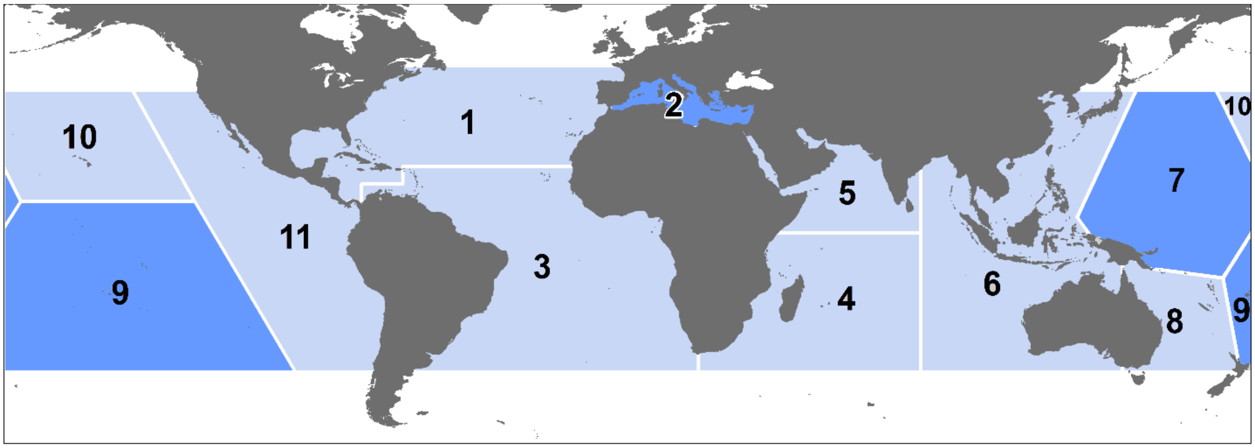
Recovery Goals

See the 2007 Final Recovery Plan for the Hawaiian monk seal for complete down listing/delisting criteria for each of the four following recovery goals.

1. Improve the survivorship of females, particularly juveniles, in sub-populations of the northwestern Hawaiian Islands.
2. Maintain the extensive field presence during the breeding season in the northwestern Hawaiian Islands.
3. Ensure the continued natural growth of the Hawaiian monk seal in the main Hawaiian Islands by reducing threats including interactions with recreational fisheries, disturbance of mother-pup pairs, disturbance of hauled out seals, and exposure to human domestic animal diseases.
4. Reduce the probability of the introduction of infectious diseases into the Hawaiian monk seal population.

8.2.3 Green Sea Turtle, North Atlantic, Central North Pacific, and Eastern Pacific DPSs

The green sea turtle is globally distributed and commonly inhabits nearshore and inshore waters, occurring throughout tropical, subtropical and, to a lesser extent, temperate waters (Figure 10).



Threatened (light blue ■) and endangered (dark blue ■) green turtle DPSs:
1. North Atlantic, 2. Mediterranean, 3. South Atlantic, 4. Southwest Indian, 5. North Indian, 6. East Indian-West Pacific, 7. Central West Pacific, 8. Southwest Pacific, 9. Central South Pacific, 10. Central North Pacific, and 11. East Pacific.

Figure 10. Map depicting range and DPS boundaries for green turtles.

The green sea turtle is the largest of the hardshell marine turtles, growing to a weight of 350 pounds (159 kilograms) and a straight carapace length of greater than 3.3 feet (1 meter) (Figure 11). The species was listed under the ESA on July 28, 1978.

Figure 11. Green sea turtle. Photos: Mark Sullivan, NOAA (left), Andy Bruckner, NOAA (right).

On April 6, 2016, NMFS listed eleven DPSs of green sea turtles under the ESA; eight as threatened and three as endangered. The DPSs considered in this opinion that occur within in the action area are: the North Atlantic, South Atlantic, Central North Pacific, and East Pacific (Table 10). The DPSs analyzed further are the North Atlantic, Central North Pacific, and East Pacific (see Section 8.1.2 above for discussion of South Atlantic DPS).

Table 10. Summary of green sea turtle listing and recovery plan information.

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Chelonia mydas	Green Turtle	North Atlantic (4 sub-populations)	Threatened	2015	81 FR 20057 04/06/2016 43 FR 32800 07/28/1978	FR N/A 1991	63 FR 46693 Puerto Rico 1998
Chelonia mydas	Green Turtle	Central North Pacific	Threatened	2015	81 FR 20057 04/06/2016 43 FR 32800 07/28/1978	N/A	None Designated
Chelonia mydas	Green Turtle	East Pacific	Threatened	2015	81 FR 20057 04/06/2016 43 FR 32800 07/28/1978	63 FR 28359 1998	None Designated

We used information available in the 2007 five-year review (NMFS and USFWS 2007a) and 2015 Status Review (Seminoff et al. 2015) to summarize the life history, population dynamics and status of the species, as follows.

Life History

Age at first reproduction for females is twenty to forty years. Green sea turtles lay an average of three nests per season with an average of 100 eggs per nest. The remigration interval (i.e., return to natal beaches) is two to five years. Nesting occurs primarily on beaches with intact dune structure, native vegetation and appropriate incubation temperatures during summer months. After emerging from the nest, hatchlings swim to offshore areas and go through a post-hatchling pelagic stage where they are believed to live for several years. During this life stage, green sea turtles feed close to the surface on a variety of marine algae and other life associated with drift lines and debris. Adult turtles exhibit site fidelity and migrate hundreds to thousands of kilometers from nesting beaches to foraging areas. Green sea turtles spend the majority of their lives in coastal foraging grounds, which include open coastlines and protected bays and lagoons. Adult green turtles feed primarily on seagrasses and algae, although they also eat jellyfish, sponges and other invertebrate prey.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes population growth rate, genetic diversity, and distribution as it relates to the green sea turtle.

Worldwide, nesting data at 464 sites indicate that 563,826 to 564,464 females nest each year (Seminoff et al. 2015). Table 11 shows by DPS the number of nesting females, nesting sites and the percentage of nesting females at the largest nesting site.

Table 11. Green sea turtle nesting abundance in each DPS (Seminoff et al. 2015).

DPS	Abundance Estimate (nesting females)	Number of Nesting Sites	Largest Nesting Site	Percentage at largest nesting site
North Atlantic	167,424	73	Tortuguero, Costa Rica	79%
Central North Pacific	3,846	12	East Island, French Frigate Shoals, Hawaii	96%
East Pacific	20,062	39	Colola, Mexico	58%

Population Growth Rate

Many nesting sites worldwide suffer from a lack of consistent, standardized monitoring, making it difficult to characterize population growth rates for a DPS. Available information on the population growth rates and trends for each of the DPSs is presented below.

8.2.3.1 North Atlantic DPS

Compared to other DPSs, the North Atlantic DPS exhibits the highest nester abundance, with approximately 167,424 females at seventy-three nesting sites and available data indicate an increasing trend in nesting. The largest nesting site in the North Atlantic DPS is in Tortuguero, Costa Rica, which hosts seventy-nine percent of nesting females for the DPS (Seminoff et al. 2015).

There are no reliable estimates of population growth rate for the DPS as a whole, but estimates have been developed at a localized level. Modeling by Chaloupka et al. (2008a) using data sets of 25 years or more show the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9 percent, and the Tortuguero, Costa Rica, population growing at 4.9 percent.

8.2.3.2 Central North Pacific DPS

There are thirteen known nesting sites for the Central North Pacific DPS, with an estimated 3,846 nesting females. The DPS is very thoroughly monitored, and it is believed there is little chance that there are undocumented nesting sites. The largest nesting site is at French Frigate Shoals, Hawaii, which hosts ninety-six percent of the nesting females for the DPS (Seminoff et al. 2015).

Nesting surveys have been conducted since 1973 for green turtles in the Central North Pacific DPS. Nesting abundance at East Island, French Frigate Shoals, increases at 4.8 percent annually.

8.2.3.3 East Pacific DPS

There are thirty-nine nesting sites for the East Pacific DPS, with an estimated 20,062 nesting females. The largest nesting site is at Colola, Mexico, which hosts fifty-eight percent of the nesting females for the DPS (Seminoff et al. 2015).

There are no estimates of population growth. Only one nesting site in the East Pacific DPS at Colola, Mexico, has sufficient long-term data to determine population trends. Data analysis indicates that the population there is increasing and is likely to continue to do so.

Genetic Diversity

Globally, the green turtle is divided into eleven DPSs; available information on the genetic diversity for each of the DPSs is presented below.

8.2.3.4 North Atlantic DPS

The North Atlantic DPS has a globally unique haplotype, which was a factor in defining the discreteness of the population for the DPS. Evidence from mitochondrial deoxyribonucleic acid (DNA) studies indicates that there are at least 4 independent nesting subpopulations in Florida, Cuba, Mexico and Costa Rica (Seminoff et al. 2015). More recent genetic analysis indicates that designating a new western Gulf of Mexico management unit might be appropriate (Shamblin et al. 2016).

8.2.3.5 Central North Pacific DPS

The majority of nesting for the Central North Pacific DPS is centered at one site on French Frigate Shoals, and there is little diversity in nesting areas. Overall, the Central North Pacific DPS has a relatively low level of genetic diversity and stock sub-structuring (Seminoff et al. 2015).

8.2.3.6 East Pacific DPS

Rare and unique haplotypes are present in the East Pacific DPS. Genetic sampling has identified four regional stocks for this DPS: Revillagigedos Archipelago, Mexico; Michoacán, Mexico; Central America (Costa Rica); and Galápagos Islands, Ecuador (Seminoff et al. 2015).

Distribution

The green sea turtle occupies the coastal waters of over 140 countries worldwide; nesting occurs in more than eighty countries. The green sea turtle is distributed in tropical, subtropical, and to a lesser extent, temperate waters (Figure 12).

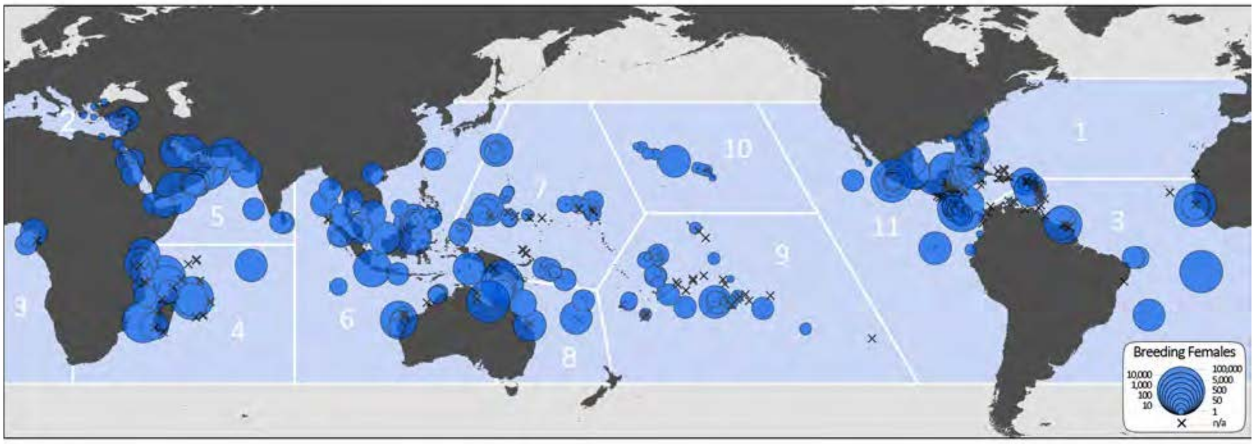


Figure 12. Map of all *Chelonia mydas* nesting sites indicating delineation of DPSs (Seminoff et al. 2015).

8.2.3.7 North Atlantic DPS

Green turtles from the North Atlantic DPS range from the boundary of South and Central America (7.5°N, 77°W) in the south, throughout the Caribbean, the Gulf of Mexico, and the U.S. Atlantic coast to New Brunswick, Canada (48°N, 77°W) in the north. The range of the DPS then extends due east along latitudes 48°N and 19°N to the western coasts of Europe and Africa Figure 12). Nesting occurs primarily in Costa Rica, Mexico, Florida and Cuba (Figure 13).

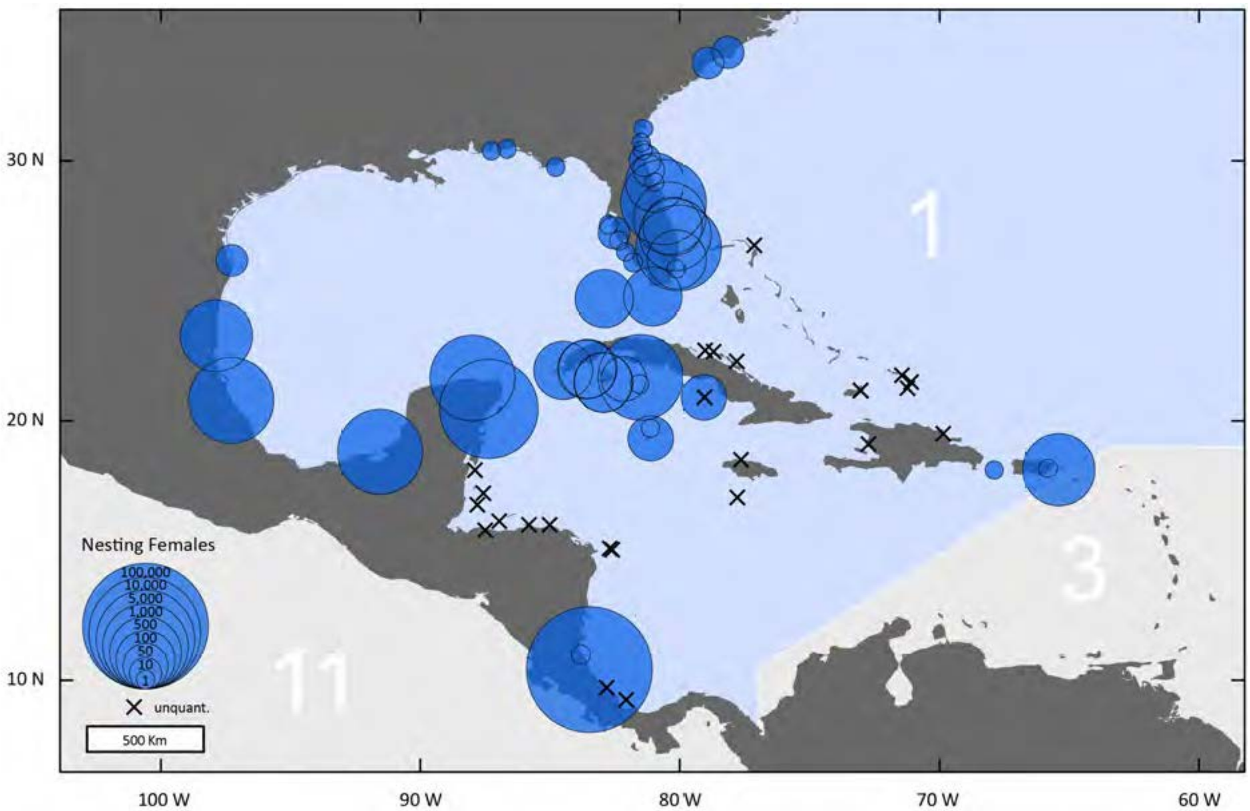


Figure 13. Close up of nesting distribution of green turtles in the western North Atlantic DPS (water body labeled '1'). Size of circles indicates estimated nester abundance. Locations marked with an 'x' indicate sites lacking abundance information (Seminoff et al. 2015).

The green turtle status review team identified 73 nesting sites within the North Atlantic DPS although some represent numerous individual beaches (Seminoff et al. 2015). The four regions that support high density nesting concentrations are Costa Rica (Tortuguero), Mexico (Campeche, Yucatan, and Quintana Roo), United States (Florida), and Cuba. Of the 73 nesting sites in the DPS, 16 nests were found in Texas from 2011 to 2012 (Seminoff et al. 2015). Once juveniles reach a certain age, they leave pelagic habitat and reside in nearshore foraging grounds (Troëng et al. 2005) which includes the area of South Padre Island, Port Isabel, and the Brownsville ship channel along the Texas coast. The presence of green sea turtles in and around the Brownsville ship channel are documented (Gorga 2010; NMFS 2014a; U.S. Army Corps of Engineers 2014). Figure 14 shows 128 individual green sea turtles in and around the Brownsville ship channel according to Sea Turtle, Inc. data from 2009-2017. Twelve additional turtles are also in this same geographic area, without the Global Positioning System (GPS) coordinates of their exact location. Data for 2018 totals fifty-two additional turtles to date; GPS coordinates for these turtles have not yet been provided.

* Dots that aren't visible because they are clustered are indicated with text boxes and arrows. All the individual turtles indicated at each location are alive.

Figure 14. Locations of green sea turtle stranding data in and around the Port of Brownsville ship channel from 2009 to 2017 (M. Devlin, Sea Turtle, Inc., pers. comm. to NMFS, February 26, 2018).

8.2.3.8 Central North Pacific DPS

Green turtles in the Central North Pacific DPS are found in the Hawaiian Archipelago and Johnston Atoll. The major nesting site for the DPS is at East Island, French Frigate Shoals, in the Northwestern Hawaiian Islands; lesser nesting sites are found throughout the Northwestern Hawaiian Islands and the Main Hawaiian Islands (Figure 15).

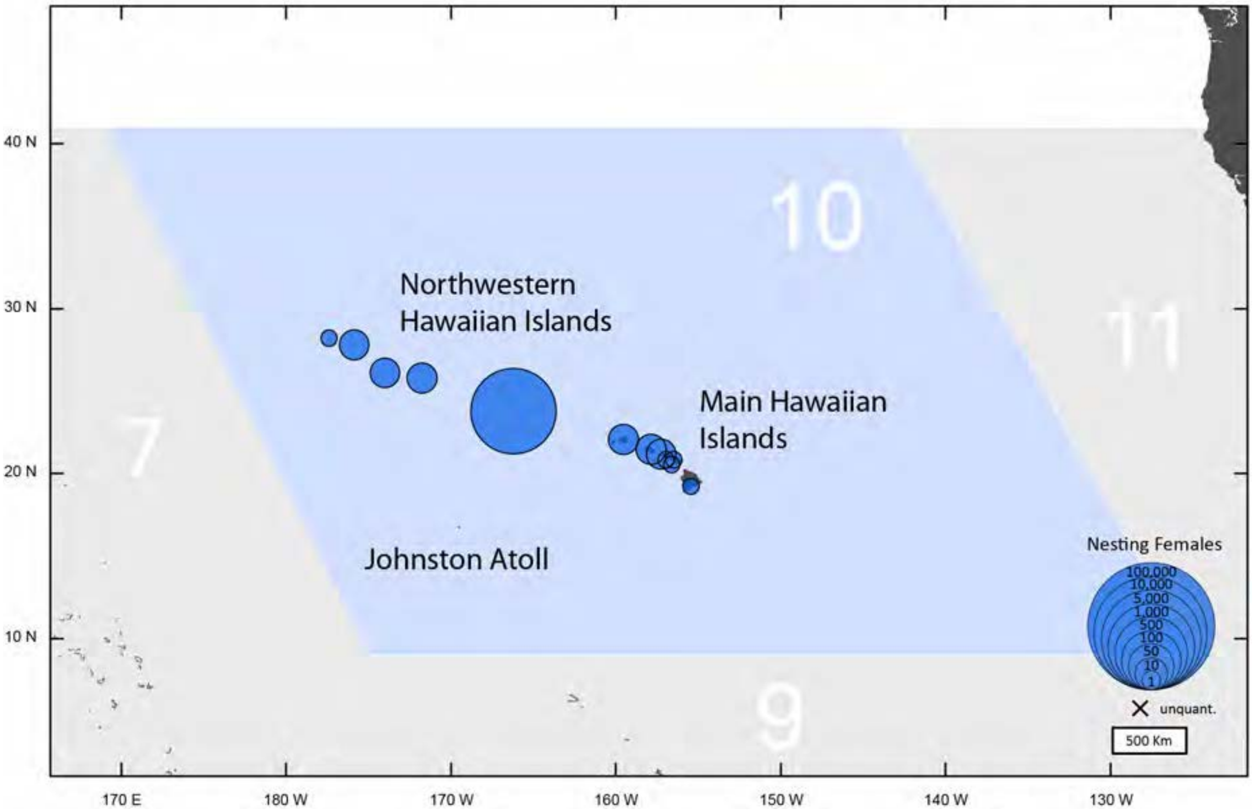


Figure 15. Nesting distribution of green turtles in the Central North Pacific DPS (water body labeled '10'). Size of circles indicates estimated nester abundance (Seminoff et al. 2015).

8.2.3.9 East Pacific DPS

Green turtles in the East Pacific DPS are found from the California/Oregon border south to central Chile (Figure 16). Major nesting sites occur at Michoacán, Mexico, and the Galápagos Islands, Ecuador. Smaller nesting sites are found on the Pacific coast of Costa Rica, and in the Revillagigedos Archipelago, Mexico. Scattered nesting occurs in Columbia, Ecuador, Guatemala and Peru (Seminoff et al. 2015).

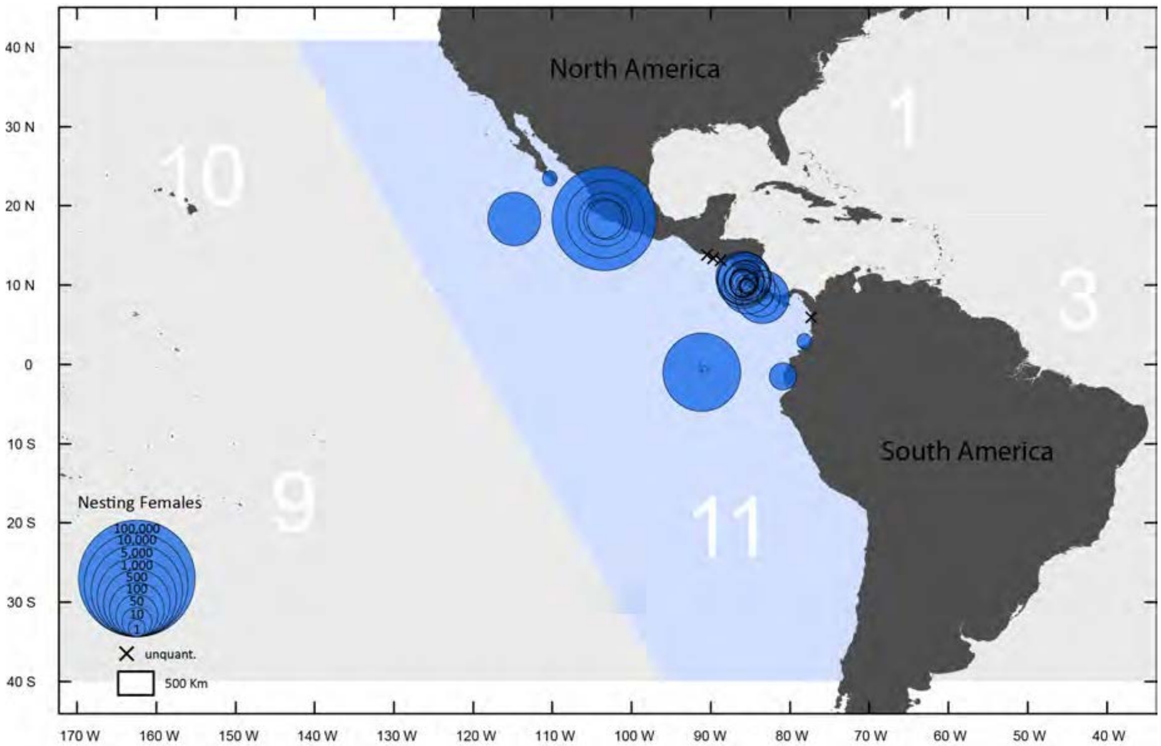


Figure 16. Nesting distribution of green turtles in the East Pacific DPS (water body labeled '11'). Size of circles indicates estimated nester abundance. Locations marked with an 'x' indicate sites lacking abundance information (Seminoff et al. 2015).

Status

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

8.2.3.10 North Atlantic DPS

Historically, green turtles in the North Atlantic DPS were hunted for food, which was the principle cause of the population's decline. Apparent increases in nester abundance for the North Atlantic DPS in recent years are encouraging but must be viewed cautiously, as the datasets represent a fraction of a green sea turtle generation, up to 50 years. While the threats of pollution, habitat loss through coastal development, beachfront lighting, and fisheries bycatch continue, the North Atlantic DPS appears to be somewhat resilient to future perturbations.

8.2.3.11 Central North Pacific DPS

Green turtles in the Hawaiian Archipelago were subjected to hunting pressure for subsistence and commercial trade, which was largely responsible for the decline in the region. Though the practice has been banned, there are still anecdotal reports of harvest. Incidental bycatch in fishing gear, ingestion of marine debris, and the loss of nesting habitat due to sea level rise are current threats to the population. Although these threats persist, the increase in annual nesting abundance, continuous scientific monitoring, legal enforcement and conservation programs are all factors that favor the resiliency of the DPS.

8.2.3.12 East Pacific DPS

The population decline for the East Pacific DPS was primarily caused by commercial harvest of green turtles for subsistence and other uses (e.g., sea turtle oil as a cold remedy). Conservation laws are in place in several countries across the range of the DPS, but enforcement is inconsistent, limiting effectiveness. Incidental bycatch in commercial fishing gear, continued harvest, coastal development and beachfront lighting are all continuing threats for the DPS. The observed increases in nesting abundance for the largest nesting aggregation in the region (Michocán, Mexico), a stable trend at Galápagos, and record high numbers at sites in Costa Rica suggest that the population is resilient, particularly in Mexico.

Designated Critical Habitat

8.2.3.13 North Atlantic DPS

On September 2, 1998, NMFS designated critical habitat for green sea turtles, which include coastal waters surrounding Culebra Island, Puerto Rico. Seagrass beds surrounding Culebra provide important foraging resources for juvenile, subadult and adult green sea turtles. Additionally, coral reefs surrounding the island provide resting shelter and protection from predators. This area provides important developmental habitat for the species. Activities that may affect the critical habitat include beach renourishment, dredge and fill activities, coastal construction, and freshwater discharge. The designated critical habitat for the North Atlantic DPS of green sea turtle is not located within the action area.

Recovery Goals

See the 1998 and 1991 recovery plans for the Pacific, East Pacific and Atlantic populations of green turtles for complete down-listing/delisting criteria for recovery goals for the species (NMFS and USFWS 1991; NMFS and USFWS 1998b). Broadly, recovery plan goals emphasize the need to protect and manage nesting and marine habitat, protect and manage populations on nesting beaches and in the marine environment, increase public education, and promote international cooperation on sea turtle conservation topics.

8.2.4 Hawksbill Sea Turtle

The hawksbill turtle has a circumglobal distribution throughout tropical and, to a lesser extent, subtropical oceans (Figure 17).

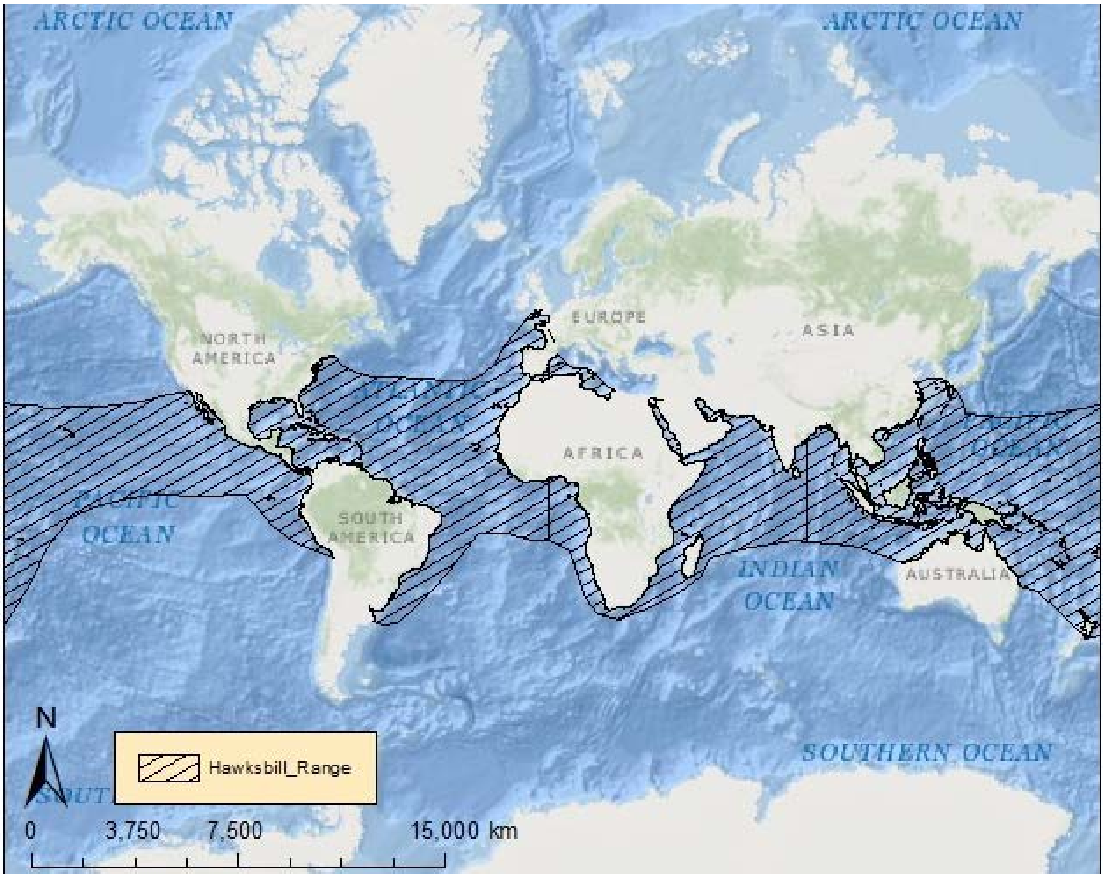


Figure 17. Map identifying the range of the endangered hawksbill sea turtle.

The hawksbill sea turtle has a sharp, curved, beak-like mouth and a "tortoiseshell" pattern on its carapace, with radiating streaks of brown, black, and amber (Figure 18).

Figure 18. Hawksbill sea turtle. Photos: Tom Moore, NOAA (left), Jordan Wilkerson (right).

The species was first listed under the Endangered Species Conservation Act and listed as endangered under the ESA since 1973 (Table 12).

Table 12. Summary of hawksbill sea turtle listing and recovery plan information.

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Eretmochelys imbricata	Hawksbill Turtle	None Designated	Endangered	2013	35 FR 8491 6/2/1970	57 FR 38818 Atlantic 1992	63 FR 46693 Puerto Rico 1998
Eretmochelys imbricata	Hawksbill Turtle	None Designated	Endangered	2013	35 FR 8491 6/2/1970	63 FR 28359 Pacific 1998	None Designated

We used information available in the 2007 and 2013 five-year reviews (NMFS and USFWS 2007b; NMFS and USFWS 2013a) to summarize the life history, population dynamics and status of the species, as follows.

Life History

Hawksbill sea turtles reach sexual maturity at twenty to forty years of age. Females return to their natal beaches every two to five years to nest and nest an average of three to five times per season. Clutch sizes are large (up to 250 eggs). Sex determination is temperature dependent, with warmer incubation producing more females. Hatchlings migrate to and remain in pelagic habitats until they reach approximately twenty two to twenty five centimeters in straight carapace length. As juveniles, they take up residency in coastal waters to forage and grow. As adults, hawksbills use their sharp beak-like mouths to feed on sponges and corals. Hawksbill sea turtles are highly migratory and use a wide range of habitats during their lifetimes (Musick and Limpus 1997; Plotkin 2003). Satellite tagged turtles have shown significant variation in movement and migration patterns. Distance traveled between nesting and foraging locations ranges from a few hundred to a few thousand kilometers (Horrocks et al. 2001; Miller et al. 1998).

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes population growth rate, genetic diversity, and distribution as it relates to the hawksbill sea turtle.

Population Growth Rate

Surveys at eighty eight nesting sites worldwide indicate that 22,004 to 29,035 females nest annually (NMFS and USFWS 2013a). In general, hawksbills are doing better in the Atlantic and

Indian Ocean than in the Pacific Ocean, where despite greater overall abundance, a greater proportion of the nesting sites are declining.

From 1980 to 2003, the number of nests at three primary nesting beaches (Rancho Nuevo, Tepehuajes, and Playa Dos) increased fifteen percent annually (Heppell et al. 2005); however, due to recent declines in nest counts, decreased survival at other life stages, and updated population modeling, this rate is not expected to continue (NMFS and USFWS 2013a).

Genetic Diversity

Populations are distinguished generally by ocean basin and more specifically by nesting location. Our understanding of population structure is relatively poor. Genetic analysis of hawksbill sea turtles foraging off the Cape Verde Islands identified three closely-related haplotypes in a large majority of individuals sampled that did not match those of any known nesting population in the western Atlantic, where the vast majority of nesting has been documented (McClellan et al. 2010; Monzón-Argüello et al. 2010). Hawksbills in the Caribbean seem to have dispersed into separate populations (rookeries) after a bottleneck roughly 100,000 to 300,000 years ago (Leroux et al. 2012).

Distribution

The hawksbill has a circumglobal distribution throughout tropical and, to a lesser extent, subtropical waters of the Atlantic, Indian, and Pacific Oceans. In their oceanic phase, juvenile hawksbills can be found in Sargassum mats; post-oceanic hawksbills may occupy a range of habitats that include coral reefs or other hard-bottom habitats, sea grass, algal beds, mangrove bays and creeks (Bjorndal and Bolten 2010; Musick and Limpus 1997).

Status

Long-term data on the hawksbill sea turtle indicate that sixty-three sites have declined over the past twenty to one hundred years (historic trends are unknown for the remaining twenty-five sites). Recently, twenty-eight sites (sixty-eight percent) have experienced nesting declines, ten have experienced increases, three have remained stable, and forty-seven have unknown trends. The greatest threats to hawksbill sea turtles are overharvesting of turtles and eggs, degradation of nesting habitat, and fisheries interactions. Adult hawksbills are harvested for their meat and carapace, which is sold as tortoiseshell. Eggs are taken at high levels, especially in southeast Asia where collection approaches one hundred percent in some areas. In addition, lights on or adjacent to nesting beaches are often fatal to emerging hatchlings and alters the behavior of nesting adults. The species’ resilience to additional perturbation is low.

Designated Critical Habitat

On September 2, 1998, NMFS established critical habitat for hawksbill sea turtles around Mona and Monito Islands, Puerto Rico. 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. The designated critical habitat for the hawksbill sea turtle is not located within the action area.

Recovery Goals

See the 1992 Recovery Plan for the U.S. Caribbean, Atlantic and Gulf of Mexico (NMFS and USFWS 1993) and the 1998 Recovery Plan for the U.S. Pacific populations (NMFS and USFWS 1998c) of hawksbill sea turtles, for complete down listing/delisting criteria for each of their respective recovery goals. The following items were the top recovery actions identified to support in the Recovery Plans:

1. Identify important nesting beaches.
2. Ensure long-term protection and management of important nesting beaches.
3. Protect and manage nesting habitat; prevent the degradation of nesting habitat caused by seawalls, revetments, sand bags, other erosion-control measures, jetties and breakwaters.
4. Identify important marine habitats; protect and manage populations in marine habitat.
5. Protect and manage marine habitat; prevent the degradation or destruction of important [marine] habitats caused by upland and coastal erosion.
6. Prevent the degradation of reef habitat caused by sewage and other pollutants.
7. Monitor nesting activity on important nesting beaches with standardized index surveys.
8. Evaluate nest success and implement appropriate nest-protection on important nesting beaches.
9. Ensure that law-enforcement activities prevent the illegal exploitation and harassment of sea turtles and increase law-enforcement efforts to reduce illegal exploitation.
10. Determine nesting beach origins for juveniles and subadult populations.

8.2.5 Kemp's Ridley Sea Turtle

The Kemp's ridley turtle is considered to be the most endangered sea turtle, internationally (Groombridge 1982; Zwinenberg 1977). Its range extends from the Gulf of Mexico to the Atlantic coast, with nesting beaches limited to a few sites in Mexico and Texas (Figure 19).

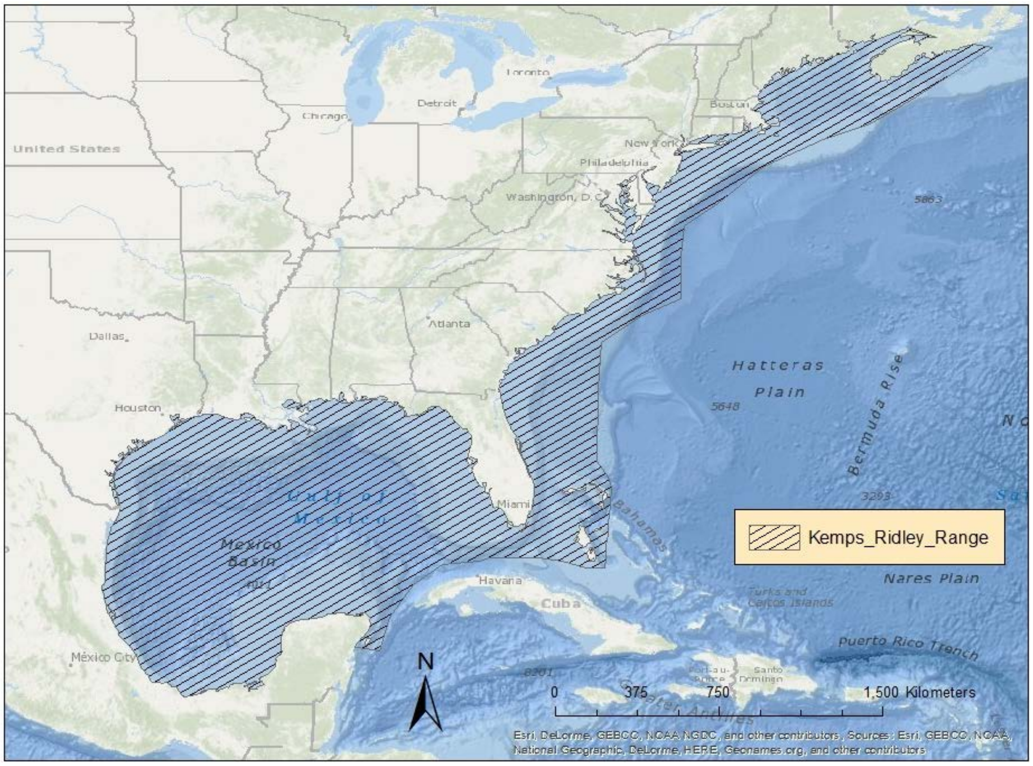


Figure 19. Map identifying the range of the Kemp's ridley sea turtle.

Kemp's ridley sea turtles are the smallest of all sea turtle species, with a nearly circular top shell and a pale yellowish bottom shell (Figure 20).

Figure 20. Kemp's ridley sea turtle. Photos: National Oceanic and Atmospheric Administration (left), National Park Service (right).

The species was first listed under the Endangered Species Conservation Act and listed as endangered under the ESA since 1970 (Table 13).

Table 13. Summary of Kemp's ridley sea turtle listing and recovery plan information.

Lepidochelys kempii	Kemp's Ridley Turtle	None Designated	Endangered	2015	35 FR 18319 12/02/1970	75 FR 12496 U.S. Caribbean, Atlantic, and Gulf of Mexico 2011	None Designated

We used information available in the revised recovery plan (NMFS and USFWS 2011) and the 2015 five-year review (NMFS and USFWS 2015) to summarize the life history, population dynamics and status of the species, as follows.

Life History

Females mature at twelve years of age. The average remigration is two years. Nesting occurs from April to July in large arribadas, primarily at Rancho Nuevo, Mexico. Females lay an average of 2.5 clutches per season. The annual average clutch size is ninety-seven to one hundred eggs per nest. The nesting location may be particularly important because hatchlings can more easily migrate to foraging grounds in deeper oceanic waters, where they remain for approximately two years before returning to nearshore coastal habitats. Juvenile Kemp's ridley sea turtles use these nearshore coastal habitats from April through November, but move towards more suitable overwintering habitat in deeper offshore waters (or more southern waters along the Atlantic coast) as water temperature drops. Adult habitat largely consists of sandy and muddy areas in shallow, nearshore waters less than 120 feet (37 meters) deep, although they can also be found in deeper offshore waters. As adults, Kemp's ridleys forage on swimming crabs, fish, jellyfish, mollusks, and tunicates (NMFS and USFWS 2011).

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the Kemp's ridley sea turtle.

Population Growth Rate

Of the sea turtles species in the world, the Kemp's ridley has declined to the lowest population level. Nesting aggregations at a single location (Rancho Nuevo, Mexico) were estimated at 40,000 females in 1947. By the mid-1980s, the population had declined to an estimated 300 nesting females. In 2014, there were an estimated 10,987 nests and 519,000 hatchlings released from three primary nesting beaches in Mexico (NMFS and USFWS 2015). The number of nests in Padre Island, Texas has increased over the past two decades, with one nest observed in 1985, four in 1995, fifty in 2005, 197 in 2009, and 119 in 2014 (NMFS and USFWS 2015).

From 1980 to 2003, the number of nests at three primary nesting beaches (Rancho Nuevo, Tepehuajes, and Playa Dos) increased fifteen percent annually (Heppell et al. 2005); however, due to recent declines in nest counts, decreased survival at other life stages, and updated population modeling, this rate is not expected to continue (NMFS and USFWS 2015).

Genetic Diversity

Genetic variability in Kemp’s ridley turtles is considered to be high, as measured by heterozygosis at microsatellite loci (NMFS and USFWS 2011). Additional analysis of the mitochondrial DNA taken from samples of Kemp’s ridley turtles at Padre Island, Texas, showed six distinct haplotypes, with one found at both Padre Island and Rancho Nuevo (Dutton et al. 2006).

Distribution

The Kemp's ridley occurs in the Gulf of Mexico and along the U.S. Atlantic coast (TEWG 2000). Kemp’s ridley sea turtles have occasionally been found in the Mediterranean Sea, which may be due to migration expansion or increased hatchling production (Tomás and Raga 2008). The vast majority of individuals stem from breeding beaches at Rancho Nuevo on the Gulf of Mexico coast of Mexico. During spring and summer, juvenile Kemp’s ridleys occur in the shallow coastal waters of the northern Gulf of Mexico from south Texas to north Florida. In the fall, most Kemp’s ridleys migrate to deeper or more southern, warmer waters and remain there through the winter (Schmid 1998). As adults, many turtles remain in the Gulf of Mexico, with only occasional occurrence in the Atlantic Ocean (NMFS and USFWS 2011).

Designated Critical Habitat

No critical habitat has been designated for Kemp’s ridley sea turtles.

Recovery Goals

See the 2011 Final Bi-National (United States and Mexico) Revised Recovery Plan for Kemp’s ridley sea turtles for complete down listing/delisting criteria for each of their respective recovery goals (NMFS and USFWS 2011). The following items were identified as priorities to recover Kemp’s ridley sea turtles:

1. Protect and manage nesting and marine habitats.
2. Protect and manage populations on the nesting beaches and in the marine environment.
3. Maintain a stranding network.

4. Manage captive stocks.
5. Sustain education and partnership programs.
6. Maintain, promote awareness of and expand United States and Mexican laws.
7. Implement international agreements.
8. Enforce laws.

8.2.6 Leatherback Sea Turtle and Designated Critical Habitat

The leatherback sea turtle is unique among sea turtles for its large size, wide distribution (due to thermoregulatory systems and behavior), and lack of a hard, bony carapace. It ranges from tropical to subpolar latitudes, worldwide (Figure 21).

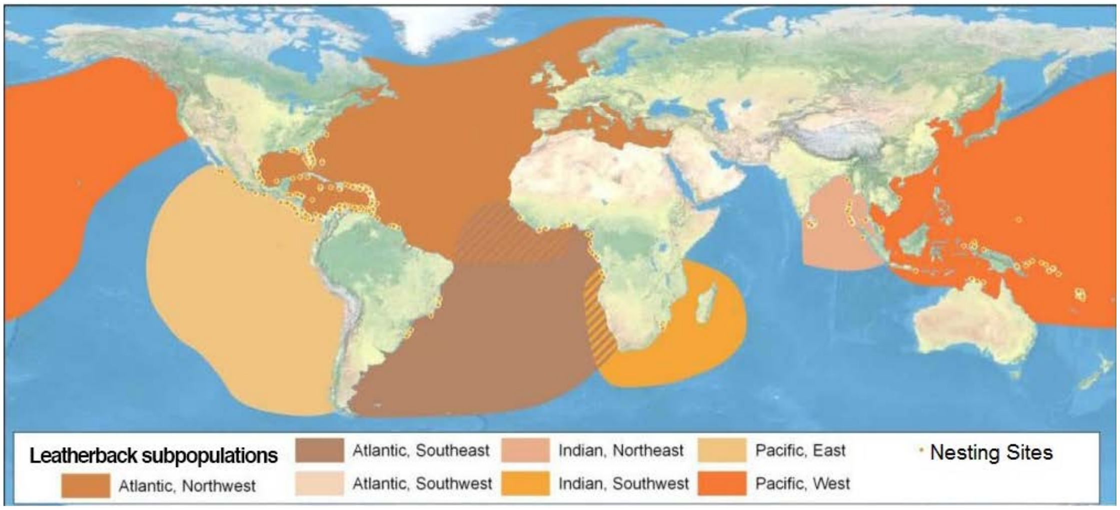


Figure 21. Map identifying the range of the leatherback sea turtle with the seven subpopulations and nesting sites. Adapted from (Wallace et al. 2010a).

Leatherbacks are the largest living turtle, reaching lengths of six feet long, and weighing up to one ton. Leatherback sea turtles have a distinct black leathery skin covering their carapace with pinkish white skin on their belly (Figure 22).

Figure 22. Leatherback sea turtle adult and hatchling. Photos: R. Tapilatu (left), N. Pilcher (right).

The species was first listed under the Endangered Species Conservation Act and listed as endangered under the ESA since 1970 (Table 14).

Table 14. Summary of leatherback sea turtle listing and recovery plan information.

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Dermochelys coriacea	Leatherback Turtle	None Designated	Endangered	2013	35 FR 8491 6/2/1970	FR N/A Atlantic 1992	44 FR 17710 U.S. Virgin Islands 1979
Dermochelys coriacea	Leatherback Turtle	None Designated	Endangered	2013	35 FR 8491 6/2/1970	63 FR 28359 Pacific 1998	77 FR 4170 Pacific 2012

We used information available in the 2013 five-year review (NMFS and USFWS 2013b) and the critical habitat designations to summarize the life history, population dynamics and status of the species, as follows.

Life History

Age at maturity has been difficult to ascertain, with estimates ranging from five to twenty-nine years (Avens et al. 2009; Spotila et al. 1996). Females lay up to seven clutches per season, with more than sixty-five eggs per clutch and eggs weighing greater than eighty grams (Reina et al. 2002; Wallace et al. 2007). The number of leatherback hatchlings that make it out of the nest on to the beach (i.e., emergent success) is approximately fifty percent worldwide (Eckert et al. 2012). Females nest every one to seven years. Natal homing, at least within an ocean basin, results in reproductive isolation between five broad geographic regions: eastern and western Pacific, eastern and western Atlantic, and Indian Ocean. Leatherback sea turtles migrate long, transoceanic distances between their tropical nesting beaches and the highly productive temperate waters where they forage, primarily on jellyfish and tunicates. These gelatinous prey are relatively nutrient-poor, such that leatherbacks must consume large quantities to support their body weight. Leatherbacks weigh about thirty-three 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. 2005; Wallace et al. 2006). Sea turtles must meet an energy threshold before returning to nesting beaches. Therefore, their remigration intervals (the time between nesting) are dependent upon foraging success and duration (Hays 2000; Price et al. 2004).

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes abundance, population growth rate, genetic diversity, and spatial distribution as it relates to the leatherback sea turtle.

Population Growth Rate

Leatherbacks are globally distributed, with nesting beaches in the Pacific, Atlantic, and Indian oceans. Detailed population structure is unknown, but is likely dependent upon nesting beach location. Based on estimates calculated from nest count data, there are between 34,000 and 94,000 adult leatherbacks in the North Atlantic (TEWG 2007). In contrast, leatherback populations in the Pacific are much lower. Overall, Pacific populations have declined from an estimated 81,000 individuals to less than 3,000 total adults and subadults (Spotila et al. 2000). Deriving abundance estimates from nest counts divided by average clutch frequency from (NMFS and USFWS 2013b) give as West Pacific subpopulation estimate of 562 nesting females. Population abundance in the Indian Ocean is difficult to assess due to lack of data and inconsistent reporting. Available data from southern Mozambique show that approximately ten females nest per year from 1994 to 2004, and about 296 nests per year counted in South Africa (NMFS and USFWS 2013b).

Population growth rates for leatherback sea turtles vary by ocean basin. Counts of leatherbacks at nesting beaches in the western Pacific indicate that the subpopulation has been declining at a rate of almost six percent per year since 1984 (Tapilatu et al. 2013). Leatherback subpopulations in the Atlantic Ocean, however, are showing signs of improvement. Nesting females in South Africa are increasing at an annual rate of four to 5.6 percent, and from nine to thirteen percent in Florida and the U.S. Virgin Islands (TEWG 2007), believed to be a result of conservation efforts.

Genetic Diversity

Analyses of mitochondrial DNA from leatherback sea turtles indicates a low level of genetic diversity, pointing to possible difficulties in the future if current population declines continue (Dutton et al. 1999). Further analysis of samples taken from individuals from rookeries in the Atlantic and Indian oceans suggest that each of the rookeries represent demographically independent populations (NMFS and USFWS 2013b).

Distribution

Leatherback sea turtles are distributed in oceans throughout the world. Leatherbacks occur throughout marine waters, from nearshore habitats to oceanic environments (Shoop and Kenney 1992). Movements are largely dependent upon reproductive and feeding cycles and the oceanographic features that concentrate prey, such as frontal systems, eddy features, current boundaries, and coastal retention areas (Benson et al. 2011).

Status

The leatherback sea turtle is an endangered species whose once large nesting populations have experienced steep declines in recent decades. The primary threats to leatherback sea turtles include fisheries bycatch, harvest of nesting females, and egg harvesting. Because of these threats, once large rookeries are now functionally extinct, and there have been range-wide reductions in population abundance. Other threats include loss of nesting habitat due to development, tourism, and sand extraction. 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. Plastic ingestion is common in leatherbacks and can block gastrointestinal tracts leading to death. Climate change may alter sex ratios (as temperature determines hatchling sex), range (through expansion of foraging habitat), and habitat (through the loss of nesting beaches, because of sea-level rise. The species' resilience to additional perturbation is low.

Designated Critical Habitat

On March 23, 1979, leatherback critical habitat was identified adjacent to Sandy Point, St. Croix, Virgin Islands from the 183 meter isobath to mean high tide level between 17° 42'12" N and 65°50'00" W (Figure 23). 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. The designated critical habitat is within the Sandy Point National Wildlife Refuge.

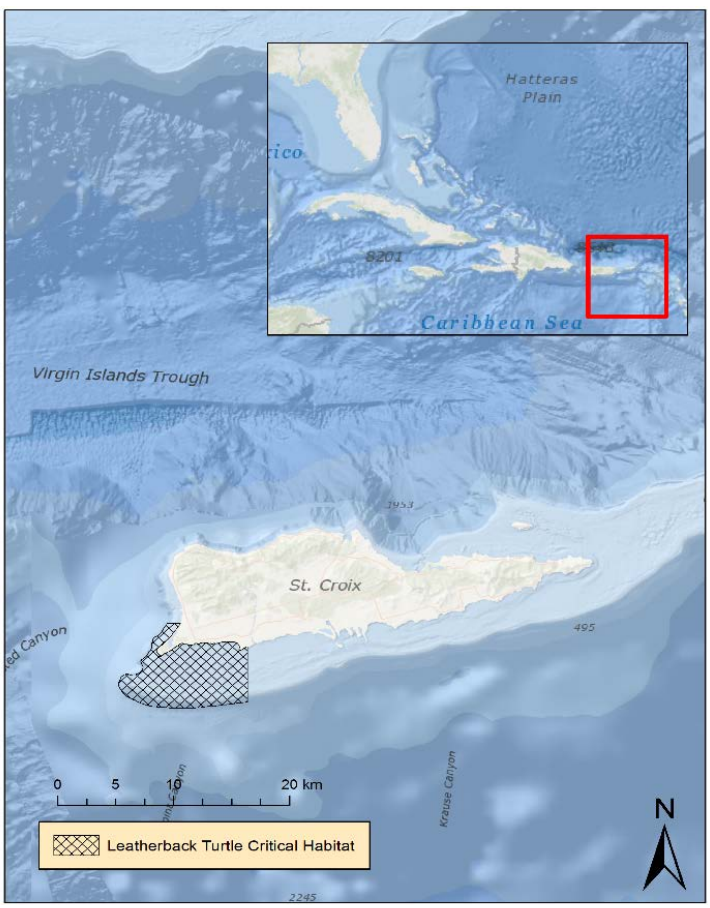


Figure 23. Map depicting leatherback sea turtle designated critical habitat in the U.S. Virgin Islands.

On January 20, 2012, NMFS issued a final rule to designate additional critical habitat for the Pacific leatherback sea turtle. This designation includes approximately 43,798 square kilometers stretching along the California coast from Point Arena to Point Arguello east of the 3000 m depth contour; and 64,760 square kilometers stretching from Cape Flattery, Washington to Cape Blanco, Oregon east of the 2,000 meters depth contour (Figure 24). The designated areas comprise approximately 108,558 square kilometers of marine habitat and include waters from the ocean surface down to a maximum depth of 80 meters. They were designated specifically because of the occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae (i.e., jellyfish), of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks.

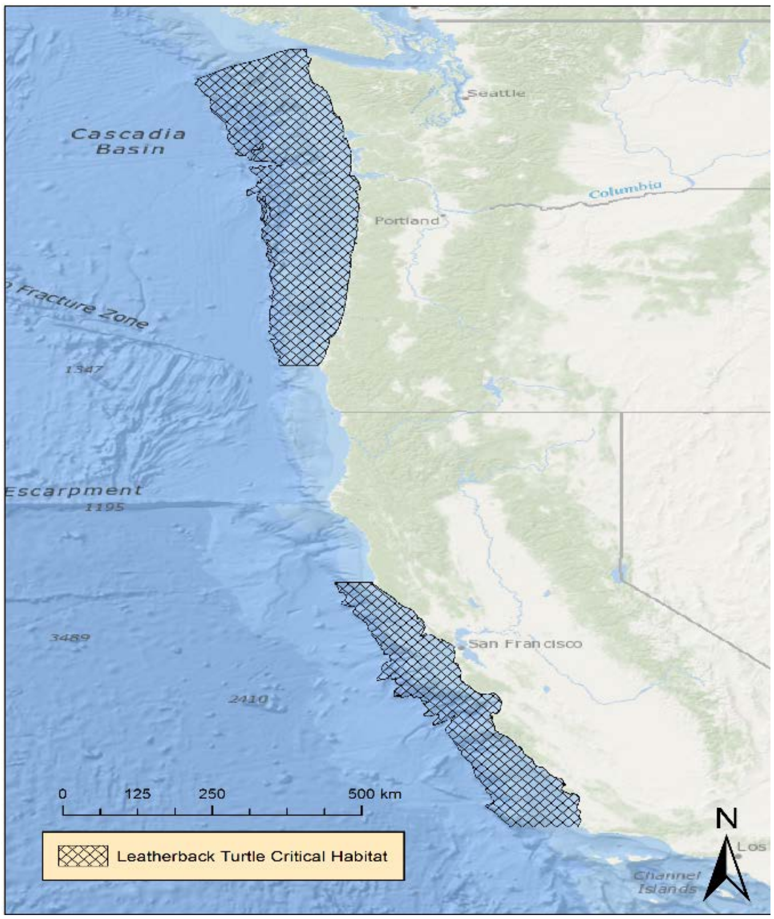


Figure 24. Map depicting leatherback sea turtle designated critical habitat along the U.S. Pacific coast.

Recovery Goals

See the U.S. Pacific (NMFS and USFWS 1998a) and U.S. Caribbean, Gulf of Mexico and Atlantic Recovery Plans (NMFS and USFWS 1992) for leatherback sea turtles for complete down listing/delisting criteria for each of their respective recovery goals. The following items were the top five recovery actions identified to support in the Leatherback Five Year Action Plan:

1. Reduce fisheries interactions.
2. Improve nesting beach protection and increase reproductive output.
3. International cooperation.
4. Monitoring and research.
5. Public engagement.

8.2.7 Loggerhead Sea Turtle, Northwest Atlantic Ocean DPS and Designated Critical Habitat

Loggerhead sea turtles are circumglobal, and are found in the temperate and tropical regions of the Indian, Pacific and Atlantic Oceans (Figure 25).

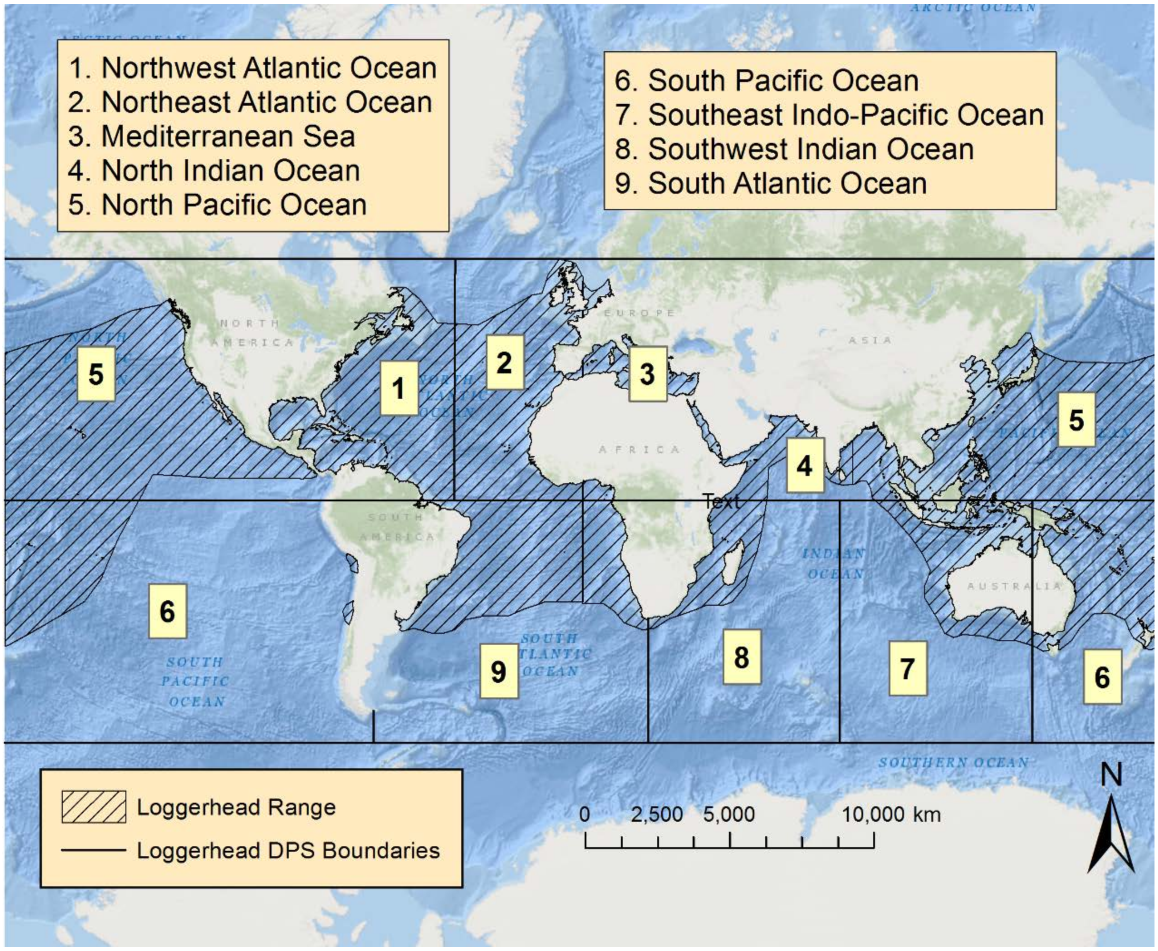


Figure 25. Map identifying the range and DPS boundaries of the loggerhead sea turtle.

The loggerhead sea turtle is distinguished from other turtles by its large head and powerful jaws (Figure 26).

Figure 26. Loggerhead sea turtle. Photos: National Oceanic and Atmospheric Administration (left), Marco Giuliano, Fondazione Cetacea (right).

The species was first listed as threatened under the ESA in 1978. On September 22, 2011, the NMFS designated nine DPSs of loggerhead sea turtles: South Atlantic Ocean and Southwest Indian Ocean as threatened as well as Mediterranean Sea, North Indian Ocean, North Pacific Ocean, Northeast Atlantic Ocean, Northwest Atlantic Ocean, South Pacific Ocean, and Southeast Indo-Pacific Ocean as endangered. The DPS considered in this opinion that occurs within in the action area is the Northwest Atlantic Ocean (Table 15).

We used information available in the 2009 status review (Conant et al. 2009) and the final listing rule to summarize the life history, population dynamics and status of the species, as follows.. Recent ocean-basin scale genetic analysis supports this conclusion, with additional differentiation apparent based upon nesting beaches (Shamblin et al. 2014).

Table 15. Summary of loggerhead sea turtle listing and recovery plan information.

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Caretta caretta	Loggerhead Turtle	Northwest Atlantic Ocean	Threatened	2009	76 FR 58868 09/22/2011 43 FR 32800 07/28/1978	74 FR 2995 2008	79 FR 39856 Atlantic/Gulf of Mexico 2014

Life History

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

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes abundance, population growth rate, genetic diversity, and distribution as it relates to the loggerhead sea turtle.

There is general agreement that the number of nesting females provides a useful index of the species' population size and stability at this life stage, even though there are doubts about the ability to estimate the overall population size (Bjorndal et al. 2005). Adult nesting females often account for less than one percent of total population numbers. The global abundance of nesting female loggerhead turtles is estimated at 43,320 to 44,560 (Spotila 2004). Abundance estimates for the loggerhead sea turtle Northwest Atlantic Ocean DPS and recovery units (prior to the designation of DPSs) are found in Table 16.

Table 16. Loggerhead sea turtle abundance estimates for the Northwest Atlantic Ocean DPS and recovery units.

DPS	Abundance Estimate	Citations
Northwest Atlantic Ocean	53,000 to 92,000 nests annually	(NMFS SEFSC 2009)
Northern Recovery Unit	5,215 nests, on average; 1989 to 2008	(NMFS and USFWS 2008)
Peninsular Florida Recovery Unit	>10,000 nesting females annually	(Ehrhart et al. 2003)
Greater Caribbean Recovery Unit	903 to 2,700 nests annually	(Ehrhart et al. 2003; NMFS and USFWS 2008; Zurita et al. 2003)
Dry Tortugas Recovery Unit	246 nests annually	(NMFS and USFWS 2007c)
Gulf of Mexico Recovery Unit	100 to 999 nesting females annually	(NMFS SEFSC 2009)

Population Growth Rate

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

8.2.7.1 Northwest Atlantic Ocean DPS

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

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

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

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

The Greater Caribbean Recovery Unit encompasses nesting subpopulations in Mexico to French Guiana, the Bahamas, and the Lesser and Greater Antilles. The majority of nesting for this recovery unit occurs on the Yucatán peninsula, in Quintana Roo, Mexico, with 903 to 2,331 nests annually (Zurita et al. 2003). Other significant nesting sites are found throughout the Caribbean, and including Cuba, with approximately 250 to 300 nests annually (Ehrhart et al. 2003) and over one hundred nests annually in Cay Sal in the Bahamas (NMFS and USFWS 2008).

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

Genetic Diversity

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

8.2.7.2 Northwest Atlantic Ocean DPS

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

Distribution

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

8.2.7.3 Northwest Atlantic Ocean DPS

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

Status

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

8.2.7.4 Northwest Atlantic Ocean DPS

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

Designated Critical Habitat

8.2.7.5 Northwest Atlantic Ocean DPS

NMFS has designated critical habitat for the Northwest Atlantic Ocean DPS loggerhead sea turtles. On July 10, 2014, NMFS and FWS designated critical habitat for the Northwest Atlantic Ocean DPS loggerhead sea turtles along the U.S. Atlantic and Gulf of Mexico coasts from North Carolina to Mississippi. These areas contain one or a combination of nearshore reproductive habitat, winter area, breeding areas, and migratory corridors. The critical habitat is categorized into 38 occupied marine areas and 685 miles of nesting beaches (Figure 27).

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

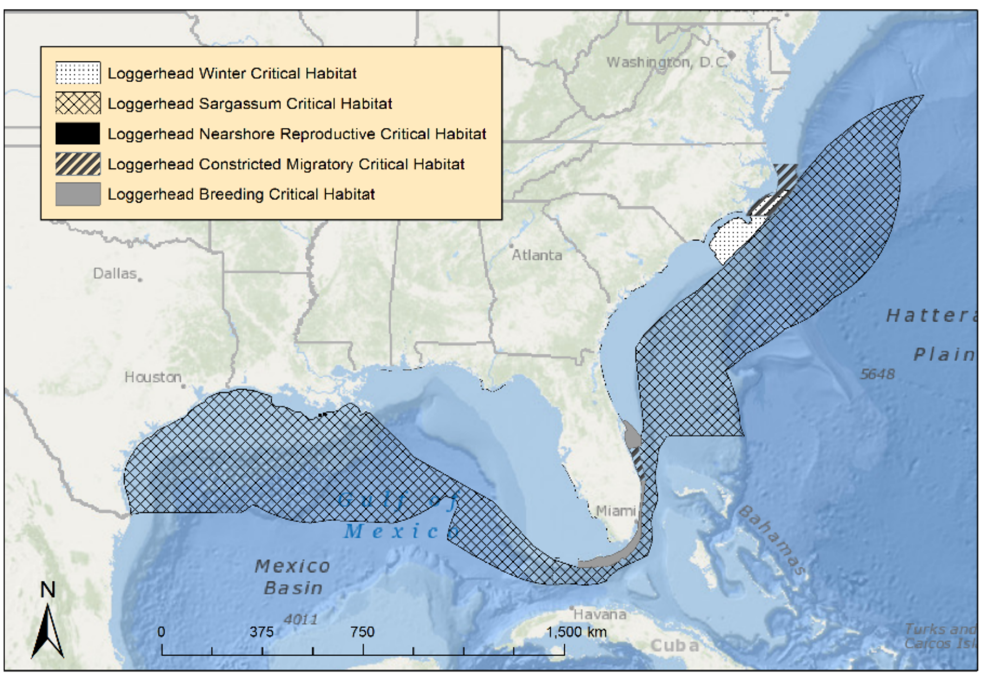


Figure 27. Map depicting loggerhead turtle designated critical habitat.

Recovery Goals

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

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

8.2.8 Atlantic Sturgeon and New York Bight Designated Critical Habitat

Sturgeon are among the most primitive of the bony fishes. The Atlantic sturgeon is a long-lived (approximately 60 years), late maturing, iteroparous, estuarine dependent species (ASSRT 2007; Dadswell 2006). Atlantic sturgeon are anadromous, spawning in freshwater but spending most of their subadult and adult life in the marine environment. They can grow to approximately 14 feet long and can weigh up to 800 pounds. Atlantic sturgeon are bluish-black or olive brown dorsally (on their back) with paler sides, a white belly, and have five major rows of dermal "scutes" (Figure 28).

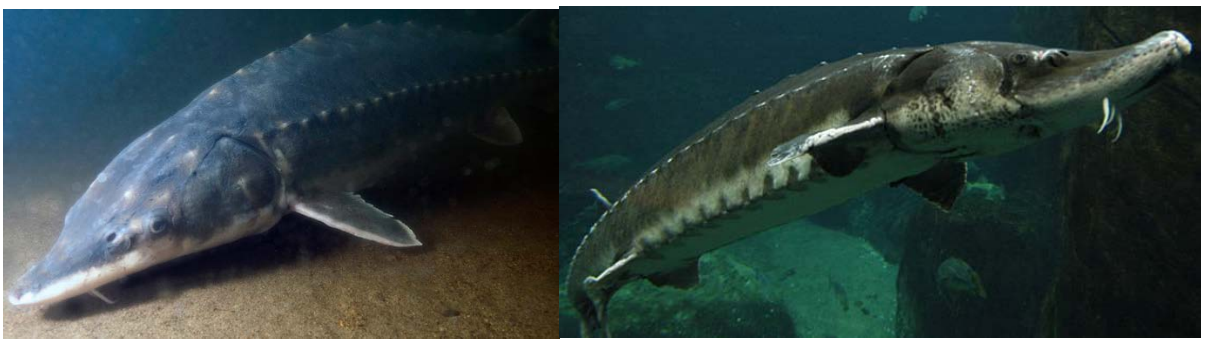


Figure 28. Atlantic sturgeon. Photo: Robert Michelson (left), NOAA (right).

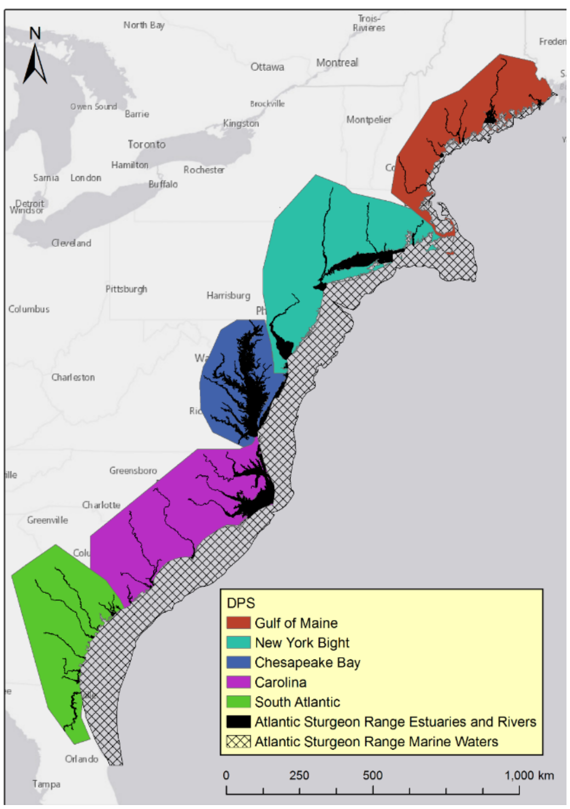


Figure 29. Range and boundaries of the five Atlantic sturgeon DPSs.

Five DPSs of Atlantic sturgeon were listed under the ESA in 2012 (Table 17). The Gulf of Maine DPS is listed as threatened while the New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs are listed as endangered (Figure 29).

Table 17. Summary of Atlantic sturgeon listing and recovery plan information.

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acipenser oxyrinchus oxyrinchus	Atlantic sturgeon	Gulf of Maine	Threatened	2007	77 FR 5880 02/06/2012	N/A	82 FR 39160 Gulf of Maine 2017
Acipenser oxyrinchus oxyrinchus	Atlantic sturgeon	New York Bight	Endangered	2007	77 FR 5880 02/06/2012	N/A	82 FR 39160 New York Bight 2017
Acipenser oxyrinchus oxyrinchus	Atlantic sturgeon	Chesapeake Bay	Endangered	2007	77 FR 5880 02/06/2012	N/A	82 FR 39160 Chesapeake Bay 2017
Acipenser oxyrinchus oxyrinchus	Atlantic sturgeon	Carolina	Endangered	2007	77 FR 5914 02/06/2012	N/A	82 FR 39160 Carolina 2017
Acipenser oxyrinchus oxyrinchus	Atlantic sturgeon	South Atlantic	Endangered	2007	77 FR 5914 02/06/2012	N/A	82 FR 39160 South Atlantic 2017

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

Life History

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

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

Sturgeon eggs are highly adhesive and are deposited on the bottom substrate, usually on hard surfaces (e.g., cobble) (Smith and Clugston 1997). Hatching occurs approximately 94 to 140 hours after egg deposition, and larvae assume a demersal existence (Smith et al. 1980). The yolk sac larval stage is completed in about 8 to 12 days, during which time the larvae move downstream to rearing grounds over a 6 to 12-day period (Kynard and Horgan 2002). During the first half of their migration downstream, movement is limited to nighttime. During the day, larvae use benthic structure (e.g., gravel matrix) as refugia (Kynard and Horgan 2002). During the latter half of migration when larvae are more fully developed, movement to rearing grounds occurs both day and night. The larvae grow rapidly and are 4” to 5 1/2" long at a month old (MSPO 1993). At this size, the young sturgeon bear teeth and have sharp, closely spaced spine-tipped scutes. As growth continues, they lose their teeth, the scutes separate and lose their sharpness.

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

Atlantic sturgeon have been aged to 60 years (Mangin 1964), but this should be taken as an approximation because the age validation studies conducted to date show ages cannot be reliably estimated after 15 to 20 years (Stevenson and Secor 2000). Vital parameters of sturgeon populations generally show clinal variation with faster growth, earlier age at maturation, and shorter life span in more southern systems. Spawning intervals range from one to five years for male Atlantic sturgeon (Collins et al. 2000; Smith 1985) and three to five years for females (Schueller and Peterson 2010; Stevenson and Secor 2000). Fecundity of Atlantic sturgeon is correlated with age and body size, ranging from approximately 400,000 to 8 million eggs (Dadswell 2006; Smith et al. 1982; Van Eenennaam and Doroshov 1998). The average age at which 50 percent of Atlantic sturgeon maximum lifetime egg production is achieved is estimated to be 29 years, approximately 3 to 10 times longer than for most other bony fish species (Boreman 1997).

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

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes population growth rate, genetic diversity, and distribution as it relates to the Atlantic sturgeon.

Population Growth Rate

8.2.8.1 Gulf of Maine DPS

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

8.2.8.2 New York Bight DPS

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

8.2.8.3 Chesapeake Bay DPS

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

8.2.8.4 Carolina DPS

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

8.2.8.5 South Atlantic DPS

Precise estimates of population growth rate (intrinsic rates) for the South Atlantic DPS are unknown due to lack of long-term abundance data. During the last two decades, Atlantic sturgeon have been observed in most South Carolina coastal rivers, although it is not known if all rivers support a spawning population (Collins and Smith 1997). The Altamaha River supports the healthiest Atlantic sturgeon populations in the South Atlantic DPS. In a telemetry study by Peterson et al. (2008), most tagged adult Atlantic sturgeon were found between river kilometer 215 and 420 in October and November when water temperatures were appropriate for spawning. The status review team (ASSRT 2007) found that, overall, the South Atlantic DPS had a moderate risk (less than 50 percent chance) of becoming endangered over the next 20 years.

Genetic Diversity

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

Distribution

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

8.2.8.6 Gulf of Maine DPS

The Gulf of Maine DPS includes all anadromous Atlantic sturgeons that are spawned in the watersheds from the Maine/Canadian border and, extending southward, all watersheds draining into the Gulf of Maine as far south as Chatham, Massachusetts (Figure 29). The geomorphology of most small coastal rivers in Maine is not sufficient to support Atlantic sturgeon spawning populations, except for the Penobscot and the estuarial complex of the Kennebec, Androscoggin, and Sheepscot rivers. Spawning still occurs in the Kennebec and Androscoggin Rivers, and may occur in the Penobscot River. Atlantic sturgeon have more recently been observed in the Saco, Presumpscot, and Charles rivers.

8.2.8.7 New York Bight DPS

The natal river systems of the New York Bight DPS span from the Connecticut River south to the Delaware River (Figure 29). The Connecticut River has long been known as a seasonal aggregation area for subadult Atlantic sturgeon, and both historical and contemporary records document presence of Atlantic sturgeon in the river as far upstream as Hadley, Massachusetts (Pacileo 2003; Savoy 1992). The upstream limit for Atlantic sturgeon on the Hudson River is the Federal Dam at the fall line, approximately river kilometer 246 (Dovel 1983; Kahnle et al. 2007). In the Delaware River, there is evidence of Atlantic sturgeon presence from the mouth of the Delaware Bay to the head-of-tide at the fall line near Trenton on the New Jersey side and Morrisville on the Pennsylvania side of the River, a distance of 220 river kilometers (Breece et al. 2013).

Of the Navy's origination and destination ports for the proposed action, New York Bight DPS of Atlantic sturgeon are found in the port of Philadelphia on the Delaware River. This port is also part of the critical habitat designation (Unit 4) for the New York Bight DPS (Figure 30).

8.2.8.8 Chesapeake Bay DPS

The natal river systems of the Chesapeake Bay DPS span from the Susquehanna River south to the James River (Figure 29).

8.2.8.9 Carolina DPS

The natal river systems of the Carolina DPS span from the Roanoke River, North Carolina south to the Santee-Cooper system in South Carolina (Figure 29). The Carolina DPS ranges from the Santee-Cooper River to the Albemarle Sound and consists of seven extant populations; one population (the Sampit River) is believed to be extirpated.

8.2.8.10 South Atlantic DPS

The natal river systems of the South Atlantic DPS span from Edisto south to the St. Mary's River (Figure 29). Seventy-six Atlantic sturgeon were tagged in the Edisto River during a 2011 to 2014 telemetry study (Post et al. 2014). Fish entered the river between April and June and were detected in the saltwater tidal zone until water temperature decreased below 25 degrees Celsius. They then moved into the freshwater tidal area, and some fish made presumed spawning migrations in the fall around September to October. Atlantic sturgeon in the Savannah River were documented displaying similar behavior three years in a row—migrating upstream during the fall and then being absent from the system during spring and summer. Forty three Atlantic sturgeon larvae were collected in upstream locations (river kilometer 113 to 283) near presumed spawning locations (Collins 1997).

Status

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

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

Designated Critical Habitat

NMFS designated critical habitat for each ESA-listed DPS of Atlantic sturgeon in August of 2017 (Table 17). PBFs determined to be essential for Atlantic sturgeon reproduction and recruitment include (1) suitable hard bottom substrate in low salinity waters for settlement of fertilized eggs, refuge, growth, and development of early life stages, (2) transitional salinity zones for juvenile foraging and physiological development, (3) water of appropriate depth and absent physical barriers to passage, (4) unimpeded movement of adults to and from spawning sites, and (5) water quality conditions that support spawning, survival, growth, development, and recruitment. The action area for this opinion overlaps with Atlantic sturgeon New York Bight DPS critical habitat units located in the Delaware River (Figure 30).

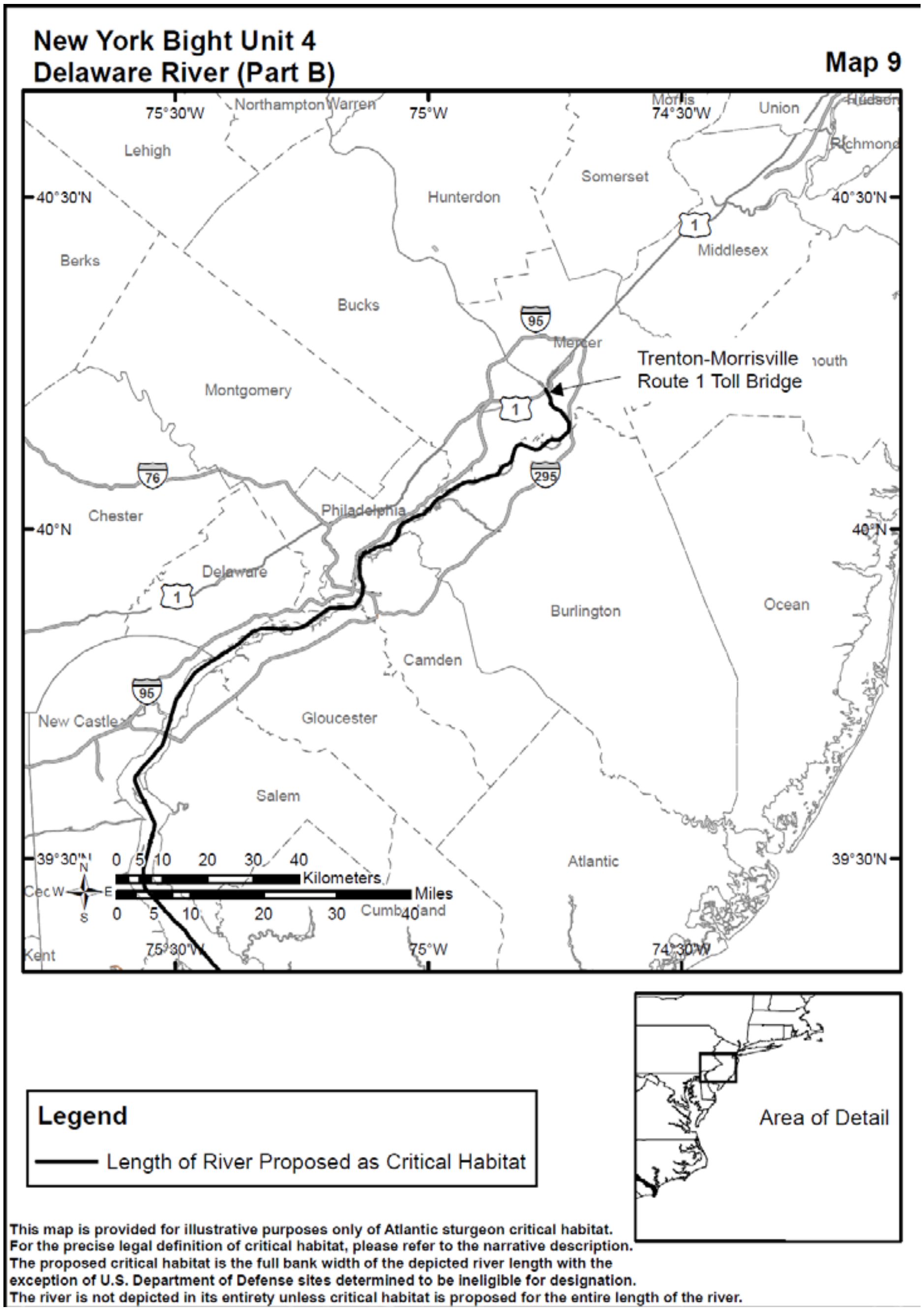


Figure 30. Map of designated critical habitat Unit 4 for the New York Bight DPS of Atlantic sturgeon which include the port of Philadelphia.

Recovery Goals

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

8.2.9 Bocaccio, Puget Sound/Georgia Basin DPS and Designated Critical Habitat

The bocaccio is a long-lived, large species of rockfish, occupying the eastern Pacific Ocean in waters from California to Alaska. Puget Sound/Georgia Basin DPS bocaccio are those that reside in the Puget Sound/Georgia Basin (Figure 31).

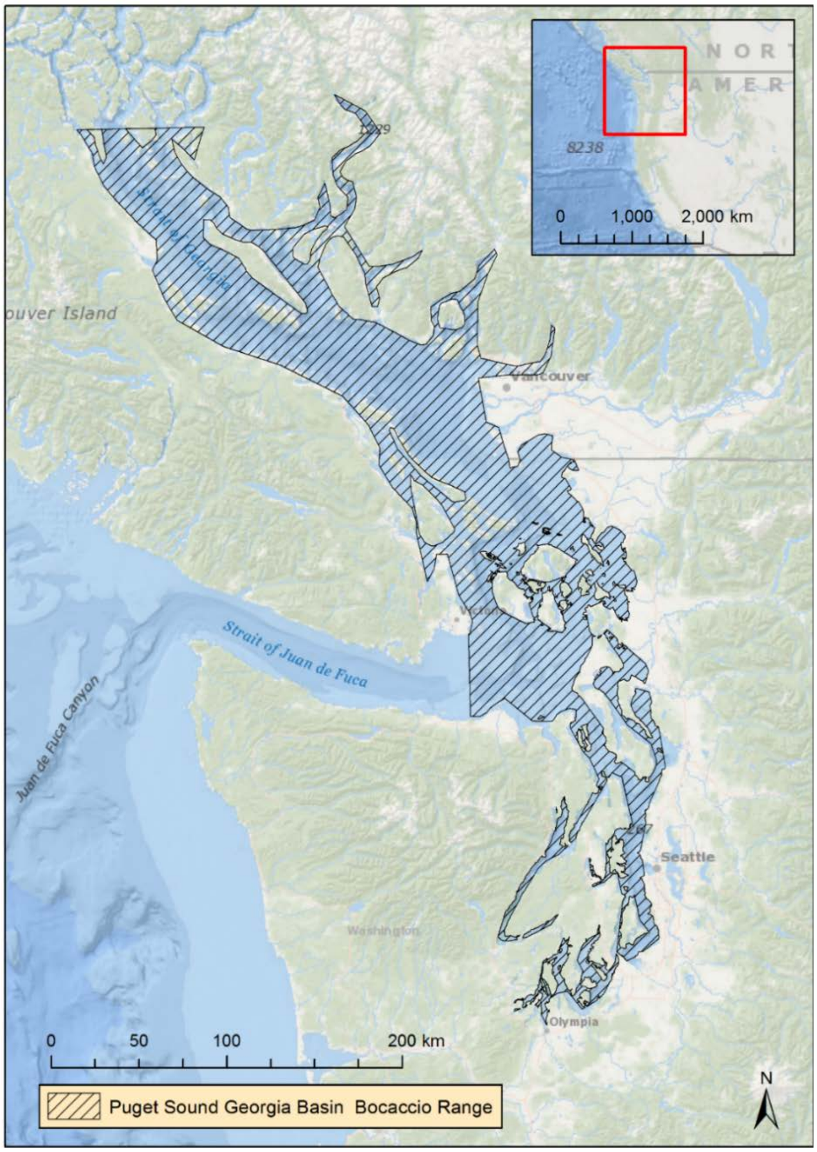


Figure 31. Map identifying the range of the Puget Sound/Georgia Basin DPS of bocaccio.

Bocaccio are a large (three feet/one meter) Pacific rockfish, olive to burnt orange-brown, with a distinctively long jaw (Figure 32). The Puget Sound/Georgia Basin DPS was first listed as endangered by NMFS on April 28, 2010 (Table 18).

Figure 32. Bocaccio. Photo: Mary Nishimoto, NOAA.

The listing was updated on January 23, 2017, when NMFS amended the listing description to include fish residing within the Puget Sound/Georgia Basin rather than fish originating from the Puget Sound/Georgia Basin.

Table 18. Summary of bocaccio listing and recovery plan information.

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Sebastes paucispinis	Bocaccio	Puget Sound/Georgia Basin	Endangered	2016	75 FR 22276 04/28/2010	2017	79 FR 68041 Puget Sound 2017

We used information available in the status review (NMFS 2016i) and recent scientific publications to summarize the life history, population dynamics, and status of the species, as follows.

Life History

Bocaccio larvae and young juveniles tend to be found in offshore regions (1 to 148 kilometers offshore), associated with the surface and occasionally with floating kelp mats (NMFS 2016i). As adults, fish move into waters 18 to 30 meters deep and occupy rocky reefs (Carr 1983; Eschmeyer et al. 1983; Feder et al. 1974; Johnson 2006; Love and Yoklavich 2008). As adults, bocaccio may be found in depths of 12 to 478 meters, but tend to remain in shallow waters on the continental shelf (20 to 250 meters), still associating mostly with reefs or other hard substrate, but may move over mud flats.

Bocaccio are live-bearers with internal fertilization. Once females become mature (at 54 to 61 centimeters total length), they produce 20,000 to 2.3 million eggs annually, with the number increasing as females age and grow larger (Echeverria 1987; Hart 1973; Love et al. 2002).

However, either sex has been known to attain sexual maturity as small as 35 centimeters or three years of age. In recent years as populations have declined, average age at sexual maturity may have declined as well (Echeverria 1987; Hart 1973; Love et al. 2002; MacCall 2002). Mating occurs between August and November, with larvae born between January and April (NMFS 2016i).

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

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes population growth rate, genetic diversity, and distribution as it relates to the Puget Sound/Georgia Basin DPS bocaccio.

Population Growth Rate

There is no current population abundance estimate for the Puget Sound/Georgia Basin DPS bocaccio. There is a lack of long-term information on this DPS for bocaccio abundance, although among rockfish of the Puget Sound, bocaccio appear to have undergone a particular decline. This was likely because of the removal of the largest, most fecund individuals of the population due to overfishing and the frequent failure of recruitment classes, possibly because of unfavorable climactic/oceanographic conditions (MacCall and He 2002). The rate of decline for rockfish in Puget Sound has been estimated at 3.1 to 3.8 percent annually for the period 1977 to 2014 (NMFS 2016i).

Genetic Diversity

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

Distribution

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

Status

Bocaccio resistance to depletion and recovery is also hindered by demographic features (Love et al. 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 2002). Over the past 30 years, 1977, 1984, and 1988 are the only years in which recruitment appears to have been significant successes. Recruitment success appears to be linked to oceanographic/climactic patterns and may be related to cyclic warm/cool ocean periods, with cool periods having greater success (Love et al. 1998b; MacCall 1996; Moser et al. 2000; Sakuma and Ralson 1995). Harvey et al. (2006) suggested that bocaccio may have recently diverted resources from reproduction, potentially resulting in additional impairment to recovery.

Designated Critical Habitat

Critical habitat for the Puget Sound/Georgia Basin DPS for bocaccio, canary rockfish, and yelloweye rockfish was finalized in 2014. The critical habitat designation was updated in 2017 when canary rockfish were delisted. The specific areas designated for bocaccio include approximately 1,184.75 square miles (3,068.5 square kilometers) of marine habitat in Puget Sound, Washington (Figure 33). Designated habitat was divided into two units: nearshore, to support juveniles, and deeper, rocky habitat for adults. Features essential for adult bocaccio (greater than 30 meters deep) include sufficient prey resources, water quality, and rocks or highly rugose habitat. For juvenile bocaccio, features essential for their conservation include sufficient prey resources and water quality. The Puget Sound Naval Shipyard at Naval Base Kitsap was excluded from critical habitat designation for Puget Sound/Georgia Basin bocaccio.

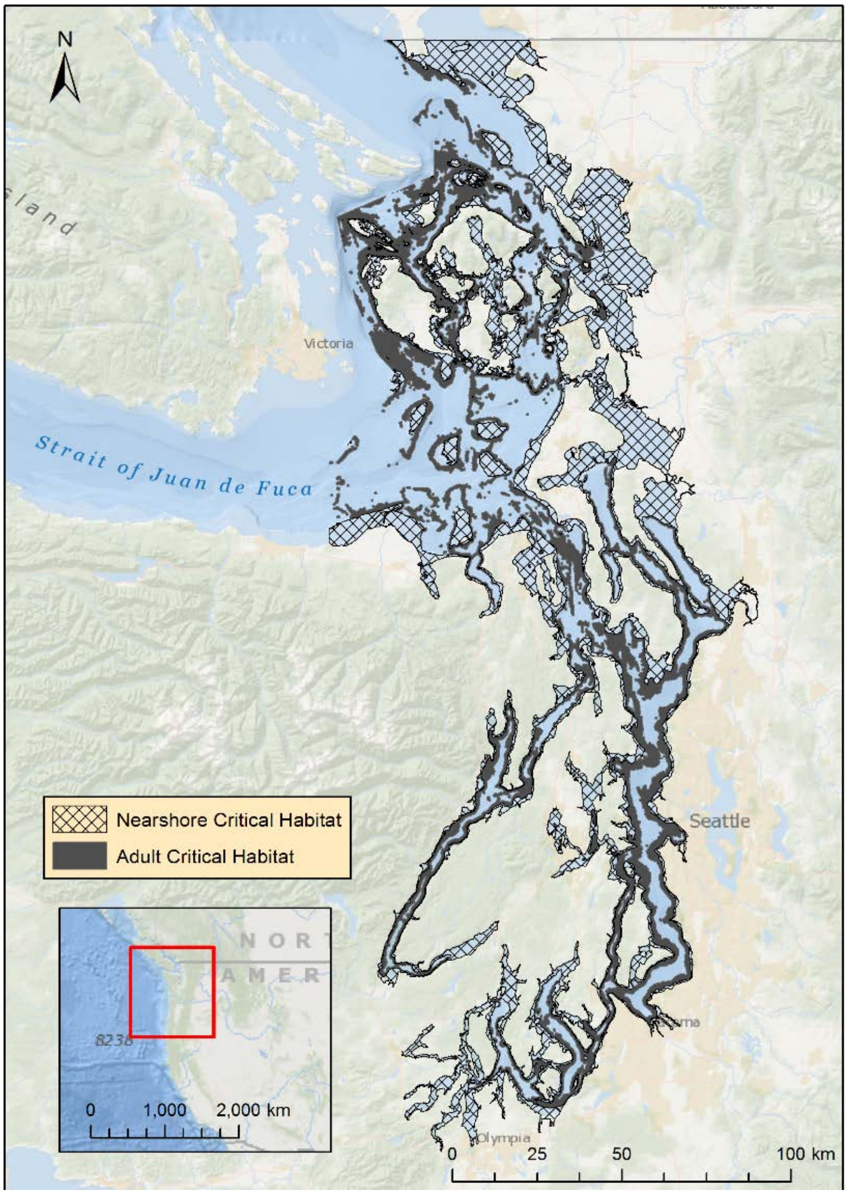


Figure 33. Map of designated critical habitat for the bocaccio Puget Sound/Georgia Basin DPS.

Recovery Goals

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

1. Improve our knowledge of the current and historical status of the yelloweye rockfish and bocaccio and their habitats.

2. Reduce or eliminate existing threats to listed rockfish from fisheries/anthropogenic mortality.
3. Reduce or eliminate existing threats to listed rockfish habitats and restore important rockfish habitat.

8.2.10 Chinook Salmon, Puget Sound ESU and Designated Critical Habitat

The Puget Sound ESU includes naturally spawned Chinook salmon originating from rivers flowing into Puget Sound from the Elwha River (inclusive) eastward, including rivers in Hood Canal, South Sound, North Sound and the Strait of Georgia. Twenty-six artificial propagation programs are included as part of the ESU (Figure 34).

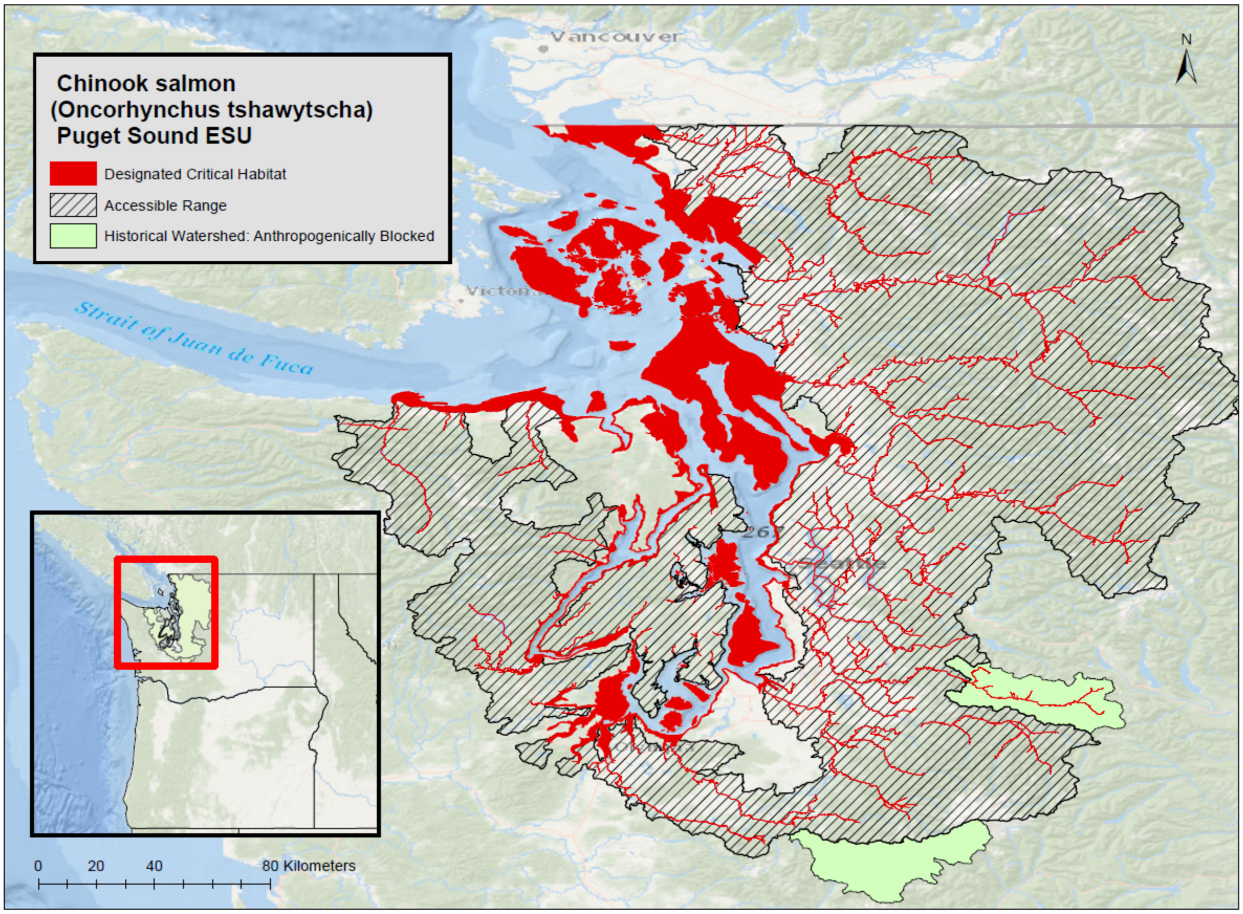


Figure 34. Geographic range and designated critical habitat of Chinook salmon, Puget Sound ESU.

Chinook salmon, also referred to as king salmon, are the largest of the Pacific salmon. Spawning adults are olive to dark maroon in color, without conspicuous streaking or blotches on the sides (Figure 35). Spawning males are darker than females, and have a hooked jaw and slightly humped back. They can be distinguished from other spawning salmon by the color pattern, particularly the spotting on the back and tail, and by the dark, solid black gums of the lower jaw (Moyle 2002).

Figure 35. Adult Chinook salmon. Photo: NOAA (left).

On March 24, 1999, NMFS listed the Puget Sound ESU of Chinook salmon as a "threatened" species. The listing was revisited and confirmed as "threatened" in 2005 (Table 19).

Table 19. Summary of Chinook salmon, Puget Sound Evolutionarily Significant Unit listing and recovery plan information.

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus tshawytscha	Chinook salmon	Puget Sound ESU	Threatened	2011	70 FR 37160 06/28/2005	2007	70 FR 52630 Puget Sound 2005

Life History

Puget Sound Chinook salmon populations exhibit both early-returning (August) and late-returning (mid-September and October) adult Chinook salmon spawners (Healey 1991). Juvenile Chinook salmon within the Puget Sound generally exhibit an "ocean-type" life history. However, substantial variation occurs with regard to juvenile residence time in freshwater and estuarine environments. Juvenile Chinook that remain in freshwater longer have a "river-type" life history. Some rivers may have Chinook that exhibit both strategies. In addition, Hayman (Hayman et al. 1996) described three juvenile life histories for Chinook salmon with varying freshwater and estuarine residency times in the Skagit River system in northern Puget Sound. In this system, 20 percent to 60 percent of sub-yearling migrants rear for several months in freshwater habitats while the remaining fry migrate to rear in the Skagit River estuary and delta (Beamer et al. 2005). Juveniles in tributaries to Lake Washington exhibit both a stream rearing and a lake rearing strategy. Lake rearing fry are found in highest densities in nearshore shallow (less than 1 meter) habitat adjacent to the opening of tributaries or at the mouth of tributaries where they empty into the lake (Tabor et al. 2006). Puget Sound Chinook salmon also has several estuarine rearing juvenile life history types that are highly dependent on estuarine areas for rearing

(Beamer et al. 2005). In the estuaries, fry use tidal marshes and connected tidal channels including dikes and ditches developed to protect and drain agricultural land. During their first ocean year, immature Chinook salmon use nearshore areas of Puget Sound during all seasons and can be found long distances from their natal river systems (Brennan et al. 2004).

Upon entering Puget Sound, juvenile Chinook salmon are nearshore obligate foraging in shallow areas with protective cover, such as tidally influenced sandy beaches and zones vegetated with eelgrass (*Zostera marina*) (Healey et al. 1991). Cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson et al. 1981; MacFarlane and Norton 2002; Sommer et al. 2001a). In the marine environment, as juvenile Chinook salmon increase in size, they move away from the shore into deeper water and feed voraciously on zooplankton, larval crustaceans and juvenile fishes, plankton, and terrestrial insects (Healey et al. 1991; MacFarlane and Norton 2002). Chinook salmon grow rapidly in the ocean environment, with growth rates dependent on water temperatures and food availability.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes population growth rate, genetic diversity, and distribution as it relates to Chinook salmon, Puget Sound ESU.

Population Growth Rate

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

Available data on total abundance since 1980 indicate that although abundance trends have fluctuated between positive and negative for individual populations, there are widespread negative trends in natural-origin Chinook salmon spawner abundance across the ESU (NMFS NWFSC 2015). Productivity remains low in most populations, and hatchery-origin spawners are present in high fractions in most populations outside of the Skagit watershed. Available data now shows that most populations have declined in abundance over the past 7 to 10 years. Further, escapement levels for all populations remain well below the Technical Recovery Team (TRT) planning ranges for recovery, and most populations are consistently below the spawner-recruit levels identified by the TRT as consistent with recovery (NMFS NWFSC 2015).

Genetic Diversity

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

Distribution

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

Status

All Puget Sound Chinook salmon populations are well below escapement abundance levels identified as required for recovery to low extinction risk in the recovery plan. In addition, most populations are consistently below the productivity goals identified in the recovery plan as necessary for recovery. Although trends vary for individual populations across the ESU, most populations have declined in total natural origin recruit abundance since the last status review; while natural origin recruit escapement trends are severely depressed, they have remained mostly stable since 1995. Several of the risk factors identified in the previous status review (Good et al. 2005) are still present, including high fractions of hatchery fish in many populations and widespread loss and degradation of water quality and habitat. Although this ESU's total abundance is a greatly reduced from historic levels, recent abundance levels do not indicate that the ESU is at immediate risk of extinction. This ESU remains relatively well distributed over 22 populations in 5 geographic areas across the Puget Sound. Although current trends are concerning, the available information indicates that this ESU remains at moderate risk of extinction (NMFS 2011a).

Designated Critical Habitat

Critical habitat was designated for Puget Sound Chinook salmon on September 2, 2005. It includes 1,683 kilometers of stream channels, 41 square kilometers of lakes, and 3,512 kilometers of nearshore marine habitat (Figure 36). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, nearshore marine areas, and offshore marine areas. The PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. The Puget Sound Naval Shipyard at Naval Base Kitsap was excluded from critical habitat designation for Puget Sound Chinook salmon.

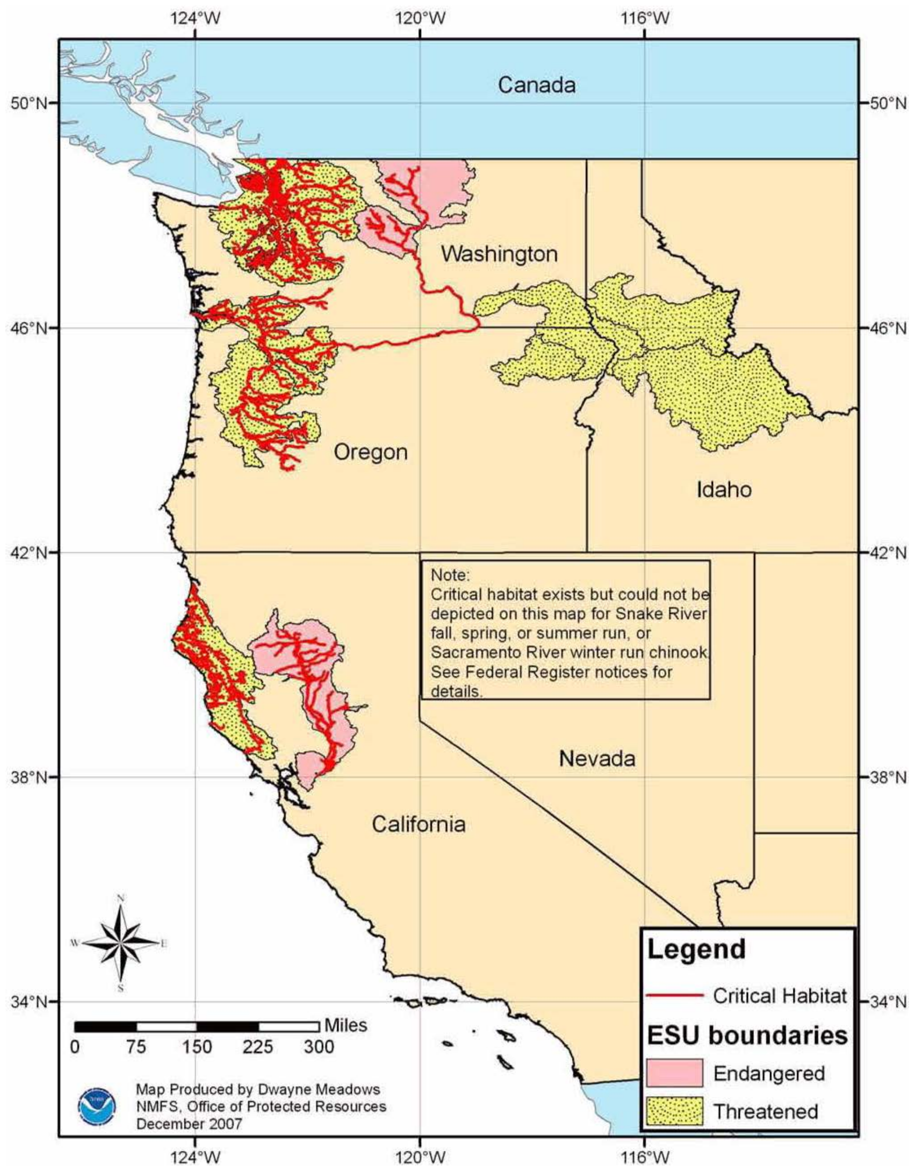


Figure 36. Map of designated critical habitat for Chinook salmon.

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

Hydroelectric development and flood control also obstruct Puget Sound Chinook salmon migration in several basins. The most functional PBFs are found in northwest Puget Sound: the

Skagit River basin, parts of the Stillaguamish River basin, and the Snohomish River basin where federal land overlap with critical habitat designated for the Puget Sound Chinook salmon. However, estuary PBFs are degraded in these areas by reduction in the water quality from contaminants, altered salinity conditions, lack of natural cover, and modification and lack of access to tidal marshes and their channels from the installation of floodgates and other structures.

Recovery Goals

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

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

8.2.11 Chum Salmon, Hood Canal Summer-run ESU and Designated Critical Habitat

The chum salmon, Hood Canal summer-run ESU includes naturally spawned summer-run chum salmon originating from Hood Canal and its tributaries as well as from Olympic Peninsula Rivers between Hood Canal and Dungeness Bay (inclusive) (Figure 37). Also, summer-run chum salmon includes four artificial propagation programs.

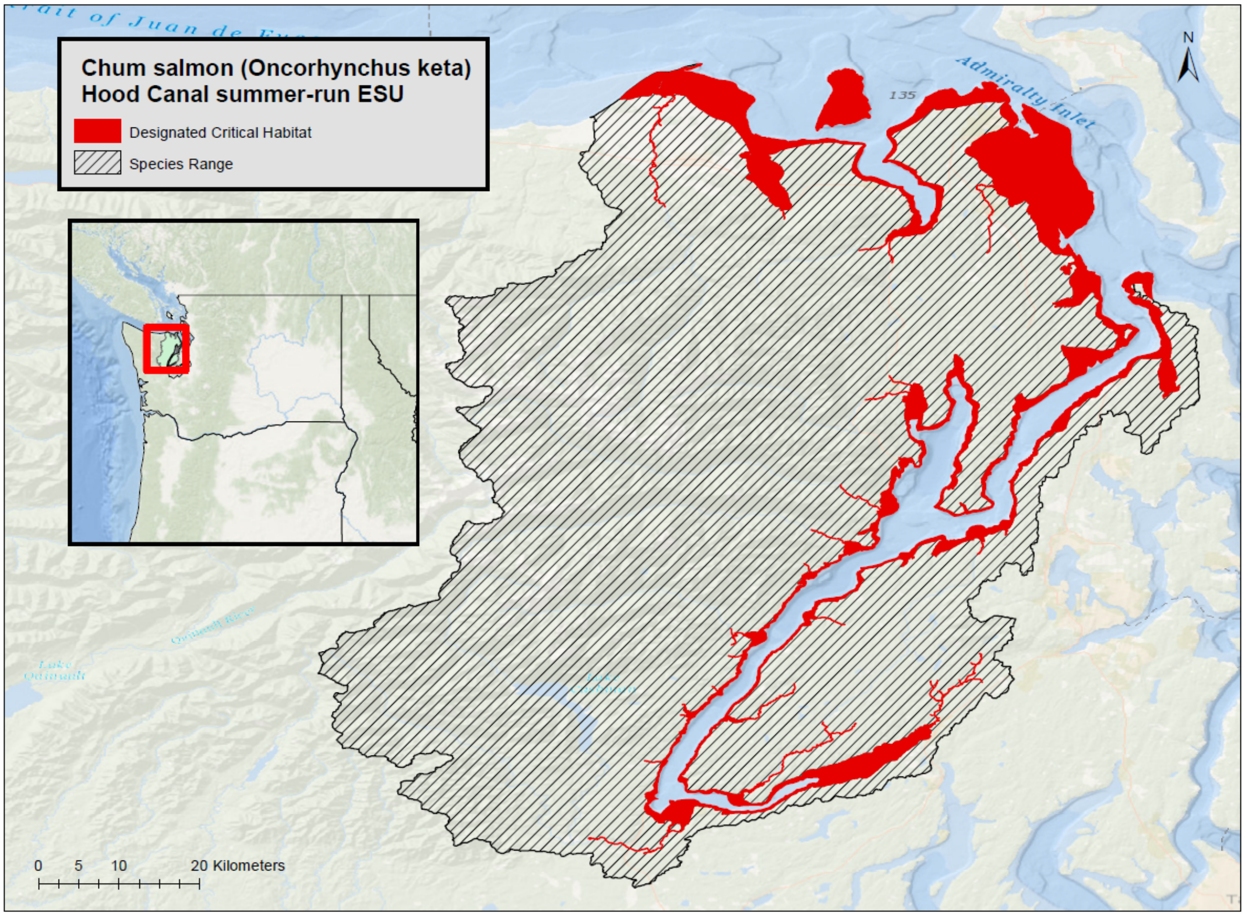


Figure 37. Geographic range and designated critical habitat of chum salmon, Hood Canal summer-run ESU.

Chum salmon are an anadromous (i.e., adults migrate from marine to freshwater streams and rivers to spawn) and semelparous (i.e., they spawn once and then die) fish species. Adult chum salmon are typically between eight and fifteen pounds, but they can get as large as 45 pounds and 3.6 feet long. Males have enormous canine-like fangs and a striking calico pattern body color (front two-thirds of the flank marked by a bold, jagged, reddish line and the posterior third by a jagged black line) during spawning (Figure 38). Females are less flamboyantly colored and lack the extreme dentition of the males. Ocean stage chum salmon are metallic greenish-blue along the back with black speckles. Chum salmon have the widest natural geographic and spawning distribution of the Pacific salmonids.

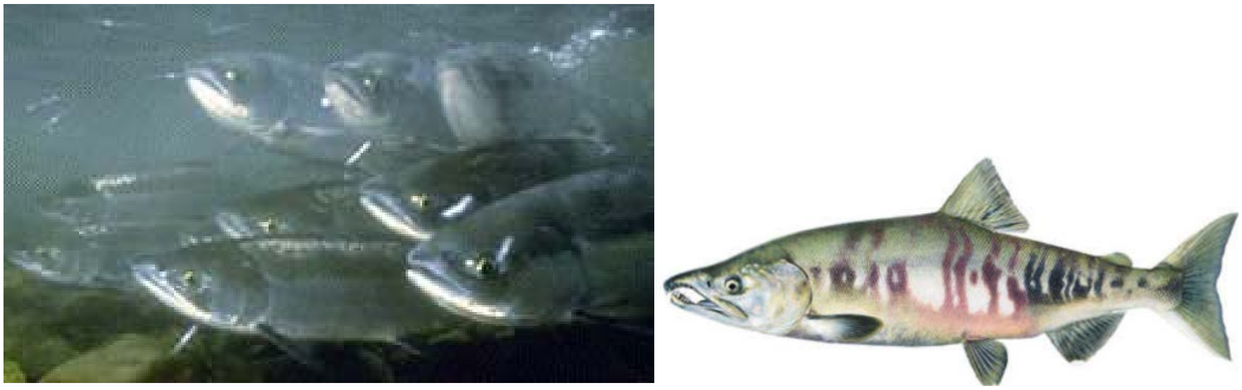


Figure 38. Chum salmon. Photo: NOAA (left).

On March 25, 1999, NMFS listed the Hood Canal summer-run ESU and the Columbia River ESU of chum salmon as threatened. NMFS reaffirmed the status of these two ESUs as threatened on June 28, 2005 (Table 20).

Table 20. Summary of chum salmon, Hood Canal summer-run ESU listing and recovery plan information.

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus keta	Chum salmon	Hood Canal summer-run	Threatened	2011	70 FR 37160 06/28/2005	2005	70 FR 52630 Washington 2005

Life History

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

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

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: population growth rate, genetic diversity, and distribution as it relates to the chum salmon, Hood Canal summer-run ESU.

Population Growth Rate

Of the sixteen populations that comprise the Hood Canal Summer-run chum ESU, seven are considered "functionally extinct" (Skokomish, Finch Creek, Anderson Creek, Dewatto, Tahuya, Big Beef Creek and Chimicum). NMFS examined average escapements (geometric means) for five-year intervals and estimated trends over the intervals for all natural spawners and for natural-origin only spawners. For both populations, abundance was relatively high in the 1970s, lowest for the period 1985-1999, and high again for the most recent 10 years (NMFS NWFSC 2015).

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

Genetic Diversity

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

Distribution

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

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

Status

The two most recent status reviews indicate some positive signs for the Hood Canal summer-run chum salmon ESU. Diversity has increased from the low levels seen in the 1990s due to both the reintroduction of spawning aggregates and the more uniform relative abundance between populations; considered a good sign for viability in terms of spatial structure and diversity (NMFS NWFSC 2015). Spawning distribution within most streams was also extended further upstream with increased abundance. At present, spatial structure and diversity viability parameters for each population nearly meet the viability criteria (NMFS NWFSC 2015). Spawning abundance has remained relatively high compared to the low levels observed in the early 1990's (NMFS NWFSC 2015). Natural-origin spawner abundance has shown an increasing trend since 1999, and spawning abundance targets in both populations were met in some years (NMFS NWFSC 2015). Despite substantive gains towards meeting viability criteria in the Hood Canal and Strait of Juan de Fuca summer chum salmon populations, the ESU still does not meet all of the recovery criteria for population viability at this time (NMFS NWFSC 2015). Overall, the Hood Canal Summer-run chum salmon ESU remains at a moderate risk of extinction.

Designated Critical Habitat

NMFS designated critical habitat for Hood Canal Summer-run chum salmon in 2005 (Table 20). Critical habitat includes freshwater spawning, freshwater rearing, freshwater migration, estuarine areas free of obstruction, nearshore marine areas free of obstructions and offshore marine areas with good water quality. The PBFs (formerly *primary constituent elements*) that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity (Figure 37).

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

Recovery Goals

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

8.2.12 Eulachon, Southern DPS and Designated Critical Habitat

The eulachon is a small, cold-water species of anadromous fish, occupying the eastern Pacific Ocean in nearshore waters to depths of about 1,000 feet (300 meters) from California to the Bering Sea. Eulachon will return to their natal river spawn. Southern DPS eulachon are those that spawn in rivers south of the Nass River in British Columbia to the Mad River in California (Figure 39) (NMFS WCR 2016).

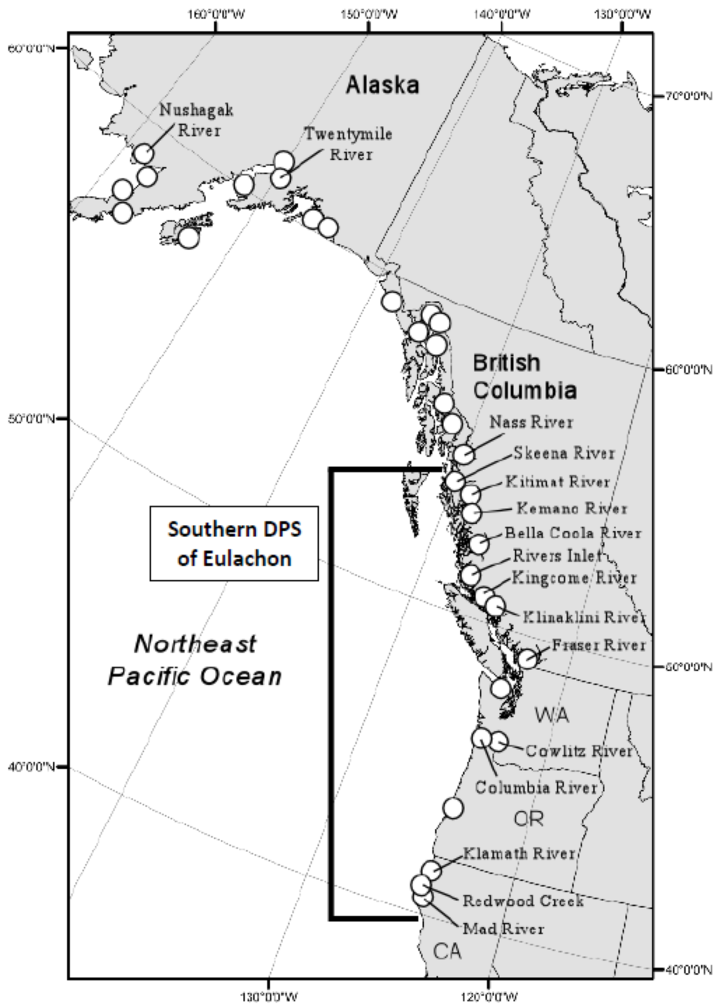


Figure 39. Map identifying the range of the eulachon Southern DPS (NMFS WCR 2016).

Eulachon are a small (8.5 inches, 21.5 centimeters) anadromous fish, with brown or blue backs, silver on their sides, and white underneath (Figure 40).

Figure 40. Eulachon. Photo: NOAA Alaska Fisheries Science Center.

The Southern DPS was first listed as threatened by NMFS on March 18, 2010 (Table 21).

Table 21. Summary of eulachon, Southern DPS listing and recovery plan information.

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Thaleichthys pacificus	Eulachon	Southern	Threatened	2016	75 FR 13012 03/18/2010	Draft Recovery Plan 2016	76 FR 515 WA/OR/CA 2011

We used information available in the status review (Gustafson et al. 2010), the updated status review (Gustafson 2016), the 5-year review (NMFS WCR 2016), and recent scientific publications to summarize the life history, population dynamics and status of the species, as follows.

Life History

Although primarily marine, eulachon return to freshwater to spawn. For the Southern DPS eulachon, most spawning occurs in the Columbia River and its tributaries. Spawning usually occurs between ages two and five. Spawning is strongly influenced by water temperatures, and the timing of migration typically occurs between December and June, when water temperatures are between zero degrees Celsius and 10 degrees Celsius (Gustafson 2016). In the Columbia River and further south, spawning occurs from late January to March (Hay and McCarter 2000). Further north, the peak of eulachon runs in Washington State is from February through March (Hay and McCarter 2000). Females lay between 7,000 and 60,000 eggs over sand, course gravel or detrital substrate. Eggs attach to gravel or sand and incubate for 30 to 40 days after which larvae drift to estuaries and coastal marine waters. In their first year of life, juveniles are found along the continental shelf (Gustafson 2016; Wydoski and Whitney 1979). Adult eulachon are found in coastal and offshore marine habitats. With the exception of some individuals in Alaska, eulachon generally die after spawning (Gustafson 2016). The maximum known lifespan is nine years of age, but 20 to 30 percent of individuals live to four years and most individuals survive to three years of age, although spawning has been noted as early as two years of age. 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.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: population growth rate, genetic diversity, and distribution as it relates to the Southern DPS eulachon.

Population Growth Rate

There is no current population abundance estimate for the Southern DPS eulachon. There is a lack of long-term information on Southern DPS eulachon abundance, although the available fisheries landings data indicate a steep decline in the early to mid-1990s (Gustafson 2016). Data from fisheries surveys show that abundance can vary from year to year. There is no population growth rate available for Southern DPS eulachon, although some indices show an increasing temporal trend. (Gustafson 2016).

Genetic Diversity

Southern DPS eulachon are genetically distinct from eulachon in the northern parts of its range (i.e., Alaska). Recent genetic analysis indicates that the Southern DPS exhibits a regional population structure, with a three-population southern Columbia-Fraser group, coming from the Cowlitz, Columbia, and Fraser rivers (Candy et al. 2015; Gustafson 2016).

Distribution

Adult and juvenile Southern DPS eulachon can be found in the Pacific Ocean, along the continental shelf, in waters from 50 to 200 meters deep (Gustafson 2016). Adults are most frequently found in the Columbia River and its tributaries (e.g., Cowlitz River, Sandy River), and sometimes in the Klamath River, California.

Status

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). The species 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. Warmer water temperatures associated with climate change could alter the timing of spawning, and the availability of prey for larval and juvenile eulachon (NMFS WCR 2016). 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).

Designated Critical Habitat

On October 20, 2011, NMFS designated critical habitat for Southern DPS eulachon. Sixteen areas were designated in the states of Washington, Oregon, and California. Eulachon critical habitat is not found within the action area.

Recovery Goals

See the 2016 Draft Recovery Plan for the Southern DPS eulachon, for complete down listing/delisting criteria for each of their respective recovery goals (NMFS WCR 2016). The following items were the top recovery actions identified to support in the Draft Recovery Plan:

1. Implement outreach and education strategies.
2. Conduct strategic research on eulachon.
3. Develop biological viability targets.
4. Conduct strategic research on eulachon habitats.
5. Conduct research on threats, including in marine and freshwater habitat, bycatch, predation, dams and water diversions, water quality, and others.
6. Assess regulatory measures, inadequacy of existing regulatory mechanisms.
7. Develop a research, monitoring, evaluation, and adaptive management plan.

8.2.13 Green Sturgeon, Southern DPS and Designated Critical Habitat

The North American green sturgeon, *Acipenser medirostris*, is an anadromous fish that occurs in the nearshore Eastern Pacific Ocean from Alaska to Mexico (Moyle 2002) (Figure 41).

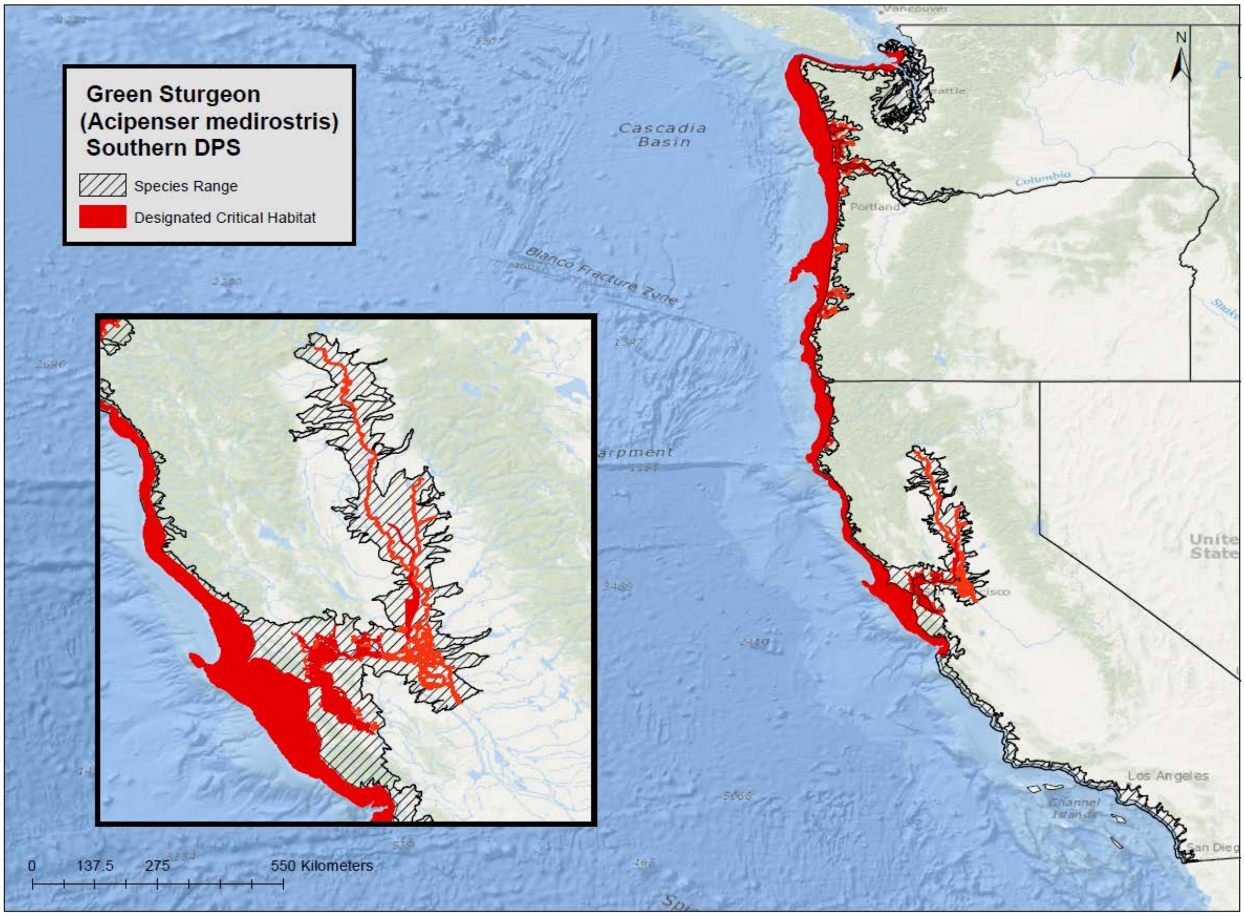


Figure 41. Geographic range (within the contiguous United States) and designated critical habitat for green sturgeon, Southern DPS.

Green sturgeon are long-lived, late-maturing, iteroparous, anadromous species that spawn infrequently in natal streams, and spend substantial portions of their lives in marine waters. Although they are members of the class of bony fishes, the skeleton of sturgeons is composed mostly of cartilage. Sturgeon lack scales; however, they have five rows of characteristic bony plates on their body (called scutes). Green sturgeon have an olive green to dark green back, a yellowish green-white belly (Adams et al. 2002) (Figure 42). NMFS has identified two DPSs (DPS) of green sturgeon; northern and southern (Israel et al. 2009).

Figure 42. Green sturgeon. Photos: Toz Soto, Karuk Tribe Fisheries Dept. (left); K. Newell and J. Daly, NOAA (right).

In 2006, NMFS determined that the Southern DPS green sturgeon warranted listing as a threatened species under the ESA (Table 22). Green sturgeon have been observed in large concentrations in the summer and autumn within coastal bays and estuaries along the west coast of the United States, including the Columbia River estuary, Willapa Bay, Grays Harbor, San Francisco bay and Monterey bay (Huff et al. 2012; Lindley et al. 2011; Lindley et al. 2008; Moser and Lindley 2007).

Table 22. Summary of green sturgeon, Southern DPS listing and recovery plan information.

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acipenser medirostris	Green Sturgeon	Southern	Threatened	2015	71 FR 17757 04/07/2006	Outline 2010	74 FR 52300 2009

Life History

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

Green sturgeon are believed to spend the majority of their lives in nearshore oceanic waters, bays, and estuaries. Younger green sturgeon reside in fresh water, with adults returning to freshwater to spawn when they are about 15 years of age and more than 4 feet (1.3 meters) in size. In preparation for spawning, adult Southern DPS green sturgeon enter San Francisco Bay between mid-February and early-May, and migrate rapidly (on the order of a few weeks) up the Sacramento River (Heublein et al. 2009). Spawning occurs from April through early July, with peaks of activity that depend on a variety of factors including water temperature and water flow rates (Poytress et al. 2009; Poytress et al. 2010). Post-spawn fish typically congregate and hold for several months in a few deep pools in the upper mainstem Sacramento River near spawning sites and migrate back downstream when river flows increase in fall. They re-enter the ocean during the winter months (November through January) and begin their marine migration north along the coast (California Fish Tracking Consortium database).

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

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: population growth rate, genetic diversity, and distribution as it relates to the Southern DPS green sturgeon.

Population Growth Rate

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

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

Genetic Diversity

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

Distribution

This species is found along the west coast of Mexico, the United States, and Canada. Green sturgeon are the most broadly distributed, wide-ranging, and most marine-oriented species of the sturgeon family. The green sturgeon ranges from Mexico to at least Alaska in marine waters, and is observed in bays and estuaries up and down the west coast of North America. Tagged Southern DPS green sturgeon subadults and adults have been detected in coastal marine waters from Monterey Bay, California to Graves Harbor, Alaska, including the Strait of Juan de Fuca and Puget Sound (74 FR 52299)(AquaMaps 2016; Lindley et al. 2011), Washington estuaries within the action area for the proposed Navy ship tow program. Lindley et al. (2011) reported that green sturgeon use the Puget Sound estuary at a low rate, but fish were detected within this estuary in both winter and summer months. More recent preliminary tag results (i.e., data that are in the process of being validated) suggest that green sturgeon may be using the Puget Sound estuary at a higher rate than previously thought. Up to 70 transmitters having tag codes identified as green sturgeon were detected at Admiralty Inlet, the entrance to Puget Sound, between October 2012 and February 2018 (M. Moser, NMFS, pers. comm. to R. Salz, NMFS, February 7, 2019). Although not confirmed, based on the proportions of meta-populations found in other Pacific Northwest estuaries (i.e., Columbia River, Willapa Bay, and Grays Harbor)(Israel et al. 2009; Moser et al. 2016; Schreier et al. 2016), we anticipate the green sturgeon detected at Admiralty Inlet likely represent a mix of both Southern DPS and Northern DPS green sturgeon.

Status

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

Climate change has the potential to impact Southern DPS green sturgeon in the future, but it is unclear how changing oceanic, nearshore and river conditions will affect the Southern DPS overall (NMFS 2015b).

Designated Critical Habitat

Critical habitat was designated for Southern DPS green sturgeon on October 9, 2009, and includes marine, coastal bay, estuarine, and freshwater areas (Figure 41).

Recovery Goals

The final recovery plan for Southern DPS green sturgeon has not been released. The recovery outline (NMFS 2010a) indicates that the recovery potential for Southern DPS green sturgeon is considered moderate to high; however, certain life history characteristics (e.g., long-lived, delayed maturity) indicate recovery could take many decades, even under the best circumstances. According to the recovery outline key recovery needs and implementation measures identified include additional spawning and egg/larval habitat as well as additional research and monitoring (NMFS 2010a).

8.2.14 Gulf Sturgeon

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

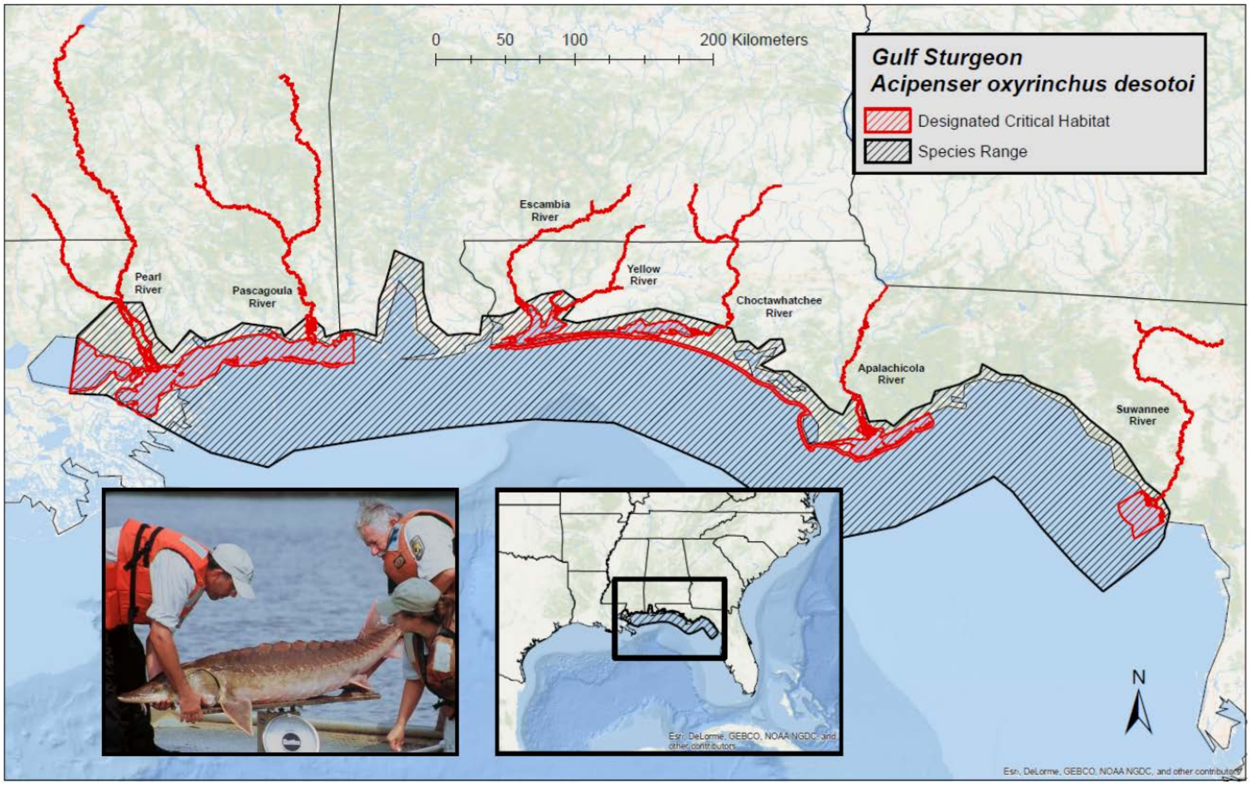


Figure 43. Geographic range and designated critical habitat of the Gulf Sturgeon.

Gulf sturgeon are nearly cylindrical fish with an extended snout, vertical mouth, five rows of scutes (bony plates surrounding the body), four barbels (slender, whisker-like feelers anterior to the mouth used for touch and taste), and a heterocercal (upper lobe is longer than lower) caudal fin (Figure 44). Adults range from 6 to 8 feet in length and weigh up to 200 pounds; females grow larger than males (USFWS 2009b).

Figure 44. Gulf sturgeon. Photo: Dewayne Fox, Delaware State University.

The Gulf sturgeon was listed as Threatened on September 30, 1991 (Table 23).

Table 23. Summary of gulf sturgeon listing and recovery plan information.

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acipenser oxyrinchus desotoi	Gulf Sturgeon	Entire Population	Threatened	2009	56 FR 49653 09/30/1991	1995	68 FR 13370 2003

Life History

Gulf sturgeon are long-lived, with some individuals reaching at least 42 years in age. Surveys in the Suwannee River suggest that a more common maximum age may be around 25 years (Sulak and Clugston 1999). Age at sexual maturity for females ranges from 8 to 17 years, and for males from 7 to 21 years (Huff 1975). In general, gulf sturgeon spawn up-river in spring, spend winter months in near-shore marine environments, and utilize pre- and post-spawn staging and nursery areas in the lower rivers and estuaries (Heise et al. 2005; Heise et al. 2004). There is some evidence of autumn spawning in the Suwannee River, however there is uncertainty as to whether this spawning is due to environmental conditions or represents a genetically distinct population (Randall and Sulak 2012). Gulf sturgeon spawn at intervals ranging from 3 to 5 years for females and 1 to 5 years for males (Fox et al. 2000; Smith 1985). The spring migration to up-river spawning sites begins in mid-February and continues through May. Fertilization is external; females deposit their eggs in the upper reaches of and show preference for hard, clean substrate (e.g. bedrock covered in gravel and small cobble).

Upon hatching from their eggs, gulf sturgeon larvae spend the first few days of life sheltered in interstitial spaces at the spawning site (Kynard and Parker 2004). At the onset of feeding, age-0 gulf sturgeon disperse and are often found on shallow sandbars and rippled sand shoals (less than 4 meters depth) (Sulak and Clugston 1998). Young-of-the-year spend 6 to 10 months slowly working their way downstream feeding on aquatic insects (e.g., mayflies and caddisflies), worms (oligochaetes), and bivalve mollusks, and arrive in estuaries and river mouths by mid-winter (Sulak and Clugston 1999) where they will spend their next 6 years developing. After spawning, adult gulf sturgeon migrate downstream to summer resting and holding areas in the mid to lower reaches of the rivers where they may hold until November (Wooley and Crateau 1985). While in freshwater adults lose a substantial amount of their weight, but regain it upon entering the estuaries. Sub adult and non-spawning adults also spend late spring through fall in these holding areas (Foster and Clugston 1997). By early December all adult and sub-adult gulf sturgeon return to the marine environment to forage on benthic (bottom dwelling) invertebrates along the shallow nearshore (2 to 4 meter depth), barrier island passes, and in unknown off-shore locations in the gulf (Carr et al. 1996; Fox et al. 2002; Huff 1975; Ross et al. 2009). Juvenile gulf sturgeon overwinter in estuaries, river mouths, and bays; juveniles do not enter the nearshore/offshore marine environments until around age 6 (Sulak and Clugston 1999). Gulf sturgeon show a high degree of river-specific fidelity (Rudd et al. 2014). Adult and sub-adult gulf sturgeon fast while in freshwater environments and are almost entirely dependent on the estuarine/marine environment for food (Gu et al. 2001; Wooley and Crateau 1985). Some juveniles (ages 1 to 6) will also fast in the freshwater summer holding areas, but the majority feed year round in the estuaries, river mouths, and bays (Sulak et al. 2009).

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: population growth rate, genetic diversity, and distribution as it relates to Gulf sturgeon.

Population Growth Rate

Currently, seven rivers are known to support reproducing populations of Gulf sturgeon. The most recent abundance estimates reported in the 5-Year Review (USFWS 2009b) (Table 24).

Table 24. Gulf sturgeon abundance estimates by river and year, with confidence intervals for the seven major rivers with reproducing populations. Table modified from (USFWS 2009b).

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

[a] Estimates refer to numbers of individuals greater than a certain size, which varies between studies depending on sampling gear, and in some cases, numbers of individuals that use a particular portion of the river. Refer to original publication for details.
[b] Large confidence intervals around the mean estimates reflect the low capture probability in mark-recapture studies.

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

Genetic Diversity

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

Distribution

Gulf sturgeon inhabit coastal rivers from Louisiana to Florida during the warmer months, and the Gulf of Mexico and its estuaries and bays in the cooler months. Gulf sturgeon are anadromous: adults spawn in freshwater and migrate into marine waters in the fall to forage and overwinter. Juvenile Gulf sturgeon stay in the river for about the first 2 to 3 years. Gulf sturgeon return to their natal stream to spawn. Gulf sturgeon initiate movement up to the rivers between February and April and migrate back out to the Gulf of Mexico between September and November.

Status

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

Designated Critical Habitat

NMFS and the U.S. Fish and Wildlife Service (USFWS) jointly designated Gulf sturgeon critical habitat on April 18, 2003 (50 CFR §226.214). The agencies designated seven riverine areas (Units 1 to 7) encompassing 2,783 river kilometers and seven estuarine/marine areas (Units 8 to 14) encompassing 6,042 square kilometers as critical habitat based on the PBFs that support the species (Figure 43). PBFs considered essential for the conservation of Gulf Sturgeon are abundant food items, riverine spawning sites with substrates suitable for egg deposition and development, riverine aggregation areas, a flow regime necessary for normal behavior, growth, and survival, water and sediment quality necessary for normal behavior, growth, and viability of all life stages, and safe and unobstructed migratory pathways.

Recovery Goals

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

8.2.15 Shortnose Sturgeon

The shortnose sturgeon is the smallest of the three sturgeon species that occur in eastern North America; they grow up to 4.7 feet (1.4 meters) and weigh up to 50.7 pounds (23 kilograms). It has a short, conical snout with four barbells in front of its large underslung mouth. Five rows of bony plates (called scutes) occur along its body: one on the back, two on the belly, and one on each side. The body coloration is generally olive-yellow to gray or bluish on the back, and milky-white to dark yellow on the belly. The peritoneum (body cavity lining) is black (Figure 45).

Figure 45. Shortnose sturgeon. Photo: Nancy Haley, NOAA (left), University of Maine (right).

Shortnose sturgeon were initially listed as endangered on March 11, 1967 under the Endangered Species Preservation Act of 1966. In 1994, the species was listed as endangered throughout its range under the ESA (Table 25). Critical habitat has not been designated for shortnose sturgeon. Shortnose sturgeon occur along the Atlantic Coast of North America, from the St. John River in Canada to the St. Johns River in Florida (Figure 46).

Table 25. Summary of shortnose sturgeon listing and recovery plan information.

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Acipenser brevirostrum	Shortnose sturgeon	N/A	Endangered	2010	32 FR 4001 03/11/1967	63 FR 69613 1998	None Designated

Life History

The shortnose sturgeon is a relatively slow growing, late maturing, and long-lived fish species. The maximum recorded size of shortnose sturgeon was collected from the Saint John River, Canada, measuring 143 centimeters total length and weighing 23 kilograms (Dadswell et al. 1984). Shortnose sturgeon typically live longer in the northern portion of their range compared to the southern portion (Dadswell et al. 1984; Gilbert 1989). The maximum ages reported of female shortnose sturgeon by river system include 67 years for the St. John River (New Brunswick), 40 years for the Kennebec River, 37 years for the Hudson River, 34 years for the Connecticut River, 20 years for the Pee Dee River, and ten years for the Altamaha River (Dadswell et al. 1984; Gilbert 1989). Female shortnose sturgeon generally outlive and outgrow males, which seldom exceed 30 years of age (Dadswell et al. 1984; Gilbert 1989). Shortnose sturgeon also exhibit sexually dimorphic growth and maturation patterns across latitudes (Dadswell et al. 1984). In the northern parts of its range, males reach maturity at 5 to 11 years, while females mature between 7 and 18 years. Shortnose sturgeon in southern rivers typically grow faster, mature at younger ages (2 to 5 years for males and 4 to 5 for females), but attain smaller maximum sizes than those in the north which grow throughout their longer lifespans (Dadswell et al. 1984).

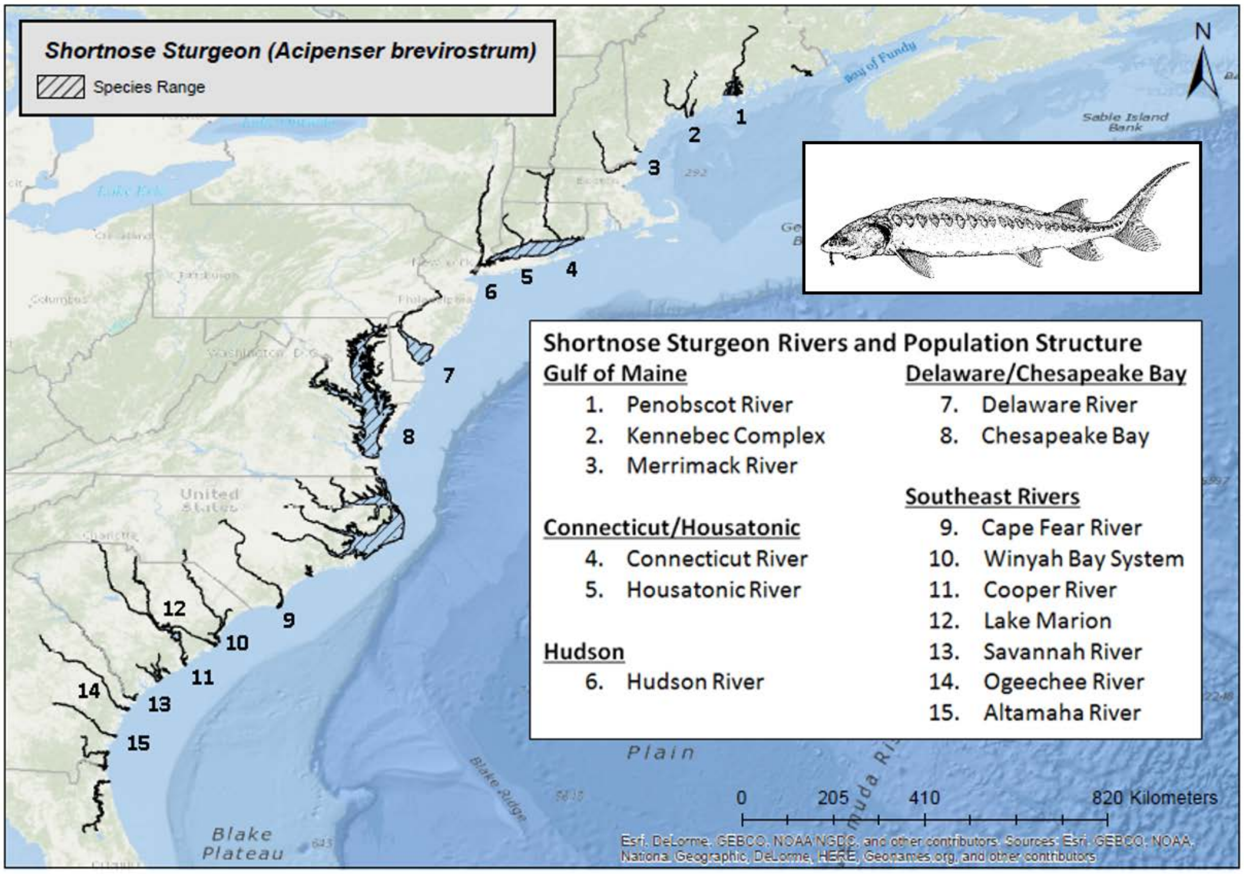


Figure 46. Geographic range of shortnose sturgeon.

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

Shortnose sturgeon have been found in waters with temperatures as low as 2 to 3ºC (Dadswell et al. 1984) and as high as 34ºC (Heidt and Gilbert 1979). However, temperatures above 28ºC are thought to adversely affect shortnose sturgeon (Kynard 1997). Shortnose sturgeon are known to occur at a wide range of depths from a minimum depth of 0.6 m up to 30 m (Dadswell 1979; Dadswell et al. 1984). Shortnose sturgeon exhibit tolerance to a wide range of salinities from freshwater (Taubert 1980) to waters with salinity of 30 parts-per-thousand (Holland and Yelverton 1973). Shortnose sturgeon typically occur in the deepest parts of rivers or estuaries where suitable oxygen and salinity levels are present (Gilbert 1989).

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

Estimates of annual egg production for shortnose sturgeon are difficult to calculate and are likely to vary greatly in this species because females do not spawn every year. Fecundity estimates that have been made range from 27,000 to 208,000 eggs/female, with a mean of 11,568 eggs/kg body weight (Dadswell et al. 1984). At hatching, shortnose sturgeon are 7 to 11 millimeters long and resemble tadpoles (Buckley and Kynard 1981). In 9 to 12 days, the yolk sac is absorbed and the sturgeon develops into larvae which are about 15 millimeters total length (Buckley and Kynard 1981). Sturgeon larvae are believed to begin downstream migrations at about 20 millimeters total length.

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

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: population growth rate, genetic diversity, and distribution as it relates to shortnose sturgeon.

Population Growth Rate

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

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

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

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

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

Genetic Diversity

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

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

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

Distribution

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

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

Status

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

Designated Critical Habitat

No critical habitat has been designated for the shortnose sturgeon.

Recovery Goals

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

8.2.16 Steelhead Trout, Puget Sound DPS and Designated Critical Habitat

This DPS, or DPS, includes naturally spawned anadromous *Oncorhynchus mykiss* (steelhead) originating below natural and manmade impassable barriers from rivers flowing into Puget Sound from the Elwha River (inclusive) eastward, including rivers in Hood Canal, South Sound, North Sound and the Strait of Georgia (Figure 47). Also, steelhead from six artificial propagation programs.

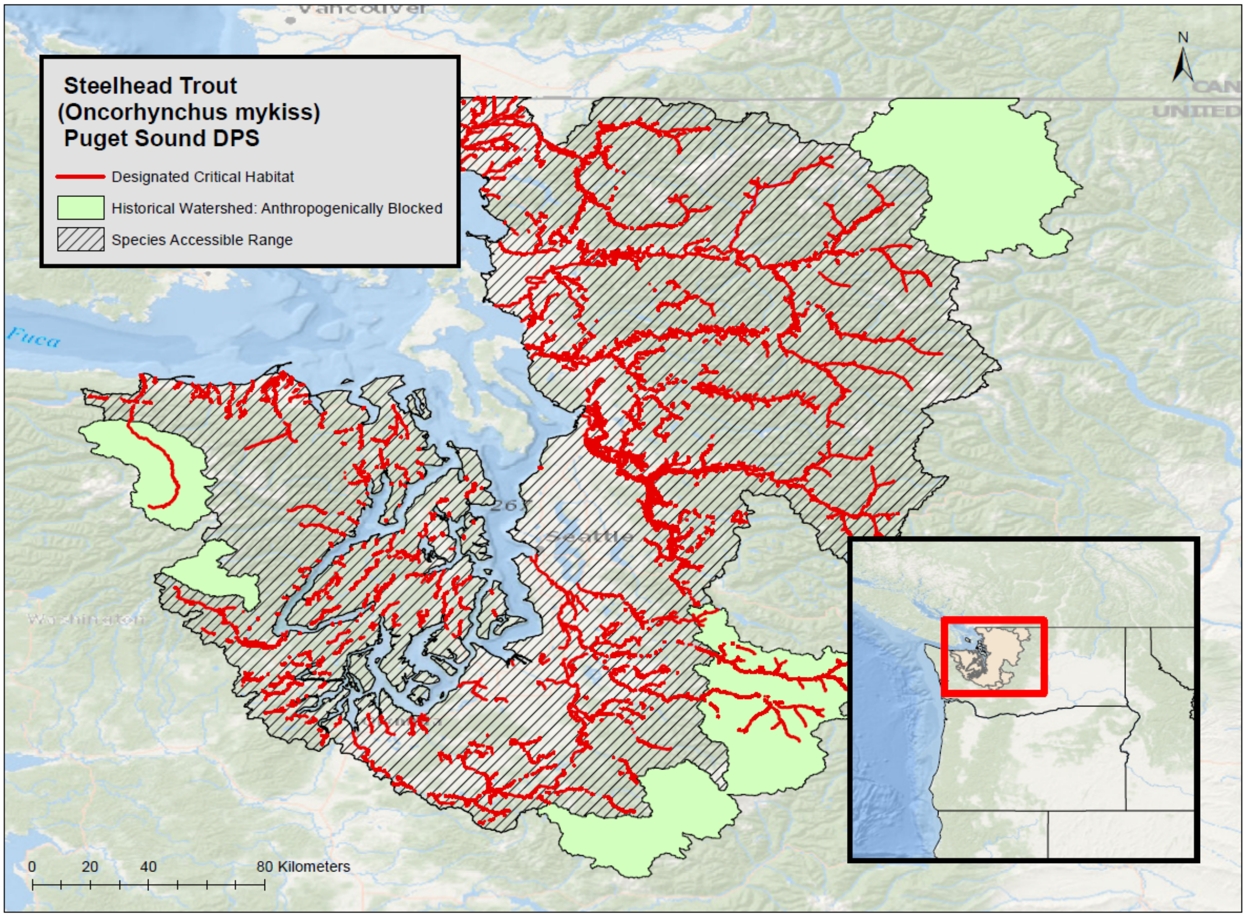


Figure 47. Geographic range and designated critical habitat for the Puget Sound DPS of steelhead trout.

Steelhead are dark-olive in color, shading to silvery-white on the underside with a speckled body and a pink-red stripe along their sides (Figure 48). Those migrating to the ocean develop a slimmer profile, becoming silvery in color, and typically growing larger than rainbow trout that remain in fresh water. Steelhead trout grow to 55 pounds (25 kilograms) in weight and 45 inches (120 centimeters) in length, though average size is much smaller.

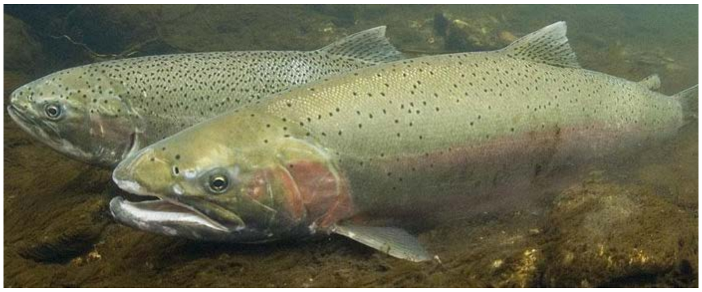



Figure 48. Steelhead trout. Photo: NOAA (left).

On May 11, 2007 NMFS listed the Puget Sound DPS of steelhead as threatened (Table 26).

Table 26. Summary of steelhead trout, Puget Sound DPS listing and recovery plan information.

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Oncorhynchus mykiss	Steelhead Trout	Puget Sound	Threatened	2011	72 FR 26722 05/11/2007	None	81 FR 9251 2016

Life History

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

The majority of juveniles reside in the river system for two years with a minority migrating to the ocean as one or three-year olds. Smoltification and seaward migration occur from April to mid-May. The ocean growth period for Puget Sound steelhead ranges from one to three years in the ocean (Busby et al. 1996). Juveniles or adults may spend considerable time in the protected marine environment of the fjord-like Puget Sound during migration to the high seas.

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: population growth rate, genetic diversity, and distribution as it relates to Puget Sound steelhead.

Population Growth Rate

Abundance of adult steelhead returning to nearly all Puget Sound Rivers has fallen substantially since estimates began for many populations in the late 1970s and early 1980s. Inspection of geometric means of total spawner abundance from 2010 to 2014 indicates that 9 of 20 populations evaluated had geometric mean abundances fewer than 250 adults and 12 of 20 had fewer than 500 adults.

Smoothed trends in abundance indicate modest increases since 2009 for 13 of the 22 populations. Between the two most recent five-year periods (2005 to 2009 and 2010 to 2014), the geometric mean of estimated abundance increased by an average of 5.4 percent. For seven populations in the Northern Cascades subregion, the increase was 3 percent; for five populations in Central and South Puget Sound subregion, the increase was 10 percent; and for six populations in Hood Canal and Strait of Juan de Fuca subregion, the increase was 4.5 percent. However, several of these upward trends are not statistically different from neutral, and most populations remain small. Long-term (15-year) trends in natural spawners are predominantly negative (NMFS NWFSC 2015).

Genetic Diversity

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

Distribution

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

Status

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

Designated Critical Habitat

NMFS designated critical habitat for Puget Sound steelhead on February 2, 2016 (Figure 47). Critical habitat includes freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, nearshore marine areas, and offshore marine areas. The essential PBFs that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity necessary to support spawning, incubation and larval development, juvenile growth and mobility, and adult survival. Of 70 assessed watersheds, 41 were assigned a high and 18 were assigned a medium conservation value. The remaining watersheds were either of low conservation value, or have been proposed to be excluded for economic considerations. The Puget Sound Naval Shipyard at Naval Base Kitsap was excluded from critical habitat designation for Puget Sound DPS steelhead.

Recovery Goals

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

8.2.17 Yelloweye Rockfish, Puget Sound/Georgia Basin DPS and Designated Critical Habitat

Yelloweye rockfish occur throughout most of the eastern Pacific Ocean ranging from northern Baja California to the Aleutian Islands, Alaska. The Puget Sound/Georgia Basin DPS is located along the coastal/inlet waters off the state of Washington and province of British Columbia and is the only population listed under the ESA (Figure 49).

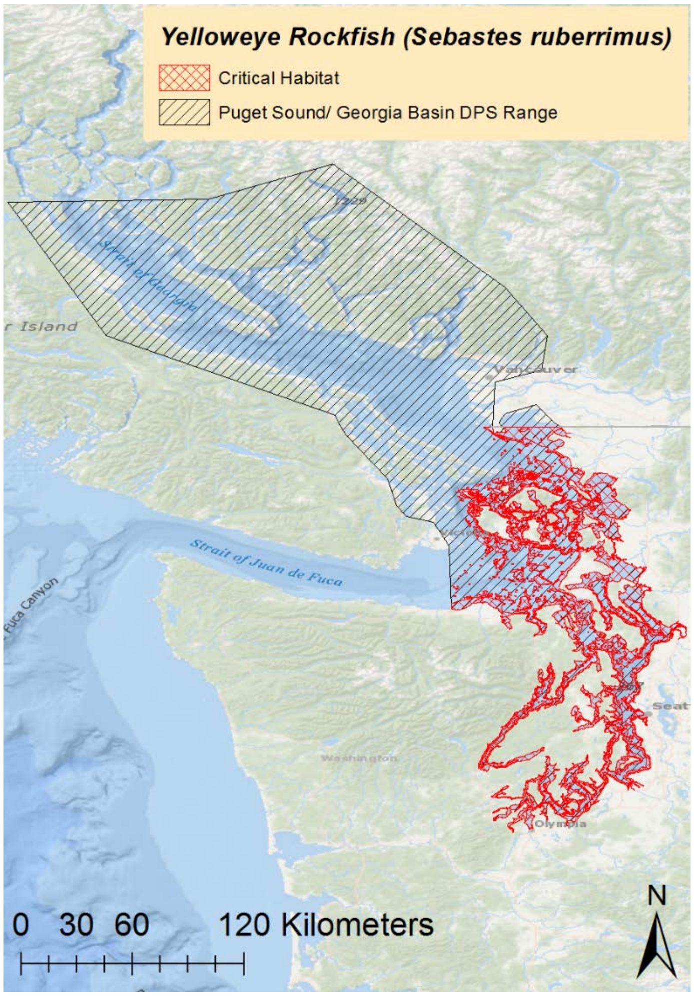


Figure 49. Geographic range and designated critical habitat for yelloweye rockfish, Puget Sound/Georgia Basin DPS.

Yelloweye rockfish is one of the largest species belonging to the genus *Sebastes*. They are orange-red to orange-yellow in color and may have black fin tips with bright yellow eyes. Adults usually have a light to white stripe on the lateral line; juveniles have two light stripes, one on the lateral line and a shorter one below the lateral line (Yamanaka et al. 2006) (Figure 50).

Figure 50. Yelloweye rockfish. Photo: NOAA.

Puget Sound/Georgia Basin yelloweye rockfish were listed on the ESA as threatened on April 28, 2010 (Table 27).

Table 27. Summary of yelloweye rockfish, Puget Sound/Georgia Basin DPS listing and recovery plan information.

Species	Common Name	DPS	ESA Status	Recent Review Year	Listing	Recovery Plan	Critical Habitat
Sebastes ruberrimus	Yelloweye Rockfish	Puget Sound/ Georgia Basin	Threatened	2016	75 FR 22276 04/28/2010	2017	79 FR 68041 2014

Population Dynamics

The following is a discussion of the species' population and its variance over time. This section includes: population growth rate, genetic diversity, and distribution as it relates to yelloweye rockfish, Puget Sound/Georgia Basin DPS.

Population Growth Rate

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

Yelloweye rockfish were 2.4 percent of the rockfish harvest in the North Sound during the 1960s, 2.1 percent of the harvest during the 1980s, and further decreased to an average of one percent from 1996 to 2002 (Palsson et al. 2009). In Puget Sound proper, yelloweye rockfish were 4.4 percent of the rockfish harvest during the 1960s, 0.4 percent during the 1980s, and 1.4

percent from 1996 to 2002 (Palsson et al. 2009). By the 2000s, evidence of fewer older fish in the population prevailed. Since overall fish numbers in the database were also much lower, it is difficult to determine if size truncation occurred.

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

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

Genetic Diversity

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

Distribution

The leading factors affecting diversity are the relatively small home ranges of juveniles and subadults (Love et al. 2002) and low population size of all life stages. Yelloweye rockfish spatial structure and connectivity are likely threatened by the apparently severe reduction of fish numbers throughout Hood Canal and South Puget Sound. At 2,330 square kilometers, Puget

Sound is a small geographic area compared with the entire yelloweye rockfish range in the northeastern Pacific.

Status

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

Designated Critical Habitat

Critical habitat was designated for Puget Sound/Georgia Basin yelloweye rockfish on November 13, 2014, when NMFS published a final rule in the Federal Register. The critical habitat in the United States is spread amongst five interconnected, biogeographic basins (San Juan/Strait of Juan de Fuca basin, Main basin, Whidbey basin, South Puget Sound, and Hood Canal) based upon presence and distribution of adult and juvenile yelloweye rockfish, geographic conditions, and habitat features (Figure 49). The Puget Sound Naval Shipyard at Naval Base Kitsap was excluded from critical habitat designation for Puget Sound/Georgia Basin yelloweye rockfish.

Recovery Goals

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

9 Environmental Baseline

The "environmental baseline" includes the past and present impacts of all Federal, state, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 C.F.R. §402.02). The environmental baseline for this opinion includes the effects of several activities affecting the survival and recovery of proposed or listed species as well as their proposed or designated critical habitats in the action area.

Since the action area for this consultation is so large, encompassing ports throughout the continental United States and Hawaii, as well as open ocean environments along the tow routes, it is not practical to provide a detailed description of the environmental baseline in all locations throughout the entire action area. Instead, we provide a more general overview of several past and ongoing anthropogenic threats to ESA-listed species within the action area. We then provide a description and characterization of the origination and destination ports for the proposed action.

9.1 Fisheries

Bycatch occurs when fisheries interact with living marine resources (e.g., marine mammals, sea turtles, non-market fish species, corals, or seabirds) that are not the target species for commercial sale. Bycatch represents a global threat to many ESA-listed species. Populations of marine megafauna (e.g., turtles, mammals, sharks) can be particularly sensitive to the detrimental effects of bycatch due to life history parameters such as slow growth, late age at maturity, and low reproductive rates (Hall et al. 2017). Highly migratory, transboundary species that spend large amounts of time in ocean jurisdictions lacking adequate bycatch mitigation measures, monitoring, or enforcement are often most vulnerable to this threat.

While mitigation and minimization measures have reduced fisheries bycatch in the United States in recent years, large numbers of ESA-listed species are still routinely captured in federal and state commercial fisheries targeting other species. Some ESA-listed species also interact with recreational hook-and-line fisheries. Fisheries management plans (FMPs) developed for federally regulated fisheries with ESA-listed species bycatch are required to undergo section 7 consultation, including a NMFS issued opinion and an ITS. The ITS includes the anticipated amount of take (lethal and nonlethal) and reasonable and prudent measures with specific terms and conditions for mitigating and minimizing the adverse effects of the proposed action on ESA-listed species and designated critical habitat. Some state-managed fisheries with ESA-listed species bycatch have also been the subject of section 7 consultations with NMFS for issuance of ESA section 10(a)(1)(B) incidental take permits (ITPs). ITPs are issued based on NMFS approval of a state's Conservation Plan, which includes ESA-listed species mitigation and minimization measures.

9.1.1 Cetaceans

Killer Whale, Southern Resident DPS

Entrapment and entanglement in fishing gear is a frequently documented source of human-caused mortality in cetaceans (see Dietrich et al. 2007). Materials entangled tightly around a body part may cut into tissues, enable infection, and severely compromise an individual's health (Derraik 2002). Entanglements also make animals more vulnerable to additional threats (e.g., predation and ship strikes) by restricting agility and swimming speed. The majority of cetaceans that die from entanglement in fishing gear likely sink at sea rather than strand ashore, making it difficult to accurately determine the extent of such mortalities. When not immediately fatal, entanglement or ingestion of fishing gear can impede the ability of marine mammals to feed and can cause injuries that eventually lead to infection and death (Cassoff et al. 2011; Moore and Van der Hoop 2012; Wells et al. 2008). There are also costs likely to be associated with nonlethal entanglements in terms of energy and stress (Moore and Van der Hoop 2012). Necropsies of stranded whales have found that ingestion of net pieces, ropes, and other fishing debris has resulted in gastric impaction and ultimately death (Jacobsen et al. 2010). As with ship strikes, entanglement or entrapment in fishing gear likely has the greatest impact on populations of ESA-listed species with the lowest abundance (e.g., Kraus et al. 2016). Nevertheless, all species of cetacean may face threats from derelict fishing gear.

In addition to these direct impacts, cetaceans may also be subject to indirect impacts from fisheries. Many cetacean species are known to feed on species of fish that are harvested by humans (Carretta et al. 2016). Thus, competition with humans for prey is a potential concern. Reductions in fish populations, whether natural or human-caused, may affect the survival and recovery of ESA-listed populations. Even species that do not directly compete with human fisheries could be indirectly affected by fishing activities through changes in ecosystem dynamics.

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

9.1.2 Pinnipeds

Hawaiian Monk Seal

Hawaiian monk seals have one of the highest documented entanglement rates of any pinniped species, and marine debris and derelict fishing gear are chronic forms of pollution affecting the northwest Hawaiian Islands. There is serious concern for the entanglement of monk seals, especially since the number of monk seals found entangled has not changed nor has there been a reduction in the accumulation rates of marine debris in the northwest Hawaiian Islands. The principle, direct fishery interaction threat currently facing monk seals are main Hawaiian Islands recreational net fisheries. In particular, gillnets and shore-cast gear, which are managed by the State of Hawaii, are known to cause monk seal mortalities (NMFS 2007b). Two monk seals drowned in recreational gillnets on Oahu in 2007 (NMFS 2007b). Fishing in the U.S. Exclusive Economic Zone (EEZ) around the Hawaiian Archipelago is managed by NMFS through the advisory of the Western Pacific Regional Fishery Management Council. In 2016, NMFS released the Main Hawaiian Islands Monk Seal Management Plan. This plan is an important step in addressing challenges between seals and humans, including an outline for fishery partnerships to reduce harmful monk seal-fishery interactions (NMFS 2016d).

Of 43 monk seal/fishery interactions in the main Hawaiian Islands from 1982 to 2006, one seal was found dead in a nearshore gillnet in 1994, a second seal was disentangled by recreational divers from a nearshore gillnet in 2002, and a third seal was temporarily entangled in a nearshore gillnet in 2005, but escaped unaided (NMFS 2007e). In October 2006, a female, juvenile Hawaiian monk seal was found drowned in a gill net about three months after her weaning. In June 2007, an adult male Hawaiian monk seal at Makua Beach on the island of Oahu was found dead and densely wrapped in gill netting. The monk seal drowned in the illegal net that was not properly tagged with identification markers under regulations passed by the State of Hawaii in March 2007. A total of 38 seals have been observed with embedded hooks in the main Hawaiian Islands during 1982 to 2006 (Caretta et al. 2005; NMFS 2007e). A monk seal was found dead in 1995 with a hook lodged in its esophagus. For most of the interactions, the hooks were not always recovered, and it was not possible to attribute each hooking event to a specific fishery. Among hooks that could be identified, the sources included nearshore fisheries (esp. for *Caranx sp.* in the main Hawaiian Islands) in state waters, and bottomfish (handline) fisheries in state and federal waters (NMFS 2007e). There is a continuing need for intervention for Hawaiian monk seals in the main Hawaiian Islands to remove embedded hooks from recreational fishing; however this effort does not remedy the interaction problem itself.

9.1.3 Sea Turtles

Bycatch of ESA-listed sea turtles occurs in a diversity of fisheries throughout the broad geographic oceanic ranges of these species. Sea turtle bycatch occurs in both large-scale commercial fishing operations as well as small-scale, artisanal fisheries throughout the world. Fishing gears that are known to interact with sea turtles include trawls, longlines, purse seines, gillnets, pound nets, dredges and to a lesser extent, pots and traps (Finkbeiner et al. 2011; Lewison et al. 2013).

Sea turtle bycatch rates (i.e., individuals captured per unit of fishing effort) and mortality rates (i.e., individuals killed per number captured) can vary widely both within and across particular fisheries due to a combination of factors. These include gear types and gear configurations, fishing methods (e.g., depth fished, soak times), fishing locations, fishing seasons, time fished (i.e., day versus night), and turtle handling and release techniques used (Lewison et al. 2013; Wallace et al. 2010b). Henwood (1987) found a strong positive correlation between shrimp trawl tow time and mortality rate of turtles bycaught in commercial shrimp trawlers. Similarly, Murray (2009) found that sea turtle mortality rates in sink gillnet gear was largely a function of soak time. Differences in bycatch rates among gear deployment practices and gear configurations have driven many of the bycatch reduction strategies in longline ships (Lewison et al. 2013; Watson et al. 2005). Shallow-set longlines (less than 50 meters) have been shown to result in higher turtle bycatch rates than deeper sets (Beverly et al. 2009; Gilman et al. 2006); leatherbacks are caught more often during nighttime longline sets compared to daytime sets; increased longline soak times have resulted in higher catches of loggerhead turtles (Gilman et al. 2006); and switching from J-shaped hooks with squid bait to circle hooks with fish bait resulted in significant declines in loggerhead (83 percent) and leatherback (90 percent) bycatch in the Hawaii longline swordfish fishery (Gilman et al. 2007). Estimated turtle mortality rates from capture in longline gear have also been shown to vary widely (8 percent to over 30 percent) depending on numerous factors including hook type used, set depth, and hook location (Casale et al. 2008; Chaloupka et al. 2004).

If mortality is not directly observed during gear retrieval, it may occur after the turtle is released due to physiological stress and injury suffered during capture. Entanglement in fishing gear and/or plastics can result in severe ulcerative dermatitis, and amputation of flippers (Orós et al. 2005). Although rates of post-release mortality and serious injury are essential to understanding the impact of bycatch on sea turtle populations, it is a major knowledge gap for many fisheries that interact with turtles (Lewison et al. 2013).

There have been some major advancements in sea turtle bycatch reduction technologies and management approaches in the past few decades. Direct gear and fishery modifications such as changes to bait type, modifying gear to make it less visible or attractive to sea turtles, making gear less likely to cause direct mortality, or changing the way that gear is deployed are all examples of bycatch mitigation techniques that have been employed to reduce sea turtle bycatch in trawl, passive net, and longline large-scale fisheries (Hall et al. 2017; Lewison et al. 2013). Time-area closures have also proven effective at reducing sea turtle bycatch in commercial fishing gear (Dunn et al. 2011). Swimmer et al. (2017) analyzed 20 years of U.S. longline observer data from the Atlantic and Pacific Ocean basins during periods before and after sea turtle bycatch reduction regulations to assess the effectiveness of the regulations. They found that in two federally managed longline fisheries, rates of sea turtle bycatch significantly declined after the regulations. Capture probabilities were lowest when using a combination of circle hooks (versus J-hooks) and fish bait (versus squid bait). In the Atlantic (all regions), rates declined by 40 and 61 percent for leatherback and loggerhead turtles, respectively, after the regulations. In the Pacific shallow set fishery, mean bycatch rates declined by 84 and 95 percent, for leatherback and loggerhead turtles, respectively, for the post-regulation period (Swimmer et al. 2017).

In 2003, NMFS developed a National Bycatch Strategy that identified concrete actions necessary for reducing bycatch in U.S. fisheries (Benaka and Dobrzynski 2004). This document was recently updated and expanded to enhance the effectiveness of existing bycatch reduction approaches. The 2016 National Bycatch Reduction Strategy identifies several key objectives including: (1) improved monitoring of bycatch and bycatch mortality, (2) conduct research to improve bycatch estimates, (3) implement management measures to further reduce the effects of bycatch, and (4) more emphasis on enforcement to ensure compliance with bycatch measures (NMFS 2016e). The most effective way to monitor sea turtle bycatch is to place trained observers aboard fishing ships. Although observer programs have increased in recent decades, many fisheries still lack the level of observer coverage necessary to produce reliable estimates of bycatch and associated mortalities needed to assess fishery impacts on ESA-listed species. In 2007, NMFS established a new regulation (72 FR 43176) to annually review sea turtle interactions across fisheries, identify those that require monitoring, and require fishermen to accommodate observers if requested. This annual process should help NMFS and the fishing industry learn more about sea turtle interactions with fishing operations, continually evaluate existing measures to reduce sea turtle takes, and determine whether additional measures to address prohibited sea turtle takes may be necessary to avoid exceeding established take limits.

Atlantic Ocean Basin, Gulf of Mexico and Caribbean

The primary turtle species captured in U.S. fisheries in Atlantic and Gulf of Mexico is the loggerhead (Moore et al. 2009). The southeastern United States comprises one of the largest aggregate nesting rookeries for loggerhead sea turtles in the world, and the continental shelf provides critical ontogenetic habitats for this population. Thus, because a large number of individuals are present throughout areas of high fishing activity, loggerheads interact with a greater number of fishing fleets and gear types in the Atlantic than other sea turtle species (Moore et al. 2009).

The Southeast shrimp trawl fishery in the Atlantic and Gulf of Mexico has historically accounted for the overwhelming majority (up to 98 percent) of sea turtle bycatch in U.S. fisheries (Finkbeiner et al. 2011). Regulations that went into effect in the early 1990's require shrimp trawlers in the Atlantic and Gulf of Mexico to modify their gear with turtle excluder devices (TEDs) designed to allow turtles to escape trawl nets and avoid drowning. Although mitigation measures have greatly reduced the impact on sea turtle populations, the shrimp trawl fishery is still responsible for large numbers of turtle mortalities each year. The Gulf of Mexico fleet accounts for a large percentage of the sea turtle bycatch in this fishery. In 2010, the Gulf of

Mexico shrimp trawl fishery had an estimated bycatch mortality of 5,166 turtles (18 leatherback, 778 loggerhead, 486 green and 3,884 Kemp's ridley). By comparison, the southeast Atlantic fishery had an estimated bycatch mortality of 1,033 turtles (8 leatherback, 673 loggerhead, 28 green and 324 Kemp's ridley) in 2010 (NMFS 2014c).

The U.S. Atlantic pelagic longline fishery began in the early 1960s. This fishery is currently comprised of five distinct fishing sectors: Gulf of Mexico yellowfin tuna fishery; southern Atlantic swordfish fishery; Mid-Atlantic and New England swordfish and tuna fishery; U.S. Atlantic Distant Water swordfish fishery; and the Caribbean tuna and swordfish fishery. The pelagic longline fishery mainly interacts with leatherback sea turtles and pelagic juvenile loggerhead sea turtles. The estimated average annual bycatch in this fishery (all geographic areas combined) between 1992 to 2002 was 912 loggerhead interactions (including seven captured dead) and 846 leatherback interactions (including 11 captured dead) (NMFS 2004). These mortality estimates do not account for post-release mortality, which historically was likely substantial (NMFS 2014c).

Since 2004, bycatch estimates for both loggerheads and leatherbacks in pelagic longline gear have been well below the average prior to implementation of gear regulations under the section 7 consultation reasonable and prudent measure (Garrison, Stokes, & Fairfield, 2012). The pelagic longline fishery resulted in an estimated 259 loggerhead and 268 leatherback sea turtle interactions in 2014 (NMFS, 2015). NMFS is currently reviewing a request for reinitiation of section 7 consultation on the pelagic longline fishery.

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

Gillnets and bottom trawls are commonly used gears by many of the commercial fisheries operating in the northeastern U.S. Atlantic EEZ from North Carolina through Maine. These fisheries are also known to interact with large numbers of sea turtles, particularly loggerheads. Following an event in which over 200 sea turtle carcasses washed ashore in an area where large mesh gillnetting had been occurring, NMFS published new restrictions for the use of gill nets with larger than 8-inch stretched mesh, in the EEZ off of North Carolina and Virginia (67 FR 71895, December 3, 2002). This rule was in response to a direct need to reduce the impact of this fishery on sea turtles. The rule was subsequently modified on April 26, 2006, by requiring the use of gillnets with greater than or equal to 7-inch stretched mesh when fished in federal waters from the North Carolina/South Carolina border to Chincoteague, Virginia.

In 2013, NMFS issued a “batched” section 7 opinion on the following fisheries: Northeast multispecies; monkfish; spiny dogfish; Atlantic bluefish; Northeast skate complex; mackerel/squid/butterfish; and summer flounder /scup/black sea bass (NMFS, 2013). Gill net gear is used by five of the seven fisheries, and bottom trawl gear is used by six of the seven fisheries covered by this opinion. The “batched” FMP opinion includes an ITS (amended March 10, 2016) that exempts take of ESA-listed sea turtles.

In 2009, NMFS issued an opinion on the continued authorization of the Gulf of Mexico stone crab fishery that occurs off Florida’s west coast (NMFS 2009). Commercial and recreational crab traps and pots authorized in this fishery can adversely affect sea turtles via entanglement and forced submergence. Sea turtles occasionally interact with commercial gillnets targeting king mackerel, Spanish mackerel, and cobia managed under the coastal migratory pelagics FMP. In 2015, NMFS issued an opinion on reinitiation of a section 7 consultation on the FMP for coastal migratory pelagics in the Atlantic and Gulf of Mexico (NMFS 2015a). The South Atlantic commercial snapper-grouper fishery occurs in the U.S. federal waters from North Carolina through Florida. Four fishing methods are used in this fishery: vertical line (handline, hydraulic, or electric), longline, black sea bass pots, and powerheads or spears. In 2006, NMFS issued an opinion on the continued authorization of the South Atlantic snapper-grouper fishery (NMFS 2006b). In 2011, NMFS issued an opinion on the continued authorization of the reef fish fisheries in the Gulf of Mexico (NMFS 2011b). Fishing gears used in these fisheries that may interact with sea turtles include commercial bottom longlines and bandit rigs (a type of vertical line gear), and recreational hook-and-line gear. This opinion also addresses the anticipated take of sea turtles by these fisheries due to ship strikes.

Various fishing gears (e.g., trawls, pots, pound nets and gillnets) used in state waters from Maine through Texas are known to incidentally take sea turtles. However, information on turtle bycatch in these coastal, nearshore fisheries is often sparse. State managed fisheries bycatch estimates are often based on extremely low observer coverage, and historically sea turtles have not always been included in the data collection effort. Thus, these data may provide insights into gear interactions that could potentially occur but are often not indicative of the magnitude of the overall threat to sea turtles from bycatch in state fisheries. Although the past and current effects of state managed fisheries on sea turtles is currently not determinable, NMFS believes that ongoing state fishing activities may be responsible for seasonally high levels of observed sea turtles strandings in state waters on both the Atlantic and Gulf of Mexico coasts.

In 2002, NMFS issued an opinion on sea turtle conservation measures for the Virginia pound net fishery in response to a proposed NMFS rule that prohibited pound net leaders measuring 12 inches and greater (stretched mesh) and pound net leaders with stringers in the Virginia waters of the mainstem Chesapeake Bay and portions of Virginia tributaries from May 8 to June 30 each year. Another formal consultation on the Virginia pound net fishery was completed in 2004 in response to new information on the effects of the fishery and issuance of a proposed rule prohibiting the use of offshore pound net leaders. The 2004 opinion included an ITS for the anticipated incidental take of loggerhead (507), leatherback (2), Kemp's ridley (103), and green sea turtles (3) in the Virginia pound net fishery. Section 7 consultation on this action was reinitiated in 2017.

In 2013, NMFS issued an incidental take section 10(a)(1)(B) permit (Permit No. 16230) to the state of North Carolina authorizing the take of listed sea turtles incidental to large and small mesh gill net fishing. Authorized take in this fishery includes an estimated 165 green sea turtle mortalities and 49 Kemp’s ridley mortalities. For hawksbill, leatherback, and loggerhead turtles, incidental take was expressed in terms of observed numbers (from observer program data) since insufficient observer data existed to model an estimated annual take level for these species.

Pacific Ocean Basin

This subsection describes the impact of U.S. commercial fisheries bycatch on sea turtle populations within the Pacific Ocean basin, including fisheries operations along the U.S. West Coast and the Pacific Islands region. The primary turtle species of concern for U.S. fisheries bycatch in the Pacific are leatherbacks and loggerheads, due to their critical conservation status (Moore et al. 2009).

The west coast longline fishery operates in the north Pacific ocean, mainly from the U.S. EEZ west to 140 degrees west longitude and from the equator to 35 degrees north (NMFS 2016b). This fishery primarily targets bigeye tuna, although other tuna and non-tuna species are also caught and retained. As of 2016 there was only one boat participating in this fishery, although fishing effort is expected to increase in the future (NMFS 2016b).

Domestic longline fishing around Hawaii consists of two separately managed fisheries: a deep-set fishery that primarily targets bigeye tuna and a shallow-set fishery that targets swordfish. The shallow-set fishery operates almost entirely north of Hawaii. The deep-set fishery operates primarily to the south of Hawaii between the equator and 35 degrees north, although in some years this fishery expands northward and overlaps with the shallow-set fishery.

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

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

In 2012, NMFS issued an opinion on the continued operation of the Hawaii shallow-set longline fishery (NMFS 2012a). Sea turtle incidental take was authorized over a continuous two-year calendar period in the ITS. In 2014, NMFS issued an opinion on the continued operation of the Hawaii deep-set longline fishery (NMFS 2014b). Sea turtle incidental take was authorized over a three-year period in the ITS.

The West coast drift gillnet fishery targets swordfish and thresher sharks in the U.S. EEZ and adjacent high seas off the coasts of California, Oregon, and Washington (NMFS 2013). In 2001, NMFS established Pacific Sea Turtle Conservation Areas that prohibit drift gillnet fishing in large portions of the historical fishing grounds, either seasonally or conditionally, to protect endangered leatherback and loggerhead sea turtle populations (66 FR 44549; August 24, 2001). Oregon and Washington state laws currently prohibit landings caught with drift gillnet gear, although ships still fish drift gillnets in federal waters off these states and land their catch in California. The drift gillnet fishery can also be closed during El Niño events in order to reduce bycatch of loggerhead turtles that move further north on the warm El Niño currents from Mexico into U.S. waters (72 FR 31756, June 8, 2007). In 2013, NMFS issued an opinion on the continued authorization of the West Coast drift gillnet fishery (NMFS 2013).

9.1.4 Fishes

Atlantic and Shortnose Sturgeon

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

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

Several federally regulated fisheries that may encounter Atlantic sturgeon have fishery management plans (FMPs) that have undergone section 7 consultation with NMFS. On December 16, 2013, NMFS issued a “batched” section 7 opinion on the following fisheries: Northeast multispecies; monkfish; spiny dogfish; Atlantic bluefish; Northeast skate complex; mackerel/squid/butterfish; and summer flounder /scup/black sea bass. The majority (73 percent) of all Atlantic sturgeon bycatch mortality in New England and Mid-Atlantic waters is attributed to the monkfish sink gill net fishery (ASMFC 2007). Observer data from 2001 to 2006 shows 224 recorded interactions between the monkfish fishery and Atlantic sturgeon, with 99 interactions resulting in death, a 44 percent mortality rate. For all seven fisheries combined, the following take of Atlantic sturgeon was authorized annually: 1,331 trawl interactions of which 42 may be lethal and 1,229 gill net interactions of which 155 may be lethal. The 2012 NMFS opinion on the Southeast shrimp trawl fishery exempted the take of Atlantic sturgeon as follows: 1,731 total interactions, including 243 captures of which 27 are expected to be lethal every three years. In 2012, NMFS provided an updated opinion on the Federal shark fisheries, including the smoothhound fishery on ESA-listed species. For the federal smoothhound fishery and shark fisheries combined, NMFS exempted the take of 321 Atlantic sturgeon over a three-year span, with 66 of those takes expected to be lethal.

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

Gulf Sturgeon

Direct take of Gulf sturgeon is prohibited in all four states within the current range of the species. However, fisheries directed at other species that employ various trawling and entanglement gear in areas that gulf sturgeon regularly occupy pose a risk of incidental bycatch. Gulf sturgeon are occasionally incidentally captured in state managed shrimp fisheries in bays and sounds along the northern Gulf of Mexico. Gulf sturgeon bycatch has also been documented in entanglement gear (trammel and gill nets) used to target gar in the Pearl River in southeast Louisiana (USFWS and NMFS 2009). While state regulations prohibit the taking or possession of gulf sturgeon (including roe), there is no available data to determine bycatch capture or mortality rates (NMFS 2014c).

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

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

Federal fisheries that NMFS authorizes in the Gulf of Mexico have likely had a minor impact on Gulf sturgeon. This is because Gulf sturgeon occur in the Gulf of Mexico only during winter months and during that time, most migrate alongshore and to barrier island habitats within shallower state waters (NMFS 2014c).

Bocaccio and Yelloweye Rockfish

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

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

Green Sturgeon

Take of Southern DPS green sturgeon in federal fisheries was prohibited as a result of the ESA 4(d) protective regulations issued in June of 2010 (75 FR 30714). Green sturgeon are occasionally encountered as bycatch in Pacific groundfish fisheries (Al-Humaidhi 2011), although the impact of these fisheries on green sturgeon populations is estimated to be small (NMFS 2012b). NMFS (2012b) estimates between 86 and 289 Southern DPS green sturgeon are annually encountered as bycatch in the state-regulated California halibut bottom trawl fishery.

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

Eulachon

The main source of eulachon bycatch are the west coast shrimp fisheries (NMFS 2017e). Offshore trawl fisheries for ocean shrimp (*Pandalus jordani*) occur off the west coast of North America from the west coast of Vancouver Island to Cape Mendocino, California (Hannah and Jones 2007) and in British Columbia, Canada. *Pandalus jordani* is known as the smooth pink shrimp in British Columbia, ocean pink shrimp or smooth pink shrimp in Washington, pink shrimp in Oregon, and Pacific Ocean shrimp in California. The ocean shrimp season is open April 1 through October 31 in California, Oregon and Washington and ships deliver catch to shore-based processors. Total coast-wide ocean shrimp landings have ranged from a low of 1,888 metric tons in 1957 to a high of 46,494 metric tons in 2015 (NMFS 2017e).

Prior to 2000, bycatch in the ocean shrimp fishery ranged from 32 to 61 percent of the total catch (Hannah and Jones 2007). Eulachon occur as bycatch in shrimp trawl fisheries off the coasts of Washington, Oregon, California, and British Columbia (Gustafson et al. 2010). Ward et al. (2015) found that the coastal areas just south of Coos Bay, Oregon; between the Columbia River and Grays Harbor, Washington; and just south of La Push, Washington were consistent hotspots of eulachon bycatch across years. The previously depressed and currently increasing abundance of the Southern DPS of eulachon (James et al. 2014) are likely contributing to the increased levels of eulachon bycatch reported for 2012 to 2014. The dramatic increases in the level of eulachon bycatch in both the Washington and Oregon ocean shrimp trawl fisheries in 2012 and 2013 occurred in spite of regulations requiring the use of bycatch reduction devices. It is unclear why bycatch ratios were highest in the Washington, intermediate in the Oregon, and lowest in the California sectors of the ocean shrimp trawl fishery in 2012 and 2013. However, the bycatch ratio increased in Oregon and decreased in Washington in 2014 compared to the previous two-year period. Use of bycatch reduction devices in offshore shrimp trawl fisheries, which was mandated beginning in 2002 in California and 2003 in Washington and Oregon has substantially reduced bycatch of fin fish in these fisheries (Frinodig et al. 2009; Hannah and Jones 2007).

Chinook Salmon and Chum Salmon

Commercial and recreational fisheries also result in “non-landed mortality” on chinook and other species which varies by the type of gear. Even fisheries designed to be selective either for species or to harvest specially marked hatchery fish will have some mortality associated with the hooking and handling of the released fish. These include fish that are brought to the boat but are released because they are too small (may die from hooking trauma), fish that are hooked but drop off before they are brought to the boat, and fish that die from entanglement in gillnet or purse seine gear and drop out before being landed. For each type of fishery (commercial troll, recreational, net, etc.), harvest managers add between 5 and 50 percent to the total catch to account for fish deaths due to release, drop-off and other harvest related impacts (PSTT and WDFW 2004).

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

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

Steelhead Trout

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

Most Puget Sound streams have insufficient catch and escapement data to calculate exploitation rates for natural steelhead (NMFS 2016f). Populations with sufficient data include those in the Skagit, Green, Nisqually, Puyallup, and Snohomish rivers. Exploitation rates differ widely among the different rivers, but all have declined since the 1970s and 1980s. Exploitation rates on natural steelhead during the earlier period averaged between 10 percent and 40 percent, with some populations in the central and south parts of Puget Sound, such as the Green and Nisqually river populations, experiencing exploitation rates over 60 percent. Exploitation rates on natural steelhead over the past decade have been stable and generally less than 5 percent. Current exploitation rates are low enough that they unlikely to substantially reduce spawner abundance for most steelhead populations in Puget Sound, and these rates are expected to continue for the near future (NMFS NWFSC 2015).

9.2 Ship Strike

Marine habitats occupied by ESA-listed species often feature both heavy commercial and recreational ship traffic. Ship strikes represent a recognized threat to large, air breathing marine species including whales, pinnipeds and sea turtles. This threat is increasing as commercial shipping lanes cross important breeding and feeding habitats and as ESA-listed species populations recover and populate new areas or areas where they were previously extirpated (Swingle et al. 1993; Wiley et al. 1995). As ships continue to become faster and more widespread, an increase in ship interactions with ESA-listed species is expected.

All sizes and types of ships can hit whales, but most lethal and severe injuries are caused by ships eighty meters (262.5 feet) or longer (Laist et al. 2001). For whales, studies show that the probability of fatal injuries from ship strikes increases as ships operate at speeds above twenty-six kilometers per hour (14 knots) (Laist et al. 2001). Evidence suggests that not all whales killed as a result of ship strike are detected, particularly in offshore waters, and some detected carcasses are never recovered while those that are recovered may be in advanced stages of decomposition that preclude a definitive cause of death determination (Glass et al. 2010). Most whales killed by ship strike likely end up sinking rather than washing up on shore. Kraus et al. (2005) estimated that 17 percent of ship strikes of North Atlantic right whales are actually detected. The number of documented cetacean mortalities from ship strike is much lower than the actual number of mortalities, especially for less buoyant species such as blue, humpback, and fin whales (Rockwood et al. 2017). Rockwood et al. (2017) modeled ship strike mortalities of blue, humpback and fin whales off California using carcass recovery rates of five and seventeen percent and conservatively estimated that ship strike mortality may be as high as 7.8, 2.0, and 2.7 times the recommended limit for blue, humpback and fin whales stocks in this area respectively.

Of the eleven species known to be hit by ships, in the northern hemisphere fin whales are stuck most frequently, but right whales, humpback whales, sperm whales, and gray whales are also struck (Laist et al. 2001; Peel et al. 2018; Vanderlaan and Taggart 2007). A review by Jensen and Silber (2004) of the NMFS' ship strike database revealed fin whales as the most frequently confirmed victims of ship strikes (twenty-six percent of the 292 documented ship strikes), with most collisions occurring off the U.S. East coast. In some areas, one-third of all fin whale and right whale strandings appear to involve ship strikes (Laist et al. 2001). The effects of ship strikes are particularly profound on species with low abundance, such as North Atlantic right whales. Ship strikes represent one of the greatest threats to the continued existence of North Atlantic right whales. Between 1999 and 2006, ships were confirmed to have struck 22 North Atlantic right whales, killing thirteen of these whales (Jensen and Silber 2004; Knowlton and Kraus 2001). From 2006 to 2010, ten instances of mortality stemming from ship collision were documented (Waring et al. 2013). However, with the implementation of the 2008 mandatory right whale ship strike reduction rule and increased communication through the usage of the

Automatic Identification System, reported instances of North Atlantic right whale mortalities from ship strikes have significantly decreased (Conn and Silber 2013). As a result of the rule, speed restrictions of ten knots or less for ships 65 feet in length or greater were implemented for several areas along the western Atlantic during specified times of the year (50 C.F.R. §224.105). From 2008 to 2014, only two reported instances of mortalities were recorded for North Atlantic right whales due to ship strike, resulting in a nearly 80 to 90 percent reduction of occurrence from previous time spans (Henry et al. 2016; Henry et al. 2015; Waring et al. 2015). However, the results of necropsies for five of the North Atlantic right whales found dead in the 2017 Unusual Mortality Event indicate evidence of blunt force trauma consistent with ship strikes (Daoust et al. 2017). The number of observed physical injuries to humpback whales as a result of ship collisions has increased in Hawaiian waters (Glockner-Ferrari et al. 1987; Lammers et al. 2007), possibly partly stemming from humpback whale population growth. From 1991 to 2010, eight ship strikes of humpback whales in California waters were documented. Ship strikes were implicated in the deaths of five blue whales in the Pacific Ocean from 2004 to 2008.

In rare instances, killer whales are injured or killed by collisions with passing ships and powerboats, primarily from being struck by the propeller blades. Three killer whale mortalities from a ship collision have been reported for Washington and British Columbia since the 1960s, two of these reported in 2006 (Baird and Baird 2006; Gaydos and Raverty 2007). Five non-lethal collisions between ships and killer whales have also been documented in the region since the 1990s (Baird 2001; NMFS 2008). In most instances, researchers reported only minor visible injuries from which the whales were able to recover from NMFS.

Sea turtles must surface to breathe and several species are known to bask at the surface for long periods making them more susceptible to ship strike. Ship strikes have been identified as one of the important mortality factors in several nearshore turtle habitats worldwide (Denkinger et al. 2013). However, available information is sparse regarding the overall magnitude of this threat or the impact on sea turtle populations globally. Although sea turtles can move somewhat rapidly, they apparently are not adept at avoiding ships that are moving at more than 4 km per hour; most ships move far faster than this in open water (Hazel and Gyuris 2006; Hazel et al. 2007; Work et al. 2010). Hazel et al. (2007) suggests that green turtles may use auditory cues to react to approaching ships rather than visual cues, making them more susceptible to strike as ship speed increases. Since turtles that were previously killed or injured as a result of some other stressor (e.g., fishing net entanglement or disease) may be more susceptible to a ship strike, it is not always known what proportion of ship wounds were sustained ante-mortem versus post mortem (or post injury). In one study from Virginia, Barco et al. (2016) found that all fifteen dead loggerhead turtles encountered with signs of acute ship interaction were apparently normal and healthy prior to being struck by a ship.

High levels of ship traffic in nearshore areas along the U.S. Atlantic and Gulf of Mexico coasts result in frequent sea turtle ship strikes. The incidence of propeller wounds of stranded turtles from the U.S. Atlantic and Gulf of Mexico doubled from about ten percent in the late 1980s to about twenty percent in 2004. Singel et al. (2007) reported a tripling of boat strike injuries in Florida from the 1980's to 2005. Over this time period, in Florida alone over 4,000 (approximately 500 live; approximately 3500 dead) sea turtle strandings were documented with propeller wounds, which represents 30 percent of all sea turtle strandings for the state (Singel et al. 2007). These studies suggest that the threat of ship strikes to sea turtles may be increasing over time as ship traffic continues to increase in the United States.

Ship strikes were identified as a source of mortality for green sea turtles in Hawaii waters, although reported incidence rates among stranded turtles are not as high as in the southeastern United States. Chaloupka et al. (2008b) reported that 2.5 percent of green turtles found dead on Hawaiian beaches between 1982 and 2003 had been killed by boat strike. Ship strikes have also been reported as a potentially important threat to sea turtle populations by researches in other parts of the world including the Canary Islands (Orós et al. 2005), Italy (Casale et al. 2010), and the Galápagos Islands (Denkinger et al. 2013; Parra et al. 2011).

Sturgeon are also susceptible to ship strikes due to their large size and frequent use of coastal waterways with heavy commercial ship traffic. The factors relevant to determining the risk to sturgeon from ship strikes are currently unknown, but are likely related to size and speed of the ships, navigational clearance (i.e., depth of water and draft of the ship) in the area where the ship is operating, and the behavior of sturgeon in the area (e.g., foraging, migrating, etc.). The ASSRT determined Atlantic sturgeon in the Delaware River are at a moderately high risk of extinction because of ship strikes, and sturgeon in the James River are at a moderate risk from ship strikes (ASSRT 2007). Balazik et al. (2012) estimated up to 80 sturgeon were killed between 2007 and 2010 in these two river systems. Brown and Murphy (2010) examined 28 dead Atlantic sturgeon from the Delaware River from 2005 through 2008 and found that fifty percent of the mortalities resulted from apparent ship strikes, and seventy-one percent of these (ten out of fourteen) had injuries consistent with being struck by a large ship. Eight of the fourteen ship-struck sturgeon were adult-sized fish which, given the time of year the fish were observed, were likely migrating through the river to or from the spawning grounds. Ship strikes may also be threatening Atlantic sturgeon populations in the Hudson River where large ships move from the river mouth to ports upstream through narrow shipping channels. The channels are dredged to the approximate depth of the ships, usually leaving less than six feet of clearance between the bottom of ships and the river bottom. Any aquatic life along the bottom is at risk of being sucked up through the large propellers of these ships.

Large Atlantic sturgeon are most often killed by ship strikes because their size means they are unable to pass through the ship's propellers without making contact. Shortnose sturgeon may not be as susceptible due to their smaller size in comparison to Atlantic sturgeon. There has been only one confirmed incidence of a ship strike on a shortnose sturgeon in the Kennebec River, and two suspected ship strike mortalities in the Delaware River (SSSRT 2010). Green sturgeon and smalltooth sawfish may also been susceptible to ship strikes but there is no available information on this threat to these species.

9.3 Coastal Development and Land Use Changes

The modification and destruction of habitat remains one of the primary threats to many threatened and endangered species. In this section, we summarize the impacts of general anthropogenic stressors associated with coastal development and other land use changes on the aquatic habitats used by ESA-listed species. The effects of human activities on aquatic habitats are discussed in more detail in subsequent sections addressing the following specific threats: dredging, oil pollution, contaminants, nutrient loading, marine debris, and sound.

Many stream, riparian, and coastal areas within the action area have been degraded by the effects of land and water use associated with urbanization, road construction, forest management, agriculture, mining, transportation, water development, and other human activities. Development activities contribute to a variety of interrelated factors that lead to the decline of ESA-listed anadromous fish species considered in this opinion. These include reduced in-channel and off-channel habitat, restricted lateral channel movement, increased flow velocities, increased erosion, decreased cover, reduced prey sources, increased contaminants, increased water temperatures, degraded water quality, and decreased water quantity.

Urbanization and increased human population density within a watershed result in changes in stream habitat, water chemistry, and the biota (plants and animals) that live there. In many cases, these changes negatively impact species, particularly those with small population sizes like some of the ESA-listed species within the action area. The most obvious effect of urbanization is the loss of natural vegetation, which results in an increase in impervious cover and dramatic changes to the natural hydrology of urban and suburban streams (O'Driscoll et al. 2010). Urbanization generally results in land clearing, soil compaction, modification and/or loss of riparian buffers, and modifications to natural drainage features. The increased impervious cover in urban areas leads to increased volumes of runoff, increased peak flows and flow duration, and greater stream velocity during storm events (Bledsoe and Watson 2001; Booth et al. 1995). Runoff from urban areas also contains chemical pollutants from vehicles and roads, industrial sources, and residential sources (Connor et al. 2003). Urban runoff is typically warmer than receiving waters and can significantly increase temperatures, particularly in smaller streams (O'Driscoll et al. 2010).

Municipal wastewater treatment plants replace septic systems, resulting in point discharges of nutrients and other contaminants not removed in the processing (Booth et al. 1995). Municipalities with combined sewer/stormwater overflows or older treatment systems may directly discharge untreated sewage following heavy rainstorms. Urban and suburban nonpoint and point source discharges affect water quality and quantity in basin surface waters (O'Driscoll et al. 2010). Dikes and levees constructed to protect infrastructure and agriculture have isolated floodplains from their river channels and restricted fish access (Bayley 1995). The many miles of roads and rail lines that parallel streams within the action area have degraded stream bank conditions and decreased floodplain connectivity by adding fill to floodplains (O'Driscoll et al. 2010). Culvert and bridge stream crossings have similar effects and create additional problems for fish when they act as physical or hydraulic barriers that prevent fish access to spawning or rearing habitat, or contribute to adverse stream morphological changes upstream and downstream of the crossing itself.

9.4 Dredging

Riverine, nearshore, and offshore coastal areas are often dredged to support commercial shipping, recreational boating, construction of infrastructure, and marine mining. Dredging in fish spawning and nursery grounds modifies habitat quality, and limits the extent of available habitat in some rivers where anadromous fish habitat has already been impacted by the presence of dams. Negative indirect effects of dredging include changes in DO and salinity gradients in and around dredged channels (Campbell and Goodman 2004; Jenkins et al. 1993; Secor and Niklitschek 2001). Dredging operations may also pose risks to anadromous fish species by destroying or adversely modifying benthic feeding areas, disrupting spawning migrations, and filling spawning habitat with resuspended fine sediments. As benthic omnivores, sturgeon in particular may be sensitive to modifications of the benthos which affect the quality, quantity and availability of prey species. Dredging and filling impact important habitat features of Atlantic sturgeon as they disturb benthic fauna, eliminate deep holes, and alter rock substrates (Smith and Clugston 1997). Hatin et al. (2007) reported avoidance behavior by Atlantic sturgeon during dredging operations, and McQuinn and Nellis (2007) found that Atlantic sturgeon were substrate dependent and avoided dredge spoil dumping grounds. The eulachon biological review team identified dredging as a low to moderate threat to the species in the Fraser and Columbia rivers, and a low threat for eulachon in mainland British Columbia rivers due to less dredging activity here (FR 75 13012). They noted that dredging during eulachon spawning was particularly detrimental, as eggs associated with benthic substrates are likely to be destroyed. In Canada, dredging is not allowed in the Fraser River during early March to June to protect spawning eulachon.

In addition to the indirect impacts described above, hydraulic dredging operations can directly harm large marine animals (e.g., sturgeon and sea turtles) by lethally entraining them through the dredge drag-arms and impeller pumps. Large animals that are entrained in hydraulic dredges rarely survive the encounter. Hopper dredges, in particular, are capable of moving relatively quickly compared to turtles and fish which can be overtaken and entrained by the suction draghead of the advancing dredge. Between 1990 and 2005, ten Atlantic sturgeon were reported killed by hopper dredges (ASSRT 2007). Atlantic sturgeon have been taken in both hydraulic pipeline and bucket-and-barge operations in the Cape Fear River, North Carolina (Moser and Ross 1995). Dickerson (2006) reported 15 Atlantic sturgeon taken in dredging activities conducted by the U.S. Army Corps of Engineers (USACE) from 1990 to 2010, most captured by hopper dredge. Notably, these reports include only those trips when an observer was on board to document capture and the actual number directly killed by dredging operations may be much greater.

An estimated 609 incidental takes (lethal or sublethal interactions) of sea turtles were documented from hopper dredging activity in the southeastern U.S. from 1980 through 2006 (Dickerson et al. 2007). Reductions in dredge entrainment rates for sea turtles have been achieved through mitigation measures including gear modifications, operational changes, time-area restrictions, and the capture and relocation of turtles away from dredge sites (Dickerson et al. 2007). Dickerson et al. (2007) studied the effectiveness of turtle relocation trawling in reducing the incidental take of sea turtles in hopper dredge operations. They found that relocation trawling can be an effective management option provided that a substantial amount of trawling effort is conducted either at the onset of dredging or early in the project.

Dredging operations also emit sounds at levels that could potentially disturb individuals of many marine taxa. Depending on the type of dredge, peak sound pressure levels from 100 to 140 dB re 1 micro Pascal (μPa) were reported in one study (Clarke et al. 2003). As with pile driving, most of the sound energy associated with dredging is in the low-frequency range, less than 1000 Hz (Clarke et al. 2003).

9.5 Pollution

Many different types of pollution can adversely affect ESA-listed species and habitats within the action area. In this section, we focus on four major categories of marine and estuarine pollution: oil pollution, contaminants and pesticides, nutrient loading and algal blooms, and marine debris. Considering the large area covered by the proposed action, we do not attempt to provide a detailed analysis of the effects of pollution throughout the entire action area. Instead, this section provides a more general discussion of the four pollution categories above, including the stressor pathways and anticipated effects on ESA-listed resources, with an emphasis on geographic areas, habitats or species that are particularly susceptible to these threats.

9.5.1 Oil Pollution

Oil released into the marine environment contains aromatic organic chemicals known to be toxic to a variety of marine life (Yender et al. 2002). Oil spills can impact wildlife directly through three primary pathways: (1) ingestion—when animals swallow oil particles directly or consume prey items that have been exposed to oil, (2) absorption—when animals come into direct contact with oil, and (3) inhalation—when animals breath volatile organics released from oil or from "dispersants" applied by response teams in an effort to increase the rate of degradation of the oil in seawater. Direct exposure to oil can cause acute damage including skin, eye, and respiratory irritation, reduced respiration, burns to mucous membranes such as the mouth and eyes, diarrhea, gastrointestinal ulcers and bleeding, poor digestion, anemia, reduced immune response, damage to kidneys or liver, cessation of salt gland function, reproductive failure, and death (NOAA 2003; Vargo et al. 1986). Nearshore spills or large offshore spills that reach shore can oil beaches on which sea turtles lay their eggs, causing birth defects or mortality in the nests (NOAA 2003). Disruption of other essential behaviors, such as breeding, communication, and feeding may also occur. The loss of invertebrate communities due to oiling or oil toxicity would also decrease prey availability for hawksbill, Kemp's ridley, and loggerhead sea turtles (NOAA 2003). Sea turtles species which commonly forage on crustaceans and mollusks may be vulnerable to oil ingestion due to oil adhering to the shells of these prey and the tendency for these organisms to bioaccumulate toxins found in oil (NOAA 2003). Seagrass beds may be particularly susceptible to oiling as oil contacts grass blades and sticks to them, hampering photosynthesis and gas exchange (Wolfe et al. 1988). If spill cleanup is attempted, mechanical damage to seagrass can result in further injury and long-term scarring. Loss of seagrass due to oiling would be important to green sea turtles, as this is a significant component of their diets (NOAA 2003). Sea turtles are known to ingest and attempt to ingest tar balls, which can block their digestive systems, impairing foraging or digestion and potentially causing death (NOAA 2003).

The Gulf of Mexico is an area of high-density offshore oil extraction with chronic, low-level spills and occasional massive spills (such as the *Deepwater Horizon* oil spill, *IXTOC I* oil well blowout and fire in the Bay of Campeche in 1979, and the explosion and destruction of a loaded supertanker, the Mega Borg, near Galveston in 1990). Oil spills remain a significant threat to marine ecosystems in the Gulf of Mexico due to the large amount of extraction and refining activity in the region. There are approximately 4,000 oil and gas structures in the northern Gulf of Mexico, 90 percent of which are off Louisiana and Texas (USN 2009).

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

In addition to oil spills, routine oil discharges into the northern Gulf of Mexico (not including oil spills) account for roughly 88,200 barrels of petroleum per year from municipal and industrial wastewater treatment plants, and roughly 19,250 barrels from produced water discharges during oil and gas operations (MMS 2007; USN 2008). Another major source of oil found in the northern Gulf of Mexico is natural seepage. Estimates of natural seepage are highly imprecise, ranging from 120,000 to 980,000 barrels of oil annually (MacDonald et al. 1993; MMS 2007).

9.5.2 Contaminants and Pesticides

Coastal habitats are often in close proximity to major sources of pollutants and contaminants, which make their way into the marine environment from industrial, domestic and agricultural sources. Sources include wastewater treatment plants, septic systems, industrial facilities, agriculture, animal feeding operations, and improper refuse disposal. Agricultural discharges, as well as discharges from large urban centers, contribute contaminants as well as coliform bacteria to coastal watersheds. Contaminants can be carried long distances from terrestrial or nearshore sources and ultimately accumulate in offshore pelagic environments (USCOP 2004). The Gulf of Mexico, in particular, is a major sink for pollution from a variety of marine and terrestrial sources, which ultimately can interfere with ecosystem health and particularly that of ESA-listed species and their habitats. The Mississippi River drains 80 percent of the U.S. cropland (including the fertilizers, pesticides, herbicides, and other contaminants that are applied to it) and discharges into the Gulf of Mexico (MMS 1998).

Chemical contaminants, particularly those that are persistent in the environment, are a particular concern for marine animals that often occupy high trophic positions. Persistent organic pollutants (POPs), which include legacy pesticides (e.g., DDT, chlordane), legacy industrial-use chemicals (e.g., polychlorinated biphenyls), and emerging contaminants of concern (e.g., polybrominated diphenyl ethers, perfluorinated compounds) accumulate in fatty tissues of marine organisms and are magnified through the food chain, leading upper trophic predators to be highly exposed (National Academies of Sciences and Medicine 2016). High concentrations of PCBs and DDT have been reported in tissues of marine mammals in most parts of the world, particularly in coastal regions adjacent to heavy coastal development and/or industry. These legacy POPs have been linked to a number of adverse health effects including endocrine disruption, reproductive impairment or developmental effects, and immune dysfunction or disease susceptibility (National Academies of Sciences and Medicine 2016). Polybrominated diphenyl ethers commonly used as flame retardants, are another class of POPs that have spread globally in the environment and have also been reported in a broad array of marine mammal species (National Academies of Sciences and Medicine 2016).

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

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

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

The life histories of sturgeon species (i.e., long lifespan, extended residence in estuarine habitats, benthic foraging) predispose them to long-term, repeated exposure to environmental contamination and potential bioaccumulation of heavy metals and other toxicants (Dadswell 1979). Heavy metals and organochlorine compounds accumulate in sturgeon tissue, but their long-term effects are not well studied (Ruelle and Keenlyne 1993). Shortnose sturgeon collected from the Delaware and Kennebec Rivers had total toxicity equivalent concentrations of polychlorinated dibenzo-p-dioxins, polychlorinated dibenzofurans, PCBs, dichlorodiphenyl dichloroethene, aluminum, cadmium, and copper all above adverse effect concentration levels reported in the literature (Brundage III 2008). Dioxin and furans were detected in ovarian tissue from shortnose sturgeon caught in the Sampit River/Winyah Bay system (South Carolina).

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

Dwyer et al. (2005) compared the relative sensitivities of common surrogate species used in contaminant studies to 17 ESA-listed species including Atlantic sturgeon. The study examined 96-hour acute water exposures using early life stages where mortality is an endpoint. Chemicals tested were carbaryl, copper, 4-nonphenol, pentachlorophenal and permethrin. Of the ESA-listed species, Atlantic sturgeon were ranked the most sensitive species tested for four of the five chemicals (Atlantic and shortnose sturgeon were found to be equally sensitive to permethrin). Additionally, a study examining the effects of coal tar, a byproduct of the process of destructive distillation of bituminous coal, indicated that components of coal tar are toxic to shortnose sturgeon embryos and larvae in whole sediment flow-through and coal tar elutriate static renewal (Kocan et al. 1993).

Despite the vast evidence to suggest that marine animals are exposed to anthropogenic, as well as natural, chemicals capable of producing significant toxic effects, only a few studies have actually examined the impacts on population survival or reproductive rates. Such observational assessments are inherently challenging due to the difficulty in controlling for confounding or interacting variables, as well as the sublethal but chronic nature of chemical contaminant effects, and the difficulty of observing mortality or reproductive endpoints, particularly in long-lived species such as cetaceans and sea turtles (National Academies of Sciences and Medicine 2016). The Environmental Protection Agency (EPA) has consulted with NMFS on re-registration of over 30 active ingredients found in commercial herbicides, insecticides and fungicides. Since 2009, NMFS has issued seven batched opinions on the effects of these active ingredients on ESA-listed Pacific salmon and steelhead (see https://www.fisheries.noaa.gov/national/consultations/pesticide-consultations for details). Based on the opinions competed to date, NMFS has issued several jeopardy and/or adverse modification determinations for Pacific salmon and steelhead populations on the EPA's proposed registration of the active ingredients in many pesticides and herbicides.

9.5.3 Nutrient Loading and Algal Blooms

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

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

9.5.4 Marine Debris

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

Marine debris is a significant concern for ESA-listed species, particularly sea turtles and marine mammals. The initial developmental stages of all turtle species are spent in the open sea. During this time both juvenile turtles and their buoyant food are drawn by advection into fronts (convergences, rips, and drift lines). The same process accumulates large volumes of marine debris, such as plastics and lost fishing gear, in ocean gyres (Carr 1987). An estimated four to twelve million metric tons of plastic enter the oceans annually (Jambeck et al. 2015). It is thought that some sea turtles eat plastic because it closely resembles jellyfish, a common natural prey item (Schuyler 2014). Ingestion of plastic debris can block the digestive tract which can cause turtle mortality as well as sub-lethal effects including dietary dilution, reduced fitness, and absorption of toxic compounds (Laist et al. 1999; Lutcavage et al. 1997). Santos et al. (2015) found that a surprisingly small amount of plastic debris was sufficient to block the digestive tract and cause death. They reported that 10.7 percent of green turtles in Brazilian waters were killed by plastic ingestion, while 39.4 percent had ingested enough plastic to have killed them. These results suggest that debris ingestion is a potentially important source of turtle mortality, one that may be masked by other causes of death. Gulko and Eckert (2003) estimated that between one-third and one-half of all sea turtles ingest plastic at some point in their lives. A more recent study by Schuyler et al. (2016) estimates that 52 percent of sea turtles globally have ingested plastic debris. Schuyler et al. (2016) synthesized the factors influencing debris ingestion by turtles into a global risk model, taking into account the area where turtles are likely to live, their life history stage, the distribution of debris, the time scale, and the distance from stranding location. They found that oceanic life stage turtles are at the highest risk of debris ingestion. Based on this model, olive ridley turtles are the most at-risk species; green, loggerhead, and leatherback turtles were also found to be at a high and increasing risk from plastic ingestion (Schuyler 2014). The regions of highest risk to global turtle populations are off the east coasts of the United States, Australia, and South Africa; the east Indian Ocean, and southeast Asia. In addition to ingestion risks, sea turtles can also become entangled in marine debris such as fishing nets, monofilament line, and fish-aggregating devices or FADs (Laist et al. 1999; Lutcavage et al. 1997; NRC 1990). Turtles are particularly vulnerable to ghost nets due to their tendency to use floating objects for shelter and as foraging stations (Dagorn et al. 2013; Kiessling 2003).

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

9.6 Anthropogenic Sound

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

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

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

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

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

This section is divided into subsections addressing the impacts of anthropogenic sound from the following major sources: ships and commercial shipping; seismic surveys; military activities; active sonar; explosions; and pile driving and construction.

9.6.1 Ship Sound and Commercial Shipping

Much of the increase in sound in the ocean environment over the past several decades is due to increased shipping, as ships become more numerous and of larger tonnage (Hildebrand 2009; Mckenna et al. 2012; NRC 2003). Shipping constitutes a major source of low-frequency sound in the ocean (Hildebrand 2004), particularly in the Northern Hemisphere where the majority of ship traffic occurs. While commercial shipping ships contribute a large portion of oceanic anthropogenic noise, other sources of maritime traffic can be present in large numbers and impact the marine environment. These include recreational boats, whale-watching boats, research ships, and ships associated with oil and gas activities. Individual ships produce unique acoustic signatures, although these signatures may change with ship speed, ship load, and activities that may be taking place on the ship. Sound levels are typically higher for the larger and faster ships. Peak spectral levels for individual commercial ships are in the frequency band of ten to 50 Hz and range from 195 dB re: μPa^2-s at 1 m for fast-moving (greater than 20 knots) supertankers to 140 dB re: μPa^2-s at 1 m for smaller ships (NRC 2003). Although large ships emit predominantly low frequency sound, studies report broadband sound from large cargo ships above two kHz, which may interfere with important biological functions of cetaceans (Holt 2008). At frequencies below 300 Hz, ambient sound levels are elevated by 15 to 20 dB when exposed to sounds from ships at a distance (McKenna et al. 2013).

9.6.2 Seismic Surveys

Offshore seismic surveys involve the use of high energy sound sources operated in the water column to probe below the seafloor for oil and gas exploration. The Bureau of Ocean Energy Management has issued permits for over 12,000 seismic surveys (2D and 3D) in the Gulf of Mexico since the 1953 passage of the Outer Continental Shelf Lands Act and has consulted with NMFS under the ESA and MMPA on seismic survey permits for oil and gas exploration off the Atlantic coast. Seismic surveys are also used for scientific research, to identify possible seafloor or shallow-depth geologic hazards, and to locate potential archaeological resources and benthic habitats that should be avoided.

Two major categories of seismic surveys conducted within the action area are: (1) deep seismic surveys which include ocean bottom, vertical seismic profile or borehole, 2D, 3D, 4D and wide azimuth surveys, and (2) high resolution surveys. Deep seismic survey acoustic sources consist of airgun arrays while receiver arrays consist of hydrophones or geophones encased in plastic tubing called streamers. When an airgun array fires, an acoustic energy pulse is emitted and reflected or refracted back from the seafloor. These reflected/refracted acoustic signals create pressure fluctuations, which are detected and recorded by the streamers. Seismic airguns generate intense low-frequency sound pressure waves capable of penetrating the seafloor and are fired repetitively at intervals of 10 to 20 seconds for extended periods (NRC 2003). Most of the energy from airguns is directed vertically downward, but significant sound emission also extends horizontally. Peak sound pressure levels from airguns usually reach 235 to 240 decibels at dominant frequencies of five to 300 Hz (NRC 2003). High-resolution surveys collect data on surface and near-surface geology used to identify archaeological sites, potential shallow geologic and manmade hazards for engineering, and site planning for bottom-founded structures. High-resolution surveys may use airguns but also use other sound sources such as sub-bottom profilers (at 2.5 to 7 kHz), echosounders (single-beam at 12 to 240 kHz; multibeam at 50 to 400 kHz), boomers (at 300 to 3,000 Hz), sparkers (at 50 to 4,000 Hz), compressed high intensity radar pulse sub-bottom profiler (at 4 to 24 kHz), pingers (at 2 kHz), and side-scan sonars (16 to 1,500 kHz). These sound sources are typically powered either mechanically or electromagnetically.

A study of ambient sound in the North Atlantic showed that airgun activity contributes significantly to ocean sound levels and can appear to be more continuous than other impulsive sounds because of the reverberation from the surface and seabed (Nieukirk et al. 2004). Exposure of cetaceans to very strong impulsive sound sources from airgun arrays can result in auditory damage, such as changes to sensory hairs in the inner ear, which may temporarily or permanently impair hearing by decreasing the range of sound an animal can detect within its normal hearing ranges (reviewed in Finneran 2015). A temporary threshold shift results in a temporary change to hearing sensitivity, and the impairment can last minutes to days, but full recovery of hearing sensitivity is expected. At higher received levels, particularly in frequency ranges where animals are more sensitive, permanent threshold shift can occur, meaning lost auditory sensitivity is unrecoverable. Either of these conditions can result from exposure to a single pulse or from the accumulation of multiple pulses, in which case each pulse need not be as loud as a single pulse to have the same accumulated effect. Since there is frequency overlap between airgun array sounds and vocalizations of ESA-listed cetaceans, particularly baleen whales and to some extent sperm whales, seismic surveys could mask these calls at some of the lower frequencies for these species.

ESA-listed cetaceans are expected to exhibit a wide range of behavioral responses as a consequence of being exposed to seismic airgun sound fields and echosounders. Baleen whales are expected to mostly exhibit avoidance behavior, and may also alter their vocalizations. Sperm whales are expected to exhibit less overt behavioral changes, but may alter foraging behavior, including vocalizations. These responses are expected to be temporary with behavior returning to a baseline state shortly after the seismic source becomes inactive or leaves the area. Individual whales exposed to sound fields generated by seismic airguns could also exhibit responses not readily observable, such as stress (Romano et al. 2002), that may have adverse effects. Other possible responses to impulsive sound sources like seismic airguns include neurological effects, bubble formation, resonance effects, and other types of organ or tissue damage (Cox et al. 2006; Southall et al. 2007; Tal et al. 2015; Zimmer and Tyack 2007), but similar to stress, these effects are not readily observable.

As with cetaceans, ESA-listed sea turtles may exhibit a variety of different responses to sound fields associated with seismic airguns and echosounders. Avoidance behavior and physiological responses from airgun exposure may affect the natural behaviors of sea turtles (McCauley et al. 2000). McCauley et al. (2000) conducted trials with caged sea turtles and an approaching-departing single airgun to gauge behavioral responses of green and loggerhead sea turtles. Their findings showed behavioral responses to an approaching airgun array at 166 dB re: one micro Pascal rms and avoidance around 175 dB re: 1 μPa rms. From measurements of a seismic ship operating 3D air gun arrays in 100 to 120 meters water depth this corresponds to behavioral changes at around two kilometers and avoidance around one kilometer.

9.6.3 Active Sonar

A wide range of sonar systems are in use for both civilian and military applications. Active sonar can emit high-intensity acoustic energy and can receive reflected and/or scattered energy. The primary sonar characteristics that vary with application are the frequency band, signal type (pulsed or continuous), rate of repetition, and source level. Sonar systems can be divided into three categories, depending on their primary frequency of operation; low frequency for one kHz and less, mid frequency for one to ten kHz, and high frequency for ten kHz and greater (Hildebrand 2004). Low frequency active (LFA) sonar systems are designed for long-range detection (Popper et al. 2014).. Signal transmissions are emitted in patterned sequences that may last for days or weeks. An example of a LFA sonar system is the Navy Surveillance Towed Array Sensor System (SURTASS), discussed in more detail below. Mid-frequency military sonars include tactical Anti-Submarine Warfare sonars that are designed to detect submarines, depth sounders, and communication sonars. High-frequency military sonars includes those incorporated into weapons (torpedoes) or weapon countermeasures (mine countermeasures or anti-torpedo devices), as well as sidescan sonar for seafloor mapping. Commercial sonars are designed for fish finding, depth sounding, and sub-bottom profiling. They typically generate sound at frequencies of 3 to 200 kHz, with source levels ranging from 150 to 235 dB re 1μPa at 1 meter (Hildebrand 2004). Depth sounders and sub-bottom profilers are operated primarily in nearshore and shallow environments, however, fish finders are operated in both deep and shallow areas.

9.6.4 Military Training and Testing Activities

The Navy conducts training, testing, and other military readiness activities on range complexes, throughout coastal and offshore areas in the U.S. and on the high seas. During training, existing and established weapon systems and tactics are used in realistic situations to simulate and prepare for combat. Activities include routine gunnery, missile, surface fire support, amphibious assault and landing, bombing, sinking, torpedo, tracking, and mine exercises. Testing activities are conducted for different purposes and include at-sea research, development, evaluation, and experimentation. The Navy performs testing activities to ensure that its military forces have the latest technologies and techniques available to them. The majority of the training and testing activities the Navy conducts in the action area are similar, if not identical, to activities that have been occurring in the same locations for decades.

Navy activities produce sound and visual disturbances to marine mammals and sea turtles. Potential impacts from harassment due to Navy activities include changes from foraging, resting, milling, and other behavioral states that require lower energy expenditures to traveling, avoidance, and behavioral states that require higher energy expenditures. Sound produced during Navy training and testing activities is also expected to result in instances of a temporary threshold shift and permanent threshold shift in marine mammals and sea turtles. The Navy training and testing activities constitute a federal action and take of ESA-listed marine mammals and sea turtles resulting from these Navy activities have previously undergone section 7 consultations. Through these consultations with NMFS, the Navy has implemented monitoring and mitigation measures to reduce the potential effects of underwater sound from military training and testing activities on ESA-listed resources in the action area. Mitigation measures include employing visual observers and implementing mitigation zones when training and testing using active sonar or explosives.

The Navy's SURTASS sonar system has a vertical line array of 18 elements operating between 100 and 500 Hz. Signals projected include combinations of swept frequency and tones pulses, totaling up to 100 seconds in length with individual signals of the order of 10 seconds. SURTASS LFA has a coherent low frequency signal with a duty cycle of less than 20 percent, operating for a maximum of only 255 hours per year for each system (or 432 hours per year in the past) or a total of 10.6 days per year. Prior to 2017, the Navy had only used SURTASS LFA sonar in the western and central North Pacific Ocean. However, in 2017 the Navy requested programmatic section 7 consultation for the operation of SURTASS LFA sonar from August 2017 through August 2022 in the non-polar region of the world's oceans (including within the action area). The consultation was concluded in August 2017 (NMFS 2017d) and considered the Navy's SURTASS LFA program as well as specific SURTASS LFA operations.

The Air Force conducts training and testing activities on range complexes on land and in U.S. waters. Aircraft operations and air-to-surface activities may occur in the action area (e.g., off Florida). Air Force activities generally involve the firing or dropping of munitions (e.g., bombs, missiles, rockets, and gunnery rounds) from aircraft towards targets located on the surface, though Air Force training exercises may also involve boats. These activities have the potential to impact ESA-listed species by physical disturbance, boat strikes, debris, ingestion, and effects from sound and pressure produced by detonations. Air Force training and testing activities constitute a federal action and take of ESA-listed sea turtles considered for these Air Force activities have previously undergone separate section 7 consultations.

9.6.5 Underwater Explosions

An underwater explosion is composed of an initial shock wave, followed by a succession of oscillating bubble pulses. A shock wave is a compression wave that expands radially out from the detonation point of an explosion. The direct shock wave results in the peak shock pressure (compression) and the reflected wave at the air-water surface produces negative pressure (expansion). Explosions are described by metrics such as amplitude, energy and time-space characteristics of the pressure wave (Popper et al. 2014). In the case of detonations, the pressure wave is very pronounced from the sudden release of high energy, and the resulting shock wave can result in injury or death of animals closest to the site. The amount of explosives used is the primary factor affecting how large an area is impacted, but extent of impacts is also affected by the depth of the charge, type of explosive, and whether bulk or shape charges are used. The U.S. Bureau of Ocean Energy Management requires that oil and gas structures must be removed from the sea floor within one year of lease termination. Many of these structures are removed by explosively severing the underwater supportive elements, which produces a shock wave that kills, injures, or disrupts marine life in the blast radius (Gitschlag et al. 1997).

Impacts of underwater explosives on sea turtles and whales could include death, injuries to internal organs, auditory damage, physical discomfort, and behavior disruptions. There is considerable variability in the effects of explosive blasts on fish species; research suggests that there is far more damage to fish species with swim bladders than to species lacking these air chambers (Hastings and Popper 2005). Lethal injuries result from massive trauma or combined trauma to internal organs as a result of close proximity to the point of detonation. Types of lethal injuries include massive lung hemorrhage, gastrointestinal tract injuries (contusions, ulcerations, and ruptures), and concussive brain damage, cranial and skeletal (shell) fractures, hemorrhage, or massive inner ear trauma (Ketten 1995). Examples of nonlethal injuries include eardrum rupture, bruising, and immobilization of severely stunned animals. Stunned animals beneath the water may drown or become vulnerable to other impacts while they are immobilized. Delayed complications arising from nonlethal injuries may ultimately result in the death of the animal because of increased risks from secondary infection, predation, or disease.

9.6.6 Pile Driving and Construction Sound

Industrial activities and construction both in the ocean and along the shoreline can contribute to underwater noise. Pile driving is commonly used for the construction of foundations for a large number of structures including bridges, buildings, retaining walls, harbor facilities, offshore wind turbines, and offshore structures for the oil and gas industry. Pile driving during construction activities is of special concern because it generates noise with a very high source level. During pile installation, noise is produced when the energy from construction equipment is transferred to the pile and released as pressure waves into the surrounding water and sediments. The impulsive sounds generated by impact pile driving are characterized by a relatively rapid rise time to a maximal pressure value followed by a decay period that may include a period of diminishing, oscillating maximal and minimal pressures (Illingworth and Rodkin 2001; Illingworth and Rodkin 2007; Reyff 2012). The amount of noise produced by pile driving depends on a variety of factors, including the type and size of the impact hammer, size of the pile, the properties of the sea floor, and the depth of the water. The predominant energy in pile impact impulses is at frequencies below approximately 2000 Hz with most occurring below 1000 Hz (Laughlin 2006; Reyff 2008; Reyff 2012). Pressure levels from 190 to 220 dB re 1 μPa were reported for piles of different sizes in a number of studies (NMFS 2006a). The majority of the sound energy associated with pile driving is in the low frequency range (less than 1,000 Hz) (Illingworth and Rodkin Inc. 2001; Illingworth and Rodkin Inc. 2004; Reyff 2003). Impact pile driving occurs over small spatial and temporal scales and produces high-intensity, low-frequency, impulsive sounds with high peak pressures that can be detected by mammals, sea turtles and other marine species (Dow Piniak et al. 2012). The expected type of injury to sea turtles and marine mammals is caused by pressure wave damage to hair cells, ear canals, or ear drums as these structures compress and expand with passage of the wave. Vibratory pile driving produces a continuous sound with peak pressures lower than those observed in impulses generated by impact pile driving (Popper et al. 2014).

9.7 Invasive species

Introduction of non-native species is considered one of primary threats to ESA-listed species (Anttila et al. 1998; Pimentel et al. 2004; Wilcove and Chen 1998). Clavero and Garcia-Bertro (2005) found that invasive species were a contributing cause to over half of the extinct species in the International Union for Conservation of Nature (IUCN) database for which an extinction cause could be determined (for 75 percent of extinct species a specific cause could not be determined); invasive species were the only cited cause in 20 percent of those cases. Invasive species consistently rank as one of the top threats to the world's oceans (Pughiuc 2010; Raaymakers 2003; Raaymakers and Hilliard 2002; Terdalkar et al. 2005; Wambiji et al. 2007). A variety of vectors are thought to be responsible for introducing aquatic non-native species including, but not limited to, aquarium and pet trades, recreation, ballast water discharges from ocean-going ships, and hull fouling. Common impacts of invasive species are alteration of habitat and nutrient availability, as well as altering species composition and diversity within an ecosystem (Strayer 2010). The Nonindigenous Aquatic Nuisance Prevention and Control Act of 1990, Title I of P.L. 101-646 (104 Stat. 4761, 16 U.S.C. 4701, enacted November 29, 1990) established a broad Federal program to prevent introduction of and control the spread of introduced aquatic species and the brown tree snake. The USFWS, U.S. Coast Guard, EPA, USACE, and NOAA were assigned new responsibilities including membership on an Aquatic Nuisance Species Task Force. The Task Force's Strategic Plan (2013-2017 https://www.anstaskforce.gov/) addresses coordination, prevention, early detection/rapid response, control/management, restoration, education/outreach, research, and funding.

A more thorough discussion of nonindigenous species in the action area, and how nonindigenous species have or may affect ESA-listed resources is included in the *Effects of the Action* of this opinion (Section 10.1.3).

9.8 Climate change

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

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

Global annually averaged surface air temperature has increased by about 1.8 degrees Fahrenheit (1.0 degrees Celsius) over the last 115 years (1901 to 2016) (Wuebbles et al. 2017). The globally-averaged combined land and ocean surface temperature data, as calculated by a linear trend, show a warming of approximately 1.0°C from 1901 through 2016 (Hayhoe 2018). This period is now the warmest in the history of modern civilization. These global trends are expected to continue over climate timescales. The magnitude of climate change beyond the next few decades will depend primarily on the amount of greenhouse gases (especially carbon dioxide) emitted globally. A set of four scenarios was developed by the Intergovernmental Panel on Climate Change (IPCC) to ensure that starting conditions, historical data, and projections are employed consistently across the various branches of climate science. The scenarios are referred to as representative concentration pathways (RCPs), which capture a range of potential greenhouse gas emissions pathways and associated atmospheric concentration levels through 2100 (IPCC 2014). The RCP scenarios drive climate model projections for temperature, precipitation, sea level, and other variables: RCP2.6 is a stringent mitigation scenario; RCP4.5 and RCP6.0 are intermediate scenarios; and RCP8.5 is a scenario with no mitigation or reduction in the use of fossil fuels. IPCC future global climate predictions and national and regional climate predictions included in the Fourth National Climate Assessment for U.S. states and territories (USGCRP 2018) use the RCP scenarios. The increase of global mean surface temperature change by 2100 is projected to be 0.3 to 1.7°C under RCP2.6, 1.1 to 2.6°C under RCP4.5, 1.4 to 3.1°C under RCP6.0, and 2.6 to 4.8°C under RCP8.5 with the Arctic region warming more rapidly than the global mean under all scenarios (IPCC 2014).

Changes in surface, atmospheric, and oceanic temperatures and other climatic changes have resulted in melting glaciers, diminishing snow cover, shrinking sea ice, rising sea levels, ocean acidification, and increasing atmospheric water vapor. Global average sea level has risen by about seven to eight inches since 1900, with almost half (about three inches) of that rise occurring since 1993. Human-caused climate change has made a substantial contribution to this rise since 1900, contributing to a rate of rise that is greater than during any preceding century in at least 2,800 years (Wuebbles et al. 2017). Global sea level rise has already affected the U.S.; the incidence of daily tidal flooding is accelerating in more than 25 Atlantic and Gulf Coast cities. Global average sea levels are expected to continue to rise by at least several inches in the next 15 years and by one to four feet by 2100. Sea level rise will be higher than the global average on the East and Gulf Coasts of the U.S. (Wuebbles et al. 2017). Climate change has been linked to changing ocean currents as well. Rising carbon dioxide levels have been identified as a reason for a poleward shift in the Eastern Australian Current, shifting warm waters into the Tasman Sea and altering biotic features of the area (Poloczanska et al. 2009). Similarly, the

Kuroshio Current in the western North Pacific (an important foraging area for juvenile sea turtles) has shifted southward as a result of altered long-term wind patterns over the Pacific Ocean (Poloczanska et al. 2009).

Changes in air and sea surface temperatures can affect marine ecosystems in several ways. Direct effects decreases in sea ice and changes in ocean acidity, precipitation patterns, and sea level. Indirect effects of climate change include altered reproductive seasons/locations, shifts in migration patterns, reduced distribution and abundance of prey, and changes in the abundance of competitors and/or predators. Variations in sea surface temperature can affect an ecological community's composition and structure, alter migration and breeding patterns of fauna and flora and change the frequency and intensity of extreme weather events. For species that undergo long migrations, individual movements are usually associated with prey availability or habitat suitability. If either is disrupted, the timing of migration can change or negatively impact population sustainability (Simmonds and Eliott. 2009). Over the long term, increases in sea surface temperature can also reduce the amount of nutrients supplied to surface waters from the deep sea leading to declines in fish populations (EPA 2010), and, therefore, declines in those species whose diets are dominated by fish. Acevedo-Whitehouse and Duffus (2009) proposed that the rapidity of environmental changes, such as those resulting from global warming, can harm immunocompetence and reproductive parameters in wildlife to the detriment of population viability and persistence.

The potential for invasive species to spread may increase under the influence of climatic change. If water temperatures warm in marine ecosystems, native species may shift poleward to cooler habitats, opening ecological niches that can be occupied by invasive species introduced via ships ballast water or other sources (Philippart et al. 2011; Ruiz et al. 1999). Invasive species that are better adapted to warmer water temperatures can also outcompete native species that are physiologically geared towards lower water temperatures (Lockwood and Somero 2011). Altered ranges can also result in the spread of novel diseases to new areas via shifts in host ranges (Simmonds and Eliott. 2009). For example, it has been suggested that increases in harmful algal blooms could result from increases in sea surface temperature (Simmonds and Eliott. 2009). Moore et al. (2011) estimated that the impacts of a dinoflagellate establishment would likely intensify with a warming climate, resulting in roughly 13 more days of potential bloom conditions per year by the end of the 21st century.

Climate change will likely have its most pronounced effects on vulnerable species whose populations are already in tenuous positions (Williams et al. 2008). As such, we expect the risk of extinction to listed species to rise with the degree of climate shift associated with global warming. Increasing atmospheric temperatures have already contributed to documented changes in the quality of freshwater, coastal, and marine ecosystems and to the decline of endangered and threatened species populations (Karl 2009; Mantua et al. 1997).

Changes in the marine ecosystem caused by global climate change (e.g., ocean acidification, salinity, oceanic currents, dissolved oxygen levels, nutrient distribution) could influence the distribution and abundance of lower trophic levels (e.g., phytoplankton, zooplankton, submerged aquatic vegetation, crustaceans, mollusks, forage fish), ultimately affecting primary foraging areas of ESA-listed species including marine mammals, sea turtles, and fish. Marine species ranges are expected to shift as they align their distributions to match their physiological tolerances under changing environmental conditions (Doney et al. 2011). Hazen et al. (2013) examined top predator distribution and diversity in the Pacific Ocean in light of rising sea surface temperatures using a database of electronic tags and output from a global climate model. They predicted up to a 35 percent change in core habitat area for some key marine predators in the Pacific Ocean, with some species predicted to experience gains in available core habitat and some predicted to experience losses. Notably, leatherback turtles were predicted to gain core habitat area, whereas loggerhead turtles and blue whales were predicted to experience losses in available core habitat. McMahon and Hays (2006) predicted increased ocean temperatures will expand the distribution of leatherback turtles into more northern latitudes. The authors noted this is already occurring in the Atlantic Ocean. MacLeod (2009) estimated, based upon expected shifts in water temperature, 88 percent of cetaceans will be affected by climate change, with 47 percent predicted to experience unfavorable conditions (e.g., range contraction). Willis-Norton et al. (2015) acknowledged there will be both habitat loss and gain, but overall climate change could result in a 15 percent loss of core pelagic habitat for leatherback turtles in the eastern South Pacific Ocean.

Sea turtles occupy a wide range of terrestrial and marine habitats, and many aspects of their life history have been demonstrated to be closely tied to climatic variables such as ambient temperature and storminess (Hawkes et al. 2009). Sea turtles have temperature-dependent sex determination, and many populations produce highly female-biased offspring sex ratios, a skew likely to increase further with global warming (Newson et al. 2009; Patrício et al. 2017). Genetic analyses and behavioral data suggest that populations with temperature-dependent sex determination may be unable to evolve rapidly enough to counteract the negative fitness consequences of rapid global temperature change (Hays 2008 as cited in Newson et al. 2009). Altered sex ratios have been observed in sea turtle populations worldwide (Fuentes et al. 2009a; Mazaris et al. 2008; Reina et al. 2008; Robinson et al. 2008). This does not yet appear to have affected population viabilities through reduced reproductive success, although average nesting and emergence dates have changed over the past several decades by days to weeks in some locations (Poloczanska et al. 2009). Hayes et al. (2010) suggests that because of the increased frequency of male loggerhead breeding (based on visits to breeding sites) versus female breeding, the ability of males to breed with many females and the ability of females to store sperm and fertilize many clutches, skewed sex ratios due to climate change could be compensated for in some turtle populations and population effects may be ameliorated. However, such a fundamental shift in population demographics may cause a fundamental instability in the viability of some populations. In addition to altering sex ratios, increased temperatures in sea turtle nests can result in reduced incubation times (producing smaller hatchling), reduced clutch size, and reduced nesting success due to exceeded thermal tolerances (Azanza-Ricardo et al. 2017; Fuentes et al. 2010; Fuentes et al. 2011; Fuentes et al. 2009b).

Other climatic aspects, such as extreme weather events, precipitation, ocean acidification and sea level rise also have potential to affect marine turtle populations. Changes in global climatic patterns will likely have profound effects on the coastlines of every continent, thus directly impacting sea turtle nesting habitat (Wilkinson and Souter 2008). In some areas, increases in sea level alone may be sufficient to inundate turtle nests and reduce hatching success by creating hypoxic conditions within inundated eggs (Caut et al. 2009; Pike et al. 2015). Flatter beaches, preferred by smaller sea turtle species, would likely be inundated sooner than would steeper beaches preferred by larger species (Hawkes et al. 2014). Relatively small increases in sea level can result in the loss of a large proportion of nesting beaches in some locations. For example, a study in the northwestern Hawaiian Islands predicted that up to 40 percent of green turtle nesting beaches could be flooded with 0.9 meters of sea level rise (Baker et al. 2006). The loss of nesting beaches would have catastrophic effects on sea turtle populations globally if they are unable to colonize new beaches that form, or if the newly formed beaches do not provide the habitat attributes (sand depth, temperature regimes, refuge) necessary for egg survival.

Changing patterns of coastal erosion and sand accretion, combined with an anticipated increase in the number and severity of extreme weather events, may further exacerbate the effects of sea level rise on turtle nesting beaches (Wilkinson and Souter 2008). Climate change is expected to affect the intensity of hurricanes through increasing sea surface temperatures, a key factor that influences hurricane formation and behavior (EPA 2010). The intensity of tropical storms in the Atlantic Ocean, Caribbean, and Gulf of Mexico has risen noticeably over the past 20 years and six of the 10 most active hurricane seasons have occurred since the mid-1990s (EPA 2010). Extreme weather events may directly harm sea turtles, causing “mass” strandings and mortality (Poloczanska et al. 2009). Studies examining the spatio-temporal coincidence of marine turtle nesting with hurricanes, cyclones and storms suggest that cyclical loss of nesting beaches, decreased hatching success and hatchling emergence success could occur with greater frequency in the future due to global climate change (Hawkes et al. 2009). Pike et al. (2006) concluded that warming sea surface temperatures may lead to potential fitness consequences in sea turtles resulting from altered seasonality and duration of nesting. Sea turtles may expand their range as temperature-dependent distribution limits change (McMahon and Hays 2006). Warming ocean temperatures may extend poleward the habitat which sea turtles can utilize (Poloczanska et al. 2009).

Global climate change may affect the ESA-listed fish species and DPSs considered in this opinion. Thermal changes of just a few degrees Celsius can substantially alter fish protein metabolism (McCarthy and Houlihan 1997), response to aquatic contaminants (Reid 1997), reproductive performance (Van Der Kraak and Pankhurst 1997), smolt development (McCormick et al. 1997), species distribution limits (McCarthy and Houlihan 1997), and community structure of fish populations (Schindler 2001). Apart from direct changes to anadromous fish survival, increased water temperatures may alter habitat, including the availability of prey (ISAB 2007).

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

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

Changes in precipitation patterns are anticipated as a result of global climate change. The increased rainfall predicted in some areas may increase runoff and scour spawning areas, and flooding events could cause temporary water quality issues. Increased extremes in river flow (i.e., periods of flooding and low flow) can alternatively disrupt and fill in spawning habitat that sturgeon and salmon rely upon (ISAB 2007). In some areas, longer and more frequent droughts are predicted, in combination with increased water withdrawal for human use, may cause loss of habitat including loss of access to spawning habitat. Drought conditions in the spring may also expose eggs and larvae in rearing habitats. If a river becomes too shallow or flows become intermittent, anadromous fish may become susceptible to strandings or habitat restrictions. Low flow and drought conditions are also expected to cause additional water quality issues. Any of the conditions associated with climate change are likely to disrupt river ecology, causing shifts in community structure and the type and abundance of prey. Additionally, cues for spawning migration and spawning could occur earlier in the season, which might affect prey availability in rearing habitat. Overall, it is likely that global climate change would increase pressures on the survival and recovery of ESA-listed sturgeon and salmon populations considered in this opinion.

9.9 Characterization of Origination and Destination Ports

In this section we characterize the origination and destinations ports proposed by the Navy for their inactive ship tow program. Table 28 provides summary information for each port, including water type, salinity, and water temperature (winter and summer).

9.9.1 Baltimore, Maryland

The Port of Baltimore is ranked as the sixteenth largest port in the U.S. in terms of total trade (American Association of Port Authorities 2016b). It is the deepest harbor in the Chesapeake Bay area. The Baltimore Harbor estuary is the fifteen-mile tidal region of the lower Patapsco River. The Patapsco River Watershed (which includes seven sub-watersheds) drains directly into the Inner Harbor of Baltimore. The port of Baltimore is deep (46 to 50 feet) with mesohaline/brackish water with average salinities ranging from around seven to ten ppt.

The Chesapeake Bay has become home to more than 200 species. Some of the well-known invasives include blue and flathead catfish, zebra mussels, Chinese mitten crab, veined rapa whelk, Japanese shore crab, and northern snakehead. The Invasive Catfish Task Force was established in 2012 by the Chesapeake Bay Program's Sustainable Fisheries Goal Implementation Team to identify management options and respond to the spread of these species.

Table 28. Summary information of origin and destination ports[1].

Origin and Destination Ports	Water Type[2]	Average Port Salinity (parts per thousand)	Water Temperature Low (Winter, degrees Fahrenheit)	Water Temperature High (Summer, degrees Fahrenheit)
Baltimore, Maryland	Mesohaline	7.6-10 ppt	40°F	82°F
Beaumont, Texas	Oligohaline	1.0-4.0 ppt	55°F	91°F
Brownsville, Texas	Euhaline (marine)	34-37 ppt	66°F	82°F
Mayport, Florida	Mesohaline	14.5 ppt	58°F	83°F
New Orleans, Louisiana	Oligohaline	2-5 ppt	61°F	84°F
Norfolk, Virginia	Polyhaline	18-21 ppt	46°F	80°F
Pearl Harbor, Hawaii	Euhaline (marine)	34 ppt	74°F	82°F
Philadelphia, Pennsylvania	Freshwater	0.0-0.10 ppt	37°F	77°F
Puget Sound, Washington	Polyhaline	24-30 ppt	48°F	55°F
San Diego, California	Euhaline (marine)	34 ppt	59 °F	72 °F

[1] Salinity and temperature information obtained from the following sources: NOAA Tides and Currents https://tidesandcurrents.noaa.gov; Sea Temperature Information https://seatemperature.info/; Chesapeake Bay Program https://www.chesapeakebay.net/; NOAA National Centers for Environmental Information https://www.nodc.noaa.gov/dsdt/cwtg/all_meanT.html; World Sea Temperatures https://www.seatemperature.org/; USGS Water Data for the Nation https://nwis.waterdata.usgs.gov/nwis; Water Data for Texas https://waterdatafortexas.org/coastal; (Tolan 2007); (University of North Florida 2017); State of Washington Department of Ecology https://ecology.wa.gov/Research-Data/Monitoring-assessment/Puget-Sound-and-marine-monitoring

[2] Water types defined as follows: freshwater 0.0 – 0.5 ppt; oligohaline 0.5 – 5.0 ppt; mesohaline 5.0 – 18 ppt; polyhaline 18 – 30 ppt; euhaline (marine) 30 ppt or greater (Harte Research Institute 2018; Montagna et al. 2012).

9.9.2 Beaumont, Texas

The Port of Beaumont is a deep-water channel (36 to 40 feet) that is 400 feet wide near the mouth of the Neches River. It is the fifth busiest port in the U.S. for total trade (American Association of Port Authorities 2016b) and the fifty-eighth busiest in the world in terms of tonnage (American Association of Port Authorities 2016a). The Neches Basin is the third largest river basin in Texas and forth largest by average flow volume. The Neches River flows from headwaters in Van Zandt County to its confluence with Sabine Lake which drains into the Gulf of Mexico. As part of the proposed action, the Navy would berth inactive ships in the Neches River at the Maritime Administration (MARAD) Beaumont Reserve Fleet, which is located about ten miles upstream from Sabine Lake estuary (Figure 51). Salinities in this section of the Neches River fluctuate depending on tides, river flow, season, and the operating status (i.e., open or closed) of the saltwater barrier located several miles upstream of the Port of Beaumont. In addition, large differences in surface and bottom water salinities have been reported in the Lower Neches River, indicative of limited mixing of surface and bottom waters (Academy of Natural Sciences 2006). Based on several sources of salinity information, we categorize the Beaumont Reserve Fleet location as oligohaline, with average salinities likely ranging from about 1.0 to 4.0 ppt (Academy of Natural Sciences 2006; LA Coast 2018; NOAA 2018b; NOAA 2018c). Salinities are expected to be greater in the lower portion than in the upper portion of the water column, and during periods of low river flow (i.e., drought years).

According to the Texas Parks and Wildlife, the threat of invasives entering estuaries from the Gulf of Mexico is increasing, because there are so many ways to enter the waters. Some marine invasive threats throughout the Gulf of Mexico include the brown mussel, Australian spotted jellyfish, Asian green mussel, and lionfish as well as marine plants such as the water hyacinth, hydrilla, and giant salvinia. Texas Parks and Wildlife has documented reports of the presence of hydrilla and giant salvinia in the upper Neches River as of 2011. Giant salvinia, currently the top aquatic invasive plant management priority is capable of doubling its population in two to eight days under the right conditions. According to the Texas Department of Agriculture, hydrilla is one of the state’s worst invaders and covers 75,000 to 100,000 surface acres of water found in nearly 100 of Texas reservoirs.

9.9.3 Brownsville, Texas

The Port of Brownsville is ranked as the sixty-sixth largest port in the U.S. in terms of total trade (American Association of Port Authorities 2016b). Brownsville is part of the Boca Chica subdelta of the Rio Grande. The Port is at the southern terminus of the Gulf Intracoastal Waterway near the mouth of the Rio Grande and Lower Rio Grande Valley plain. Port staff and members of the Brownsville Navigation District are working to secure funding to deepen the port from 42 feet to 52 feet, to allow for bigger cargo ships. If funding is found, the project would start in 2020. Brownsville is part of the Rio Grande River system which flows from south-central Colorado and flows into the Gulf of Mexico. The salinity in Brownsville is around 34 to 37 ppt.

Invasive aquatic species are of a high concern in the state of Texas and the Texas Parks and Wildlife restricts the importation and possession of approximately 600 species of fishes, shellfishes, and aquatic plants. Boaters in the state are required by law to remove all harmful plants and animals from boats and trailers before leaving the vicinity of a lake, river, or bay. The two species of the greatest concern are zebra mussels and giant salvinia. Texas Parks and Wildlife has a report of water lettuce in Brownsville as of 2011.

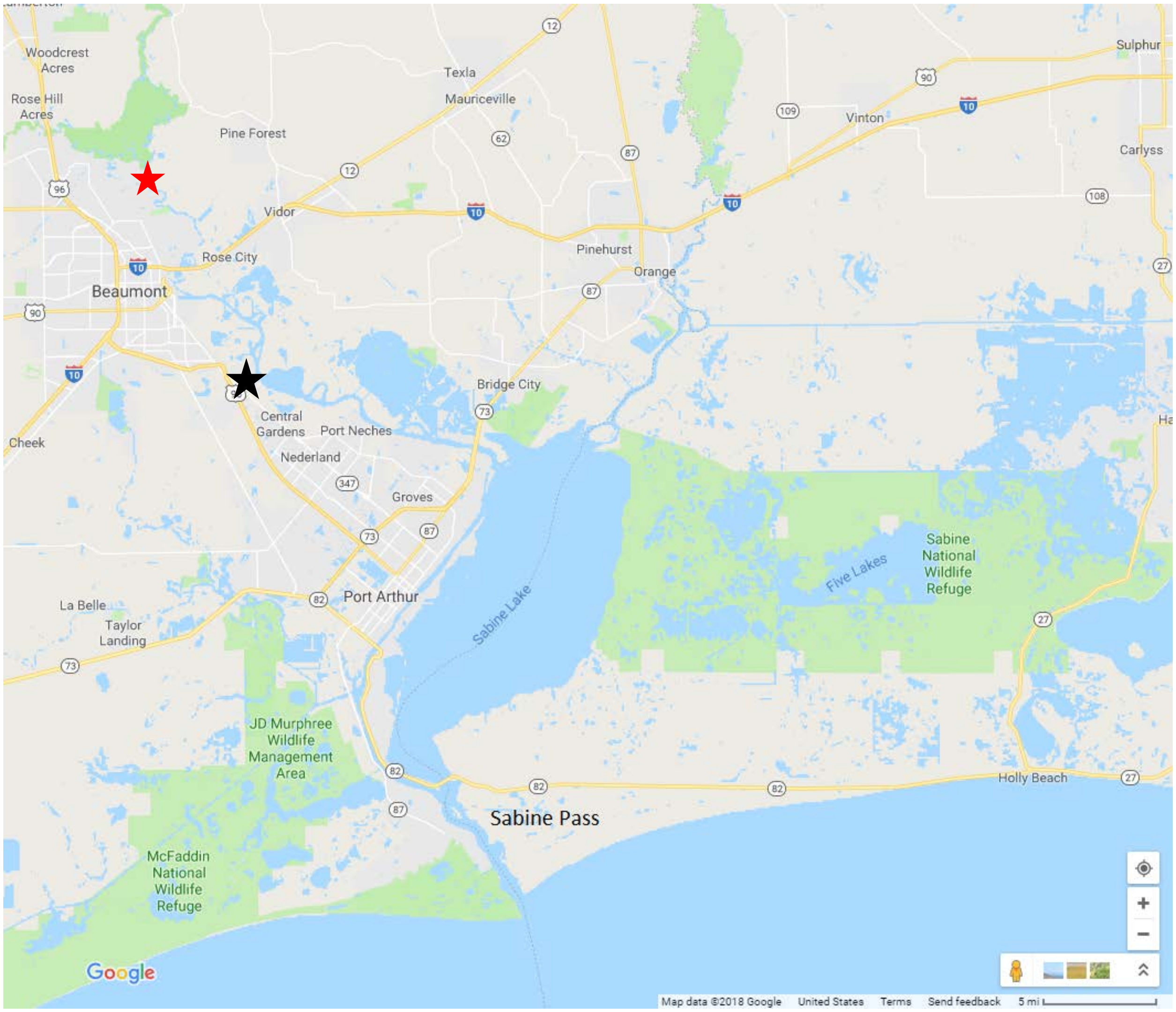


Figure 51. Map showing the location of the Beaumont Reserve Fleet (black star) on the Neches River, the saltwater barrier (red star), Sabine Lake, and Sabine Pass out to the northern Gulf of Mexico.

9.9.4 Mayport, Florida

Mayport is located near the northern mouth (lower basin) of the St. Johns River via Jacksonville as it empties into the Atlantic Ocean. The St. Johns River is the longest river in the state of Florida and is separated into three basins and two watersheds. Because the river flows in a northerly direction, the upper basin is located in the headwaters of the river at its southernmost point. Naval Station Mayport was commissioned in 1942 and has grown to become the third largest Navy fleet concentration in the U.S. It was reactivated in 2008 after being deactivated in 1950. The average salinity value for Mayport Florida from the State of the Lower St. Johns River Basin River Report is 14.5 ppt (University of North Florida 2017).

The lower St. Johns river basin is commonly divided into three ecological zones based on salinity differences. The mesohaline riverine zone is the most northern ecological zone and stretches from the Atlantic Ocean, through Mayport, to the Fuller Warren Bridge in Jacksonville. The mesohaline riverine zone is typically deeper and well-mixed with an average salinity of 14.5 ppt and a fast flow rate (Hendrickson and Konwinski 1998; Malecki et al. 2004; University of North Florida 2017). Over the past ten years, the status of non-native aquatic species has been "unsatisfactory" and the trend "worsening" (University of North Florida 2017). The estimated number of invasive species was 56 in 2008 and increased to 80 in 2016. These species include submerged aquatic plants, mollusks, fish, crustaceans, amphibians, jellyfish, mammals, reptiles, and algae. For example, the spread of the Cuban tree frog and lionfish is of increasing concern (University of North Florida 2017).

9.9.5 New Orleans, Louisiana

The Port of New Orleans is a deep-draft (36 to 40 feet) multipurpose port. The port handles about ninety million short tons of cargo every year and is the forth busiest port in the U.S. (American Association of Port Authorities 2016b). The Port of New Orleans and Port of South Louisiana combined form one of the largest port systems in the world by bulk tonnage and among the top ten in the world by volume handled (American Association of Port Authorities 2016a). The port is located at the center of the Lower Mississippi River complex of Louisiana. Due to the sheer volume of freshwater discharge from the river and its outlets, the delta can be classified as a mixing zone for fresh and salt water. Saltwater intrusion occurs when freshwater flow decreases in volume, allowing heavy saltwater from the Gulf to move upstream (U.S. Army Corps of Engineers 2016). Chabrek (1972) divided coastal marshes based on salinity into four vegetation zones: fresh, intermediate, brackish, and saline. The area of New Orleans is located in the "intermediate" zone and salinity for this area typically ranges from two to five ppt (i.e., oligohaline), however instances of saltwater intrusion are becoming more common.

The Mississippi River drains 41 percent of the 48 contiguous states of the U.S. The basin covers more than 1,245,000 square miles, includes all or parts of 31 states and two Canadian provinces, and roughly resembles a funnel which has its spout at the Gulf of Mexico (U.S. Army Corps of Engineers 2018). In Louisiana, millions of dollars of business and thousands of jobs are related to the handling, financing, processing and transporting of cargo passing through the Lower Mississippi River. The Louisiana coast is home to 25 percent of commercial fisheries in the continental U.S., with more than one billion pounds caught annually. The fisheries have a dockside value of $291 million and a recreation value of $944 million. The catch is comparable to the entire Atlantic seaboard and is triple that of the remaining states along the Gulf of Mexico (U.S. Army Corps of Engineers 2018).

The Louisiana Department of Wildlife and Fisheries have efforts to produce a demand for to invasive species of Asian carp: the silver and bighead. Both species are firmly established throughout the Mississippi and Atchafalaya Rivers. Reports have shown the presence of giant tiger prawns in Vermillion and Barataria Bays. The Northern snakehead, which is native to Asia, has been found in Louisiana, including in the Mississippi River, and is a current threat.

9.9.6 Norfolk, Virginia

The Port of Norfolk is located on Sewell's Point, a peninsula of land within the independent city of Norfolk. It is located at the mouth of the salt-water port of Hampton Roads, and is bordered on three sides with Willoughby Bay to the north, Hampton Roads to the west, and the Lafayette River to the south. The James River, Nansemond River, and Elizabeth River converge into Hampton Roads which leads out into the Chesapeake Bay. The average salinity in Norfolk, which can fluctuate widely due to the proximity to freshwater inputs, averages around 18 to 21 ppt.

According to the Virginia Invasive Species website, the northern snakehead is found in the Potomac River and many of its tributaries in Virginia. The Chinese mitten crab was found in the Chesapeake Bay in 2006 and a number of individuals have been found around the Bay. The rusty crayfish has become an aggressive and destructive invader of West Virginia. The zebra mussel was discovered in a quarry pond in northern Virginia in 2003. Early detection and control efforts have thus far prevented zebra mussel from establishing in the state. The invasive aquatic plant water hyacinth have been found in Norfolk with additional risk of invasion from giant salvinia, water spinach, and beach vitex.

9.9.7 Pearl Harbor, Hawaii

Pearl Harbor is located on the southern coast of the island of Oahu, Hawaii. Joint Base Pearl Harbor-Hickham provides services similar to other larger ports. It is one of the Navy's busiest, completing about 65,000 boat runs and transporting 2.4 million passengers and 200,000 vehicles each year (Commander Navy Region Hawaii 2018). Pearl Harbor is the largest estuary on the Islands and has the largest freshwater spring complex in the islands (Englund et al. 2000) with shoreline, submarine, diffuse, and non-point source groundwater inputs. Surface water salinities within Pearl Harbor vary from about 22 to 34.5 ppt (Kelly 2012) but are closer to 34 ppt in the port locations where inactive Navy ships are located.

The Hawaii Invasive Species Council was established in 2003 for the special purpose of providing policy level direction, coordination, and planning among state departments, federal agencies, and international and local initiatives for the control and eradication of harmful invasive species infestations throughout the State and for preventing the introduction of other invasive species that may be potentially harmful. Hawaii is a unique place due in part to its geographic isolation and volcanic origin. Up to 346 alien marine algae and invertebrate species are currently established in Hawaiian waters through different vectors including aquaculture, unmanaged ballast water, and ship biofouling.

Some marine invasive species currently being managed in Hawaii include snowflake coral, peacock grouper, tilapia, upside-down jellyfish, keyhole sponge. Freshwater invasive species include: banded jewel cichlid, smallmouth bass, green swordtail, convict cichlid, guppies, suckermouth catfish, water fern, Asiatic clam, freshwater eel, Tahitian prawn, and apple snail. Five species of alien algae have become invasive in Hawaii: prickly seaweed, leather mudweed, hookweed, smothering seaweed, and gorillo ogo.

9.9.8 Philadelphia, Pennsylvania

The Port of Philadelphia is ranked as the thirtieth largest port in the U.S. in terms of total trade (American Association of Port Authorities 2016b) and the eighty-third busiest in the world by bulk tonnage (American Association of Port Authorities 2016a). According to Philadelphia Water Department, Philadelphia is part of the Delaware River watershed which has seven sub-watersheds: Delaware District, Schuylkill, Pennypack, Tookany/Tacony-Frankford, Darby-Cobbs, Poquessing, and Wissahickon. The Delaware River Basin extends from the Catskill Mountains in New York and stretches more than 330 miles through four states draining through the Delaware Estuary to the Atlantic Ocean. The entire watershed drains nearly 13,000 square miles, of which Philadelphia’s contribution is only 40 square miles into the Delaware River. The Delaware Estuary provides drinking water for over one million people (Partnership for the Delaware Estuary 2017). The location of the salt line in the Delaware River fluctuates as streamflow increases or decreases. The port of Philadelphia is considered freshwater with salinities around zero ppt (Partnership for the Delaware Estuary 2017).

The state of Pennsylvania is currently managing the following invasive species: European ruffe, sea lamprey, hydrilla, spiny water flea, purple loosestrife, Eurasian watermilfoil, Asian clam, and red-eared slider (turtle). Aquatic invasive species with action plans include: Asian carp complex, didymo (alga), golden alga, viral hemorrhagic septicemia (disease), and the water chestnut. The state has banned some aquatic species including: bighead carp, black carp, European rudd, quagga mussel, round goby, ruffe, rusty crayfish, silver carp, snakehead (all species), tubenose goby, and zebra mussel.

9.9.9 Puget Sound, Washington

The Puget Sound Naval Shipyard is located in Bremerton, Washington on the Sinclair Inlet. The Sinclair Inlet is an estuary in the middle of Puget Sound, west of Seattle. Puget Sound is the second largest estuary in the U.S. (after the Chesapeake Bay) and is fed by highly seasonal freshwater from the Olympic and Cascade Mountain watersheds. Puget Sound is considered polyhaline with salinity ranging from about 24 to 30 ppt.

Some invasive species that have established in Puget Sound include: cordgrass, Japanese eelgrass, *Sargassum*, purple varnish/mahogany clam, gallo mussel, Atlantic/Eastern oyster drill, Japanese oyster drill, Asian mudsnail, Bamboo worm, seven species of tunicate, red swamp crayfish, and Eurasian water milfoil.

9.9.10 San Diego, California

The port of San Diego was established in 1962 and serves the port's five member cities: Chula Vista, Coronado, Imperial Beach, National City, and San Diego. The port cares for thirty-four miles of waterfront with an economic impact of $8.3 billion in 2015. The U.S. Bureau of Transportation Statistics ranked the port one of America's top thirty containership ports bringing in nearly 3,000,000 metric tons of cargo per year. Naval Base San Diego is the second largest surface ship base in the U.S. and is the principal home of the Pacific Fleet, consisting of over fifty ships and 190 tenant commands.

The first introduction of invasive marine species into San Diego Bay may have come from the ships used by the early Spanish explorers. Some invasive species have been in their current ecosystem for so long that they were assumed to be natives until recent genetic analyses proved otherwise. According to a 1999 study, the following invasive species taxa are present in San Diego Bay: one species of marine algae, one marine protozoan, forty-seven marine invertebrates, and five marine fish. There are also twenty-eight species of invasive coastal plants. In total, at least eighty-two non-native species are found in the Bay. The San Diego Bay Integrated Natural Resources Management Plan has a complete listing of all the invasive species found in and around San Diego Bay (Naval Base Coronado 2013).

10 EFFECTS OF THE ACTION

Section 7 regulations define "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 (50 C.F.R. §402.02). Indirect effects are those that are caused by the proposed action and are later in time, but are reasonably certain to occur.

The jeopardy analysis relies upon the regulatory definition of "to jeopardize the continued existence of a listed species," which is "to engage in an action that would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 C.F.R. §402.02). Therefore, the jeopardy analysis considers both survival and recovery of the species.

The destruction and adverse modification analysis considers whether the action produces "a direct or indirect alteration that appreciably diminished the value of critical habitat for the conservation of a listed species. Such alterations may include, but are not limited to, those that alter the PBFs essential to the conservation of a species or that preclude or significantly delay development of such features." 50 C.F.R. 402.02.

This section evaluates the (1) exposure of ESA-listed species to the potential stressors resulting from the proposed action, (2) the range of responses those species that are exposed may exhibit, and (3) the consequences of the responses to the individuals that have been exposed, the populations those individuals represent, and the species those populations comprise. Exposure analyses identify the ESA-listed species and critical habitats that are likely to co-occur with the proposed action's effects on the environment in space and time, and identify the nature of that co-occurrence. Response analyses identify the potential and likely responses of ESA-listed species and habitats exposed to stressors associated with the proposed action.

Our effects analysis is divided into three parts. First, in Section 10.1 we discuss those activities and stressors that we determined are not likely to adversely affect (NLAA) any ESA-listed species or critical habitat. These include ship movement between origination and destination ports, ship dismantling, and non-indigenous species introductions as a result of towing inactive Navy ships between ports. Next, in Section 10.2 we evaluate the effects of in-water hull cleaning on ESA-listed species and critical habitat. We found that in-water hull cleaning was the only activity that would likely result in adverse effects on ESA-listed species. Our rationale for reaching the effects determinations for each stressor and ESA-listed species (or designated critical habitat) is explained below. Finally, in Section 10.3 we summarize our effects determinations for each ESA-listed species and critical habitat evaluated in our effects analysis.

10.1 Activities and Stressors Not Likely to Adversely Affect ESA-listed Species

This section includes a discussion of those activities and associated stressors that we determined are not likely to adversely affect (NLAA) any ESA-listed species or critical habitat. These include ship movement between origination and destination ports, ship dismantling, and non-indigenous species introductions as a result of towing inactive Navy ships between ports. We determined that any adverse effects resulting from ship movements on the ESA-listed species or designated critical habitats in the action area would be either discountable (as is the case for ship strike) or insignificant (as is the case for ship sound and disturbance). We determined that any adverse effects resulting from the ship dismantling process, including water and sediment quality degradation and sound, on any ESA-listed species or designated critical habitat in the action area are extremely unlikely to occur and thus would be discountable. We also determined that adverse effects to any ESA-listed species or critical habitat resulting from exposure to an invasive species introduced as a result of the proposed action is extremely unlikely to occur and thus is discountable. This determination assumes the Navy will implement the agreed upon mitigation measures described in Section 4.

10.1.1 Ship Interaction Effects Analysis

In this subsection we evaluate the effects of towing inactive Navy ships on ESA-listed species (see Section 8.2 for complete list of species) first from ship strike, and then from ship sound and/or disturbance.

10.1.1.1 Ship Strike

Ship and tow line strike could potentially affect ESA-listed marine mammals, sea turtles and fish species occurring within the action area. Turtles may use auditory cues to react to approaching ships rather than visual cues, making them more susceptible to strike as ship speed increases (Hazel et al. 2007). In the rare event they are encountered by a tugboat towing an inactive ship, sea turtles are expected to exhibit avoidance behavior. Similarly, killer whales and monk seals would be expected to avoid the slow moving tugs and towed inactive ships. Given the slow tow speeds and small number of ships towed each year, a marine mammal or sea turtle strike occurring as a result of the proposed action is extremely unlikely to occur and thus would be discountable.

Large ships that transit through shipping channels typically draft close to the bottom of the channel, which increases the likelihood of interactions with bottom-dwelling fish such as the ESA-listed sturgeon species considered in this opinion. Transit routes for the proposed action will take ships to and from Philadelphia through the Delaware River and Delaware Bay, where they may encounter Atlantic and shortnose sturgeon. Ships towed out of Norfolk may also overlap with sturgeon migratory routes to and from the James River. The Atlantic Sturgeon Status Review Team (ASSRT) determined that the Delaware River subpopulation is at a moderately high risk of extinction because of ship strikes, and sturgeon in the James River are at a moderate risk from ship strikes (ASSRT 2007). Brown and Murphy (2010) examined 28 dead Atlantic sturgeon from the Delaware River from 2005 through 2008 and found that fifty percent of the mortalities resulted from apparent ship strikes, and 71 percent of these (10 out of 14) had injuries consistent with being struck by a large ship. Shortnose sturgeon may not be as susceptible due to their smaller size in comparison to the larger Atlantic sturgeon, for which ship strikes have been documented more frequently. There has been only one confirmed incidence of a ship strike on a shortnose sturgeon in the Kennebec River, and two suspected ship strike mortalities in the Delaware River (SSSRT 2010). There are no reported incidents of ship strike of Atlantic or shortnose sturgeon in the St John's River or of green sturgeon in Puget Sound.

While ship strike remains a general threat to sturgeon within portions of the action area, the likelihood that the proposed action would result in a ship strike remains very low given the low tow speeds, infrequent towing events anticipated, and the minimal time that a ship would be in areas where a strike may occur. We anticipate an average of nine ships will be towed per year between any of the origin-destination port combinations included in this program. Even if all nine ship tows in a given year involved ports with ESA-listed sturgeon (i.e., Philadelphia, Norfolk, Mayport, or Puget Sound), this would still represent an extremely small fraction of the total ship traffic within these estuaries on an annual basis. The Delaware River shoreline has six major petroleum refineries that process nearly one million barrels of crude oil per day, as well as other chemicals associated with the refining process, making it one of the major petroleum infrastructures in the United States (Altiok et al. 2012). The ports of Philadelphia, South Jersey and Wilmington combined make one of the largest general cargo port complexes in the nation involving container, general cargo and bulk terminals. Commercial ship traffic is expected to increase in the future as the Delaware River's main channel is currently being deepened from 40 to 45 feet to accommodate larger ships into various port terminals (Altiok et al. 2012). The 50-foot deep Port of Virginia, which includes three marine terminals (Norfolk International, Portsmouth Marine and Newport News Marine), is one of the major commercial shipping areas along the U.S. Atlantic coast. In 2017, total import/export throughput at the Port of Virginia was 2.84 million TEUs (twenty-foot equivalent unit) (POV 2018). In addition to commercial traffic, Norfolk's naval facility is one of the largest shipyards in the world; inactive fleet ships towing represents a very small fraction of the total large ship activity at this port. Given the relatively small number of ESA-listed sturgeon that are struck by ships each year, a ship strike resulting from towing an inactive ship is extremely unlikely to occur and thus would be discountable.

Atlantic sturgeon early life stages (e.g., eggs and larvae) could also be affected by the movement of large ships through shipping channels to and from the port of Philadelphia. Sturgeon eggs are highly adhesive and are deposited on the bottom substrate, usually on hard surfaces (e.g., cobble) (Smith and Clugston 1997). Hatching occurs approximately 94 to 140 hours after egg deposition, and larvae assume a demersal existence (Smith et al. 1980). Propellers on ships with deep drafts relative to depth can stir up the water column and disturb or entrain sturgeon early life stages that are either attached to the bottom substrate or drifting in the lower part of the water column, resulting in injury or death (Killgore et al. 2001). For the proposed action to adversely affect Atlantic sturgeon early life stages, the towing route would have to overlap both temporally and spatially with sturgeon spawning activity. The likelihood of this occurring is low considering the small number of inactive ships that would be towed each year through areas where sturgeon are known to spawn. In addition, the shipping channels used for towing inactive ships are high ship traffic areas for commercial shipping and other maritime activities. The sediment and lower water column in these areas are likely disturbed by large ships on a regular basis, thus removing or displacing many of the fish eggs and larvae that may settle in these channels.

Other ESA-listed fish species that occur within the action area (i.e., Pacific salmonids, eulachon, bocaccio, and yelloweye rockfish) could potentially be susceptible to ship strikes, although reported ship strikes for these species are extremely rare. Due to the slow ship tow speeds, high fish maneuverability, distribution of these species throughout the water column, and their relatively small body size, a ship strike of one of these fish species as a result of the proposed action is extremely unlikely to occur and thus would be discountable.

We have no information to indicate that ESA-listed species have been struck by towed inactive Navy ships in the past. Although the tug, tow cable, and towed ship may affect ESA-listed species encountered along the proposed tow routes, the chance that such an encounter would result in an adverse effect due to an injury or behavioral disturbance is extremely remote because of the low probability that an ESA-listed species would overlap with the infrequent towing events (i.e., estimated less than ten per year). In summary, given the low speeds, infrequency of transit, and the expected low density of ESA-listed species along the tow routes, the likelihood of ships encountering ESA-listed species and posing a strike risk is extremely unlikely to occur and thus would be discountable.

10.1.1.2 Ship Sound and Disturbance

Sound from contracted tug ships and towed Navy ships may be detectable to ESA-listed mammals, sea turtles and fish, although the density of species in the open ocean is so low that they are unlikely to be encountered. Nearshore species are more likely to be encountered in transit in the estuaries where origination and destination ports occur. However, these areas are already heavily trafficked by ships and the infrequency of towed Navy ships is not expected to substantially increase noise levels above background or ambient conditions. Any response elicited from ESA-listed marine mammals, sea turtles or fish due to ship noise is expected to be in the form of behavioral avoidance or interruption in behavior. We believe any behavioral response of ESA-listed species to ship noise will be of limited duration and magnitude. The temporary disturbance of marine mammals, sea turtles or fish associated with the anticipated infrequent ship interactions as part of the proposed action are not expected to result in injury or reduced fitness as we do not anticipate any significant disruption of breeding, feeding, or sheltering to occur. Therefore, any potential effects from ship sound or ship avoidance behavior resulting from the proposed action would be undetectable, not measurable, or so minor that they cannot be meaningfully evaluated, and thus would be insignificant.

10.1.2 Ship Breaking Effects Analysis

For this consultation, ship breaking (or dismantling) is considered an interrelated action associated with the Navy's proposed action. In this subsection, we evaluate the effects of ship breaking on the ESA-listed species (see Section 8.2 for complete list of species). We also evaluate the effects of shipbreaking on the following species' designated critical habitats that would potentially overlap with (or are in close proximity to) destination port areas where ship breaking may be conducted as an interrelated action: killer whale Southern Resident DPS, Chinook salmon Puget Sound DPS, steelhead Puget Sound DPS, bocaccio Georgia Basin/Puget Sound DPS, yelloweye rockfish Georgia Basin/Puget Sound DPS, Atlantic sturgeon New York Bight DPS, and Atlantic sturgeon Chesapeake Bay DPS. We discuss water and sediment quality degradation related stressors associated with shipbreaking, and sound-related stressors associated with shipbreaking.

10.1.2.1 Water and Sediment Quality Degradation

If not contained and disposed of properly during the shipbreaking process, hazardous materials (e.g., PCBs, petroleum products, asbestos) commonly found in ships have the potential to affect ESA-listed species and critical habitats. Ship dismantling companies that are awarded contracts to tow and dismantle inactive ships are responsible for all work associated with the removal and proper disposal of hazardous materials. Contractors must comply with all applicable federal, state, and local environmental laws and regulations during the processing, use, or disposal of any material under an awarded contract. Shipbreaking companies must submit an Environmental Compliance Plan as part of the bid process. Bidders must demonstrate how the shipbreaking facility will ensure safe and environmentally sound management of all hazardous materials and wastes removed from a ship recycled at the facility, including information for asbestos, PCBs, fuels and oils, bilge/ballast water, heavy metals, paints and coatings, waste water/sludge, ozone depleting substances and other potential hazardous materials. In addition, bidders must certify and/or verify that the dismantling facility has developed, implemented, and maintains a Spill Prevention, Control and Countermeasures Plan and a Stormwater Pollution Prevention Plan. The bidder must also reveal any Notices of Violations, fines or proposed fines, convictions or citations associated with environmental compliance, and whether the bidder has been the subject of any judicial or administrative proceeding related to the violation of any applicable law related to environmental compliance.

Based on the Navy's BE (NUWC 2017a), we anticipate an average of four inactive ships will be towed for the purpose of dismantlement per year as part of the Navy's program. While ships could be dismantled at any of the seven different destination ports proposed, based on recent history we expect most shipbreaking would occur in the port of Brownsville, Texas, considered the shipbreaking capital of the United States. While ESA-listed sea turtles are known to occur at this port, we anticipate that the number of turtles in the immediate vicinity of where shipbreaking would occur to be low, further reducing the likelihood of exposure to this activity. There is no ESA designated critical habitat within the port of Brownsville.

Based on the requirements for environmental compliance related to the shipbreaking process described above, and the low anticipated density of ESA-listed species in the immediate vicinity of this activity, we have determined that the potential effects to ESA-listed species and critical habitats associated with exposure to contaminant or hazardous material discharged from the shipbreaking process is extremely unlikely to occur and thus would be discountable.

10.1.2.2 Sound

Noise from shipbreaking activities would likely be detectable to ESA-listed species if they were in close proximity to the ship breaking facility. However, we expect ESA-listed species to be rare in the industrialized portions of ports where shipbreaking facilities are located due to the number of ships, amount of noise, and possible reduced water quality from the general industrialization of the immediate area and development of surrounding areas. Further, it is unlikely that shipbreaking activities associated with the proposed action would significantly increase underwater noise levels above the baseline level at these high traffic, industrialized ports. Therefore, we determine that the potential for effects to ESA-listed species and critical habitat associated with underwater noise from shipbreaking activities is extremely unlikely to occur and thus would be discountable.

10.1.3 Nonindigenous Aquatic Species Effects Analysis

Nonindigenous aquatic species (NAS) are defined as species that enter a body of water or aquatic ecosystem outside of their historic or native range (USGS 2018). The successful establishment and spread of a NAS in a new environment depends upon multiple factors, including climate, food resources, its reproductive strategy, competition, predation, and parasitism (Carlton 1996). When a NAS threatens the diversity or abundance of native species, the ecological stability of infested waters, or any commercial, agricultural, aquacultural or recreational activities dependent on such waters, it is referred to as an aquatic nuisance species (ANS).

10.1.3.1 Spread of Nonindigenous Aquatic Species Via Hull Fouling

The global movement of large ships is considered one of the primary vectors for the introduction of NAS (Davidson et al. 2009). Nonindigenous species are introduced by ships through ballast water exchange and hull fouling. Hull fouling (or biofouling) involves the attachment of generally bottom-associated sessile and sedentary organisms to a ship’s surface in contact with water over time. Hull fouling begins with the growth of a biofilm or slime layer of bacteria and algae. This is typically followed by a succession of early colonizers such as barnacles, bryozoans, and tube worms and then mollusks, sponges, sea squirts, and seaweeds. Highly developed fouling communities may also provide microhabitats for mobile organisms, such as fish, crabs, and sea stars.

Molnar et al. (2008a) reported shipping as a potential vector for up to 69 percent of the 329 ANS they analyzed. Of those, 69 percent (227 species) introduced by shipping, 39 percent were exclusively introduced by hull fouling, 31 percent exclusively by ballast water, and the other 30 percent by both vectors. Based on life history characteristics, Bax et al. (2003) identified hull fouling as the probable mechanism of invasion for over 75 percent of introduced species in Australia and for about 35 percent of introduced species in San Francisco Bay. Davidson et al. (2009) examined the relative contribution of hull fouling as a vector to established aquatic species invasions at eight temperate global locations, where invasions have been well documented: Belgium, Germany and Ireland in Europe; Puget Sound, San Francisco Bay and Los Angeles/Long Beach on the U.S. West Coast; and New Zealand and Port Philip Bay (Melbourne, Australia) in the southern Hemisphere. A total of 838 records of 553 different invasive species have been recorded across these sites, including 102 algal species, 129 arthropods, 75 molluscs, and 56 bryozoans. For some taxa, including ascidians and cnidarians, hull fouling has been widely assigned as the sole or primary vector causing invasive species occurrences. Hull fouling was considered the primary vector (having transferred most invasives) at Los Angeles/Long Beach and both southern Hemisphere sites in the Davidson et al. study, mainly because high proportions of molluscs and arthropods at these sites were considered ship fouling incursions while at other sites these taxa were vectored by other means.

Based on a literature review of 36 hull-fouling studies, Davidson et al. (2009) reported a total of 1,128 species from 21 phyla or divisions recorded attached to ship’s hulls. Arthropods, including 73 different barnacle species, were most commonly recorded (33 percent of all species), followed by annelids, molluscs and bryozoans. Llansó and Sillett (2008) summarized biofouling surveys of obsolete ships originating in Suisan Bay, California that were part of the U.S. Maritime Administration’s non-retention ship disposal program. The authors reported an extensive fouling community dominated by barnacles, bryozoans, isopod crustaceans, and amphipods in the pre-cleaning surveys.

The inactive Naval ships considered in this opinion are particularly susceptible to hull biofouling given their large hull and niche surface areas, long lay-up times, general lack of hull maintenance, and slow tow speeds. These ships tend to have dense and extensive biofouling assemblages in both hull and non-hull areas that greatly exceed fouling levels of most operational military and commercial ships (Davidson et al. 2008b). All submerged areas of the hull, including appendages and niche areas are subject to the accumulation of biofouling (McClay et al. 2015). Niche areas are problematic because they are difficult to access and clean.

Most hull cleaning systems are designed to clean open, flat or broadly curved hull plates, rather than the partially closed or shielded niche areas (McClay et al. 2015). Cleaning the smooth sides and bottom of a hull through the process of physical scrubbing, heating or water jet blasting is a straightforward process, but niche areas that are not reachable by area cleaning machines require individual human attention and specialized tools (McClay et al. 2015).

Davidson et al. (2009) evaluated the effects of duration-since-dry-docking (i.e., age of antifouling paint) on biofouling accumulation and found a positive relationship between richness of biofouling species and duration since dry-docking. In addition to greater species diversity than active ships, large inactive ships typically have more growth and greater biomass of hull fouling organisms. For example, a recent biofouling survey conducted on the Navy's aircraft carrier the ex-Independence, after the ship had been inactive in Puget Sound for nearly 20 years, documented dense biofouling on all parts of the ship (NUWC 2017b). From the bow of the ship and 300 feet aft the average biofouling growth was about two inches thick with a biomass wet weight ranging from less than 0.1 to 2.0 kilograms per square meter. The biomass across the entire wetted hull surface was estimated to be 54,010 kg (geometric mean of total wet weight) comprised of water (76 percent) calcareous matter, i.e. shells (18 percent) and organic matter (6 percent) (NUWC 2017b). The elevated level of biofouling found on inactive ships is expected to increase the risk of transfers and invasions on a per-ship basis because of higher numbers of propagules per inoculation (Drake and Lodge 2006; Grevstad 1999). From the literature, it is clear that invasion risk is increased by ships with high-density, high-diversity, and repeated transfers of NAS (Davidson et al. 2008a).

Furthermore, slow tow speeds and long in-water residence times at destination ports increase the probability of species successfully recruiting from the ship to the port environment. Based on the Navy's BE (NUWC 2017a), we anticipate that about half of the towed ships will be moved to another Navy shipyard rather than dismantled at the destination port. Ships scheduled for dismantling at the destination port also pose an invasive species risk. Current post-voyage ship dismantling practice involves ships being dragged out of the water iteratively over a period of weeks as the ship is taken apart, thus increasing the risk of invasive species dislodging from the hull and becoming established in the new environment.

10.1.3.2 Potential Effects on ESA-listed Species and Critical Habitat

When non-native plants and animals are introduced into habitats where they do not naturally occur, they can have significant impacts on ecosystems and native fauna and flora. The effects that NAS can have on the existing community of native species in the areas where they become established range from adverse to no effect to beneficial effects (Schlaepfer et al. 2011). Non-native species can reduce native species abundance and distribution, and reduce local biodiversity by out-competing native species for food and habitat. They may also displace food items preferred by native predators, disrupting the natural food web. There are many reported instances in the literature where the establishment of an invasive species has had catastrophic effects on native species and ecosystems, causing high mortality or extinction (Chew 1990; Ciguarria and Elston 1997; Haupt et al. 2010; Josefsson and Andersson 2011; Lleonart et al. 2003; Minchin and Rosenthal 2002; Moncheva and Kamburska 2002; Nehring 2006; Nell 2002; Tsikhon-Lukanina and Reznichenko 1991; Vinogradov et al. 1989; Zeitsev and Ozturk 2001) or fundamentally altering an ecosystem through a single introduction (Castilla et al. 2004; Colwell 1996; Hecky et al. 2004; Wallentinus and Nyberg 2007). Some introductions can be beneficial, providing food for listed sea turtles (Russell and Balazs 2009), increasing biodiversity (Nyberg 2007), and establishing new habitat used by native species (Castilla et al. 2004; Crooks 1998; Daehler and Strong 1996; Grosholz 2002; Paulo da Cunha Lana 1991; Wallentinus and Nyberg 2007).

Uncertainty resulting from the lack of empirical evidence regarding the threat posed by invasive species remains high (Dueñas et al. 2018). As such, it is difficult to predict how particular ESA-listed species would respond to the introduction of a NAS. The impacts of most invasive species are not well known, and for the vast majority no quantitative information is available (Dueñas et al. 2018; Jeschke et al. 2014). In many instances, the impacts from invasive species on native species can be difficult to detect, can only be assessed after long-term studies, and are particularly difficult to identify at the ecosystem level (Parker et al. 1999; Simberloff 2011; Simberloff et al. 2013). Additionally, since most ESA-listed species are impacted by multiple threats, it is often difficult to distinguish invasive species impacts from those caused by other synergistic stressors (Didham et al. 2007; MacDougall and Turkington 2005). The synergistic effects of the invasive species threat with other threats, such as climate change or habitat alteration, can exacerbate the spread and the negative effects of invasive species (Dukes and Mooney 1999; Simberloff 2000).

Studies that have attempted to quantify the role of invasive species in terms of the endangerment or extinction of native species have reported varying results. Wilcove et al. (1998) reported that habitat loss was the greatest threat to imperiled species within the United States (threatening 85 percent of the species classified as imperiled), followed by invasive species (threatening 50 percent of species). By comparison, of the imperiled species on the IUCN Red List with known threats, only six percent list effects from invasive species as contributing to their decline (Gurevitch and Padilla 2004). Gurevitch and Padilla (2004) also note that of the 21 total marine species in the IUCN database listed to have gone extinct (four mammals, 11 birds, one fish, four molluscs, one alga), none are attributed to invasive alien species; most were extinct prior to 1900 and before many modern invasions. Dueñas et al. (2018) conducted a systematic literature review of the available scientific evidence on invasive species' interactions with all endangered and threatened species protected under the ESA. The review found scientific evidence available for 116 endangered or threatened species (8.5 percent of all ESA-listed species), of which 85 species (6.2 percent) were reported as being negatively impacted by invasive species (Dueñas et al. 2018). Of the 85 ESA-listed species negatively impacted by invasive species, seven were marine species: four sea turtle species adversely impacted by terrestrial invasive predators on nesting beaches; and three salmonids adversely impacted by non-indigenous species of trout (Dueñas et al. 2018).

The introduction of hull-fouling ANS may have both direct and indirect effects on ESA-listed sea turtles and fish species. Indirect effects may include changes to benthic habitat and/or changes to invertebrate prey. These effects could result from ANS preying upon, outcompeting, or smothering organisms that may be critical to benthic habitat or food chains. The alteration of a species' habitat or food chain could potentially lead to behavioral disturbance in the form of requiring animals to travel farther or could cause fitness consequences if the animal is unable to feed. In some instances, we would expect NAS introduced via hull fouling to suppress an ecologically similar native species low in the food web (e.g., sessile benthic invertebrate) and the impacts of that invasion have a negligible effect to the ESA-listed species or critical habitats considered in this opinion. For example, Kemp's ridley and loggerhead sea turtles are generalist feeders and it is unlikely additional biofouling species would impact the ability of these species to locate food even if they were to co-occur with biofouling invasion areas. Nonindigenous species can also enhance the available prey options of ESA-listed turtles. For example, Russell and Balazs (2009) found a beneficial effect of invasive algae providing a significant new food source for green sea turtles in Hawaii.

Sturgeon feed on a variety of benthic invertebrates including mollusks, polychaeta worms, crustaceans, gastropods, shrimps, pea crabs, decapods, amphipods, isopods, and small fishes in the marine environment (Collins et al. 2008; Guilbard et al. 2007; Savoy 2007; Savoy and Benway 2004). Many of the prey items of ESA-listed sturgeon likely have an ecologically similar function or niche as the hull fouling species that could potentially be introduced via ship hulls as part of the proposed action. Both positive and negative impacts have been reported in the literature from the introduction of nonindigenous benthic invertebrate species on fish species (Schlaepfer et al. 2011). Vinson and Baker (2008) found that the nonindigenous mudsnail (*Potamopyrgus antipodarum*) was an abundant prey item for native fish in the western United States. However, when native rainbow trout (*oncorhynchus mykiss*) fed exclusively on mudsnails they lost 0.5 percent of their body weight per day. This study suggests that, in some cases, even if nonindigenous invertebrate species can provide a new food source, the resulting effect can still be detrimental to native species if the nonindigenous prey is not as nutritionally valuable as the native prey items that it is replacing.

Invertebrates can have major impacts on the ecosystems they invade. Benthic invertebrates, such as mussels, polychaetes, and hydroids can become dominant filter feeders, greatly reducing the amount of organic energy that is available to native taxa in the water column (NMFS 2012d). This transfer of energy from the water column into the benthos fundamentally alters the ecology of the host habitat, resulting in less prey available for other filter feeders. Adverse effects of this include reduced body condition, growth, survival, and reproduction of native pelagic organisms at the same or similar trophic level as the invader if the native competitor cannot adapt to another food source. These changes would be manifested to a greater or lesser degree up the food chain to higher trophic level organisms in the habitat, including ESA-listed sturgeon and sea turtles (NMFS 2012d). For example. European green crabs (*Carcinus maenas*) have invaded both the U.S. East and West coasts, having trophic scale effects to both environments. In Massachusetts Bay, green crabs prey upon native mussels and oysters, altering community structure (Grosholz 2002; Lafferty and Kuris 1996; Pimentel et al. 2004).

ANS can also alter the physical habitats they invade and modify habitat features that are important to the native fauna. The introduced periwinkle, *Littorina littorea*, ranging along the Atlantic Coast from Canada to the mid-Atlantic, is highly-influential in the sedimentation process; because individuals cumulatively engage in so much grazing, some bottom habitats have become dominated by hard-bottom instead of soft bottom as they formerly were (Bertness 1984; Carlton 1999; Wallentinus and Nyberg 2007). Significant declines in soft-sediment habitats and fringing salt marshes are attributed, at least partially, to the invasion of this species, possibly due to consumption of marsh grasses, such as *S. alterniflora* (Bertness 1984). Species normally adapted to living in soft-bottom systems are gradually replaced by species better adapted for hard-bottom substrates.

Invasive plants can also cause widespread habitat alteration, including native plant displacement, changes in benthic and pelagic animal communities, altered sediment deposition, altered sediment characteristics, and shifts in chemical processes such as nutrient cycling (Grout et al. 1997; Ruiz et al. 1999; Wigand et al. 1997). Introduced seaweeds alter habitat by colonizing previously unvegetated areas, while algae form extensive mats that exclude most native taxa, dramatically reducing habitat complexity and the ecosystem services provided by it (Wallentinus and Nyberg 2007). Invasive algae can alter native habitats through a variety of impacts, including trapping sediment, reducing the number of suspended particles that reach the benthos for benthic suspension and deposit feeders, reducing light availability, and adversely impacting foraging for a variety of animals (Britton-Simmons 2004; Gribsholt and Kristensen 2002; Levi and Francour 2004; Sanchez et al. 2005).

Introduced parasites and diseases can impact native species both directly and indirectly through the food chain which native species rely upon. Parasitism by hull fouling ANS could lead to fitness consequences for sea turtles and biofouling of turtle shells can increase drag, resulting in increased energy expenditure during movement. However, sea turtle shells are often already heavily fouled by native epibionts (Frazier et al. 1985), and we have no information to indicate that nonnative epibionts have resulted in additional impacts from this potential stressor. Pathogens and species with toxic effects can have not only direct effects to ESA-listed species, but also may affect essential critical habitat features or indirectly affect the species through ecosystem-mediated impacts. There are a number of ANS that have the potential to expel toxins at low levels, which can become problematic for other members of the ecosystem if their population grows to very large sizes, resulting in very large amounts of toxins being released. Ship mediated ANS can include organisms that cause harmful algal blooms that can produce toxic compounds, including brevetoxin, saxitoxin and microcystins. The brown alga, *Aureococcus anophagefferens*, causes brown tide when it blooms, causing diebacks of eelgrass habitat due to blooms decreasing light availability and failure of scallops and mussels to recruit (Doblin et al. 2004). Red tide dinoflagellates (*Alexandrium fundyense*) have been introduced via ballast water discharges and have the potential to undergo extreme seasonal population fluctuations. During bloom conditions, high levels of neurotoxins are released into local and regional surface water and air that can cause illness and death in fishes, sea turtles, marine mammals, and invertebrates (as well as their larvae) (Hallegraeff and Bolch 1992; Hallegraeff 1998; Hamer et al. 2001; Hamer et al. 2000; Lilly et al. 2002; McMinn et al. 1997). 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.

10.1.3.3 Exposure Analysis

For our exposure analysis, we assess the risk that ESA-listed species and designated critical habitat would be exposed to the introduction of new NAS as a result of the Navy's inactive ship tow program. First, we consider the risk of exposure to a NAS introduced via ballast water. Next, we discuss the risk of exposure to a hull fouling NAS introduced along the ship tow route while in transit. Finally, we evaluate the risk of exposure to a hull fouling NAS introduced at each of the proposed destination ports.

NAS Introduction Via Ballast Water

Ballast water is required for towing inactive ships for list, trim, and stability purposes (Navy 2018a). When inactive ships are towed for transfer purposes to ports consisting of inactive fleet sites, ballast water will not be removed when the ship arrives at the destination port. Prior to towing the ship in the future, ballast water may be added, transferred between tanks, or removed. If any ballast water is removed, it will be sent offsite for treatment and disposal in accordance with all federal, state, and local laws and regulations (Navy 2018a). For inactive ships being towed for dismantlement, ballast water will be removed as part of the dismantlement process and is under the cognizance of the Navy contractor. As part of the dismantling process, the contractor will pump the ballast water ashore and send it for treatment and disposal in accordance with all federal, state, and local laws and regulations (Navy 2018a). The privately owned and operated tug boat will operate under all U.S. Coast Guard laws and regulations, thus reducing the risk of introducing a NAS through ballast water contained in the tug boat. Therefore, we find it extremely unlikely (to the point of being discountable) that a NAS would be introduced via ballast water as part of the proposed action.

Hull Fouling NAS Introduced Along the Ship Tow Route While in Transit

Due to the slow tow speeds, it is unlikely that hull-fouling species would release, either involuntarily or voluntarily, from the hull while the ship is being towed. If a NAS should release during towing, successful establishment would require (1) settlement on suitable substrate in an area with favorable environmental conditions (i.e., flow, available food, water temperature, and salinity range) in order to survive, and (2) a sufficient number of propagules and environmental conditions suited for reproduction. Hull fouling organisms on inactive ships originate in the relatively shallow, nearshore, estuarine waters of the Navy's proposed origination ports. Environmental conditions at these origination ports are very different from conditions found in the offshore, deepwater habitats that make up the large majority of each tow route. Considering the hydrodynamics and other environmental conditions of the open ocean or major shipping channel environments where hull fouling organisms could potentially release from an inactive ship, successful establishment of a NAS in these environments is extremely unlikely (Floerl and Inglis 2003). In summary, we find it extremely unlikely (to the point of being discountable) that a hull fouling NAS would be introduced in transit along the ship tow route as part of the proposed action.

Hull Fouling NAS Introduced at the Proposed Destination Ports

The proposed action involves the towing of inactive ships to and from many possible combinations of origin and destination ports. Since this is a programmatic consultation, we do not know the details of each particular action such as which ships would be moved to which destination ports, when they would be moved, or what specific biofouling communities may be present on those ships at the time of towing. Even if information on hull-fouling communities were available for each ship, invasive species science is not well developed enough to predict specific nonindigenous species' probabilities of invasion (NMFS 2012d) or how ESA-listed species may respond to a NAS introduction at a particular destination port. Given the limited available information, it is not possible to accurately quantify the exposure of ESA-listed species (i.e., numbers, life stages) to NAS introductions as a result of the proposed action. For this reason, we conduct a more general, qualitative assessment of the relative risk of exposure to a hull fouling NAS introduced as a result of the proposed action at each of the identified destination ports.

Our exposure analysis roadmap is as follows:

1. Evaluate the relative risk of NAS introductions from inactive Navy ships in relation to the risk of NAS introductions to U.S. ports from all ships.

2. Identify and evaluate measureable factors that will affect the risk of a new NAS introduction as a result of towing inactive Navy ships (i.e., towing frequency, hull cleaning, salinity changes, temperature changes, tow duration, and similarity of species composition between ports).
3. Qualitatively assess the risk of a NAS introduction associated with towing an inactive Navy ship for each origination-destination port combination, based on the information from Steps 1 and 2 above.
4. Based on the results of our qualitative risk analysis (Step 3) and the anticipated towing frequencies (from Step 2), assess the cumulative risk of a NAS introduction at each destination port as a result of towing multiple ships over the course of the Navy's inactive ship tow program.
5. Based on the analysis in Step 4, determine which ESA-listed species and/or critical habitats we anticipate would likely be exposed to a NAS introduction as a result of the proposed action.

Step 1: Evaluate the relative risk of NAS introductions from inactive Navy ships in relation to the risk of NAS introductions to U.S. ports from all ships.

Hull fouling is a major vector in the spread of NAS globally due primarily to the large volume of shipping traffic between coastal ports throughout the world. To further evaluate the likelihood that a NAS would be introduced as a result of the proposed action, we need to consider the anticipated number of inactive Navy ships towed (i.e., nine per year on average) in relation to the total volume of shipping traffic, and the estimated rate of hull fouling NAS introductions in U.S. ports. It is important to note that our discussion of the proposed action in relation to all other ship traffic is not intended to minimize the effects of the proposed action by comparing the relative magnitude of effect. Rather we compare the number of inactive Navy ships towed annually to the total volume of ship traffic, along with information on the annual rate of NAS introductions, as a means to evaluate the risk of a NAS introduction from the proposed action. Analyzing risk in this manner supports our ability to predict the likelihood of the Navy's proposed action to cause an effect to ESA-listed species and designated critical habitat.

Based on information obtained from the National Exotic Marine and Estuarine Species Information System (NEMESIS) database (Fofonoff et al. 2018), NMFS (2012c) estimated the number of documented hull fouling NAS that are introduced in U.S. ports annually. In querying the database, they limited their search to only include hull-fouling species documented in the years 2008 through 2011. Based on these search criteria, NMFS (2012c) estimated that the annual number of newly documented hull-fouling NAS in waters of the United States ranged from 10 to 20 species with a mean of 13.5 species per year. It should be noted that the NEMESIS database does not include all NAS introductions in the United States; rather, it represents only those introductions that have been documented. The number documented likely underestimates the total number of introductions since many waters of the United States lack invasive species monitoring programs. Alternatively, this approach may somewhat overestimate the number of hull fouling species introduced by ship hulls since hull fouling species can also be introduced by other vectors such as ballast water or aquaculture. Despite these caveats, the NMFS (2012c) estimate (i.e., 10 to 20) is likely a reasonable approximation of the number of hull fouling NAS introduced by ship hulls in U.S. ports each year.

Large commercial ships (those displacing greater than 300 metric tons) arrive over 90,000 times per year in hundreds of different ports in U.S. coastal waters, including the Great Lakes (Miller et al. 2007). Approximately half of these arrivals are from ports of call outside the United States and Canada; the remainder arrive directly from another port within North America. Added to these commercial ships is the movement of a large fleet of active U.S. military ships between multiple domestic ports, and the movement of a small fleet of inactive MARAD obsolete ships, primarily to Brownsville, Texas for dismantling.

If the risk of a hull fouling NAS introduction were roughly equivalent for all 90,000 large "active" ships arriving in U.S. ports annually, given the relatively small number of NAS introductions estimated per year (approximately 10 to 20), it would be highly unlikely that a hull fouling NAS would be introduced by any of the nine (on average) inactive Navy ships towed per year. However, in general, we would expect the risk of a hull fouling NAS introduction to be greater for ships that have been inactive for long periods of time compared to active ships. The inactive Navy ships considered in this opinion are particularly susceptible to hull biofouling given their long periods of inactivity at origination ports, lack of hull maintenance, slow tow speeds, and (for those not being dismantled) long stays at destination ports. Therefore, based on the available information, we anticipate that, on average, in the absence of hull cleaning mitigation the risk of a hull-fouling NAS introduction from an inactive Navy ship is substantially greater than from an active ship.

In the absence of hull cleaning mitigation or large differences in salinity between origination and destination ports, inactive Navy ships would likely have a much greater risk of a NAS introduction compared to the average commercial ship or active military ship. From 2007 to 2009, Sylvester et al. (2011) surveyed the hulls of 40 large commercial ships (including bulk, container, general cargo carriers, oil and chemical tankers, roll-on/roll-off cargo ships, and one cable layer) in the Canadian ports of Halifax (east coast) and Vancouver (west coast). The average length of these ships was 646 feet, the average sailing speed was 16 knots, and the average time since these ships were last painted was a little under two years. Biosurveys of the hulls showed the average number of species per ship was 33 and the average propagule pressure (i.e. individual organisms per ship) was around 2,860. Thus, even the hulls of active ships that have been painted within the past few years and that move at relatively fast speeds (compared to towed ships) can carry a diverse biofouling community with several thousand organisms.

Sylvester et al. (2011) found that propagule pressure was highly variable among ships, and increased with time spent in previous ports-of-call and time since last application of antifouling paint. Sylvester et al. (2011) reported high propagule pressures (as high as 600,000 individuals per ship) for commercial ships with particularly long stays in a given port and long periods between hull paintings. For large Navy ships that have been inactive at a given port for many years without hull husbandry, we would expect propagule pressures to be more similar to commercial ships on the higher end of the study by Sylvester et al. (2011) (i.e., hundreds of thousands) than to the average propagule pressure reported in this study of around 3,000 individual organisms.

Biofouling surveys conducted on inactive Navy and MARAD ships before and after in-water hull cleaning have shown this mitigation measure can be very effective at reducing the risk of NAS introductions (Davidson et al. 2008b; Llansó and Sillett 2008; NUWC 2017b). In particular, these studies show that hull cleaning can significantly increase the amount of bare space found on the hull, greatly reduce the overall biomass, and remove most of the three-dimensional biofouling community. Based on pre and post hull cleaning biosurveys of the ex-Independence and Orion, we would expect hull cleaning to reduce the propagule pressure (number of individual organisms) on inactive Navy ships by several orders of magnitude. We expect the differential risk between inactive Navy ships to be largely commensurate with active commercial ships, on average, in scenarios where hull cleaning mitigation is conducted. Although the risk of a NAS introduction would not be completely eliminated, with hull cleaning mitigation (either in-water or dry-docking) the risk from inactive Navy ships should be reduced to a level more similar, on average, to the tens of thousands of active commercial ships that regularly move between U.S. coastal ports each year. Based our discussion above, considering the very large number of ships and the relatively small number of hull fouling NAS introductions in U.S. ports each year, the risk of a NAS introduction for a ship that has undergone hull cleaning mitigation should be extremely small.

Other studies of inactive ship tows, where hull cleaning was not conducted, have shown that exposing hull fouling organisms to an extreme change in salinity (i.e., freshwater to saltwater or visa-versa) can also greatly reduce the risk of invasive species introductions by reducing the likelihood that species can survive and become established in their new environment (Brock et al. 1999; Davidson et al. 2008a). Whereas Navy ships that are not hull cleaned would likely have a rich, diverse, and large biofouling community prior to departure, very few (if any) of these organisms would be expected to survive a major, prolonged change in salinity (either along the tow route or at the destination port), and fewer still would be expected to reproduce and become established in their new environment. Considering the very large number of ships and the relatively small number of hull fouling NAS introductions in U.S. ports each year, the risk of a NAS introduction from ships with hull fouling organisms subjected to a major and prolonged salinity change should also be relatively small.

The relative risk of a NAS introduction from inactive Navy ships for each possible ship tow route is discussed in more detail in Step 2 below.

Step 2: Identify and evaluate measureable factors that will affect the risk of a new NAS introduction as a result of towing inactive Navy ships (i.e., towing frequency, hull cleaning, salinity changes, temperature changes, tow duration, and similarity of species composition between ports).

The process by which a species becomes a biological invader, at a location where it does not naturally occur, can be divided into a series of sequential stages (transport, introduction, establishment and spread) (Blackburn et al. 2015). A species' success at passing through each of these stages depends, in a large part, on the propagule pressure available to assist making each transition (Lockwood et al. 2005). Propagule pressure is defined as the quantity, quality, and frequency with which propagules (individual organisms or their larvae, eggs, spores, etc.) are introduced to a given region to which they are not native (NRC 2011). Propagule pressure affects transport and introduction by moderating the likelihood that abundant (and widespread) species are deliberately or accidentally translocated (Blackburn et al. 2015). Propagule pressure affects establishment success by moderating the stochastic processes (demographic, environmental, genetic or Allee) to which small, introduced populations will be vulnerable (Blackburn et al. 2015). Based on a thorough review of the invasive species literature, Lockwood et al. (2005) found evidence for a positive effect of propagule pressure on nonindigenous population establishment. They conclude that propagule pressure (both size and number) is a key element to understanding why some introduced populations fail to establish whereas others succeed. Propagule pressure may also affect invasive spread because of persistent genetic effects determined by the numbers of individuals involved in the establishment phase (Blackburn et al. 2015). Propagule pressure incorporates estimates of the absolute number of individuals involved in any one release event (propagule size) and the number of discrete release events (propagule number). For the proposed action, this means that the likelihood of a NAS becoming established in a particular port will be positively related to both the degree of hull fouling on inactive vessels towed to that port and to the towing frequency of inactive vessels to that port. By significantly reducing propagule pressure, hull cleaning mitigation reduces the likelihood that any species that survives both the hull cleaning process and towing would be able to successfully reproduce and become established in their new environment.

The risk of ESA-listed species exposure to NAS introduced by a towed inactive Navy ship will depend on numerous factors including: propagule pressure or level of biofouling (e.g., biomass, species diversity, structural complexity); salinity and/or temperature differences between origination and destination ports; salinity and/or temperature differences between the ports and the tow route; ship tow duration; and the current level of invasive species establishment at the destination port (i.e., likelihood that the NAS is not already established). While there are other factors that could affect the likelihood of a NAS introduction (e.g., presence of potential predators, pathogens and/or parasites; availability of suitable habitat and/or substrate; biofouling species dispersal mode), our analysis focuses on those factors that we can measure and evaluate within the context of this opinion. As such, our evaluation of the relative risk of a NAS introduction resulting from a towed inactive Navy ship is based on the following six factors:

1. Tow Frequency: How many inactive ships will be towed between each origination/destination port combination over a given time frame?
2. Mitigation: Will hull-cleaning (either in-water or dry-docking) be conducted on the inactive ship prior to departure? Alternatively, has the ship's hull been painted within the previous three years?
3. Salinity Changes: Will the hull-fouling species experience major changes in salinity due to difference between origination port and destination port, or one of the port and sections of the tow route?
4. Temperature Changes: Will the hull-fouling species experience major changes in water temperature due to difference between origination port and destination port, or one of the ports and sections of the tow route?
5. Tow Duration: How long will it take to tow the inactive ship from the origination port to the destination port?
6. Similarity of Species Composition between Ports: Based on geographic distance between origination and destination ports, how similar do we expect the species composition at the two ports to be (i.e., what is the likelihood that hull-fouling species will already be established, either as a native species or NAS, at the destination port)?

<u>Tow Frequency</u>

As discussed above, the likelihood of a NAS becoming established in a particular port will be positively related to both the degree of hull fouling on inactive vessels towed to that port and to the towing frequency of inactive vessels to that port. With all other factors equal, we would expect propagule pressure and the associated risk of an invasive species introduction to increase with increasing towing frequency.

In their BE, the Navy indicated that, on average, nine inactive ships would be towed each year. On June 25, 2018 the Navy provided NMFS with a supplemental table indicating the anticipated average number of ship tows for each origination/destination port combination over a five-year period (Table 29). We base our effects analysis on the anticipated ship tow frequencies estimated for each scenario as shown in Table 29. Based on supplemental information provided by the

Navy (after submission of the BE), an entry of "0" in Table 29 indicates that the towing of an inactive ship between these ports would be a very rare event, and would likely not exceed one ship every 25 years, on average.

Table 29. The anticipated average number of ship tows for each origination/destination port combination over a five-year period.

Origin Port	Destination Port						
	Baltimore	Beaumont	Brownsville	New Orleans	Pearl Harbor	Philadelphia	Puget Sound
Beaumont[1]	0		1	0	0	0	0
Mayport	0	0	0	0	0	1	0
Norfolk	0	1	0	0	0	8	0
San Diego	0	0	0	0	7	0	2
Pearl Harbor	0	0	4	1		0	0
Philadelphia	1	0	9	5	0		0
Puget Sound	0	0	4	1	0	0	

Note: Grey shading indicates that any ship towed to one of these three ports will be dismantled. A "0" indicates that the towing of an inactive ship between these ports would be a very rare event, and would likely not exceed once every 25 years, on average.

<u>Mitigation</u>

Hull husbandry includes the removal of biofouling organisms during dry dock periods, as well in-water hull cleaning. Ships may also be kept in dry-dock for long enough periods to result in the desiccation of hull fouling organisms. Hull cleaning has been shown to be an effective mitigation measure for minimizing the risk of NAS by appreciably reducing the number, biomass, and diversity of hull fouling organisms. Pre-tow hull cleaning should have the added benefit of reducing the number of organisms that can attach to the hull while the ship is in transit (i.e., fewer surface areas for attaching, cementing, burrowing, embedding etc.), further reducing the potential for species transfer to result from the proposed action (Llansó and Sillett 2008).

Diver surveys conducted after in-water biofouling removal was conducted on the ex-Independence reported that 99 percent of the hull was free from biofouling organisms post-cleaning (NUWC 2017b). Davidson et al. (2008b) reported a significant reduction in organism cover on the underwater surfaces (i.e., propeller, rudder, propeller shafts, struts, transverse hull transects and sea chests) of the Orion, a MARAD ship that was inactive for 13 years, post-cleaning using the self-propelled, diver driven SCAMP® cleaning machine. The mean percent area of exposed hull surface increased from 10.9 percent per sample prior to scrubbing to 62.7 percent after scrubbing. A large increase in bare space (a combination of 'hull surface' and 'organism scars' categories) was consistent across all depths of the hull and on the surfaces of the ship's running gear. This increase in bare space coincided clearly with reductions in encrusting species, barnacles, and filamentous biofouling (Davidson et al. 2008b). These studies, comparing biofouling levels pre and post hull-cleaning, indicate that this mitigation method is highly effective at reducing propagule pressure, and therefore, the likelihood of an NAS being introduced or becoming established. Despite the large reported decreases in hull area covered and propagule pressure, Davidson et al. (2008b) found that the number of species on the Orion decreased only slightly (from 37 to 30) between initial (pre-cleaning) and final (post-cleaning) biosurveys. Based on our review of the literature, we calculated an average of 48 species (pre hull cleaning) on the hulls of inactive Navy and MARAD ships ranging in size from 337 to 887 feet in length (Brock et al. 1999; Davidson et al. 2008a; Davidson et al. 2008b). After hull cleaning, the average species count would be expected to be somewhat lower and closer to the average of 33 species for commercial ships hulls as reported by Sylvester et al. (2011). Thus, in-water hull cleaning of inactive ships can significantly reduce but not entirely eliminate invasive species risk, as viable specimens from many species may still be found post-cleaning.

An alternative form of mitigation to minimize invasive species risk from towing inactive ships is hull painting. As noted above, we expect that, on average, the risk of a hull-fouling NAS introduction from an inactive Navy ship is substantially greater than from an active commercial ship. This is largely due to the fact that antifouling hull paint on inactive ships is typically much older than antifouling hull paint on active ships, and therefore less effective at protecting the hull from biofouling organisms. While this is likely the case for most of the inactive fleet, there could be some inactive ships whose antifouling hull paint is more similar in age to that of active ships. For example, Navy ships that have only recently been inactivated may have been painted within the past few years while they were still active. We would expect that the risk of an invasive species introduction from an inactive ship that has been recently painted (i.e., within three years) to be substantially smaller than a typical inactive ship, and more similar to that of a typical active commercial ship.

<u>Changes in Salinity and Water Temperature</u>

Depending on the tow route, hull fouling organisms on inactive Navy ships may be exposed to considerable changes in environmental conditions (including physical, chemical, and biological) during transit. Temperature and salinity are often considered the primary physical factors determining survival of hull fouling communities during voyages through a range of oceanographic conditions. We would expect a greater risk of introducing species between regions with similar environmental conditions than between regions with large differences in environmental conditions, particularly when considering salinity levels and temperature ranges.

Changes in salinity can impact the ability of hull-fouling organisms to survive and reproduce successfully in a new environment. High salinity is lethal to freshwater plants and animals because the cells of the organism have either a lack of water or an excess of ions (or have both) that can result in a range of toxic effects (Hart et al. 1991). Experimental and observational studies have shown that salinity tolerance varies among NAS; some NAS cannot tolerate large salinity variation in their new habitats, while others can (Choi et al. 2016).

Washing ships and associated gear with saltwater is a standard and effective method to prevent the spread of freshwater invasive species (Matheson et al. 2007). Most freshwater species are not expected to survive the long duration (i.e., weeks to months) tows through saltwater. In their analysis of transoceanic shipping vectors to the Great Lakes, Colautti et al. (2003) conclude that it is unlikely that hull fouling has contributed substantially to the Great Lakes invasive species biota since taxa would have to survive transit through highly saline oceanic water prior to establishing in the Great Lakes. Likewise, most hull fouling species potentially transferred by ships towed from saltwater origination ports to any freshwater destination port are not expected to survive and reproduce successfully at the freshwater destination port (Foster et al. 2016). In a study from the United Kingdom, Foster et al. (2016) found a negative relationship between marinas' proximity to a freshwater source and the number of NAS established at the marina. Hypo-saline and hyper-saline hull treatments have shown success as an environmentally safe and cost-effective mitigation method for minimizing the risk of invasive species introductions (Brock et al. 1999; Jute and Dunphy 2017). Most hull fouling species would not be expected to survive such treatments, although a few rare exceptions have been noted in the literature (e.g., the bivalve *Mya arenaria* has been observed within 0 to 32 ppt (Powers et al. 2006)).

From the available literature, very few species can tolerate major salinity changes from either: a) very low salinity (i.e., complete freshwater to less than three ppt) to euhaline conditions (i.e., 30 ppt or greater) or visa-versa, or b) from complete freshwater to polyhaline conditions (i.e., 18-30 ppt) or visa-versa. However, there are several known species that can tolerate a wide range of salinities in between these extremes. The transfer of biofouling species between ports with salinities in the mesohaline range (i.e., 5 to 18 ppt), polyhaline range (i.e., 18 to 30 ppt) or euhaline range (i.e., 30 to 35 ppt) can lead to the introduction of a NAS. Some examples (from the Navy BE) of NAS with wide salinity tolerance ranges that are already established in origination ports considered in this opinion include: sea vase (*Ciona intestinalis*), 5 to 40 ppt; sea grape tunicate (*Molgula manhattensis*), 10 to 35 ppt; green crab, 4 to 54 ppt; green mussel (*Perna viridis*), 18 to 33 ppt; and Japanese wireweed (*Sargassum muticum*), 7 to 34 ppt (NUWC 2017a).

The duration of the treatment (i.e. freshwater or saltwater immersion time) is also a factor that could affect the ability of species to survive changes in salinity. Initially, mussels and other hull fouling species may react to salinity shock by temporarily closing their valves, suspending ventilation and feeding (van der Gaag et al. 2016). However, this cannot be maintained for long periods and adaptation to higher salinity must eventually occur or the organism will not survive. In an experimental study of mussels from the North Sea, van der Gaag et al. (2016) found the animals could survive salinity shocks within the favorable salinity range, but that most individuals died within a few weeks of exposure to the unfavorable salinity range. For the proposed action, the Navy anticipates that 90 percent of vessels towed between ocean basins (i.e., Pacific to Atlantic or visa-versa) would traverse through the Panama Canal. As such, hull fouling organisms on these vessels would be towed through freshwater for anywhere from about eight hours to two days, depending on the travel time through the canal. While this freshwater immersion would likely reduce the survival of some saltwater hull fouling organisms, we also anticipate that some other saltwater species would be able to survive this relatively short duration immersion through the freshwater canal.

Hull fouling invertebrate species (e.g., mussels, barnacles, bryozoans etc.) have variable body temperatures and are considered ectothermic organisms since they gain heat from the external environment. Because physiological rates are temperature sensitive, an ectotherm's behavioral and ecological performance — even its fitness — can be influenced by body temperature (Huey and Kingsolver 1989). Thus, thermal sensitivity is an important factor in determining whether a species can adapt to changes in water temperature. Ship tow scenarios that involve a large temperature shift from origination to destination port, or large temperature changes along the tow route, will likely result in a lower survival rate of hull fouling species, and therefore a lower risk of NAS introduction compared to scenarios where temperature fluctuation are minor to moderate. Yuan et al. (2016) found that changes in temperature had a more rapid effect on the survival of two bivalve species than did changes in salinity. As discussed above, while some hull fouling species may be able to shut down for up to a few weeks to avoid salinity shock, temperature effects may be more sudden as organisms may be less adept at adjusting to such changes. Yuan et al. (2016) suggest that changes in both temperature and salinity can work synergistically to reduce the likelihood of NAS survival and establishment.

While temperature changes could affect the survival of some hull fouling species, many species, particularly invasives, are known to occur over a wide range of water temperatures. Some examples (from the Navy BE) of NAS with relatively wide temperature tolerance ranges that are already established in origination ports considered in this opinion include the sea vase (*Ciona intestinalis*), 37 to 86 degrees Fahrenheit and the white crust tunicate (*Didemnum vexillum*), 28 to 75 degrees Fahrenheit (NUWC 2017a).

Brock et al. (1999) reported on the effects of salinity and temperature changes on the biofouling community attached to the hull of the U.S.S. Missouri. After sitting inactive for five years, the Missouri was moved from Bremerton, Puget Sound (salinity approximately 24 to 30 ppt; water temperature 48 to 55 degrees Fahrenheit) up into the brackish Columbia River (salinity approximately 2 to 10 ppt) for nine days, before being towed to Pearl Harbor, Hawaii (salinity approximately 30 ppt; water temperature 74 to 82 degrees Fahrenheit). A total of 116 different biofouling taxa were identified initially when the ship was in Bremerton. Only 11 species were initially alive when the ship reached Pearl Harbor, five of which were either recruited in transit or were already found in Hawaii. A biosurvey conducted 83 days after arrival in Pearl Harbor did not find any living organisms. Brock et al. (1999) concluded that short-term freshwater exposure (i.e., nine days) coupled with a large increase in water temperature appear to be very effective at mitigating the risk of invasion from a temperate, marine biofouling community. In a follow-up study, Apte et al. (2000) reported that one NAS mussel (*Mytilus galloprovincialis*) that was attached to the hull of the Missouri spawned successfully shortly after the ship arrived in Pearl Harbor. However, based on subsequent assessments of the nonnative marine biota of the main Hawaiian islands, it does not appear as if this species became permanently established in Hawaii (Carlton and Eldredge 2015).

Davidson et al. (2008a) quantified the biofouling communities of two inactive MARAD ships (Florence and Point Loma), one stationary for one decade and the other for two decades, before and after their final transit from California to Texas. During the 43-day voyage, organisms encountered salinity variation that ranged between zero (Panama Canal) and at least 37 ppt (Brownsville, Texas) and temperatures that varied between 49.8 degrees Fahrenheit and 88.9 degrees Fahrenheit. Pre-departure biofouling surveys across both ships detected 22 species of macroinvertebrates (meiofauna, protists, and other microorganisms were not surveyed in this study). A decrease in biofouling extent and reduction in the complex, three-dimensional matrix of fouling was found across both ships upon arrival in Texas. A large increase in species richness (from 22 to 57 species) was found for the two ships combined as 48 new species recorded in Brownsville had presumably attached during transit or after the ship reached the destination port (Davidson et al. 2008a). Only nine of the 22 initial species from Suisun Bay, California were recorded from the two ships in Brownsville; four of these were already established in Texas. Of the five species that were not already established in Texas, only three were alive upon arrival in Brownsville. Similar to the Brock et al. findings, this study showed that the combination of temperature and salinity changes can effectively reduce the risk of NAS introductions by reducing the number, biomass and diversity of hull fouling organisms. Although the total number of species increased in this study due to species that attached in transit, these open water species are likely less adapted to survive and become established in the marine estuarine or freshwater locations of the destination ports for the proposed action.

<u>Tow Duration</u>

In addition to salinity and water temperature, tow time is another variable that can affect the survival of hull fouling species. Hull fouling species that become established in estuarine and freshwater locations of the destination ports are less adapted to survive in the open ocean environment of the transit route. Although we did not find any studies that specifically address the effects of tow time on a biofouling community, it is reasonable to assume that longer tow times would result in a lower number, biomass and diversity of hull fouling organisms at the destination port (assuming all other factors are constant).

Similarity of Species Composition between Ports

Although biofouling organisms are more likely to survive trips with shorter tow durations, origination and destination ports that are relatively close to one another, and that have similar salinities, would be expected to have similar species (both native and nonnative) compositions. Thus, the risk of a new NAS (i.e., one that is not already present at a destination port) becoming established is likely very small if the origination port is relatively close to the destination port. Even for more distant ports, it is probable that at least some of the biofouling organisms that could be transported from origination ports are already present at destination ports given the historical and ongoing movement of ships between these ports. For example, Brock et al. (1999) reported that out of 76 biofouling species identified on the hull of the U.S.S. Missouri which originated in Puget Sound, 11 (14.4 percent) were already known to occur in Hawaiian waters.

Step 3: Qualitatively assess the risk of a new NAS introduction as a result of towing an inactive Navy ship for each origination-destination port combination, based on the information from Steps 1 and 2 above.

Using the information from Steps 1-2 above, next we assess the risk of a new NAS introduction as a result of towing an inactive Navy ship for each origination-destination port combination. In this step we assess the risk from towing one ship from origination to destination port. In Step 4 (below), we evaluate the cumulative risk, over the course of the inactive ship tow program, of a new NAS introduction for each destination port by combining the risk assessment for a single ship tow with the anticipated towing frequencies for the various origination-destination port scenarios.

As discussed above, for ship tow scenarios that involve hull cleaning mitigation we anticipate the risk of exposure to a NAS introduction from an inactive Navy ship would be reduced to a level of risk comparable, on average, to large commercial ships. For any given ship, this risk is extremely small considering that tens of thousands of large commercial ships arrive in U.S. ports each year (Miller et al. 2007), and only approximately 10-20 hull fouling NAS are introduced in U.S. ports each year (NMFS 2012c). Thus, for all ship tow scenarios that involve hull cleaning mitigation, we find there is an *extremely low* risk of a newly introduced NAS as a result of the proposed action.

This risk assessment is predicated on the assumption that towing of the inactive ship would commence within a relatively short time period after hull cleaning has been completed. The effectiveness of hull cleaning as a mitigation measure decreases the longer the inactive ships sits in the origination port after hull cleaning, as new hull fouling organisms may attach prior to departure. For all ship tow scenarios involving hull cleaning mitigation the Navy contractor would tow the ship as soon as practicable upon completion of hull cleaning (R. Johnson, Navy, pers. comm. to R. Salz, NMFS, October 17, 2018). For purposes of our analysis, for all scenarios involving hull cleaning mitigation we assume that once hull cleaning has been completed the inactive ship would not be in the water at the origination port for more than four weeks prior to departure.

Similarly, for ship tow scenarios that involve a major salinity change, either from origination port to destination port or for some prolonged portion of the tow route, based on the available information we find there is an *extremely low* risk of a newly introduced NAS as a result of the proposed action. *Major salinity change* is defined as either (1) from a very low salinity (i.e., between one and four ppt, on average) to a euhaline environment (i.e., 30 ppt or greater, on average) or visa-versa <u>or</u> (2) from freshwater to either a polyhaline (i.e., from 18-30 ppt, on average) or a euhaline environment, or visa-versa;

For all other ship tow scenarios (i.e. those that do not involve <u>either</u> hull cleaning mitigation <u>or</u> a major salinity change as defined above) we consider a combination of several factors in determining the likelihood of a NAS introduction. These include moderate or large salinity changes from origination port to destination port, temporary or short-term salinity changes during towing, and temperature changes from origination port to destination port or during towing, tow duration, and anticipated similarity of species composition between origination port and destination port. For example, since fewer species would be expected to survive a moderate to large change in salinity, ship tow scenarios that involve a moderate to large (but not major) change in salinity would be expected to have a lower likelihood of a NAS introduction than scenarios where the salinity change is minor. Similarly, large temperature changes and long tow durations would be expected to reduce the survivability of hull fouling species, thus decreasing the likelihood of a NAS introduction.

For ports that are in relatively close proximity to one another, short tow durations and similar climates should increase the likelihood that species will survive towing. However, such ports would also be expected to have similar species compositions, thus decreasing the likelihood of a new NAS introduction (i.e., one that is not already established at the destination port). To evaluate anticipated similarity of species composition between ports we use a three item scale as follows: *very similar*, *somewhat similar*, or *not similar*. For purposes of our analysis, we define similarity of species between ports as follows:

- *Very similar* species composition: ports that are connected to the same major water body (e.g., Gulf of Mexico), have similar latitudes (i.e., within one or two degrees), have similar water temperatures (i.e., within ten degrees Fahrenheit, on average), and are in relatively close proximity to each other (i.e., ship tow durations of one week or less).
- *Similar* species composition: ports that have similar latitudes (i.e., within one or two degrees Fahrenheit), have similar water temperatures (i.e., within ten degrees Fahrenheit), and are in relatively close proximity to each other (i.e., ship tow durations of one week or less).
- *Not similar* species composition: all other port to port comparisons.

We use the following four item categorical scale to qualitatively assess the risk of a NAS introduction from towing an inactive Navy ship for each ship tow scenario:

1. *extremely low risk*: ship tow scenarios that involve one of the following:
   a. Hull cleaning mitigation;
   b. Hull painted with antifouling paint within three years prior to towing;
   c. Major salinity change between origination and destination ports (see above for definition of *major salinity change*);
   d. Major change in salinity between one (or both) of the ports involved in the tow and the salinity along a stretch of the tow route lasting for a minimum of two weeks, on average.

2. *very low risk*: ship tow scenarios that involve one of the following:
   a. Very similar species composition between origination and destination ports;
   b. Very large salinity difference (i.e., greater than 20 ppt, on average) between origination and destination ports and very large temperature difference (i.e., greater than 25 degrees Fahrenheit, on average) between origination and destination ports.
   c. Major difference in salinity between one (or both) of the ports involved in the tow and the salinity along a stretch of the tow route lasting for a minimum of one week, on average.

3. *low risk*: ship tow scenarios that involve one of the following:

   a. Moderate port-to-port salinity change (i.e., between 10 and 15 ppt, on average) and *somewhat similar* species composition between ports;
   b. Large port-to-port salinity change (i.e., between 15 and 20 ppt, on average).

Salinity comparisons between ports were made using the salinity range mid-points for each port. Water temperature comparisons were made using the average low (winter) temperatures and the average high (summer) temperatures found at each port. For salinity and temperature ranges and more specific information about each port refer to Section 9.9 *Characterization of Origination and Destination Ports* above. Changes in salinity and temperature along the tow route were based on supplemental information provided by the Navy (Navy 2018e) as shown in the tables in Appendix 17.

For purposes of this analysis, based on information provided by the Navy, we assume that about 90 percent of ships towed from the Pacific to the Atlantic/Gulf of Mexico or visa-versa would go through the Panama Canal. Ships towed from saltwater ports through the freshwater Panama Canal would likely have a lower risk of a NAS introduction compared to ships that are not immersed in freshwater along the transit route. We assume tow routes through the Panama Canal would expose hull fouling organisms to a freshwater immersion lasting for between eight hours and two days. According to the Navy, only very large ships (e.g., aircraft carriers) that cannot go through the Panama Canal would travel between ocean basins through the Strait of Magellan.

NAS Introduction Risk for Ships Towed to Baltimore, Maryland

Since the salinity in the port of Baltimore is brackish and ESA-listed species are present at this port, the Navy has agreed to conduct hull cleaning mitigation (either in-water cleaning or dry-docking) for all inactive ships towed to Baltimore. The only exception would be for inactive ships that have had their hulls painted within the past three years. Therefore, we determine that the NAS introduction risk from inactive Navy ships towed to Baltimore from any of the proposed origination ports is *extremely low*.

NAS Introduction Risk for Ships Towed to Brownsville, Texas

Since Brownsville is a saltwater port and ESA-listed species are present at this port, the Navy has agreed to conduct hull cleaning mitigation (either in-water cleaning or dry-docking) for all inactive ships towed to Brownsville from any saltwater or brackish origination port. The only exception would be for inactive ships that have had their hulls painted within the past three years. Therefore, we find that the NAS introduction risk from inactive Navy ships towed to Brownsville from a ship originating in Mayport, Norfolk, Pearl Harbor, Puget Sound or San Diego is *extremely low*. We also find that the NAS introduction risk from inactive Navy ships towed to the saltwater port of Brownsville from the freshwater port of Philadelphia is *extremely low* due to the major salinity difference between these ports. Hull fouling species on inactive ships in the very low salinity (range from about 1 to 4 ppt), oligohaline port of Beaumont would not be expected to survive in the euhaline port of Brownsville with salinities ranging from 34 to 37 ppt. As such, we find that the NAS introduction risk from inactive Navy ships towed from Beaumont to Brownsville is *extremely low* due to the major salinity difference between these ports.

NAS Introduction Risk for Ships Towed to Pearl Harbor, Hawaii

Since there are ESA-listed species present at the saltwater port of Pearl Harbor, the Navy has agreed to conduct hull cleaning mitigation for inactive ships towed to this port from any saltwater or brackish origination port. The only exception would be for inactive ships that have had their hulls painted within the past three years. Therefore, we find that the NAS introduction risk from inactive Navy ships towed to Pearl Harbor from a ship originating in Mayport, Norfolk, San Diego or Puget Sound is *extremely low*. We also find the NAS introduction risk from inactive Navy ships towed to the saltwater port of Pearl Harbor from the freshwater port of Philadelphia is *extremely low* due to the major salinity difference between these ports. Hull fouling species on inactive ships in the very low salinity (range from about 1 to 4 ppt), oligohaline port of Beaumont would not be expected to survive in the euhaline port of Pearl Harbor with an average salinity around 34 ppt. As such, we find that the NAS introduction risk from inactive Navy ships towed from Beaumont to Pearl Harbor is *extremely low* due to the major salinity difference between these ports.

NAS Introduction Risk for Ships Towed to Puget Sound, Washington

Since there are ESA-listed species present at the saltwater port of Bremerton (Puget Sound), the Navy has agreed to conduct hull cleaning mitigation for inactive ships bound for Puget Sound from any saltwater or brackish origination port. The only exception would be for inactive ships that have had their hulls painted within the past three years. Therefore, we find that the NAS introduction risk from inactive Navy ships towed to Puget Sound from a ship originating in Mayport, Norfolk, San Diego or Pearl Harbor is *extremely low*. We also find that the NAS introduction risk from inactive Navy ships towed to the saltwater port of Puget Sound from the freshwater port of Philadelphia is *extremely low* due to the major salinity difference between these ports.

Most hull fouling species on inactive ships located in the very low salinity (range from about 1 to 4 ppt), oligohaline port of Beaumont would not be expected to survive in the polyhaline port of Bremerton, Puget Sound with salinities ranging from around 24 to 30 ppt. In addition to the very large differences in salinity from port to port, hull fouling organisms from Beaumont would be subjected to a major salinity change while being towed through saltwater during transit for several weeks to month, depending on the tow route (see Appendix 17). Based on the anticipated salinity differences (both between ports and along most of the long duration tow route), we find that the NAS introduction risk from inactive Navy ships towed from Beaumont to Puget Sound is *extremely low*.

NAS Introduction Risk for Ships Towed to Philadelphia, Pennsylvania

Since there are ESA-listed species present at the freshwater port of Philadelphia, the Navy has agreed to conduct hull cleaning mitigation for inactive ships towed to Philadelphia from any freshwater or brackish origination port. The only exception would be for inactive ships that have had their hulls painted within the past three years. As such, we find that the NAS introduction risk from inactive Navy ships towed from Beaumont or Mayport to Philadelphia is *extremely low*. Hull fouling species from high salinity (i.e., polyhaline or euhaline) origination ports would not be expected to survive in the freshwater port of Philadelphia. Therefore, we find that the NAS introduction risk from inactive Navy ships towed from Pearl Harbor, Puget Sound, San Diego or Norfolk to Philadelphia is also *extremely low*.

NAS Introduction Risk for Ships Towed to Beaumont, Texas

Since ESA-listed species are not present at the port of Beaumont, the Navy action would not include hull cleaning mitigation (either in-water cleaning or dry-docking) for any inactive ships towed to Beaumont, regardless of the origination port. As discussed previously, for our analysis we consider Beaumont an oligohaline port with average salinities ranging from around one to four ppt (Academy of Natural Sciences 2006; LA Coast 2018; NOAA 2018b; NOAA 2018c) .

We find that the NAS introduction risk from inactive Navy ships towed to the oligohaline port of Beaumont from the saltwater ports of either San Diego or Pearl Harbor is *extremely low* due to the major salinity difference between these ports.

Hull fouling species originating in the polyhaline port (24 to 30 ppt) of Bremerton, Puget Sound would need to tolerate a wide range of salinities during the long duration ship tow to Beaumont. All inactive ships would be towed through euhaline waters (around 34 to 36 ppt) for several weeks to months before reaching the very low salinity destination port of Beaumont with average salinities around one to four ppt. Except for very large inactive ships (e.g., aircraft carriers), most ships towed between these two ports would also travel through the freshwater Panama Canal. Thus, most hull fouling organisms on inactive Navy ships towed from Puget Sound to Beaumont would be submerged in freshwater for from eight hours up to two days before entering the saltwater Gulf of Mexico. In addition to the widely fluctuating salinities during towing, the salinity difference between origination port and destination port for this tow scenario is also very large (i.e., from around 27 ppt to 3 ppt, on average). Considering the major changes in salinity along the tow route and between the tow route and the destination port, and the very large salinity difference between the origination and destination port, we find that the NAS introduction risk from inactive Navy ships towed from Puget Sound to Beaumont is *extremely low*.

Hull fouling species towed from Mayport to Beaumont would need to tolerate a moderate change in average salinity (-12 ppt), small change in water temperature (-3 to +8 degrees Fahrenheit, depending on the season) and a short tow duration (four days, on average). Differences in salinity (+22 ppt) and winter water temperature (+22 degrees Fahrenheit) between Mayport and much of the tow route are larger than between ports. However, the very short tow duration (3.5. days, on average) may reduce the effectiveness of such in-transit environmental changes at minimizing the risk of a NAS introduction. We anticipate the risk of a new NAS introduction from towing inactive ships from Mayport to Beaumont would be reduced due to the fact that these ports are expected to have *somewhat similar* species compositions. Considering these factors, we find that the NAS introduction risk from inactive Navy ships towed from Mayport to Beaumont is *low*.

Hull fouling species originating in the polyhaline port of Norfolk (18 to 21 ppt) would be towed in the saltwater Atlantic Ocean and Gulf of Mexico before reaching the very low salinity destination port of Beaumont, with average salinities ranging from one to four ppt. The average tow duration from Norfolk to Beaumont is 15 days. Given the location of these ports in relation to the open ocean and the anticipated ship tow speeds, we assume that for at least two weeks of the trip hull fouling organisms would be towed through high salinity, euhaline waters (i.e., 34 ppt or greater). In addition to the widely fluctuating salinities during towing, the salinity difference between origination port and destination port for this tow scenario is also large (i.e., from around 20 ppt to 3 ppt, on average). If towing occurs during winter, water temperatures could fluctuate along the tow route from around 46 degrees Fahrenheit in the port of Norfolk to 86 degrees Fahrenheit in the Gulf of Mexico. Considering the major changes in salinity between the tow route and the destination port, and the large salinity difference between the origination and destination ports, we find that the NAS introduction risk from inactive Navy ships towed from Norfolk to Beaumont is *extremely low*.

Freshwater biofouling species on Navy ships in Philadelphia would be towed through saltwater in transit to Beaumont, thus significantly reducing the likelihood of survival. The average tow duration from Philadelphia to Beaumont is 20 days. Allowing for a few days of towing through fresh or brackish water on either end of the tow route, we assume that for at least two weeks the ship would be towed through saltwater. Due to a major salinity change in-transit lasting for two or more weeks during towing, we find that the NAS introduction risk from inactive Navy ships towed from Philadelphia to Beaumont is *extremely low*.

<u>NAS Introduction Likelihood for Ships Towed to New Orleans, Louisiana</u>

Since we have determined that there are no ESA-listed species present at the port of New Orleans, the Navy action would not include hull cleaning mitigation for inactive ships towed to New Orleans, regardless of the origination port. For our analysis, we consider New Orleans an oligohaline port with average salinities ranging from around two to five ppt. We find that the NAS introduction risk from inactive Navy ships towed to the oligohaline port of New Orleans (2 to 5 ppt) from the saltwater ports of either San Diego or Pearl Harbor (34 ppt) is *extremely low* due to the major salinity difference between these ports.

Given their close geographic proximity to each other, shared connection to the Gulf of Mexico, similar latitudes, and similar water temperatures, we anticipate the ports of Beaumont and New Orleans would have *very similar* species compositions. Thus, the risk of introducing a new NAS in New Orleans from a ship originating in Beaumont is significantly reduced. As such, very find that the NAS introduction risk from inactive Navy ships towed from Beaumont to New Orleans is *very low*.

Hull fouling species originating in the polyhaline port (24 to 30 ppt) of Bremerton, Puget Sound would need to tolerate a wide range of salinities during the long duration ship tow to New Orleans. All inactive ships would be towed through euhaline waters (around 34 to 36 ppt) for several weeks to months before reaching the very low salinity destination port of New Orleans, with average salinities ranging from two to five ppt. Except for very large inactive ships (e.g., aircraft carriers), most ships towed between these two ports would also travel through the freshwater Panama Canal. Thus, most hull fouling organisms on inactive Navy ships towed from

Puget Sound to New Orleans would be submerged in freshwater for from eight hours up to two days before entering the saltwater Gulf of Mexico. In addition to the widely fluctuating salinities during towing, the salinity difference between origination port and destination port for this tow scenario is also very large (i.e., from around 27 ppt to 4 ppt, on average). Considering the major changes in salinity along the tow route and between the tow route and the destination port, and the very large salinity difference between the origination and destination ports, we find that the NAS introduction risk from inactive Navy ships towed from Puget Sound to New Orleans is *extremely low*.

Hull fouling species towed from Mayport to New Orleans would need to tolerate a moderate change in average salinity (-11 ppt) but virtually no change in water temperature. Differences in salinity (+22 ppt) and winter water temperature (+22 degrees Fahrenheit) between Mayport and much of the tow route are larger than between ports. However, the very short tow duration (two days, on average) may reduce the effectiveness of such in transit environmental changes at minimizing the risk of a NAS introduction. We anticipate the risk of a new NAS introduction from towing inactive ships from Mayport to New Orleans would be reduced due to the fact that these ports are expected to have *somewhat similar* species compositions. Considering these factors, we find that the NAS introduction risk from inactive Navy ships towed from Mayport to New Orleans is *low*.

Hull fouling species originating in the polyhaline port of Norfolk (18 to 21 ppt) would be towed in the saltwater Atlantic Ocean and Gulf of Mexico before reaching the very low salinity destination port of New Orleans, with average salinities ranging from two to five ppt. The average tow duration from Norfolk to New Orleans is 12 days. Given the location of these ports in relation to the open ocean and the anticipated ship tow speeds, we assume that for at least one week of the trip hull fouling organisms would be towed through high salinity, euhaline waters (i.e., 34 ppt or greater). In addition to the widely fluctuating salinities during towing, the salinity difference between origination port and destination port for this tow scenario is also large (i.e., from around 20 ppt to 4 ppt, on average). If towing occurs during winter, water temperatures could fluctuate along the tow route from around 46 degrees Fahrenheit in the port of Norfolk to 86 degrees Fahrenheit in the Gulf of Mexico. Considering the major changes in salinity along the tow route and between the tow route and the destination port, and the large salinity difference between the origination and destination ports, we find that the NAS introduction risk from inactive Navy ships towed from Norfolk to New Orleans is *very low*.

Freshwater biofouling species on Navy ships in Philadelphia would be towed through saltwater during transit to New Orleans, thus significantly reducing the likelihood of survival. The average tow duration from Philadelphia to New Orleans is 19 days. Allowing for a few days of towing through fresh or brackish water on either end of the tow route, we assume for at least two weeks the ship would be towed through saltwater. Due to a major salinity change in-transit lasting for two or more weeks during towing, we find that the NAS introduction risk from inactive Navy ships towed from Philadelphia to New Orleans is *extremely low*.

Step 4: Qualitatively assess the cumulative risk, over the course of the inactive ship tow program, of a new NAS introduction for each destination port by combining the risk assessment for a single ship tow (from Step 3) with the anticipated towing frequencies (see Step 2) for each origination-destination port scenario.

In Step 3 (above) we qualitatively assessed the risk of a new NAS introduction as a result of towing an inactive Navy ship for each origination-destination port combination. Since this is a programmatic opinion with no sunset date, we must evaluate the cumulative risks associated with the Navy's on-going program. In this step we qualitatively assess the cumulative risk, over the course of the inactive ship tow program, of a new NAS introduction for each destination port by combining the risk assessment for each ship tow scenario with the anticipated towing frequencies.

As shown in Table 30 we found the risk of introducing a NAS was *extremely low* for the large majority of inactive ship tow scenarios. These origination-destination port combinations (i.e., shaded in green) would all involve either (1) some form of hull cleaning mitigation, (2) hull painting within the past three years, or (3) a major change in salinity (either between ports or for some extended duration during towing). As discussed previously, for inactive Navy ships that have been either hull cleaned prior to towing or painted within the previous three years, we anticipate the risk of a NAS introduction should be reduced to a level more similar, on average, to the tens of thousands of active commercial ships that regularly move between U.S. coastal ports each year. Considering this very large number of commercial ships and the relatively small number of hull fouling NAS introductions in U.S. ports each year, we find that the introduction of a NAS at any given port as a result of towing inactive Navy ships that have either undergone hull cleaning mitigation or been painted within the previous three years is extremely unlikely to occur and thus would be discountable. Similarly, based on scientific studies of salinity tolerance ranges for hull fouling species, we find that the introduction of a NAS at any given port as a result of towing inactive Navy ships with towing scenarios that involve a major salinity change for more than two weeks is extremely unlikely to occur and thus would be discountable. Therefore, we determine that the risk of a NAS introduction resulting from the Navy's inactive ship tow program to be discountable for the following destination ports: Baltimore, Brownsville, Pearl Harbor, Philadelphia, and Puget Sound.

Table 30. The qualitative level of risk (green = extremely low; yellow = very low; red = low) of a new NAS introduction from towing an inactive Navy ship, and the anticipated average number of inactive ships towed over a five-year period for each origination/destination port.

Origin Port	Destination Port						
	Baltimore	Beaumont	Brownsville	New Orleans	Pearl Harbor	Philadelphia	Puget Sound
Beaumont	0		1	0	0	0	0
Mayport	0	0	0	0	0	1	0
Norfolk	0	1	0	0	0	8	0
San Diego	0	0	0	0	7	0	2
Pearl Harbor	0	0	4	1		0	0
Philadelphia	1	0	9	5	0		0
Puget Sound	0	0	4	1	0	0	

Note: A “0” indicates that the towing of an inactive ship between these ports would be a very rare event, and would likely not exceed once every 25 years, on average.

For an inactive ship towed to the destination port of Beaumont from either San Diego, Pearl Harbor, Puget Sound, Norfolk or Philadelphia, we found the risk of a NAS introduction was also *extremely low*. As such, we find a NAS introduction in Beaumont as a result of towing inactive ships from these origination ports is extremely unlikely to occur and thus would be discountable.

For an inactive ship towed from Mayport to Beaumont, we found that the risk of a NAS introduction was *low*. However, the Navy has indicated that this ship tow scenario is unlikely to occur (i.e., would likely not exceed one ship every 25 years). Considering this extremely low towing frequency along with the *low risk* associated with towing one inactive ship, we find a NAS introduction in Beaumont as a result of towing inactive ships from Mayport is also extremely unlikely to occur and thus would be discountable.

For an inactive ship towed to the destination port of New Orleans from either San Diego, Pearl Harbor, Puget Sound or Philadelphia, we found the risk of a NAS introduction was also *extremely low*. As such, we find a NAS introduction in New Orleans as a result of towing inactive ships from these origination ports is extremely unlikely to occur and thus would be discountable.

For an inactive ship towed from Beaumont or Norfolk to New Orleans, we found that the risk of a NAS introduction was *very low*. For an inactive ship towed from Mayport to New Orleans, we found that the risk of a NAS introduction was *low*. However, the Navy has indicated that these ship tow scenarios (i.e., from either Beaumont, Norfolk or Mayport to New Orleans) are unlikely to occur (i.e., would not likely exceed one ship every 25 years). Considering this extremely low towing frequency, along with the *very low risk* (from Beaumont or Norfolk) or *low risk* (from Mayport) associated with towing an inactive ship, we find a NAS introduction in New Orleans as a result of towing inactive ships from Beaumont, Norfolk or Mayport is extremely unlikely to occur and thus would be discountable.

Step 5: Based on the analysis in Step 4, determine which ESA-listed species and/or critical habitats we anticipate would likely be exposed to a NAS introduction as a result of the proposed action.

The results of our NAS introduction risk analysis indicate that a NAS introduction from inactive Navy ships towed to any of the proposed destination ports from any of the proposed origination ports is extremely unlikely to occur and thus would be discountable. This determination is based on the assumption that the Navy would conduct the agreed upon mitigation measures as described in Section 4 of this opinion. If this mitigation does not occur, the Navy must reinitiate consultation under 50 CFR § 402.16. Because we find it discountable that a new NAS would be introduced into any destination port as a result of the proposed action, it is not likely that ESA-listed species or their critical habitats in these ports, or in nearby areas adjacent to these ports, would be exposed to the adverse effects from a NAS introduction.

10.2 In-water Hull Cleaning Effects Analysis

As part of the mitigation measures agreed to during the course of section 7 consultation, the Navy could conduct hull cleaning mitigation in any of the seven proposed origination ports (Navy 2018b). In cases where the Navy determines dry docking is not practicable, the Navy will conduct hull cleaning mitigation with the ship in the water (Navy 2018c). In this subsection, we evaluate the effects of in-water hull cleaning on the ESA-listed species and critical habitat that may be affected by this activity (see Section 8.2 for complete list of species).

First, we discuss the potential effects of sound-related stressors associated with in-water hull cleaning (Section 10.2.1). We found that the potential effects of sound from in-water hull cleaning on any ESA-listed species would be insignificant. Next, we evaluate the effects of water and sediment quality degradation related stressors associated with in-water hull cleaning (Section 10.2.2). We found that sediment quality degradation from in-water hull cleaning would likely result in adverse effects on the following benthic fish species: Atlantic sturgeon (New York Bight and South Atlantic DPSs), shortnose sturgeon, and green sturgeon Southern DPS. The effects of stressors from this activity, therefore, are carried forward to our Integration and Synthesis section (Section 12) for these species. For all other ESA-listed species and critical habitats analyzed, we found that in-water hull cleaning would not likely result in adverse effects (i.e. NLAA).

10.2.1 Sound

In-water hull cleaning would also introduce noise into the aquatic environment. We do not have specific information on sound levels emitted from underwater hull cleaning equipment. However, given that the origination ports are industrialized areas with heavy ship traffic and other anthropogenic sounds sources, ambient noise levels are expected to be relatively high in general. In addition, because the equipment would be operated by divers underwater (i.e., operated at sound levels within Occupational Safety and Health Administration Permissible Exposure Limits that would not harm human hearing), we would not expect that sound levels would be high enough to adversely affect any ESA-listed species that could occur sporadically in proximity to the ship cleaning operation. Therefore, we find that the potential effects of exposure to noise from in-water hull cleaning would be undetectable, not measurable, or so minor that they cannot be meaningfully evaluated, and thus would be insignificant.

10.2.2 Water and Sediment Quality Degradation

In-water hull cleaning involves the active removal of biofouling organisms and varying amounts of antifouling paint, which can increase localized environmental loading of copper, and possibly other metals such as zinc or lead (Forbes 1996; NUWC 2017b). Copper is toxic to aquatic organisms (mainly dissolved copper due to its bioavailability), which makes it a good biocide, but its use as an antifouling agent can cause unintended consequences when paint containing copper is discharged as particulates into the water column (USDOT 2012). The main environmental concerns regarding contamination from copper-based antifoulants include: (1) bioaccumulation, (2) ecotoxicological effects and subsequent changes to local ecology and biodiversity, and (3) effects on ecosystem function (i.e. microbial and geochemical processes that regulate the cycling, bioavailability, and fate of micro- and macronutrients) (Macleod and Eriksen 2009). Marine organisms vary in their relative accumulation and tolerance of trace metals, due in part to differences in uptake pathways and metals changing bioavailability and speciation (Clement et al. 2010). Prolonged exposure to high concentrations of copper can impact fish in various ways, ranging from impacting sensory organs to causing mortality (Woody 2007). Common sub-lethal effects of copper and zinc in marine organisms include inhibited growth and settlement of larvae, interference with respiration, metabolism, reproduction and/or abnormal behaviors (Clement et al. 2010). Benthic community dynamics (e.g. species composition, abundance and diversity) and functions (e.g. microbial mineralization processes) are known to be highly dependent on the physical, geological, chemical and biological conditions of the local environment (Clement et al. 2010). When metal concentrations in sediments reach extremely high levels, benthic assemblages can be adversely affected to the point that the regular functioning of such communities is comprised (Macleod and Eriksen 2009).

There are several pathways by which metals in antifoulants can become environmentally available: (1) dissolution from the active paint surface, (2) ablation or physical damage of the painted surface, and (3) release from accumulated paint in the sediments (Macleod and Eriksen 2009). Water toxicity associated with heavy metal loading is mediated by the biological, physicochemical and hydrographic conditions of the surrounding environment. Once released from a coating, dissolved copper may take a number of chemical forms in natural seawater environments, including the hydrated free copper ion (Cu2+), dissolved organic copper (labile and inert), inorganic copper complexes, and colloidal and particulate copper (Morel 1983). In general, only the free copper ion represents the bioavailable fraction of copper in the marine environment (Morel 1983). Biofilms such as organic copper-binding ligands in coastal estuaries have been shown to effectively buffer copper toxicity even at relatively high copper loadings (Buck and Bruland 2005; Rivera-Duarte et al. 2005). Hydrographic conditions, such as currents, wind turbulence, tidal exchange, and naturally occurring organic matter, can also play a role in diminishing toxicity from antifouling paint in bays or bodies of water with more open geography. Copper concentrations that exceed the binding capacity of natural ligands can lead to potentially toxic copper conditions (Rivera-Duarte et al. 2005), which can be an issue in harbors and marinas, where there are large concentrations of active ships and where water circulation may be limited.

While in-water hull cleaning may actively release copper, zinc and other metal particles over a short time frame, continuous passive leaching from ship hull paint is the primary contributor to ambient toxicity (NUWC 2017b). For example, Valkirs et al. (1994) found that 99 percent of the dissolved copper loading in San Diego Bay was contributed by antifouling paints on pleasure craft (65 percent) and active Naval ships (34 percent), compared to less than one percent contributed from biofouling removal. The "life span" of antifouling paints is highly variable based on the thickness and type of paint on the hull. Any cuprous-oxide bearing antifouling coating that remains on an inactive ship can release some copper into seawater, but the levels of copper released decrease over time (i.e., with a concurrent decrease in efficacy at preventing marine fouling). While the effective life span of the hull coating's anti-fouling ability is about thirteen years, the active ingredients in the hull coating will continue to degrade over time to the point at which no active ingredients remain. Valkirs et al. (1994) found that low copper release rates from anti-fouling paint was likely the result of the age of the paints. This was consistent with the hulls being heavily fouled as the copper release rate was not high enough to prevent biofouling. Thus, the older the antifouling paint is the less of the active ingredient will be present to potentially enter the water column during hull cleaning.

In 1970 the Navy began formulating organotin antifouling paints containing TBT (Schatzberg 1987). Although TBT was shown to be highly effective at protecting hulls from fouling organisms, growing concerns over the environmental impacts eventually led to a ban on its use in antifouling paints. In October 1987, the EPA issued a preliminary determination to cancel certain registrations of TBT, in which the agency proposed, among other things, to cancel all registrations which exceeded a daily release rate of four micrograms (Showalter and Savarese 2005). In 1988, Congress enacted a partial ban on TBT antifouling paints, eliminating the need for EPA action (Showalter and Savarese 2005). A global ban on TBT entered into force in September 2008 (WWF 2008). The Navy cancelled their last specification for paints containing TBT (DOD-P-24588) on August 14, 1985, before the partial U.S. ban or the global ban (R. Johnson, Navy, pers. comm. to R. Salz, NMFS, October 15, 2018). Although TBT was likely still available on the market when some inactive Navy ships were last painted, based on available information, there are no current inactive Navy ships that have hull coatings containing TBT (R. Johnson, Navy, pers. comm. to R. Salz, NMFS, October 15, 2018).

In addition to heavy metals, the active removal of biofouling organisms from a ship releases organic matter to the immediate aquatic environment. Water bodies can be adversely impacted by elevated concentrations of organic matter. The decay of excess organic matter could result in increases of nutrients, primary productivity (e.g. plankton blooms), and associated decreases in DO, which can lead to hypoxic or anoxic conditions. Hypoxia is a condition where the amount of DO in the water is not adequate for aerobic organisms, while anoxia is a complete lack of DO. Any biofouling (e.g., algal growth, shellfish) removed from the ship during in-water hull cleaning would be expected to settle quickly to the seafloor. Some biofouling organisms would be expected to survive and reattach to bottom substrates, whereas some may not survive and undergo biochemical decomposition that could potentially drive increases in nutrients and decreases in DO (Valkirs et al., 1994) (Figure 52). A portion of the released organic material is also consumed by other organisms, buffering the effect of organic material release. Also, similar to metal loadings, the effect from organic matter loading is diminished by hydrographic conditions. Currents, tidal effects, and wind-induced water turbulence add oxygen to the water, buffering the impacts from organic matter loading. In-water hull cleaning can also increase turbidity and decrease water clarity, although we would anticipate these effects to be temporary and localized.

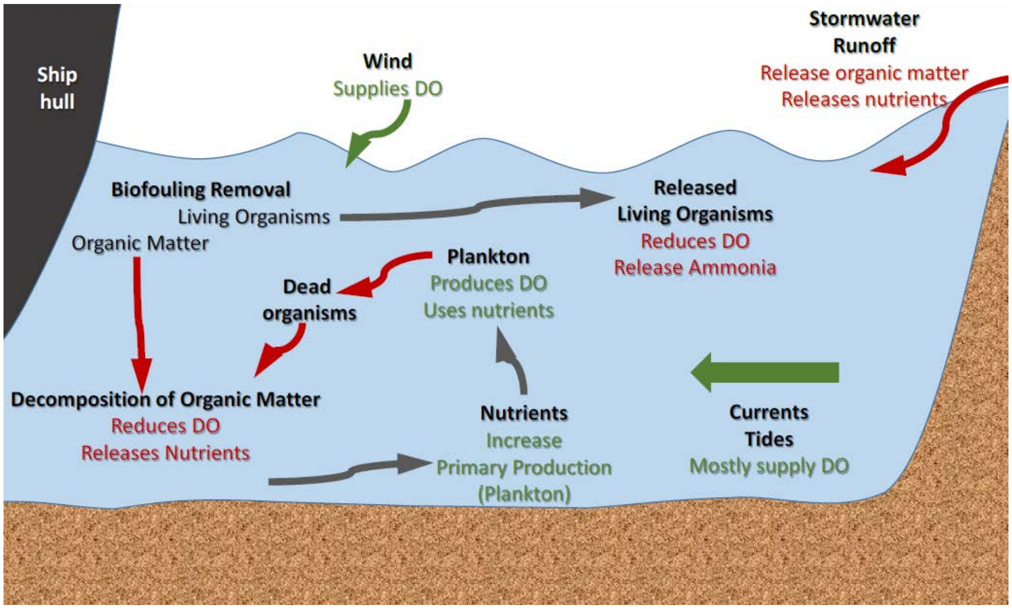


Figure 52. Simplified diagram of the processes affecting the release of organic matter in a coastal marine embayment (NUWC 2017b).

Naval Ocean Systems Center (1981) evaluated water column copper concentrations in the vicinity of in-water hull cleaning operations in San Diego Bay. They found that during hull cleaning operations, water column copper concentrations rapidly decreased with distance from the ship, reaching ambient levels within 50 to 100 meters of the ship. Residence time of the dissolved copper in the vicinity of the cleaning operation was short; ambient levels were reached within one to three hours, dependent on the tidal cycle. This study also found that much of the copper released during the hull cleaning operation was in particulate form, and quickly incorporated into bottom sediments. Slight increases in dissolved and particulate concentrations of lead and zinc were found in the immediate vicinity of the hull cleaning operations, but the concentrations were not considered environmentally significant (Naval Ocean Systems Center 1981). BOD, a measure of the degree of pollution, is the amount of dissolved oxygen that must be present in water in order for microorganisms to decompose the organic matter in the water. BOD increased slightly in some samples taken in the vicinity of in-water hull cleaning operations, but decreased rapidly to ambient concentrations both with distance from the cleaning and after cessation of the cleaning activity (Naval Ocean Systems Center 1981).

The decommissioned ship ex-Independence had been moored in Bremerton (Puget Sound, Washington) from September 1998 until March 2017 when its hull was cleaned in preparation for towing to Brownsville, Texas. The ship's hull was last painted in December 1986. Water quality monitoring was conducted over four sampling events, which included before removal (Event 1, Baseline, November 9 to 10, 2016), during removal (Event 2, During-removal, January 10, 2017), at the end of removal (Event 3, Week-post-removal, January 31, 2017), and 40 days after removal was completed (Event 4, Month-post-removal, March 7, 2017). Water samples were analyzed for dissolved and total copper and zinc, nutrients, DOC, BOD, and turbidity. In addition, in-situ sensors were utilized during the study to provide continuous monitoring of temperature, salinity, pH, DO, and turbidity within the water column at each sampling site. In summary, there was no evidence of any parameter exceeding regulatory thresholds and no evidence of persistent water quality impacts from the ex-Independence biofouling removal operation (NUWC 2017b). The water quality report on the effects of cleaning the hull found statistically significant increases of total and dissolved copper, turbidity and nitrate. No negative impacts to water quality were found from total and dissolved zinc, DO, nitrite, ammonia, DOC or BOD (NUWC 2017b). All parameters returned to baseline levels and were similar to reference conditions within 40 days after biofouling removal was completed (NUWC 2017b). Dissolved copper and zinc, DO, and turbidity did not exceed EPA water quality standards for aquatic life and were less than or equal to 25 percent of the threshold range for exceedance.

As noted above, since the antifouling paint of this ship's hull was over 30 years old, much of the copper and other metal contaminants had likely already leached out into the port prior to the in-water hull cleaning operation. Inactive Navy ships that have been painted more recently may have higher copper release rates as a result of in-water hull cleaning than those found for the ex-Independence.

It should be noted that hull cleaning of the ex-Independence occurred in Puget Sound in the middle of winter, a time of year when the effects on DO would be less of a concern. The effects on DO from in-water hull cleaning would be of greater concern if conducted in the summer months in Puget Sound, or more generally in warmer water origination ports. Effects on DO could be further exacerbated if in-water hull cleaning co-occurs with algal blooms at the origination ports. Through the course of the section 7 consultation process, the Navy has agreed, to the maximum extent practicable, to conduct in-water hull cleanings in the ports of Philadelphia, Pennsylvania, Norfolk, Virginia and Bremerton (Puget Sound), Washington during optimal timeframes (i.e., seasonal work windows) to minimize the potential for adverse effects to ESA-listed resources (Navy 2017). The effects of in-water hull cleaning on water quality and potential impacts on ESA-listed species should, therefore, be lessened since we anticipate the Navy would generally try to conduct this activity during colder months of the year.

In-water hull cleaning of inactive Navy ships can also result in bottom sediment contamination from copper or zinc found in paint particles dislodged from the hull during cleaning. Sediment monitoring was conducted in Sinclair Inlet, Puget Sound to assess potential impacts to sediment quality from in-water hull cleaning of the Navy ex-Independence (Johnston et al. 2018). Sediment chemistry sampling was performed at six locations within the zone of influence near the ship prior to the start of cleaning operations (pre-removal) and another sampling event conducted approximately one month after the ship was moved from Sinclair Inlet and about two months after biofouling removal was completed (post-removal). Data from the 0 to 10 centimeter grab samples taken before and after the project showed that there were no statistically significant (at the $p \leq 0.05$ level) differences in concentrations of total Cu and Zn, grain size, and TOC at the site after biofouling removal was completed compared to before removal (Johnston et al. 2018). However, the lack of statistical significance between the two sampling periods was due to the high variability for both sampling periods, and post-removal sediment Zn concentrations were statistically different (about twice as high) from pre-removal concentrations at the $p \leq 0.10$ level (Johnston et al. 2018). While concentrations of total Cu did not exceed the Washington State Sediment Management Standards Maximum Chemical Criteria (MCC) of 390 micrograms per gram dry weight in any of the pre-removal 0 to 10 centimeter grabs, three (50 percent) of the post-removal 0 to 10 centimeter grabs exceeded this benchmark. These results suggest copper and zinc from the ship's hull were deposited in the sediment below as a result of in-water hull cleaning. However, since high levels of copper were also found in post-removal sediment core samples (i.e., 17 percent of samples exceeded the MCC benchmark), disturbance of the seafloor during hull cleaning may have also contributed to the higher copper levels found in some of the 0 to 10 centimeter post-removal grabs (Johnston et al. 2018).

Metal bioavailability and the potential for metal exposure to cause adverse effects to benthic organisms were evaluated by collecting sediment cores from the site and conducting Acid Volatile Sulfide (AVS) and Simultaneously Extracted Metals (SEM) measurements of core sections in the top 0 to 25 centimeters (0 to 10 inches) of sediment located under the former location of ship's hull. The findings from this study showed that the potential impact of biofouling removal performed on the ex-Independence to the benthic community from the release of copper and zinc was low and that the benthic community in the area of the vessel was not adversely degraded (Johnston et al. 2018). However, as noted in the report, there is uncertainty about this finding due to the variability and interferences associated with the method, including analytical inaccuracy, seasonal variation of AVS, sample inhomogeneity, and other sources of variation.

Another potential effect on water and sediment quality from in-water hull cleaning is the resuspension of contaminants (e.g., PCBs and mercury) in bottom sediments where inactive Navy ships are docked. Resuspension could possibly occur from disturbance of the seafloor caused by hull cleaning divers or from the large volume and weight of organic materials scrapped from the ship's hull when it falls to the seafloor. In soft bottom ports, material falling from the ship would not necessarily remain on the surface but would "splash" on the bottom sinking into the muck and disturbing the bottom conditions (Johnston et al. 2018). Resuspension sampling has not been conducted by the Navy as part of previous in-water hull cleaning monitoring efforts. We have no information to determine whether or not sediment resuspension occurs during in-water hull cleaning, or (if it occurs) whether or not it has more than a minor, short-term effect on water and sediment quality within the origination port. Sediment resuspension would likely be more of a concern in origination ports with high current levels of sediment contamination, such as the port in Bremerton, Puget Sound which has been designated as an EPA Superfund site.

10.2.2.1 Effects on ESA-listed Species

In this section, we discuss the effects of in-water hull cleaning on ESA-listed species. First, we identify which species could potentially overlap spatially with effects from in-water hull cleaning. We then evaluate the likelihood of ESA-listed species exposure to the effects of water quality degradation and then to the effects of sediment quality degradation.

Species That Could Potentially Overlap with Effects from In-water Hull Cleaning

The information presented above suggests potential impacts from in-water hull cleaning would be highly localized. Therefore, the only ESA-listed species that could potentially be exposed to decreased water or sediment quality would be those occurring in the immediate vicinity of where in-water hull cleaning would take place. Although we lack site specific information regarding the presence or abundance of ESA-listed species at proposed origination ports, based on the available information on species ranges and preferred habitats (see Section 8.2 *Status of Species* above), we conservatively assume that any of the following ESA-listed species could overlap spatially with in-water hull cleaning activities: killer whale Southern resident DPS, Hawaiian monk seal, chinook salmon Puget Sound DPS, steelhead Puget Sound DPS, Atlantic sturgeon (New York Bight DPS, Chesapeake Bay DPS, and South Atlantic DPSs), shortnose sturgeon, green sturgeon Southern DPS, eulachon, green sea turtle (Central North Pacific, Eastern Pacific and North Atlantic DPSs), hawksbill, Kemp’s ridley, leatherback, loggerhead Northwest Atlantic Ocean DPS, juvenile bocaccio Puget Sound/Georgia Basin DPS, and juvenile yelloweye rockfish Puget Sound/Georgia Basin DPS. Adult rockfish species (i.e., bocaccio and yelloweye rockfish) are found primarily in deeper waters of Puget Sound and would not be expected to overlap with hull cleaning activities.

Effect of Water Quality Degradation on ESA-listed Species

Due to the age of the antifouling paint on many inactive ships, a large percentage of the copper applied to inactive ships’ hulls has likely already been released into the environment over the long period of time since most inactive ships have been painted (e.g., Valkirs et al. 1994). However, monitoring studies conducted during in-water hull cleaning have shown that some copper can still be released even on ships that had not been painted in several decades. Although in-water hull cleaning can impact levels of DO, turbidity, nitrate, organic debris and dissolved metals in the water column, monitoring studies have shown these conditions to be very temporary, decreasing rapidly to ambient concentrations after cessation of the cleaning activity. For large Navy ships, in-water cleaning would likely be conducted over a long period (i.e., weeks to months) such that biofouling material would be released slowly over time, allowing it to disperse within the environment.

Many of the ESA-listed species that may be found at ports where in-water hull cleaning would be conducted feed primarily in the water column and are not likely to ingest or be directly exposed to contaminants (NMFS 2016g). These include killer whales, Hawaiian monk seals, Chinook salmon, steelhead, sea turtles, eulachon, juvenile bocaccio and juvenile yelloweye rockfish. Given the small timeframe in which copper or zinc would be suspended in the water column (minutes to hours) after hull cleaning is completed, it is unlikely that these ESA-listed species would consume or be exposed to these metals in the water column for prolonged periods of time as a result of hull cleaning. In addition, we expect the densities of ESA-listed species in the immediate vicinity of where in-water hull cleaning would occur to be very low, further reducing the likelihood of exposure. Marine mammals, fish and turtles, in general, would be expected to avoid areas where in-water hull cleaning is being conducted due to the noise, debris and general disturbance such activities produce. Given the extremely small area where in-water hull cleaning would be conducted in relation to the total available habitat within origination ports, any effects of avoiding these areas for the short duration during active hull cleaning would be undetectable, not measurable, or so minor that they cannot be meaningfully evaluated, and thus would be insignificant.

Through the course of the section 7 consultation process, the Navy has agreed to try to conduct in-water hull cleaning during seasonal work windows designed to minimize the potential impacts on ESA-listed species. In Philadelphia, the Navy would, to the maximum extent practicable, conduct in-water hull cleaning from November 1 through March 15 to avoid impacts to adult Atlantic sturgeon during spawning migrations and early life stages of Atlantic sturgeon (Navy 2017)(Navy 2017)(Navy 2017)(Navy 2017)(R. Johnson, Navy, pers. comm. to R. Salz, NMFS, October 18, 2018). In Norfolk, the Navy would, to the maximum extent practicable, conduct in-water hull cleanings from November through February to avoid impacts to migrating Atlantic sturgeon and the seasonal occurrence of sea turtles. In Puget Sound, the Navy would, to the maximum extent practicable, conduct in-water hull cleanings from December through February to avoid impacts to migrating salmon and sensitive life stages of ESA-listed rockfish species (Navy 2017)(R. Johnson, Navy, pers. comm. to R. Salz, NMFS, October 15, 2018). Although juvenile ESA-listed fish species (i.e., sturgeon and/or salmon) could be present in these ports year-round, conducting in-water hull cleaning at these ports during the winter would avoid anadromous fish spawning migrations to freshwater habitats and the seasonal occurrences of other ESA-listed species or sensitive life stages. The effects on water quality, including DO, turbidity, and the possibility of synergistic effects with co-occurring algal blooms, would also likely be minimized if in-water hull cleaning is conducted during winter months.

For inactive ships which are too large to travel through the Panama Canal (i.e., aircraft carriers or vessels of similar size), towing actions have a preferred seasonal window for going around Cape Horn through the Strait of Magellan. This window is during the Southern Hemisphere's summer months due to the likelihood of severe weather in other seasons. Considering this preferred window for towing, in-water hull cleaning of aircraft carriers (or ships of similar size) in preparation for towing between ocean basins (e.g., Pacific to Atlantic/Gulf of Mexico) would likely be conducted during the winter months since all origination ports are located in the Northern Hemisphere. Thus, in-water hull cleaning of the largest inactive ships, which generally have the greatest extent of biofouling, would likely occur during the winter for ship tow scenarios around Cape Horn.

In summary, given the short time duration of this stressor, small area affected, expected low densities of ESA-listed species in the affected areas, and the anticipated timing of most in-water hull cleanings (i.e., winter months), we find direct exposure to the effects of water quality degradation is extremely unlikely to occur and thus would be discountable. As discussed above, while some ESA-listed marine mammals, turtles and fish would be expected to avoid areas where in-water hull cleaning in actively being conducted, any effects associated with avoidance of these areas would be undetectable, not measurable, or so minor that they cannot be meaningfully evaluated, and thus would be insignificant.

ESA-listed species that feed in the water column could be exposed to contaminants indirectly through bioaccumulation up the food chain. However, given the highly localized nature of this stressor, any indirect effects would be undetectable, not measurable, or so minor that they cannot be meaningfully evaluated, and thus would be insignificant. As discussed above, resuspension of contaminated sediments could represent another potential stressor caused by in-water hull cleaning, particularly in Bremerton, Puget Sound which is a designated Superfund site. However, at present we have no information to determine whether or not sediment resuspension occurs during in-water hull cleaning, nor to evaluate the magnitude of the impact this stressor may have on ESA-listed species, if it does occur.

Effect of Sediment Quality Degradation on ESA-listed Species

A potentially longer term (but still highly localized) stressor from in-water hull cleaning is from copper and possibly other metals (i.e., zinc or lead) that settle onto the seafloor and mix with the benthic environment. Solid phase exposure can occur when metals become available to benthic fauna through ingestion and filtering of contaminated organic matter and any associated sediments (Clement et al. 2010). Metals can accumulate over time in sediments, particularly organically enriched sediments which have a higher capacity to bind and accumulate copper (Macleod and Eriksen 2009). If benthic fish normally feed in the port areas where in-water hull cleaning would occur, the risk of exposure to contaminated sediments can last for years or decades, as copper is highly conservative once introduced into bottom sediments (Petersen et al.

1997; Ranke 2002). The only way to reduce sediment copper levels in a local area is by dispersal or removal in the form of either biological bioturbation or hydrological resuspension (Smith and Williamson 1986). Macleod and Eriksen (2009) analyzed the ecological impacts of copper-based antifoulants used in the salmon farming industry in Tasmania. They found no major acute effects of antifoulant release in the water-column. Instead, they concluded that the greatest risk appears to be the potential for build-up of copper and zinc in the sediments around the farms.

As discussed above, sediment monitoring conducted to assess potential impacts to sediment quality from in-water hull cleaning of the Navy ex-Independence found no statistically significantly ($p \leq 0.05$) differences in concentrations of total copper or zinc between pre-removal and post-removal samples and that the potential impact on the benthic community from metal exposure associated with biofouling removal was low (Johnston et al. 2018). As noted by the authors, the non-significant results may have been due to small sample sizes and high variability in the monitoring data. Metal contaminant levels in post-removal samples were generally higher compared to pre-removal samples, and copper levels exceeded the state of Washington MCC benchmark for one-half of the post-removal samples taken. Since copper and zinc from antifouling paint particles can accumulate and persist in sediments over time, we expect the likelihood of exposure to contaminated sediments at a particular origination port would be influenced by the relative frequency of in-water hull cleanings conducted at each port. As the total load of copper and zinc within the sediment increases, so does the likelihood of exposure through ingestion of sediment or benthic prey. The accumulation of copper and zinc in sediments over time also increase the likelihood of exposure to contaminant levels that could result in adverse effects on benthic feeding fish. While the risk of exposure to such levels from in-water hull cleaning of one inactive ship may be relatively low, the cumulative risk from in-water hull cleaning many ships at a given port over the course of the Navy's inactive ship towing program would be considerably greater.

During, and for some period after, in-water hull cleaning benthic fish species may be attracted to hull fouling prey organisms that have been removed from the hull and settled onto the seafloor. Shells and other calcareous material scrapped from the hull also provide attachment substrate for organisms to colonize. In-water hull cleaning of the ex-Independence in Puget Sound resulted in increased structure and diversity of the benthic community, the effects of which lasted for at least two months after hull cleaning was completed (Johnston et al. 2018). Benthic associated species, such as sturgeon, that occupy port areas where in-water hull cleaning may occur are more likely to be exposed to elevated levels of sediments contaminated with copper and other metals as a result of hull cleaning. Bocaccio and yelloweye rockfish are also benthic associated species in their adult life stage. However, adult rockfish occupy deeper waters of Puget Sound and we do not anticipate overlap between in-water hull cleaning activities and adult rockfish. While juvenile bocaccio and yelloweye rockfish could overlap with in-water hull cleaning activities at the port of Bremerton, as discussed above, in this life stage rockfish feed primarily in the water column and are not likely to ingest or be directly exposed to copper or zinc in bottom sediments below where in-water hull cleaning would occur. As such, we find direct exposure of juvenile bocaccio and yelloweye rockfish to the effects of sediment quality degradation is extremely unlikely and is therefore discountable.

Exposure to contaminated sediments could contribute to the bioaccumulation of heavy metals in sturgeon located in origination port environments. Heavy metals and organochlorine compounds accumulate in sturgeon tissue, but their long-term effects are not known (Ruelle and Keenlyne 1993). Life histories of sturgeon species (i.e., long lifespan, extended residence in estuarine habitats, benthic foraging) predispose them to long-term, repeated exposure to environmental contamination and potential bioaccumulation of heavy metals and other toxicants (Dadswell 1979). Sublethal effects may include impacts to sensory organs or reproductive impairments, although there has been little work on the effects of contaminants on sturgeon to date. High levels of contaminants in several other fish species are associated with reproductive impairment (Billsson 1998; Cameron et al. 1992; Giesy et al. 1986; Hammerschmidt et al. 2002), reduced survival of larval fish (McCauley et al. 2015; Willford et al. 1981), delayed maturity and posterior malformations (Billsson 1998).

As discussed in Section 4 of this opinion, based on the agreed upon mitigation, the Navy would conduct hull cleaning mitigation on inactive ships towed from Puget Sound to Brownsville, Baltimore, or Pearl Harbor. The Navy anticipates towing about four ships every five years from Puget Sound to Brownsville (Navy 2018b). Ships towed from Puget Sound to either Baltimore or Pearl Harbor would also require hull cleaning, although these ship tow scenarios are expected to occur very rarely. Hull cleaning mitigation can be conducted with the ship in the water or in dry dock. Although the Navy is required to clean vessel hulls in dry dock to the greatest extent practicable (81 FR 69753), there are many factors (see *Section 4.1 Ship Hull and Underwater Component Cleaning Mitigation* for details) that could limit the availability of dry dock space and the practicability of this hull cleaning method (Navy 2018d). Considering these factors, for purposes of our effects analysis we conservatively assume that all hull cleaning mitigation would be conducted with the ship in the water. If we conservatively assume that all hull cleaning mitigation in Puget Sound would be conducted in-water (i.e., none in dry dock), the expected frequency of in-water hull cleanings in Puget Sound is approximately one ship per year. As discussed above, since copper is highly conservative once introduced into bottom sediments (Petersen et al. 1997; Ranke 2002), the risk of exposure for benthic fish species can last for years or decades.

Puget Sound is within the Southern DPS green sturgeon's range and fish from this population have been recorded in this estuary in both summer and winter months (AquaMaps 2016; Lindley et al. 2008)(74 FR 52299). Preliminary (unvalidated) tag detection data from Admiralty Inlet suggest that Southern DPS green sturgeon may be using the Puget Sound estuary at a higher rate than was previously thought (see *Status of Species* section 8.2.13 for details) Monitoring data for determining Southern DPS green sturgeon densities for particular geographic locations within Puget Sound (e.g., Sinclair Inlet or Port of Bremerton) are not available. Although we expect the occurrence of this species in the Port of Bremerton to be a rare event, considering the expected frequency of in-water hull cleanings at this port, the conservative nature of copper and zinc in bottom sediments, and that this is a programmatic opinion on a program with no sunset date, we find it reasonably likely that green sturgeon Southern DPS would be exposed to sediments contaminated with copper (and possibly zinc) found in antifouling paints as a result of the Navy's inactive ship tow program.

Based on the agreed upon mitigation, the Navy would conduct hull cleaning (either in-water or dry dock) on ships located in Mayport prior to towing to Philadelphia (Navy 2018b). The Navy anticipates towing about one ship every five years from Mayport to Philadelphia. On very rare occasions, ships towed from Mayport to any of the following destination ports would also require hull cleaning: Baltimore, Brownsville, Pearl Harbor, or Puget Sound. If we conservatively assume that all hull cleaning mitigation in Mayport would be conducted in-water (i.e., none in dry dock), the expected frequency of in-water hull cleanings in Mayport is about one to two ships every five years. Although the short-term risk of exposure may be relatively low given this frequency of in-water hull cleanings, the risk would likely increase over the course of the Navy's inactive ship tow program due to the cumulative nature of copper and other metals in bottom sediments. The port of Mayport, located near the mouth of the St. Johns River, is considered within the species range for the South Atlantic DPS Atlantic sturgeon and the shortnose sturgeon (ASSRT 2007; FFWCC 2007; SSSRT 2010). Densities of Atlantic and shortnose sturgeon in the St. Johns River are expected to be relatively low considering the low number of recent sightings and that this estuary represents the southern extent of the range for both species. Although these species may only rarely be found in the port of Mayport, considering the expected frequency of in-water hull cleanings at this port, the conservative nature of copper in bottom sediments, and that this is a programmatic opinion on a program with no sunset date, we find it reasonably likely that South Atlantic DPS Atlantic sturgeon and shortnose sturgeon would be exposed to sediments contaminated with copper (and possibly other metals) found in antifouling paints as a result of in-water hull cleaning conducted in Mayport as part of the Navy's inactive ship tow program.

Based on the agreed upon mitigation, the Navy would conduct hull cleaning (either in-water or dry dock) on ships located in Philadelphia prior to towing to Baltimore (Navy 2018b). The Navy anticipates towing about one ship every five years from Philadelphia to Baltimore. If we assume that all hull cleaning mitigation in Philadelphia would be conducted in-water (i.e., none in dry dock), the expected frequency of in-water hull cleanings in Philadelphia is about one ship every five years. Although the short-term risk of exposure may be relatively low given this frequency of in-water hull cleanings, the risk would likely increase over the course of the Navy's inactive ship tow program due to the cumulative nature of copper and zinc in bottom sediments.

The port of Philadelphia, located on the Delaware River, is within the species' range for the New York Bight DPS Atlantic sturgeon and the shortnose sturgeon (ASSRT 2007; SSSRT 2010). In the Delaware River, there is evidence of Atlantic sturgeon presence from the mouth of the Delaware Bay to the head-of-tide at the fall line near Trenton on the New Jersey side and Morrisville on the Pennsylvania side of the River, a distance of 220 river kilometers (Breece et al. 2013). The stretch of the river that includes the port of Philadelphia is also part of the critical habitat designation (Unit 4) for the New York Bight DPS (see Figure 30) (82 FR 39160). The Delaware River estuary represents one of just two known spawning subpopulation for the New York Bight DPS. The Delaware River once supported the largest spawning subpopulation of Atlantic sturgeon in the United States, with 3,200 metric tons of landings in 1888 (ASSRT 2007; Secor and Waldman 1999). The most recent estimate of the adult Atlantic sturgeon spawning population in the Delaware River is 1,305 individuals (NMFS 2017b). A study by Hale et al. (2016) estimated that 3,656 (95% confidence interval from 1,935 to 33,041) juveniles (ages 0 to 1) used the Delaware River estuary as a nursery in 2014.

For shortnose sturgeon, the Delaware River subpopulation represents the second highest estimated adult abundance (12,000) of all estuaries within the species range (SSSRT 2010). O'herron et al. (1993) used sonic tracking to study the movement of shortnose sturgeon in the Delaware River from 1983-1987. They found that after spawning in the nontidal river above Trenton (from late March through April), sturgeon traveled rapidly downstream into the tidal portion of the river near Philadelphia, where they remained through the end of May. Before the end of June, most sturgeon returned upstream and re-entered the upper tidal river near Trenton, where most apparently remained for the summer and winter. Thus, the stretch of the Delaware River between Philadelphia to just above Trenton is used intensively by shortnose sturgeon (O'herron et al. 1993).

Considering the importance of the stretch of the Delaware River that includes the port of Philadelphia as sturgeon habitat, the expected frequency of in-water hull cleanings at this port, the conservative nature of copper in bottom sediments, and that this is a programmatic opinion on a program with no sunset date, we find it reasonably likely that New York Bight DPS Atlantic sturgeon and shortnose sturgeon would be exposed to sediments contaminated with copper (and possibly other metals) found in antifouling paints as a result of in-water hull cleaning conducted in Philadelphia as part of the Navy's inactive ship tow program.

Based on the agreed upon mitigation, the Navy would conduct hull cleaning (either in-water or dry dock) on ships located in Norfolk prior to towing to Baltimore, Brownsville, Pearl Harbor or Puget Sound (Navy 2018b). The Navy anticipates the towing frequency for each of these routes to be extremely low (i.e., less than one ship every 25 years). Atlantic sturgeon from the

Chesapeake Bay DPS could potentially be exposed to the effects of in-water hull cleaning conducted in the port of Norfolk, Virginia. However, given the extremely low towing frequency, we find the exposure of Atlantic sturgeon to copper or other metals in bottom sediments resulting from in-water hull cleaning in Norfolk is extremely unlikely to occur and thus would be discountable.

In summary, we find that the following ESA-listed sturgeon species would likely be adversely affected by contaminated sediments as a result of in-water hull cleaning mitigation conducted as part of the Navy's inactive ship tow program: green sturgeon Southern DPS, Atlantic sturgeon South Atlantic DPS, Atlantic sturgeon New York Bight DPS, and shortnose sturgeon. Only those individual fish foraging within the immediate vicinity of where in-water hull cleaning is conducted would be affected. The total area affected by contaminated sediments, as well as the level of contamination, should be a factor of the frequency of in-water hull cleanings at each port (i.e., the more in-water hull cleanings conducted at a port, the greater the area contaminated and the level of contamination). The affected area will also vary depending on environmental conditions at each origination port at the time of in-water hull cleaning. For example, in San Diego Bay, water column copper concentrations rapidly decreased with distance, reaching ambient levels within 50 to 100 meters of the ship following in-water hull cleaning (Naval Ocean Systems Center 1981; NUWC 2017b). In Puget Sound, the area most likely impacted by material released during biofouling removal (i.e., area of influence) from the ex-Independence extended in one direction for over 1,300 meters from the inactive ship (NUWC 2017b). These areas of influence were in regard effects on water quality. We wound expect the areal extent of sediment affected to be much smaller. Sediment monitoring conducted for the ex-Independence hull cleaning was confined to the area immediately below the location where the ship was moored. Based on modeling results that is where the vast majority of material was expected to settle to the bottom (i.e. zone of influence) (Johnston et al. 2018).

The number of individual Southern DPS green sturgeon that would be adversely affected is expected to be very small given the extremely localized nature of this stressor and the expected very low density of this species in the port of Bremerton. Similarly, the number of South Atlantic DPS Atlantic sturgeon and shortnose sturgeon that would be adversely affected in Mayport is also expected to be very small. A relatively small number of in-water hull cleanings are expected to occur in Mayport and sturgeon densities (both Atlantic and shortnose) are expected to be very low in the St Johns River (Florida), which represents the southern extent of the range for these species and is not known to support a spawning population of sturgeon. Since the stretch of the Delaware River that passes through Philadelphia is widely used by Atlantic (New York Bight DPS) and shortnose sturgeon, we expect more individual sturgeon to pass through this port as compared to Bremerton or Mayport. However, since a very small number of in-water hull cleanings are expected to occur in this port (about one every five years, on average), the area affected (and level of sediment contamination) at this port as a result of in-water hull cleaning would still be relatively small compared to the available sturgeon habitat in this stretch of the river. As such, we expect only a very small proportion of the Atlantic and shortnose sturgeon Delaware River subpopulations would forage within the affected area, and thus be exposed to sediments contaminated by in-water hull cleaning.

For all other ESA-listed species that may overlap with in-water hull cleaning activities, we find that direct adverse effects from water and sediment quality degradation to be extremely unlikely to occur and thus would be discountable. We also find that any indirect effects from the bioaccumulation of heavy metals up the food chain would be undetectable, not measurable, or so minor that they cannot be meaningfully evaluated, and thus would be insignificant.

10.2.2.2 Effects on Designated Critical Habitat

Designated critical habitat for the following species could potentially overlap with (or is in close proximity to) in-water hull cleaning activities conducted as part of the proposed action: Southern Resident DPS killer whale, Chinook salmon Puget Sound DPS, steelhead Puget Sound DPS, bocaccio Georgia Basin/Puget Sound DPS, yelloweye rockfish Georgia Basin/Puget Sound DPS, and Atlantic sturgeon New York Bight DPS. As part of our effects analysis, we evaluate the effects of in-water hull cleaning of inactive Navy ships at origination ports on those essential PBFs of designated critical habitat that may be affected (Table 31).

The Puget Sound Naval Shipyard at Naval Base Kitsap was excluded from critical habitat designation for Puget Sound/Georgia Basin yelloweye rockfish and bocaccio, Puget Sound DPS steelhead, Puget Sound Chinook salmon, and Southern Resident DPS killer whale. Most slime, marine growth, and anti-fouling paint particles that would be released during in-water hull cleaning would be expected to settle on the substrate in close proximity to the inactive ship being cleaned. Thus, the habitat areas that would be most exposed to in-water hull cleaning stressors (i.e., those directly below the ship) are excluded from critical habitat designation. While it is possible that small amounts of hull debris could drift into adjacent areas that are designated as critical habitat, the impacts from such drift would likely be short-term and at undetectable levels. Therefore, any effects on the essential PBFs of ESA designated critical habitat in Puget Sound from in-water hull cleaning would be undetectable, not measurable, or so minor that they cannot be meaningfully evaluated, and thus would be insignificant.

NMFS did not exclude the in-water parts of the Philadelphia Navy Yard Annex Reserve Basin and Piers that occur in the Delaware River critical habitat unit of the Atlantic sturgeon New York Bight DPS. As such, critical habitat areas located adjacent to the Navy Yard could be directly exposed to debris from in-water hull cleaning. Water quality impacts of hull cleaning could affect the Atlantic sturgeon essential physical and biological feature related to oxygen levels. Sturgeon are more sensitive to low level DO conditions than other fishes, possessing limited behavioral and physiological capacity to respond to hypoxia. In experiments on Atlantic sturgeon, the effect of oxygen level on routine metabolism, consumption, feeding metabolism,

Table 31. Summary of essential physical and biological features (PBFs) of designated critical habitat that may be affected by in-water hull cleaning[1].

Species	Habitat Type	Essential PBFs
Killer whale Southern Resident DPS	General	Water quality to support growth and development; prey species of sufficient quantity, quality and availability to support individual growth, reproduction and development, as well as overall population growth; and inter-area passage conditions to allow for migration, resting, and foraging.
Chinook salmon Puget Sound DPS and Steelhead Puget Sound DPS	Estuarine habitat	Estuarine areas free of obstruction with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation.
Bocaccio Georgia Basin/Puget Sound DPS and Yelloweye rockfish Georgia Basin/Puget Sound DPS	Nearshore juvenile habitat	1) quantity, quality, and availability of prey species to support individual growth, survival, reproduction, and feeding opportunities, and 2) water quality and sufficient levels of DO to support growth, survival, reproduction, and feeding opportunities.
Atlantic sturgeon New York Bight DPS	Riverine migratory, spawning and rearing habitat	Water, especially in the bottom meter of the water column, with the temperature, salinity, and oxygen values that, combined, support: (1) spawning; (2) annual and inter-annual adult, subadult, larval, and juvenile survival; and (3) larval, juvenile, and subadult growth, development, and recruitment.

[1] Note: This table only lists those essential PBFs that we determined may be affected by in-water hull cleaning. Other essential PBFs of critical habitat for these species that are not affected by this activity are not provided here.

growth, and survival has been shown to be conditional on temperature (Secor and Niklitschek 2001). To avoid lethal and sublethal effects, sturgeon must avoid temperatures above 33.7 degrees Celsius with complete oxygen saturation (Ziegeweid et al. 2008b) or hypoxic conditions, particularly as water exceeds about 15 degrees Celsius (Secor and Gunderson 1998). Increased nutrients released from the hull could result in algal blooms and subsequent die-offs, which may lead to decreased levels of DO. However, based on the post-hull cleaning water quality reports discussed above, we would expect increased nutrient loading to be a short-term effect with levels returning to baseline within a few hours from cleaning. If DO levels are impacted, the impact would likely be short-term and limited to a small area below and immediately adjacent to the ship. Sturgeon in the area would be expected to swim to an unaffected area to avoid stress. Also, as discussed above, since we expect most in-water hull cleaning in Philadelphia would occur from November through early March, any adverse effects on sturgeon related to DO levels would likely be minimized. In addition, based on the Navy's approach for determining scenarios when mitigation would be required (i.e., like-to-like salinities and ESA-listed species at destination port), we expect very few hulls would be cleaned in this port. Based on current port salinities and ESA-listed species distribution, the hulls of inactive ships originating in Philadelphia would only need to be cleaned if they were being towed to Baltimore (about one every five years, on average). Based on the factors above, we find that any effects on Atlantic sturgeon New York Bight DPS designated critical habitat from in-water hull cleaning would be not measurable or so minor that they cannot be meaningfully evaluated, and thus would be insignificant.

10.3 Effects Determinations Summary

In this section we summarize our effects determinations for each ESA-listed species and critical habitat evaluated in our effects analysis.

10.3.1 ESA-listed Species and Critical Habitat Not Likely to be Adversely Affected

In Section 8.1 above (*Species and Critical Habitat Not Carried Forward*), we identified those ESA-listed species and critical habitats that were not likely to be adversely affected (NLAA) by the proposed action, and therefore did not need to be evaluated further in our effects analysis. Based on the results of our effects analysis, we determine that the ESA-listed species and critical habitats listed below are not likely to be adversely affected (NLAA) by any activities associated with the proposed action. These species and critical habitats are either not likely to be exposed to stressors resulting from the proposed action or, if exposed, any adverse effects would likely be insignificant (i.e., undetectable, not measurable, or so minor that they cannot be meaningfully evaluated) or discountable (i.e., extremely unlikely to occur).

NLAA ESA-listed Species

- Killer whale, Southern Resident DPS
- Hawaiian monk seal
- Sea Turtles
    - Green: Central North Pacific DPS and North Atlantic DPS
    - Hawksbill
    - Leatherback
    - Kemp’s ridley
    - Loggerhead Northwest Atlantic DPS
- Fishes
    - Atlantic sturgeon DPSs: Chesapeake Bay, Carolina and Gulf of Maine
    - Gulf sturgeon
    - Bocaccio: Puget Sound/Georgia Basin DPS
    - Chinook salmon: Puget Sound ESU
    - Chum salmon: Hood Canal summer-run ESU
    - Eulachon: Southern DPS

  - Steelhead trout: Puget Sound DPS
  - Yelloweye rockfish: Puget Sound/Georgia Basin DPS

<u>NLAA Designated Critical Habitat</u>

- Killer whale, Southern Resident DPS
- Sea Turtles
  - Leatherback
  - Loggerhead: Northwest Atlantic Ocean DPS
- Fishes
  - Atlantic sturgeon New York Bight DPS
  - Gulf sturgeon
  - Bocaccio: Puget Sound/Georgia Basin DPS
  - Chinook salmon: Puget Sound ESU
  - Chum salmon: Hood Canal summer-run ESU
  - Green sturgeon: Southern DPS
  - Steelhead trout: Puget Sound DPS
  - Yelloweye rockfish: Puget Sound/Georgia Basin DPS

10.3.2 ESA-listed Species Likely to be Adversely Affected

- Fishes
  - Atlantic sturgeon New York Bight DPS
  - Atlantic sturgeon South Atlantic DPS
  - Green sturgeon Southern DPS
  - Shortnose sturgeon

11 CUMULATIVE EFFECTS

"Cumulative effects" are those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation (50 C.F.R. §402.02). Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

This section attempts to identify the likely future changes and their impact on ESA-listed species and their critical habitats in the action area. This section is not meant to be a comprehensive socio-economic evaluation, but a brief outlook on future changes in the environment. Projections are based upon recognized organizations producing best-available information and reasonable rough-trend estimates of change stemming from these data. However, all changes are based upon projections that are subject to error and alteration by complex economic and social interactions.

During this consultation, we searched for information on future state, tribal, local, or private (non-Federal) actions reasonably certain to occur in the action area. We did not find any information about non-Federal actions other than what has already been described in the *Environmental Baseline* (Section 9), most of which we expect would continue in the future. An increase in these activities could similarly increase their effects on ESA-listed species and for some, an increase in the future is considered reasonably certain to occur. Given current trends in global population growth, threats associated with climate change, invasive species, pollution, habitat degradation, dredging, fisheries bycatch, ship strikes, and anthropogenic sound are likely to continue to increase in the future, although the projected increase in effects from these activities may be somewhat countered by future conservation and management measures. For many of the activities and associated threats identified in the *Environmental Baseline*, and other unforeseen threats, the magnitude of increase and the significance of any anticipated effects remain unknown. The best scientific and commercial data available provide little specific information on any long-term effects of these potential sources of disturbance on populations of ESA-listed species. Thus, this opinion assumed effects in the future would be similar to those in the past and, therefore, are reflected in the anticipated trends described in the *Species and Critical Habitat Evaluated* (Section 8) and *Environmental Baseline* (Section 9).

12 INTEGRATION AND SYNTHESIS

The *Integration and Synthesis* section is the final step in our assessment of the risk posed to species and critical habitat as a result of implementing the proposed action. In this section, we add the *Effects of the Action* (Section 10) to the *Environmental Baseline* (Section 9) and the *Cumulative Effects* (Section 11) to formulate the agency's opinion as to whether the proposed action is likely to: (1) reduce appreciably the likelihood of both the survival and recovery of a ESA-listed species in the wild by reducing its numbers, reproduction, or distribution; or (2) reduce the value of designated or proposed critical habitat for the conservation of the species. These assessments are made in full consideration of the *Species and Critical Habitat Evaluated* (Section 8).

In our *Effects of the Action*, we determined that several of the activities conducted as part of the proposed action or agreed upon mitigation measures would result in only insignificant or discountable effects on ESA-listed species and critical habitat. We summarize our analysis of these activities and associated stressors as follows:

- Given the low speed and infrequency of transit, and the expected density of ESA-listed species along the tow routes, the likelihood of ships encountering ESA-listed species and posing a strike risk is extremely unlikely to occur and thus would be discountable.
- The temporary disturbance of ESA-listed species associated with the anticipated infrequent ship interactions as part of the proposed action are not expected to result in injury or reduced fitness as we do not anticipate any significant disruption of breeding, feeding, or sheltering to occur. Therefore, any potential effects from ship sound or ship avoidance behavior resulting from the proposed action would be undetectable, not measurable, or so minor that they cannot be meaningfully evaluated, and thus would be insignificant.
- Exposure to contaminants or hazardous material discharged from the shipbreaking process is extremely unlikely to occur and thus would be discountable.
- The potential for adverse effects associated with underwater noise from in-water hull cleaning and shipbreaking activities is extremely unlikely to occur and thus would be discountable.
- Due to the Navy's protocols for handling ballast water, a NAS introduction via ballast water either from the tug or the towed inactive ship, is extremely unlikely to occur and thus would be discountable.
- Due to the Navy's agreed to hull cleaning mitigation measures and other ship tow scenario factors, a NAS introduction from towing inactive ships to any of the proposed destination ports from any of the proposed origination ports is extremely unlikely to occur and thus would be discountable.

- During the towing of the ship and prior to reaching the destination port, the likelihood of a NAS releasing from a slowly towed inactive ship, finding suitable substrate for attachment, reproducing successfully and becoming established in the new environment (i.e., either open ocean or shipping channel) is extremely unlikely to occur and thus would be discountable.

Based on our effects analysis, adverse effects to ESA-listed species are likely to result from sediment quality degradation resulting from in-water hull cleaning (Section 10.2.2). We find it reasonably likely that the following ESA-listed species of sturgeon would be adversely affected by contaminated sediments as a result of the proposed action: green sturgeon Southern DPS, Atlantic sturgeon South Atlantic DPS, Atlantic sturgeon New York Bight DPS, and shortnose sturgeon. Our Integration and Synthesis section focuses on the anticipated adverse effects associated with this stressor as a result of in-water hull cleaning mitigation. The following discussions summarize the probable risks that sediment quality degradation from in-water hull cleaning pose to threatened and endangered species that are likely to be exposed. These summaries integrate our exposure and response analyses from above.

12.1 Atlantic Sturgeon

The major anthropogenic stressors that contributed to the sharp decline of Atlantic sturgeon populations were commercial fisheries, habitat curtailment and alteration from dams and dredging, and degraded water quality. While Atlantic sturgeon are still at risk from anthropogenic threats, some of the major threats that contributed to sharp population declines in the past have either been eliminated or reduced. Efforts made over the past few decades to reduce the impact of these threats have slowed the rate of decline for many sturgeon populations. Directed commercial fishing for Atlantic sturgeon which decimated the population during the 20th century, was prohibited coast-wide by the late 1990's. Several dams within Atlantic sturgeon historic ranges have been removed or naturally breached. Implementation of the Clean Water Act resulted in water quality improvements throughout the species' range, although many rivers and estuaries occupied by sturgeon are still not meeting their designated use water quality standards. Water quality in the Hudson has improved markedly since the 1980s and is no longer considered a major threat to this subpopulation (ASSRT 2007). Similarly, restrictions on dredging in the upper portions of the Delaware have likely had beneficial impacts on Atlantic sturgeon spawning success. It is likely that some current threats to Atlantic sturgeon will increase in the future. These include global climate change, population growth, and land use changes. However, it is difficult to predict the magnitude of these threats in the future or their impact on sturgeon populations.

12.1.1 New York Bight DPS

The New York Bight DPS of Atlantic sturgeon is listed as endangered and includes seven river systems, only three of which are known spawning subpopulations: Delaware, Hudson River, with intermittent spawning in the Connecticut. Long-term surveys indicate that the Hudson River subpopulation has been stable or slightly increasing in abundance since 1995 (ASSRT 2007). In their opinion on the sturgeon research program, NMFS (2017c) assigned the Hudson and Delaware Atlantic sturgeon subpopulations health category ratings of "high" based on the following health index criteria: regular spawning; juveniles present and progressing through age classes; no major ongoing threats; a few minor threats including water quality and impingement/entrainment; and relatively large estimated effective population sizes. The estimated adult Atlantic sturgeon population size is 3,000 for the Hudson River and 1,305 for the Delaware River (Kahnle et al. 2007; O'Leary et al. 2014). The estimated New York Bight DPS overall abundance (juveniles and adults across all subpopulations) is 33,210 fish ((NMFS 2017c; Wirgin et al. 2015).

From our *Effects of the Action* (Section 10), we determined that the Atlantic sturgeon New York Bight DPS would likely be affected by sediment quality degradation resulting from in-water hull cleaning in the port of Philadelphia, Pennsylvania. In particular, it is reasonably likely that Atlantic sturgeon would be exposed to sediments contaminated with copper (and possibly zinc) found in antifouling paints removed from the hulls of inactive ships. Such exposure could contribute to the bioaccumulation of heavy metals in Atlantic sturgeon found in this port. Sublethal effects may include impacts to sensory organs or reproductive impairments.

Only those fish foraging within the immediate vicinity of where in-water hull cleaning is conducted would be affected. Therefore, the number of individual sturgeon that would be adversely affected is expected to be relatively small given the extremely small area affected in relation to available sturgeon habitat, and the expected low density of sturgeon that would be foraging in the affected area. The number of fish that would be exposed and the available foraging habitat area that would be affected are both an extremely small proportion of the Delaware River subpopulation size and riverine habitat, and an even smaller proportion of the New York Bight DPS as a whole. Therefore, we do not expect that the proposed action will reduce appreciably, the likelihood of both the survival and recovery of Atlantic sturgeon New York Bight DPS by reducing their numbers, reproduction, or distribution.

12.1.2 South Atlantic DPS

The South Atlantic DPS of Atlantic sturgeon is listed as endangered and includes ten river systems. Currently, this DPS supports six known spawning subpopulations: Ashepoo, Combahee and Edisto (ACE) Basin, Savannah, Ogeechee, Altamaha, Satilla, and St. Mary's. Atlantic sturgeon are also known to occur in the St Johns River in Florida but spawning has not been confirmed in this river. NMFS (2017c) rated the Savannah and Altamaha subpopulations as health category "high" based on the following health index criteria: average adult survival rate; regular spawning; juveniles present and progressing through age classes; and relatively large estimated effective population sizes. The ACE, Ogeechee, and Satilla subpopulations were rated by NFMS as health category “medium” as these systems are somewhat smaller compared to the Altamaha and Savannah, and juvenile progression through age classes has not been confirmed in these systems. There are no known major threats in any of the ten river populations within this DPS; several face minor threats, mainly water quality and bycatch. The estimated South Atlantic DPS overall abundance (juveniles and adults across all subpopulations) is 13,555 fish ((NMFS 2017c; Wirgin et al. 2015).

From our *Effects of the Action* (Section 10), we determined that the Atlantic sturgeon South Atlantic DPS would likely be affected by sediment quality degradation resulting from in-water hull cleaning in the port of Mayport, Florida. In particular, it is reasonably likely that Atlantic sturgeon would be exposed to sediments contaminated with copper (and possibly zinc) found in antifouling paints removed from the hulls of inactive ships. Such exposure could contribute to the bioaccumulation of heavy metals in sturgeon found in this port. Sublethal effects may include impacts to sensory organs or reproductive impairments.

Only those fish foraging within the immediate vicinity of where in-water hull cleaning is conducted would be affected. Therefore, the number of individual sturgeon that would be adversely affected is expected to be very small given the extremely localized nature of this stressor and the expected very low density of this ESA-listed species in the affected area. The density of Atlantic sturgeon in the port of Mayport is expected to be particularly low since the St. Johns River represents the southern extent of this species’ range along the Atlantic coast and spawning is not known to occur in this river. The number of fish that would be exposed and the available foraging habitat area that would be affected are both an extremely small proportion of the South Atlantic DPS as a whole. Therefore, we do not expect that the proposed action will reduce appreciably, the likelihood of both the survival and recovery of Atlantic sturgeon South Atlantic DPS by reducing their numbers, reproduction, or distribution.

12.2 Shortnose Sturgeon

The shortnose sturgeon is listed as endangered and includes 29 identified populations. The largest shortnose sturgeon adult populations are found in the Northeastern rivers: Hudson 56,708 adults (Bain et al. 2007); Delaware 12,047 (ERC 2006); and Saint John’s greater than 18,000 adults (Dadswell 1979). The major anthropogenic stressors that contributed to the sharp decline of sturgeon populations were commercial fisheries, habitat curtailment and alteration from dams and dredging, and degraded water quality. While sturgeon are still at risk from anthropogenic threats, some of the major threats that contributed to sharp population declines in the past have either been eliminated or reduced. Efforts made over the past few decades to reduce the impact of these threats have slowed the rate of decline for many sturgeon populations. Several dams within shortnose sturgeon historic ranges have been removed or naturally breached. Implementation of the Clean Water Act resulted in water quality improvements throughout the species’ range, although many rivers and estuaries occupied by sturgeon are still not meeting their designated use water quality standards. Restrictions on dredging in the upper portions of the Delaware have likely had beneficial impacts on shortnose sturgeon spawning success. It is likely that some current threats to shortnose sturgeon will increase in the future. These include global climate change, population growth, and land use changes. However, it is difficult to predict the magnitude of these threats in the future or their impact on sturgeon populations.

Major threats to shortnose sturgeon, defined as threats that if altered could lead to recovery, are currently identified for four river systems: dams in the Connecticut, Santee, and Cooper Rivers and water quality in the St. Mary's River. One or more minor threats, defined as threats that likely result in a low level of mortality, have been identified for several other river populations. The most prevalent minor threats to shortnose sturgeon are water quality (ten populations), bycatch (eight populations), and power plant impingement/entrainment (six populations) (SSSRT 2010).

Population trend estimates are available for six shortnose sturgeon spawning stocks: St John, Kennebec, Hudson, and Satilla are all decreasing slightly (-1 percent); Delaware and Ogeechee are stable (zero percent). The shortnose sturgeon Status Review Team evaluated the extinction risk for three shortnose subpopulations (Hudson, Cooper, and Altamaha) and concluded that the estimated probability of extinction was zero for all three under the default assumptions, despite the long (100-year) horizon and the relatively high year-to-year variability in fertility and survival rates (SSSRT 2010). NMFS (2017c) assigned the Delaware River shortnose sturgeon subpopulations a health category rating of "high" based on the following health index criteria: regular spawning; juveniles present and progressing through age classes; no major ongoing threats; a few minor threats including water quality and impingement/entrainment; and a stable population trend.

From our *Effects of the Action* (Section 10), we determined that shortnose sturgeon would likely be affected by sediment quality degradation resulting from in-water hull cleaning in the ports of Philadelphia, Pennsylvania and Mayport, Florida. In particular, it is reasonably likely that shortnose sturgeon would be exposed to sediments contaminated with copper found in antifouling paints removed from the hulls of inactive ships. Such exposure could contribute to the bioaccumulation of heavy metals in sturgeon found in these two ports. Sublethal effects may include impacts to sensory organs or reproductive impairments. Only those fish foraging within the immediate vicinity of where in-water hull cleaning is conducted would be affected. Therefore, the number of individual sturgeon that would be adversely affected is expected to be very small given the extremely localized nature of this stressor and the expected low density of this ESA-listed species in the affected area. The number of fish exposed and the available foraging habitat area affected both represent an extremely small proportion of the shortnose sturgeon subpopulations for the Delaware and St Johns rivers, and an even smaller proportion of the species population and foraging habitat as a whole. Therefore, we do not expect that the proposed action will reduce appreciably, the likelihood of both the survival and recovery of shortnose sturgeon by reducing their numbers, reproduction, or distribution.

12.3 Green Sturgeon, Southern DPS

The green sturgeon Southern DPS is listed as threatened. The estimated total number of adults in the Southern DPS population is 1,348 ± 524 (Mora 2015; NMFS 2015b). The final rule listing Southern DPS green sturgeon indicates that the principle factor for the decline in the DPS is the reduction of spawning to a limited area in the Sacramento River caused primarily by impoundments. Green sturgeon also face threats related to water temperature, water flow, and from commercial and recreational bycatch. Climate change has the potential to impact Southern DPS green sturgeon in the future, but it is unclear how changing oceanic, nearshore and river conditions will affect the Southern DPS overall (NMFS 2015b).

From our *Effects of the Action* (Section 10), we determined that green sturgeon would likely be affected by sediment quality degradation resulting from in-water hull cleaning in the port of Bremerton in Puget Sound. In particular, it is reasonably likely that green sturgeon would be exposed to sediments contaminated with copper found in antifouling paints removed from the hulls of inactive ships. Such exposure could contribute to the bioaccumulation of heavy metals in sturgeon found in these two ports. Sublethal effects may include impacts to sensory organs or reproductive impairments.

Only those fish foraging within the immediate vicinity of where in-water hull cleaning is conducted would be affected. We expect the occurrence of Southern DPS green sturgeon in the Port of Bremerton to be a relatively rare event. Therefore, the number of individual sturgeon that would be adversely affected is expected to be very small given the extremely localized nature of this stressor and the expected very low density of this ESA-listed species in the affected area. . The number of fish exposed and the available foraging habitat area affected both represent an extremely small proportion of the DPS population and foraging habitat as a whole. Therefore, we do not expect that the proposed action will reduce appreciably, the likelihood of both the survival and recovery of green sturgeon Southern DPS by reducing their numbers, reproduction, or distribution.

13 CONCLUSION

We find that the Navy's action is **not likely to adversely affect** blue whale, Southern right whale, false killer whale Main Hawaiian islands insular DPS, fin whale, humpback whale Central America DPS and Mexico DPS, North Atlantic right whale, North Pacific right whale, sei whale, sperm whale, Guadalupe fur seal, green sea turtle East Pacific DPS and South Atlantic DPS, loggerhead North Pacific DPS, loggerhead South Atlantic DPS, loggerhead South Pacific DPS, olive ridley: Mexico's Pacific Coast breeding colonies and all other areas, Chinook salmon ESUs (California Coastal, Central Valley spring-run, Lower Columbia River, Sacramento River winter-run, Snake River fall-run, Snake River spring/summer-run, Upper Columbia River spring-run, Upper Willamette River), chum salmon Columbia River ESU, coho salmon ESUs (Central California Coast, Lower Columbia River, Oregon Coast, Southern Oregon/Northern California Coasts), giant manta ray, Nassau grouper, oceanic whitetip shark, scalloped hammerhead shark DPSs (Central and Southwest Atlantic, and Eastern Pacific), smalltooth sawfish U.S. portion of range DPS, largetooth sawfish, sockeye salmon Snake River ESU, steelhead trout DPSs (California Central Valley, Central California Coast, Lower Columbia River, Middle Columbia River, Northern California, Snake River Basin, South-Central California Coast, Southern California, Upper Columbia River, Upper Willamette River), black abalone, white abalone, boulder star coral, elkhorn coral, lobed star coral, mountainous star coral, pillar coral, rough cactus coral, staghorn coral, Gulf of Mexico Bryde's whale (proposed), killer whale Southern Resident DPS, Hawaiian monk seal, green sea turtle Central North Pacific DPS, green sea turtle North Atlantic DPS hawksbill sea turtle, loggerhead sea turtle Northwest Atlantic DPS**,** leatherback sea turtle, Kemp's ridley sea turtle, Atlantic sturgeon Carolina DPS, Atlantic sturgeon Gulf of Maine DPS, Atlantic sturgeon Chesapeake DPS**,** Gulf sturgeon, bocaccio Puget Sound/Georgia Basin DPS, Chinook salmon Puget Sound ESU, chum salmon Hood Canal summer-run ESU, eulachon Southern DPS, sockeye salmon Ozette Lake ESU, steelhead trout Puget Sound DPS, yelloweye rockfish: Puget Sound/Georgia Basin DPS; thus, it is also not likely to jeopardize the continued existence or recovery of these species.

We find that the proposed action is also **not likely to adversely affect designated critical habitat** for North Atlantic right whale, black abalone, elkhorn coral, staghorn coral, false killer whale Main Hawaiian islands insular DPS, Hawaiian monk seal, killer whale Southern Resident DPS, leatherback sea turtle, loggerhead Northwest Atlantic Ocean DPS, Atlantic sturgeon New York Bight DPS, Gulf sturgeon, bocaccio Puget Sound/Georgia Basin DPS, Chinook salmon Puget Sound ESU, chum salmon Hood Canal summer-run ESU, green sturgeon Southern DPS, steelhead trout Puget Sound DPS, yelloweye rockfish Puget Sound/Georgia Basin DPS; thus, no destruction or adverse modification of the designated critical habitat for these species is anticipated.

After reviewing the current status of the ESA-listed species, the environmental baseline within the action area, the effects of the proposed action, any effects of interrelated and interdependent actions, and cumulative effects, it is NMFS' opinion that the proposed action is **not likely to jeopardize the continued existence of the Atlantic sturgeon, New York Bight DPS**.

After reviewing the current status of the ESA-listed species, the environmental baseline within the action area, the effects of the proposed action, any effects of interrelated and interdependent actions, and cumulative effects, it is NMFS' opinion that the proposed action is **not likely to jeopardize the continued existence of the Atlantic sturgeon, South Atlantic DPS**. Designated critical habitat for this species does not overlap with the action area; therefore, none will be affected.

After reviewing the current status of the ESA-listed species, the environmental baseline within the action area, the effects of the proposed action, any effects of interrelated and interdependent actions, and cumulative effects, it is NMFS' opinion that the proposed action is **not likely to jeopardize the continued existence of the shortnose sturgeon.** No critical habitat has been designated or proposed for this species; therefore, none will be affected.

After reviewing the current status of the ESA-listed species, the environmental baseline within the action area, the effects of the proposed action, any effects of interrelated and interdependent actions, and cumulative effects, it is NMFS' opinion that the proposed action is **not likely to jeopardize the continued existence of the green sturgeon, Southern DPS**.

14 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and Federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. "Take" is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by regulation to include significant habitat modification or degradation that results in death or injury to ESA-listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. NMFS has not yet defined "harass" under the ESA in regulation. However, on December 21, 2016, NMFS issued interim guidance on the term "harass," defining it as an action that "creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering" (NMFS 2016c). For purposes of this consultation, we relied on NMFS' interim definition of harassment to evaluate when the proposed activities are likely to harass ESA-listed species. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Section 7(o)(2) provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA if that action is performed in compliance with the terms and conditions of this ITS.

14.1 Amount or Extent of Take

Section 7 regulations require NMFS to specify the impact of any incidental take of endangered or threatened species; that is, the amount or extent, of such incidental taking on the species (50 C.F.R. §402.14(i)(1)(i)). The amount of take represents the number of individuals that are expected to be taken by the proposed actions. The extent of take represents the "extent of land or marine area that may be affected by an action" and may be used if we cannot assign numerical limits for animals that could be incidentally taken during the course of an action (51 FR 19953). Where it is not practical to quantify the number of individuals that are expected to be taken by the action, a surrogate (e.g., similarly affected species or habitat or ecological conditions) may be used to express the amount or extent of anticipated take (C.F.R. §402.14(i)(1)(i)). A surrogate may be used when the following three conditions are met. The ITS: (i) describes the causal link between the surrogate and take of the listed species, (ii) explains why it is not practical to express the amount or extent of anticipated take or to monitor take-related impacts in terms of individuals of the listed species, and (iii) sets a clear standard for determining when the level of anticipated take has been exceeded.

14.1.1 Incidental Take from the Adverse Effects Resulting from In-Water Hull Cleaning

Take is authorized for the following species identified in our effects analysis as likely to be adversely affected by exposure to contaminated sediments resulting from in-water hull cleaning of inactive Navy ships: Atlantic sturgeon (New York Bight and South Atlantic DPSs), shortnose sturgeon, and green sturgeon Southern DPS. We anticipate this take will be in the form of nonlethal harm or injury to individuals resulting from the effects of exposure to sediments contaminated with copper and possibly other metals found on hull paints. The Navy ports where in-water hull cleaning of inactive ships would be conducted represent the extent of the marine areas within which ESA-listed species would be taken as a result of exposure to contaminated sediments. In general, we would expect low densities of ESA-listed species in these heavily industrialized, high traffic port locations compared to surrounding areas. The best available scientific and commercial information that exists currently does not include species density information for such localized areas for estimating take in terms of numbers of individuals. Therefore, specifying the amount of take in numbers of individuals is not possible. Moreover, monitoring take-related impacts from sediment contamination is not practical due to the following: (1) Sampling sturgeon located in Navy shipyards would not be technically feasible due to the extremely small numbers of sturgeon likely present in these locations, and (2) the highly migratory nature of sturgeon makes it infeasible to attribute elevated contaminant levels to a particular site or activity (i.e., in-water hull cleaning). Since it is not practical to express the amount of anticipated take or to monitor take-related impacts from sediment contamination at origination ports in terms of individuals of the ESA-listed species, we must use a surrogate measure to express the amount or extent of incidental take.

From our *Effects of the Action* (Section 10.2.2), we found that green sturgeon would likely be exposed to the effects of in-water hull cleaning conducted in the Puget Sound port of Bremerton. Atlantic sturgeon from the New York Bight DPS and South Atlantic DPS would likely be exposed to the effects of in-water hull cleaning conducted in the ports of Philadelphia and Mayport, respectively. Shortnose sturgeon would likely be exposed to the effects of in-water hull cleaning conducted in the ports of Philadelphia and Mayport. These port areas, where in-water hull cleaning would be conducted, represent the extent of the marine areas within which ESA-listed species may be taken as a result of exposure to contaminated sediments. We find it reasonable to assume that there is a positive relationship between the extent of bottom area sediments contaminated with copper or other metals found in antifouling paints and the number of individuals of ESA-listed species that will be taken. That is, we anticipate that the larger the contaminated area is, the more individual sturgeon would be exposed. As a surrogate, this measure would be exceeded if the proposed action resulted in sediment contamination in areas beyond the area of influence where in-water hull cleaning is conducted. The area of influence is defined as the area most likely to be impacted by material released during in-water hull cleaning. This area can be determined by linking the particle-tracking model, General NOAA Operational Modeling Environment (GNOME)(NOAA 2018a), to output from hydrodynamics models for the particular water bodies where in-water hull cleaning is conducted (SSC Pacific and NUWC 2016). Therefore, based on the available information and the assumptions stated above, we find a causal link between the area extent and the take of listed species. In addition, we can determine when this surrogate measure has been exceeded (i.e., through sediment contaminant monitoring [see Section 14.4 below]). For the foregoing reasons, the three criteria for using a surrogate have been met, and the anticipated area extent of sediment contamination is a suitable surrogate for specifying the amount or extent of incidental take (Table 32).

Table 32. Surrogate measure of take of green sturgeon Southern DPS, Atlantic sturgeon (New York Bight and South Atlantic DPSs), and shortnose sturgeon expressed as the anticipated area extent of bottom sediment contaminated by in-water hull cleaning as a result of the proposed action.

ESA-listed Species	Surrogate for take expressed as the anticipated area extent of bottom sediment contaminated by in-water hull cleaning conducted as a result of the proposed action.
Green sturgeon Southern DPS	The portion of the Navy port in Bremerton (Puget Sound), Washington that would be used for in-water hull cleaning of inactive Navy ships, including the bottom sediment area most likely to be impacted by material released during hull cleaning (i.e., area of influence).
Atlantic sturgeon New York Bight DPS	The portion of the Navy port in Philadelphia, Pennsylvania that would be used for in-water hull cleaning of inactive Navy ships, including the bottom sediment area most likely to be impacted by material released during hull cleaning (i.e., area of influence).
Atlantic sturgeon South Atlantic DPS	The portion of the Navy port in Mayport, Florida that would be used for in-water hull cleaning of inactive Navy ships, including the bottom sediment area most likely to be impacted by material released during hull cleaning (i.e., area of influence).
Shortnose sturgeon	The portion of the Navy ports in Philadelphia, Pennsylvania and in Mayport, Florida that would be used for in-water hull cleaning of inactive Navy ships, including the bottom sediment area most likely to be impacted by material released during hull cleaning (i.e., area of influence).

14.2 Effects of the Take

In this opinion, we determined that the amount or extent of anticipated take, coupled with other effects of the proposed action, is not likely to result in jeopardy to any of the ESA-listed species or destruction or adverse modification of any ESA designated critical habitat evaluated in this opinion. Refer to Section 12 *Integration and Synthesis* for details regarding how we reached these determinations.

14.3 Reasonable and Prudent Measures

The measures described below are nondiscretionary, and must be undertaken by the Navy so that they become binding conditions for the exemption in section 7(o)(2) to apply. Section 7(b)(4) of the ESA requires that when a proposed agency action is found to be consistent with section 7(a)(2) of the ESA and the proposed action may incidentally take individuals of ESA-listed species, NMFS will issue a statement that specifies the impact of any incidental taking of endangered or threatened species. To minimize such impacts, reasonable and prudent measures, and term and conditions to implement the measures, must be provided. Only incidental take resulting from the agency actions and any specified reasonable and prudent measures and terms and conditions identified in the ITS are exempt from the taking prohibition of section 9(a), pursuant to section 7(o) of the ESA.

"Reasonable and prudent measures" are nondiscretionary measures to minimize the amount or extent of incidental take (50 C.F.R. §402.02). NMFS believes the reasonable and prudent measures described below are necessary and appropriate to minimize the impacts of incidental take on threatened and endangered species.

Reasonable and Prudent Measure #1: The Navy shall implement mitigation measures to reduce the potential for species attached to an inactive ship's hull to be transported and introduced to areas outside of their natural range, and subsequently become established in the new location and potentially impact ESA-listed resources.

Reasonable and Prudent Measure #2: For inactive ship tow scenarios that involve hull cleaning mitigation, the Navy shall, to the maximum extent practicable, use hull cleaning methods that minimize adverse effects on ESA-listed species found at the origination port.

Reasonable and Prudent Measure #3: The Navy shall monitor and report on activities conducted as part of the proposed action to allow us to evaluate the effectiveness of mitigation measures and verify assumptions regarding the risk to ESA-listed species made in the effects analysis of this opinion.

14.4 Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, the Navy must comply with the following terms and conditions, which implement the reasonable and prudent measures described above. These include the take minimization, monitoring and reporting measures required by the section 7 regulations (50 C.F.R. §402.14(i)). These terms and conditions are non-discretionary. If the Navy fails to ensure compliance with these terms and conditions and their implementing reasonable and prudent measures, the protective coverage of section 7(o)(2) may lapse.

The following Term and Condition implements Reasonable and Prudent Measure #1:

1) The Navy shall implement all mitigation measures agreed to during the course of section 7 consultation with NMFS as described in Section 4 of this programmatic opinion.

The following Term and Condition implements Reasonable and Prudent Measure #2:

1) For all inactive ship tow scenarios involving in-water hull cleaning in the origination ports of Philadelphia, Mayport, or Puget Sound, the Navy shall, to the maximum extent practicable, use in-water hull cleaning methods that minimize impacts to surrounding water bodies by collecting organisms, paint particles and other materials dislodged by the cleaning process for upland disposal (McClay et al. 2015).

The following Terms and Conditions implement Reasonable and Prudent Measure #3:

1) For all inactive ship tow scenarios involving in-water hull cleaning in the origination ports of Philadelphia, Mayport, or Puget Sound, the Navy shall monitor water and sediment quality in the area of influence where the impacts of this activity may occur. This term and condition is necessary to evaluate whether the adverse effects of in-water hull cleaning on ESA-listed species exceed the anticipated adverse effects analyzed for this programmatic opinion.

   Water and Sediment Quality Monitoring Protocols
   At a minimum, four water and sediment quality monitoring sampling events shall be conducted: (1) before hull cleaning to establish a baseline, (2) during active hull cleaning, (3) at the end of hull cleaning, and (4) one month after hull cleaning. The water and sediment quality monitoring methods used and report format shall follow those used for monitoring of biofouling removal from the inactive Navy ship ex-Independence. For details see Water Quality Monitoring of Biofouling Removal from the ex-USS Independence (NUWC 2017b) and Biofouling Removal from the ex-INDEPENDENCE (CV 62) Moored in Sinclair Inlet, Puget Sound, WA: Sediment Monitoring Report (Johnston et al. 2018). Pre-monitoring activities shall include identification of the “area of influence” which is defined as the area most likely to be impacted by material released during biofouling removal. As was done for the ex-Independence water quality monitoring, the area of influence can be determined by linking the particle-tracking model, General NOAA Operational Modeling Environment (GNOME)(NOAA 2018a), to output from hydrodynamics models for the particular water bodies where in-water hull cleaning is conducted (SSC Pacific and NUWC 2016).

   In addition to the water quality parameters monitored for hull cleaning the ex-Independence, water and sediment quality monitoring for this term and condition shall address concerns that in-water hull cleaning may result in the resuspension of contaminants (e.g., PCBs and mercury) in bottom sediments where inactive Navy ships are docked. Contaminant resuspension monitoring shall include, but is not necessarily limited to, (1) monitoring bottom sediments within the area of influence at different time periods (i.e. before, during, and after in-water hull cleaning) to determine the types and levels of contaminants present, (2) water quality monitoring that includes monitoring the levels of contaminants (i.e., those found in sediment samples) at varying depths within the water column and at different time periods (i.e. before, during, and after in-water hull cleaning), and (3) an overall evaluation of the extent to which resuspension of contaminants is occurring as a result of in-water hull cleaning activities, including the specific contaminants that are being resuspended in the water column, and the temporal and spatial extent of occurrence.

For any inactive Navy ship last painted prior to 1985 that is located in the origination ports of Philadelphia, Mayport, or Puget Sound, water quality monitoring for this term and condition shall specifically include TBT monitoring. TBT monitoring is needed to address concerns that in-water hull cleaning may result in the release of TBT, which still may be present on hulls last painted prior to 1985, into the water column and bottom sediments. TBT monitoring would not be required for any ship decommissioned in the year 2000 or later, since such vessels would have been active for 15 years since the Navy prohibited the use of paints containing TBT, and would have subsequently been repainted with non-TBT paints. TBT monitoring would also not be required for any ship which the Navy has documentation indicating hull paints last used on that particular ship did not contain TBT.

Level of Monitoring Required

There are many factors that can influence the impact that in-water hull cleaning activities may have on sediment and water quality. These include environmental variables (e.g., port location, water chemistry, time of year, water temperature, salinity, tides, and current), ship specific variables (e.g., ship size and dimensions, degree of hull fouling, access to niche areas, age of paint, and type of paint), and the particular hull cleaning methods employed (e.g., metal brushes versus soft brushes, or use of debris capture technologies). Therefore, the adverse effects on water and sediment quality may differ greatly for different in-water hull cleaning events.

Considering the multiple factors involved, initially a high level of monitoring is needed to evaluate whether the adverse effects of in-water hull cleaning on ESA-listed species exceed the anticipated adverse effects analyzed for this programmatic opinion. Thus, this term and condition requires the Navy to initially monitor water and sediment quality each time in-water hull cleaning is conducted on an inactive ship in the origination ports of Philadelphia, Mayport, or Puget Sound. Over the course of the Navy's inactive ship tow program, this level of monitoring will result in the accumulation of a large volume of information about the effects of in-water hull cleaning on water and sediment quality under different environmental conditions, ship specific variables, and hull cleaning methods. This cumulative monitoring information can then be used to evaluate whether the initial level of monitoring is still warranted for all scenarios. Monitoring results may indicate that the effects on water and sediment quality are fairly consistent across in-water hull cleaning events under similar conditions. For example, an evaluation of multiple monitoring reports may indicate that the effects of in-water hull cleaning inactive ships last painted within ten years, in the port of Philadelphia, during winter months are relatively minor and short-term. Similarly, monitoring results may indicate that the effects on water and sediment quality are negligible when using an effective debris capture technology for in-water hull cleaning small to medium sized vessels.

Based on an analysis of the cumulative monitoring information, the Navy may propose particular in-water hull cleaning scenarios for which the level of water and sediment quality monitoring could be reduced or for which monitoring would no longer be required. Proposals for reducing or eliminating the monitoring requirement for particular scenarios shall be submitted by the Navy to NMFS in writing as part of the triennial inactive ship tow program report (see Term and Condition #3 below). NMFS will evaluate all such proposals for approval/disapproval and respond to the Navy's in writing within two months of receiving the triennial report. Changes in the monitoring levels required by this term and condition that are approved by NMFS through this process would not necessarily require reinitiation of formal consultation.

2) The Navy shall prepare and submit a water and sediment quality monitoring report to the NMFS Interagency Cooperation Division within twelve months of conducting water and sediment quality monitoring as described in Term and Condition #1 above. As needed, the Navy can request an extension to this timeline from NMFS on a case-by-case basis.

3) The Navy shall prepare and submit a triennial (i.e., every three years) inactive ship tow program report to the NMFS Interagency Cooperation Division summarizing Navy inactive ship tow program activities covered by this opinion, including the mitigation measures agreed to during the course of this section 7 programmatic consultation. The triennial report shall include a discussion of the Navy's implementation of the nondiscretionary Terms and Conditions which implement the Reasonable and Prudent Measures, and implementation of any discretionary Conservation Recommendations (Section 15 below). Any Navy proposals for reducing or eliminating the monitoring requirement for particular in-water hull cleaning scenarios shall also be included in the triennial reports (see Term and Condition #1, *Level of Monitoring Required*). In addition, the triennial report shall include a section describing any progress made by the Navy in the area of in-water hull cleaning capture technologies (e.g., testing results, planned future testing, implementation schedule etc.). The first report, which will cover activities from the completion date of this consultation through December 31, 2021, shall be submitted to the Interagency Cooperation Division by July 1, 2022. Subsequent triennial program reports shall be due on July 1 every three years after the first report (i.e., 2025, 2028, 2031… etc.).

15 Conservation Recommendations

Section 7(a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of threatened and endangered species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on ESA-listed species or designated critical habitat, to help implement recovery plans or develop information (50 C.F.R. §402.02).

In order for NMFS' Office of Protected Resources ESA Interagency Cooperation Division to be kept informed of actions minimizing or avoiding adverse effects on, or benefiting, ESA-listed species or their designated critical habitat, the Navy should notify the NMFS ESA Interagency Cooperation Division of any conservation recommendations they implement in their final action.

1. We recommend that, as practicable, the Navy dry dock <u>all</u> inactive ships prior to towing to minimize the risk of ESA-listed species and critical habitat being adversely affected by the introduction of a hull fouling NAS and to minimize the potential water and sediment quality impacts from in-water hull cleaning. This recommendation applies to all origination-destination port ship tow scenarios.
2. We recommend that the Navy coordinate with the appropriate state agencies (from both origination and destination port states) and affected Tribes as part of the inactive ship tow planning process. State agencies and Tribal nations may be able to provide new information, local expertise, and site-specific recommendations for minimizing the risks of the proposed action and agreed upon mitigation measures on ESA-listed resources.
3. For <u>all</u> inactive ship tow scenarios involving hull cleaning mitigation (either in-water or in dry dock), we recommend the Navy conduct biological surveys to document the extent of biofouling on all underwater components (i.e., hull and niche areas listed above to the maximum extent practicable) of the inactive ship. This recommendation would support evaluation of assumptions made in this opinion regarding the effectiveness of hull cleaning as a mitigation measure for reducing the risk of a NAS introduction. Biological surveys to determine species composition, density, and abundance of fouling organisms can be conducted at different stages including before hull cleaning mitigation, after hull cleaning but prior to departure, and sometime after arrival at the destination port.
4. For all inactive ship tow scenarios that do not involve hull cleaning mitigation, we recommend the Navy conduct biological surveys to document the extent of biofouling on all underwater components of the inactive ship. This recommendation would support evaluation of assumptions made in this opinion regarding the risk of a NAS introduction from ship tow scenarios without hull cleaning mitigation. Biological surveys to determine species composition, density, and abundance of fouling organisms can be conducted prior to departure and sometime after arrival at the destination port.

5. For <u>all</u> inactive ship tow scenarios involving hull cleaning mitigation (either in-water or in dry dock), we recommend the Navy require its contractor to commence towing as soon as practicable, but no later than four weeks, upon completion of mitigation to minimize the attachment of additional hull-fouling organisms to the hull in the period between hull cleaning and ship departure.
6. To minimize the probability of ship strike of ESA-listed species, we recommend that the Navy ensure that the contracted tow companies employ the following measures: (1) Crew members of the tug boats towing the inactive ship will serve as lookouts to avoid potential collisions with ESA-listed species; (2) Strongly encourage towing operators to take the Navy's Marine Species Awareness Training; (3) Whenever marine mammals or sea turtles are sighted, the crew members of the tug boats will increase vigilance and take reasonable and prudent actions to avoid collisions or activities that might result in close interactions between the ships and animals; (4) Any interactions between contracted tug ships and ESA-listed species will be logged by contracted tug operators and reported to the NMFS Office of Protected Resources by the Navy via the triennial reports.
7. For all inactive ship tow scenarios involving towing in the Atlantic Ocean, we recommend the Navy ensure that the towing contractor is familiar with the Right Whale Ship Strike Reduction Rule (50 CFR 224.105) and associated compliance measures. Refer to the NMFS website for details: https://www.fisheries.noaa.gov/national/endangered-species-conservation/reducing-ship-strikes-north-atlantic-right-whales.
8. We recommend that the Navy collaborate with hull coating manufacturers and conduct research on the effectiveness of non-toxic antifouling hull applications including silicon based compounds and epoxy coatings (McClay et al. 2015).
9. We recommend that the Navy continue its research on in-water hull cleaning methods that minimize impacts to surrounding water bodies by collecting organisms, paint particles and other materials dislodged by the cleaning process for upland disposal (McClay et al. 2015).
10. We recommend that the Navy monitor the effects of shipbreaking activities on water quality, and report any potential exposure of ESA-listed resources to contaminants or hazardous materials discharged from the shipbreaking process to the NMFS Office of Protected Resources.
11. We recommend that the Navy adhere to and fully implement, to the maximum extent practicable and permitted by law, all requirements of Executive Order 13112 - Invasive Species[3], as described in Section 2 Federal Agency Duties (64 FR 6183). This includes

the following subsections: Section 2(a)(2)(i) *prevent the introduction of invasive species;* Section 2(a)(2)(v) *conduct research on invasive species and develop technologies to prevent introduction and provide for environmentally sound control of invasive species;* Section 2(a)(3) *not authorize, fund, or carry out actions that it believes are likely to cause or promote the introduction or spread of invasive species in the United States or elsewhere unless, pursuant to guidelines that it has prescribed, the agency has determined and made public its determination that the benefits of such actions clearly outweigh the potential harm caused by invasive species; and that all feasible and prudent measures to minimize risk of harm will be taken in conjunction with the actions* (61 FR 6183).

16 Reinitiation Notice

This concludes formal consultation on the Navy's program for towing inactive ships from their existing berths to dismantling facilities or other inactive ship sites. As 50 C.F.R. §402.16 states, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if:

(1) The amount or extent of taking specified in the ITS is exceeded;

(2) New information reveals effects of the agency action that may affect ESA-listed species or critical habitat in a manner or to an extent not previously considered;

(3) The identified action is subsequently modified in a manner that causes an effect to ESA-listed species or designated critical habitat that was not considered in this opinion. This may include, but is not limited to, differences between actual towing frequencies for each inactive ship tow scenario and the anticipated (or estimated) towing frequencies (from) which the effects analysis for this opinion was based on; or

(4) A new species is listed or critical habitat designated under the ESA that may be affected by the action.

17 APPENDIX: SEASONAL SEA SURFACE TEMPERATURE AND SALINITY AT DIFFERENT LOCATIONS ALONG INACTIVE SHIP TOW ROUTES

The tables below were provided by the Navy as supplemental information to be used by NMFS in the effects analysis for this opinion.

Table 33. East Coast Possible Tow Scenarios: Seasonal Sea Surface Temperature and Salinity Changes (Navy 2018d).

Location	December-February Season			June-August Season			Yearly Average		
Name	Temperature[1]		Salinity[2]	Temperature		Salinity	Temperature		Salinity
	°C	°F	(ppt)	°C	°F	(ppt)	°C	°F	(ppt)
Gulf of Mexico	26.4	79.5	36.2	29.6	85.2	36.4	27.8	82.0	36.3
Brownsville, TX	19.6	67.3	35.8	28.6	83.4	36.4	24.7	76.5	35.7
New Orleans, LA[3]	17.4	63.4	0.20	29.1	84.4	0.20	23.5	74.2	0.21
Straits of Florida	25.6	78.1	36.0	29.1	84.4	36.0	27.3	81.1	36.0
Hatteras, NC	19.9	67.8	36.5	26.5	79.7	36.0	23.2	73.7	36.4
Philadelphia, PA [4, 5]	4.3	39.8	0.25	22.8	73.0	0.25	13.7	56.7	0-0.5

Notes:

- Seasons are calculated monthly averages
- Bold locations are ports locations, while non-bold numbered locations are representative offshore locations
- Location numbers correspond to numbers in tow route figures

Temperature data source: NASA Goddard Space Flight Center, Ocean Ecology Laboratory, Ocean Biology Processing Group; (2014): MODIS-Aqua Ocean Color Data; http://dx.doi.org/10.5067/AQUA/MODIS_OC.2014.0. Ocean Color Data, NASA OB.DAAC. Dataset accessed [2018-07-16]

[2] Salinity data source (unless otherwise noted)**:** Frank Wentz, Simon Yueh, and Gary Lagerloef. 2014. Aquarius Level 3 Sea Surface Salinity Standard Mapped Image Monthly Data V3.0. Ver. 3.0. PO.DAAC, CA, USA. Dataset accessed [2018-07-16]

[3] Closest data available was along the Mississippi River next to Baton Rouge. Salinity may differ closer to the port. (Source: https://waterdata.usgs.gov/nwis/monthly?referred_module=sw&site_no=07374000&por_07374000_61169=197857,00480,61169,2004-10,2015-06&format=html_table&date_format=YYYY-MM-DD&rdb_compression=file&submitted_form=parameter_selection_list)

[4] Average of salinity range

[5] Salinity values range from 0- >18 ppt throughout the Delaware River and 0- 0.5 in the upper portion near Philadelphia (Sources: http://www.delawareestuary.org/wp-content/uploads/2018/01/Chp3-water-quality.pdf; http://www.nap.usace.army.mil/Portals/39/docs/Civil/Deepening/Environmental/Hydrodynamic%20and%20Salinity%20Modeling.pdf)

Table 34. West Coast Possible Tow Scenarios through the Panama Canal: Seasonal Sea Surface Temperature and Salinity Changes (Navy 2018d).

Location	December-February Season			June-August Season			Yearly Average		
Name	Temperature[1]		Salinity[2]	Temperature		Salinity	Temperature		Salinity
	°C	°F	(ppt)	°C	°F	(ppt)	°C	°F	(ppt)
Bremerton, WA	8.1	46.5	31.4	10.5	51.0	31.6	9.7	49.5	31.7
San Diego, CA	15.2	59.3	34.2	22.2	72.0	32.4	17.6	63.6	33.6
Henderson Seamount	19.4	67.0	33.8	21.8	71.2	33.9	20.4	68.8	33.9
Pearl Harbor, HI	24.3	75.7	34.5	27.0	80.6	34.3	25.9	78.5	34.3
Albatross Plateau	29.0	84.1	34.2	29.4	84.9	33.9	29.3	84.8	34.1
Guardian Seamount	24.8	76.6	34.9	28.1	82.5	33.9	26.4	79.5	34.2
Panama Canal [3]	26.7	80.1	0.0	28.5	83.3	0.0	27.8	82.0	0.03
Gulf of Mexico	26.4	79.5	36.2	29.6	85.2	36.4	27.8	82.0	36.3
Brownsville, TX	19.6	67.3	35.8	28.6	83.4	36.4	24.7	76.5	35.7
New Orleans, LA [4]	17.4	63.4	0.2	29.1	84.4	0.2	23.5	74.2	0.214
Straits of Florida	25.6	78.1	36.0	29.1	84.4	36.0	27.3	81.1	36.0
Hatteras, NC	19.9	67.8	36.5	26.5	79.7	36.0	23.2	73.7	36.4
Philadelphia, PA [5]	4.3	39.8	0.255	22.8	73.0	0.25	13.7	56.7	0-0.56

Notes:

- Seasons are calculated monthly averages
- Bold locations are ports locations, while non-bold numbered locations are representative offshore locations
- Location numbers correspond to numbers in tow route figures

Temperature data source: NASA Goddard Space Flight Center, Ocean Ecology Laboratory, Ocean Biology Processing Group; (2014): MODIS-Aqua Ocean Color Data; http://dx.doi.org/10.5067/AQUA/MODIS_OC.2014.0. Ocean Color Data, NASA OB.DAAC. Dataset accessed [2018-07-16]

[2] Salinity data source (unless otherwise noted)**:** Frank Wentz, Simon Yueh, and Gary Lagerloef. 2014. Aquarius Level 3 Sea Surface Salinity Standard Mapped Image Monthly Data V3.0. Ver. 3.0. PO.DAAC, CA, USA. Dataset accessed [2018-07-16]

[3] Salinities range from 0-29 ppt throughout the Panama Canal Locks (Sources: http://www.dtic.mil/dtic/tr/fulltext/u2/a378475.pdf; http://people.wku.edu/charles.smith/biogeog/HILD1939.htm; https://link.springer.com/article/10.1007%2FBF00354602)

[4] Closest data available was along the Mississippi River next to Baton Rouge. Salinity may differ closer to the port. (Source: https://waterdata.usgs.gov/nwis/monthly?referred_module=sw&site_no=07374000&por_07374000_61169=197857,00480,61169,2004-10,2015-06&format=html_table&date_format=YYYY-MM-DD&rdb_compression=file&submitted_form=parameter_selection_list)

[5] Average of salinity range [6] Salinity values range from 0- >18 ppt throughout the Delaware River and 0- 0.5 in the upper portion near Philadelphia (Sources: http://www.delawareestuary.org/wp-content/uploads/2018/01/Chp3-water-quality.pdf; http://www.nap.usace.army.mil/Portals/39/docs/Civil/Deepening/Environmental/Hydrodynamic%20and%20Salinity%20Modeling.pdf

Table 35. West Coast Possible Tow Scenarios through the Strait of Magellan: Seasonal Sea Surface Temperature and Salinity Changes (Navy 2018d).

Location	December-February Season			June-August Season [6]			Yearly Average		
Name	Temp[1]		Salinity[2]	Temp		Salinity	Temp		Salinity
	°C	°F	(ppt)	°C	°F	(ppt)	°C	°F	(ppt)
Bremerton, WA	8.1	46.5	31.4	10.5	51.0	31.6	9.7	49.5	31.7
San Diego, CA	15.2	59.3	34.2	22.2	72.0	32.4	17.6	63.6	33.6
Henderson Seamount	19.4	67.0	33.8	21.8	71.2	33.9	20.4	68.8	33.9
Pearl Harbor, HI	24.3	75.7	34.5	27.0	80.6	34.3	25.9	78.5	34.3
Albatross Plateau	29.0	84.1	34.2	29.4	84.9	33.9	29.3	84.8	34.1
Guardian Seamount	24.8	76.6	34.9	28.1	82.5	33.9	26.4	79.5	34.2
Nazca Ridge	20.2	68.4	35.2	17.7	63.9	35.4	18.7	65.6	35.4
Strait of Magellan	9.6	49.3	32.4	4.4[6]	40.0	34.8	7.0	44.6	33.5
Vema Seachannel	23.5	74.4	36.1	19.1	66.5	36.3	21.0	69.7	36.1
Romanche Fracture Zone	27.1	80.7	36.0	26.3	79.4	36.3	26.7	80.1	36.1
Guiana Basin	26.7	80.0	36.2	29.4	84.9	36.4	27.6	81.7	34.0
Gulf of Mexico	26.4	79.5	36.2	29.6	85.2	36.4	27.8	82.0	36.3
Brownsville, TX	19.6	67.3	35.8	28.6	83.4	36.4	24.7	76.5	35.7
New Orleans, LA [3]	17.4	63.4	0.20	29.1	84.4	0.20	23.5	74.2	0.21[3]
Straits of Florida	25.6	78.1	36.0	29.1	84.4	36.0	27.3	81.1	36.0
Hatteras, NC	19.9	67.8	36.5	26.5	79.7	36.0	23.2	73.7	36.4
Philadelphia, PA [4, 5]	4.3	39.8	0.25[4]	22.8	73.0	0.25[4]	13.7	56.7	0-0.5[5]

Notes:

- Seasons are calculated monthly averages
- Bold locations are ports locations, while non-bold numbered locations are representative offshore locations
- Location numbers correspond to numbers in tow route figures

Temperature data source: NASA Goddard Space Flight Center, Ocean Ecology Laboratory, Ocean Biology Processing Group; (2014): MODIS-Aqua Ocean Color Data; http://dx.doi.org/10.5067/AQUA/MODIS_OC.2014.0. Ocean Color Data, NASA OB.DAAC. Dataset accessed [2018-07-16]

[2] Salinity data source (unless otherwise noted)**:** Frank Wentz, Simon Yueh, and Gary Lagerloef. 2014. Aquarius Level 3 Sea Surface Salinity Standard Mapped Image Monthly Data V3.0. Ver. 3.0. PO.DAAC, CA, USA. Dataset accessed [2018-07-16]

[3] Closest data available was along the Mississippi River next to Baton Rouge. Salinity may differ closer to the port. (Source: https://waterdata.usgs.gov/nwis/monthly?referred_module=sw&site_no=07374000&por_07374000_61169=197857,00480,61169,2004-10,2015-06&format=html_table&date_format=YYYY-MM-DD&rdb_compression=file&submitted_form=parameter_selection_list)

[4] Average of salinity range

[5] Salinity values range from 0- >18 ppt throughout the Delaware River and 0- 0.5 in the upper portion near Philadelphia (Sources: http://www.delawareestuary.org/wp-content/uploads/2018/01/Chp3-water-quality.pdf; http://www.nap.usace.army.mil/Portals/39/docs/Civil/Deepening/Environmental/Hydrodynamic%20and%20Salinity%20Modeling.pdf)

[6] West coast tows generally do not occur during Southern Hemisphere winter and fall (June-December)

18 References

Academy of Natural Sciences, T. 2006. 2003 Neches River Biological Survey near Beaumont, Texas for Mobil Oil Corporation, DuPont Beaumont and Lower Neches Valley Authority. Patrick Center for Environmental Research, Academy of Natural Sciences of Philadelphia, Philadelphia, PA.

Acevedo-Whitehouse, K., and A. L. J. Duffus. 2009. Effects of environmental change on wildlife health. Philosophical Transactions of the Royal Society of London B Biological Sciences 364(1534):3429-3438.

Adams, P. B., and coauthors. 2007. Population status of North American green sturgeon, Acipenser medirostris. Environmental Biology of Fishes 79(3-4):339-356.

Adams, P. B., C. B. Grimes, S. T. Lindley, and M. L. Moser. 2002. Status review for North American green sturgeon, Acipenser medirostris. National Marine Fisheries Service, Southwest Fisheries Science Center, Santa Cruz, CA.

Akçakaya, H., and W. Root. 2007. RAMAS red list professional. Spatial and temporal data analysis for threatened species classifications under uncertainty. New York, NY: Applied Biomathematics.

Al-Humaidhi, A. 2011. Analysis of green sturgeon bycatch by sector and time in the West Coast Groundfish Fishery. 3pp. Included as Attachment 2 to: National Marine Fisheries Service. 2011. Endangered Species Act Section 7.

Alava, J. J., and coauthors. 2006. Loggerhead sea turtle (*Caretta caretta*) egg yolk concentrations of persistent organic pollutants and lipid increase during the last stage of embryonic development. Science of the Total Environment 367(1):170-181.

Altiok, A. Almaz, and A. Ghafoori. 2012. Modeling and Analysis of Vessel Traffic in Delaware River and Bay Area: Risk Assessment and Mitigation Study. Rutgers, The State University of New Jersey, Piscataway, NJ 08854.

American Association of Port Authorities. 2016a. U.S. Port Rankings by Cargo Tonnage 2016. AAPA Port Industry Statistics http://www.aapa-ports.org/unifying/content.aspx?ItemNumber=21048.

American Association of Port Authorities. 2016b. World Port Ranking. http://www.aapa-ports.org/unifying/content.aspx?ItemNumber=21048.

Andrews, A. H., and coauthors. 2005. Bomb radiocarbon and lead-radium disequilibria in otoliths of bocacio rockfish (*Sebstes paucispinis*): a determination of age and longevity for a difficult-to-age fish. Marine and Freshwater Research 56:517-528.

Anttila, C. K., C. C. Daehler, N. E. Rank, and D. R. Strong. 1998. Greater male fitness of a rare invader (Spartina alterniflora, Poaceae) threatens a common native (Spartina foliosa) with hybridization. American Journal of Botany 85:1597-1601.

Apte, S., B. S. Holland, L. S. Godwin, and J. P. Gardner. 2000. Jumping ship: a stepping stone event mediating transfer of a non-indigenous species via a potentially unsuitable environment. Biological invasions 2(1):75-79.

AquaMaps. 2016. Reviewed distribution maps for Acipenser medirostris (Green sturgeon), with modelled year 2100 native range map based on IPCC A2 emissions scenario.

ASMFC. 1998. Amendment 1 to the interstate fishery management plan for Atlantic sturgeon.

ASMFC. 2007. Special Report to the ASMFC Atlantic Sturgeon Management Board: Estimation of Atlantic Sturgeon Bycatch in Coastal Atlantic Commercial Fisheries of New England and the Mid-Atlantic Atlantic States Marine Fisheries Commission, Arlington, VA

ASSRT. 2007. Status Review of Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*). Atlantic Sturgeon Status Review Team Report to National Marine Fisheries Service, Northeast Regional Office, Gloucester, MA.

Avens, L., J. C. Taylor, L. R. Goshe, T. T. Jones, and M. Hastings. 2009. Use of skeletochronological analysis to estimate the age of leatherback sea turtles *Dermochelys coriacea* in the western North Atlantic. Endangered Species Research 8(3):165-177.

Azanza-Ricardo, J., and coauthors. 2017. Possible Effect of Global Climate Change on Caretta caretta (Testudines, Cheloniidae) Nesting Ecology at Guanahacabibes Peninsula, Cuba. Chelonian conservation and Biology.

Bagley, D. A., W. E. Redfoot, and L. M. Ehrhart. 2013. Marine turtle nesting at the Archie Carr NWR: Are loggerheads making a comeback? Pages 167 *in* T. Tucker, and coeditors, editors. Thirty-Third Annual Symposium on Sea Turtle Biology and Conservation. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Baltimore, Maryland.

Bailey, K. M., and E. D. Houde. 1989. Predation on eggs and larvae of marine fishes and the recruitment problem. Advances in Marine Biology 25:1-83.

Bain, M. B. 1997. Atlantic and shortnose sturgeons of the Hudson River: common and divergent life history attributes. Pages 347-358 *in* Sturgeon Biodiversity and Conservation. Springer.

Bain, M. B., and coauthors. 2007. Recovery of a US endangered fish. PLoS One 2(1):e168.

Baird, R. W. 2001. Status of killer whales, Orcinus orca, in Canada. Canadian field-naturalist 115(4):676-701.

Baird, R. W., and R. W. Baird. 2006. Killer whales of the world: natural history and conservation. Voyageur Press.

Baker, J. D., A. L. Harting, T. A. Wurth, and T. C. Johanos. 2011. Dramatic shifts in Hawaiian monk seal distribution predicted from divergent regional trends. Marine Mammal Science 27(1):78-93.

Baker, J. D., C. L. Littnan, and D. W. Johnston. 2006. Potential effects of sea level rise on the terrestrial habitats of endangered and endemic megafauna in the Northwestern Hawaiian Islands. Endangered Species Research 2:21-30.

Balazik, M. T., D. J. Farrae, T. L. Darden, and G. C. Garman. 2017. Genetic differentiation of spring-spawning and fall-spawning male Atlantic sturgeon in the James River, Virginia. PLoS One 12(7):e0179661.

Balazik, M. T., G. C. Garman, J. P. Van Eenennaam, J. Mohler, and L. C. Woods III. 2012. Empirical evidence of fall spawning by Atlantic Sturgeon in the James River, Virginia. Transactions of the American Fisheries Society 141(6):1465-1471.

Balazik, M. T., and J. A. Musick. 2015. Dual annual spawning races in Atlantic Sturgeon. PLoS One 10(5):e0128234.

Balazs, G. H. 1991. Research Plan for Marine Turtle Fibropapilloma: Results of a December 1990 Workshop. US Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service [Southwest Fisheries Science Center].

Barbieri, E. 2009. Concentration of heavy metals in tissues of green turtles (*Chelonia mydas*) sampled in the Cananéia Estuary, Brazil. Brazilian Journal of Oceanography 57(3):243-248.

Barco, S., and coauthors. 2016. Loggerhead turtles killed by vessel and fishery interaction in Virginia, USA, are healthy prior to death. Marine Ecology Progress Series 555:221-234.

Bartol, S. M., and D. R. Ketten. 2006. Turtle and tuna hearing. Pages 98-105 *in* Y. Swimmer, and R. Brill, editors. Sea turtle and pelagic fish sensory biology: Developing techniques to reduce sea turtle bycatch in longline fisheries, NOAA Technical Memo.

Bartol, S. M., J. A. Musick, and M. Lenhardt. 1999. Auditory evoked potentials of the loggerhead sea turtle (*Caretta caretta)*. Copeia 3:836-840.

Bartron, M., S. Julian, and J. Kalie. 2007. Genetic assessment of Atlantic sturgeon from the Chesapeake Bay: Temporal comparison of juveniles captured in the Bay.

Baulch, S., and C. Perry. 2014. Evaluating the impacts of marine debris on cetaceans. Marine Pollution Bulletin 80(1-2):210-221.

Bax, N., A. Williamson, M. Aguero, E. Gonzalez, and W. Geeves. 2003. Marine invasive alien species: a threat to global biodiversity. Marine Policy 27(4):313-323.

Bayley, P. B. 1995. Understanding large river: floodplain ecosystems. BioScience 45(3):153-158.

Beamer, E. M., B. Hayman, and D. Smith. 2005. Appendix C: Linking freshwater habitat to Skagit Chinook salmon recovery. Skagit River System Cooperative and Washington Department of Fish and Wildlife.

Beauvais, S. L., S. B. Jones, S. K. Brewer, and E. E. Little. 2000. Physiological measures of neurotoxicity of diazinon and malathion to larval rainbow trout (*Oncorhynchus mykiss*) and their correlation with behavioral measures. Environmental Toxicology and Chemistry 19(7):1875-1880.

Benaka, L. R., and T. J. Dobrzynski. 2004. The national marine fisheries service's national bycatch strategy. Marine Fisheries Review 66(2):1-8.

Benson, S. R., and coauthors. 2011. Large-scale movements and high-use areas of western Pacific leatherback turtles, *Dermochelys coriacea*. Ecosphere 2(7):1-27.

Berg, J. J., M. S. Allen, and K. J. Sulak. 2007. Population assessment of the Gulf of Mexico sturgeon in the Yellow River, Florida. Pages 365 *in* American Fisheries Society Symposium. American Fisheries Society.

Bertness, M. D. 1984. Habitat and Community Modification by An Introduced Herbivorous Snail. Ecology 65(2):370-381.

Beverly, S., D. Curran, M. Musyl, and B. Molony. 2009. Effects of eliminating shallow hooks from tuna longline sets on target and non-target species in the Hawaii-based pelagic tuna fishery. Fisheries Research 96(2):281-288.

Billsson, K., L. Westerlund, M. Tysklind, and P. Olsson. 1998. Developmental disturbances caused by chlorinated biphenyls in zebrafish (*Brachydanio rerio*). Marine Environmental Research 46:461-464.

Bjorndal, K. A., and A. B. Bolten. 2010. Hawksbill sea turtles in seagrass pastures: success in a peripheral habitat. Marine Biology 157:135-145.

Bjorndal, K. A., A. B. Bolten, and M. Y. Chaloupka. 2005. Evaluating trends in abundance of immature green turtles, *Chelonia mydas*, in the greater Caribbean. Ecological Applications 15(1):304-314.

Blackburn, T. M., J. L. Lockwood, and P. Cassey. 2015. The influence of numbers on invasion success. Molecular ecology 24(9):1942-1953.

Bledsoe, B. P., and C. C. Watson. 2001. Effects of urbanization on channel instability. Journal of the American Water Resources Association 33(5):1077-1090.

Booth, R. K., J. D. Kieffer, B. L. Tufts, K. Davidson, and A. T. Bielak. 1995. Effects of late-season catch and release angling on anaerobic metabolism, acid-base status, survival, and gamete viability in wild Atlantic salmon (Salmo salar). Canadian Journal of Fisheries and Aquatic Sciences 52(2):283-290.

Boreman, J. 1997. Sensitivity of North American sturgeons and paddlefish to fishing mortality. Environmental Biology of Fishes 48(1-4):399-405.

Breece, M. W., M. J. Oliver, M. A. Cimino, and D. A. Fox. 2013. Shifting Distributions of adult Atlantic Sturgeon amidst post-industrialization and future impacts in the Delaware River: a maximum entropy approach. PLoS One 8(11):e81321.

Brennan, J. S., K. F. Higgins, J. R. Cordell, and V. A. Stamatiou. 2004. Juvenile salmon composition, timing, distribution, and diet in the marine nearshore waters of central Puget Sound in 2001-2002 King County Department of Natural Resources and Parks, Seattle, Washington:164 p.

Britton-Simmons, K. H. 2004. Direct and indirect effects of the introduced alga Sargassum muticum on benthic, subtidal communities of Washington State, USA. Marine Ecology-Progress Series 277:61-78.

Brock, R., J. H. Bailey-Brock, and J. Goody. 1999. A case study of efficacy of freshwater immersion in controlling introduction of alien marine fouling communities: the USS Missouri. Pacific Science 53(3):223.

Brown, J. J., and G. W. Murphy. 2010. Atlantic sturgeon vessel-strike mortalities in the Delaware Estuary. Fisheries 35(2):72-83.

Brundage III, H. M. 2008. Final Report of Investigations of shortnose sturgeon early life stages in the Delaware River, Spring 2007 and 2008.

Buck, K. N., and K. W. Bruland. 2005. Copper speciation in San Francisco Bay: a novel approach using multiple analytical windows. Marine Chemistry 96(1-2):185-198.

Buckley, J., and B. Kynard. 1981. Spawning and rearing of shortnose sturgeon from the Connecticut River. The Progressive Fish-Culturist 43(2):74-76.

Buckley, J., and B. Kynard. 1985. Yearly movements of shortnose sturgeons in the Connecticut River. Transactions of the American Fisheries Society 114(6):813-820.

Busby, P. J., and coauthors. 1996. Status review of steelhead from Washington, Oregon, and California. U.S. Department of Commerce, Northwest Fisheries Science Center, NMFS-NWFSC-27, Seattle, Washington.

Bushaw-Newton, K., and K. Sellner. 1999. NOAA's State of the Coast Report. National Oceanic and Atmospheric Administration.

Byles, R. A. 1988. Behavior and ecology of sea turtles from Chesapeake Bay, Virginia. Dissertation. The College of William and Mary in Virginia.

Cameron, P., J. Berg, V. Dethlefsen, and H. Von Westernhagen. 1992. Developmental defects in pelagic embryos of several flatfish species in the southern North Sea. Netherlands Journal of Sea Research 29(1):239-256.

Campbell, J. G., and L. R. Goodman. 2004. Acute sensitivity of juvenile shortnose sturgeon to low dissolved oxygen concentrations. Transactions of the American Fisheries Society 133(3):772-776.

Candy, J. R., and coauthors. 2015. Population differentiation determined from putative neutral and divergent adaptive genetic markers in Eulachon (Thaleichthys pacificus, Osmeridae), an anadromous Pacific smelt. Molecular Ecology Resources 15(6):1421-1434.

Capper, A., L. J. Flewelling, and K. Arthur. 2013. Dietary exposure to harmful algal bloom (HAB) toxins in the endangered manatee (Trichechus manatus latirostris) and green sea turtle (Chelonia mydas) in Florida, USA. Harmful Algae 28:1-9.

Caretta, J. V., and coauthors. 2005. U.S. Pacific marine mammal stock assessments: 2004. NOAA-TM-NMFS-SWFSC-358.

Carlton, J. T. 1996. Pattern, process, and prediction in marine invasion ecology. Biological Conservation 78(1):97-106.

Carlton, J. T. 1999. Molluscan invasions in marine and estuarine communities. Malacologia 41(2):439-454.

Carlton, J. T., and L. G. Eldredge. 2015. Update and revisions of the marine bioinvasions of Hawai'i: The introduced and cryptogenic marine and estuarine animals and plants of the Hawaiian archipelago. Bish. Mus. Bull. Zool 9:25-47.

Carr, A. 1987. Impact of nondegradable marine debris on the ecology and survival outlook of sea turtles. Marine Pollution Bulletin 18(6):352-356.

Carr, M. H. 1983. Spatial and temporal patterns of recruitment of young-of-the-year rockfishes (Genus *Sebastes*) into a central California kelp forest. San Francisco State University.

Carr, S., F. Tatman, and F. Chapman. 1996. Observations on the natural history of the Gulf of Mexico sturgeon (Acipenser oxyrinchus de sotoi Vladykov 1955) in the Suwannee River, southeastern United States. Ecology of Freshwater Fish 5(4):169-174.

Carretta, J. V., and coauthors. 2017. U.S. Pacific marine mammal stock assessments: 2016, NOAA-TM-NMFS-SWFSC-577.

Carretta, J. V., and coauthors. 2016. U.S. Pacific marine mammal stock assessments: 2015.

Casale, P., and coauthors. 2010. Sea turtle strandings reveal high anthropogenic mortality in Italian waters. Aquatic Conservation: Marine and Freshwater Ecosystems 20(6):611-620.

Casale, P., D. Freggi, and M. Rocco. 2008. Mortality induced by drifting longline hooks and branchlines in loggerhead sea turtles, estimated through observation in captivity. Aquatic Conservation: Marine and Freshwater Ecosystems 18(6):945-954.

Cassoff, R. M., and coauthors. 2011. Lethal entanglement in baleen whales. Diseases of aquatic organisms 96(3):175-185.

Castilla, J. C., N. A. Lagos, and M. Cerda. 2004. Marine ecosystem engineering by the alien ascidian Pyura praeputialis on a mid-intertidal rocky shore. Marine Ecology-Progress Series 268:119–130.

Caut, S., E. Guirlet, and M. Girondot. 2009. Effect of tidal overwash on the embryonic development of leatherback turtles in French Guiana. Marine Environmental Research 69(4):254-261.

Chabrek, R. H. 1972. Vegetation, water and soil characteristics of the Louisiana coastal region, LSU Agricultural Experiment Station Reports 147.

Chaloupka, M., and coauthors. 2008a. Encouraging outlook for recovery of a once severely exploited marine megaherbivore. Global Ecology and Biogeography 17(2):297-304.

Chaloupka, M., P. Dutton, and H. Nakano. 2004. Status of sea turtle stocks in the Pacific. FAO Fisheries Report (738):135-164.

Chaloupka, M., T. M. Work, G. H. Balazs, S. K. Murakawa, and R. Morris. 2008b. Cause-specific temporal and spatial trends in green sea turtle strandings in the Hawaiian Archipelago (1982–2003). Marine Biology 154(5):887-898.

Chambers, R. C., D. D. Davis, E. A. Habeck, N. K. Roy, and I. Wirgin. 2012. Toxic effects of PCB126 and TCDD on shortnose sturgeon and Atlantic sturgeon. Environmental Toxicology and Chemistry 31(10):2324-2337.

Chew, K. K. 1990. Global bivalve shellfish introductions. World Aquaculture 21(3):9-22.

Choi, F. M., C. C. Murray, T. W. Therriault, and E. A. Pakhomov. 2016. Intertidal invasion patterns in Canadian ports. Marine Biology 163(9):183.

Ciguarria, J., and R. Elston. 1997. Independent introduction of Bonamia ostreae, a parasite of Ostrea edulis, to Spain. Diseases of aquatic organisms 29:157-158.

Clarke, D., C. Dickerson, and K. Reine. 2003. Characterization of underwater sounds produced by dredges. Third Specialty Conference on Dredging and Dredged Material Disposal, Orlando, Florida.

Clavero, M., and E. Garcıa-Berthou. 2005. Invasive species are a leading cause of animal extinctions. Trends in Ecology and Evolution 20(3):110.

Clement, D., N. Keeley, and R. Sneddon. 2010. Ecological relevance of Copper (Cu) and Zinc (Zn) in sediments beneath fish farms in New Zealand. Report.

Colautti, R. I., and coauthors. 2003. Spatial and temporal analysis of transoceanic shipping vectors to the Great Lakes. Island Press: Washington DC.

Collins, M. R., C. Norwood, and A. Rourk. 2008. Shortnose and Atlantic sturgeon age-growth, status, diet, and genetics. 2006-0087-009: October 25, 2006-June 1, 2008 final report. South Carolina Department of Natural Resources, Charleston, South Carolina.

Collins, M. R., S. G. Rogers, T. I. Smith, and M. L. Moser. 2000. Primary factors affecting sturgeon populations in the southeastern United States: fishing mortality and degradation of essential habitats. Bulletin of Marine Science 66(3):917-928.

Collins, M. R., S. G. Rogers, and T. I. J. Smith. 1996. Bycatch of sturgeons along the southern Atlantic coast of the USA. North American Journal of Fisheries Management 16(1):24 - 29.

Collins, M. R., and T. I. J. Smith. 1997. Distributions of shortnose and Atlantic sturgeons in South Carolina. American Fisheries Society Symposium 17(4):955-1000.

Collins, M. R. S., T. I. J. 1997. Distributions of shortnose and Atlantic sturgeons in South Carolina. North American Journal of Fisheries Management 17(4):955-1000.

Colwell, R. R. 1996. Global climate and infectious disease: the cholera paradigm. Science 274(5295):2025-2031.

Command, N. S. S. 2006. Naval Ship's Technical Manual (NSTM), Chapter 81, Waterborne Underwater Hull Cleaning of Navy Ships. Naval Sea Systems Command.

Commander Navy Region Hawaii. 2018. Port and Air Operations. https://cnic.navy.mil/regions/cnrh/om/port_and_air_operations.html.

Conant, T. A., and coauthors. 2009. Loggerhead sea turtle (*Caretta caretta*) 2009 status review under the U.S. Endangered Species Act. National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Conn, P. B., and G. K. Silber. 2013. Vessel speed restrictions reduce risk of collision-related mortality for North Atlantic right whales. Ecosphere 4(4):art43.

Connor, T. O., D. Sullivan, M. Clar, and B. Barfield. 2003. Considerations in the design of treatment best management practices (BMPs) to improve water quality. Proceedings of the Water Environment Federation 2003(4):1186-1205.

Cope, W., and coauthors. 2011. Assessing water quality suitability for shortnose sturgeon in the Roanoke River, North Carolina, USA with an in situ bioassay approach. Journal of Applied Ichthyology 27(1):1-12.

Cox, T. M., and coauthors. 2006. Understanding the impacts of anthropogenic sound on beaked whales. Journal of Cetacean Research and Management 7(3):177-187.

Craig, J. K., and coauthors. 2001. Ecological effects of hypoxia on fish, sea turtles, and marine mammals in the northwestern Gulf of Mexico. American Geophysical Union, Washington, D.C.

Crooks, J. A. 1998. Habitat alteration and community-level effects of an exotic mussel, Musculista senhousia. Marine Ecology-Progress Series 162:137-152.

Dadswell, M. J. 1979. Biology and population characteristics of the shortnose sturgeon *Acipenser brevirostrum* LeSueur 1818 (Osteichthyes: Acipenseridae), in the Saint John River Estuary, New Brunswick, Canada. Canadian Journal of Zoology 57(11):2186-2210.

Dadswell, M. J. 2006. A review of the status of Atlantic sturgeon in Canada, with comparisons to populations in the United States and Europe. Fisheries 31(5):218-229.

Dadswell, M. J., B. D. Taubert, T. S. Squiers, D. Marchette, and J. Buckley. 1984. Synopsis of biological data on shortnose sturgeon, *Acipenser brevirostrum* LeSueur 1818. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Seattle, Washington.

Daehler, C. C., and D. R. Strong. 1996. Status, prediction and prevention of introduced cordgrass Spartina spp. invasions in PAcific estuaries, USA. Biological Conservation 78:51-58.

Dagorn, L., K. N. Holland, V. Restrepo, and G. Moreno. 2013. Is it good or bad to fish with FADs? What are the real impacts of the use of drifting FADs on pelagic marine ecosystems? Fish and Fisheries 14(3):391-415.

Daoust, P.-Y., E. L. Couture, T. Wimmer, and L. Bourque. 2017. Incident Report: North Atlantic Right Whale Mortality Event in the Gulf of St. Lawrence, 2017. Collaborative Report Produced by: Canadian Wildlife Health Cooperative, Marine Animal Response Society, and Fisheries and Oceans Canada., http://www.cwhc-rcsf.ca/docs/technical_reports/Incident%20Report%20Right%20Whales%20EN.pdf.

Davenport, J., J. Wrench, J. McEvoy, and V. Carnacho-Ibar. 1990. Metal and PCB concentrations in the "Harlech" leatherback. Marine Turtle Newsletter 48:1-6.

Davidson, I., M. Sytsma, and G. Ruiz. 2009. Ship fouling: a review of an enduring worldwide vector of nonindigenous species. Aquatic Bioinvasions Research and Policy Institute.

Prepared for the California State Lands Commission, Marine Invasive Species Program. 47pp.

Davidson, I. C., L. D. McCann, P. W. Fofonoff, M. D. Sytsma, and G. M. Ruiz. 2008a. The potential for hull-mediated species transfers by obsolete ships on their final voyages. Diversity and Distributions 14(3):518-529.

Davidson, I. C., L. D. McCann, M. D. Sytsma, and G. M. Ruiz. 2008b. Interrupting a multi-species bioinvasion vector: The efficacy of in-water cleaning for removing biofouling on obsolete vessels. Marine Pollution Bulletin 56(9):1538-1544.

Day, R. D., A. L. Segars, M. D. Arendt, A. M. Lee, and M. M. Peden-Adams. 2007. Relationship of blood mercury levels to health parameters in the loggerhead sea turtle (Caretta caretta). Environmental Health Perspectives 115(10):1421.

de Stephanis, R., J. Giménez, E. Carpinelli, C. Gutierrez-Exposito, and A. Cañadas. 2013. As main meal for sperm whales: Plastics debris. Marine Pollution Bulletin 69(1-2):206-214.

Deng, X. 2000. Artificial reproduction and early life stages of the green sturgeon (Acipenser medirostris).

Denkinger, J., and coauthors. 2013. Are boat strikes a threat to sea turtles in the Galapagos Marine Reserve? Ocean & Coastal Management 80:29-35.

Derraik, J. G. B. 2002. The pollution of the marine environment by plastic debris: a review. Marine Pollution Bulletin 44(9):842-852.

DeRuiter, S. L., and K. L. Doukara. 2012. Loggerhead turtles dive in response to airgun sound exposure. Endangered Species Research 16:55-63.

Dickerson. 2006. Observed takes of sturgeon and turtles from dredging operations along the Atlantic Coast.

Dickerson, D., and coauthors. 2007. Effectiveness of relocation trawling during hopper dredging for reducing incidental take of sea turtles. Pages 509-530 *in* R. E. Randell, editor 2007 World Dredging Conference, Lake Buena Vista, Florida.

Didham, R. K., J. M. Tylianakis, N. J. Gemmell, T. A. Rand, and R. M. Ewers. 2007. Interactive effects of habitat modification and species invasion on native species decline. Trends Ecol Evol 22(9):489-496.

Dietrich, K. S., V. R. Cornish, K. S. Rivera, and T. A. Conant. 2007. Best practices for the collection of longline data to facilitate research and analysis to reduce bycatch of protected species. NOAA Technical Memorandum NMFS-OPR-35. 101p. Report of a workshop held at the International Fisheries Observer Conference Sydney, Australia, November 8,.

Dionne, P. E., G. B. Zydlewski, M. T. Kinnison, J. Zydlewski, and G. S. Wippelhauser. 2013. Reconsidering residency: characterization and conservation implications of complex migratory patterns of shortnose sturgeon (*Acispenser brevirostrum*). Canadian Journal of Fisheries and Aquatic Sciences 70(1):119-127.

Doblin, M. A., and coauthors. 2004. Transport of the Harmful Bloom Alga Aureococcus anophagefferens by Oceangoing Ships and Coastal Boats. Applied and Environmental Microbiology 70(11):6495-6500.

Doney, S. C., and coauthors. 2011. Climate change impacts on marine ecosystems.

Dorn, M. W. 2002. Advice on west coast rockfish harvest rates from Bayesian meta-analysis of stock-recruit relationships. North American Journal of Fisheries Management 22:280-300.

Dovel, W. L., and T. J. Berggren. 1983. Atlantic sturgeon of the Hudson River estuary, New York. New York Fish and Game Journal 30:140-172.

Dow Piniak, W. E., S. A. Eckert, C. A. Harms, and E. M. Stringer. 2012. Underwater hearing sensitivity of the leatherback sea turtle (*Dermochelys coriacea*): Assessing the potential effect of anthropogenic noise. U.S. Dept of the Interior, Bureau of Ocean Energy Management, Headquarters, Herndon, VA.

Drake, J. M., and D. M. Lodge. 2006. Allee effects, propagule pressure and the probability of establishment: risk analysis for biological invasions. Biological invasions 8(2):365-375.

Drake, J. S., and coauthors. 2010. Status review of five rockfish species in Puget Sound, Washington: bocaccio (*Sebastes paucispinis*), canary rockfish (*S. pinniger*), yelloweye rockfish (*S. ruberrimus*), greenstriped rockfish (*S. elongatus*), and redstripe rockfish (*S. proriger*). U.S. Department of Commerce, NOAA Technical Memo, NMFS-NWFSC-108.

Dueñas, M.-A., and coauthors. 2018. The role played by invasive species in interactions with endangered and threatened species in the United States: a systematic review. Biodiversity and Conservation:1-13.

Dukes, J. S., and H. A. Mooney. 1999. Does global change increase the success of biological invaders? Trends Ecol Evol 14(4):135-139.

Dunn, D. C., A. M. Boustany, and P. N. Halpin. 2011. Spatio-temporal management of fisheries to reduce by-catch and increase fishing selectivity. Fish and Fisheries 12(1):110-119.

Dutton, P. H., B. W. Bowen, D. W. Owens, A. Barragan, and S. K. Davis. 1999. Global phylogeography of the leatherback turtle (*Dermochelys coriacea*). Journal of Zoology 248:397-409.

Dutton, P. H., V. Pease, and D. Shaver. 2006. Characterization of mtDNA variation among Kemp's ridleys nesting on Padre Island with reference to Rancho Nuevo genetic stock. Pages 189 *in* M. Frick, A. Panagopoulou, A. F. Rees, and K. Williams, editors. Twenty-Sixth Annual Conference on Sea Turtle Conservation and Biology.

Dwyer, F., and coauthors. 2005. Assessing contaminant sensitivity of endangered and threatened aquatic species: Part III. Effluent toxicity tests. Archives of Environmental Contamination and Toxicology 48(2):174-183.

Echeverria, T. W. 1987. Thirty-four species of California rockfishes: maturity and seasonality of reproduction. Fisheries Bulletin 85(2):229-250.

Eckert, K. L., B. P. Wallace, J. G. Frazier, S. A. Eckert, and P. C. H. Pritchard. 2012. Synopsis of the biological data on the leatherback sea turtle (*Dermochelys coriacea*). U.S. Department of Interior, Fish and Wildlife Service, Biological Technical Publication.

Ehrhart, L. M., D. A. Bagley, and W. E. Redfoot. 2003. Loggerhead turtles in the Alantic Ocean: Geographic distribution, abundance, and population status. Pages 157-174 *in* A. B. Bolten, and B. E. Witherington, editors. Loggerhead Sea Turtles. Smithsonian Institution Press, Washington D.C.

Englund, R. A., and coauthors. 2000. Biodiversity of freshwater and estuarine communities in lower Pearl Harbor, Oahu, Hawaii with observations on introduced species. Bishop Museum Technical Report HBS 2000-001.

EPA. 2010. Climate Change Indicators in the United States: Weather and Climate. Pages 14 *in*. Evironmental Protection Agency.

ERC. 2006. Final report of Shortnose Sturgeon population studies in the Delaware River, January 1999 through March 2003. Environmental Research and Consulting.

Erickson, D., and J. E. Hightower. 2007. Oceanic distribution and behavior of green sturgeon. Pages 197-211 *in* Anadromous Sturgeons: Habitats, Threats, and Management: Proceedings of the Symposium" Anadromous Sturgeons--Status and Trends, Anthropogenic Impacts, and Essential Habitats" Held in Quebec City, Quebec, Canada, August 11-13, 2003. American Fisheries Society.

Eschmeyer, W. N., E. S. Herald, and H. Hammann. 1983. A Field Guide to Pacific Coast Fishes of North America. Houghton Mifflin Company, Boston, Massachusetts.

Farrae, D. J., W. C. Post, and T. L. Darden. 2017. Genetic characterization of Atlantic sturgeon, *Acipenser oxyrinchus oxyrinchus*, in the Edisto River, South Carolina and identification of genetically discrete fall and spring spawning. Conservation Genetics:1-11.

Fay, C., and coauthors. 2006. Status review for anadromous Atlantic salmon (Salmo salar) in the United States.

Feder, H. M., C. H. Turner, and C. Limbaugh. 1974. Observations on fishes associated with kelp beds in southern California. Fisheries Bulletin 160:144.

FFWCC. 2007. Shortnose sturgeon population evaluation in the St. Johns River, FL: has there ever been a shortnose sturgeon population in Florida’s St. Johns River?

Finkbeiner, E. M., and coauthors. 2011. Cumulative estimates of sea turtle bycatch and mortality in USA fisheries between 1990 and 2007. Biological Conservation.

Finneran, J. J. 2015. Noise-induced hearing loss in marine mammals: A review of temporary threshold shift studies from 1996 to 2015. Journal of the Acoustical Society of America 138(3):1702-1726.

Flewelling, L. J., and coauthors. 2005. Brevetoxicosis: red tides and marine mammal mortalities. Nature 435(7043):755.

Floerl, O., and G. J. Inglis. 2003. Boat harbour design can exacerbate hull fouling. Austral Ecology 28(2):116-127.

Fofonoff, P. W., G. M. Ruiz, B. Steves, C. Simkanin, and J. T. Carlton. 2018. National Exotic Marine and Estuarine Species Information System.

Forbes, D. J. 1996. Characteristics and Treatment of Wastewater Generated During Underwater Hull Cleaning Operations of US Navy Ships. DTIC Document.

Ford, M. J., and coauthors. 2011. Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Pacific Northwest. U.S. Department of Commerce, NOAA Technical Memo, NMFS-NWFSC-113.

Foster, A. M., and J. P. Clugston. 1997. Seasonal migration of Gulf sturgeon in the Suwannee River, Florida. Transactions of the American Fisheries Society 126(2):302-308.

Foster, V., R. J. Giesler, A. M. W. Wilson, C. R. Nall, and E. J. Cook. 2016. Identifying the physical features of marina infrastructure associated with the presence of non-native species in the UK. Marine Biology 163(8):173.

Fox, D. A., J. E. Hightower, and F. M. Parauka. 2000. Gulf sturgeon spawning migration and habitat in the Choctawhatchee River system, Alabama–Florida. Transactions of the American Fisheries Society 129(3):811-826.

Fox, D. A., J. E. Hightower, and F. M. Parauka. 2002. Estuarine and nearshore marine habitat use by Gulf sturgeon from the Choctawhatchee River system, Florida. Pages 111-126 *in* American Fisheries Society Symposium.

Francis, C. D. J. R. B. 2013. A framework for understanding noise impacts on wildlife: An urgent conservation priority. Frontiers in Ecology and the Environment 11(6):305-313.

Frazier, J., and coauthors. 1985. Epizoan Communities on Marine Turtles. Marine Ecology 6:127-140.

Frinodig, A., and coauthors. 2009. Review of the California trawl fishery for Pacific ocean shrimp, *Pandalus jordani*, from 1992 to 2007. Marine Fisheries Review 71(2):1-13.

Fritts, M. W., C. Grunwald, I. Wirgin, T. L. King, and D. L. Peterson. 2016. Status and genetic character of Atlantic Sturgeon in the Satilla River, Georgia. Transactions of the American Fisheries Society 145(1):69-82.

Fuentes, M., M. Hamann, and C. J. Limpus. 2009a. Past, current and future thermal profiles of green turtle nesting grounds: Implications from climate change. Journal of Experimental Marine Biology and Ecology 383(1):56-64.

Fuentes, M. M. P. B., M. Hamann, and C. J. Limpus. 2010. Past, current and future thermal profiles of green turtle nesting grounds: Implications from climate change. Journal of Experimental Marine Biology and Ecology 383:56-64.

Fuentes, M. M. P. B., C. J. Limpus, and M. Hamann. 2011. Vulnerability of sea turtle nesting grounds to climate change. Global Change Biology 17:140-153.

Fuentes, M. M. P. B., and coauthors. 2009b. Proxy indicators of sand temperature help project impacts of global warming on sea turtles in northern Australia. Endangered Species Research 9:33-40.

Fujihara, J., T. Kunito, R. Kubota, and S. Tanabe. 2003. Arsenic accumulation in livers of pinnipeds, seabirds and sea turtles: Subcellular distribution and interaction between arsenobetaine and glycine betaine. Comparative Biochemistry and Physiology C Toxicology and Pharmacology 136(4):287-296.

García-Fernández, A. J., and coauthors. 2009. Heavy metals in tissues from loggerhead turtles (*Caretta caretta*) from the southwestern Mediterranean (Spain). Ecotoxicology and Environmental Safety 72(2):557-563.

Gardner, S. C., M. D. Pier, R. Wesselman, and J. A. Juarez. 2003. Organochlorine contaminants in sea turtles from the Eastern Pacific. Marine Pollution Bulletin 46(9):1082-1089.

Gaydos, J. K., and S. Raverty. 2007. Killer Whale Stranding Response, August 2007 Final Report.

Giesy, J. P., J. Newsted, and D. L. Garling. 1986. Relationships between chlorinated hydrocarbon concentrations and rearing mortality of chinook salmon (*Oncorhynchus tshawytscha*) eggs from Lake Michigan. Journal of Great Lakes Research 12(1):82-98.

Gilbert, C. R. 1989. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Mid-Atlantic Bight) : Atlantic and shortnose sturgeons. Coastal Ecology Group, Waterways Experiment Station, U.S. Dept. of the Interior, Fish and

Wildlife Service, Research and Development, National Wetlands Research Center, Vicksburg, MS, Washington, DC.

Gilman, E., and coauthors. 2007. Reducing sea turtle interactions in the Hawaii-based longline swordfish fishery. Biological Conservation 139(1):19-28.

Gilman, E., and coauthors. 2006. Reducing sea turtle by-catch in pelagic longline fisheries. Fish and Fisheries 7(1):2-23.

Gilman, E. L., J. Ellison, N. C. Duke, and C. Field. 2008. Threats to mangroves from climate change and adaptation options: A review. Aquatic Botany 89(2):237-250.

Gitschlag, G. R., B. A. Herczeg, and T. R. Barcak. 1997. Observations of sea turtles and other marine life at the explosive removal of offshore oil and gas structures in the Gulf of Mexico. Gulf Research Reports 9(4):247-262.

Glass, A. H., T. V. N. Cole, and M. Garron. 2010. Mortality and serious injury determinations for baleen whale stocks along the United States and Canadian Eastern Seaboards, 2004-2008. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Glockner-Ferrari, D. A., M. J. Ferrari, and D. McSweeney. 1987. Occurrence of abnormalities, injuries, and strandings of humpback whales in Hawaiian waters. Seventh Biennial Conference on the Biology of Marine Mammals, 5-9 December Miami Florida. p.26.

Godley, B. J., D. R. Thompsonà, and R. W. Furness. 1999. Do heavy metal concentrations pose a threat to marine turtles from the Mediterranean Sea? Marine Pollution Bulletin 38(6):497-502.

Good, T. P., R. S. Waples, and P. Adams. 2005. Updated status of federally listed ESUs of West Coast salmon and steelhead. U.S. Department of Commerce, NMFS-NWFSC-66, Seattle, Washington.

Gordon, A. N., A. R. Pople, and J. Ng. 1998. Trace metal concentrations in livers and kidneys of sea turtles from south-eastern Queensland, Australia. Marine and Freshwater Research 49(5):409-414.

Gorga, C. C. T. 2010. Foraging ecology of green turtles (*Chelonia mydas*) on the Texas coast, as determined by stable isotope analysis. Master’s thesis. Texas A&M University, College Station, Texas.

Grevstad, F. S. 1999. Experimental invasions using biological control introductions: the influence of release size on the chance of population establishment. Biological invasions 1(4):313-323.

Gribsholt, B., and E. Kristensen. 2002. Effects of bioturbation and plant roots on salt marsh biogeochemistry: a mesocosm study. Marine Ecology-Progress Series 241:71–87.

Groombridge, B. 1982. Kemp's ridley or Atlantic ridley, *Lepidochelys kemii* (Garman 1880). Pages 201-208 *in* B. Groombridge, and L. Wright, editors. The IUCN Amphibia, Reptilia Red Data Book.

Grosholz, E. 2002. Ecological and evolutionary consequences of coastal invasions. Trends in Ecology and Evolution 17(1):22-27.

Grout, J. A., C. D. Levings, and J. S. Richardson. 1997. Decomposition rates of purple loosestrife (Lythrum salicaria) and Lyngbyei's sedge (Carexlyngbyei) in the Fraser River Estuary. Estuaries 20(1):96-102.

Grunwald, C., L. Maceda, J. Waldman, J. Stabile, and I. Wirgin. 2008. Conservation of Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus*: delineation of stock structure and distinct population segments. Conservation Genetics 9(5):1111-1124.

Gu, B., D. Schell, T. Frazer, M. Hoyer, and F. Chapman. 2001. Stable carbon isotope evidence for reduced feeding of Gulf of Mexico sturgeon during their prolonged river residence period. Estuarine, Coastal and Shelf Science 53(3):275-280.

Guilbard, F., J. Munro, P. Dumont, D. Hatin, and R. Fortin. 2007. Feeding ecology of Atlantic sturgeon and lake sturgeon co-occurring in the St. Lawrence estuarine transition zone. Pages 85 *in* American Fisheries Society Symposium. American Fisheries Society.

Gulko, D., and K. L. Eckert. 2003. Sea Turtles: An Ecological Guide. Mutual Publishing, Honolulu, Hawaii.

Gulland, F. M., and A. J. Hall. 2007. Is marine mammal health deteriorating? Trends in the global reporting of marine mammal disease. EcoHealth 4(2):135-150.

Gurevitch, J., and D. K. Padilla. 2004. Are invasive species a major cause of extinctions? Trends Ecol Evol 19(9):470-474.

Gustafson, R. G., editor. 2016. Status Review Update of Eulachon (*Thaleichthys pacificus*) Listed under the Endangered Species Act: Southern Distinct Population Segement.

Gustafson, R. G., M. J. Ford, D. Teel, and J. S. Drake. 2010. Status review of eulachon (*Thaleichthys pacificus*) in Washington, Oregon,and California. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center.

Hager, C., J. Kahn, C. Watterson, J. Russo, and K. Hartman. 2014. Evidence of Atlantic Sturgeon spawning in the York River system. Transactions of the American Fisheries Society 143(5):1217-1219.

Hale, E. A., and coauthors. 2016. Abundance Estimate for and Habitat Use by Early Juvenile Atlantic Sturgeon within the Delaware River Estuary. Transactions of the American Fisheries Society 145(6):1193-1201.

Hall, J. W., T. I. J. Smith, and S. D. Lamprecht. 1991. Movements and habitats of shortnose sturgeon, *Acipenser brevirostrum*, in the Savannah River. Copeia 1991(3):695-702.

Hall, M., E. Gilman, H. Minami, T. Mituhasi, and E. Carruthers. 2017. Mitigating bycatch in tuna fisheries. Reviews in Fish Biology and Fisheries:1-28.

Hallegraeff, G., and C. Bolch. 1992. Transport of diatom and dinoflagellate resting spores in ships' ballast water: implications for plankton biogeography and aquaculture. Journal of Plankton Research 14(8):1067-1084.

Hallegraeff, G. M. 1998. Transport of toxic dinoflagellates via ships' ballast water: bioeconomic risk assessment and efficacy of possible ballast water management strategies. Marine Ecology-Progress Series 168:297-309.

Halstead, B. W., P. S. Auerbach, and D. R. Campbell. 1990. A colour atlas of dangerous marine animals. Wolfe Medical Publications Ltd, W.S. Cowell Ltd, Ipswich, England.

Hamer, J. P., I. Lucas, and T. McCollin. 2001. Harmful dinoflagellate resting cysts in ships' ballast tank sediments:

potential for introduction into English and Welsh waters. Phycologia 40(3):246-255.

Hamer, J. P., T. McCollin, and I. Lucas. 2000. Dinofagellate Cysts in Ballast Tank

Sediments: Between Tank Variability. Marine Pollution Bulletin 40(9):731-733.

Hammerschmidt, C. R., M. B. Sandheinrich, J. G. Wiener, and R. G. Rada. 2002. Effects of dietary methylmercury on reproduction of fathead minnows. Environmental science & technology 36(5):877-883.

Hannah, R. W., and S. A. Jones. 2007. Effectiveness of bycath reduction devices (BRDs) in the ocean shrimp (*Pandalus jordani*) trawl fishery. Fisheries Research 85(1-2):217-225.

Hanson, M. B., and coauthors. 2010. Species and stock identification of prey consumed by endangered southern resident killer whales in their summer range. Endangered Species Research 11:69-82.

Hart, B. T., and coauthors. 1991. A review of the salt sensitivity of the Australian freshwater biota. Hydrobiologia 210(1-2):105-144.

Hart, J. L. 1973. Pacific fishes of Canada. Bulletin of the Fisheries Research Board of Canada 180.

Harte Research Institute. 2018. Freshwater Inflows: Salinity. Harte Research Institute for Gulf of Mexico Studies, Texas A&M University, Corpus Christi.

Harvey, C. J., N. Tolimieri, and P. S. Levin. 2006. Changes in body size, abundance, and energy allocation in rockfish assemblages of the Northeast Pacific. Ecological Applications 16(4):1502-1515.

Hastings, M. C., and A. N. Popper. 2005. Effects of sound on fish. California Department of Transportation.

Hatin, D., J. Munro, F. Caron, and R. D. Simons. 2007. Movements, home range size, and habitat use and selection of early juvenile Atlantic sturgeon in the St. Lawrence estuarine transition zone. Pages 129 *in* American Fisheries Society Symposium. American Fisheries Society.

Haupt, T. M., C. L. Griffiths, T. B. Robinson, and A. F. G. Tonin. 2010. Oysters as vectors of marine aliens, with notes on four introduced species associated with oyster farming in South Africa. African Zoology 45(1):52-62.

Hawkes, L. A., A. C. Broderick, H. Godfrey, B. Godley, and M. J. Witt. 2014. The impacts of climate change on marine turtle reproductoin success. Pages 287-310 *in* B. Maslo, and L. Lockwood, editors. Coastal Conservation. Cambridge University Press, Cambridge.

Hawkes, L. A., A. C. Broderick, M. H. Godfrey, and B. J. Godley. 2009. Climate change and marine turtles. Endangered Species Research 7(2):137-154.

Hay, D. E., and P. B. McCarter. 2000. Status of the eulachon *Thaleichthys pacificus* in Canada. Department of Fisheries and Oceans Canada. Canadian Stock Assessment Secretariat, Ottawa, Ontario.

Hayes, G. C., S. Fossette, K. A. Katselidis, G. Schofield, and M. B. Gravenor. 2010. Breeding Periodicity for male sea turtles, operational sex ratios, and implications in the face of climate change. Society for Conservation Biology DOI 10.1111/j.1523-1739.2010.01531.x. OR In Press.

Hayhoe, K., D.J. Wuebbles, D.R. Easterling, D.W. Fahey, S. Doherty, J. Kossin, W. Sweet, R. Vose, and M. Wehner. 2018. Our Changing Climate. Pages 72-144 *in* D. R. Reidmiller, C.W. Avery, D.R. Easterling, K.E. Kunkel, K.L.M. Lewis, T.K. Maycock, and B.C. Stewart, editor. In Impacts, Risks, and Adaptation in the United States: Fourth National

Climate Assessment, Volume II. U.S. Global Change Research Program, Washington, DC.

Hayman, R. A., E. M. Beamer, and R. E. McClure. 1996. FY 1995 Skaig River chinook restoration research. Report by Skagit System Cooperative, La Conner, Washington:54p + Appendices.

Hays, G. C. 2000. The implications of variable remigration intervals for the assessment of population size in marine turtles. J Theor Biol 206(2):221-7.

Hazel, J., and E. Gyuris. 2006. Vessel-related mortality of sea turtles in Queensland, Australia. Wildlife Research 33(2):149-154.

Hazel, J., I. R. Lawler, H. Marsh, and S. Robson. 2007. Vessel speed increases collision risk for the green turtle *Chelonia mydas*. Endangered Species Research 3:105-113.

Hazen, E. L., and coauthors. 2013. Predicted habitat shifts of Pacific top predators in a changing climate. Nature Climate Change 3(3):234.

HCCC. 2005. Hood Canal and Eastern Strait of Juan de Fuca Summer Chum Salmon Recovery Plan.

Healey, M., C. Groot, and L. Margolis. 1991. Life history of chinook salmon (Oncorhynchus tshawytscha). Pacific salmon life histories:313-393.

Healey, M. C. 1991. Life history of Chinook salmon (*Oncorhynchus tshawytscha*). Pages 311-394 *in* C. Groot, and L. Margolis, editors. Pacific salmon life histories. University of British Columbia Press, Vancouver, Canada.

Hecky, R. E., and coauthors. 2004. The nearshore phosphorus shunt: a consequence of ecosystem engineering by dreissenids in the Laurentian Great Lakes. Canadian Journal of Fisheries and Aquatic Sciences 61(7):1285-1293.

Heidt, A., and R. Gilbert. 1979. Movements of shortnose sturgeon, *Acipenser brevirostrum*, in the Altamaha River, Georgia. ASB Bulletin 26.

Heise, R., W. Slack, S. T. Ross, and M. Dugo. 2005. Gulf sturgeon summer habitat use and fall migration in the Pascagoula River, Mississippi, USA. Journal of Applied Ichthyology 21(6):461-468.

Heise, R. J., W. T. Slack, S. T. Ross, and M. A. Dugo. 2004. Spawning and associated movement patterns of Gulf sturgeon in the Pascagoula River drainage, Mississippi. Transactions of the American Fisheries Society 133(1):221-230.

Hendrickson, J., and J. Konwinski. 1998. Seasonal nutrient import-export budgets for the Lower St. Johns River, Florida. St. Johns River Waste Management District, Report prepared for the Florida Department of Environmental Protection Contract No. WM598.

Henry, A. G., and coauthors. 2016. Serious Injury and Mortality Determinations for Baleen Whale Stocks along the Gulf of Mexico, United States East Coast, and Atlantic Canadian Provinces, 2010-2014. Northeast Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, Woods Hole, Massachusetts.

Henry, A. G., and coauthors. 2015. Serious injury determinations for baleen whale stocks along the Gulf of Mexico, United States east coast and Atlantic Canadian provinces, 2009-2013. NOAA Northeast Fisheries Science Center Reference Document.

Henwood, T. A., and W. E. Stuntz. 1987. Analysis of sea turtle captures and mortalities during commercial shrimp trawling. Fishery Bulletin 85:813-817.

Heppell, S. S., and coauthors. 2005. A population model to estimate recovery time, population size, and management impacts on Kemp's ridley sea turtles. Chelonian Conservation and Biology 4(4):767-773.

Heublein, J. C., J. T. Kelly, C. E. Crocker, A. P. Klimley, and S. T. Lindley. 2009. Migration of green sturgeon, Acipenser medirostris, in the Sacramento River. Environmental Biology of Fishes 84(3):245-258.

Hildebrand, J. 2004. Impacts of anthropogenic sound on cetaceans. Unpublished paper submitted to the International Whaling Commission Scientific Committee SC/56 E 13.

Hildebrand, J. A. 2009. Metrics for characterizing the sources of ocean anthropogenic noise. Journal of the Acoustical Society of America 125(4):2517.

Hiruki, L. M., and T. J. Ragen. 1992. A compilation of historical monk seal, *Monachus schauinslandi*, counts. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Holland, B. F., and G. F. Yelverton. 1973. Distribution and biological studies of anadromous fishes offshore North Carolina. Division of Commercial and Sports Fisheries, North Carolina Department of Natural and Economic Resources.

Holt, M. M. 2008. Sound exposure and Southern Resident killer whales (*Orcinus orca*): A review of current knowledge and data gaps. U.S. Department of Commerce, NMFS-NWFSC-89.

Horrocks, J. A., and coauthors. 2001. Migration routes and destination characteristics of post-nesting hawksbill turtles satellite-tracked from Barbados, West Indies. Chelonian Conservation and Biology 4(1):107-114.

Houston, J. J. 1988. Status of green sturgeon, Acipenser medirostris, in Canada. Canadian field-naturalist 102:286-290.

Huey, R. B., and J. G. Kingsolver. 1989. Evolution of thermal sensitivity of ectotherm performance. Trends Ecol Evol 4(5):131-135.

Huff, D. D., S. T. Lindley, B. K. Wells, and F. Chai. 2012. Green sturgeon distribution in the Pacific Ocean estimated from modeled oceanographic features and migration behavior. PLoS One 7(9):e45852.

Huff, J. A. 1975. Life history of Gulf of Mexico sturgeon, Acipenser oxrhynchus desotoi, in Suwannee River, Florida.

Illingworth and Rodkin Inc. 2001. Noise and vibration measurements associated with the pile installation demonstration project for the San Francisco-Oakland Bay Bridge east span, final data report.

Illingworth and Rodkin Inc. 2004. Conoco/Phillips 24-inch steel pile installation – Results of underwater sound measurements. Letter to Ray Neal, Conoco/Phillips Company.

Illingworth, R., and R. Rodkin. 2001. Noise and vibration measurements associated with the pile installation demonstration project for the San Francisco-Oakland Bay Bridge east span. Technical Report of Illingworth & Rodkin, Inc., Petaluma, CA.

Illingworth, R., and R. Rodkin. 2007. Compendium of pile driving sound data. Prepared for the California Department of Transportation, Sacramento, CA.

IPCC. 2014. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva, Switzerland.

ISAB. 2007. Climate change impacts on Columbia River Basin fish and wildlife. Independent Scientific Advisory Board.

Israel, J. A., K. J. Bando, E. C. Anderson, and B. May. 2009. Polyploid microsatellite data reveal stock complexity among estuarine North American green sturgeon (Acipenser medirostris). Canadian Journal of Fisheries and Aquatic Sciences 66(9):1491-1504.

Israel, J. A., J. F. Cordes, M. A. Blumberg, and B. May. 2004. Geographic patterns of genetic differentiation among collections of green sturgeon. North American Journal of Fisheries Management 24(3):922-931.

Jacobsen, J. K., L. Massey, and F. Gulland. 2010. Fatal ingestion of floating net debris by two sperm whales (Physeter macrocephalus). Marine Pollution Bulletin 60(5):765-767.

Jambeck, J. R., and coauthors. 2015. Plastic waste inputs from land into the ocean. Science 347(6223):768-771.

James, M. C., R. A. Myers, and C. A. Ottensmeyer. 2005. Behaviour of leatherback sea turtles, *Dermochelys coriacea*, during the migratory cycle. Proceedings of the Royal Society Biological Sciences Series B 272(1572):1547-1555.

Jenkins, W. E., T. I. J. Smith, L. D. Heyward, and D. M. Knott. 1993. Tolerance of shortnose sturgeon, *Acipenser brevirostrum*, juveniles to different salinity and dissolved oxygen concentrations. Pages 476-484 *in* Annual Conference of the Southeastern Association of Fish and Wildlife Agencies.

Jensen, A. S., and G. K. Silber. 2004. Large whale ship strike database. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.

Jeschke, J. M., and coauthors. 2014. Defining the impact of non-native species. Conservation Biology 28(5):1188-1194.

Johnson, D. W. 2006. Predation, habitat complexity, and variation in density-dependent mortality of temperate reef fishes. Ecology 87:1179-1188.

Johnson, O., and coauthors. 1997. Status review of chum salmon from Washington, Oregon, and California, Seattle, WA.

Johnston, R. K., E. Arias, D. E. Marx Jr., I. Rivera-Duarte, and P. J. Earley. 2018. Biofouling Removal from the ex-INDEPENDENCE (CV 62) Moored in Sinclair Inlet, Puget Sound, WA: Sediment Monitoring Report. Navy, SSC Pacific, San Diego, CA.

Josefsson, M., and B. Andersson. 2011. The environmental consequences of alien species in the Swedish Lakes Mälaren, Hjälmaren,Vänern and Vättern. Ambio 30(8):514-521.

Jute, A., and B. Dunphy. 2017. The potential efficacy and application of freshwater and hypersaline immersion to control the spread of a marine invasive species. Biological invasions 19(4):1137-1141.

Kahn, J., and M. Mohead. 2010. A protocol for use of shortnose, Atlantic, Gulf, and green sturgeons. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.

Kahnle, A. W., K. A. Hattala, and K. A. McKown. 2007. Status of Atlantic sturgeon of the Hudson River Estuary, New York, USA. American Fisheries Society Symposium 56:347-363.

Karl, T. R. 2009. Global climate change impacts in the United States. Cambridge University Press.

Keller, J. M., and coauthors. 2005. Perfluorinated compounds in the plasma of loggerhead and Kemp's ridley sea turtles from the southeastern coast of the United States Environmental Science and Technology 39(23):9101-9108.

Keller, J. M., J. R. Kucklick, M. A. Stamper, C. A. Harms, and P. D. McClellan-Green. 2004. Associations between organochlorine contaminant concentrations and clinical health parameters in loggerhead sea turtles from North Carolina, USA. Environmental Health Perspectives 112(10):1074.

Keller, J. M., P. D. McClellan-Green, J. R. Kucklick, D. E. Keil, and M. M. Peden-Adams. 2006. Effects of organochlorine contaminants on loggerhead sea turtle immunity: Comparison of a correlative field study and *in vitro* exposure experiments. Environmental Health Perspectives 114(1):70-76.

Kelly, J. L. 2012. Identification and quantification of submarine groudwater discharge in the Hawaiian Islands. Dissertation. University of Hawai'i at Mānoa, Honolulu, Hawai'i.

Ketten, D. R. 1995. Estimates of blast injury and acoustic trauma zones for marine mammals from underwater explosions. Pages 391-407 *in* R. A. Kastelein, J. A. Thomas, and P. E. Nachtigall, editors. Sensory Systems of Aquatic Mammals. De Spil Publishers, Woerden.

Kieffer, M., and B. Kynard. 1996a. Spawning of shortnose sturgeon in the Merrimack River, Massachusetts. Transactions of the American Fisheries Society 125:179-186.

Kieffer, M. C., and B. Kynard. 1993. Annual movements of shortnose and Atlantic sturgeons in the Merrimack River, Massachusetts. Transactions of the American Fisheries Society 122(6):1088-1103.

Kieffer, M. C., and B. Kynard. 1996b. Spawning of the shortnose sturgeon in the Merrimack River, Massachusetts. Transactions of the American Fisheries Society 125(2):179-186.

Kiessling, I. 2003. Finding solutions: derelict fishing gear and other marine debris in northern Australia. National Oceans Office.

Killgore, K. J., S. T. Maynord, M. D. Chan, and R. P. Morgan. 2001. Evaluation of Propeller-Induced Mortality on Early Life Stages of Selected Fish Species. North American Journal of Fisheries Management 21(4):947-955.

King, T., B. Lubinski, and A. Spidle. 2001. Microsatellite DNA variation in Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) and cross-species amplification in the Acipenseridae. Conservation Genetics 2(2):103-119.

King, T. L., and coauthors. 2014. A Nuclear DNA Perspective on Delineating Evolutionarily Significant Lineages in Polyploids: The Case of the Endangered Shortnose Sturgeon (*Acipenser brevirostrum*). PLoS One 9(8):e102784.

Knowlton, A. R., and S. D. Kraus. 2001. Mortality and serious injury of northern right whales (*Eubalaena glacialis*) in the western North Atlantic Ocean. Journal of Cetacean Research and Management Special Issue 2:193-208.

Kocan, R., M. Matta, and S. Salazar. 1993. A laboratory evaluation of Connecticut River coal tar toxicity to shortnose sturgeon (*Acipenser brevirostrum*) embryos and larvae. Final Report to the National Oceanic and Atmospheric Administration, Seattle, Washington.

Kraus, S. D., and coauthors. 2005. North Atlantic right whales in crisis. Science 309(5734):561-562.

Kraus, S. D., and coauthors. 2016. Recent Scientific Publications Cast Doubt on North Atlantic Right Whale Future. Frontiers in Marine Science.

Kretzmann, M. B., N. J. Gemmell, and A. Meyer. 2001. Microsatellite analysis of population structure in the endangered Hawaiian monk seal. Conservation Biology 15(2):457-466.

Kretzmann, M. B., and coauthors. 1997. Low genetic variability in the Hawaiian monk seal. Conservation Biology 11(2):482-490.

Kynard, B. 1997. Life history, latitudinal patterns, and status of the shortnose sturgeon, *Acipenser brevirostrum*. Environmental Biology of Fishes 48(1-4):319-334.

Kynard, B., and coauthors. 2016. Life history and status of Shortnose Sturgeon (*Acipenser brevirostrum* LeSueur, 1818). Journal of Applied Ichthyology 32(S1):208-248.

Kynard, B., and M. Horgan. 2002. Ontogenetic behavior and migration of Atlantic sturgeon, *Acipenser oxyrinchus oxyrinchus*, and shortnose sturgeon, *A. brevirostrum*, with notes on social behavior. Environmental Biology of Fishes 63(2):137-150.

Kynard, B., and E. Parker. 2004. Ontogenetic behavior and migration of Gulf of Mexico sturgeon, Acipenser oxyrinchus desotoi, with notes on body color and development. Environmental Biology of Fishes 70(1):43-55.

LA Coast. 2018. The Calcasieu-Sabine Basin. https://lacoast.gov/reports/static/HILCP_3.pdf.

Lafferty, K. D., and A. M. Kuris. 1996. Biological Control of Marine Pests. Ecology 77(7):1989-2000.

Laist, D. W., J. M. Coe, and K. J. O'Hara. 1999. Marine debris pollution. Pages 342-366 *in* J. R. Twiss Jr., and R. R. Reeves, editors. Conservation and Management of Marine Mammals. Smithsonian Institution Press, Washington, D.C.

Laist, D. W., A. R. Knowlton, J. G. Mead, A. S. Collet, and M. Podesta. 2001. Collisions between ships and whales. Marine Mammal Science 17(1):35-75.

Lammers, M. O., A. A. Pack, and L. Davis. 2007. Trends in whale/vessel collisions in Hawaiian waters. IWC Scientific Committee, Anchorage, Alaska.

Lamont, M. M., I. Fujisaki, and R. R. Carthy. 2014. Estimates of vital rates for a declining loggerhead turtle (Caretta caretta) subpopulation: implications for management. Marine Biology 161(11):2659-2668.

Larson, Z. S., and M. R. Belchik. 2000. A preliminary status review of eulachon and Pacific lamprey in the Klamath River Basin. Yurok Tribal Fisheries Program, Klamath, California.

Lassalle, G., P. Crouzet, J. Gessner, and E. Rochard. 2010. Global warming impacts and conservation responses for the critically endangered European Atlantic sturgeon. Biological Conservation 143(11):2441-2452.

Laughlin, J. 2006. Underwater sound levels associated with pile driving at the Cape Dissapointment boat launch facility, wave barrier project. Washington State Parks Wave Barrier Project Underwater Technical Report.

Lee, D. S., and coauthors. 1980. Atlas of North American freshwater fishes. North Carolina State Museum of Natural History Raleigh.

Leroux, R. A., and coauthors. 2012. Re-examination of population structure and phylogeography of hawksbill turtles in the wider Caribbean using longer mtDNA sequences. Journal of Heredity 103(6):806-820.

Levi, F., and P. Francour. 2004. Behavioural response of Mullus surmuletus to habitat modification by the invasive macroalga Caulerpa taxifolia. Journal of fish biology 64:55–64.

Lewison, R., and coauthors. 2013. Fisheries bycatch of marine turtles lessons learned from decades of research and conservation. Pages 329-351 *in* The biology of sea turtles, volume III.

Lilly, E. L., D. M. Kulis, P. Gentien, and D. M. Anderson. 2002. Paralytic shellfish poisoning toxins in France linked to a human-introduced strain of Alexandrium catenella from the western Pacific: evidence from DNA and toxin analysis. Journal of Plankton Research 24(5):443-452.

Lindley, S. T., and coauthors. 2011. Electronic tagging of green sturgeon reveals population structure and movement among estuaries. Transactions of the American Fisheries Society 140(1):108-122.

Lindley, S. T., and coauthors. 2008. Marine migration of North American green sturgeon. Transactions of the American Fisheries Society 137(1):182-194.

Llansó, R. J., and K. Sillett. 2008. Hull biofouling of Suisun Bay reserve fleet vessel Occidental Victory before and after transit from California to Texas. US Maritime Administration. Washington DC.

Lleonart, M., J. Handlinger, and M. Powell. 2003. Spionid mudworm infestation of farmed abalone (Haliotis spp.). Aquaculture 221(1-4):85-96.

Lockwood, B. L., and G. N. Somero. 2011. Invasive and native blue mussels (genus Mytilus) on the California coast: The role of physiology in a biological invasion☆. Journal of Experimental Marine Biology and Ecology.

Lockwood, J. L., P. Cassey, and T. Blackburn. 2005. The role of propagule pressure in explaining species invasions. Trends in Ecology and Evolution 20(5).

Love, M. S., J. E. Caselle, and K. Herbinson. 1998a. Declines in nearshore rockfish recruitment and populations in the southern California Bight as measured by impingement rates in coastal electrical power generating stations. Fishery Bulletin 96(3):492-501.

Love, M. S., J. E. Caselle, and W. Van Buskirk. 1998b. A severe decline in the commercial passenger fishing vessel rockfish (*Sebastes* spp.) catch in the Southern California Bight, 1980-1996. California Cooperative Oceanic Fisheries Investigations Reports 39:180-195.

Love, M. S., and M. Yoklavich. 2008. Habitat characteristics of juvenile cow cod, *Sebastes levis* (Scorpaenidae), in Southern California. Environmental Biology of Fishes 82:195-202.

Love, M. S., M. M. Yoklavich, and L. Thorsteinson. 2002. The Rockfishes of the Northeast Pacific. University of California Press, Berkeley, California.

Lubchenco, J., and coauthors. 2010. Deepwater Horizon/BP oil budget: What happened to the oil? USGS, NMFS, and DOI, editors.

Lutcavage, M., and J. A. Musick. 1985. Aspects of the biology of sea turtles in Virginia. Copeia 1985(2):449-456.

Lutcavage, M. E., P. Plotkin, B. E. Witherington, and P. L. Lutz. 1997. Human impacts on sea turtle survival. Pages 387-409 *in* P. L. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, Florida.

MacCall, A. D. 1996. Patterns of low frequency variability in fish populations of the California Current. California Cooperative Oceanic Fisheries Investigation Report.

MacCall, A. D. 2002. Use of known-biomass production models to determine productivity of west coast groundfish stocks. North American Journal of Fisheries Management 22:272-279.

MacCall, A. D. 2003. Status of bocaccio off California in 2003. Pacific Fishery Management Council, Portland, Oregon.

MacCall, A. D., and X. He. 2002. Status review of the southern stock of bocaccio (*Sebastes paucispinis*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, Santa Cruz Laboratory, Santa Cruz, California.

MacDonald, I. R., and coauthors. 1993. Natural oil slicks in the Gulf of Mexico visible from space. Journal of Geophysical Research 98(C9):16,351-16,364.

MacDougall, A. S., and R. Turkington. 2005. Are invasive species the drivers or passengers of change in degraded ecosystems? Ecology 86(1):42-55.

Macfadyen, G., T. Huntington, and R. Cappell. 2009. Abandoned, lost or otherwise discarded fishing gear. Food and Agriculture Organization of the United Nations (FAO).

MacFarlane, R. B., and E. C. Norton. 2002. Physiological ecology of juvenile chinook salmon (Oncorhynchus tshawytscha) at the southern end of their distribution, the San Francisco Estuary and Gulf of the Farallones, California. Fishery Bulletin 100(2):244-257.

MacLeod, C. D. 2009. Global climate change, range changes and potential implications for the conservation of marine cetaceans: a review and synthesis. Endangered Species Research 7(2):125-136.

Macleod, C. K., and R. S. Eriksen. 2009. A review of the ecological impacts of selected antibiotics and antifoulants currently used in the Tasmanian salmonid farming industry (Marine Farming Phase). Fisheries Research and Development Corporation.

Malecki, L. M., J. R. White, and K. R. Reddy. 2004. Nitrogen and phosphorus flux rates from sediment in the Lower St. Johns River. Journal of Environmental Quality 33(4):1545-1555.

Mangin, E. 1964. Growth in length of three North American Sturgeon: *Acipenser oxyrhynchus*, Mitchill, *Acipenser fulvescens*, Rafinesque, and *Acipenser brevirostris* LeSueur. Limnology 15:968-974.

Mansfield, K. L. 2006. Sources of mortality, movements and behavior of sea turtles in Virginia. The College of William and Mary.

Mantua, N. J., S. R. Hare, Y. Zhang, J. M. Wallace, and R. C. Francis. 1997. A Pacific interdecadal climate oscillation with impacts on salmon production. Bulletin of the american Meteorological Society 78(6):1069-1079.

Martin, K. J., and coauthors. 2012. Underwater hearing in the loggerhead sea turtles (*Caretta caretta*): a comparison of behvioral and auditory evoked potential audiograms. Journal of Experimental Biology 215:3001-3009.

Masuda, A. 2010. Natal Origin of Juvenile Loggerhead Turtles from Foraging Ground in Nicaragua and Panama Estimated Using Mitochondria DNA.

Matheson, F., A. Dugdale, R. Wells, A. Taumoepeau, and J. Smith. 2007. Efficacy of saltwater solutions to kill introduced freshwater species and sterilise freshwater fishing nets. New Zealand Department of Conservation Research and Development Series 259:24.

Mazaris, A. D., A. S. Kallimanis, S. P. Sgardelis, and J. D. Pantis. 2008. Do long-term changes in sea surface temperature at the breeding areas affect the breeding dates and reproduction performance of Mediterranean loggerhead turtles? Implications for climate change. Journal of Experimental Marine Biology and Ecology.

McCarthy, I., and D. Houlihan. 1997. The effect of temperature on protein metabolism in fish: The possible consequences for wild Atlantic salmon(Salmo salar L.) stocks in Europe as a result of global warming. Pages 51-77 *in* Society of Experimental Biology Seminar Series.

McCauley, D. J., and coauthors. 2015. Marine defaunation: animal loss in the global ocean. Science 347(6219):1255641.

McCauley, R. D., and coauthors. 2000. Marine seismic surveys - A study of environmental implications. APPEA Journal:692-708.

McClay, T., C. Zabin, I. Davidson, R. Young, and D. Elam. 2015. Vessel Biofouling Prevention and Management Options Report U.S. Coast Guard Research & Development Center

McClellan, C. M., J. Braun-McNeill, L. Avens, B. P. Wallace, and A. J. Read. 2010. Stable isotopes confirm a foraging dichotomy in juvenile loggerhead sea turtles. Journal of Experimental Marine Biology and Ecology 387:44-51.

McCormick, S. D., J. M. Shrimpton, and J. D. Zydlewski. 1997. Temperature effects on osmoregulatory physiology of juvenile anadromous fish. Pages 279-301 *in* C. M. W. a. D. G. McDonald, editor. Global warming: implications for freshwater and marine fish. Cambridge University Press, Cambridge, United Kingdom.

McCracken, M. L. 2000. Estimation of Sea Turtle Take and Mortality in the Hawaiian Longline Fisheries. National Marine Fisheries Service, Southwest Fisheries Science Center.

Mcdonald, M. A., J. A. Hildebrand, and S. M. Wiggins. 2006. Increases in deep ocean ambient noise in the Northeast Pacific west of San Nicolas Island, California. Journal of the Acoustical Society of America 120(2):711-718.

Mckenna, M. F., D. Ross, S. M. Wiggins, and J. A. Hildebrand. 2012. Underwater radiated noise from modern commercial ships. Journal of the Acoustical Society of America 131(2):92-103.

McKenna, M. F., D. Ross, S. M. Wiggins, and J. A. Hildebrand. 2013. Relationship between container ship underwater noise levels and ship design, operational and oceanographic conditions. Scientific reports 3.

McMahon, C. R., and G. C. Hays. 2006. Thermal niche, large-scale movements and implications of climate change for a critically endangered marine vertebrate. Global Change Biology 12:1330-1338.

McMinn, A., G. Hallegraeff, P. Thomson, A. V. Jenkinson, and H. Heijnis. 1997. Cyst and radionucleotide evidence for the recent introduction of the toxic dinoflagellate Gymnodinium catenatum into Tasmanian waters. Marine Ecology-Progress Series 161:65-172.1.

McQuinn, I. H., and P. Nellis. 2007. An acoustic-trawl survey of middle St. Lawrence Estuary demersal fishes to investigate the effects of dredged sediment disposal on Atlantic sturgeon and lake sturgeon distribution. Pages 257 *in* American Fisheries Society Symposium. American Fisheries Society.

Miller, A., K. Lion, M. Minton, and G. Ruiz. 2007. Status and trends of ballast water management in the United States. Third Biennial Report of the National Ballast Information Clearinghouse.(January 2004 to December 2005). Smithsonian Environmental Research Center. Report Submitted to the United States Coast Guard.

Miller, J. D., K. A. Dobbs, C. J. Limpus, N. Mattocks, and A. M. Landry Jr. 1998. Long-distance migrations by the hawksbill turtle, *Eretmochelys imbricata*, from north-eastern Australia. Wildlife Research 25(1):89-95.

Miller, R., and E. Brannon. 1982. The origin and development of life history patterns in Pacific salmonids. Pages 296-309 *in* Proceedings of the Salmon and Trout Migratory Behavior Symposium. Edited by EL Brannon and EO Salo. School of Fisheries, University of Washington, Seattle, WA.

Miller, T., and G. Shepherd. 2011. Summary of discard estimates for Atlantic sturgeon. Population Dynamics Branch, Northeast Fisheries Science Center 47.

Minchin, D., and H. Rosenthal. 2002. Exotics for stocking and aquaculture, making correct decisions. Pages 206-216 *in* E. Leppakoski, S. Gollasch, and S. Olenin, editors. Invasive Aquatic Species of Europe.

MMC. 2007. Marine mammals and noise: A sound approach to research and management. Marine Mammal Commission.

MMS. 1998. Pages III-3 to III-72 in Gulf of Mexico OCS oil and gas lease sales 171, 174, 177, and 180—Western Planning Area. Minerals Management Service, New Orleans, Louisiana.

MMS. 2007. Gulf of Mexico OCS oil and gas lease sales: 2007-2012, Western planning area sales 204, 207, 210, 215, and 218; Central planning area sales 205, 206, 208, 213, 216, and 222. Final environmental impact statement. Minerals Management Service.

Moein, S. E., and coauthors. 1995. Evaluation of seismic sources for repelling sea turtles from hopper dredges. Pages 75-78 *in* L. Z. Hales, editor. Sea Turtle Research Program: Summary report. Prepared for U.S. Army Corps of Engineers, South Atlantic, Atlanta GA and U.S. Naval Submarine Base, Kings Bay, GA.

Moncheva, S. P., and L. T. Kamburska. 2002. Plankton stowaways in the Black Sea - Impacts on biodiversity and ecosystem health. Pages 47-51. CIESM Workshop Monographs [CIESM Workshop Monogr.]. 2002. *in* Alien marine organisms introduced by ships in the Mediterranean and Black seas.

Montagna, P., T. A. Palmer, and J. B. Pollack. 2012. Hydrological changes and estuarine dynamics, volume 8. Springer Science & Business Media.

Monzón-Argüello, C., C. Rico, A. Marco, P. López, and L. F. López-Jurado. 2010. Genetic characterization of eastern Atlantic hawksbill turtles at a foraging group indicates major undiscovered nesting populations in the region. Journal of Experimental Marine Biology and Ecology.

Moore, J. E., and coauthors. 2009. A review of marine mammal, sea turtle and seabird bycatch in USA fisheries and the role of policy in shaping management. Marine Policy 33(3):435-451.

Moore, M. J., and J. M. Van der Hoop. 2012. The painful side of trap and fixed net fisheries: chronic entanglement of large whales. Journal of Marine Biology 2012.

Moore, S. K., N. J. Mantua, and E. P. Salathé. 2011. Past trends and future scenarios for environmental conditions favoring the accumulation of paralytic shellfish toxins in Puget Sound shellfish. Harmful Algae 10(5):521-529.

Mora, E. A. 2015. Personal communication via email with Phaedra Doukakis regarding green sturgeon DIDSON survey and results. UC Davis May 6, 2015.

Morel, F. M. M. 1983. Principles of aquatic chemistry. Wiley-Interscience, New York, NY.

Moser, H. G., and coauthors. 2000. Abundance and distribution of rockfish (*Sebastes*) larvae in the Southern California Bight in relation to environmental conditions and fishery exploitation. California Cooperative Oceanic Fisheries Investigations Reports (41):132-147.

Moser, M., and coauthors. 2016. Biology and life history of green sturgeon (Acipenser medirostris Ayres, 1854): state of the science. Journal of Applied Ichthyology 32:67-86.

Moser, M. L., and S. T. Lindley. 2007. Use of Washington estuaries by subadult and adult green sturgeon. Environmental Biology of Fishes 79(3-4):243.

Moser, M. L., and S. W. Ross. 1995. Habitat use and movements of shortnose and Atlantic sturgeons in the lower Cape Fear River, North Carolina. Transactions of the American Fisheries Society 124(2):225.

Moyle, P., P. Foley, and R. Yoshiyama. 1992. Status of green sturgeon, Acipenser medirostris. California. Final Report submitted to National Marine Fisheries Service, Terminal Island, CA.

Moyle, P. B. 2002. Inland fishes of California. Revised and Expanded. University of California Press, Berkeley, California.

MSPO. 1993. Kennebec River Resource Management Plan. Maine State Planning Office Paper 78, http://digitalmaine.com/spo_docs/78.

Murray, K. T. 2009. Characteristics and magnitude of sea turtle bycatch in US mid-Atlantic gillnet gear. Endangered Species Research 8(3):211-224.

Murray, K. T. 2011. Interactions between sea turtles and dredge gear in the US sea scallop (*Placopecten magellanicus*) fishery, 2001–2008. Fisheries Research 107(1):137-146.

Murray, K. T. 2015. Estimated loggerhead (*Caretta caretta*) interactions in the Mid-Atlantic scallop dredge fishery, 2009-2014. US Dept Commer, Northeast Fish Sci Cent Ref Doc:15-20.

Musick, J. A., and C. J. Limpus. 1997. Habitat utilization and migration in juvenile sea turtles. Pages 137-163 *in* P. Lutz, and J. A. Musick, editors. The Biology of Sea Turtles. CRC Press, Boca Raton, Florida.

Nakamoto, R. J., T. T. Kisanuki, and G. H. Goldsmith. 1995. Age and growth of Klamath River green sturgeon (*Acipenser medirostris*). U.S. Fish and Wildlife Service, Klamath River Fishery Resource Office.

National Academies of Sciences, E., and Medicine. 2016. Approaches to understanding the cumulative effects of stressors on marine mammals. National Academies Press.

Naval Base Coronado. 2013. Integraged natural resources management plan. Tierra Data, Inc., Escondido, CA.

Naval Ocean Systems Center. 1981. Progress Report: Effect of Organotin Antifoulling Leachates in the Marine Environment, In: Command, N.S.S. (Ed.), Environmental Assessment of Fleetwide use of Organotic Antifouling Paint, San Diego, California.

Navy. 2017. INACTSHIPS responses to NMFS final three questions (2, 3, & 13). Supplemental document provided to NMFS as part of the Navy's Inactive Ship Tow ESA Section 7 Programmatic Consultation. Naval Sea Systems Command, Environmental Planning Branch.

Navy. 2018a. Ballast Response 2018-02-12 1400. Supplemental document provided to NMFS as part of the Navy's Inactive Ship Tow ESA Section 7 Programmatic Consultation. Naval Sea Systems Command, Environmental Planning Branch.

Navy. 2018b. Inactive Ships Risk Matrix and Mitigation Measures_20180907. Supplemental document provided to NMFS as part of the Navy's Inactive Ship Tow ESA Section 7 Programmatic Consultation. Naval Sea Systems Command, Environmental Planning Branch.

Navy. 2018c. Navy's Hull Cleaning Description_20180817. Supplemental document provided to NMFS as part of the Navy's Inactive Ship Tow ESA Section 7 Programmatic Consultation. Naval Sea Systems Command, Environmental Planning Branch.

Navy. 2018d. Process for Dry Dock Selection_Final_07SEP18. Supplemental document provided to NMFS as part of the Navy's Inactive Ship Tow ESA Section 7 Programmatic Consultation. Naval Sea Systems Command, Environmental Planning Branch.

Navy. 2018e. Temp and Salinity Changes and Effects on NIS_from Navy. Supplemental document provided to NMFS as part of the Navy's Inactive Ship Tow ESA Section 7 Programmatic Consultation. Naval Sea Systems Command, Environmental Planning Branch.

Nehring, S. 2006. NOBANIS – Invasive alien species fact sheet-Crassostrea gigas. Online Database of the North European and Baltic Network on Invasive Alien Species.

Nell, J. 2002. The Australian oyster industry. World Aquaculture 613:8-10.

Newson, S. E., and coauthors. 2009. Indicators of the impact of climate change on migratory species. Endangered Species Research 7(2):101-113.

Nieukirk, S. L., K. M. Stafford, D. K. Mellinger, R. P. Dziak, and C. G. Fox. 2004. Low-frequency whale and seismic airgun sounds recorded in the mid-Atlantic Ocean. Journal of Acoustical Society of America 115:1832-1843.

NMFS. 1998. Recovery plan for the shortnose surgeon (*Acipenser brevirostrum*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, Maryland.

NMFS. 2004. ESA Section 7 reinitiation of consultation on the Atlantic Pelagic Longline Fishery for Highly Migratory Species. NOAA, National Marine Fisheries Service.

NMFS. 2005. Status review update for Puget Sound steelhead. National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle, Washington.

NMFS. 2006a. Biological opinion on the issuance of section lO(a)(l)(A) permits to conduct scientific research on the southern resident killer whale (Orcinus orca) distinct population segment and other endangered or threatened species. National Marine Fisheries Service, Northwest Region, Seattle, Washington.

NMFS. 2006b. Continued Authorization of Snapper-Grouper Fishing in the U.S. South Atlantic Exclusive Economic Zone (EEZ) as Managed under the Snapper-Grouper Fishery Management Plan (SGFMP) of the South Atlantic Region,, including Amendment 13C to the SGFMP. NOAA, National Marine Fisheries Service, Southeast Regional Office, Protected Resources Division.

NMFS. 2006c. Final supplement to the Shared Strategy's Puget Sound salmon recovery plan. National Marine Fisheries Service, Northwest Region, editor, Seattle.

NMFS. 2007a. Final Supplement to the Hood Canal and Eastern Strait of Juan de Fuca Summer Chum Salmon Recovery Plan, Portland, Oregon.

NMFS. 2007b. Hawaiian monk seal (*Monachus schauinslandi*) 5-year review: Summary and evaluation. National Marine Fisheries Service, Pacific Islands Regional Office, Honolulu, Hawaii.

NMFS. 2007c. Hawaiian monk seal (*Monachus schauinslandi*). 5-year review: Summary and evaluation. National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

NMFS. 2007d. Puget Sound salmon recovery plan. Submitted by the Shared Strategy Development Committee, Seattle, Washington.

NMFS. 2007e. Recovery plan for the Hawaiian monk seal (*Monachus schauinslandi*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

NMFS. 2008. Recovery Plan for Southern Resident Killer Whales (*Orcinus orca*). National Marine Fisheries Service, Northwest Region, Seattle, Washington.

NMFS. 2009. The Continued Authorization of Fishing under the Fishery Management Plan for the Stone Crab Fishery of the Gulf of Mexico. NOAA, National Marine Fisheries Service, Southeast Regional Office, Protected Resources Division.

NMFS. 2010a. Federal Recovery Outline North American Green Sturgeon Southern Distinct Population Segment. N. M. F. Service, editor, Southwest Region.

NMFS. 2010b. Shortnose Sturgeon Status Review Team. A Biological Assessment of Shortnose Sturgeon (*Acipenser brevirostrum*). Report to National Marine Fisheries Service. Pages 417 pp *in*, Northeast Regional Office.

NMFS. 2011a. 5-Year Review: Summary & Evaluation of Puget Sound Chinook Hood Canal Summer Chum Puget Sound Steelhead. National Marine Fisheries Service Northwest Region Portland, OR.

NMFS. 2011b. Continued Authorization of Reef Fish Fishing under the Gulf of Mexico Reef Fish Fishery Management Plan. NOAA, National Marine Fisheries Service, Southeast Regional Office, Protected Resources Division.

NMFS. 2012a. Biological opinion on the continued operation of the Hawaii-based shallow-set longline swordfish fishery under amendment 18 to the FMP for pelagic fisheries of the Western Pacific Region. NOAA, National Marine Fisheries Service, Pacific Islands Region, Protected Resources Division.

NMFS. 2012b. Biological Opinion on the Operation of the Pacific Coast Groundfish Fishery. NMFS, Northwest Region.

NMFS. 2012c. 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.

NMFS. 2012d. ESA Section 7 Consultation on an assessment of the United States Coast Guard's National Ballast Water Management Program and Initial Numerical Standard. U.S. Dept. of Commerce, NOAA, NMFS, Silver Spring, MD.

NMFS. 2013. Biological opinion on the continued management of the drift gillnet fishery under the FMP for U.S. West Coast fisheries for highly migratory species. NOAA, National Marine Fisheries Service, Southwest Region, Protected Resources Division.

NMFS. 2014a. Biological opinion on the Brazos Island Harbor Channel Improvement Project. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Regional Office, St. Petersburg, Florida.

NMFS. 2014b. Biological opinion on the continued operation of the Hawaii-based deep-set pelagic longline fishery. NOAA, National Marine Fisheries Service, Pacific Islands Region, Protected Resources Division.

NMFS. 2014c. Reinitiation of Endangered Species Act (ESA) Section 7 Consultation on the Continued Implementation of the Sea Turtle Conservation Regulations under the ESA and the Continued Authorization of the Southeast U.S. Shrimp Fisheries in Federal Waters under the Magnuson-Stevens Fishery Management and Conservation Act. NOAA. NMFS, Southeast Regional Office, Protected Resources Division.

NMFS. 2015a. Reinitiation of ESA Section 7 Consultation on the Continued Authorization of the FMP for Coastal Migratory Pelagic Resources in the Atlantic and Gulf of Mexico under the Magnuson-Stevens Fishery Management and Conservation Act. NOAA, National Marine Fisheries Service, Southeast Regional Office, Protected Resources Division.

NMFS. 2015b. Southern Distinct Population Segment of the North American Green Sturgeon (Acipenser medirostris); 5-year Review: Summary and Evaluation. W. C. R. National Marine Fisheries Service, editor, Long Beach, CA.

NMFS. 2016a. Draft Rockfish Recovery Plan: Puget Sound/Georgia Basin yelloweye rockfish (*Sebastes ruberrimus*) and bocaccio (*Sebastes paucispinis*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service, West Coast Regional Office, Seattle, Washington.

NMFS. 2016b. ESA Section 7(a)(2) Biological Opinion for the Continued Opertation of the West Coast-based Deep-set Longline Fishery managed under the FMP for U.S. West Coast HMS. NOAA, National Marine Fisheries Service, West Coast Region.

NMFS. 2016c. Interim Guidance on the Endangered Species Act Term "Harass." December 2016. Protected Resources Management. National Marine Fisheries Service Procedural Instruction 02-110-10. .

NMFS. 2016d. Main Hawaiian Islands monk seal management plan. National Marine Fisheries Service, Pacific Islands Region, Honolulu, Hawaii.

NMFS. 2016e. National Bycatch Reduction Strategy. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Silver Spring, MD.

NMFS. 2016f. Puget Sound chinook salmon, Hood Canal summer-run chum salmon, and Puget Sound steelhead 5-year review: Summary and Evaluation. National Marine Fisheries Service, West Coast Region, Portland, Oregon.

NMFS. 2016g. Request for informal consultation under section 7(a)(2) of the ESA and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Response regarding United States Navy Naval Sea Systems Command Inactive Ships Office proposed contracted towing and dismantling of inactive U.S. Navy vessel ex-Independence. NOAA, National Marine Fisheries Service, Silver Spring, Maryland.

NMFS. 2016h. Southern Resident Killer Whales (*Orcinus orca*) 5-year Review: Summary and Evaluation. National Marine Fisheries Service, West Coast Region, Seattle, Washington.

NMFS. 2016i. Yelloweye rockfish (*Sebastes ruberrimus*), canary rockfish (*Sebastes pinniger*), and bocaccio (*Sebastes paucispinis*) of the Puget Sound/Georgia Basin. Pages 131 *in* Five-Year Review: Summary and Evaluation, West Coast Region Seattle, WA.

NMFS. 2017a. 5-Year Review: Summary & Evaluation of Puget Sound Chinook Hood Canal Summer Chum Puget Sound Steelhead, Portland, Oregon.

NMFS. 2017b. Biological and Conference Opinion on the Proposed Implementation of a Program for the Issuance of Permits for Research and Enhancement Activities on Atlantic and Shortnose Sturgeon Pursuant to Section 10(a) of the Endangered Species Act. Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service.

NMFS. 2017c. Biological and Conference Opinion on the Proposed Implementation of a Program for the Issuance of Permits for Research and Enhancement Activities on Threatened and Endangered Sea Turtles Pursuant to Section 10(a) of the Endangered Species Act, Silver Spring, Maryland.

NMFS. 2017d. Biological and Conference Opinion on United States Navy's Surveillance Towed Array Sensor System Low Frequency Active Sonar Routine Training, Testing, and Military Operations from August 2017 through August 2022. Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, FPR-2016-9181, Silver Spring, Maryland.

NMFS. 2017e. Recovery plan for the southern distinct population segment of Eulachon (*Thaleichthys pacificus*). National Marine Fisheries Service, West Coast Region, Protected Resources Division, Portland, Oregon.

NMFS, and USFWS. 1991. Recovery plan for U.S. population of Atlantic green turtle *Chelonia mydas*. National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service, Washington, D.C.

NMFS, and USFWS. 1992. Recovery plan for leatherback turtles (*Dermochelys coriacea*) in the U.S. Caribben, Atlantic, and Gulf of Mexico. National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service, Washington, D.C.

NMFS, and USFWS. 1993. Recovery plan for hawksbill turtles (*Eretmochelys imbricata*) in the U.S. Caribbean Sea, Atlantic Ocean, and Gulf of Mexico. National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service, St. Petersburg, Florida.

NMFS, and USFWS. 1998a. Recovery plan for the U.S. Pacific populations of the leatherback turtle (*Dermochelys coriacea*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service, Washington, D.C.

NMFS, and USFWS. 1998b. Recovery plan for U.S. Pacific populations of the green turtle (*Chelonia mydas*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 1998c. Recovery plan for U.S. Pacific populations of the hawksbill turtle (*Eretmochelys imbricata*). National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, MD.

NMFS, and USFWS. 2007a. Green sea turtle (*Chelonia mydas*) 5-year review: summary and evaluation. National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 2007b. Hawksbill sea turtle (*Eretmochelys imbricata*) 5-year review: Summary and evaluation. National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 2007c. Loggerhead sea turtle (*Caretta caretta*) 5-year review: Summary and evaluation. National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 2008. Recovery plan for the northwest Atlantic population of the loggerhead sea turtle (*Caretta caretta*): Second revision. National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 2011. Bi-National Recovery Plan for the Kemp's Ridley Sea Turtle (*Lepidochelys kempii*), Second Revision. National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 2013a. Hawksbill sea turtle (*Eretmochelys imbricata*) 5-year review: Summary and evaluation National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service, Silver Spring, Maryland.

NMFS, and USFWS. 2013b. Leatherback sea turtle (*Dermochelys coriacea*) 5-year review: Summary and evaluation. National Oceanic and Atmospheric Administration National Marine Fisheries Service, Office of Protected Resources and U.S. Fish and Wildlife Service, Southeast Region, Jacksonvill Ecological Services Office, Jacksonville, Florida.

NMFS, and USFWS. 2015. Kemp's ridley sea turtle (*Lepidochelys kempii*) 5-year review: Summary and evaluation. National Oceanic and Atmospheric Administration, National Marine Fisheries Service and U.S. Fish and Wildlife Service.

NMFS BRT. 1997. Status review of chum salmon from Washington, Oregon, and California. West Coast Chum Biological Review Team, U.S. Department of Commerce, NOAA Technical Memo.

NMFS BRT. 2003. Updated statues of federally listed ESUs of West Coast salmon and steelhead. West Coast Chinook Salmon Biological Review Team, Seattle, Washington.

NMFS NWFSC. 2015. Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Pacific northwest. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center.

NMFS SEFSC. 2009. An assessment of loggerhead sea turtles to estimate impacts of mortality reductions on population dynamics. NMFS Southeast Fisheries Science Center Contribution PRD-08/09-14.

NMFS WCR. 2016. Eulachon (*Thaleichthys pacificus*) 5-year review: Summary and evaluation. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, West Coast Region, Portland, Oregon.

NOAA. 2003. Oil and sea turtles: Biology, planning, and response. National Oceanic and Atmospheric Administration, National Ocean Service, Office of Response and Restoration.

NOAA. 2018a. GNOME (General NOAA Operational Modeling Environment). NOAA Ocean Service, Office of Response and Restoration.

NOAA. 2018b. Port Beaumount- Northern Gulf of Mexico Operational Forecast System (NGOFS) https://tidesandcurrents.noaa.gov/ofs/ofs_station.shtml?stname=Port%20Beaumount&ofs=ng&stnid=sn0601c&subdomain=sn.

NOAA. 2018c. Rainbow Bridge- Northern Gulf of Mexico Operational Forecast System (NGOFS). https://tidesandcurrents.noaa.gov/ofs/ofs_station.shtml?stname=Rainbow%20Bridge&ofs=ng&stnid=sn0501c&subdomain=sn.

NRC. 1990. Sea turtle mortality associated with human activities. National Academy Press, National Research Council Committee on Sea Turtle Conservation, Washington, D.C.

NRC. 2003. Ocean noise and marine mammals. National Academy Press, Washington, DC.

NRC. 2005. Marine mammal populations and ocean noise. Determining when noise causes biologically significant effects. National Academy of Sciences, Washington, D. C.

NRC. 2011. Assessing the relationship between propagule pressure and invasion risk in ballast water. National Research Council, National Academies Press.

NUWC. 2017a. Revised biological evaluation for species listed under the Endangered Species Act under National Marine Fisheries Service jurisdiction for the towing of inactive ships. Naval Sea Systems Command Inactive Ships Office (SEA 21I), Naval Undersea Warfare Center Division, Environmental Division, Mission Environmental Planning Program, Newport, Rhode Island.

NUWC. 2017b. Water Quality Monitoring of Biofouling Removal from the ex-USS INDEPENDENCE (CV 62). Final Report prepared by the Space and Naval Warfare Systems Center Pacific and Naval Underwater Warfare Center Newport., San Diego, CA.

Nyberg, C. D. 2007. Introduced marine macroalgae and habitat modifiers - their ecological role and significant attributes. Dissertation; Bibliography. Goeteborg University, Goeteborg, Sweden.

O'Driscoll, M., S. Clinton, A. Jefferson, A. Manda, and S. McMillan. 2010. Urbanization effects on watershed hydrology and in-stream processes in the southern United States. Water 2(3):605-648.

O'Hara, J., and J. R. Wilcox. 1990. Avoidance responses of loggerhead turtles, *Caretta caretta*, to low frequency sound. Copeia 2:564-567.

O'herron, J. C., K. W. Able, and R. W. Hastings. 1993. Movements of shortnose sturgeon (Acipenser brevirostrum) in the Delaware River. Estuaries 16(2):235-240.

O'Leary, S. J., K. J. Dunton, T. L. King, M. G. Frisk, and D. D. Chapman. 2014. Genetic diversity and effective size of Atlantic sturgeon, *Acipenser oxyrhinchus oxyrhinchus* river spawning populations estimated from the microsatellite genotypes of marine-captured juveniles. Conservation Genetics 15(5):1173-1181.

OCNO. 2016. Report to Congress on the Annual Long-Range Plan for Construction of Naval Vessels for Fiscal Year 2017. Office of the Chief ofNa\'al Operations, Deputy Chief of Naval Operations (Integration of Capabilities and Resources) (N8) Washington, D.C.

Oros, J., O. M. Gonzalez-Diaz, and P. Monagas. 2009. High levels of polychlorinated biphenyls in tissues of Atlantic turtles stranded in the Canary Islands, Spain. Chemosphere 74(3):473-478.

Orós, J., A. Torrent, P. Calabuig, and S. Déniz. 2005. Diseases and causes of mortality among sea turtles stranded in the Canary Islands, Spain (1998–2001). Diseases of aquatic organisms 63(1):13-24.

Pacileo, T. S. D. 2003. Movements and important habitats of subadult Atlantic sturgeon in Connecticut waters. Transactions of the American Fisheries Society 132:1-8.

Palsson, W. A., and coauthors. 2009. The biology and assessment of rockfishes in Puget Sound. Washington Department of Fish and Wildlife, Fish Management Division, Fish Program.

Parker, I. M., and coauthors. 1999. Impact: toward a framework for understanding the ecological effects of invaders. Biological invasions 1(1):3-19.

Parker, S. J., H. I. McElderry, P. S. Rankin, and R. W. Hannah. 2006. Buoyancy regulation and barotrauma in two species of nearshore rockfish. Transactions of the American Fisheries Society 135:1213-1223.

Parks, S. E. 2003. Response of North Atlantic right whales (Eubalaena glacialis) to playback of calls recorded from surface active groups in both the North and South Atlantic. Marine Mammal Science 19(3):563-580.

Parks, S. E. 2009. Assessment of acoustic adaptations for noise compensation in marine mammals. 2009 ONR Marine Mammal Program Review, Alexandria, Virginia.

Parra, M., S. L. Deem, and E. Espinoza. 2011. Green Turtle (Chelonia mydas) Mortality in the Galápagos Islands, Ecuador During the 2009-2010 Nesting Season. Marine Turtle Newsletter (130):10.

Parrish, F. A., M. P. Craig, T. J. Ragen, G. J. Marshall, and B. M. Buhleier. 2000. Identifying diurnal foraging habitat of endangered Hawaiian monk seals using a seal-mounted video camera. Marine Mammal Science 16(2):392-412.

Parsons, K. M., K. C. B. III, J. K. B. Ford, and J. W. Durban. 2009. The social dynamics of southern resident killer whales and conservation implications for this endangered population. (Orcinus orca). Animal Behaviour 77(4):963-971.

Partnership for the Delaware Estuary. 2017. Technical report for the Delaware Estuary and Basin 2017. Pages 379 pp *in* L. Haaf, S. Demberger, D. Kreeger, and E. Baumbach, editors. Partnership for the Delaware Estuary. PDE Report No. 17-07.

Patrício, A. R., and coauthors. 2017. Balanced primary sex ratios and resilience to climate change in a major sea turtle population. Marine Ecology Progress Series 577:189-203.

Paulo da Cunha Lana, C. G. 1991. Influence of Spartina alterniflora on structure and temporal variability of macrobenthic associations in a tidal flat of Paranagua Bay (southeastern Brazil). Marine Ecology Progress Series 73:231-244.

Peel, D., J. N. Smith, and S. Childerhouse. 2018. Vessel Strike of Whales in Australia: The Challenges of Analysis of Historical Incident Data. Frontiers in Marine Science 5.

Petersen, W., E. Willer, and C. Willamowski. 1997. Remobilization of trace elements from polluted anoxic sediments after resuspension in oxic water. Water, Air, and Soil Pollution 99(1-4):515-522.

Peterson, D., and coauthors. 2008. Annual run size and genetic characteristics of Atlantic sturgeon in the Altamaha River, Georgia. Transactions of the American Fisheries Society 137:393-401.

Peterson, D. L., M. B. Bain, and N. Haley. 2000. Evidence of declining recruitment of Atlantic sturgeon in the Hudson River. North American Journal of Fisheries Management 20(1):231-238.

Philippart, C. J. M., and coauthors. 2011. Impacts of climate change on European marine ecosystems: Observations, expectations and indicators☆. Journal of Experimental Marine Biology and Ecology.

Pike, D. A., R. L. Antworth, and J. C. Stiner. 2006. Earlier nesting contributes to shorter nesting seasons for the loggerhead seaturtle, *Caretta caretta*. Journal of Herpetology 40(1):91-94.

Pike, D. A., E. A. Roznik, and I. Bell. 2015. Nest inundation from sea-level rise threatens sea turtle population viability. Royal Society Open Science 2:150127.

Pilot, M., M. E. Dahlheim, and A. R. Hoelzel. 2010. Social cohesion among kin, gene flow without dispersal and the evolution of population genetic structure in the killer whale (*Orcinus orca*). Journal of Evolutionary Biology 23(1):20-31.

Pimentel, D., R. Zuniga, and D. Morrison. 2004. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics.

Pine, W., and S. Martell. 2009. Status of Gulf sturgeon Acipenser oxyrinchus desotoi in the Gulf of Mexico: a document prepared for review, discussion, and research planning. Gulf sturgeon annual working group meeting, Cedar Key, Florida.

Piner, K. R., J. R. Wallace, O. S. Hamel, and R. Mikus. 2006. Evaluation of ageing accuracy of bocaccio (*Sebastes paucispinis*) rockfish using bomb radiocarbon. Fisheries Research 77(2):200-206.

Pitcher, T. J. 1986. Functions of shoaling behaviour in teleosts. Springer.

Plotkin, P. 2003. Adult migrations and habitat use. Pages 225-241 *in* L. Lutz, J. A. Musick, and J. Wyneken, editors. Biology of sea turtles, volume II. CRC Press, Boca Raton, Florida.

Poeta, G., E. Staffieri, A. Acosta, and C. Battisti. 2017. Ecological effects of anthropogenic litter on marine mammals: A global review with a “black-list” of impacted taxa.

Poloczanska, E. S., C. J. Limpus, and G. C. Hays. 2009. Vulnerability of marine turtles in climate change. Pages 151-211 *in* D. W. Sims, editor. Advances in Marine Biology, volume 56. Academic Press, Burlington.

Popper, A. N., and coauthors. 2014. Sound exposure guidelines for fishes and sea turtles: A technical report prepared by ANSI-Accredited Standards Committee S3/SC1 and registered with ANSI. Springer, New York, NY.

Post, W., T. Darden, D. Peterson, M. Loeffler, and C. Collier. 2014. Research and management of endangered and threatened species in the southeast: riverine movements of Shortnose and Atlantic sturgeon. South Carolina Department of Natural Resources, Project NA10NMF4720036, Final Report, Charleston.

POV. 2018. The Port of Virginia: Fast Facts. http://www.portofvirginia.com/about/fast-facts/.

Powers, S. P., M. A. Bishop, J. H. Grabowski, and C. H. Peterson. 2006. Distribution of the invasive bivalve Mya arenaria L. on intertidal flats of southcentral Alaska. Journal of Sea Research 55(3):207-216.

Poytress, W. R., J. J. Gruber, and J. Van Eenennaam. 2009. 2008 Upper Sacramento River Green Sturgeon spawning habitat and larval migration surveys. Final Annual Report to US Bureau of Reclamation, US Fish and Wildlife Service.

Poytress, W. R., J. J. Gruber, and J. Van Eenennaam. 2010. 2009 Upper Sacramento River Green Sturgeon spawning habitat and larval migration surveys. Annual Report to US Bureau of Reclamation, US Fish and Wildlife Service.

Poytress, W. R., J. J. Gruber, and J. Van Eenennaam. 2013. 2012 Upper Sacramento River Green Sturgeon spawning habitat and young of the year migration surveys. Annual Report to US Bureau of Reclamation, US Fish and Wildlife Service.

Price, E. R., and coauthors. 2004. Size, growth, and reproductive output of adult female leatherback turtles *Dermochelys coriacea*. Endangered Species Research 5:1-8.

PSTT, and WDFW. 2004. Puget Sound hatchery strategies for steelhead, coho salmon, chum salmon, sockeye salmon, and pink salmon. Puget Sound Treaty Tribes and Washington Department of Fish and Wildlife, Olympia, Washington.

Pughiuc, D. 2010. Invasive species: Ballast water battles. Seaways.

Quattro, J., T. Greig, D. Coykendall, B. Bowen, and J. Baldwin. 2002. Genetic issues in aquatic species management: the shortnose sturgeon (*Acipenser brevirostrum*) in the southeastern United States. Conservation Genetics 3(2):155-166.

Raaymakers, S. 2003. The GEF/UNDP/IMO global ballast water management programme integrating science, shipping and society to save our seas. Proceedings of the Institute of Marine Engineering, Science and Technology Part B: Journal of Design and Operations (B4):2-10.

Raaymakers, S., and R. Hilliard. 2002. Harmful aquatic organisms in ships' ballast water - Ballast water risk assessment, 1726-5886, Istanbul, Turkey.

Rabalais, N. N., R. E. Turner, and D. Scavia. 2002. Beyond science into policy: Gulf of Mexico hypoxia and the Mississippi River. BioScience 52(2):129-142.

Radtke, L. 1966. Distribution of smelt, juvenile sturgeon, and starry flounder in the Sacramento-San Joaquin Delta with observations on food of sturgeon. Ecological studies of the Sacramento-San Joaquin Estuary, Part II:115-119.

Ragen, T. J. 1999. Human activities affecting the population trends of the Hawaiian monk seal. American Fisheries Society Symposium 23:183 - 194.

Ralston, S., and J. N. Ianelli. 1998. When lengths are better than ages: The complex case of bocaccio. University of Alaska, Fairbanks, Alaska.

Ramseur, J. L. 2010. Deepwater Horizon oil spill: the fate of the oil. Congressional Research Service, Library of Congress Washington, DC.

Randall, M., and K. Sulak. 2012. Evidence of autumn spawning in Suwannee River Gulf sturgeon, Acipenser oxyrinchus desotoi (Vladykov, 1955). Journal of Applied Ichthyology 28(4):489-495.

Ranke, J. 2002. Persistence of antifouling agents in the marine biosphere. Environmental science & technology 36(7):1539-1545.

Reid, S. D., D. G. McDonald, and C. M. Wood. 1997. Interactive effects of temperature and pollutant stress. Pages 325-349 *in* C. M. W. a. D. G. McDonald, editor. Global warming: implications for freshwater and marine fish. Cambridge University Press, Cambridge, United Kingdom.

Reilly, C. A., T. W. Echeverria, and S. Ralston. 1992. Interannual variation and overlap in the diets of pelagic juvenile rockfish (genus *Sebastes*) off central California. Fishery Bulletin 90(3):505-515.

Reina, R. D., P. A. Mayor, J. R. Spotila, R. Piedra, and F. V. Paladino. 2002. Nesting ecology of the leatherback turtle, *Dermochelys coriacea*, at Parque Nacional Marino Las Baulas, Costa Rica: 1988-1989 to 1999-2000. Copeia 2002(3):653-664.

Reina, R. D., J. R. Spotila, F. V. Paladino, and A. E. Dunham. 2008. Changed reproductive schedule of eastern Pacific leatherback turtles Dermochelys coriacea following the 1997–98 El Niño to La Niña transition. Endangered Species Research.

Reyff, J. A. 2003. Underwater sound levels associated with constnlction of the Benicia-Martinez Bridge. lllingworth & Rodkin, Inc.

Reyff, J. A. 2008. Underwater sound pressure levels associated with marine pile driving: assessment of impacts and evaluation of control measures. Journal of Transportation Research Board CD 11-S:481-490.

Reyff, J. A. 2012. Underwater sounds from unattenuated and attenuated marine pile driving. Pages 439-444 *in* A. N. Popper, and A. D. Hawkins, editors. The effects of noise on aquatic life. Springer Science + Business Media, LLC, New York, NY.

Richardson, W. J. 1995. Marine mammal hearing. Pages 205-240 *in* C. R. W. J. G. J. Richardson, C. I. Malme, and D. H. Thomson, editors. Marine Mammals and Noise. Academic Press, San Diego, California.

Ridgway, S. H., E. G. Wever, J. G. McCormick, J. Palin, and J. H. Anderson. 1969. Hearing in the giant sea turtle, *Chelonia mydas*. Proceedings of the National Academy of Science 64:884-890.

Rivera-Duarte, I., and coauthors. 2005. Copper toxicity to larval stages of three marine invertebrates and copper complexation capacity in San Diego Bay, California. Environmental science & technology 39(6):1542-1546.

Robinson, R. A., and coauthors. 2008. Travelling through a warming world: climate change and migratory species. Endangered Species Research.

Rockwood, R. C., J. Calambokidis, and J. Jahncke. 2017. High mortality of blue, humpback and fin whales from modeling of vessel collisions on the U.S. West Coast suggests population impacts and insufficient protection. PLoS One 12(8):e0183052.

Rogillio, H., and coauthors. 2001. Status, movement, and habitat use of Gulf Sturgeon in the Lake Pontchartrain basin, Louisiana. Louisiana Department of Wildlife and Fisheries and National Wildlife Foundation, Shell Marine Habitat Program, Final Report, Baton Rouge.

Romano, T. A., and coauthors. 2002. Immune response, stress, and environment: Implications for cetaceans. Pages 253-279 *in* Molecular and Cell Biology of Marine Mammals. Krieger Publishing Co., Malabar, Florida.

Rosenthal, H., and D. Alderdice. 1976. Sublethal effects of environmental stressors, natural and pollutional, on marine fish eggs and larvae. Journal.

Ross, S., R. Heise, M. Dugo, and W. T. Slack. 2001. Movement and Habitat use of the Gulf Sturgeon (Acipenser oxyrinchus desotoi) in the Pascagoula Drainage of Mississippi: year V. Project No. E-1, Segment 16. University of Southern Mississippi, Mississippi Museum of Natural Science.

Ross, S. T., and coauthors. 2009. Estuarine and coastal habitat use of Gulf sturgeon (Acipenser oxyrinchus desotoi) in the north-central Gulf of Mexico. Estuaries and Coasts 32(2):360-374.

Ruckelshaus, M., and coauthors. 2002. Planning ranges and preliminary guidelines for the delisting and recovery of the Puget Sound Chinook salmon evolutionarily significant unit. Puget Sound Technical Recovery Team. National Marine Fisheries Service, Northwest Fisheries Science Center, editor, Seattle.

Rudd, M. B., R. N. Ahrens, W. E. Pine III, and S. K. Bolden. 2014. Empirical, spatially explicit natural mortality and movement rate estimates for the threatened Gulf Sturgeon (Acipenser oxyrinchus desotoi). Canadian Journal of Fisheries and Aquatic Sciences 71(9):1407-1417.

Ruelle, R., and K. D. Keenlyne. 1993. Contaminants in Missouri River pallid sturgeon. Bulletin of Environmental Contamination and Toxicology 50(6):898-906.

Ruiz, G. M., P. Fofonoff, and A. H. Hines. 1999. Non-indigenous species as stressors in estuarine and marine communities: Assessing invasion impacts and interactions. Limnology and Oceanography 44(3):950–972.

Russell, D. J., and G. H. Balazs. 2009. Dietary shifts by green turtles (Chelonia mydas) in the Kane‘ohe Bay region of the Hawaiian Islands: A 28-year study. Pacific Science 63(2):181-192.

Sahoo, G., R. K. Sahoo, and P. Mohanty-Hejmadi. 1996. Distribution of heavy metals in the eggs and hatchlings of olive ridley sea turtles, *Lepidochelys olivacea*, from Gahirmatha, Orissa. Indian Journal of Marine Sciences 25(4):371-372.

Sakuma, K. M., and S. Ralson. 1995. Distributional patterns of late larval groundfish off central California in relation to hydrographic features during 1992 and 1993. California Cooperative Oceanic Fisheries Investigations Reports 36:179-192.

Salo, E. O. 1991. Life history of chum salmon (Oncorhynchus keta). Pages 231–309 *in* C. G. a. L. Margolis, editor. Pacific salmon life histories. University of British Columbia Press, Vancouver, B.C.

Sanchez, I., C. Fernandez, and J. Arrontes. 2005. Long-Term Changes in the Structure of Intertidal Assemblages after Invasion by Sargassum Muticum (Phaeophyta)1. Journal of Phycology 41(5):942-949.

Sands, N. J., K. Rawson, K.P. Currens, W.H. Graeber, M.H., Ruckelshaus, R.R. Fuerstenberg, and J.B. Scott. 2009. Determination of Independent Populations and Viability Criteria for the Hood Canal Summer Chum Ssalmon Evolutionarily Significant Unit. National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce.

Santos, R. G., R. Andrades, M. A. Boldrini, and A. S. Martins. 2015. Debris ingestion by juvenile marine turtles: an underestimated problem. Marine Pollution Bulletin 93(1):37-43.

Savoy, T. 2007. Prey eaten by Atlantic sturgeon in Connecticut waters. Pages 157 *in* American Fisheries Society Symposium. American Fisheries Society.

Savoy, T., and J. Benway. 2004. Food habits of shortnose sturgeon collected in the lower Connecticut River from 2000 through 2002. American Fisheries Society Monograph 9:353-360.

Savoy, T., L. Maceda, N. K. Roy, D. Peterson, and I. Wirgin. 2017. Evidence of natural reproduction of Atlantic sturgeon in the Connecticut River from unlikely sources. PLoS One 12(4):e0175085.

Savoy, T. S., Deborah. 1992. Anadromous fish studies in Connecticut waters. Department of Environmental Protection, AFC-20-1.

Schatzberg, P. 1987. Organotin Antifouling Hull Paints and the US Navy--A Historical Perspective. Pages 1324-1333 *in* OCEANS'87. IEEE.

Schindler, D. W. 2001. The cumulative effects of climate warming and other human stresses on Canadian freshwaters in the new millennium. Canadian Journal of Fisheries and Aquatic Sciences 58(1):18-29.

Schlaepfer, M. A., D. F. Sax, and J. D. Olden. 2011. The potential conservation value of non-native species. Conservation Biology 25(3):428-437.

Schmid, J. R. 1998. Marine turtle populations on the west-central coast of Florida: Results of tagging studies at the Cedar Keys, Florida, 1986-1995. Fishery Bulletin 96(3):589-602.

Scholz, N. L., and coauthors. 2000. Diazinon disrupts antipredator and homing behaviors in chinook salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences 57(9):1911-1918.

Schreier, A., O. P. Langness, J. A. Israel, and E. Van Dyke. 2016. Further investigation of green sturgeon (Acipenser medirostris) distinct population segment composition in non-natal estuaries and preliminary evidence of Columbia River spawning. Environmental Biology of Fishes 99(12):1021-1032.

Schueller, P., and D. L. Peterson. 2010. Abundance and recruitment of juvenile Atlantic Sturgeon in the Altamaha River, Georgia. Transactions of the American Fisheries Society 139(5):1526-1535.

Schultz, J. K., J. D. Baker, R. J. Toonen, and B. W. Bowen. 2009. Extremely low genetic diversity in the endangered Hawaiian monk seal (*Monachus schauinslandi*). Journal of Heredity 100(1):25-33.

Schultz, J. K., J. D. Baker, R. J. Toonen, A. L. Harting, and B. W. Bowen. 2011. Range-wide genetic connectivity of the Hawaiian monk seal and implications for translocation. Conservation Biology 25(1):124-132.

Schuyler, Q. A. 2014. Ingestion of marine debris by sea turtles. Doctoral dissertation. The University of Queensland.

Schuyler, Q. A., and coauthors. 2016. Risk analysis reveals global hotspots for marine debris ingestion by sea turtles. Global Change Biology 22(2):567-576.

Secor, D. H., and T. E. Gunderson. 1998. Effects of hypoxia and temperature on survival, growth, and respiration of juvenile Atlantic sturgeon, *Acipenser oxyrinchus*. Fishery Bulletin 96:603-613.

Secor, D. H., and E. J. Niklitschek. 2001. Hypoxia and sturgeons. University of Maryland Center for Environmental Science, Chesapeake Biological Laboratory, Technical Report Series No. TS-314-01-CBL, Solomons, Maryland.

Secor, D. H., and E. J. Niklitschek. 2002. Sensitivity of sturgeons to environmental hypoxia: A review of physiological and ecological evidence. Pages 61-78 *in* R. V. Thurston, editor. Fish Physiology, Toxicology, and Water Quality, volume EPA/600/R-02/097. U.S.

Environmental Protection Agency, Office of Research and Development, Ecosystems Research Division, Athens, Georgia.
Secor, D. H., and J. R. Waldman. 1999. Historical abundance of Delaware Bay Atlantic sturgeon and potential rate of recovery. Pages 203-216 *in* American Fisheries Society Symposium.
Seesholtz, A. M., M. J. Manuel, and J. P. Van Eenennaam. 2015. First documented spawning and associated habitat conditions for green sturgeon in the Feather River, California. Environmental Biology of Fishes 98(3):905-912.
Seminoff, J. A., and coauthors. 2015. Status review of the green turtle (*Chelonia mydas*) under the Endangered Species Act. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.
Shamblin, B. M., and coauthors. 2014. Geographic patterns of genetic variation in a broadly distributed marine vertebrate: New insights into loggerhead turtle stock structure from expanded mitochondrial DNA sequences. PLoS ONE 9(1):e85956.
Shamblin, B. M., and coauthors. 2012. Expanded mitochondrial control region sequences increase resolution of stock structure among North Atlantic loggerhead turtle rookeries. Marine Ecology Progress Series 469:145-160.
Shamblin, B. M., and coauthors. 2016. Mexican origins for the Texas green turtle foraging aggregation: A cautionary tale of incomplete baselines and poor marker resolution. Journal of Experimental Marine Biology and Ecology:10 pp.
Shared Strategy for Puget Sound. 2007. Puget Sound salmon recovery plan. Volume 1, recovery plan, Seattle.
Shoop, C. R., and R. D. Kenney. 1992. Seasonal distributions and abundances of loggerhead and leatherback sea turtles in waters of the northeastern United States. Herpetological Monographs 6:43-67.
Showalter, S., and J. Savarese. 2005. Restrictions on the use of marine antifouling paints containing tributyltin and copper. Sea Grant Law Center. MASGP:04-052.
Simberloff, D. 2000. Global climate change and introduced species in United States forests. Science of the Total Environment 262(3):253-261.
Simberloff, D. 2011. How common are invasion-induced ecosystem impacts? Biological invasions 13(5):1255-1268.
Simberloff, D., and coauthors. 2013. Impacts of biological invasions: what's what and the way forward. Trends Ecol Evol 28(1):58-66.
Simeone, C. A., F. M. Gulland, T. Norris, and T. K. Rowles. 2015. A systematic review of changes in marine mammal health in North America, 1972-2012: the need for a novel integrated approach. PLoS One 10(11):e0142105.
Simmonds, M. P., and W. J. Eliott. 2009. Climate change and cetaceans: Concerns and recent developments. Journal of the Marine Biological Association of the United Kingdom 89(1):203-210.
Simpfendorfer, C. A., and coauthors. 2011. Environmental influences on the spatial ecology of juvenile smalltooth sawfish (*Pristis pectinata*): results from acoustic monitoring. PLoS One 6(2):e16918.
Singel, K., A. Foley, and R. Bailey. 2007. Navigating Florida's waterways: Boat-related strandings of marine turtles in Florida. Proceedings 27th Annual Symposium on Sea Turtle Biology and Conservation, Myrtle Beach, SC. International Sea Turtle Society.

Smith, D. G., and R. B. Williamson. 1986. Heavy metals in the New Zealand aquatic environment: A Review. Ministry of Works and Development, Wellington, NZ.

Smith, T., D. Marchette, and R. Smiley. 1982. Life history, ecology, culture and management of Atlantic sturgeon, Acipenser oxyrhynchus oxyrhynchus, Mitchill. South Carolina. South Carolina Wildlife Marine Resources. Resources Department, Final Report to US Fish and Wildlife Service Project AFS-9 75.

Smith, T. I. 1985. The fishery, biology, and management of Atlantic sturgeon, *Acipenser oxyrhynchus,* in North America. Environmental Biology of Fishes 14(1):61-72.

Smith, T. I., E. K. Dingley, and D. E. Marchette. 1980. Induced spawning and culture of Atlantic sturgeon. The Progressive Fish-Culturist 42(3):147-151.

Smith, T. I. J., and J. P. Clugston. 1997. Status and management of Atlantic sturgeon, *Acipenser oxyrinchus*, in North America. Environmental Biology of Fishes 48(1-4):335-346.

Sorte, C. J., S. L. Williams, and J. T. Carlton. 2010. Marine range shifts and species introductions: comparative spread rates and community impacts. Global Ecology and Biogeography 19(3):303-316.

Southall, B. L., and coauthors. 2007. Marine mammal noise exposure criteria: Initial scientific recommendations. Aquatic Mammals 33(4):411-521.

Spotila, J. R. 2004. Sea turtles: A complete guide to their biology, behavior, and conservation. John Hopkins University Press, Baltimore. 227p.

Spotila, J. R., and coauthors. 1996. Worldwide population decline of *Dermochelys coriacea*: Are leatherback turtles going extinct? Chelonian Conservation and Biology 2(2):209-222.

Spotila, J. R., R. D. Reina, A. C. Steyermark, P. T. Plotkin, and F. V. Paladino. 2000. Pacific leatherback turtles face extinction. Nature 405:529-530.

SSC Pacific and NUWC. 2016. Project Work Plan for Monitoring Biofouling Removal from the ex-INDEPENDENCE (CV 62) Moored in Sinclair Inlet, Puget Sound, WA. Prepared by Space and Naval Warfare Systems Center Pacific and Naval Underwater Warfare Systems Center Newport Division for Naval Sea Systems Command, Washington, DC.

SSSRT. 2010. A Biological Assessment of Shortnose Sturgeon (*Acipenser brevirostrum*). Report to National Marine Fisheries Service, Northeast Regional Office, Gloucester, MA.

Stabile, J., J. R. Waldman, F. Parauka, and I. Wirgin. 1996. Stock structure and homing fidelity in Gulf of Mexico sturgeon (Acipenser oxyrinchus desotoi) based on restriction fragment length polymorphism and sequence analyses of mitochondrial DNA. Genetics 144(2):767-775.

Stein, A. B., K. D. Friedland, and M. Sutherland. 2004a. Atlantic sturgeon marine distribution and habitat use along the northeastern coast of the United States. Transactions of the American Fisheries Society 133(3):527-537.

Stein, A. B., K. D. Friedland, and M. Sutherland. 2004b. Atlantic sturgeon marine bycatch and mortality on the continental shelf of the Northeast United States. North American Journal of Fisheries Management 24(1):171-183.

Stevenson, J. 1997. Life history characteristics of Atlantic sturgeon (*Acipenser oxyrinchus*) in the Hudson River and a model for fishery management. Master's thesis. University of Maryland, College Park.

Stevenson, J., and D. Secor. 2000. Age determination and growth of Hudson River Atlantic sturgeon, *Acipenser oxyrinchus*. Fishery Bulletin 98(1):153-166.

Storelli, M. M., G. Barone, and G. O. Marcotrigiano. 2007. Polychlorinated biphenyls and other chlorinated organic contaminants in the tissues of Mediterranean loggerhead turtle *Caretta caretta*. Science of the Total Environment 373(2-3):456-463.

Storelli, M. M., G. Barone, A. Storelli, and G. O. Marcotrigiano. 2008. Total and subcellular distribution of trace elements (Cd, Cu and Zn) in the liver and kidney of green turtles (*Chelonia mydas*) from the Mediterranean Sea. Chemosphere 70(5):908-913.

Strayer, D. L. 2010. Alien species in fresh waters: ecological effects, interactions with other stressors, and prospects for the future. Freshwater Biology 55:152-174.

Sulak, K., and J. Clugston. 1999. Recent advances in life history of Gulf of Mexico sturgeon, Acipenser oxyrinchus desotoi, in the Suwannee River, Florida, USA: a synopsis. Journal of Applied Ichthyology 15(4-5):116-128.

Sulak, K., and coauthors. 2009. Defining winter trophic habitat of juvenile Gulf Sturgeon in the Suwannee and Apalachicola rivermouth estuaries, acoustic telemetry investigations. Journal of Applied Ichthyology 25(5):505-515.

Sulak, K. J., and J. P. Clugston. 1998. Early life history stages of Gulf sturgeon in the Suwannee River, Florida. Transactions of the American Fisheries Society 127(5):758-771.

Swimmer, Y., and coauthors. 2017. Sea Turtle Bycatch Mitigation in US Longline Fisheries. Frontiers in Marine Science 4:260.

Swingle, W. M., S. G. Barco, T. D. Pitchford, W. A. Mclellan, and D. A. Pabst. 1993. Appearance of juvenile humpback whales feeding in the nearshore waters of Virginia. Marine Mammal Science 9(3):309-315.

Sylvester, F., and coauthors. 2011. Hull fouling as an invasion vector: can simple models explain a complex problem? Journal of Applied Ecology 48(2):415-423.

Szymanski, M. D., and coauthors. 1999. Killer whale (Orcinus orca) hearing: Auditory brainstem response and behavioral audiograms. The Journal of the Acoustical Society of America 106:1134.

Tabor, R. A., H. A. Gearns, C. M. McCoy III, and S. Camacho. 2006. Nearshore habitat use by Chinook salmon in lentic systems of the Lake Washington basin. U.S. Fish and Wildlife Service, Lacy, Washington.

Tal, D., H. Shachar-Bener, D. Hershkovitz, Y. Arieli, and A. Shupak. 2015. Evidence for the initiation of decompression sickness by exposure to intense underwater sound. Journal of Neurophysiology 114(3):1521-1529.

Tapilatu, R. F., and coauthors. 2013. Long-term decline of the western Pacific leatherback, *Dermochelys coriacea*: A globally important sea turtle population. Ecosphere 4:15.

Taubert, B. D. 1980. Reproduction of shortnose sturgeon (*Acipenser brevirostrum*) in Holyoke Pool, Connecticut River, Massachusetts. Copeia:114-117.

Terdalkar, S., A. S. Kulkarni, S. N. Kumbhar, and J. Matheickal. 2005. Bio-economic risks of ballast water carried in ships, with special reference to harmful algal blooms. Nature, Environment and Pollution Technology 4(1):43-47.

TEWG. 2000. Assessment update for the Kemp's ridley and loggerhead sea turtle populations in the western North Atlantic. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Turtle Expert Working Group.

TEWG. 2007. An assessment of the leatherback turtle population in the Atlantic Ocean. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Turtle Expert Working Group.
Tolan, J. M. 2007. El Nino-Southern Oscillation impacts translated to the watershed scale: Estuarine salinity patterns along the Texas Gulf Coast, 1982 to 2004. Estuarine, Coastal and Shelf Science 72:247-260.
Tolimieri, N., and P. S. Levin. 2005. The roles of fishing and climate in the population dynamics of bocaccio rockfish. Ecological Applications 15:458-468.
Tomás, J., and J. A. Raga. 2008. Occurrence of Kemp's ridley sea turtle (*Lepidochelys kempii*) in the Mediterranean. Marine Biodiversity Records 1:1-2.
Tonnes, D., and coauthors. 2016. Yelloweye rockfish (*Sebastes ruberrimus*), canary rock-fish (*Sebastes pinniger*), and bocaccio (*Sebastes paucispinis*) of the Puget Sound/Georgia Basin: 5-Year Review: Summary and Evaluation. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources, Seattle, Washington.
Tracy, C. 1990. Memorandum: Green sturgeon meeting and comments. State of Washington Department of Fisheries.
Troëng, S., D. R. Evans, E. Harrison, and C. J. Lagueux. 2005. Migration of green turtles *Chelonia mydas* from Tortuguero, Costa Rica. Marine Biology 148(2):435-447.
Tsikhon-Lukanina, E. I., and O. G. Reznichenko. 1991. Diet of the ctenophore Mnemiopsis in the Black Sea as a function of size. Oceanology 31(3):320-323.
Twiner, M. J., and coauthors. 2011. Concurrent exposure of bottlenose dolphins (Tursiops truncatus) to multiple algal toxins in Sarasota Bay, Florida, USA. PLoS One 6(3):e17394.
Twiner, M. J., and coauthors. 2012. Comparative analysis of three brevetoxin-associated bottlenose dolphin (Tursiops truncatus) mortality events in the Florida panhandle region (USA). PLoS One 7(8):e42974.
U.S. Army Corps of Engineers. 2014. Brazos Island Harbor, Texas channel improvement project; Final integrated feasibility report - Environmental Assessment. Galveston District Southwestern Division, Galveston, Texas.
U.S. Army Corps of Engineers. 2016. Draft integrated reevaluation report and supplemental environmental impact statement, New Orleans, Louisiana.
U.S. Army Corps of Engineers. 2018. Lower Mississippi - A brief history. Mississippi Valley Division.
University of North Florida. 2017. State of the Lower St. Johns River Basin, Florida: River Report. Prepared for the Environmental Protection Board, City of Jacksonville, Florida, Jacksonville, Florida.
USCOP. 2004. An ocean blueprint for the 21st century. Final report. U.S. Commission on Ocean Policy, Washington, D. C.
USDOT. 2012. In-water Hull Cleaning Summary Report, US DOT Maritime Administration, Alameda California. U.S Department of Transportation, U.S. Maritime Administration.
USFWS. 1995. Gulf Sturgeon Recovery Plan. Pages 170 *in* U. S. F. W. Service, editor. Gulf States Marine Fisheries Commission, Atlanta, GA.
USFWS. 2004. Fisheries Resources Annual Report. Pages 47 *in* U. S. F. W. Service, editor, Panama City, FL.

USFWS. 2007. Fisheries Resources Annual Report. Pages 37 *in* U. S. F. W. Service, editor, Panama City, Fl.

USFWS. 2009a. Fisheries Resources Annual Report. Pages 56 *in* U. S. F. W. Service, editor, Panama City, FL.

USFWS. 2009b. Gulf Sturgeon (Acipenser oxyrinchus destoi) 5-year review: summary and evaluation. S. R. USFWS, Panama City Ecological Services Field Office, National Marine Fisheries Service, Office of Protected Resources, editor, St. Petersburg, FL.

USFWS, and NMFS. 1998. Endangered Species Act Consultation Handbook: Procedures for Conducting Section 7 Consultations and Conferences. U.S. Fish and Wildlife Service and National Marine Fisheries Service.

USFWS, and NMFS. 2009. Gulf sturgeon (*Acipenser oxyrinchus desotoi*) 5-year review: Summary and evaluation. U.S. Fish and Wildlife Service and National Marine Fisheries Service.

USGCRP. 2018. Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment, Volume II. U.S. Global Change Research Program, Washington, DC, USA.

USGS. 2018. Nonindigenous Aquatic Species Database. Gainesville, FL.

USN. 2008. Biological evaluation for the Gulf of Mexico rangle complex. U.S. Navy.

USN. 2009. Gulf of Mexico range complex final environmental impact statement/overseas environmental impact statement (EIS/OEIS) volume 1 (version 3). United States Navy, Norfolk, Virginia.

Valkirs, A., and coauthors. 1994. Environmental effects from in-water hull cleaning of ablative copper antifouling coatings. DTIC Document.

Van de Merwe, J. P. V., and coauthors. 2009. Chemical contamination of green turtle (*Chelonia mydas*) eggs in peninsular Malaysia: Implications for conservation and public health. Environmental Health Perspectives 117(9):1397-1401.

van der Gaag, M., G. van der Velde, S. Wijnhoven, and R. S. Leuven. 2016. Salinity as a barrier for ship hull-related dispersal and invasiveness of dreissenid and mytilid bivalves. Marine Biology 163(7):147.

Van Der Kraak, G., and N. W. Pankhurst. 1997. Temperature effects on the reproductive performance of fish. Pages 159-176 *in* C. M. W. a. D. G. McDonald, editor. Global warming: implications for freshwater and marine fish. Cambridge University Press, Cambridge, United Kingdom.

Van Dolah, F. M. 2005. Effects of harmful algal blooms. Marine mammal research: Conservation beyond crisis:85-99.

Van Eenennaam, J., and S. Doroshov. 1998. Effects of age and body size on gonadal development of Atlantic sturgeon. Journal of fish biology 53(3):624-637.

Van Eenennaam, J. P., and coauthors. 2006. Reproductive conditions of the Klamath River green sturgeon. Transactions of the American Fisheries Society 135(1):151-163.

Vanderlaan, A. S., and C. T. Taggart. 2007. Vessel collisions with whales: The probability of lethal injury based on vessel speed. Marine Mammal Science 23(1):144-156.

Vargo, S., P. Lutz, D. Odell, E. Van Vleet, and G. Bossart. 1986. Study of the effects of oil on marine turtles. Available from the National Technical Information Service, Springfield VA. 22161, as PB 87-199923 1.

Vinogradov, M. Y., E. A. Shuskina, E. I. Musayeva, and P. Y. Sorokin. 1989. A newly acclimated species in the Black Sea: the ctenophore Mnemiopsis leidyi (Ctenophora: Lobata). Oceanology 29(2):220-224.

Vinson, M. R., and M. A. Baker. 2008. Poor growth of rainbow trout fed New Zealand mud snails Potamopyrgus antipodarum. North American Journal of Fisheries Management 28(3):701-709.

Waldman, J., C. Grunwald, J. Stabile, and I. Wirgin. 2002. Impacts of life history and biogeography on the genetic stock structure of Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus*, Gulf sturgeon *A. oxyrinchus desotoi*, and shortnose sturgeon *A. brevirostrum*. Journal of Applied Ichthyology 18(4-6):509-518.

Waldman, J. R., and coauthors. 2013. Stock origins of subadult and adult Atlantic Sturgeon, *Acipenser oxyrinchus*, in a non-natal estuary, Long Island Sound. Estuaries and Coasts 36(2):257-267.

Waldman, J. R., and I. I. Wirgin. 1998. Status and restoration options for Atlantic sturgeon in North America. Conservation Biology 12(3):631-638.

Wallace, B. P., and coauthors. 2010a. Regional managment units for marine turtles: A novel framework for prioritizing conservation and research across multiple scales. PLoS One 5(12):e15465.

Wallace, B. P., S. S. Kilham, F. V. Paladino, and J. R. Spotila. 2006. Energy budget calculations indicate resource limitation in Eastern Pacific leatherback turtles. Marine Ecology Progress Series 318:263-270.

Wallace, B. P., and coauthors. 2010b. Global patterns of marine turtle bycatch. Conservation Letters 3(3):131-142.

Wallace, B. P., and coauthors. 2007. Maternal investment in reproduction and its consequences in leatherback turtles. Oecologia 152(1):37-47.

Wallentinus, I., and C. D. Nyberg. 2007. Introduced marine organisms as habitat modifiers. Marine Pollution Bulletin 55(7-9):323-332.

Wambiji, N., P. Gwada, E. Fondo, S. Mwangi, and M. K. Osore. 2007. Preliminary results from a baseline survey of the port of Mombasa: with focus on molluscs. 5th Western Indian Ocean Marine Science Association Scientific Symposium; Science, Policy and Management pressures and responses in the Western Indian Ocean region, Durban, South Africa.

Waring, C. P., and A. Moore. 2004. The effect of atrazine on Atlantic salmon (Salmo salar) smolts in fresh water and after sea water transfer. Aquatic toxicology 66(1):93-104.

Waring, G., Elizabeth Josephson, Katherine Maze-Foley, and P. E. Rosel. 2013. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments - 2012. National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Waring, G. T., E. Josephson, K. Maze-Foley, and P. E. Rosel. 2015. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments - 2014, NOAA Tech Memo NMFS NE 231.

Washington Department of Fish and Wildlife (WDFW). 1993. 1992 Washington state salmon and steelhead stock inventory (SASSI) WDFW and Western Washington Treaty Indian Tribes, Olympia, Washington.

Watson, J. W., S. P. Epperly, A. K. Shah, and D. G. Foster. 2005. Fishing methods to reduce sea turtle mortality associated with pelagic longlines. Canadian Journal of Fisheries and Aquatic Sciences 62(5):965-981.

WDFW, and ODFW. 2001. Washington and Oregon eulachon management plan. Washington Department of Fish and Wildlife and Oregon Department of Fish and Wildlife.

WDFW, and PNPTT. 2000. Summer chum salmon conservation initiative. Washington Department of Fish and Wildlife and the Point No Point Treaty Tribes.

Wells, R. S., and coauthors. 2008. Consequences of injuries on survival and reproduction of common bottlenose dolphins (Tursiops truncatus) along the west coast of Florida. Marine Mammal Science 24(4):774-794.

Wigand, C., J. C. Stevenson, and J. C. Cornwell. 1997. Effects of different submersed macrophytes on sediment biogeochemistry. Aquatic Botany 56:233-244.

Wilcove, D. S., and L. Y. Chen. 1998. Management costs for endangered species. Conservation Biology 12(6):1405-1407.

Wilcove, D. S., D. Rothstein, J. Dubow, A. Phillips, and E. Losos. 1998. Quantifying threats to imperiled species in the United States. BioScience 48(8):607-615.

Wiley, D. N., R. A. Asmutis, T. D. Pitchford, and D. P. Gannon. 1995. Stranding and mortality of humpback whales, *Megaptera novaeangliae*, in the mid-Atlantic and southeast United States, 1985-1992. Fishery Bulletin 93(1):196-205.

Wilkinson, C., and D. Souter. 2008. Status of Caribbean coral reefs after bleaching and hurricanes in 2005. Global Coral Reef Monitoring Network, and Reef and Rainforest Research Centre, Townsville.

Willford, W., and coauthors. 1981. Chlorinated hydrocarbons as a factor in the reproduction and survival of lake trout (*Salvelinus namaycush*). U.S. Dept. of the Interior, U.S. Fish and Wildlife Service.

Williams, S. E., L. P. Shoo, J. L. Isaac, A. A. Hoffmann, and G. Langham. 2008. Towards an integrated framework for assessing the vulnerability of species to climate change. PLoS Biol 6(12):e325.

Willis-Norton, E., and coauthors. 2015. Climate change impacts on leatherback turtle pelagic habitat in the Southeast Pacific. Deep Sea Research Part II: Topical Studies in Oceanography 113:260-267.

Wirgin, I., and coauthors. 2005. Range-wide population structure of shortnose sturgeon *Acipenser brevirostrum* based on sequence analysis of the mitochondrial DNA control region. Estuaries 28(3):406-421.

Wirgin, I., C. Grunwald, J. Stabile, and J. Waldman. 2007. Genetic evidence for relict Atlantic sturgeon stocks along the mid-Atlantic coast of the USA. North American Journal of Fisheries Management 27(4):1214-1229.

Wirgin, I., C. Grunwald, J. Stabile, and J. R. Waldman. 2010. Delineation of discrete population segments of shortnose sturgeon *Acipenser brevirostrum* based on mitochondrial DNA control region sequence analysis. Conservation Genetics 11(3):689-708.

Wirgin, I., L. Maceda, C. Grunwald, and T. King. 2015. Population origin of Atlantic sturgeon *Acipenser oxyrinchus oxyrinchus* by-catch in US Atlantic coast fisheries. Journal of fish biology 86(4):1251-1270.

Witherington, B., P. Kubilis, B. Brost, and A. Meylan. 2009. Decreasing annual nest counts in a globally important loggerhead sea turtle population. Ecological Applications 19(1):30-54.

Wolfe, S. H., J. A. Reidenauer, and D. B. Means. 1988. An ecological characterization of the Florida Panhandle. U.S. Fish and Wildlife Service and MMS, New Orleans, Louisiana.

Woodbury, D., and S. Ralston. 1991. Interannual variation in growth rates and back-calculated birth-date distributions of pelagic juvenile rockfishes (*Sebastes* spp.) off the central California coast. Fishery Bulletin 89(3):523-533.

Woody, C. A. 2007. Effects on Freshwater Food Chains and Salmon: A Literature Review. Fisheries Research and Consulting:1-18.

Wooley, C. M., and E. J. Crateau. 1985. Movement, microhabitat, exploitation, and management of Gulf of Mexico sturgeon, Apalachicola River, Florida. North American Journal of Fisheries Management 5(4):590-605.

Work, P. A., A. L. Sapp, D. W. Scott, and M. G. Dodd. 2010. Influence of small vessel operation and propulsion system on loggerhead sea turtle injuries. Journal of Experimental Marine Biology and Ecology 393(1-2):168-175.

Wuebbles, D. J., and coauthors. 2017. Executive Summary. Pages 12-34 *in* D. J. Wubbles, and coeditors, editors. Climate Science Special Report: Fourth National Climate Assessment, volume 1. U.S. Global Change Research Program, Washington D.C.

WWF. 2008. Tributyltin canned. WWF Global, http://wwf.panda.org/?145704/tributyltin-canned.

Wydoski, R., and R. Whitney. 1979. Inland fishes of Washington. University of Washington Press.

Yamanaka, K. L., and coauthors. 2006. A review of yelloweye rockfish Sebastes ruberrimus along the Pacific coast of Canada: Biology, distribution, and abundance trends. Canadian Science Advisory Secretariat Research Document, 2006/076.

Yender, R., J. Michel, and C. Lord. 2002. Managing seafood safety after an oil spill. National Oceanic and Atmospheric Administration, National Ocean Service, Office of Response and Restoration, Seattle, Washington.

Yuan, W. S., L. J. Walters, S. A. Brodsky, K. R. Schneider, and E. A. Hoffman. 2016. Synergistic effects of salinity and temperature on the survival of two nonnative bivalve molluscs, Perna viridis (Linnaeus 1758) and Mytella charruana (d'Orbigny 1846). Journal of Marine Biology 2016.

Zeitsev, Y., and B. Ozturk. 2001. Exotic species in the Aegean, Marmara, Black, Azov, and Caspian Seas.1-267.

Ziegeweid, J., C. Jennings, and D. Peterson. 2008a. Thermal maxima for juvenile shortnose sturgeon acclimated to different temperatures. Environmental Biology of Fishes 82(3):299-307.

Ziegeweid, J. R., C. A. Jennings, D. L. Peterson, and M. C. Black. 2008b. Effects of salinity, temperature, and weight on the survival of young-of-year shortnose sturgeon. Transactions of the American Fisheries Society 137:1490-1499.

Zimmer, W. M. X., and P. L. Tyack. 2007. Repetitive shallow dives pose decompression risk in deep-diving beaked whales. Marine Mammal Science 23(4):888-925.

Zurita, J. C., and coauthors. 2003. Nesting loggerhead and green sea turtles in Quintana Roo, Mexico. Pages 125-127 *in* J. A. Seminoff, editor. Proceedings of the 22nd Annual Symposium on Sea Turtle Biology and Conservation, Miami, Florida.

Zwinenberg, A. J. 1977. Kemp's ridley, *Lepidochelys kempii* (Garman 1880), undoubtedly the most endangered marine turtle today (with notes on the current status of *Lepidochelys olivacea*). Bulletin of the Maryland Herpetological Society 13(3):378-384.