UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
Region 1
5 Post Office Square, Suite 100
BOSTON, MA 02109-3912

CERTIFIED MAIL – RETURN RECEIPT REQUESTED

December 28, 2016

NOAA National Marine Fisheries Service
Protected Resources Division
55 Great Republic Drive
Gloucester, MA 01930

Attn: Mrs. Kimberly Damon-Randall

Re: Re-Issuance of the National Pollutant Discharge Elimination System (NPDES) General Permit for Remediation Activity Discharges – The Remediation General Permit (RGP); NPDES Permit MAG910000 and NHG910000

Dear Mrs. Damon-Randall,

The U.S. Environmental Protection Agency Region 1 (EPA) is proposing to reissue an NPDES general permit for remediation activity discharges to certain waters of the United States in the Commonwealth of Massachusetts and the State of New Hampshire as described below. This letter is to request Endangered Species Act (ESA) concurrence from your office for the proposed reissuance of the RGP. EPA has made the determination that the proposed reissuance may affect, but is not likely to adversely affect, any listed threatened or endangered species or their critical habitat under the jurisdiction of NMFS under the ESA of 1973, as amended. EPA's supporting analysis is provided below.

For your convenience, a copy of the draft RGP and fact sheet was previously provided. This information, as well as all appendices to the draft RGP, can also be found at: https://www3.epa.gov/region1/npdes/rgp.html. The Notice of Availability of the draft RGP was published in the Federal Register on Thursday, August 18, 2016.

1. Proposed Action

Section 301(a) of the Clean Water Act (the Act) provides that the discharge of pollutants is unlawful except in accordance with a NPDES permit unless such a discharge is otherwise authorized by the Act. The NPDES permit program must regulate the discharge of point sources of pollutants to waters of the United States under 40 CFR § 122.1(b)(1). EPA is proposing to reissue the RGP for sites located in Massachusetts and New Hampshire which discharge as a result of remediation activities grouped into eight general categories: 1) Petroleum-related site remediation; 2) Non-petroleum-related site remediation; 3) Contaminated/formerly contaminated

site dewatering; 4) Pipeline and tank dewatering; 5) Aquifer pump testing; 6) Well development/rehabilitation; 7) Dewatering/remediation of collection structures; and 8) Dredge-related dewatering. Once final, the Draft RGP will replace the RGP that expired on September 9, 2015 and has been administratively continued. The RGP will provide authorization to discharge to certain waters of the Commonwealth of Massachusetts and the State of New Hampshire. Discharges to certain receiving waters, such as Class A waters; Outstanding Resource Waters in New Hampshire; Ocean Sanctuaries in Massachusetts; Discharges to territorial seas; or discharges which are inconsistent with the State Coastal Zone Management program will not be authorized under the permit. See Section I.D. of the Fact Sheet for a complete listing of eligibility requirements and coverage exclusions.

The effluent generated from these point sources are all generated by substantially similar operations, which involve remediation, dewatering and dewatering-/remediation-related activities conducted at contaminated or formerly contaminated sites. These discharges may contain a variety of conventional, non-conventional and toxic pollutants. The pollutants of concern for a given individual site depend upon the type of influent. Pollutants may include one or more individual pollutant parameters from chemical groups present or likely present at contaminated or formerly contaminated sites, such as: 1) inorganics (e.g., metals, solids, nutrients); 2) non-halogenated volatile organic compounds (VOCs) (e.g., benzene, toluene, ethylbenzene and xylenes); 3) halogenated VOCs (e.g., chlorinated solvents); 4) non-halogenated semi-volatile organic compounds (SVOCs) (e.g., polycyclic aromatic hydrocarbons); 5) halogenated SVOCs (e.g., polychlorinated biphenyls); and 6) fuels parameters (e.g., petroleum hydrocarbons, petroleum additives and oxygenates). The Draft RGP contains provisions for the variations expected across sites and activities.

The RGP was first issued by EPA Region 1 on September 9, 2005 (2005 RGP) and reissued on September 10, 2010 (2010 RGP). Since September 9, 2005, EPA has authorized approximately 750 discharges under the RGP. EPA issued authorization to discharge under the 2010 RGP to 275 sites located in Massachusetts and 23 sites located in New Hampshire. The types of sites EPA expects to request coverage under the RGP are not expected to change. The majority of sites EPA expects to authorize under this General Permit will discharge a small volume of water, intermittently, for a short period, following treatment. The treatment processes allowed under this General Permit include: 1) Adsorption/Absorption, 2) Advanced Oxidation Processes, 3) Air Stripping; 4) Granulated Activated Carbon ("GAC")/Liquid Phase Carbon Adsorption; 5) Ion Exchange; 6) Precipitation/Coagulation/Flocculation; and 7) Separation/Filtration. Permittees are required to develop, implement, and maintain a Best Management Practices (BMP) plan to prevent or minimize the concentration of pollutants (biological, chemical and physical) in the effluent discharged to surface waters.

The RGP establishes Notice of Intent (NOI), Notice of Change (NOC), and Notice of Termination (NOT) requirements, effluent limitations and requirements, and standard and special conditions for sites that discharge 1.0 million gallons per day (MGD) or less in Massachusetts and New Hampshire. The Draft RGP includes "end-of-pipe" effluent limitations that all permittees are required to meet for effluent flow, pH, temperature and 58 pollutant parameters for discharges from sites based on the type of remediation activity and the receiving water of the discharge.

The permit includes technology-based effluent limits as well as water-quality based effluent limits, when a water-quality based effluent limit is more stringent than a technology-based limit for a pollutant. All discharges eligible for coverage under the RGP are subject to **"end-of-pipe" effluent limitations** and requirements, regardless of the type of site. Effluent limitations for inorganic pollutants apply to all sites. In addition, the effluent limitation for any pollutant applies to any site where that pollutant is present. Effluent limitations for all other pollutant parameters may or may not apply, and depend on the activity category of a site, the contamination type subcategory, and the classification of the receiving water.

Part 2 of the Draft RGP includes the Effluent Limitations and Monitoring Requirements (including frequency) for the Commonwealth of Massachusetts and State of New Hampshire, and the Special Conditions (including Best Management Practices) for both states. Section III of the Fact Sheet provides an explanation of the effluent limitations under this General Permit. The effluent limitations for all pollutants are identical, except where the appropriate State allows calculation of water quality-based effluent limitations adjusted for available dilution. Although the water-quality based effluent limits do allow for consideration of available dilution (See Appendix V for sites in Massachusetts and Appendix VI for sites in New Hampshire), the RGP does *not* establish mixing zones. Therefore, the applicable limitations and monitoring requirements are the same for all sites excepting the site-specific variation in the activities, the types of contaminants, and the receiving water(s). Further, the RGP contains conditions for toxicity testing and/or a priority pollutant scan if warranted. In addition, EPA may require individual permits be issued if actual environmental conditions (including the preservation of endangered species) are not adequately addressed by this general permit.

Part 4 of the Draft RGP indicates additional monitoring and other sampling requirements, including record keeping and reporting requirements. Monitoring and reporting are required under the permit for all discharges in order to ensure compliance with state (MA: 314 CMR 4.00; NH: Env-Wq 1700) and federal surface water quality standards to ensure that the water quality of the receiving water is protected. All discharges must be monitored and reported in accordance with the permit. The permit will authorize discharge up to 1.0 MGD. The inclusion of a maximum effluent flow is a change from the expired permit. Although, actual effluent flow has typically been reported at flow rates significantly less than 1.0 MGD at sites covered under the RGP, EPA will consider discharges above 1 MGD, on a case by case basis. In such cases, EPA will take into consideration any ESA-listed species and critical habitat within the vicinity of the discharge when evaluating the appropriateness of such a site's request for coverage.

The permit also requires remediation sites to conduct acute whole effluent toxicity (WET) testing of a proposed discharge. The results from the acute WET testing will provide EPA with a better understanding of any adverse synergistic/cumulative impact the discharge has on living species. The 2016 RGP specifically excludes coverage to facilities whose discharge(s) are likely to jeopardize the continued existence of listed threatened or endangered species or the critical habitat of such species.

In addition to the numeric effluent limitations, the draft RGP also contains several non-numeric technology-based effluent limitations and water quality requirements. For example, the RGP retains requirements for permittees to develop, implement, and maintain a BMP Plan and to

document how both the non-numeric technology-based and numeric effluent limitations are being met through the selection, design, installation, and implementation of control measures (including BMPs). The RGP includes several specific BMPs of all permittees, including pollutant minimization and waste management. The RGP also retains restrictions on discharges of chemicals and additives that are commonly used during remediation activities or for treatment directly that could be present in discharges. The purpose of these requirements is to prevent or minimize the concentration of pollutants (biological, chemical and physical) in the wastewater discharged to surface waters. The BMP Plan, the specific BMPs required of all permittees, and conditions for the discharge of chemicals and additives is discussed in more detail in Section III.D of the Fact Sheet.

This RGP will replace the previous RGP that expired September 9, 2015, and has been administratively continued for permittees until the permit is reissued. The Notice of Availability of this Draft RGP was published in the Federal Register on August 18, 2016. After a 30-day comment period, EPA will address any significant comments and make the necessary revisions. After being published in the Federal Register, the final permit will then be reissued. EPA's reissuance of this RGP will be for a subsequent five year permit term.

Section I.A.1 of the Fact Sheet highlights the changes that were made from the expired permit. Key changes include: additional limitations or monitoring requirements for pollutants either not included or not limited in the expired RGP; revised limitations for multiple pollutants, including more stringent limitations for metals; additional BMP requirements; increased specificity for sampling requirements, including additional Notice of Intent (NOI) sampling requirements (of both effluent and upstream ambient water and acute Whole Effluent Toxicity testing, for certain sites).

The Massachusetts Department of Environmental Protection (MassDEP) and the New Hampshire Department of Environmental Services (NHDES) will review the protectiveness of the permit and provide water quality certification. In addition, EPA expects MassDEP to issue the RGP as a state permit in Massachusetts.

2. Description of the Action Area

The Action Area is defined as "all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action." 50 CFR §402.02. The entire universes of facilities that will apply for and obtain coverage under the RGP is unknown at the time the draft permit is published for public comment. The Action Area could include any surface water in Massachusetts and New Hampshire, excluding those waterbodies to which discharges are not authorized (See Section 1.D of the Fact Sheet and Part 1.3 of the Draft Permit). For example, discharges are not authorized under the RGP to: Class A waters in Massachusetts and New Hampshire; Outstanding Resource Waters in New Hampshire; Ocean Sanctuaries; and the territorial seas.

Although the Action Area could encompass numerous surface waters in Massachusetts and New Hampshire, for the purposes of this consultation, the Action Area of the General Permit will be restricted to those waters where there is a known presence of ESA species or designated critical habitat. Existing discharges to these waterbodies will be considered in evaluating the effects of

EPA's reissuance of the General Permit on listed species and critical habitat. Currently, there are several waterbodies where EPA has considered whether ESA species could be impacted by permitted discharges: 1) the Connecticut River (from Turner's Falls, downstream through Holyoke (including Holyoke Dam region); 2) the Merrimack River below the Essex Dam (Merrimack River Dam) in Lawrence and downstream (including Haverhill); 3) Cape Cod Bay; 4) the Taunton River; 5) Massachusetts Bay; 6) the Piscataqua River/Great Bay Estuary in New Hampshire; and 7) coastal embayments and nearshore marine waters of Massachusetts and New Hampshire. EPA has also considered the land areas adjacent to these waterbodies.

To establish the Action Area, EPA also considered other areas in which the effects of the action are likely to occur. This assessment considers direct and indirect effects of the action on listed species or critical habitat, together with the effects of other activities that are interrelated or interdependent with the action, that will be added to the environmental baseline. Indirect effects are those that are caused by the proposed action and are later in time, but still are reasonably certain to occur. Interrelated actions are those that are part of a larger action and depend on the larger action for their justification. Interdependent actions are those that have no independent utility apart from the action under consideration". 50 CFR §402.02. Thus, the action area for this General Permit may include:

- Flow pathway to discharge area
- Discharge area
- Area extending a short distance from the discharge area during discharge

The Action Area of the RGP includes site discharges to the waterbodies described below. Baseline information for each waterbody is also provided. This aided in the analysis of any impacts that remediation activity discharges might have on the ESA listed species or their critical habitat (which is discussed in Section 4 of this document). As previous noted above, approximately 750 sites in Massachusetts and New Hampshire have been covered under the RGP since 2005. EPA expects that a portion of these facilities will reapply for coverage when the RGP is reissued. Therefore, EPA believes that it is appropriate to use discharge data from current and recently covered permittees to predict the effect of future discharges on ESA species and critical habitat: discharges from the sites are sufficiently similar to warrant coverage under a general permit (see Section I.B. of the draft RGP fact sheet) and are considered representative in determining impacts to aquatic species.

a. Connecticut River

The Connecticut River Watershed is the largest river ecosystem in New England, encompassing approximately 11,000 square miles and spanning over four New England states, including Vermont, New Hampshire, Massachusetts, and Connecticut (Executive Office of Environmental Affairs, n.d.). From its origin near the Canadian border, the 410-mile Connecticut River flows southward to form the boundary between New Hampshire and Vermont (Carr & Kennedy, 2008). The Upper Connecticut River, the name for the river in NH and VT, spans approximately 255 miles. In New Hampshire, the river begins in the town of Pittsburg, NH (at the outlet of Fourth Connecticut Lake), flows through 26 communities, and drains approximately 3,046 square miles

(NHDES, 2008). The Connecticut River (in both NH and VT) was designated into the NH Rivers Management and Protection Program in 1992 (NHDES, 2008).

The river then enters Massachusetts (near the Town of Northfield) and drains all or part of 45 municipalities before entering Connecticut (near the Towns of Agawam and Longmeadow) (Executive Office of Environmental Affairs, n.d.). The Middle Connecticut River usually refers to the stretch from Massachusetts through Central Connecticut, while the Lower Connecticut River includes the portion in southern CT which then empties into Long Island Sound. This assessment will focus on the lower Connecticut River (including waters in Massachusetts downstream of Turner Falls), based on the population and distribution of ESA listed species, described in Section 3, below. EPA did not evaluate sites that will discharge to tributaries of the Connecticut River in this assessment. EPA assumes that tributary discharges will cause insignificant or discountable water quality impacts, if any, to the habitat of the mainstem of the Connecticut River due to the extremely high dilution and mixing of the small volume discharges with the receiving water tributaries.

According to NH's final 2012 303(d) list, eighteen segments of the Connecticut River were listed as impaired waters in NH that require a TMDL (NHDES, 2014). The most common impairment was pH, while lead, aluminum, and benthic-macroinvertebrate bioassessments were listed as occasional impairments under the aquatic life use category. However, the prioritization for development of TMDLs to address these concerns was categorized as "Low."

The Connecticut River is classified in the Massachusetts Surface Water Quality Standards as a Class B – warm water fishery (Carr & Kennedy, 2008). Segments MA34-01, MA34-02, MA34-03, MA34-04, and MA34-05, which cover the length of the Connecticut River from the New Hampshire/Massachusetts state line in the north to Massachusetts/Connecticut state line in the south, were listed as Category 5 – Impaired waters that requires a TMDL (MassDEP, 2013). The listed impairments included bacterial contamination from *E. coli* and nutrient enrichment from wet weather discharges, such as combined sewage outflows; high turbidity (total suspended solids or TSS); flow regime and streamside alterations from anthropologic activities including nearby hydro-electric facilities; and PCBs in fish tissue from unknown sources.

b. Merrimack River

The Merrimack River is the second largest river in New England and its watershed drains approximately 5,014 square miles as it travels from the White Mountain region of New Hampshire to east-central Massachusetts (NHDES, 2008). The Upper Merrimack River begins at the confluence of the Pemigewasset and Winnipesaukee Rivers (near Franklin, NH), and then flows for approximately 30 miles to the town of Bow, NH. Although the Upper Merrimack River flows through Concord, NH, almost 80% of the land within three quarter miles of the river is currently undeveloped as forest, farm, or wetland (NHDES, 2008). As such, this stretch of the river has a high level of water quality, provides valuable habitat for plants and animals, and was designated under the NH Rivers Management and Protection Program in 1990 (NHDES, 2008). A Designated River is managed and protected for its outstanding natural and cultural resources (NHDES, 2014). The Lower Merrimack River in NH was also designated under the NH Rivers Management and Protection Program (NHDES, 2008). This segment begins at the Merrimack-

Bedford town line and flows approximately 15 miles through Merrimack and then Nashua, before entering the Commonwealth of Massachusetts.

According to NH's 2012 303(d) list, three sections of the Upper Merrimack River (near Concord and Bow) were listed as impaired for pH, dissolved oxygen or aluminum (NHDES, 2014). Five segments of the Lower Merrimack River, including areas near Manchester and Nashua, were also on the 303(d) list. Likewise, these segments were impaired for pH, dissolved oxygen or aluminum, under the aquatic life use category.

Approximately 24% of the Merrimack River Watershed is located in Massachusetts. However, the Commonwealth of MA defines the Merrimack River Watershed on a smaller scale by excluding the Nashua, SuAsCo, Shawsheen River Watersheds, and all of the NH watersheds. (Executive Office of Environmental Affairs, 2001). This watershed encompasses all or parts of 24 MA communities. It also includes over 50 miles of the Merrimack River, from the New Hampshire border until it flows into the Atlantic Ocean at Newburyport and Salisbury.

As previously mentioned, the Massachusetts Surface Water Quality Standards (SWQS) assign all inland and coastal and marine waters to classes according to the intended beneficial uses of those waters (MassDEP, 2006). The Merrimack River in Massachusetts is classified as Class B, warm water fishery from the New Hampshire border to Haverhill (near the confluence of the Little River), while the 22-mile tidal section from Haverhill to the ocean is designated as Class SB (Meek & Kennedy, 2010).

According to the Massachusetts Year 2012 Integrated List of Waters, new water quality assessments were conducted for five specific watersheds and/or drainage areas, including the Merrimack River Watershed. Based on that data, the Merrimack River (from the state line to the mouth near the Atlantic Ocean) as well as other water bodies within the watershed were listed as Category 5 (MassDEP, 2013). Waters that fall under Category 5 are impaired waters that require a Total Maximum Daily Load, or TMDL, because the waterbodies are not meeting designated uses under technology-based controls. Pollutants include pathogens, such as coliform and *E. coli*, PCBs and mercury in fish tissue, and phosphorus (total). Wet weather discharges, including those from point sources, combined sewer overflow and urban runoff, are the major sources for the pathogens and nutrients. Atmospheric deposition causes the mercury in fish tissue, while the specific source of the PCBs is unknown (Executive Office of Environmental Affairs, 2001).

The Merrimack River Watershed does have a draft Pathogen TMDL (MADEP, Regioni, & International, Draft Pathogen TMDL for the Merrimack River Watershed). TMDLs determine the amount of a pollutant that a waterbody can safely assimilate without violating water quality standards. The TMDL process is designed to assist states and watershed stakeholders in the implementation of water quality-based controls specifically targeted to identify source(s) of pollution in order to restore and maintain the quality of their water resources. It should also be noted that EPA approved the Northeast Regional Mercury Total Maximum Daily Load (TMDL) on December 20, 2007 (CTDEP, et al., 2007). The TMDL applies to all six New England states as well as the state of New York. It outlines a strategy for reducing mercury concentrations in fish in Northeast fresh waterbodies so that water quality standards can be met. A final addendum

to this TMDL for the state of Massachusetts was finalized in September of 2012 (MassDEP, 2012).

c. Cape Cod Bay

The state of Massachusetts encompasses two geological provinces, namely the Coastal Plain and the New England Upland (MassDEP, 2013); Cape Cod (and the islands) form the coastal plain. The Cape Cod Watershed extends 70 miles into the Atlantic Ocean and is surrounded by the salt waters of Buzzards Bay, Cape Cod Bay, the Atlantic Ocean, and Nantucket Sound. The watershed includes the 15 towns that comprise Barnstable County. It also encompasses a drainage area of approximately 440 square miles and includes 559 miles of coastline, 360 ponds, 145 public water supply wells, and 8 areas of Critical Environmental Concern (ACEC) (EOEEA, c). In addition to the highly significant environmental resources of these ACEC, such as the Inner Cape Cod Bay, the Cape also supports a number of Class SA waters, including the waters in and adjacent to the Cape Cod National Seashore (MassDEP, 2006). As stated previously in this document, dewatering discharges to ACECs (along with other categories listed in Section 1.D. of the Fact Sheet) are not eligible under this RGP.

Based upon the 2004 Cape Cod Watershed Assessment, one of the greatest threats to water quality on the Cape was (and continues to be) excessive nutrients, particularly nitrogen (MassDEP, 2011). Some of the water recharging the Cape Cod Aquifer is wastewater discharge from on-site septic systems, municipal wastewater treatment plants, irrigation, or road runoff (MassDEP, 2011). The assessment concluded that increased population, intense development pressures, and sprawling land use patterns on Cape Cod resulted in increased non-point source pollution and loss of open space, habitat, and biodiversity. Pathogens, particularly fecal coliform and Enterococcus, are other common pollutants that can impair various water bodies in the Cape (MassDEP, 2013).

The 2004 – 2008 Surface Water Quality Assessment Report for Cape Cod Coastal Drainage Areas provided an assessment of five river segments (15.4 miles), 63 lake segments (5649 acres), and 89 estuarine/embayment segments (42.363 mi^2) (MassDEP, 2011). Water quality assessments for over 100 water bodies were also conducted for some of the drainage areas in the Cape Cod Watershed and incorporated into Massachusetts Year 2012 Integrated List of Waters (MassDEP, 2013).

Multiple studies and efforts have taken place to counteract the impairment issues in the Cape. The Massachusetts Estuaries Program (MEP), which represents a partnership between entities such as the UMASS-Dartmouth School of Marine Science and Technology (SMAST) and MassDEP, has resulted in the development of 66 nitrogen TMDLs for waters in the Cape Cod and Buzzards Bay drainage systems. According to MA's 2012 Integrated List of Waters report, the MEP will continue their efforts to develop nitrogen criteria and TMDLs for coastal waters. The project plans estimate that TMDLs for an additional 12 embayments will be developed each year (MassDEP, 2013). Also, a Pathogen Total Maximum Daily Load for the Cape Cod Watershed was approved in August 2009, and an addendum was approved in August 2012 (MassDEP, I, & International, Final Pathogen TMDL for the Cape Cod Watershed, 2009); (MassDEP, 2013).

d. Taunton River

The Taunton River Watershed, which encompasses 562 square miles, is the second largest watershed in the state of Massachusetts (Executive Office of Energy and Environmental Affairs, b). The Taunton River starts in the Town of Bridgewater and travels approximately 40 miles before ending in Rhode Island's Mount Hope Bay, which is part of Narragansett Bay. Since tidal influences reach 19.0 miles inland, this provides a unique habitat within the Taunton River Watershed for fresh and salt-water aquatic, terrestrial, and biological species (Executive Office of Energy and Environmental Affairs, b). Only sites in Massachusetts or New Hampshire (but not Rhode Island) are eligible for the RGP. Therefore, only the portion of the Taunton River included in the action area (i.e., the Massachusetts portion of the River) will be included in the assessment.

The uppermost segment of the mainstem Taunton River (MA62-01) is classified as a Class B, Warm Water Fishery while the lower three downstream portions (MA62-02, MA 62-03, and MA 62-04) are classified as Class SB (Estuary) with SFR/CSO as a qualifier.

Of the four segments of the mainstem Taunton River that were assessed as part of MassDEP's 2001 Water Quality Assessment of the Taunton River Watershed, all three of the lower downstream portions were listed as impaired for pollutants such as pathogens and organic enrichment/low dissolved oxygen and identified as being impacted by the discharge of CSOs (Rojko, Tamul, & Kennedy, 2005). The 20.4 miles of the uppermost portion of the Taunton River, down to the Route 24 bridge in Taunton/Raynham, was assessed as supporting aquatic life; other uses were not assessed. Massachusetts' Year 2012 Integrated List of Waters continued to list the two lower most segments (MA 62-03 and MA 62-04) of the Taunton River as impaired Category 5 waters, or "Waters Requiring a TMDL" (MassDEP, 2013). They were listed as not supporting fish or other aquatic life because of low dissolved oxygen from wet weather discharges (which includes point source and a combination of stormwater, SSO, or CSO). They also did not support shellfish harvesting because of fecal coliform. Since a Final Pathogen TMDL for the Taunton River Watershed was approved on June 16, 2011, Segment MA62-02 of the Taunton River mainstem was no longer classified as Category 5 (MassDEP, I, & International, Final Pathogen TMDL for the Taunton River Watershed, 2011); (MassDEP, 2013).

e. Massachusetts Bay

Massachusetts Bay is described as the offshore water that occupies a wide, triangular indentation of the eastern coast of Massachusetts, extending from Cape Ann to Plymouth Harbor, a distance of 42 miles. The depth inland from the middle of this ocean base line to Boston is about 22 miles. The northern shore of Massachusetts Bay is generally characterized as rocky, while the southern areas are typically comprised of marshy and sandy areas. The shoreline area throughout the bay is irregular and indented by numerous large and small bays, forming the harbors of Gloucester, Salem, Marblehead, Lynn, and Boston. The bay contains a number of islands along the shores, especially in the entrance to Boston Harbor.

The bay's most prominent submerged feature is the kidney-shaped plateau called Stellwagen Bank, which lies at the bay's eastern edge. Stellwagen Bank is a shallow, primarily sandy feature, curving in a southeast to northwest direction for 19 miles. There are also relatively deep areas of the bay, including Stellwagen Basin.

In general, Massachusetts Bay is heavily influences by regional oceanographic processes in the larger Gulf of Maine. During the winter months, waters of the bay are well mixed and reflect salinity and other characteristics of the Gulf of Maine. From April to October, however, there is sufficient stratification to partially isolate the deeper waters of Massachusetts Bay. The mean current, driven principally by the near shore coastal current in the western Gulf of Maine, moves in a counterclockwise direction around Massachusetts Bay.

f. Piscataqua River/Great Bay

Formed by the confluence of the Salmon Falls and Cocheco rivers, the Piscataqua River originates at the boundary of Dover, New Hampshire, and Eliot, Maine, and flows southeasterly for approximately 13 miles to Portsmouth Harbor (and the Atlantic Ocean) (USACE, 2014). The drainage basin of the river is approximately 1,495 square miles (3,870 km^2), and it encompasses the additional watersheds of the Great Works River and five rivers, namely the Bellamy, Oyster, Lamprey, Squamscott, and Winnicut, whose freshwaters all flow into the Great Bay. Since the Piscataqua River is a tidal estuary, it also brings salt water into the Great Bay with the tides (NH DES, 2014).

New Hampshire's Great Bay is one of the largest estuaries on the Atlantic Coast and it's also unique because the estuary is set apart from the coastline, approximately 10 miles inland. Although Great Bay has been designated by the U.S. EPA as one of only 28 "estuaries of national significance," there is concern about this ecosystem's health (NH DES, 2014). According to the 2013 State of Our Estuaries Report, which is compiled by the Piscataqua Region Estuaries Partnership every three years, 15 of the 22 key indicators used to assess the health of the estuaries were negative and/or had cautionary results (Piscataqua Region Estuaries Partnership, 2014). For example, concentrations of dissolved inorganic nitrogen (the most reactive form of nitrogen) have significantly increased over the long term, suspended sediment conditions have increased over the long term, and dissolved oxygen levels are frequently too low in the tidal rivers (Piscataqua Region Estuaries Partnership, 2014).

According to NH's final 2012 303d list, which highlights impaired waters that require a TMDL, various portions of both the Piscataqua River and Great Bay were listed. This included two stretches in the Upper Piscataqua River (in Dover), two stretches in the Lower Piscataqua River (one in Newington and one Portsmouth), and three areas in Great Bay (two in Newmarket and one in Newington). For these areas, the aquatic life use was impaired for estuarine bioassessments, light attenuation, total nitrogen, and pH (for the Great Bay stretches). The fish consumption use was impaired due to mercury and polychlorinated biphenyls while the shellfishing use was impaired for dioxin, mercury, and/or polychlorinated biphenyls (NHDES, 2014).

g. Coastal Embayments and Nearshore Marine Waters

Coastal embayments and nearshore marine waters are associated with over 160 miles of coastline in Massachusetts and New Hampshire. They include the southern Massachusetts coastline, the south and east coast of Cape Cod, the coastline north of Cape Anne, and the coastline of New Hampshire from the Massachusetts border to the entrance of Great Bay. These coastal areas are in addition to the coastal embayments and nearshore marine waters described as part of the rivers and major bays discussed above. These habitats are relatively shallow and associated with coastline features that variety from rocky shorelines to marshy and sandy areas. The shoreline area of Massachusetts is irregular and indented by numerous small embayments. Aside from the Great Bay area of New Hampshire (discussed above), the coastline of that state is relatively uniform.

Because, by definition, this habitat is near the shoreline, the water quality can vary and is influenced by runoff from the land. The type and volume of runoff is related to the geology of the near shore area as well as the anthropogenic activities that take place in the coastal watersheds of Massachusetts and New Hampshire. Oceanographic effects due to currents and wind patterns may not influence the habitat of these areas as much as the impact from localized coastal land characteristics and land use activities within the respective watershed.

3. NMFS Listed Species and Critical Habitat in the Action Area

The following are federally protected ESA species under the jurisdiction of NMFS in Massachusetts and New Hampshire:

Massachusetts (2)	**New Hampshire (2)**
Atlantic sturgeon (*Acipenser oxyrinchus*)	Atlantic sturgeon (*Acipenser oxyrinchus*)
Shortnose sturgeon (*Acipenser brevirostrum*)	Shortnose sturgeon (*Acipenser brevirostrum*)

This correspondence will not discuss the effects of the action on any threatened or endangered species under the jurisdiction of the USFWS and is only intended for use during informal consultation under Section 7 of the ESA with the National Marine Fisheries Service (NMFS). According to information obtained from the NMFS website, as well as information provided via September 3, 2013 and October 26, 2016, electronic correspondence between NMFS and EPA regarding this and/or other General Permits, ESA listed species potentially present within the Action Area include two species of listed fish: 1) shortnose sturgeon (*Acipenser brevirostrum*); and 2) Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus).* NOAA's Fisheries Service announced a final decision to list five Distinct Population Segments (DPSs) of Atlantic sturgeon in 2012. Only three DPSs fall under the jurisdiction of the Northeast Region of NOAA Fisheries; these are the Gulf of Maine DPS (threatened) and the New York Bight and Chesapeake Bay DPSs which are both listed as endangered ((77 FR 5880, 2012). However, since the range of all five DPSs overlaps and extends from Canada through Cape Canaveral, FL, the other two DPS of Atlantic sturgeon, namely the endangered Carolina and South Atlantic DPSs, have also been included in this document (77 FR 5914, 2012).

In addition, the following are federally protected marine species that are present in the near coastal waters of Massachusetts and New Hampshire. These species are listed under the jurisdiction of NMFS:

Marine Reptiles (5)

Loggerhead Sea Turtle *(Caretta caretta)*
Kemp's Ridley Sea Turtle *(Lepidochelys kempii)*
Leatherback Sea Turtle *(Dermochelys coriacea)*
Green Sea Turtle *(Chelonia mydas)*
Hawksbill Sea Turtle *(Eretmochelys imbricata)***

Marine Mammals (3)

North Atlantic Right Whale *(Eubalaena glacialis)*
Fin Whale *(Balaenoptera physalus)*

** Species rare in near shore Massachusetts and New Hampshire coastal waters

Two species of federally endangered whales are found seasonally in New England waters, including those off the coast of Massachusetts. These include the North Atlantic right whale (*Eubalaena glacialis*), and the fin whale (*Balaenoptera physalus).* The Cape Cod Bay Critical Habitat Area for North Atlantic Right Whales (*Eubalaena glacialis*) falls within a portion of the Action Area. The aforementioned critical habitat is part of the broader Northeast Atlantic critical habitat, which was designated in 1994. Following review by NMFS (78 FR 53391, 2013), the North Pacific population of humpback whale (*Megaptera novaeangliae*), which previously fell within a portion of the Action Area, has been delisted. The final rule was published on September 8, 2016 and became effective October 11, 2016 (81 FR 62018, 2016). RGP outfalls (in general) do not extend any measurable distance from the shoreline. Based upon this information and the listed whales' expected distributions, contact between these three endangered whales and the projected transient RGP discharge plume is extremely unlikely to occur. A discussion of the status of these protected whales and potential impacts to these species from the federal action is included in this correspondence to support a conservative approach to the informal consultation.

Four species of ESA listed sea turtles are found seasonally in New England waters, including those off the coast of Massachusetts. These include the endangered Kemp's ridley sea turtle *(Lepidochelys kempii)*, the threatened Northwest Atlantic Distinct Population Segment (DPS) of the Loggerhead sea turtle (*Caretta caretta*), the endangered Leatherback sea turtle (*Dermochelys* coriacea), and the Green Turtle *(Chelonia mydas)*. Based upon this information and the sea turtles' expected distribution, contact between these turtle species and the projected transient RGP discharge plumes is extremely unlikely to occur. A discussion of the status of these protected sea turtles and potential impacts to these species from the federal action is included in this correspondence to support a conservative approach to the informal consultation.

ESA-listed species and critical habitat that are present in the action area are described below. For each species, EPA has summarized available information regarding: 1) Life stages present and listed species' activities (e.g., foraging, migrating, spawning, overwintering); 2) Status of listed species; 3) Listed species' population and distribution including critical habitat used by the listed species; and 4) Population risks and stressors.

a. Shortnose Sturgeon (*Acipenser brevirostrum*) – Endangered

i. Life Stages and Activities

Shortnose sturgeons are large benthic fish that mainly occupy the deep channel sections of large coastal rivers in eastern North America (Shortnose Sturgeon Status Review Team, 2010). Throughout their lifecycle, they feed on a variety of benthic insects, crustaceans, mollusks, and polychaetes (Dadswell, Taubert, Squiers, Marchette, & Buckley, 1984).

Like other sturgeon, the shortnose sturgeon is relatively slow going, late maturing and long-lived (Shortnose Sturgeon Status Review Team, 2010). Shortnose sturgeon have similar lengths at maturity (45-55 cm fork length) throughout their range, but, because sturgeon in southern rivers grow faster than those in northern rivers, southern sturgeon mature at younger ages (Dadswell, Taubert, Squiers, Marchette, & Buckley, 1984) . In the north, males reach maturity at 5 to 10 years, while females mature between 7 and 13 years (Shortnose Sturgeon Status Review Team, 2010).

> Spawning is not typically a yearly event for shortnose sturgeon in northern rivers. Based on limited data, females spawn every three to five years while males spawn approximately every two years (Dadswell, Taubert, Squiers, Marchette, & Buckley, 1984). The spawning period is estimated to last from a few days to several weeks. According to the 2010 Biological Assessment, shortnose sturgeon in northern rivers are known to migrate from overwintering locations upstream to spawning grounds during the spring when the freshwater temperatures increase to 7-9°C (Shortnose Sturgeon Status Review Team, 2010). Sturgeon spawn in upper, freshwater areas and feed and overwinter in both fresh and saline habitats. As noted in the 2010 Biological Assessment, shortnose sturgeon is often considered “anadromous,” however a more accurate term is “amphidromous.” This means that the fish move between fresh and salt water during some part of their lifecycle, but not for breeding purposes (Shortnose Sturgeon Status Review Team, 2010).

ii. Status

Shortnose sturgeon were originally listed as an endangered species by the USFWS on March 11, 1967 under the Endangered Species Preservation Act (32 FR 4001, 1967). After a government reorganization plan was implemented in the early 1970’s, NMFS assumed jurisdiction for shortnose sturgeon from the USFWS. Although the original listing notice did not document specific reasons for listing the shortnose sturgeon as endangered, a 1973 Resource Publication, issued by the US Department of the Interior, indicated that shortnose sturgeon were in peril in most of the rivers of its former range but probably not as yet extinct (United States Department of Interior, 1973) The U.S. Fish and Wildlife Service also identified pollution and overharvest in commercial fisheries as principal reasons for the species decline (United States Department of Interior, 1973) . Shortnose sturgeon remains listed as an endangered species throughout all of its range along the U.S. East Coast. NOAA Fisheries is currently conducting a status review for shortnose sturgeon to ensure that the original classification as an endangered species is still appropriate.

iii. Population and Distribution

The Shortnose Sturgeon Recovery Plan, which was finalized in 1998, identified 19 distinct populations based on the fish's strong ties to their natal river systems (Shortnose Sturgeon Status Review Team, 2010). These river systems range from the Saint John River in New Brunswick, Canada to the St. Johns River in Florida. Two populations of shortnose Sturgeon have been documented in Massachusetts waters, specifically in the following areas:

1) Merrimack River (main stem) below the Essex Dam in Lawrence, MA to the Merrimack River's mouth (Essex County);
2) Connecticut River (main stem) downstream of Turner's Falls, MA (Franklin, Hampshire, and Hampden Counties) to the Connecticut River's mouth in the state of CT (Hartford Middlesex and New London Counties);
3) Piscataqua River in New Hampshire (historically);
4) Coastal embayments and nearshore marine waters including Cape Cod Bay and Massachusetts Bay in Massachusetts and Great Bay in New Hampshire (transiently).

The state of Massachusetts encompasses 27 watersheds (MassDEP, 2013). The Action Area for the permit, as it relates to shortnose sturgeon, consists of two watersheds within Massachusetts where the species has been well documented. This includes portions of the Merrimack River Watershed and the Connecticut River Watershed. A population of endangered shortnose sturgeon is known to seasonally inhabit the Merrimack River below the Essex (also known as the Lawrence or Merrimack) Dam in Lawrence. The lower Connecticut River (including waters in Massachusetts downstream of Turner Falls) is inhabited by the endangered shortnose sturgeon (*Acipenser brevirostrum*). In addition to the mainstems of the Merrimack and Connecticut River, at least eight additional Massachusetts watersheds influence coastal embayments and nearshore marine waters that may be used by adult shortnose sturgeon.

<u>Shortnose Sturgeon in the Merrimack River</u>

According to a letter dated November 4, 2013 in which NMFS responded to EPA's request for ESA section 7 consultation regarding NPDES discharges from Lawrence Hydroelectric Project (NMFS, 2013f) ,

> There is a small population of the federally endangered shortnose sturgeon (*Acipenser brevirostrum*) in the Merrimack River. The size of this population has been estimated by tag and release studies (conducted in 1988-1990) to be 33 adults with an unknown number of juveniles and subadults…. Shortnose sturgeon in the Merrimack River are not known to exist upstream of the Essex Dam (Lawrence), which represents the first significant impediment to the upstream migration of shortnose sturgeon in this system. Sexually mature fish begin to move upriver from freshwater overwintering areas (located in the Amesbury reach) to the spawning site near Haverhill…Spawning is concentrated within a 2-km reach at river kilometers 30-32 (measured from the mouth) near Haverhill…Following spawning in late April-early May, fish move downriver. Some fish remain in a freshwater reach near Amesbury (Rocks Village to Artichoke River) for the remainder of the year while others move into a saline

reach near the lower islands for about 6 weeks prior to returning to the freshwater reach.

Since those earlier tag and release studies, more recent sampling efforts have occurred. NMFS' 2010 Shortnose Sturgeon Biological Assessment indicated that a gill net-sampling took place in the winter of 2009 in which researchers captured a total of 170 adults (Shortnose Sturgeon Status Review Team, 2010). According to NMFS, spawning near Haverhill (RKM 30-32). Eggs and larvae present in spawning grounds begin to move downstream approximately four weeks after spawning (RKM 16-32). Foraging is concentrated in the lower Merrimack near Amesbury and the lower islands (RKM 7-12). Multiple overwintering sites are located beyond the maximum salt penetration (RKM 15-29).[1]

The projected dimensions of the discharge plume of any RGP outfall in the Merrimack River below the Essex Dam are generally expected to be confined to the immediate riverbank and only extend out a minimal distance into the mainstem of the river and a minimal distance downstream of the discharge before complete mixing takes place. The expected distribution of shortnose sturgeon in the river has the potential to include the immediate riverbank of the shallow mainstem waters. Therefore, contact between all life stages of shortnose sturgeon in the Merrimack River and the projected transient RGP discharge plumes may occur.

Shortnose Sturgeon in the Connecticut River

Shortnose sturgeons inhabit the Connecticut River from the Turners Falls Dam, at RKM 198 in Turners Falls, MA, down to Long Island Sound. The Connecticut River population is separated by the Holyoke Dam, at the South Hadley Falls near RKM 140, into an upriver group (above Holyoke Dam) and a lower river group (below Holyoke Dam). Although earlier reports indicated that the shortnose sturgeon were separated with the construction of the Holyoke Dam, the 2010 Shortnose Sturgeon Biological Assessment reported that more recent "behavioral and genetic information indicates shortnose sturgeon in the Connecticut River are of a single population impeded, but not isolated, by the dam" (Shortnose Sturgeon Status Review Team, 2010).

According to NMFS, several areas of the Connecticut River have been identified as concentration areas for the shortnose sturgeon. Spawning occurs at two locations below the Turners Falls Dam/Cabot Station, depending on River conditions (RKM 193-194). A 2-km spawning site identified near Montague, MA and this is thought to be the primary spawning site for shortnose sturgeon in the Connecticut River (Kynard, Bronzi, & Rosenthal, 2012). Eggs and larvae have been documented at least 3 to 15 kilometers downstream of the spawning sites. Limited spawning may occasionally occur below the Holyoke Dam. If spawning is successful, early life stages would also be present in downstream freshwater reaches. Foraging concentrations occur above the Holyoke Dam in the Deerfield Confluence Area (DCM) RKM 144-192), and throughout the river below the Holyoke Dam (RKM 0-140), with concentrations near Holyoke (RKM 137-139), Agawam (RKM 112-120), and the lower river (RKM 0-100). Overwintering concentrations occur above the Holyoke Dam in the DCA (RKM 144-192), and below the Holyoke Dam, with concentrations near Holyoke (RKM 140), Agawam (RKM 117),

[1] *GARFO Master ESA Species Table – Shortnose Sturgeon*. National Marine Fisheries Service. Dated 4-28-2016.

Hartford (RKM 82-86), Portland (RKM ≈50), and the lower river (RKM 0-25). Adults and/or larvae have also been documented in tributaries to the Connecticut River, adults and larvae in the Deerfield River, and adults in the Westfield River.[2]

Population estimates have been completed for shortnose sturgeon in the Connecticut River, occurring both above and below the Holyoke Dam. According to the 2010 Biological Assessment, Taubert (1980) conducted the earliest population estimate for the sturgeon upstream of the dam which resulted in an estimate of 370-714 adults. More recent studies, including a 1994 mark-recapture estimate during the summer-fall foraging period of 1994 and an annual spring study of pre-spawning adults near Montague between 1994-2001 yielded estimates of 328 adults (CI of 188-1,264 adults) and a mean of 142.5 spawning adults (CI of 14-360 adults), respectively (Shortnose Sturgeon Status Review Team, 2010) . Downstream of the Holyoke Dam, researchers conducted annual estimates of foraging and wintering adults during 1989-2002. Savoy (2004) estimated that the lower river population may be as high as 1000 individuals, based on his studies that used mark-recapture techniques.

The projected dimensions of the discharge plume of any RGP outfall in the Massachusetts portion of the river downstream of Turners Falls are generally expected to be confined to the immediate riverbank and only extend out a minimal distance into the mainstem of the river and a minimal distance downstream of the discharge before complete mixing takes place. The expected distribution of shortnose sturgeon in the river has the potential to include the immediate riverbank of the shallow mainstem waters. Therefore, contact between all life stages of shortnose sturgeon in the Connecticut River and the projected transient RGP discharge plumes may occur.

Shortnose Sturgeon in the Piscataqua River and Coastal Embayments and Nearshore Marine Waters of New Hampshire

It is believed that shortnose sturgeon were historically abundant in the Piscataqua River, though there are few records of sturgeon captures (Shortnose Sturgeon Status Review Team, 2010). With few records and no current directed studies underway in this river, it is unclear whether a shortnose sturgeon population currently exists in the Piscataqua River. However, several larger river systems in the vicinity of the Piscataqua River (e.g., Merrimack, Kennebec and Androscoggin Rivers) support shortnose sturgeon populations. According to NMFS, the Piscataqua River is used seasonally by adult shortnose sturgeon for foraging and resting during spring and fall migrations, limited to days or weeks.[3]

According to information taken directly from previous communication between NMFS and EPA:

> It is clear from recent telemetry data that shortnose sturgeon tagged in the Merrimack, Kennebec, and Penobscot rivers undertake significant coastal migrations…. Telemetry data also indicates that shortnose sturgeon utilize smaller coastal river systems during these migrations. Fish moving between the Penobscot and Kennebec rivers have been documented utilizing a number

[2] See footnote 1, above.
[3] See footnote 1, above.

> of small coastal rivers in between these two larger systems (e.g., Damariscotta as well as the St. George, Medomak, and Passagasawakeag). As such, not only are inter-basin transfers between the Merrimack and GOM evident, but there also is the potential for shortnose sturgeon undertaking these migrations to utilize smaller riverine systems along the way. Therefore, NMFS will consider that shortnose sturgeon could occur in any coastal river, below the first impassable barrier as well as in nearshore coastal waters throughout the state.[4]

The projected dimensions of the discharge plume of any RGP outfall near the mouth of the Piscataqua River are generally expected to be confined to the immediate riverbank or shoreline and only extend out a minimal distance into the mainstem of the river or nearshore marine waters and a minimal distance downstream of the discharge before complete mixing takes place. The expected distribution of shortnose sturgeon in the river has the potential to include the immediate riverbank of the shallow mainstem waters, and, less frequently, in nearshore marine waters. Therefore, contact between juvenile and adult shortnose sturgeon in the Piscataqua River and adult foraging shortnose sturgeon in nearshore marine waters and the projected transient RGP discharge plumes may occur.

<u>Shortnose Sturgeon in Coastal Embayments and Nearshore Marine Waters of Massachusetts</u>

The projected dimensions of the discharge plume of any RGP outfall in coastal marine waters is generally expected to be confined to the immediate estuarine areas or shoreline and only extend out a minimal distance into the nearshore marine waters and a minimal distance downstream of the discharge before complete mixing takes place. The expected distribution of shortnose sturgeon in marine waters in Massachusetts has the potential to include the immediate shallow marine waters, albeit less frequently. The adult life stage of shortnose sturgeon is expected to occur in these coastal areas. Therefore, contact between foraging adult shortnose sturgeon in nearshore marine waters and the projected transient RGP discharge plumes may occur.

iv. Population Risks and Stressors

According to a Shortnose Sturgeon Recovery plan that was published in December 1998 to promote the conservation and recovery of the species, principal threats to the species' survival included habitat degradation or loss (resulting from dams, bridge construction, channel dredging, and pollutant discharges) and mortality (from impingement on cooling water intake screens, dredging, and bycatch from other fisheries) (NMFS, 1998) . Several natural and human-induced factors, including dams and diversions, dredging, blasting and pile driving, water quality and contaminants, climate change, and bycatch, threaten the recovery of shortnose sturgeon. The following stressor described in the 2010 Shortnose Sturgeon Biological Assessment is relevant to the proposed action:

[4] NMFS's Appendix I (NMFS-listed Species in New Hampshire) to a March 22, 2013 letter from NMFS to EPA regarding NH's Small MS4 NPDES Permit and Technical Comments on the Draft Permit

1) **Water Quality and Contaminants:** Non-point source pollution and/or point-source discharges from municipal wastewater, industrial activities, power plant cooling water or wastewater, and agricultural practices can discharge pollutants (including nutrients, chemicals and/or metals) and lead to poor water quality (NMFS, 1998); coastal and riparian areas can be particularly impacted by development and urbanization which can lead to erosion, stormwater discharges, and non-point source pollution (Shortnose Sturgeon Status Review Team, 2010); compounds associated with point-source discharges, which can include metals, dioxin, dissolved solids, phenols, and hydrocarbons, lead to changes in fish behavior, deformations, reduced egg production and survival, or mortality (Health, 1987); such chemicals can also alter the physical properties of the receiving waterbody by reducing dissolved oxygen (DO) or changing the water's temperature and/or pH (Shortnose Sturgeon Status Review Team, 2010);

According to the most recent Biological Assessment for the shortnose sturgeon, the viability of sturgeon populations was most negatively influenced by dams, dredging, poor water quality, and bycatch (Shortnose Sturgeon Status Review Team, 2010). As a whole, the greatest single threat to shortnose sturgeon was habitat degradation (Shortnose Sturgeon Status Review Team, 2010). No reliable estimate exists for the shortnose sturgeon population in the Northeastern U.S, nor is there an estimate for the total species population as a whole (NMFS, 2013e). However, the population size is obviously lower than what could be supported because of the aforementioned threats (NMFS, 2013e).

b. Atlantic Sturgeon (Acipenser oxyrinchus oxyrinchus):

1) Gulf of Maine DPS: Threatened
2) New York Bight DPS: Endangered
3) Chesapeake Bay DPS: Endangered
4) Carolina DPS: Endangered
5) South Atlantic DPS: Endangered

i. Life Stages and Activities

Atlantic sturgeon is a long-lived, late maturing, estuarine-dependent, anadromous species, feeding primarily on benthic invertebrates such as crustaceans, worms, and mollusks. Although adults spend most of their lives in marine environments, they migrate upriver to spawn in freshwater in the spring and early summer (Atlantic Sturgeon Status Review Team, 2007). According to NMFS's website, Atlantic sturgeon spawn in moderately flowing water in deep parts of large rivers. The spawning interval for males ranges from 1 to 5 years and 2 to 5 years for females. Sturgeon eggs are highly adhesive and are deposited on hard benthic substrate, such as cobble. Once eggs hatch, the larvae eventually migrate downstream using structures, like gravel matrices, as refuges. Juvenile Atlantic sturgeon continue to move further downstream into brackish waters. Adults live in coastal waters and estuaries, particularly in shallow areas with sand and gravel substrates (NMFS, 19 Nov 2013).

ii. Status

All five DPSs of Atlantic sturgeon, including the GOM, New York Bight, and Chesapeake Bay DPSs in the Northeast Region of the United States and the South Atlantic and Carolina DPSs in the Southeast Region, received a final listing under the ESA on February 6, 2012 (77 FR 5880, 2012); (77 FR 5914, 2012). The GOM distinct population segment is listed as threatened while the other four DPSs are listed as endangered. Although an earlier petition to list the Atlantic sturgeon was submitted in 1997, the status review determined that the species did not meet the requirements under the ESA at that time. However, in 1998, the Atlantic States Marine Fisheries Commission (ASMFC) did amend the 1990 Atlantic Sturgeon Fishery Management Plan to impose a 20 to 40-year moratorium on Atlantic sturgeon fisheries (Atlantic Sturgeon Status Review Team, 2007). NMFS completed a second status review in 2007 and the Natural Resources Defense Council (NRDC) petitioned NMFS to list the Atlantic sturgeon under ESA in 2009. This led to the current listing (NMFS, 19 Nov 2013).

On June 3, 2016, NMFS issued two proposed rules to designate critical habitat for the five listed distinct population segments (DPSs) of Atlantic sturgeon found in U.S. waters (Gulf of Maine, New York Bight, and Chesapeake Bay DPSs: 81 FR 35701; Carolina and South Atlantic DPSs: 81 FR 36078).

iii. Population and Distribution

Summary of Distribution & Population Trends

Distinct Population Segment (DPS)	**Range** (According to 77 FR 5580 & 77 FR 5914; Includes watersheds (rivers and tributaries) "as well as wherever these fish occur in coastal bays and estuaries and the marine environment")	**Current Spawning Location(s)** – (NMFS, 2013b)
Gulf of Maine DPS	Those spawned in watersheds from Maine/Canadian border – extending southward to all watersheds draining into Gulf of Maine as far south as Chatham, MA	Kennebec River; possibly Penobscot River
New York Bight DPS	Those spawned in the watersheds that drain into coastal waters, including Long Island Sound, the New York Bight, and Delaware Bay, from Chatham, MA to the Delaware-Maryland border of Fenwick Island.	Hudson River & Delaware River
Chesapeake Bay DPS	Spawned in watersheds that drain into the Chesapeake Bay and into coastal waters from the Delaware-Maryland border on Fenwick Island to Cape Henry, VA	James River; possibly York River (NMFS, n.d.)(NMFS CB Fact Sheet)

Carolina DPS	Spawned in watersheds from Albemarle Sound southward along the southern Virginia, North Carolina, and South Carolina coastal areas to Charleston Harbor	Roanoke, Tar-Pamlico, Cape Fear, Waccamaw, and Pee Dee Rivers; Possibly in Neuse, Santee and Cooper Rivers
South Atlantic DPS	Spawned in watersheds of the ACE (Ashepoo, Combahee, and Edisto) Basin southward along the South Carolina, Georgia, and Florida coastal areas to the St. Johns River, Florida	ACE (Ashepoo, Combahee and Edisto Rivers) Basin, Savannah River, Ogeechee River, Altamaha River, and Satilla River

On June 3, 2016, NMFS issued two proposed rules to designate critical habitat for the five listed distinct population segments (DPSs) of Atlantic sturgeon found in U.S. waters (Gulf of Maine, New York Bight, and Chesapeake Bay DPSs: 81 FR 35701; Carolina and South Atlantic DPSs: 81 FR 36078).

Atlantic sturgeon were historically present in approximately 38 rivers in the United States ranging from St. Croix, ME to Saint Johns River, FL; a historical spawning population was confirmed for 35 of those rivers. Currently, Atlantic sturgeon are present in 35 rivers, and spawning occurs in at least 20 of these rivers (Atlantic Sturgeon Status Review Team, 2007). The species has been documented in several New England rivers, including the Penobscot, Kennebec, Androscoggin, and Sheepscot Rivers in Maine; **the Piscataqua River in New Hampshire; the Merrimack River in NH and MA; the Taunton River in MA & RI; and the Connecticut River in MA and CT (ASSRT 2007).** Of these, a spawning population has only been identified in the Kennebec River, although there is possible spawning in the Penobscot. Atlantic sturgeon from all of those rivers, with the exception of the Taunton River and Connecticut River, fall under the Gulf of Maine (GOM) DPS. Sturgeon from the Taunton and Connecticut River would fall under the New York Bight (NYB) DPS.
As previously mentioned, the Action Area for this General Permit includes Massachusetts and New Hampshire waters. The Action Area, as it relates to Atlantic sturgeon, can been further narrowed to the waterways where the sturgeon exist: the Connecticut, Merrimack, Taunton and Piscataqua Rivers and the coastal embayments and nearshore marine waters of Massachusetts and New Hampshire. Atlantic sturgeon may be present in these waterbodies as follows:

1) **Merrimack River**: Atlantic sturgeon have been documented in the Merrimack River (ASSRT). According to NMFS, spawning potentially occurs due to the presence of features necessary to support reproduction and recruitment, and the estuary appears to be used as a nursery area (Atlantic Sturgeon Status Review Team, 2007). Rearing of early life stages and Young of Year occurs in nursery areas. Foraging occurs at the mouth of the river and the lower islands ((RKM 0-12). Some known overwintering sites occur at

RKM 14, 19, and 26.[5] Therefore, contact between subadult and adult (and potentially all life stages of) of Atlantic sturgeon in the Merrimack River and the projected transient RGP discharge plumes may occur.

2) **Connecticut River**: Research efforts have not specifically investigated the occurrence of Atlantic sturgeon in the upper Connecticut River, which would include the MA-portion of the river (Atlantic Sturgeon Status Review Team, 2007). According to Savoy (1996), there have been occasional reports, sightings and capture of Atlantic sturgeon in the Connecticut River, as far upstream as the area near the Holyoke Dam (150-300 cm), but most are captured within tidal waters or freshwater in the lower part of the Connecticut (Savoy, 1996). According to NMFS, captures strongly suggest that spawning is occurring. Rearing occurs spring through fall in the lower 26 RKM. Adult and subadult foraging occurs spring through fall in waters less than 50 meters in depth.[6] Therefore, contact between subadults and adult (and potentially all life stages of) Atlantic sturgeon in the Massachusetts reaches of the Connecticut River and the projected transient RGP discharge plumes may occur.

3) **Taunton River** – According to the ASSRT, Atlantic sturgeon did spawn in the Taunton River at the turn of the century (1900's); A gill net survey was conducted in the River during 1991 and 1992 to document the use of the system by sturgeon. Burkett and Kynard (1993) determined that the system is used as a nursery area for Atlantic sturgeon (Burkett & Kynard, 1993). According to NMFS, subadult and adult foraging is assumed to occur wherever suitable forage is present.[7] Therefore, contact between subadult and adult Atlantic sturgeon in the Taunton River and the projected transient RGP discharge plumes may occur.

4) **Piscataqua River**– According to the ASSRT, few Atlantic sturgeon have been captured in the Piscataqua River(Atlantic Sturgeon Status Review Team, 2007). Although the Atlantic Sturgeon Status Review Team and NHFG biologists concluded that the Great Bay Atlantic sturgeon population is likely extirpated, individuals from other populations may forage in the Piscataqua River. Also, according to NMFS, spawning potentially occurs in the Salmon Falls and Cocheco rivers based on the presence of features necessary to support reproduction and recruitment, as well as the historic capture of an adult female in spawning condition. Subadult and adult foraging is assumed to occur wherever suitable forage is present.[8] Therefore, contact between subadults and adult (and potentially all life stages of) Atlantic sturgeon in the Piscataqua River and the projected transient RGP discharge plumes may occur.

5) **Coastal Embayments and Nearshore Marine Waters**

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[5] *GARFO Master ESA Species Table – Atlantic Sturgeon*. National Marine Fisheries Service. Dated 4-28-2016.
[6] See footnote 5, above.
[7] See footnote 5, above.
[8] See footnote 5, above.

It is generally understood that subadult Atlantic sturgeon are known to travel widely and enter estuaries of non-natal rivers (77 FR 5880, 2012). Because this coastal migration is common, it is likely that subadult and adult Atlantic sturgeon are found in coastal embayments and nearshore marine water habitats of Massachusetts and New Hampshire. In June of 1981, one subadult Atlantic sturgeon was captured by New Hampshire Fish and Game (NHFG) at the mouth of the Oyster River in Great Bay (NH Fish and Game, 1981). Since 1990, the NHFG has not observed or received reports of Atlantic sturgeon of any age-class being captured in the Great Bay Estuary and its tributaries (Grout, 2006).

Subadults are known to travel widely and enter estuaries of non-natal rivers (77 FR 5880, 2012). Therefore, there is substantial mixing throughout the marine range of Atlantic sturgeon and coastal migration is common. Nonetheless according to 77 FR 5880, mixed stock analysis of Atlantic sturgeon collected along the U.S. coast indicates that Atlantic sturgeon occur most prominently in the vicinity of their natal river(s). Fish from the Gulf of Maine DPS are not commonly taken as bycatch in areas south of Chatham, MA. Additional tagging results also indicate that GOM DPS fish tend to remain within the waters of the Gulf of Maine and only occasionally venture to points south. Based on this information, EPA believes that Atlantic sturgeon from the Gulf of Maine (GOM) and the New York Bight (NYB) DPSs would most frequently fall within the Action Area of this permit. However, EPA cannot exclude the possibility that Atlantic sturgeon from any of the five DPSs may be present in the Action Area waters. This reasoning follows a similar conclusion reached by NMFS as stated in a March 22, 2013 letter from NMFS Assistant Regional Administrator Mary Colligan to EPA Water Permits Branch Chief Dave Webster regarding the New Hampshire MS4 NPDES permit (NMFS, 2013a).

Historically, each of the DPSs likely supported more than 10,000 spawning adults (Atlantic Sturgeon Status Review Team, 2007). However according to the most recent status review, the best available data support that current numbers of spawning adults for each DPS are one to two orders of magnitude smaller than historical levels (Atlantic Sturgeon Status Review Team, 2007); 77 FR 5880). As only two abundance estimates are presently available for Atlantic sturgeon riverine populations (Atlantic Sturgeon Status Review Team, 2007). The Hudson River population in New York, which is part of the NYB DPS, was estimated to have 870 spawning adult Atlantic sturgeon per year (Kahnle, Hattala, & McKown, 2007). The Altamaha River population in Georgia, which falls under the South Atlantic DPS, has 343 spawning adults per year (Schuller & Peterson, 2006). Other spawning populations within the U.S are likely to have less than 300 adults spawning per year (Atlantic Sturgeon Status Review Team, 2007).

According to 77 FR 5880, the Hudson is presumably the largest reproducing Atlantic sturgeon population. However, the final ruling indicated that all riverine populations of Atlantic sturgeon, including those in the Northeast Region, are at reduced levels from those reported historically, and are being exposed to significant threats that are ongoing and not being adequately addressed. The final ruling by NMFS stated that there are indications of increasing abundance of Atlantic sturgeon belonging to the GOM DPS, particularly in the following rivers in Maine: the Kennebec River, Penobscot River, and more recently the Saco and Presumpscot Rivers (77 FR 5880, 2012). This indicates that recolonization to rivers historically suitable for spawning may be occurring (78 FR 69310, 2013).

The projected dimensions of the discharge plume of any RGP outfall in coastal marine waters is generally expected to be confined to the immediate estuarine areas or shoreline and only extend out a minimal distance into the nearshore marine waters and a minimal distance downstream of the discharge before complete mixing takes place. The expected distribution of Atlantic sturgeon in marine waters in Massachusetts and New Hampshire has the potential to include the immediate shallow marine waters, albeit less frequently. The subadult and adult life stages of Atlantic sturgeon are expected to occur in these coastal areas. Therefore, contact between foraging subadult and adult Atlantic sturgeon in nearshore marine waters and the projected transient RGP discharge plumes may occur.

iv. Population Risks and Stressors

Historically, commercial fishing and overharvesting of Atlantic sturgeon was the primary factor that led to a wide-spread decline of their numbers. The Atlantic sturgeon is now managed under a Fishery Management Plan, which is implemented by the Atlantic States Marine Fisheries Commission (Atlantic States Marine Fisheries Commission, 1990). In 1998, the ASFMC also instituted a coast-wide 20 to 40-year moratorium on the harvest of Atlantic sturgeon. This will remain in effect until there are at least 20 protected age classes in each spawning stock of Atlantic sturgeon (Atlantic Sturgeon Status Review Team, 2007).

According to the final rulings for the Atlantic sturgeon, the threats that continue to adversely impact their abundance include bycatch in state and federally-managed fisheries, vessel strikes, persistent, degraded water quality, habitat impacts from dredging, habitat impediments including dams, and global climate change. The threat relevant to the proposed action includes:

1) **Persistent, degraded water quality**

Several of these threats for the Atlantic sturgeon coincide with those listed for the shortnose sturgeon. Therefore, the explanations previously provided are still applicable. However, the majority of these threats are not relevant to the proposed action. Further, since the Atlantic sturgeon is listed as five distinct population segments, the relevant threats are not necessarily present in the same area at the same time, nor are the effects identical. The section below highlights some of the difference in stressors or risks to each of the five DPSs as relevant to the proposed action.

<u>Gulf of Maine DPS</u>

All of the threats noted above apply to the GOM DPS. With respect to the proposed action, and according to status review, poor water quality has been identified as one of the key risks (Atlantic Sturgeon Status Review Team, 2007).

1) Many rivers in Maine, including the Androscoggin River, were heavily polluted in the past from industrial discharges from pulp and paper mills (NMFS, 2013b). However as stated in 77 FR 5880, water quality improvements have been made in the range of the GOM DPS since the passage of the CWA. According to the most recent (fourth) edition of the National Coastal Condition Report, the water quality index was listed as good to

fair for waters in the Arcadian province of the Northeast; these are the waters north of Cape Cod, MA (EPA, 2012).

New York Bight DPS

Persistent, degraded water quality also continues to pose risks to the NYB DPS (77 FR 5880, 2012).

1) Although the CWA has led to improvements in water quality, rivers in the NYB region, including the Hudson and Delaware rivers, were heavily polluted from past industrial discharges and sanitary sewer discharges (77 FR 5880, 2012).
   The most recent (fourth) edition of the National Coastal Condition Report identified that water quality was fair overall for waters in the Virginian province of the Northeast; this consists of waters south of Cape Cod through the Chesapeake Bay (EPA, 2012). These waters are quite vulnerable to the impacts of a highly populated and industrialized region. There are pockets of poor water, particularly in areas including Great Bay, NH; Narragansett Bay, RI; Long Island Sound; NY/NJ Harbor; the Delaware Estuary; and the western tributaries of Chesapeake Bay (EPA, 2012). Various issues exist including reports of low DO concentration in the summer and high ammonia-nitrogen levels in the Taunton River, impacts from coal tar leachate in the Connecticut River, and lasting PCB pollution in the Hudson River (77 FR 5880, 2012).

Chesapeake Bay DPS

Similar to the NYB DPS, degraded water quality continues to be a key threat to the Chesapeake Bay DPS of Atlantic sturgeon (77 FR 5880, 2012).

1) Decreased water quality is a significant threat because the Chesapeake Bay system is particularly vulnerable to the effects of nutrient enrichment and sedimentation from point and non-point sources. A Total Maximum Daily Load for Nitrogen, Phosphorus, and Sediments has been established, and a number of other efforts including NOAA's 2010 Chesapeake Bay Protection and Restoration Final Strategy have also been initiated (77 FR 5880, 2012). According to the final listing for the CB DPS, water quality concerns include especially low DO (as a result of the nutrient loadings) and a decrease in the availability of clean, hard substrate for Atlantic sturgeon spawning habitat (77 FR 5880, 2012).

c. North Atlantic Right Whales (*Eubalaena glacialis*), Western Stock – Endangered

Right whales are known to be the rarest of all large whale species, as well as the rarest of all marine mammal species. As such, North Atlantic right whales have a species' recovery priority number of One (1) based on the criteria in the Recovery Priority Guidelines (NOAA Fisheries, 2012). Three species of right whales exist: The North Atlantic right whale (*Eubalaena glacialis)*, the North Pacific right whale (*Eubalaena japonica)*, and the southern right whale (*Eubalaena*

australis) (NMFS, n.d.). The North Atlantic right whale is the only species applicable to this permit.

i. Life Stages and Activities

North Atlantic right whales are large baleen whales which feed on zooplankton, especially copepods. Unlike other baleen whales, right whales are skimmers. This means that they feed by continuously filtering prey through their baleen as they move through a patch of zooplankton with their mouth open (NMFS, 2005). In the western North Atlantic, calving occurs between December and March in the shallow, coastal waters of southeastern U.S. Females, in both the northern and southern hemisphere, give birth to their first calf at the average age of nine years; gestation lasts approximately 12 – 13 months (NMFS, 2005).

Feeding and nursery grounds, where nursing females feed and suckle, occur in New England waters and north to the Bay of Fundy and Scotian Shelf (NMFS, 2005). Right whales are most abundant in the coastal waters off Massachusetts, particularly Cape Cod Bay, between February and April where they have been observed feeding predominantly on dense patches of copepods (NMFS, n.d.); (NMFS, 2012).Much of the population is found in the Canadian waters in the summer through fall (NMFS, 2005)**.**

The location of some portion of the population during the winter months remains unknown, as does any breeding area(s) for the whales (NMFS, 2005). Also although there is little data on the longevity of these whales, it is believed that they live for at least 50 years (NMFS, n.d.).

ii. Status

In June of 1970, the "northern right whale" (*Eubalaena spp.*) was originally listed under the Endangered Species Conservation Act, the precursor to the ESA (35 FR 18319, 1970). Since the Endangered Species Act was established in 1973, it has remained listed. In 2008, after NMFS conducted a comprehensive review of the status of right whales in the North Atlantic and North Pacific Oceans, they concluded that the right whales in the northern hemisphere were actually two species: North Atlantic right whale *(Eubalaena glacialis)* and North Pacific right whale *(Eubalaena japonica*) (73 FR 12021, 2008). The species is also designate as depleted under the Marine Mammal Protection Act (MMPA).

NMFS approved a Final Recovery Plan for the Northern Right Whale, which included both the North Atlantic and North Pacific right whales) in December of 1991. This identified actual and potential factors that were impacting the northern right whale and provided recommendations to reduce and/or eliminate threats to the species' recovery. A revised recovery plan for the North Atlantic right whale (*Eubalaena glacialis*) was published in 2005 (NMFS, 2005).

Critical Habitat was originally designated for the Northern Right Whale in 1994 (59 FR 28805, 1994).

iii. Population and Distribution

Distribution

As previously mentioned, Western North Atlantic right whales generally range from their calving grounds in the coastal waters of southeastern United States to their feeding and nursery grounds in New England waters and the Canadian Bay of Fundy. According to the 2005 Recovery Plan, the distribution of whales seems to be tied to the distribution of their prey (NMFS, 2005). In addition to the coastal waters of the southeast, research indicates that there five other major habitats, or congregations, where Western North Atlantic right whales frequently exist. These include: the Great South Channel; Georges Bank/Gulf of Maine; Cape Cod and Massachusetts Bays; The Bay of Fundy; and the Scotian Shelf (NMFS, 2012).

Designated Critical Habitat

A wide range of human activities may impact the designated critical habitat including vessel activities, fisheries, and possible habitat degradation through pollution, sea bed mining, and oil and gas exploration (59 FR 28805, 1994).

Designated habitat for the Northern Right Whale includes two defined areas, namely Cape Cod/Massachusetts Bays and The Great South Channel (GSC) in the Northeast and waters adjacent to the coasts of George and the east coast of Florida in the Southeast US (SEUS) (59 FR 28805, 1994). The two designated areas in the Northeast serve as foraging habitats for the whales while the designated area in the Southeast is known as a winter calving ground and nursery.

The following excerpt from the final rule of Designated Habitat describes the Great South Channel (GSC):

> The GSC is a large funnel-shaped bathymetric feature at the southern extreme of the Gulf of Maine between Georges Bank and Cape Cod, MA. The GSC is one of the most used cetacean habitats off the northeastern United States (Kenney and Winn, 1986) …The channel is generally deeper to the north and shallower to the south, where it narrows and rises to the continental shelf edge. To the north, the channel opens into several deepwater basins of the Gulf of Maine. The V-shaped 100m isobath effectively delineates the steep drop-off from Nantucket Shoals and Georges Bank to the deeper basins…It is likely that a significant proportion of the western North Atlantic right whale population uses the GSC as a feeding area each spring, aggregating to exploit exceptionally dense copepod patches (59 FR 28805, 1994).

Although the Great South Channel is off of the coast of Massachusetts, its significant distance from any coastal facilities eligible under this permit precludes any adverse modification to this habitat from RGP discharges.

However, the Action Area for this general permit (as it relates to the North Atlantic right whale) does include the Massachusetts waters of Cape Cod Bay. In 59 FR 28805, Cape Cod Bay (CCB) is described as:

> a large embayment on the U.S. Atlantic Ocean off of the state of Massachusetts that is bounded on three sides by Cape Cod and the Massachusetts coastline from Plymouth, MA, south. To the north, CCB opens to Massachusetts Bay and the Gulf of Maine…The general water flow is counter-clockwise, running from the Gulf of Maine south into the western half of CCB, over to eastern CCB, and back into the Gulf of Maine through the channel between the north end of Cape Cod (Race Point) and the southeast end of Stellwagen Bank, a submarine bank that lies just north of Cape Cod…The late-winter/early spring zooplankton fauna of CCB consists primarily of copepods….The CCB may occasionally serve as a calving area, but it is more recognized for being a nursery habitat for calves that enter into the area after being born most likely in, or near, the SEUS.

The projected dimensions of the discharge plume of any RGP outfall are expected to be confined to the immediate shore and only extend out a minimal distance before complete mixing takes place. The critical habitat is not considered to extend to the immediate shoreline of Cape Cod. Contact between the critical habitat and the projected transient RGP discharge plumes is extremely unlikely to occur. Therefore, no adverse modification to critical habitat is expected.

Stellwagen Bank, is also a designated critical habitat, which is located at the mouth of Massachusetts Bay, between Cape Cod and Cape Ann. Since Stellwagen Bank is located approximately 5 miles east of Gloucester, MA and 5 miles north of Provincetown, MA, EPA believes that this distance would also preclude any contact between the discharges under this permit and the critical habitat. Therefore, no adverse modification to critical habitat is expected.

Population

According to NMFS' 2012 stock assessment of the western North Atlantic Right, the population was estimated to be at least 444 individuals in 2009 (NMFS, 2012)**.** This was based on the 1990-2009 census of individual whales, identified using photo-identification techniques. The stock assessment report emphasized that this was the minimum value of the population. Various studies indicated there was a decline in the whales' survival in the early 1980s and 1990s (NMFS, 2012). However according to an analysis of the current minimum alive population index, the geometric mean growth rate for the 1990-2009 period was 2.6% and there appears to be a positive, albeit slowly, accelerating trend in population size (NMFS, 2012).

iv. Population Risks and Stressors

Historically, the right whale population was brought to extremely low levels by commercial whaling (59 FR 28805, 1994). According to the most recent recovery plan, other anthropological activities, particularly ship collisions and entanglements in fishing gear are now the most common causes of mortality in North Atlantic right whales (NMFS, 2005). From 2005 to 2009, reports indicate that right whales had the greatest number of ship strike mortalities and serious injuries compared other large whales in the Northwest Atlantic (NMFS, 2013b)**.** Other potential

threats include habitat degradation, contaminants, climate/ecosystem change, and noise/disturbance from industrial activities and whale-watching activities (NMFS, 2005). Habitat degradation and contaminants are among additional threats and are the threats relevant to the proposed action.

1) **Habitat Degradation**: Pollution from human activities could possibly lead to habitat degradation.

2) **Contaminants in Whales**: According to the 2005 recovery plan, contaminant data on right whales have only been obtained from biopsy-derived samples (NMFS, 2005). Data from only two studies are available and the data indicated a total PCB range of 80 to 1000 ng/g wet weights (in the parts per billion range) for right whales (Woodley, Brown, Kraus, & Gaskin, 1991); (Moore, et al., 1998). Organic chemical contaminants are not considered to be the primary factors in slowing the recovery of any stocks of large whale species (O'Shea & Brownell, 1994).

EPA has determined that remediation activity discharges will have no effect on the north Atlantic right whale because the distance between the localized, on-shore remediation activities and minor, near-shore remediation activity discharges relative to the size of, and the high energy and volume in the marine waters this species is likely to inhabit, precludes contact between remediation activity discharges and this species, presently or in the future. As such EPA will not consider this species further in this analysis.

d. Fin Whale (*Balaenoptera physalus*) - Endangered

i. Life Stages and Activities

The fin whale, another type of baleen whale, is larger and faster swimming than the humpback and right whale (NMFS, 2010b); (NMFS, 2013b). They feed intensely in the summer and fast in the winter while they migrate to warmer waters (NMFS, 2010b). The overall distribution and movements of the fin whale may be based on the availability of its prey, which itself varies depending upon the geographical location (International Whaling Commission, 1992); (NMFS, 2010b). The fin whale of the western North Atlantic preys on crustaceans (mainly euphausiids or krill) and small schooling fish, including capelin, herring, and sand lance (Wynne & Schwartz, 1999); (Overholtz & Nicolas, 1979).

Little is known about the social and mating systems of fin whales (NMFS, 2013). Male fins whales achieve sexual maturity at 6-10 years of age while females become sexually mature at 7-12 years (Jefferson, Webber, & Pitman, 2008). However physical maturity is not attained for either sex until approximately 25 years of age (NMFS, 2013).Conception is believed to occur in tropical and subtropical areas during the winter months, and females give birth to a single calf after approximately 11-12 months of gestation (Jefferson, Webber, & Pitman, 2008). It has been estimated that the average calving interval is about 2 years (Christensen, Haug, & Oien, 1992).

ii. Status

The finback whale was originally listed under the Endangered Species Conservation Act of 1970 (35 FR 18319, 1970). It has maintained its listing as an endangered species when the Endangered Species Act (ESA) went into effect in 1973.

iii. Population and Distribution

Fin whales have a wide distribution throughout the world and can be found in the Atlantic, Pacific, and Southern Hemisphere (NMFS, 2010b). Although they inhabit a range of latitudes between 20-75ºN and 20-75 ºS (Perry, DeMaster, & Silber, 1999), they are most commonly found in the deep, offshore waters in temperate to polar latitudes (NMFS, 2013). As previously mentioned in Section 3.6.1, fin whales do migrate seasonally. Unlike the more evident north-south migration patterns of the humpback and right whales, the overall migratory pattern of fin whales is more complex and not currently well defined (NMFS, 2013).

According to the recent Recovery Plan, the population structure of fin whales has not been adequately defined and populations are often divided on an ocean basin level instead of strict biological evidence (NMFS, 2010b). Two named subspecies of the fin whale exist: *B. physalus* (Linnaeus 1758) in the North Atlantic and *B. physalus quoyi* (Fischer 1829) in the Southern Hemisphere (NMFS, 2010b). It is generally believed that the populations in the North Atlantic, North Pacific, and Southern Hemisphere rarely mix, if ever (NMFS, 2010b). Within the aforementioned ocean basins, there are geographical populations of fin whales. In U.S. waters, NMFS recognizes four MMA stocks: 1) the Western North Atlantic and the 2) Hawaii, 3) California/Oregon/ Washington, and 4) Alaska (Northeast Pacific) stocks of U.S. Pacific waters (NMFS, 2010b).

The fin whale is ubiquitous in the North Atlantic and occurs from the Gulf of Mexico and Mediterranean Sea northward to the edges of the Arctic ice pack (Reeves, Silber, & Payne, 1998b). They are common in waters of the U.S. Atlantic Exclusive Economic Zone, mainly from Cape Hatteras northward, up to Nova Scotia and the southeastern coast of Newfoundland (NMFS, 2013c). During aerial surveys that were conducted from 1978-1982, fin whales accounted for 46% of all large whales sighted over the continental shelf between Cape Hatteras and Nova Scotia (Waring, Josephson, Maze-Folew, & Rosel, 2012).

Although fin whales in the central and eastern North Atlantic are most abundant over the continental slope and on the shelf seaward of the 200 m isobaths (Rorvik, Jonsson, Mathisen, & Jonsgard, 1976), those off the eastern United States are generally centered along the 100-m isobaths with additional sighting spread out over shallower and deeper water (Kenney & Winn, 1986); (Hain, Ratnaswamy, Kenney, & Winn, 1992). An important feeding area for this species was identified from the Great South Channel, along the 50 meter isobaths past Cape Cod, Massachusetts, over Stellwagen Bank, and past Cape Ann to Jeffrey's Ledge (Hain, Ratnaswamy, Kenney, & Winn, 1992). Photo-identification studies in western North Atlantic feeding areas, especially in Massachusetts Bay, have indicated a high rate of annual return by fin whales to this feeding area (Seipt, Clapham, Mayo, & Hawvermale, 1990).

As mentioned earlier, the projected dimensions of the discharge plume of any RGP outfall are expected to be confined to the immediate shore and only extend out a minimal distance before

complete mixing takes place. The expected distribution of fin whales is not considered to include the immediate shoreline of the shallow, near-coastal Gulf of Maine waters. Therefore, contact between fin whales and the projected transient RGP discharge plumes is extremely unlikely to occur.

Reliable and recent estimates of fin whale abundance are available for significant portions of the North Atlantic Ocean, but neither for the North Pacific Ocean nor the Southern Ocean (NMFS, 2010b). There is insufficient data to determine population trends for the fin whale (Waring, Josephson, Maze-Folew, & Rosel, 2012). Various estimates have been provided to describe the current status of fin whales in western North Atlantic waters. However, the final 2012 stock assessment report provided the best population estimate of 3,522 (CV=0.27) for the western North Atlantic stock. This is considered the best estimate because the number is derived from the Canadian Trans-North Atlantic Sighting Survey (TNASS) which covered more of the fin whale range than other surveys (NMFS, 2013c).

Although reliable estimates of current abundance for the entire Northeast Pacific (Alaska) are not available, the final 2012 stock assessment report does provide a *minimum* estimate of 5,700 (Allen & Angliss, 2011). The best available estimate for the California/Oregon/Washington stock is 3,044, which is likely to be an underestimate (Carretta, et al., 2011). Based on a 2002 line-transect survey, the best available estimate for the Hawaii stock is 174 (Carretta, et al., 2011).

iv. Population Risks and Stressors

Historically, commercial whaling was the most significant threat to fin whales (NMFS, 2010b). Although commercial whaling of the fin whale ceased in the North Pacific Ocean in 1976, in the Southern Ocean in 1976, and in the North Atlantic Ocean in 1987 fin whales are still hunted today in Greenland under the IWC's "aboriginal subsistence whaling" scheme (NMFS, 2010b). Therefore, whaling is no longer the most significant threat, but the potential that illegal whaling and/or resumed legal whaling could adversely impact the fin whale population still exists today.

As with North Atlantic right and humpback whales, the most significant, known anthropologic threats to fin whales include collisions with vessels and entanglement in fishing gear (NMFS, 2010b). Out of all species of large whales, it is believed that fin whales are most commonly struck by large vessels (Laist, Knowlton, Mead, Collet, & Podesta, 2001). From 2005 – 2009, a study documented 12 ship strikes (9 fatal) of North Atlantic fin whales and 14 confirmed entanglements (2 fatal and 2 serious injuries) (Henry, Cole, Garron, & Hall, Mortality and Serious Injury Determinations for Baleen Whale Stocks along the Gulf of Mexico, United States and Canadian Eastern Seaboards, 2005-2009, 2011). Other threats to the fin whale include potential reduction in prey abundance due to overfishing or climate change, acoustic trauma, and habitat degradation. The threat relevant to the proposed action includes:

1) **Habitat Degradation:** According to the Recovery Plan for the fin whale, contaminants and pollutants were listed as a low threat (NMFS, 2010b). In a study by O'Shea and Brownell (1995), concentrations of organochlorine and metal contaminants in the tissues of baleen whales were low, and lower in fact that other marine mammal species.

EPA has determined that remediation activity discharges will have no effect on the fin whale because the distance between the localized, on-shore remediation activities and minor, near-shore remediation activity discharges relative to the size of, and the high energy and volume in the marine waters this species is likely to inhabit, precludes contact between remediation activity discharges and this species, presently or in the future. As such EPA will not consider this species further in this analysis.

e. Kemp's Ridley Sea Turtle (*Lepidochelys kempi)* - Endangered

i. Life Stages and Activities

The general life history pattern for Kemp's ridleys is similar to that of other sea turtles, including the loggerhead (Bolten, 2003). As summarized in the Kemp's ridleys revised recovery plan, its life history can be categorized by three overall ecosystems: 1) *Terrestrial zone* – the nesting beach where females lay eggs & eggs hatch; 2) *Neritic zone* – the nearshore marine environment that includes the water surface to ocean floor, with water depths no greater than 200 meters; and 3) *Oceanic zone* – the open ocean environment, where water depths exceed 200 meters (NMFS et al., 2011). This life history is also highlighted below:

Life Stages of Sea Turtles

Life Stage	Zone
Adult/Egg/Hatchling	Terrestrial
Early Transitional for Hatchling/Post-Hatchling	Neritic
Juvenile	Oceanic
Juvenile	Neritic
Adult	Neritic

Female Kemp's ridleys lay their nests on ocean beaches, primarily along a stretch of beach in Rancho Nuevo, Mexico, from April through July each year (NMFS et al., 2011). The Kemp's ridleys tend to nest in large, synchronized aggregations, called *arribadas*, which may be triggered by high wind speeds, especially north winds, and changes in barometric pressure (Jimenez, Filonov, Tereshchenko, & Marquex, 2005). Females lay an average of 2-3 clutches per season (Turtle Expert Working Group, 2000) and eggs typically take 45-58 days to hatch, depending on temperatures (NMFS & USFWS, 2007)..

Once hatchlings leave the nesting beaches, they quickly enter the surf and swim offshore. According to the revised recovery plan, not much is known about this 'early transitional neritic' phase in which the hatchling swims offshore and are associated with boundary currents, but *before* they are transported into the open ocean. The juveniles then feed, presumably on *Sargassum* seaweed or associated infauna, and develop in the ocean (NMFS et al., 2011).

After approximately 2 years of age, Kemp's ridleys will transition to benthic coastal habitats of the entire Gulf of Mexico and U.S. Atlantic coast and forage on benthic fauna, including a

variety of crabs (NMFS & USFWS, 2007; Turtle Expert Working Group, 2000). This movement represents the beginning of a new life stage, namely the juvenile developmental neritic stage (NMFS et al., 2011). The habitat where these juvenile Kemp's ridleys develop can be characterized as somewhat protected, temperate waters, with a depth below 50 m (NMFS et al., 2011). A variety of substrates have been documented as good foraging habitat and include seagrass beds, oyster reefs, rock outcroppings, and sandy and/or mud bottoms (NMFS & USFWS, 2007).

A large portion of the neritic juveniles resides in waters with temperatures that vary seasonally (NMFS et al., 2011). For those juveniles that forage in the Northwest Atlantic, they do migrate down the coast to more favorable (i.e.-warmer) overwintering sites when the water temperatures begin to decline each year (NMFS et al., 2011). The timing of this emigration depends upon the latitude of the foraging habitat, with earlier emigration in the more northern waters (NMFS et al., 2011). The offshore waters south of Cape Canaveral have been identified as an important overwintering area for seasonal migrants along the U.S. Atlantic coast (NMFS & USFWS, 2007). In the spring, Kemp's ridleys residing in east-central Florida waters migrate northward (NMFS & USFWS, 2007). As water temperatures continue to rise even farther northward, juvenile Kemp's ridleys and loggerheads continue their northward migration. By June, they might appear in New England waters (NMFS et al., 2011).

Although adult Kemp's ridleys occur primarily in the Gulf of Mexico, some are occasionally found on the U.S. Atlantic coast (NMFS & USFWS, 2007). Common habitat for adults are nearshore waters of 37 m or less that are rich in crabs and have a sandy or muddy bottom (NMFS & USFWS, 2007).

ii. Status

The Kemp's ridley sea turtle was originally listed under the Endangered Species Conservation Act of 1970 (35 FR 18319, 1970). It maintained its listing as an endangered species when the Endangered Species Act (ESA) went into effect in 1973. NOAA Fisheries and USFWS, which have joint jurisdiction for marine turtles, finalized the original recovery plan for Kemp's ridley turtles in the U.S. Caribbean, Atlantic and Gulf of Mexico in 1991 (NMFS, 2013). A revised bi-national (U.S. and Mexico) Recovery Plan was finalized in 2011. Since the largest nesting area occurs in Mexico, the Mexican government has played a critical role in the conservation of Kemp's ridley turtles. Since 1966, the Mexican government provided legal protection to the turtles. They implemented a complete ban on taking any species of sea turtle on May 28, 1990 (NMFS, 2013). NOAA Fisheries and USFWS were jointly petitioned in February of 2010 to designate critical habitat for Kemp's ridley sea turtles for nesting beaches along the coast of Texas and marine habitats in the Gulf of Mexico (WildEarth Guardians, 2010).

iii. Population and Distribution

The Kemp's ridley is one of the least abundant of the world's sea turtle species (NMFS, 2013b). Kemp's ridleys typically occur only in the Gulf of Mexico and the northwestern Atlantic Ocean, from Florida to New England (NMFS et al., 2011). The majority of Kemp's ridleys nest along a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico or the nearby beaches of Tepehuajes and Barra del Tordo (NMFS & USFWS, 2007); (NMFS et al., 2011). However, there

is a limited amount of nesting in the U.S, particularly in South Texas (NMFS et al., 2011). It is not known what proportion of the Kemp's ridley population migrates to U.S. Atlantic coastal waters (NMFS & USFWS, 2007).

As mentioned earlier, the projected dimensions of the discharge plume of any RGP outfall are expected to be confined to the immediate shore and only extend out a minimal distance before complete mixing takes place. The expected distribution of Kemp's ridley sea turtles is not considered to include the immediate shoreline of the shallow, near-coastal Gulf of Maine waters. Therefore, contact between Kemp's ridley sea turtles and the projected transient RGP discharge plumes is extremely unlikely to occur.

After emerging from the nest, hatchlings quickly enter the water to escape predators (NMFS et al., 2011). Although there is a brief neritic stage for hatchling/post-hatchling, not much is known of this transitional stage (NMFS et al., 2011). Post-hatchling Kemp's ridleys are believed to be carried by major oceanic currents and distributed predominantly in the Gulf of Mexico, but also in the Northwest Atlantic (NMFS & USFWS, 2007). The juveniles feed, often on *Sargassum* seaweed, and develop in the ocean (NMFS et al., 2011). After approximately 2 years of age, Kemp's ridleys will transition to benthic coastal habitats of the entire Gulf of Mexico and U.S. Atlantic coast (NMFS & USFWS, 2007); (Turtle Expert Working Group, 2000). Data indicates that developmental habitats for this life stage can occur in many coastal areas throughout the aforementioned range, and that these habitats may shift depending upon the availability of resources (Turtle Expert Working Group, 2000). Foraging areas along the U.S. coast include Charleston Harbor, Pamlico Sound, Chesapeake Bay, Delaware Bay, and Long Island Sound, North Carolina, as well as New York and New England (NMFS, 2013b). Adult Kemp's ridleys can be found in the coastal regions of the Gulf of Mexico and southeastern United States, but they are typically rare in the northeastern U.S. waters of the Atlantic (Turtle Expert Working Group, 2000).

According to the revised Recovery Plan for Kemp's ridley turtles, the nesting population is increasing exponentially, which may indicate that the population as a whole is increasing (NMFS et al., 2011). Although the number of nesting females was estimated to be 40,000 in 1947, the Kemp's ridley population declined significantly through the mid-1980's to fewer than 300 nesting females in the entire 1985 nesting season (Turtle Expert Working Group, 2000); (NMFS et al., 2011). As previously stated, egg collection was historically an extreme threat to this species' population. However, the total number of nests at Rancho Nuevo and nearby beaches started to increase in the mid-1980's, with a 14-16% increase per year from 1988 – 2003 (NMFS et al., 2011). In 2009 alone, the total number of nests recorded at Rancho Nuevo and adjacent beaches exceeded 20,000, which represented approximately 8,000 nesting females (NMFS et al., 2011). Although there is limited nesting in the United States, a record 195 nests were documented in South Texas compared to only 6 in 1996 (NMFS et al., 2011). An updated population model, which is based on the assumption that current survival rates within each life stage remain constant, predicted a 19% per year population growth from 2010 – 2020 (Heppell, et al., 2005); (NMFS et al., 2011).

iv. Population Risks and Stressors

Like other species of sea turtles, threats to Kemp's ridleys occur both on land (on nesting beaches) and in the marine environment (NMFS, 2013b). Historically, the exploitation of eggs in Mexico was a major factor in the decline of the Kemp's ridley sea turtle nesting population (NMFS & USFWS, 2007). Although poaching of eggs occasionally still takes place in Mexico, there was a dramatic decrease since official beach protection started in 1966/67 (NMFS et al., 2011).

The greatest threats to marine turtles, including Kemp's ridleys include incidental capture in fishing gear (from commercial and recreational fisheries), loss or destruction of nesting habitat, cold-stunning, pollution, and climate change. The threat relevant to the proposed action includes:

1) **Pollution:** According to NMFS's five-year review of Kemp's ridleys, exposure to heavy metals and other contaminants in the marine environment, including oil from spills or pollutants from coastal runoff, are potential threats (NMFS & USFWS, 2007). Although explicit effects on sea turtle have not been documented yet, toxins are capable of altering metabolic activities, development, and reproductive capacity (NMFS et al., 2011).

EPA has determined that remediation activity discharges will have no effect on the Kemp's Ridley sea turtle because the distance between the localized, on-shore remediation activities and minor, near-shore remediation activity discharges relative to the size of, and the high energy and volume in the marine waters this species is likely to inhabit, precludes contact between remediation activity discharges and this species, presently or in the future. As such EPA will not consider this species further in this analysis.

f. Loggerhead Sea Turtle (*Caretta caretta)* – Northwest Atlantic Ocean DPS - Threatened

i. Life Stages and Activities

As previously mentioned, the generalized life stages of loggerhead sea turtles are similar to the life stages of other turtles, including Kemp's ridley sea turtles (Heppell, Crowder, Crouse, Epperly, & Frazer, 2003). Therefore, the phases discussed in the above section for life stages and activities for Kemp's ridleys, including those that occur in the terrestrial, neritic, and oceanic zones, are applicable for this section, as well. However, recent studies have established that the loggerhead's life history is more complex than originally believed. According to a recent NMFS Biological Opinion, research is showing that both adults and most likely neritic stage juveniles continue to move between their oceanic and neritic environments rather than making discrete development shifts between the two habitats (NMFS, 2013b). Neritic refers to the inshore marine environment from the surface to the sea floor in which water depths do not exceed 200 meters.

Loggerheads nest on ocean beaches and sometimes on estuarine shorelines with suitable sand. Females appear to prefer relatively narrow, steeply sloped beaches with coarse-grained sand (NMFS & USFWS, 2008). In the Northwest Atlantic, the major nesting concentrations in the U.S. are located from North Carolina through southwest Florida (Conant, et al., 2009). The table below, which was taken from Table 3 of the Revised Recovery Plan, highlights some of the life

history parameters and key values for loggerheads that nest in the U.S. (NMFS & USFWS, 2008).

Typical values of life history parameters for loggerheads nesting in the U.S.

Life History Parameter	**Data**
Clutch size	100 – 126 eggs (Dodd 1988)
Clutch frequency (number of nests/female/season)	3 – 5.5 nests (Murphy and Hopkins (1984); Frazer and Richardson (1985); Hawkes *et al.* 2005; Scott 2006)
Nesting season	Late April – early September
Hatching season	Late June – early November
Age at sexual maturity	32-35 years (Melissa Snover, NMFS, personal communication, 2005; See Table A1-6)

Immediately after the hatchlings emerge from the nest, they are known to exhibit a period of frenzied activity. They move from their nest to the surf, swim and are swept through the surf zone, and continue swimming away from land for about 20-30 hours (NMFS & USFWS, 2008). After this frenzied phases, post-hatchlings enter a transitional, neritic phrase where they inhabit waters near the shoreline for weeks to months (NMFS & USFWS, 2008). These post-hatchlings have been described as low-energy float and wait foragers that feed upon a variety of floating items, including *Sargassum* seaweed (Witherington, Ecology of neonate loggerhead turtles inhabiting lines of downwelling near a Gulf Stream front, 2002).

Juvenile loggerheads then enter into an oceanic stage during which they spend about 75% of their time in the top 5 meters of the water column (Heppell, Crowder, Crouse, Epperly, & Frazer, 2003). Although the diet of these juveniles has not been studied extensively, they are known to be largely carnivorous; they primarily eat sea jellies and hydroids, and occasionally other organisms like snails, barnacles and crabs (NMFS & USFWS, 2008). After years of this phase, the juveniles transition from the oceanic to the neritic zone. According to the 2008 Recovery Plan, juvenile stage loggerheads in the North Atlantic commonly inhabit continental shelf waters from Cape Cod Bay, MA south though Florida, The Bahamas, and the Gulf of Mexico (NMFS & USFWS, 2008). North Atlantic subadults (as well as adults) are believed to eat a variety of organisms, including benthic invertebrates such as mollusks and benthic crabs (Burke, Standora, & SJ, 1993)**.** Matrix models estimate that this neritic juvenile stage can last from 14 to 24 years (Heppell, Crowder, Crouse, Epperly, & Frazer, 2003).

Although non-nesting adult loggerheads also inhabit the neritic zone, the habitat preference for adults differs from that of juveniles (Conant, et al., 2009). Adults prefer shallow water habitats with vast access to the open ocean, like Florida Bay, as compared to juveniles who more frequently use enclosed, shallow water estuarine habitats with limited ocean access (Conant, et al., 2009). Offshore, adults primarily inhabit continental shelf waters, from New York south through Florida, The Bahamas, Cuba, and the Gulf of Mexico (NMFS & USFWS, 2008). Loggerheads are known to make extensive seasonal migrations between foraging areas and nesting areas (NMFS & USFWS, 2008).

ii. Status

On July 28, 1978, the loggerhead turtle was initially listed as a threatened species under the Endangered Species Act throughout its range (43 FR 32800, 1978). In 2007, NMFS (which is the lead agency for marine turtles) and the U.S. Fish and Wildlife Service (which is the lead authority for the terrestrial areas/nesting beaches of sea turtles) completed a five-year status review of loggerheads. The results of this review, as well as the second revision of the Recovery Plan for the Northwest Atlantic Population, were published in 2009.

In September of 2011, NMFS listed 9 Distinct Population Segments (DPSs) of loggerhead sea turtles under the ESA (76 FR 58868, 2011). Five DPSs were listed as endangered (North Pacific Ocean, South Pacific Ocean, North Indian Ocean, Northeast Atlantic Ocean, and Mediterranean Sea) while four DPSs were listed as threatened (Northwest Atlantic Ocean, South Atlantic Ocean, Southeast Indo-Pacific Ocean, and Southwest Indian Ocean) (76 FR 58868, 2011)(. It should be noted that the Northwest Atlantic DPS was one of two DPSs originally proposed as endangered; however, it was eventually listed as threatened based on population abundance and population trends (NMFS, 2013b).

In July of 2013, NMFS proposed the designation of critical habitat for the Northwest Atlantic Ocean DPS of Loggerhead Sea Turtle (78 FR 43305, 2013). 36 occupied marine areas within the Atlantic Ocean and the Gulf of Mexico, which contain "one or a combination of nearshore reproductive habitat, winter area, breeding areas, and migratory corridors," were proposed (78 FR 43305, 2013). None of the proposed marine areas are located within or near Massachusetts' waters.

iii. Population and Distribution

Loggerhead sea turtles are the most abundant species of sea turtle found in U.S. coastal waters (NMFS, 2013b). They occur throughout the temperate and tropic regions of the Atlantic, Pacific, and Indian Oceans (Dodd, 1988). Neritic juvenile loggerheads in the Northwest Atlantic DPS inhabit continental shelf waters from Cape Cod Bay, Massachusetts, south through Florida, The Bahamas, Cuba, and the Gulf of Mexico (76 FR 58868, 2011). However, it should be noted that their presence varies with the seasons due to the changes in water temperature (NMFS, 2013b).

Although some loggerhead sea turtles occur year round in ocean waters off North Carolina, South Carolina, Georgia, and Florida, others begin to migrate to inshore waters of the Southeast United States and also move up in the U.S. Atlantic coast as coastal water temperatures warm in the spring (NMFS, 2013b). Loggerheads can appear in Virginia foraging areas as early as April/May and on the most northern foraging grounds in the Gulf of Maine in June (Shoop & Kenney, 1992). The trend is reversed in the fall as water temperatures cool (NMFS, 2013b).

As mentioned earlier, the projected dimensions of the discharge plume of any RGP outfall are expected to be confined to the immediate shore and only extend out a minimal distance before complete mixing takes place. The expected distribution of loggerhead sea turtles is not considered to include the immediate shoreline of the shallow, near-coastal Gulf of Maine waters. Therefore, contact between loggerheads and the projected transient RGP discharge plumes is extremely unlikely to occur.

According to the revised recovery plan, five recovery units were identified for the NWA DPS of loggerheads (NMFS & USFWS, 2008). These recovery units, which are based on nesting assemblages of the Northwest Atlantic DPS, are summarized below (NMFS & USFWS, 2008). Nest counts can be used to estimate the number of reproductively mature females nesting annually (NMFS, 2013b). In addition to listing the recovery units, the table also provides the population status/trend for each recovery unit (NMFS & USFWS, 2008).

Description of Recovery Units of Northwest Atlantic DPS of Loggerheads & Population Status/Trends

Recovery Unit	**Geographic Location**	**Population Status/Trends**
Northern Recovery Unit (Represents northern-most range)	Loggerheads originating from nesting beaches from Florida-Georgia border through southern Virginia	From 1989-2008, total annual nest averaged 5,215 nests with approximately 1,272 females nesting per year (NMFS & USFWS, 2008).
Peninsular Florida Recovery Unit (Largest nesting assemblage for NWA DPS)	Loggerheads originating from nesting beaches from the Florida-Georgia border through Pinellas County of West coast of FLR (excludes islands west of Key West)	From 1989-2007, total annual nest averaged 64,513 nests with about 15,735 females nesting per year (NMFS & USFWS, 2008). From 1989-2008, overall declining nesting trend of 26%
Dry Tortugas Recovery Unit	Loggerheads originating from nesting beaches throughout islands located west of Key West, FL	From 1995-2004 (excluding 2002), total annual nest averaged 246 nests with approximately 60 females nesting per year (NMFS & USFWS, 2008).
Northern Gulf of Mexico Recovery Unit (Western Extent of U.S. nesting range)	Loggerheads originating from nesting beaches from Franklin County of Northwest Gulf coast of FL through Texas	Total annual nests from 1995-2007 averaged 906 nests with approximately 221 females nesting per year (NMFS & USFWS, 2008).
Greater Caribbean Recovery Unit	Loggerheads originating from all other nesting assemblages within the Greater Caribbean	Only available estimate is from Quintana Roo, Yucatan, Mexico: range of 903-2,331 nest per year from 1987-2001 (NMFS and USFWS 2007a **Get source)**; Nesting has declined since 2001 (NMFS & USFWS, 2008).

The 2008 Recovery Plan indicated that there had been a significant, overall nesting decline within the Northwest Atlantic DPS based on standardized data collected prior to October of 2008 (NMFS & USFWS, 2008). However, with the addition of nesting data from 2008-2010, the trend line has changed; although there is now a slight negative trend, the rate of decline is not statistically different from zero (76 FR 58868, 2011).

In the summer of 2010, line transect aerial abundance surveys (from Cape Canaveral, FL to the Gulf of St. Lawrence, Canada) and turtle telemetry studies were conducted along the Atlantic coast as part of the Atlantic Marine Assessment Program for Protected Species (AMAPPS) (NMFS NEFSC, 2011). The 2010 survey found a preliminary total surface abundance estimate within the study area of about 60,000 loggerheads (or 85,000 if a portion of unidentified hard-shelled sea turtles were included (NMFS NEFSC, 2011). The calculated preliminary regional abundance estimate is about 588,000 loggerheads along the U.S. Atlantic coast, with an inter-quartile range of 382,000 – 817,000 (NMFS NEFSC, 2011). However, these estimates are considered very preliminary. It should be noted that population estimates for loggerhead sea turtles (as with other turtle species) are difficult to determine, particularly because of their life history characteristics (NMFS, 2013b).

iv. Population Risks and Stressors

The threats outlined earlier in this document for Kemp's ridley sea turtles are also applicable to other sea turtles, including loggerheads. Therefore, they will not be repeated in detail again. It is important to note that the factors that threaten sea turtles in the terrestrial zone (i.e.-on nesting beaches) often differ from those that threaten the turtles in the neritic and ocean zones. The 2008 Recovery Plan emphasized that the *highest* priority threats for the Northwest Atlantic DPS of loggerheads include Bycatch from fisheries, legal and illegal harvesting, vessel strikes, beach erosion, marine debris entanglement/ingestion, oil pollution, light pollution, and predation by native and exotic species. The threat relevant to the proposed action includes:

1) **Oil pollution:** Effects of oil pollution can occur in sea turtles at every life stage. Since sea turtles surface to breathe several times an hour and many oils float, sea turtles can repeatedly inhale and ingest oil, become covered in oil to the point of being impaired or unable to swim, or lose habitat or prey that is killed or contaminated by oil. Inhaling and swallowing oil can result in negative health effects, including hindering their overall health, growth, and survival, irritating sensitive mucus membranes around the eyes, mouth, lungs, and digestive tracts, and impacting organ function. Oil can become trapped in sea turtles' esophageal papillaes, impeding breathing. Oil compounds such as polycyclic aromatic hydrocarbons (PAHs) can be absorbed into vital organ tissues such as the lungs and liver. Oil compounds can also cause reproductive effects, interfering with development and survival. (NOAA Office of Response and Restoration, June 2016).

EPA has determined that remediation activity discharges will have no effect on the loggerhead sea turtle because the distance between the localized, on-shore remediation activities and minor, near-shore remediation activity discharges relative to the size of, and the high energy and volume in the marine waters this species is likely to inhabit, precludes contact between remediation

activity discharges and this species, presently or in the future. As such EPA will not consider this species further in this analysis.

g. Leatherback Sea Turtle (*Dermochelys coriacea)* - Endangered

Although leatherback sea turtles are listed as endangered on the species level, existing recovery plans are based upon population and management units within ocean basins. For example, the Recovery Plan for Leatherback Turtles in the U.S. Caribbean, Atlantic, and Gulf of Mexico was signed by NMFS and the USFWS in 1992, while the Recovery Plan for U.S. Pacific Populations of Leatherback Turtle was signed in 1998. The recent five-year status review for leatherback turtles also concluded that a Distinct Population Segment policy was recommended for leatherbacks. Therefore, the section below will focus on leatherback sea turtles in the U.S. Caribbean, Atlantic, and Gulf of Mexico because this includes the Action Area for this permit, namely Massachusetts and New Hampshire waters.

i. Life Stages and Activities

Leatherbacks are the largest living turtles and the only sea turtle that doesn't have a hard bony shell; instead, a leatherback's carapace (top shell) is made of leathery, oil-saturated connective tissue that lies above loosely interlocking dermal bones (NMFS & USFWS, 1992). Also unlike other sea turtles which possess chewing plates that enable them to feed on hard-bodied prey, leatherbacks have two tooth like projections that help them eat their diet of soft-bodied and gelatinous organisms, including jellyfish and salps (Pritchard, 1971); (NMFS & USFWS, 1992).

Courtship and mating for leatherbacks is believed to occur in coastal waters adjacent to nesting beaches and along migratory corridors (NMFS, 2013). Nesting beach habitat is generally associated with deep water and strong waves and oceanic currents; however, leatherbacks will also use shallow water with mud banks (Turtle Expert Working Group, 2007). Female leatherbacks appear to prefer beaches with coarse-grained sand that are also free of rocks or other abrasive substrates (Eckert, Wallace, Frazier, Eckert, & Pritchard, 2012); (NMFS & USFWS, 2013). In the United States and Caribbean, female leatherbacks nest from March through July (NMFS, 2013b). They nest frequently (ranging from 5 -7 nests per year) and nesting occurs about every 2-3 years (Eckert, Wallace, Frazier, Eckert, & Pritchard, 2012); (NMFS & USFWS, 2013) . During the nesting season, females will generally stay within 100 km of the nesting beach. However, they also undergo long distances between nesting events to forage in more temperate areas which support a high density of prey (Eckert, Wallace, Frazier, Eckert, & Pritchard, 2012); (NMFS & USFWS, 2013).

Little is known about the early life history of leatherbacks from the time they are hatchlings until they reach adulthood (NMFS & USFWS, 2013). However, one study found that leatherback juveniles remain in waters warmer than 26°C until their curved carapace length (CCL) exceeds 100 cm; this suggests that the first part of a leatherback's life is spent in tropical waters (Eckert S., 2002).

Adult leatherbacks are highly migratory and believed to be the most pelagic of all sea turtles (NMFS & USFWS, 1992). Based on evidence from tag returns and strandings in the western

Atlantic Ocean, data suggests that adult leatherback sea turtles engage in routine migrations between northern temperate and tropic waters (NMFS & USFWS, 1992). Although leatherbacks primarily eat gelatinous organisms, they also ingest other prey including crustaceans, vertebrates, and plants (Eckert, Wallace, Frazier, Eckert, & Pritchard, 2012). It is essential that leatherbacks have access to areas of high food productivity because they must consume large amounts of such food to meet their energy demands (Heaslip, Iverson, & Bowen, 2012).

ii. Status

The leatherback turtle was originally listed under the Endangered Species Conservation Act of 1970 (35 FR 8491, 1970). It maintained its listing as an endangered species when the Endangered Species Act (ESA) went into effect in 1973.

In 1988, NMFS designated critical habitat for leatherback turtles in the U.S. Virgin Islands, specifically for the coastal waters adjacent to Sandy Point, St. Croix, USVI (44 FR 17710, 1979). According to 44 FR 17710, courtship and mating for leatherbacks is believed to occur in these coastal waters which are adjacent to nesting beaches. (The USFWS had already designated a 0.2-mile-wide strip of land at Sandy Point Beach as critical habitat in 1978). Additional critical habitat for endangered leatherback sea turtles was designated in 2012. This critical habitat is located along the U.S. West Coast. It includes approximately 16,910 square miles and was designated because of the abundant occurrence of prey species for leatherback sea turtles (77 FR 4170, 2012).

iii. Population and Distribution

Leatherback sea turtles are widely distributed throughout the world's oceans, including the Atlantic, Pacific, and Indian Oceans, as well as the Mediterranean Sea (Ernst & Barbaour, 1972). These migratory sea turtles range farther than any other sea turtles (NMFS, 2013b). They also have a distinct physiology with various thermoregulatory adaptations that allow leatherbacks to tolerate colder water temperatures than other sea turtles (NMFS & USFWS, 1992). Therefore, they can be found in foraging grounds as far north as Labrador in the Western North Atlantic Ocean (NMFS & USFWS, 2013). Although leatherbacks are known as pelagic animals because they live in the open ocean, they do forage in coastal waters, including those of the U.S. continental shelf (NMFS, 2013b).

Leatherbacks nest on beaches in the tropics and sub-tropics and they forage into higher-latitude sub-polar waters (NMFS & USFWS, 2013). Although nesting sites for leatherbacks exist around the world, the largest nesting assemblages currently exist along the northern coast of South America and in Western Africa (Turtle Expert Working Group, 2007). The most significant leatherback nesting sites in the United States occur in the U.S. Virgin Islands (the aforementioned Sandy Point Beach in St. Croix), Culebra in Puerto Rico, and along the east coast of Florida (NMFS & USFWS, 2013). Tagging and satellite telemetry data indicate that the leatherback turtles from these western North Atlantic nesting beaches use the entire North Atlantic Ocean (Turtle Expert Working Group, 2007). For instance, leatherbacks that were tagged in Puerto Rico, Trinidad, and the Virgin Islands have subsequently been found on U.S. beaches of southern, mid-Atlantic, and northern states (NOAA, 2013).

According to the five-year status review, migration patterns differ by region, depending upon the local oceanographic processes, and several migration strategies may exist within breeding populations (NMFS & USFWS, 2013). For leatherbacks in the Atlantic Ocean, some made round-trip migrations from where they started through the North Atlantic Ocean heading northwest to fertile foraging areas off the Gulf of Maine, Canada, and Gulf of Mexico; others crossed the ocean to areas off western Europe and Africa; while others spent time between northern and equatorial waters (NMFS & USFWS, 2013). Extensive research has been conducted on Canadian waters, which has one of the largest seasonal foraging population of leatherbacks in the Atlantic Ocean, as well as foraging areas off Massachusetts (particularly Cape Cod Bay) (NMFS & USFWS, 2013). According to the 1991 Recovery Plan for Leatherbacks in the U.S. Caribbean, Atlantic, and Gulf of Mexico, peak sightings for leatherbacks foraging in Cape Cod Bay, Massachusetts took place in August and September (Prescott, 1988); (NMFS & USFWS, 1992).

As mentioned earlier, the projected dimensions of the discharge plume of any RGP outfall are expected to be confined to the immediate shore and only extend out a minimal distance before complete mixing takes place. The expected distribution of leatherback sea turtles is not considered to include the immediate shoreline of the shallow, near-coastal Gulf of Maine waters. Therefore, contact between leatherback sea turtles and the projected transient RGP discharge plumes is extremely unlikely to occur.

The five-year review also compiled the most recent information on abundance and population trends for leatherback sea turtles in each of the ocean basins. The most recent population size estimate for the North Atlantic alone is a range of 34,000 – 94,000 adult leatherback sea turtles (Turtle Expert Working Group, 2007). However, it should be noted that it is particularly difficult to monitor nesting population estimates and trends for adult female leatherbacks because they are known to frequently nest on different beaches (NMFS, 2013). The table below summarizes the results for only a select number of nesting assemblages, namely those nesting sites affiliated with the United States.

Leatherback nesting Population Site Location Information

Location	Data: Nests, Females	Years	Annual Number	Trend	Reference
U.S. (Florida)	Nests	1979 - 2008	63-754	Increase	(Steward, et al., 2011)
Puerto Rico (Culebra)	Nests	1993 - 2012	395 - 32	Decrease	C. Diez, Department of Natural and Environmental Resources of Puerto Rico,, unpublished data; (Diez, et al., 2010); (Ramirez-Gallego, Diez, Barriento-Munoz, White, &

					Roman, 2013)
Puerto Rico (other)	Nests	1993 - 2012	131 – 1,291	Increase	C. Diez, Department of Natural and Environmental Resources of Puerto Rico,, unpublished data;
United States Virgin Islands (Sandy Point National Wildlife Refuge, St. Croix)	Nests	1986 - 2004	143-1,008	Increase	(Dutton, Dutton, Chaloupka, & Boulon, 2005); (Turtle Expert Working Group, 2007)

Since overall increases were recorded for mainland Puerto Rico and St. Croix, U.S. Virgin Islands, this might indicate that the decline of nests in Culebra might not be an actual loss to the breeding population; instead, it might just represent a shift in nesting site (Diez, et al., 2010); (Ramirez-Gallego, Diez, Barriento-Munoz, White, & Roman, 2013).

The 5-year review did observe contrasting population trends between the Atlantic, Pacific, and Indian Oceans. For instance, leatherback nesting populations are declining dramatically in the Pacific Ocean, yet appear stable (or are increasing) in many of the nesting areas of the Atlantic Ocean and South Africa in the Indian Ocean (NMFS & USFWS, 2013). No long-term data is available for nesting areas in West Africa (Turtle Expert Working Group, 2007). Many hypotheses have been proposed to explain the disparate trend of leatherbacks in the Pacific Ocean, including the variability in resource abundance (i.e.- prey) and distribution (NMFS & USFWS, 2013). For example, the high reproductive output and consistent, high quality foraging area in the Atlantic Ocean have likely contributed to their stable/recovering populations while lower prey abundance and distribution in the Pacific Ocean might be leading to this population's decline (NMFS & USFWS, 2013).

iv. Population Risks and Stressors

As with other sea turtles, both natural and anthropogenic threats impact the leatherback sea turtles' nesting and marine habitats. Two of the greatest threats to leatherbacks worldwide include the collection of eggs and harvesting of turtles, and incidental capture in fishing gear in artisanal and commercial fishing.
According to the most recent 5-year review of leatherback, additional threats include ingestion of & Entanglement of Marine Debris, development along coastal areas, and climate change. These threats are not expected to be associated with the proposed action.

EPA has determined that remediation activity discharges will have no effect on the leatherback sea turtle because the distance between the localized, on-shore remediation activities and minor, near-shore remediation activity discharges relative to the size of, and the high energy and volume in the marine waters this species is likely to inhabit, precludes contact between remediation activity discharges and this species, presently or in the future. As such EPA will not consider this species further in this analysis.

h. Green Turtle (*Chelonia mydas*) – Threatened or Endangered Threatened for Most Populations; Endangered for breeding populations in Florida & Pacific Coast of Mexico

i. Life Stages and Activities

Similar to the Kemp's ridley, loggerhead, and leatherback sea turtles, the green turtle uses three distinct habitats throughout its lifetime. These include: 1) high-energy beaches for nesting habitat, 2) convergence zones in the open (pelagic) ocean, and 3) relatively shallow, coastal waters which serve as their benthic feeding grounds (NMFS & USFWS, 1991). According to the five-year review for the green turtle, relatively recent research has started to increase the understanding of the species, particularly during its time in the marine environment, but numerous gaps still exist (NMFS & USFWS, 2007b). This is particularly true of the oceanic phase of juvenile green turtles.

Mating occurs in the water off nesting beaches (NMFS & USFWS, 1991). Although the nesting season for the green turtle depends upon the location of the nest, females from the Florida breeding population generally nest between June and September, with the peak occurring in June and July (NMFS, 2013). Florida green turtles nest approximately 3-4 times per season (Johnson, 1994) and have a mean of 136 eggs per nest (Witherington & Ehrhart, Status of reproductive characteristics of green turtles (Chelonia mydas) nesting in Florida, 1989). Green turtles do exhibit a strong fidelity to their natal beaches and females generally lay eggs every two to four years (NMFS & USFWS, 1991).

As mentioned earlier, the projected dimensions of the discharge plume of any RGP outfall are expected to be confined to the immediate shore and only extend out a minimal distance before complete mixing takes place. The expected distribution of adult and juvenile green sea turtles is not considered to include the immediate shoreline of the shallow, near-coastal Gulf of Maine waters. Therefore, contact between green sea turtles and the projected transient RGP discharge plumes is extremely unlikely to occur.

Hatchlings leave the beach and apparently move into convergence zones in the open ocean (Carr A. , 1986). Once they reach a certain size/age, they move to coastal foraging areas, which includes both open coastline and protected bays (NMFS & USFWS, 2007b). The primary diet of adult green turtles consists of marine algae and seagrass, although some populations also forage on invertebrates (NMFS & USFWS, 2007b).

Adult green turtles participate in breeding migrations between foraging grounds and nesting areas every few years (Plotkin, 2003). They migrations can be extensive, ranging from hundreds to thousands of kilometers (NMFS & USFWS, 2007b).

ii. Status

The green sea turtle was originally listed under the ESA on July 28, 1978. All populations of the green sea turtle were listed as threatened, except for the Florida and Mexican Pacific coast breeding populations which were listed as endangered (43 FR 32800, 1978). The waters surrounding Culebra Island in Puerto Rico has been designated as critical habitat for the green turtle, largely in part to the extensive amount of turtle grass present (63 FR 46693, 1998). Since seagrasses, such as turtle grass, represent an important component of the diet of juvenile and adult green turtles, these coastal waters provide important green turtle developmental habitat (63 FR 46693, 1998).

iii. Population and Distribution

Originally, the green sea turtle was abundant in tropical and subtropical regions throughout the world (NMFS & USFWS, 2007b). Although the species have declined significantly from its high historical numbers, green turtles are still believed to inhabit the continental coastal areas of more than 140 countries (NMFS & USFWS, 2007b); (Groombridge & Luxmoore, 1989). Green turtles are known to be high mobile and they partake in complex migratory behavior throughout their lifetimes (Musick & Limpus, 1997); (Plotkin, 2003). Similar to the sea turtles mentioned earlier in this document, a notable feature of the adult green turtle's life history is the migration between nesting sites and foraging areas (NMFS & USFWS, 2007b).

Below, information will be presented about green sea turtle nesting sites and discuss the breeding population in Florida (which is the only nesting area that occurs in the United States). Green turtles spend the majority of their lives in coastal foraging grounds which include both open coastline and protected bays and/or lagoons, where prey species like marine algae and seagrass are found (NMFS & USFWS, 2007b). So in addition to nesting sites in Florida, green turtles are also found in US waters.

In the U.S. waters of the western Atlantic Ocean, large juvenile and adult green sea turtles can be found (seasonally) in foraging and/or developmental habitats that stretch from Massachusetts to Texas, including the Gulf of Mexico (NMFS & USFWS, 1991). Key feeding areas in the western Atlantic Ocean also include the upper west coast of Florida, the Florida Keys, the northwestern coast of the Yucatan Peninsula, and the aforementioned designated critical habitat near Culebra Island in Puerto Rico (NMFS, 2013b); (NMFS & USFWS, 1991). Foraging areas for the green turtle are also found throughout the Pacific Ocean and along the southwestern U.S. coast (NMFS, 2013b). However, for the eastern North Pacific Ocean, green turtles most commonly inhabit waters from San Diego south (NMFS & USFWS, 1991). The coastal waters of northwestern Mexico are known to be a particularly important foraging region for turtles that originate from mainland Mexico (NMFS & USFWS, 1991).

As previously mentioned, there has been a tremendous decline in the number of green turtles worldwide compared to historical numbers which can largely be attributed to the overharvesting

of eggs and adults (NMFS & USFWS, 2007b). After analyzing historical and recent population trends for green turtles at 32 index nesting sites around the world, the Marine Turtle Specialist Group reported a 48-65% reduction in the number of mature females that nested annually over the past 100-150 years (NMFS, 2013).

The two largest nesting populations for the green sea turtle exist outside of the United States. One nesting population where an average of 22,500 females nest per season occurs on Tortuguero, which is located on the Caribbean coast of Costa Rica (NMFS, 2013). This is the most important nesting concentration for green sea turtles in the western Atlantic (NMFS & USFWS, 2007b). The other nesting population, where an average of 18,000 female green turtles nest per season, can be found on Raine Island on Australia's Great Barrier Reef (NMFS, 2013).

The most recent 5-Year review of the green turtle provided current nesting abundance for over 40 threatened and endangered nesting concentrations among 11 ocean regions throughout the world (NMFS & USFWS, 2007b). Those ocean regions included Western-, Central-, and Eastern Atlantic Ocean, Mediterranean Sea, Western-, Northern, and Eastern Indian Ocean, Southeast Asia, and Western-, Central-, and Eastern Pacific Ocean. Of the eight nesting locations in the Atlantic/Caribbean, all but one in the Eastern Atlantic Ocean, showed stable or increasing nest count/abundance data (NMFS & USFWS, 2007b). (Although the nesting site at Bioko Island in the eastern Atlantic Ocean might be decreasing, there was not sufficient data to determine a meaningful trend (NMFS & USFWS, 2007b). Similarly, eight of the nine nesting locations in the Pacific Ocean showed stable or increasing abundance trends (NMFS & USFWS, 2007b).

It should be noted that only one of the aforementioned nesting sites is located in the United States. This is the ESA-endangered breeding population in the state of Florida. Although most nesting occurs along a six county area in east central and southeast Florida, some occasional nesting has also been documented in other parts of the state (NMFS & USFWS, 1991); (Meylan, Schroeder, & Mosier, 1995). According to the five-year review of the green turtle, nesting data collected during the 2000-2006 Statewide Nesting Beach Survey (SNBS) indicated that a mean of approximately 5,6000 nests are laid annually in Florida (NMFS & USFWS, 2007b). According to the Index Nesting Beach Survey (INBS) program, which has determined nesting trends at a specific number of beaches since 1989 and is distinct from the SNBS initiative, there has been an overall positive nesting trend for the Florida breeding population of green turtles (NMFS & USFWS, 2007b).

The green turtle breeding population along the Pacific coast of Mexico is also listed as an endangered population (43 FR 32800, 1978). The primary nesting concentration for this population (also known as black turtles) is located at Colola – Michoacan in Pacific Mexico (NMFS & USFWS, 2007b). According to the most recent five-year review, the annual mean nests for the Colola, Michoacan site from 2000-2005 was 4,326 nests (NMFS & USFWS, 2007b).

iv. Population Risks and Stressors

Green sea turtles encounter many of the same natural threats to the terrestrial and marine environments as loggerhead and Kemp's ridley sea turtles (NMFS, 2013b). Therefore, the explanations provided earlier still apply. Some of the threats, as outlined in the five-year review

of the green turtle, include the collection of eggs and harvesting of turtles (for commercial and subsistence use), coastal development including the construction of buildings, beach armoring, and sand extraction, contamination from anthropogenic disturbances, fisheries bycatch, particularly in nearshore artisanal fisheries gear, and climate change. The threat relevant to the proposed action includes:

1) **Contamination from anthropogenic disturbances:** Contamination from herbicides, pesticides, chemicals, and oil spills can directly threaten the coastal marine habitats, including the seagrass and marine algae, upon which green sea turtles rely (NMFS & USFWS, 2007b); (Lee Long, Coles, & McKenzie, 2000). Seagrass habitats are possibly the most susceptible of all coastal marine habitats because these areas, often defined as sheltered coasts with good water quality, are frequently at the downstream end of drainages from human development (Waycott, Longstaff, & Mellors, 2005). Nutrient over-enrichment caused by nitrogen and phosphorous from urban and agricultural run-off can cause excess algal growth, which in turn can smother seagrasses and lower the oxygen content of water (63 FR 46693, 1998).

Another real threat to green sea turtles includes disease, particularly fibropapillomatosis. Although the specific cause(s) of this disease remains unknown, it causes small internal and external tumors (fibropapillomas) on the soft portion of a turtle's body (NMFS & USFWS, 2007b). Fibropapilloma tumors can impair green turtles' ability to forage, breath, swim and this could potentially lead to death (George, 1997). This disease was referenced in the Recovery Plan for the U.S. Population of Atlantic Green Turtle as a threat, particularly for immature green turtles (NMFS & USFWS, 1991). Also consistent with the risks stated above, the recovery plan for the U.S. Atlantic population indicated that significant threats were coastal development, commercial fisheries and pollution (NMFS & USFWS, 1991).

EPA has determined that remediation activity discharges will have no effect on the green sea turtle because the distance between the localized, on-shore remediation activities and minor, near-shore remediation activity discharges relative to the size of, and the high energy and volume in the marine waters this species is likely to inhabit, precludes contact between remediation activity discharges and this species, presently or in the future. As such EPA will not consider this species further in this analysis.

4. Effects Determination

The environmental baseline, including water quality standards, numeric and non-numeric effluent limitations and the high dilution considered in EPA's effects determination for remediation activity discharges, is provided in Part a of this effects determination. As previously described, EPA has determined that remediation activity discharges will have no effect on north Atlantic right whale, fin whale, Kemp's Ridley sea turtle, loggerhead sea turtle, leatherback sea turtle, and green sea turtle, and as such, these species are not discussed further in this analysis. EPA has determined that remediation activity discharges may affect, but any effects will be insignificant and/or discountable on the Atlantic sturgeon and shortnose sturgeon. Consequently, EPA has concluded that the proposed action is not likely to adversely affect listed species. The potential effects on the listed species including support for EPA's determination that such

potential effects are insignificant and/or discountable are provided in Part b of this effects determination. Furthermore, EPA has determined that remediation activity discharges will have no effect on designated critical habitat for north Atlantic right whale, and as such, this critical habitat is not discussed further in this analysis. EPA has determined that remediation activity discharges are not likely to adversely affect proposed critical habitat for Atlantic sturgeon. The potential effects on proposed critical habitat including support for EPA's determination that any potential effects will be prevented or minimized such that remediation activity discharges are not likely to adversely affect proposed critical habitat are provided in Part c of this effects determination. Further, EPA has provided reference to the determinations made for indirect effects in Part d, and for interdependent and related actions in Part e of this effects determination.

a. Environmental Baseline

i. Massachusetts Waterbodies and Surface Water Quality Standards

Section 305(b) of the Federal Clean Water Act codifies the process in which waters are evaluated with respect to their capacity to support designated uses as defined in the Surface Water Quality Standards (MassDEP, 2006). The Massachusetts Surface Water Quality Standards (SWQS) define the goals for water quality in the Commonwealth of Massachusetts.

Class A waters are designated as a source of public water supply. Both Class A and Class SA (for coastal and marine waters) provide excellent habitat for fish, other aquatic life and wildlife, including for their reproduction, migration, growth and other critical functions, and for primary and second contact recreation, irrespective of whether or not such activities are allowed (MassDEP, 2006). Although the draft RGP includes certain effluent limitations applicable to Class A and SA waterbodies, unless authorized on a case-by-case basis by MassDEP, discharges to Class A and SA waters are excluded from the RGP.

Class B and Class SB waters are designated as a habitat for fish, other aquatic life, and wildlife, including for their reproduction, migration, growth and other crucial functions, and for primary and secondary contact recreation (MassDEP, 2006). The SWQS define a warm water fishery as a waterbody in which the maximum mean monthly temperature generally exceeds 68° F (20° C) during the summer months and which is not capable of sustaining a year-round population of cold water aquatic life (MassDEP, 2006). Unless authorized on a case-by-case basis by MassDEP, discharges to Class SB waters are authorized with no allowable dilution.

The Class B waterbodies in Massachusetts within the Action Area where listed species are expected to occur include the Connecticut River, the Merrimack River and the Taunton River. The Class SB waterbodies in Massachusetts within the Action Area where listed species are expected to occur include the marine shoreline areas, including Cape Cod Bay and Massachusetts Bay.

Table 1 below, summarizes the parameters for select MA SWQS. Massachusetts provides narrative water quality standards for solids (in accordance with 314 CMR 4.05(3)(b)5, and 4.05(4)(b)5). The temperature and pH limits for the applicable surface water quality standards are in accordance with 314 CMR 4.05(3)(b)2, and 4.05(4)(b)2, and 314 CMR 4.05(3)(b)3, and

4.05(4)(b)3, respectively. The Commonwealth of Massachusetts' surface water-quality standards require the use of federal water-quality criteria where a specific (toxic) pollutant could reasonably be expected to adversely affect existing or designated uses (314 CMR 4.05(5)(e)). Parts 2.1 and 2.3 of the draft RGP provides the actual effluent limitations for sites in Massachusetts, which incorporates numeric water quality standards for Massachusetts.

Table 1: Summary of Massachusetts Surface Water Quality Standards Relative to Parameter Effluent Limitations: Class B, and Class SB

Parameter	Class B	Class SB
Inorganics (solids, toxic pollutants)	314 CMR 4.05(3)(b)5. "Solids. These waters shall be free from floating, suspended and settleable solids in concentrations or combinations that would impair any use assigned to this Class, that would cause aesthetically objectionable conditions, or that would impair the benthic biota or degrade the chemical composition of the bottom."	314 CMR 4.05(4)(b)5.
	314 CMR 4.05(5)(e) "Toxic Pollutants. All surface waters shall be free from pollutants in concentrations or combinations that are toxic to humans, aquatic life or wildlife."	
Non-Halogenated Volatile Organic Compounds (petrochemicals, toxic pollutants)	314 CMR 4.05(3)(b)7. "Oil and Grease. These waters shall be free from oil, grease and petrochemicals that produce a visible film on the surface of the water, impart an oily taste to the water or an oily or other undesirable taste to the edible portions of aquatic life, coat the banks or bottom of the water course, or are deleterious or become toxic to aquatic life." 314 CMR 4.05(3)(b)8. "Taste and Odor. None in such concentrations or combinations that are aesthetically objectionable, that would impair any use assigned to this Class, or that would cause tainting or undesirable flavors	314 CMR 4.05(4)(b)7. 314 CMR 4.05(4)(b)8.

	in the edible portions of aquatic life.”	
	314 CMR 4.05(5)(e)	
Halogenated Volatile Organic Compounds (toxic pollutants)	314 CMR 4.05(5)(e)	
Non-Halogenated Semi-Volatile Organic Compounds (petrochemicals, toxic pollutants)	314 CMR 4.05(3)(b)7.	314 CMR 4.05(4)(b)7.
	314 CMR 4.05(5)(e)	
Halogenated Semi-Volatile Organic Compounds (toxic pollutants)	314 CMR 4.05(5)(e)	
Fuels Parameters (petrochemicals, toxic pollutants)	314 CMR 4.05(3)(b)7. 314 CMR 4.05(3)(b)8.	314 CMR 4.05(4)(b)7. 314 CMR 4.05(4)(b)8.
	314 CMR 4.05(5)(e)	
pH	314 CMR 4.05(3)(b)3. “pH. Shall be in the range of 6.5 through 8.3 standard units and not more than 0.5 units outside of the natural background range”	314 CMR 4.05(4)(b)3. “pH. Shall be in the range of 6.5 through 8.5 standard units and not more than 0.2 units outside of the natural background range.”
Temperature	314 CMR 4.05(3)(b)2. “Temperature. a. Shall not exceed 68°F (20°C) based on the mean of the daily maximum temperature over a seven day period in cold water fisheries, unless naturally occurring. Where a reproducing cold water aquatic community exists at a naturally occurring higher temperature, the temperature necessary to protect the community shall not be exceeded and the natural daily and seasonal temperature fluctuations necessary to protect the community shall be maintained. Temperature shall not exceed 83°F (28.3°C) in warm water fisheries. The rise	314 CMR 4.05(4)(b)2. “Temperature. a. Shall not exceed 85°F (29.4°C) nor a maximum daily mean of 80°0F (26.7°C), and the rise in temperature due to a discharge shall not exceed 1.5°F (0.8°C) during the summer months (July through September) nor 4°0F (2.2°C) during the winter months (October through June).”

	in temperature due to a discharge shall not exceed 3°F (1.7°C) in rivers and streams designated as cold water fisheries nor 5°F (2.8°C) in rivers and streams designated as warm water fisheries (based on the minimum expected flow for the month); in lakes and ponds the rise shall not exceed 3°F (1.7°C) in the epilimnion (based on the monthly average of maximum daily temperature)."	

MA SWQSs also include turbidity, dissolved oxygen and several narrative standards necessary to protect aquatic life. Part 2.2 of the draft RGP provides the actual non-numeric effluent limitations for sites in Massachusetts, which incorporates the narrative water quality standards. Part 2.4 provides additional State Permit Conditions, which Massachusetts determined were necessary to meet their SWQSs. Part 2.5 of the draft RGP provides additional non-numeric effluent limitations for sites in Massachusetts, including required best management practices (BMPs) and special conditions, which include discharge prohibitions.

NMFS has noted principal causes of aquatic impairments in Massachusetts.[9] **Table 2** below, summarizes the principal causes.

Table 2: Summary of Principal Causes of Aquatic Impairments in Massachusetts (Reporting Year 2012)

Aquatic Habitat	Principal Causes of Impairment	% of assessed waters
Rivers and Streams	Fecal coliform	23
	Escherichia coli	19
	PCBs in fish tissue	14
	Phosphorus, total	13
	Dissolved Oxygen	13
Lakes, Reservoirs and Ponds	Non-Native Aquatic Plants	48
	Mercury in Fish Tissue	14
	Eurasian Water Milfoil	14
	Dissolved Oxygen	9
	Excess Algal Growth	8
Bays and Estuaries	Fecal coliform	100
	PCBs in fish tissue	36
	Other Cause	27

[9] *National Marine Fisheries Service Endangered Species Act Section 7 Consultation Biological And Conference Opinion* for EPA's Multi-Sector General Permit for Stormwater Associated with Industrial Activity Pursuant to the National Pollutant Discharge Elimination System; Table 10; March 19, 2015.

	Estuarine Bioassessments Nitrogen, total	17 16

Of the causes of impairments listed above, remediation activity discharges are not likely to contain parameters which are a principal cause of bacteria, nuisance aquatic plants, or pollutants present because of atmospheric deposition. While remediation activity discharges are not likely to contain nutrients, EPA will assess nutrients using the parameter ammonia nitrogen, to determine if such discharges are a source of nutrients. Remediation activity discharges are expected to have insignificant effects on dissolved oxygen levels, which is described further in EPA's effects analysis. Finally, while some legacy contaminants are potentially present in remediation activity discharges (e.g., PCBs), those present are subject to effluent limitations which are as stringent as or more stringent than aquatic life criteria for such contaminants. EPA will continue to assess the additive, cumulative and/or synergistic effects of legacy contaminants and dissolved oxygen impairments through toxicity testing, or, if necessary, additional monitoring requirements.

ii. New Hampshire Waterbodies and Water Quality Regulations

The New Hampshire Surface Water Quality Regulations (Env-Wq 1700) define the goals for water quality in the state of New Hampshire. According to the New Hampshire Statute (Chapter 485-A.8) regarding the classification of waters, there are 2 classes of surface waters for the state: Class A and Class B waters.

Class A waters in New Hampshire shall be of the highest quality, and there shall be no discharge of any sewage or wastes into waters of this classification. Class A waters are a potentially acceptable water supply after adequate treatment. The State of New Hampshire does not allow discharges to Class A waters under the RGP.

Class B waters shall be of the second highest quality and shall have no objectionable physical characteristic. These waters are considered acceptable for fishing, swimming, and other recreational purposes, and, after adequate treatment, for use as water supplies.

The Class B waterbodies in New Hampshire within the Action Area where listed species are expected to occur include the Piscataqua River and the marine shoreline areas, including Great Bay.

Table 3 below, summarizes the parameters for select NH WQRs. New Hampshire provides narrative water quality standards for solids (covered under General Water Quality Criteria Env-Wq 1703.03). Env-Wq 1703.13 and 1703.18 sets the applicable surface water quality standards in New Hampshire for temperature and pH, respectively, while Env-Wq 1703.21 sets the water quality criteria for toxic substances, which includes the inorganic parameters, non-halogenated volatile organic compounds (VOCs), halogenated VOCs, non-halogenated semi-volatile organic compounds (SVOCs), and halogenated SVOCs included in the RGP.

Table 3: Summary of New Hampshire Surface Water Quality Regulations Relative to Parameter Effluent Limitations: Class B

Parameter	Class B
Inorganics (solids, toxic pollutants)	Env-Wq 1703.03 "General Water Quality Criteria. (c) The following physical, chemical and biological criteria shall apply to all surface waters: (1) All surface waters shall be free from substances in kind or quantity which: a. Settle to form harmful deposits; b. Float as foam, debris, scum or other visible substances; c. Produce odor, color, taste or turbidity which is not naturally occurring and would render it unsuitable for its designated uses; d. Result in the dominance of nuisance species… (3) Tainting substances shall not be present in concentrations that individually or in combination are detectable by taste and odor tests performed on the edible portions of aquatic organisms."
	Env-Wq 1703.21 "Water Quality Criteria for Toxic Substances. (a) Unless naturally occurring or allowed under part Env-Wq 1707, all surface waters shall be free from toxic substances or chemical constituents in concentrations or combinations that: (1) Injure or are inimical to plants, animals, humans or aquatic life; or (2) Persist in the environment or accumulate in aquatic organisms to levels that result in harmful concentrations in edible portions of fish, shellfish, other aquatic life, or wildlife which might consume aquatic life. (b) Unless allowed in part Env-Wq 1707 or naturally occurring, concentrations of toxic substances in all surface waters shall not exceed the recommended safe exposure levels of the most sensitive surface water use shown in Table 1703.1, subject to the notes as explained in Env-Wq 1703.22…"
Non-Halogenated Volatile Organic Compounds (toxic pollutants)	Env-Wq 1703.21
Halogenated Volatile Organic Compounds (toxic pollutants)	Env-Wq 1703.21
Non-Halogenated	Env-Wq 1703.21

Semi-Volatile Organic Compounds (toxic pollutants)	
Halogenated Semi-Volatile Organic Compounds (toxic pollutants)	Env-Wq 1703.21
Fuels Parameters (toxic pollutants)	Env-Wq 1703.21
pH	Env-Wq 1703.18 (b) The pH of Class B waters shall be 6.5 to 8.0, unless due to natural causes.
Temperature	Env-Wq 1703.13 (b) Temperature in class B waters shall be in accordance with RSA 485-A:8, II, and VIII. RSA 485-A:8 "VIII. In prescribing minimum treatment provisions for thermal wastes discharged to interstate waters, the department shall adhere to the water quality requirements and recommendations of the New Hampshire Fish and Game Department, the New England Interstate Water Pollution Control Commission, or the United States Environmental Protection Agency, whichever requirements and recommendations provide the most effective level of thermal pollution control."

NH SWQRs also include turbidity, dissolved oxygen and several narrative standards necessary to protect aquatic life. Part 2.2 of the draft RGP provides the actual non-numeric effluent limitations for sites in New Hampshire, which incorporates the narrative water quality standards. Part 2.4 provides additional State Permit Conditions, which New Hampshire determined were necessary to meet their SWQRs. Part 2.5 of the draft RGP provides additional non-numeric effluent limitations for sites in Massachusetts, including required best management practices (BMPs) and special conditions, which include discharge prohibitions.

NMFS has noted principal causes of aquatic impairments in New Hampshire.[10] **Table 4** below, summarizes the principal causes.

Table 4: Summary of Principal Causes of Aquatic Impairments in New Hampshire (Reporting Year 2010)

Aquatic Habitat	Principal Causes of Impairment	% of assessed waters
Rivers and Streams	Mercury pH Escherichia coli	100 20 7

[10] See footnote 9, above.

	Dissolved Oxygen Dissolved Oxygen Saturation	4 3
Lakes, Reservoirs and Ponds	Mercury pH Non-Native Aquatic Plants Dissolved Oxygen Saturation Dissolved Oxygen	100 25 0.4 0.4 0.3
Bays and Estuaries	Mercury Dioxin PCBs Impaired Estuarine Biological Assemblages Nitrogen, total	100 36 27 17 16

Of the causes of impairments listed above, remediation activity discharges are not likely to contain parameters which are a principal cause of bacteria, nuisance aquatic plants, or pollutants present because of atmospheric deposition. While remediation activity discharges are not likely to contain nutrients, EPA will assess nutrients using the parameter ammonia nitrogen, to determine if such discharges are a source of nutrients. Remediation activity discharges are expected to have insignificant effects on dissolved oxygen levels, which is described further in EPA's effects analysis. The draft RGP does not authorize the discharge of dioxin. Finally, while some legacy contaminants are *potentially* present in remediation activity discharges (e.g., PCBs), those present are subject to effluent limitations which are as stringent as or more stringent than aquatic life criteria for such contaminants. EPA will continue to assess the additive, cumulative and/or synergistic effects of legacy contaminants and dissolved oxygen impairments through toxicity testing, or, if necessary, additional monitoring requirements.

iii. Numeric Effluent Limitations for Remediation Activity Discharges

EPA reviews every NOI that is submitted requesting coverage under the general permit and coverage is not automatic. If EPA does not believe the effluent limitations and requirements of the general permit will ensure that the remediation activity discharge will either have no effect or any effects will be insignificant or discountable for listed species, EPA will not authorize the discharge under this general permit. This section describes the numeric effluent limitations that will be imposed on a remediation activity discharge when EPA determines a discharge is eligible for authorization under this general permit.

The RGP is intended for minor discharges. That is, discharges authorized under this general permit are expected to occur with low frequency (intermittent), small magnitude (low volume limited to no more than 1.0 MGD, typically approximately 0.0072 MGD to 0.072 MGD), and short duration (temporary or short-term, typically from 24 hours up to 12 months in duration). As a result, EPA expects that remediation activity discharges will either have no effect or are not likely to adversely affect listed species or their critical habitat, because effects will be insignificant (so small they cannot be detected) or discountable (extremely unlikely to occur). Because a limited amount of specific information is available regarding the effects of the expected stressors on listed species (described in Part b of this section, below) and the location of

future remediation activity discharges, EPA will impose numeric effluent limitations which ensure water quality standards will be met at the point of discharge and that any exposure to the discharge prior to full dilution would be extremely unlikely to occur or would have insignificant and/or discountable effects for all discharges. Although EPA may consider State-approved dilution when calculating effluent limitations, the permit does *not* allow mixing zones. Therefore, the numeric effluent limitations are "end-of-pipe" effluent limitations. Applicants are required to certify that the "end-of-pipe" effluent limitations will be met, as part of submitting their NOI to request coverage under the general permit. Therefore, effluent limitations will ensure the protection of aquatic life, including listed species.

The draft RGP uses both technology-based effluent limits (TBELs) as well as water-quality based effluent limits (WQBELs) for any parameter for which the TBEL may not meet numeric or narrative water quality standards at zero dilution. For the majority of parameters included in the RGP, the TBELs are as stringent as or more stringent than applicable water quality criteria. Where EPA determines that effluent limitations more stringent than TBELs are necessary to attain or maintain State or Federal water quality standards (WQSs) for the protection of both aquatic life and human health, EPA will impose WQBELs, as required. Therefore, the numeric effluent limitations are sufficiently stringent to ensure that State WQSs for the protection of both aquatic life and human health are met. §301(b)(1)(C) of the CWA. Further, the effluent limitations established in the draft RGP will ensure protection of aquatic life and maintenance of the receiving water as an aquatic habitat. EPA and/or the States may impose a more stringent effluent limitation on a case-by-case basis, when appropriate, such as when a more stringent effluent limitation is necessary to protect listed species or their critical habitat. Alternatively, EPA may require an individual permit. EPA believes that this approach further protects the aforementioned listed species and their critical habitat.

Where a remediation activity discharge is subject to WQBELs in the 2016 RGP, EPA will impose numeric effluent limitations based on aquatic life criteria, such as EPA's *National Recommended Water Quality Criteria* (NRWQC) for the protection of aquatic life. Where the effluent limitation for a parameter included in the draft RGP is a WQBEL based on WQC for the protection of aquatic life, EPA considers both acute and chronic WQC. Where aquatic life criteria are not available, EPA considers numeric effluent limitations based on human health or similar risk-based criteria, which, in the 2016 RGP, are generally based on EPA's NRWQC for the protection of human health for the consumption of organisms-only, drinking water standards such as maximum contaminant levels (MCLs) under the Safe Drinking Water Act, and State-adopted groundwater quality standards that apply conservative assumptions to derive risk-based cleanup levels. As a result, WQBELs ensure discharges meet WQSs established under Section 303 of the CWA.

Numeric effluent limitations for remediation activity discharges that are equivalent to human health- and risk-based water quality criteria such as EPA's drinking water standards, and State-adopted groundwater quality standards are imposed near or below analytical minimum levels of detection. Therefore, while human health and/or risk-based effluent limitations are not specifically derived for the protection of aquatic life, such limitations are an appropriate proxy because any potential effects to aquatic life at concentrations that could potentially occur near or below analytical levels of detection cannot be meaningfully measured, detected or evaluated.

The draft RGP authorizes discharges that *may* contain a variety of conventional, non-conventional and toxic pollutants. These discharges are subject to effluent limitations or monitoring requirements for effluent flow, pH, temperature, inorganic parameters (ammonia, chloride, total residual chlorine (TRC), total suspended solids (TSS), 14 metals, cyanide), five types of non-halogenated volatile organic compounds, 16 halogenated volatile organic compounds, 12 non-halogenated semi-volatile organic compounds, two halogenated semi-volatile organic compounds and five types of fuels parameters for discharges from sites based on the type of remediation activity and the receiving water of the discharge. Table 3 presents a complete list of the parameters covered under the RGP and the effluent limitations or monitoring requirements established. Any single discharge authorized under the RGP is likely to contain only a small combination of these potentially present pollutants.

Table 3: Effluent Limitations and Monitor-Only Requirements Included in the Draft RGP

Parameter	Effluent Limitation[1,2]	
	TBEL[3]	WQBEL[4]
a. Inorganics		
Ammonia	Report mg/L	
Chloride	Report µg/L	
Total Residual Chlorine	0.2 mg/L	FW= 11 µg/L SW= 7.5 µg/L
Total Suspended Solids	30 mg/L	
Antimony	206 µg/L	640 µg/L
Arsenic	104 µg/L	FW= 10 µg/L SW= 36 µg/L
Cadmium	10.2 µg/L	FW= 0.25 µg/L SW= 8.8 µg/L in MA SW= 9.3 µg/L in NH
Chromium III	323 µg/L	FW= 74 µg/L SW= 100 µg/L
Chromium VI	323 µg/L	FW= 11 µg/L SW= 50 µg/L
Copper	242 µg/L	FW= 9 µg/L SW= 3.1 µg/L
Iron	5,000 µg/L	FW = 1,000 µg/L
Lead	160 µg/L	FW= 2.5 µg/L SW= 8.1 µg/L
Mercury	0.739 µg/L	FW= 0.77 µg/L SW= 0.94 µg/L
Nickel	1,450 µg/L	FW= 52 µg/L SW= 8.2 µg/L
Selenium	235.8 µg/L	FW= 5.0 µg/L SW= 71 µg/L
Silver	35.1 µg/L	FW= 3.2 µg/L SW= 1.9 µg/L
Zinc	420 µg/L	FW= 120 µg/L

Parameter	Effluent Limitation[1,2]	
		SW= 81 µg/L
Cyanide	178 mg/L	FW = 5.2 µg/L SW = 1.0 µg/L
b. Non-Halogenated Volatile Organic Compounds		
Total BETX	100 µg/L	
Benzene	5.0 µg/L	
1,4 Dioxane	200 µg/L	
Acetone	7.97 mg/L	
Phenol	1,080 µg/L	300 µg/L
c. Halogenated Volatile Organic Compounds		
Carbon Tetrachloride	4.4 µg/L	1.6 µg/L in MA
1,2 Dichlorobenzene	600 µg/L	
1,3 Dichlorobenzene	320 µg/L	
1,4 Dichlorobenzene	5.0 µg/L	
Total dichlorobenzene	763 µg/L in NH	
1,1 Dichloroethane	70 µg/L	
1,2 Dichloroethane	5.0 µg/L	
1,1 Dichloroethylene	3.2 µg/L	
Ethylene Dibromide[17]	0.05 µg/L	
Methylene Chloride	4.6 µg/L	
1,1,1 Trichloroethane	200 µg/L	
1,1,2 Trichloroethane	5.0 µg/L	
Trichloroethylene	5.0 µg/L	
Tetrachloroethylene	5.0 µg/L	3.3 µg/L in MA
cis-1,2 Dichloroethylene	70 µg/L	
Vinyl Chloride	2.0 µg/L	
d. Non-Halogenated Semi-Volatile Organic Compounds		
Total Phthalates	190 µg/L	FW = 3.0 µg/L in NH SW = 3.4 µg/L in NH
Diethylhexyl phthalate	101 µg/L	2.2 µg/L
Total Group I Polycyclic Aromatic Hydrocarbons	1.0 µg/L	As Individual PAHs
Benzo(a)anthracene	As Total Group I PAHs	0.0038 µg/L
Benzo(a)pyrene		0.0038 µg/L
Benzo(b)fluoranthene		0.0038 µg/L
Benzo(k)fluoranthene		0.0038 µg/L
Chrysene		0.0038 µg/L
Dibenzo(a,h)anthracene		0.0038 µg/L
Indeno(1,2,3-cd)pyrene		0.0038 µg/L

Parameter	Effluent Limitation[1,2]	
Total Group II Polycyclic Aromatic Hydrocarbons	100 µg/L	
Naphthalene	20 µg/L	
e. Halogenated Semi-Volatile Organic Compounds		
Total Polychlorinated Biphenyls	0.000064 µg/L	
Pentachlorophenol	1.0 µg/L	
f. Fuels Parameters		
Total Petroleum Hydrocarbons	5.0 mg/L	
Ethanol	Report mg/L	
Methyl-tert-Butyl Ether	70 µg/L	20 µg/L in MA
tert-Butyl Alcohol	120 µg/L in MA 40 µg/L in NH	
tert-Amyl Methyl Ether	90 µg/L in MA 140 µg/L in NH	
Effluent Flow	Not to exceed 1.0 MGD	
pH	Class B: 6.5 to 8.3 in MA 6.5 to 8.0 in NH Class SB: 6.5 to 8.5 in MA	
Temperature	Class B: ≤68°F for cold water fishery ≤83°F for warm water fishery Class SB: ≤85 °F in MA	

Table 3 Footnotes:

[1] The following abbreviations are used in Table 2, above:
- [a] TBEL = technology-based effluent limitation
- [b] WQBEL = water quality-based effluent limitation
- [c] mg/L = milligrams per liter
- [d] avg = average
- [e] µg/L = micrograms per liter
- [f] FW = freshwater
- [g] SW = saltwater

[2] The effluent limitation and/or monitor-only requirement for any parameter listed applies to any site when the given parameter is present in discharges from that site. The effluent limitations and monitor-only requirements for certain parameters also apply to the different types of sites covered under the RGP, regardless if a parameter has been measured in discharges.

[3] For any parameter with a single effluent limitation, that effluent limitation applies to a site if that parameter is applicable to that site. For any parameter with both a TBEL and a WQBEL, the TBEL applies to a site, at a minimum, if that parameter is applicable to that site.

[4] For any parameter with both a TBEL and a WQBEL, the WQBEL applies to a site if: 1) the applicant determines the WQBEL for a parameter calculated in accordance with Appendix V or VI applies; or 2) EPA or the appropriate State determines that a WQBEL is necessary to meet State WQSs. The calculation of WQBELs shall be as follows: 1) A dilution factor may be used to calculate the WQBEL for a parameter, if allowable and approved by the appropriate State prior to the submission of the Notice of Intent to EPA; 2) The calculations are completed in accordance with the instructions provided in Appendix V for sites located in Massachusetts or Appendix VI for sites located in New Hampshire; 3) The WQBEL calculations are included in the Notice of Intent submitted to EPA; and 4) The calculated WQBEL is confirmed by EPA in writing. In the event of a calculation error, the operator will be informed of any corrected WQBEL when notified of permit coverage by EPA. EPA anticipates providing additional resources to assist applicants in following the calculation methodologies for effluent limitations in Appendix V for sites in Massachusetts and Appendix VI for sites in New Hampshire.

Applicants are required to include the calculated WQBELs that apply to their discharge in the NOI submitted to EPA, which EPA will confirm, or revise, if necessary. While metals limitations are generally included on the basis of dissolved metal in the water column and at an assumed hardness when a metal WQBEL is hardness-dependent, applicants must calculate the WQBELs that apply to their discharges using site-specific data, including site-specific hardness in accordance with State water quality standards, and receiving water concentrations of persistent pollutants (e.g., metals). Following the calculation methodology provided in the 2016 RGP, a WQBEL is adjusted for: 1) effluent and receiving water flow (i.e., the ratio of which is used to derive a dilution factor); 2) actual effluent and receiving water hardness (i.e., if a parameter is hardness-dependent); and 3) existing concentrations of these parameters in the receiving water (if appropriate). These conditions affect the allowable instream concentrations of the limited parameters and ensure any cumulative effects are considered.

Again, EPA carefully reviews each NOI submitted for coverage under the RGP. If any concerns are raised as a result of the site-specific information included in the NOI, EPA may request additional information from the operator, require an individual NPDES permit, or deny permit coverage. If applicable, EPA will also consult with the appropriate federal agency to determine how best to proceed. If a concern is specifically related to a listed species or the proposed/designated critical habitat of such species identified in this assessment, EPA will contact NMFS.

iv. Non-Numeric Effluent Limitations and Other Special Conditions for Remediation Activity Discharges

(1) Limitations on Coverage Which Pertain to Listed Species or their Critical Habitat

The draft RGP specifically excludes coverage under the RGP for discharge(s) that are likely to jeopardize the continued existence of listed threatened or endangered species or the critical habitat of such species. EPA also excludes coverage under the RGP for discharges to certain waters that results in additional protection for listed species and their critical habitat. The following discharges, which both directly and indirectly provide protection for listed species and their designated/proposed critical habitat, are expressly excluded from coverage under the RGP:

1) Discharges to Outstanding Resource Waters in Massachusetts and New Hampshire:
2) Discharges to Class A waters in New Hampshire, in accordance with RSA 485A:8, I and Env-Wq 1708.06.
3) Discharges that are likely to adversely affect any species listed as endangered or threatened under the Endangered Species Act (ESA) or result in the adverse modification or destruction of habitat that is designated as critical under ESA.
4) Discharges whose direct or indirect impacts do not prevent or minimize adverse effects on any designated Essential Fish Habitat (EFH).
5) Discharges to Ocean Sanctuaries in Massachusetts, as defined at 302 CMR 5.00.
6) Discharges to a river designated as a Wild and Scenic River, except in accordance with 16 U.S.C. 1271 *et seq*.

(2) Special Eligibility Determinations Which Pertain to Listed Species or their Critical Habitat

Discharges that are likely to adversely affect any species listed as endangered or threatened under the Endangered Species Act (ESA), including prohibited incidental take, are not eligible for coverage under this general permit. EPA reviews every NOI that is submitted requesting coverage under the general permit, makes a determination of coverage, and issues an authorization to discharge to each operator in writing. Every NOI received under the RGP is posted for a minimum of seven (7) days on EPA's RGP website to provide for public comment. EPA reviews the information related to endangered species under the jurisdiction of NMFS required in the NOI (i.e., if the discharge is located in the Connecticut, Merrimack, Piscataqua or Taunton Rivers, if the discharge is to saltwater, and whether there has been previous formal or informal consultation with NMFS) to determine eligibility under the general permit. If EPA determines that a discharge is likely to adversely affect any listed species, or may result in the take of a listed species, EPA will not authorize the discharge under this general permit, unless take has been authorized under the ESA of 1973, as amended, through a separate permit pursuant to ESA section 10(a)(1)(A) or ESA section 10(a)(1)(B), or take is exempted through an Incidental Take Statement included in an Opinion from the NMFS for that site.

Further, sites that are located in areas in which listed endangered or threatened species may be present are not automatically covered under this general permit. Sites located in Massachusetts and New Hampshire that are seeking coverage under this general permit must certify compliance with the requirements of this permit related to threatened and endangered species and critical habitat under the Endangered Species Act. The special eligibility determinations that apply to all applicants are included in Part 1.4 and Appendices I and IV of the draft RGP. Every applicant must certify that the proposed discharge to be covered is eligible for coverage under the general permit, including certification of ESA eligibility, in the NOI submitted to EPA. All applicants must respond to all questions pertaining to ESA included in the suggested NOI format (see Appendix IV, Part I of the draft RGP). Applicants who cannot certify compliance with the ESA requirements or the eligibility requirements of the general permit must contact EPA to determine if eligibility for an individual NPDES permit is possible or to discuss other possible options for the proposed discharge. EPA may also require individual permits be issued if actual environmental conditions (including the preservation of endangered species) are not adequately addressed by this general permit.

EPA will consult with NMFS for new discharges when necessary to ensure that the listed species and critical habitat under their jurisdiction are not adversely affected by the proposed discharge to ensure that the terms of the RGP adequately support a finding that the discharge has no effect or is not likely to adversely affect listed species in the action area, will prevent the take of listed species, and will have no effect or will prevent or minimize adverse effects on designated/proposed critical habitat due to remediation activity discharges.

Sites seeking coverage under this general permit also have an independent ESA obligation to ensure that their discharges do not result in any prohibited "take" of listed species. Appendix I of the draft RGP requires sites located in an area where endangered and threatened species are present and incidental take is possible to obtain an ESA section 10 permit (Incidental Take Permit) or complete formal consultation under ESA section 7 prior to submitting a NOI for RGP coverage. Applicants that are unsure whether to pursue a section 10 permit or a section 7 consultation for takings protection are instructed to confer with the NMFS. Therefore, take of a threatened or endangered species resulting from discharges or discharge-related activities under the RGP is only authorized when: 1) Take has been authorized under the ESA of 1973, as amended, through a separate permit pursuant to ESA section 10(a)(1)(A) or ESA section and 10(a)(1)(B); or 2) Take is exempted through an Incidental Take Statement included in an Opinion for a specific RGP site.

(3) Special Conditions for Remediation Activity Discharges

In addition to the non-numeric and numeric effluent limitations aforementioned, the draft RGP also contains several special conditions. First, the RGP retains requirements for permittees to develop, implement, and maintain a Best Management Practices (BMP) Plan and to document how both the non-numeric technology-based and numeric effluent limitations are being met through the selection, design, installation, and implementation of control measures (including BMPs).

Second, the RGP requires specific BMPs of all permittees, including a requirement that operators utilize pollution control technologies (i.e., treatment systems) if the end of pipe effluent limitations will not be met. The specific BMPs of all permittees are as follows:

1) An Effluent Flow BMP that requires flow control measures be used to prevent discharge(s) in exceedance of the design flow of the discharge (i.e., the maximum flow through the component with the lowest limiting capacity).
2) A Preventative Maintenance BMP that requires documented procedures and protocols, a maintenance schedule and records of completion to ensure all control measures, including all treatment system components and related appurtenances used to achieve the limitations in the general permit remain in effective operating condition and do not result in leaks, spills, and other releases of pollutants.
3) A Site Management BMP that requires control measures and management practices to ensure proper management of solid and hazardous waste, minimize run-on and runoff and prevent any erosion, stream scouring, or sedimentation caused directly or indirectly by the discharge and/or which contributes additional pollutants.
4) A Pollutant Minimization BMP that requires identification and assessment of the type and quantity of pollutants, a description of control measures used to ensure dilution is not used as a means to achieve permit effluent limitations and selection, design, installation

and proper operation and maintenance of pollution control technologies, when necessary to achieve the limitations and requirements in this general permit.

5) An Administrative Controls BMP that requires documentation, procedures, schedule and/or records of site security, employee training, corrective action and routine inspections.
6) A Quality Assurance/Quality Control (QA/QC) BMP that requires a description of monitoring requirements, sampling locations, test method and minimum level requirements, data validation and reporting requirements and a schedule for review of monitoring results.
7) Materials Management BMP that requires practices and/or control measures pertaining to good housekeeping, material compatibility, chemical and additive use, and leaks, spills, or other release containing a hazardous substance or oil.

Third, The RGP retains restrictions on discharges of chemicals and additives that are commonly used during remediation activities or for treatment directly that could be present in discharges. The purpose of this requirement is to prevent or minimize the concentration of pollutants (biological, chemical and physical) in the wastewater discharged to surface waters. Both Massachusetts and New Hampshire have narrative criteria in their water quality regulations (Massachusetts 314 CMR 4.05(5)(e) and New Hampshire Part Env-Wq 1703.21) that prohibit toxic discharges in toxic amounts. Excepting chemicals and additives authorized on a case-by-case basis, the draft RGP prohibits the addition of toxic materials (e.g., chemicals and additives) to the discharges and prohibits the discharge of pollutants in amounts that would be toxic to aquatic life.

Finally, the draft RGP requires additional conditions, including increased monitoring requirements such as process-specific monitoring (i.e., hydrostatic testing of pipelines and tanks), Whole Effluent Toxicity testing (i.e., remediation site discharges) and ambient monitoring (varies by the type of site and the type of contamination present). EPA does not currently have information regarding the toxicity of remediation activity discharges. In addition, acute effects data are not readily available for many of the parameters included in the draft RGP. In order to determine the extent of this *potential* pollutant at the sites covered under this general permit, EPA is requiring all remediation sites conduct acute Whole Effluent Toxicity testing and provide the results with the NOI submitted to EPA. Acute toxicity data are based on a duration of exposure most similar to the short duration of remediation activity discharges (temporary and short-term). EPA will review the WET testing results to ensure that such sites are not likely to have an adverse impact on living organisms, such as shortnose or Atlantic sturgeon. Collectively, the WET testing data from all RGP sites will inform EPA as to whether routine WET monitoring (or a limit) for toxicity is necessary. In addition, WET testing may be required as a condition of authorization on a case-by-case basis, if necessary to meet water quality standards.

(4) Dilution Estimates

To be conservative with the dilution estimate during remediation activity discharges, EPA chose the following flow rates for its effects determination for the riverine waterbodies in the Action Area:

1) Connecticut River at USGS gauging station 01172010: the lowest daily average flow rate over the period of record (10 years) was 5,270 CFS (3,400 million gallons per day (MGD)).
2) Merrimack River at USGS gauging station 01100000: the lowest daily average flow rate over the period of record (90 years) was 2,500 CFS (1,616 MGD).
3) Piscataqua River via Cocheco River at USGS gauging station 01072800: the median daily average flow rate over the period of record (19 years) was 101 CFS (65 MGD).
4) Taunton River at USGS gauging station 01108000: the lowest daily average flow rate over the period of record (66 years) was 74 CFS (48 MGD).

Discharges eligible for coverage under this general permit are considered minor. That is, discharges authorized under this general permit are expected to occur with low frequency (intermittent), small magnitude (low volume limited to no more than 1.0 MGD, typically approximately 0.0072 MGD to 0.072 MGD), and short duration (temporary or short-term, typically from 24 hours up to 12 months in duration). At the maximum effluent flow permitted for a remediation activity discharge, the dilution factor in each of the riverine waterbodies in the Action Area would be:

1) Connecticut River: approximately 472,223:1 at 0.0072 MGD to 3,401:1 at 1.0 MGD
2) Merrimack River: approximately 224,015:1 at 0.0072 MGD to 1,614:1 at 1.0 MGD
3) Piscataqua River via Cocheco River: approximately 9,051:1 at 0.0072 MGD to 66:1 at 1.0 MGD
4) Taunton River: approximately 6,632:1 at 0.0072 MGD to 49:1 at 1.0 MGD

Marine environments are high energy, and have a large volume of water available for dilution. Given the size of the marine waterbodies in the Action Area (i.e., Cape Cod Bay, Massachusetts Bay and Great Bay) and the high energy and volume in these marine environments relative to the low flow and the proximity to the near shore of individual remediation activity discharges, EPA assumes rapid and complete mixing of discharges with the marine waters to which the effluents may be discharged.

b. Potential Effects of the Action on Listed Species

With respect to potential effects of the proposed action on listed species, EPA considered the following potential stressors:

1) Sound
2) Dredging (Capture, Impingement, Entrainment)
3) Habitat Structure and Disturbances
4) Water Quality
5) In-Water Structures
6) Prey Quality/Quantity
7) Vessel Traffic

Because the proposed action will authorize the discharge of pollutants to surface water, EPA believes the relevant stressors are: 1) Habitat Structure and Disturbances; 2) Water Quality; and

3) Prey Quality/Quantity. EPA also examined dredge-related dewatering discharges concurrently with habitat structure and disturbances, water quality and prey quality/quantity rather than separately as "Dredging" because this stressor, as defined by NMFS in its Section 7 guidance, is not relevant to the proposed action. EPA does not believe dredging is a relevant stressor to assess because the proposed action neither authorizes nor requires dredging, which is an activity under the jurisdiction of the U.S. Army Corps of Engineers. Dredging activities in navigable waters, which require a general or individual permit under Section 401/404 of the Clean Water Act, are not eligible for coverage under the RGP. Thusly, the proposed action is not expected to result in any dredge-related interaction (capture, impingement, entrainment). Again, however, EPA has examined potential effects of surface water discharges from the dewatering of dredge (i.e., excavation) material, which may be authorized under the proposed action. EPA believes the remaining potential stressors are not relevant to the proposed action, including all other dredging-related activities. The reasons for excluding the remaining stressors are as follows:

1) EPA does not believe sound is a relevant stressor to assess because the proposed action is not expected to affect ambient noise levels.
2) EPA does not believe in-water structures is a relevant stressor to assess with respect to the proposed action because the proposed action neither authorizes nor requires the construction of in-water structures. Thusly, the proposed action is not expected to result in shading effects on prey or the construction of in-water structures that could affect normal behaviors, including passage.
3) EPA does not believe vessel traffic is a relevant stressor to assess with respect to the proposed action because the proposed action is not expected to result in any change in vessel traffic (volume, speed and/or route).

EPA's examination of the potential effects of the proposed action on listed species from the relevant stressors is presented for the following listed species: i) Shortnose sturgeon and Atlantic sturgeon.

i. Shortnose sturgeon and Atlantic sturgeon

As discussed in Section 3 of this document, the shortnose sturgeon and Atlantic sturgeon are the only ESA listed species that are likely to encounter an RGP discharge. These two sturgeon species are also the only protected species expected to inhabit the riverine environment, including the Connecticut River (downstream of Turner's Falls, Massachusetts and encompassing the area near the Holyoke Dam); the Merrimack River below the Essex Dam (Merrimack River Dam; in Lawrence, Massachusetts, including the area near Haverhill); the Taunton River in Massachusetts; the Piscataqua River in New Hampshire; and the nearshore marine waters of Massachusetts and New Hampshire. Also as discussed in Section 3 of this document, EPA identified shortnose sturgeon adult life stages in the Merrimack, and Connecticut Rivers, and transiently, in the Piscataqua River, and the nearshore marine waters of Massachusetts and New Hampshire and shortnose sturgeon early life stages in the spawning areas of the Merrimack and Connecticut Rivers. Similarly, EPA identified Atlantic sturgeon adult life stages in the Merrimack, Connecticut, Taunton, and Piscataqua Rivers, and the nearshore marine waters of Massachusetts and New Hampshire, and Atlantic sturgeon early life stages in the nursery area of the Taunton River for its evaluation of the potential effects in this effects determination.

Of the population risks and stressors identified for shortnose sturgeon in Section 3 of this letter, remediation activity discharges are most likely to adversely impact their abundance with respect to "water quality and contaminants", as noted in the 2010 Shortnose Sturgeon Biological Assessment. Similarly, of the population risks and stressors identified for Atlantic sturgeon in Section 3 of this letter, remediation activity discharges are most likely to adversely impact their abundance and habitat with respect to "persistent, degraded water quality", as noted in the final rulings for the Atlantic sturgeon. The following sections address potential effects related to habitat structure and disturbances and water quality, including potential effects related to prey quality/quantity.

Where available, effects information for shortnose sturgeon have been included in EPA's analysis. However, EPA was unable to identify specific effects information for Atlantic sturgeon. Since Atlantic sturgeon are closely related to shortnose sturgeon, the effects of the proposed action on Atlantic sturgeon are likely similar to effects of the proposed action on shortnose sturgeon. Therefore, when effects information is noted for shortnose sturgeon, EPA considers this information an acceptable surrogate for effects on Atlantic sturgeon. Where EPA could not find specific effects information for shortnose sturgeon or Atlantic sturgeon, EPA used the best available effects data for surrogate species, including other sturgeon species (e.g., green sturgeon and white sturgeon), and trout species (e.g., rainbow trout and brook trout). Where effects information was not available for surrogate species, EPA used the best available effects information, which includes potential prey species, and unrelated, but sensitive species, for EPA's analysis.

(1) Habitat Structure and Disturbance

EPA has assessed the habitat structure and disturbance effects because the proposed action may affect shortnose sturgeon or Atlantic sturgeon with respect to this stressor. While the RGP does not authorize or require dredging activities, one of the eight surface water discharge types eligible for coverage under this general permit *may* result from the dewatering of dredged (i.e., excavated) material. Therefore, EPA has also assessed water quality effects from dredging concurrently with habitat structure and disturbance only in regard to dewatering of dredged material because the proposed action may affect listed species with respect to this stressor. EPA has assessed the habitat structure and disturbance effects for the shoreline areas of Massachusetts and New Hampshire or the Connecticut River, Merrimack River, Taunton River, or Piscataqua River in this analysis. EPA considered all life stages potentially present in these areas with respect to the potential and habitat structure and disturbance effects.

As previously described, discharges eligible for coverage under this general permit are expected to occur with low frequency (intermittent), small magnitude (small volume limited to no more than 1.0 MGD), and short duration (temporary or short-term). Also as previously described, the projected dimensions of the discharge plume of any riverine RGP outfall are generally expected to be confined to the immediate riverbank and only extend out a minimal distance into the mainstem of the river and a minimal distance downstream of the discharge before complete mixing takes place. Also as previously described, dilution in the riverine waterbodies in the Action Area is high, given the conservative dilution estimates aforementioned. Similarly, the projected dimensions of the discharge plume of any estuarine or marine RGP outfall are generally expected to be confined to the immediate shoreline and only extend out a minimal

distance into the marine waters where rapid and complete mixing takes place. Also as previously described, EPA assumes rapid and complete mixing of discharges with the marine waters in the Action Area, given the size of the marine waterbodies and the high energy and volume in these marine environments relative to the low flow and the proximity to the near shore of individual remediation activity discharges. Finally, the expected distribution of Atlantic sturgeon and shortnose sturgeon in the Action Area has the potential to include the immediate riverbank of the shallow mainstem waters of the Merrimack, Connecticut, Taunton and/or Piscataqua Rivers, including the nearshore marine waters of Massachusetts and New Hampshire. Therefore, contact between Atlantic sturgeon and shortnose sturgeon in the Action Area and the projected RGP discharges may occur.

The potential effects of remediation activity discharges with respect to habitat structure and disturbance effects are likely to include a temporary increase in turbidity and/or suspended sediment or a temporary excursion in sediment regime parameters (e.g., erosivity and sediment transport) in the receiving waterbody in the immediate vicinity of an RGP outfall. To evaluate these effects, EPA focused on the parameters included in the RGP which limit the likely habitat structure and disturbance effects. The relevant individual parameters for these stressors included in the RGP are: 1) total suspended solids (TSS); and 2) effluent flow. EPA also generally evaluated turbidity with respect to TSS, and sediment regime parameters with respect to effluent flow, which are not parameters included in the RGP but are potential interrelated habitat structure and disturbance effects.

Total Suspended Solids (TSS)

The RGP controls suspended sediment through a numeric limitation for total suspended solids (TSS) and non-numeric limitations for turbidity and settleable or floating pollutants or debris. The increase in TSS levels in remediation activity discharges is expected to be minor and temporary. Given the numeric effluent limitation of 30 mg/L for total suspended solids (TSS) for all discharge types eligible for coverage under this general permit, EPA expects that remediation activity discharges have the potential to produce TSS concentrations up to 30 mg/L at the point of discharge. Upon mixing with the receiving waters, TSS concentrations are expected to dissipate rapidly to concentrations at or below the minimum level of detection (approximately 5 mg/L) given the high available dilution in the waterbodies in which sturgeon are likely to be present in the Action Area (i.e., the Connecticut, Merrimack, Piscataqua and Taunton Rivers). The temporary increase in TSS levels in remediation activity discharges is also expected to occur with low frequency (intermittent), small magnitude (small volume limited to no more than 1.0 MGD), and short duration (temporary or short-term).

Data provided by EPA's STORET database indicate that the median TSS in the Connecticut River since 2005 is 8.0 mg/L (with a maximum recorded value of 115 mg/L) at USGS station 01172010. The USGS has no recorded Total Suspended Solid (TSS) data in the Merrimack River at station 01100000 since 2003. However, since 1953, the median TSS concentration in the Merrimack River is 68 mg/L, with a maximum recorded value of 141 mg/L. The median TSS in the Taunton River is 11 mg/L (with a maximum recorded value of 170 mg/L at USGS station 01108000. The median total solids concentration for waters in the Piscataqua River Basin is 84.6 mg/L. Given the high available dilution in these waterbodies, the effect from individual

remediation activity discharges, even when discharged at the maximum allowable concentration, 30 mg/L, is not expected to change the instream solids concentration.

According to NMFS, observed impacts to listed species from elevated sediment and turbidity levels fall into several broad categories such as avoidance or behavioral responses, feeding and hunting, breeding and egg survival, habitat loss, juvenile survival and physical damage. The potential cumulative effect of these impacts includes reduced disease and parasite resistance, reduced growth, and degraded health of individual organisms in the fish community. Population reductions can take place both through direct mortality in the short term and reduced reproductive success in the long term.[11] TSS is most likely to affect sturgeon if a discharge plume causes a barrier to normal behaviors. Because discharges of TSS are expected to be minor (30 mg/L at the point of discharge) and temporary (TSS concentrations will rapidly dissipate because of high dilution), EPA expects sturgeon will either swim through any resulting sediment plume or make small evasive movements to avoid the plume. Consequently, any effect of a sediment plume caused by the proposed action on sturgeon movements or behavior is expected to be temporary and such small adjustments cannot be meaningfully measured, detected, or evaluated.

Elevated sediment and turbidity levels can also cause burial or smothering of listed species or their prey, or alter the substrate type. Studies of the effects of turbid water on fish suggest that concentrations of suspended solids can reach thousands of milligrams per liter before an acute toxic reaction is expected (Burton 1993). Available information indicates that TSS levels have been shown to have adverse effect on fish at 580 mg/L for the most sensitive species, with 1,000 mg/L more typical (see summary of scientific literature in Burton 1993); and on benthic communities at 390 mg/L (EPA 1986)). A frequently cited study by Newcombe and Jensen (1996), indicated sublethal effects (e.g. increased respiration rate) were observed in eggs and larvae of fish when exposed to TSS concentrations as low as 55 mg/L for one hour.[12] Given the numeric effluent limitation of 30 mg/L for TSS and the high available dilution in the waterbodies in which sturgeon are likely to be present in the Action Area (i.e., the Connecticut, Merrimack, Piscataqua and Taunton Rivers), EPA does not expect TSS levels to reach levels that are toxic to the listed species or their prey. Further, the RGP contains non-numeric effluent limitations which require treatment to ensure discharges remain free from pollutants in concentrations or combinations that settle to form harmful deposits and free from turbidity levels that would impair the designated uses of the receiving waters as aquatic habitat.

Effects Determination for TSS

Based on the best available information, EPA has made the determination that the habitat structure effects from TSS on shortnose sturgeon or Atlantic sturgeon will be insignificant and/or discountable because:

1) Any increase in turbidity/suspended sediment is minor and temporary such that there is no impairment of movement of individual animals or any other effect that can be meaningfully measured, detected, or evaluated, and effects are therefore insignificant:

[11] See footnote 9, above.
[12] See footnote 9, above.

Discharges eligible for coverage under this general permit are expected to occur with low frequency (intermittent), small magnitude (small volume limited to no more than 1.0 MGD), and short duration (temporary or short-term) and as such are likely to cause effects minor and temporary in nature. As such, any change to the substrate type or alteration in the depth of waters is expected to also be minor and temporary. However, if TSS is present in remediation activity discharges, the discharge must meet non-numeric limitations and a numeric technology-based limitation lower than levels that are toxic to benthic communities. Given the high available dilution in the waterbodies in the Action Area, the effect from individual remediation activity discharges, even if at the maximum allowable concentration, 30 mg/L, is not expected to change the instream solids concentration. This minor and temporary alteration is not expected to affect the way that individual animals use the Action Area or result in behavior change (e.g., foraging) in individual animals that can be meaningfully measured, detected, or evaluated and is therefore insignificant.

Consequently, the proposed action is not likely to adversely affect the shortnose sturgeon, the Atlantic sturgeon or their prey in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River, including the nearshore marine waters of Massachusetts and New Hampshire.

Effluent Flow

Discharges eligible for coverage under this general permit are considered minor. That is, discharges authorized under this general permit are expected to occur with low frequency (intermittent), small magnitude (low volume limited to no more than 1.0 MGD, typically approximately 0.0072 MGD to 0.072 MGD), and short duration (temporary or short-term, typically from 24 hours up to 12 months in duration). Effluent Flow is limited to a maximum of 1.0 million gallons per day (MGD). When a treatment system is used to meet the effluent limitations in the RGP, the draft RGP further limits effluent flow to the design flow of any treatment system in use, when the design flow is less than 1.0 MGD. In addition, the draft RGP includes non-numeric limitations and BMP requirements pertaining to flow. First, effluent flow cannot exceed the flow of or alter the structural characteristics of the receiving water. Second, flow control measures (e.g., sediment filters, splash blocks) must be used if necessary to dissipate energy and control erosion or scouring during discharge. Finally, drainage control practices must ensure that the discharges do not adversely affect existing water quality by preventing any erosion, stream scouring, or sedimentation caused directly or indirectly by the discharge and/or which contributes additional pollutants.

Potential effects of remediation activity discharges relating to effluent flow include a temporary excursion in sediment regime parameters (e.g., erosivity and sediment transport) in the receiving waterbody in the immediate vicinity of an RGP outfall. According to NMFS, erosion in aquatic systems occurs where the flow or movement of water scours loose sediment from stream banks and shorelines. An increase in the flow rate and volume of water can increase scouring and sediment transport potential. Excessive erosion can disturb soils or alter hydrology (e.g., current conditions). Excessive scouring could result in a change in water depth or substrate type. Effects may include bank erosion, downstream sediment movement, and the formation and loss of structural elements such as pools and riffles. Excessive sediment transport could result in particulate sediment covering the natural substrate, causing direct and indirect biological effects

ranging from behavioral to physiological to toxicological in aquatic species to disruption of aquatic habitats.[13] High energy waterbody types (e.g., large rivers including the Connecticut, Merrimack, Piscataqua and Taunton Rivers, and high energy marine waters) are capable of recovering more quickly from events causing excess suspended sediment (USEPA 2009).

As previously described, EPA identified the lowest daily average flow of 3,400 MGD for the Connecticut River, 1,616 MGD for the Merrimack River, 65 MGD for the Piscataqua River via the Cocheco River, and 48 MGD for the Taunton River. Given the low volume of remediation activity discharges, limited to no more than 1.0 MGD, typically approximately 0.0072 MGD to 0.072 MGD, and the high flow in the receiving waters, In addition, given the low frequency (intermittent) and short duration (temporary or short-term, typically from 24 hours up to 12 months in duration) of remediation activity discharges, and the high energy of the receiving waters, disturbance to sediment that would lead to a change in substrate type or water depth is unlikely to occur. EPA does not expect the minor and temporary discharges authorized by the RGP to impact the zone of passage for listed species. Any minor and temporary effects to the physical water current, such as the speed or direction of flow, are not expected to be detectable. Any minor and temporary effects to the chemical features, such as salinity, dissolved oxygen or temperature, are not expected to be measureable.

Effects Determination for Effluent Flow

Based on the best available information, EPA has made the determination that the habitat structure effects from effluent flow on shortnose sturgeon or Atlantic sturgeon will be insignificant and/or discountable because:

1) Any change in water depth will not change the use of the area by species and are therefore discountable; and
2) Any change in substrate type will not change the use of the area by species or diminish its quality such that there would be an effect to an individual that can be meaningfully measured, detected or evaluated and are therefore insignificant.

Consequently, the proposed action is not likely to adversely affect the listed species in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River, including the nearshore marine waters of Massachusetts and New Hampshire.

(2) Water Quality/ Prey Quality/Quantity

EPA has assessed water quality and prey quality/quantity effects because the proposed action may affect shortnose sturgeon, Atlantic sturgeon, or their prey with respect to these stressors. EPA considered water quality and prey quality/quantity effects to these species concurrently because the effects to the listed species and their prey are expected to be similar. While the RGP does not authorize or require dredging activities, one of the eight surface water discharge types eligible for coverage under this general permit *may* result from the dewatering of dredged (i.e., excavated) material. Therefore, EPA has also assessed water quality effects from dredging concurrently with water quality and prey quality/quantity only in regard to dewatering of dredged material because the proposed action may affect listed species with respect to this

[13] See footnote 9, above.

stressor. EPA has assessed the potential water quality and prey quality/quantity effects for the shoreline areas of Massachusetts and New Hampshire or the Connecticut River, Merrimack River, Taunton River, or Piscataqua River, including the coastal embayments and nearshore marine waters of Massachusetts and New Hampshire, in this analysis. EPA considered all life stages potentially present in these areas with respect to the potential water quality and prey quality/quantity effects.

To evaluate water quality and prey quality/quantity effects, EPA focused on the individual pollutants potentially present in remediation activity discharges, which are mostly likely to cause water quality and prey quality/quantity effects. The individual pollutants/parameters evaluated in this analysis include eighteen (18) inorganic parameters, five (5) non-halogenated volatile organic compound (VOC) parameters, sixteen (16) halogenated VOC parameters, twelve (12) non-halogenated semi-volatile organic compound (SVOCs) parameters, two (2) halogenated SVOC parameters, five (5) fuel-related parameters, pH, and temperature. Although the inorganic parameters (ammonia, chloride, total residual chlorine, total suspended solids, thirteen (13) metals and cyanide) are generally the most common pollutants expected in discharges authorized under the RGP, are most likely to have end-of-pipe effluent limitations which are adjusted for allowable dilution, and, along with pH are limited for all discharges, all pollutants/parameters authorized by the RGP are addressed in this section as potential stressors with water quality or prey quality/quantity effects.

The potential water quality and prey quality/quantity effects that could result from the discharge of one or more pollutants potentially present in remediation activity discharges are expected to primarily consist of acute and/or chronic effects from pollutants individually or in combination that cause the direct loss of individual listed species or their prey. In addition to direct mortality, the pollutants associated with remediation activity discharges can lead to changes in fish behavior, deformations, reduced egg production and survival (Health, 1987). These pollutants can also alter the physical properties of the receiving waterbody by causing changes in the receiving water chemistry. The majority of pollutants included in the RGP are organic compounds. According to NMFS, factors affecting whether or not an organism will experience adverse effects to a given organic substance released to the environment include:

1) The chemical released and its physical form at the time of release (solid, liquid, or vapor) and
2) Its solubility in water;
3) The chemical's affinity for lipids (log K_{ow}) or organic carbon (K_{oc}) relative to water;
4) The chemical's ability to volatilize from water (Henry's Law Constant);
5) The chemical's likelihood of concentrating in aquatic organisms (Bioconcentration Factor);
6) The chemical's toxicity in the organism; and
7) The exposure of the species or designated critical habitat to the chemical.

Aquatic organisms can be expected to experience greater exposure to more soluble substances. Other factors affecting the likelihood of an organism's exposure to the organic pollutants included in the RGP include environmental degradation and biodegradation. Based on observed effects in other non-salmonid fish, that organic pollutants could lead to decreased

growth, alterations of metabolic functions, and reduced recruitment in the listed species.[14]

The specific potential effects of individual pollutants on the listed species or their prey, if known, are included in the analysis of each individual pollutant. EPA generally included available acute and chronic toxicity values, including LC_{50} concentrations (the concentration causing mortality to 50 percent of the test organisms), when available. Where individual pollutants are substantially similar, that is, share physical and chemical properties that result in similar effects, the individual pollutants are grouped together in the analysis of their potential effects. With respect to effects that could result from the discharge of pollutants potentially present in remediation activity discharges in combination, EPA also evaluated an additional receiving water chemistry parameter, dissolved oxygen (DO), that is not a parameter included in the RGP but could nevertheless be affected by remediation activity discharges.

EPA has made the determination that the water quality effects from the pollutants in discharges authorized under the RGP, if present, on the shortnose sturgeon and the Atlantic sturgeon, will be insignificant and/or discountable in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River, or the coastal embayments/nearshore marine waters of Massachusetts and New Hampshire, because: 1) water quality standards are met at the point of discharge for all pollutants; 2) any exposure to the discharge prior to full dilution would be extremely unlikely to occur or would have insignificant effects (i.e., cannot be meaningfully measured, detected, or evaluated); and 3) any increase in turbidity/suspended sediment is minor and temporary such that there is no impairment of movement of individual animals or any other effect that can be meaningfully measured, detected, or evaluated. With respect to prey quantity/quality, EPA has made the determination that the proposed action will be insignificant because: 1) discharges cause only a minor and temporary reduction in available prey such that any effects on individual animals are not capable of being meaningfully measured, detected, or evaluated; or 2) where the proposed action could potentially cause a permanent reduction in the abundance, availability, accessibility, and quality of prey, it is so small that any effect on listed species cannot be meaningfully measured, detected, or evaluated.

Consequently, the proposed action is not likely to adversely affect the listed species or their prey in the shoreline areas of Massachusetts and New Hampshire or the Connecticut River, Merrimack River, Taunton River, or Piscataqua River, including the coastal embayments/nearshore marine waters of Massachusetts and New Hampshire. Incidental take is not anticipated to occur, nor has any take been exempted by NMFS. Evidence which supports this determination is provided for each individual pollutant (or group of pollutants) in the sections that follow. Unless otherwise provided, EPA has not reviewed scientific literature that specifically investigates the sensitivity of protected species, especially shortnose sturgeon and Atlantic sturgeon, to the very low expected levels and minimal exposures to the pollutants potentially present in remediation activity discharges.

Inorganics

[14] See footnote 9, above.

Inorganic pollutants are substances that generally do not have a chemical structure based on carbon or its derivatives. The inorganic parameters potentially present in remediation activity discharges and the expected water quality effects on shortnose, Atlantic sturgeon, or their prey, if known, are discussed in this section. EPA's determination with respect to inorganics potentially present in remediation activity discharges is made for monitor-only requirements, total residual chlorine and total suspended solids individually, and metals and cyanide, follows the information provided for each of these parameters or groups of parameters.

Ammonia and Chloride

Ammonia is subject to a monitor-only requirement. EPA selected ammonia as an indicator parameter because of its toxicity and the availability of numeric water quality criteria, including EPA's acute and chronic aquatic life NRWQC. EPA's recommended criteria for ammonia in freshwater are based on temperature, pH and the presence of certain species and life stages in the receiving water. For example, when mussels and/or salmonids are absent, early life stages are present, the pH of the receiving water is 8.3 SU and the receiving water temperature is 20°C, the recommended criteria for ammonia are as follows: 1) Acute criteria: 4.9 mg/L for a cold water fishery and 3.0 mg/L for a warm water fishery; and 2) Chronic criteria: 1.7 mg/L for a cold water fishery and 1.1 mg/L for a warm water fishery. EPA's recommended criteria for ammonia in freshwater are based on temperature, pH and salinity.[15] For example, when the receiving water temperature is 15°C, the pH of the receiving water is 7.8 SU and the receiving water salinity is 30 g/kg, the recommended acute criterion value is 16 mg/L and the recommended chronic criterion value is 2.4 mg/L.

Ammonia is highly soluble. The concentration of total ammonia, often expressed as ammonia nitrogen, is the sum of two species, the more abundant of which is the ammonium ion (NH_4^+), the less abundant of which is the non-dissociated or unionized ammonia (NH_3) molecule, which is more toxic. The ratio of these species in a given aqueous solution is dependent upon both pH and temperature. Generally, as values of pH and temperature increase, the concentration of NH_3 increases and the concentration of NH_4^+ decreases. The toxicity of total ammonia increases as pH increases.[16] In excessive quantities, nutrients such as ammonia can have adverse effects on ecosystems, and nutrient enrichment, which leads to eutrophication, often ranks as one of the top causes of water resource impairment (Bricker et al. 2008, USEPA 2014). Ammonia can also affect the dissolved oxygen level in a waterbody and lead to the development of eutrophic conditions in a waterbody.[17] Eutrophication alters the composition and species diversity of aquatic communities through intensifying competition, which can lead to replacement of native species by non-native or invasive species that are better adapted to eutrophic environments, many of which produce toxins ((Nordin 1985, Welch et al. 1988, Carpenter et al. 1998, Smith 1998, Smith et al. 1999) – after (USEPA 2000b). Eutrophication can also change productivity, in which nutrients lead to increased organic matter loading through increased productivity, which can result in cyanobacterial or algal blooms, surface scums, floating plant mats and excess

[15] See EPA's 1989 *Ambient Aquatic Life Water Quality Criteria for Ammonia (Saltwater).*
[16] *Aquatic Life Ambient Water Quality Criteria for Ammonia – Freshwater*. EPA 822-R-13-001: April 2013.
[17] See footnote 9, above.

benthic macrophytes and mortality to listed species (Carr et al. 2005, Shotts et al. 1972, Landsberg 2002, Shumway et al. 2003).

While ammonia can also be directly toxic to aquatic life, shortnose sturgeon are less sensitive to ammonia relative to other fish species, ranking 19th among 27 freshwater fish genera. The 96-h LC_{50} for fingerling shortnose sturgeon exposed to total ammonia is 36.49 mg/L at pH 8, the 96-h median-lethal total ammonia nitrogen is 149.8 +/- 55.20 mg/L and the calculated 96-h LC_{50} for un-ionized ammonia is 0.58 +/-0.213 mg/L (Fontenot et al. 1998). According to NMFS, ammonia has no bioaccumulation potential, and an estimated toxic concentration of ammonia to shortnose sturgeon is 580 µg/L. Since Atlantic sturgeon are closely related to shortnose sturgeon, the effects of the proposed action on Atlantic sturgeon are likely similar to effects of the proposed action on shortnose sturgeon. Therefore, EPA considers this estimated toxic concentration to shortnose sturgeon an acceptable surrogate for effects to Atlantic sturgeon. The best available effects information for other sensitive species include chronic exposure effect concentration data for ammonia toxicity for delta smelt, which indicates a LC_{50} of 13 mg/L for 4-day exposure of 57-day old juveniles to total ammonium (Connon et al. 2011). [18] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 1.7 µg/L and 630 µg/L for fish and daphnids, respectively.[19]

EPA does not currently have information regarding ammonia in discharges covered under this general permit. However, monitoring data available for sites with remediation and/or dewatering discharges covered under *individual* permits in Region 1 indicate that ammonia may be present in similar discharges at low concentrations.[20] In order to determine the extent of this *potential* pollutant in remediation activity discharges and to determine the frequency with which remediation activity discharges may contain ammonia, the draft RGP includes monitoring for ammonia.

Chloride is subject to a monitor-only requirement. However, on a case-by-case basis, as a requirement for CWA §401 certification, a numeric effluent limitation of 230 mg/L may be imposed for a discharge when a waterbody is listed for impairment for chloride, if necessary to meet the requirements of such certification. New Hampshire adopted EPA's chronic aquatic life water quality criterion from EPA's National Recommended Water Quality Criteria (NRWQC), 230 mg/L, into its water quality standards as numeric criterion. In Massachusetts, 310 CMR 4.05(e) includes this numeric limitation by reference to EPA's 2002 NRWQC.[21] Pursuant to 40 CFR §122.44(d)(1)(i), this limitation is necessary because where a waterbody is impaired, any addition of chloride is or may be discharged at a level which will cause, have the reasonable potential to cause, or contribute to an excursion above State WQSs. EPA's acute NRWQC is 860 mg/L.

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[18] See footnote 9, above.
[19] G.W. Suter II and C. L. Tsao. *Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1996 Revision*. U.S. Department of Energy: ES/ER/TM-96/R2, 151 pp. See footnote 19
[20] See, for example, Discharge Monitoring Reports for MA0000825, MA0001929, MA0003280, MA0003298, MA0003425 and MA0004006.
[21] EPA 822R-02-047, November 2002.

Chlorides are used heavily for road salting and are present near salt storage areas. As a result, the presence of chloride in groundwater and surface waters that comprise remediation activity discharges in Massachusetts and New Hampshire is widespread. Other sources of chloride in remediation activity discharges may include deicing salts, and stormwater runoff. EPA's NRWQC for chloride were derived based on sodium chloride toxicity test data available for twelve (12) different species in laboratory reconstituted water. While the chlorides of potassium, calcium and magnesium are generally more toxic to aquatic life than sodium, sodium is likely the most common chloride present. The relative toxicity of chlorides to sensitive species can increase as hardness values decrease.[22]

The best available effects information available were for other surrogate, or sensitive species. EPA's *Quality Criteria for Water* indicates that in an early life-stage test with rainbow trout, a chloride concentration of 2,740 mg/L killed all the exposed organisms (Spehar 1987). Based on tests on sodium chloride, the acute sensitivities of freshwater animals to chloride ranged from 1,470 mg/L for *Daphnia pulex* to 11,940 mg/L for the American eel. In the life-cycle test with *Daphnia pulex*, survival was as good as in the control treatment at chloride concentrations up to 625 mg/L (Birge et al. 1985). In an early life-stage test with the fathead minnow, *Pimephales promelas*, Birge et al. (1985) found that weight was as good as in the control treatment up to a chloride concentration of 533 mg/L.[23]

Effects Determination for Ammonia and Chloride

Based on the best available information, EPA has made the determination that the water quality effects from ammonia and chloride on shortnose sturgeon or Atlantic sturgeon will be discountable because:

1) Unless monitoring data indicate ammonia or chloride is present in remediation activity discharges, EPA assumes ammonia is not present in remediation activity discharges, such that effects are extremely unlikely to occur and are therefore discountable.

If monitoring data indicate ammonia or chloride is present in remediation activity discharges, EPA will evaluate whether water quality standards will be met at the point of discharge. Limitations will be imposed for ammonia on a case-by-case basis if EPA determines they are necessary to meet water quality standards, or an individual permit will be required. Limitations equivalent to EPA NRWQC will be imposed for chloride if EPA determines they are necessary to meet water quality standards.

With respect to prey quantity/quality, EPA has made the determination that the proposed action will be insignificant because:

1) If monitoring data indicate ammonia or chloride is present in remediation activity discharges, EPA will evaluate whether discharges cause only a minor and temporary reduction in available prey such that any effects on individual animals are not capable of being meaningfully measured, detected, or evaluated or where the proposed action could

[22] See *"Acute Toxicity of Chloride to Select Freshwater Invertebrates, September 26, 2008"*.
[23] See *Ambient Water Quality Criteria for Chloride – 1988*. EPA 440/5-88-001, February, 1988.

> potentially cause a permanent reduction in the abundance, availability, accessibility, and quality of prey, it is so small that any effect on listed species cannot be meaningfully measured, detected, or evaluated and are therefore insignificant. Limitations will be imposed if EPA determines they are necessary to meet water quality standards, or an individual permit will be required.

Consequently, the proposed action is not likely to adversely affect the shortnose sturgeon, the Atlantic sturgeon or their prey in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River, or the coastal embayments/nearshore marine waters of Massachusetts and New Hampshire.

Total Residual Chlorine (TRC)

TRC consists of the sum of free chlorine and combined chlorine. TRC is limited to 11 µg/L in freshwater and 7.5 µg/L in saltwater, equivalent to EPA's chronic aquatic life NRWQC. The Commonwealth of Massachusetts' surface water-quality standards require the use of the 2002 EPA NRWQC where a specific pollutant could reasonably be expected to adversely affect existing or designated uses (314 CMR 4.05 (5)(e)). The State of New Hampshire's water quality regulations for chlorine, found at Chapter 1700, Surface Water Quality Regulations, Part Env-Wq 1703.21(b), are equivalent to EPA's NRWQC. any discharge that contains or could contain residual chlorine must meet the water quality-based effluent limitation (WQBELs): 1) Freshwater: 11 µg/L (0.011 mg/L); or 2) Saltwater: 7.5 µg/L (0.0075 mg/L). TRC is also limited to a maximum of 0.2 mg/L (200 µg/L), regardless of dilution. The Massachusetts *Implementation Policy for the Control of Toxic Pollutants in Surface Waters*, dated February 23, 1990, states that waters shall be protected from unnecessary discharges of excess chlorine. Per this policy, the maximum effluent concentration of chlorine shall not exceed 1.0 mg/L TRC. However, EPA selected a more conservative technology based effluent limit (TBEL) for both states using best professional judgment as authorized by §402(a)(1) of the CWA. EPA selected the monthly average effluent limitation, consistent with ELGs at 40 CFR §423.12 for the Steam Electric Power Point Source Category and the technical factors supporting these limitations.

TRC may be present in discharges if operators use chlorine compounds to control bacterial growth in the treatment systems or in pipelines and tanks encounter, when disinfection of effluent co-mingled with incidental domestic sewage is necessary, or if discharges contain potable water that has been chlorinated as required in 40 CFR §141.72. Chlorine and chlorine compounds are toxic to aquatic life. However, chlorine is generally too reactive to be measured in surface water. The fate of chlorine in water has been well studied (Das 2002). Chlorine released to surface water is expected to either partition to air or dissolve (7.3 g/L at 20 °C) and then undergoes a disproportionation within seconds at environmental pH to form hydrochloric (H+ + Cl-) and hypochlorous acid (HOCl) (Cotton et al. 1999; Das 2002; EPA 1999; Farr et al. 2003; Morris 1946; Snoeyink and Jenkins 1980; Tchobanoglous and Schroeder 1985; Wang and Margerum 1994). Molecular chlorine in water at very low pH is expected to volatilize rapidly based on a Henry's law constant of 1.17×10^{-2} atm-m3/mol (Staudinger and Roberts 1996). The hypochlorous acid formed during the disproportionation of chlorine in natural waters reacts with organic and inorganic materials, ultimately forming chloride/chloride salts, oxidized inorganics, chloramines, trihalomethanes, oxygen, and nitrogen (i.e., chlorine demand) (IARC 1991;

Vetrano 2001). The equilibrium between chlorine, hypochlorous acid, and hypochlorite acid is dependent on the pH of the solution (Farr et al. 2003).[24]

Chlorine is not expected to bioaccumulate in plants or animals since it reacts with the moist tissues of living systems (Compton 1987; Schreuder and Brewer 2001; Schmittinger et al. 2006). The best available effects information available were for other potential prey, or sensitive species. Thirty-three (33) freshwater species in twenty-eight (28) genera exposed to TRC were evaluated for EPA's acute criteria development. The freshwater acute values ranged from 28 µg/L for *Daphnia magna* to 710 µg/L for the threespine stickleback, with fish and invertebrate species showing similar ranges of sensitivity. The freshwater chronic values for two invertebrate and one fish species ranged from less than 3.4 µg/L to 26 µg/L. Twenty-four (24) saltwater species in twenty-one (21) genera exposed to TRC were also evaluated for EPA's acute criteria development. The LC_{50} ranged from 26 µg/L for the eastern oyster to 1,418 µg/L for a mixture of two shore crab species, with fish and invertebrate species showing similar ranges of sensitivity. Available data indicate that aquatic plants are more resistant to chlorine than fish and invertebrate species.[25]

Effects Determination for TRC

Based on the best available information, EPA has made the determination that the water quality effects from chlorine on shortnose sturgeon or Atlantic sturgeon will be discountable because if present in a discharge authorized under the RGP:

1) Water quality standards are met at the point of discharge: The numeric water quality-based limits have been established at the chronic aquatic life criteria adopted in each state, at concentrations near or below the minimum levels of detection, and are more stringent than available effects data for the listed species (or surrogate species). TRC degrades rapidly and as a result, it is not expected to be detected in the aquatic environment. Therefore, the numeric limits will not adversely affect listed species because effects are extremely unlikely to occur and are therefore discountable.

With respect to prey quantity/quality, EPA has made the determination that the proposed action will be insignificant because if cyanide is present in a discharge authorized under the RGP:

1) Discharges are likely to cause only a minor and temporary reduction in available prey, such that any effects on individual listed species are not capable of being meaningfully measured, detected, or evaluated: Discharges eligible for coverage under this general permit are expected to occur with low frequency (intermittent), small magnitude (small volume limited to no more than 1.0 MGD), and short duration (temporary or short-term) and as such are likely to cause effects minor and temporary in nature. However, if this pollutant is present in remediation activity discharges, the discharge must meet numeric water quality-based limits established at the chronic aquatic life criteria adopted in each

[24] EPA 749-F-94-010, December, 1994; and *Toxicological Profile for Chlorine*. Agency for Toxic Substances and Disease Registry: November, 2010.

[25] *Quality Criteria for Water 1986*. Environmental Protection Agency: EPA 440/5-86-001; May 1, 1986. (EPA's "Gold Book")

> state, at concentrations near or below the minimum levels of detection, and at concentrations near or below the available effects concentrations at end-of-pipe. In addition, this pollutant is expected to dissipate rapidly because of the volatility of the parameter and the high dilution in the receiving waters to concentrations less than the minimum level of detection such that effects are likely to cause only a minor and temporary change in the abundance, distribution, quality and availability of only the most sensitive prey species. This minor and temporary loss or alteration is not expected to affect the way that individual animals use the Action Area or result in behavior change (e.g., foraging) in individual animals that can be meaningfully measured, detected, or evaluated and is therefore insignificant.

Consequently, the proposed action is not likely to adversely affect the shortnose sturgeon, the Atlantic sturgeon or their prey in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River, or the coastal embayments/nearshore marine waters of Massachusetts and New Hampshire.

Total Suspended Solids (TSS)

TSS is limited to a maximum of 30 mg/L, regardless of dilution. TSS is a conventional pollutant that may include inorganic (e.g., silt, sand, clay, and insoluble hydrated metal oxides) and organic matter (e.g., flocculated colloids and compounds that contribute to color). TSS can cause interference with proper operation and maintenance of the pollution control technologies used by operators to meet effluent limitations and requirements in this general permit. Suspended solids also provide a medium for the transport of other pollutants (e.g., hydrocarbons, metals) via adsorption. The control of TSS in the discharges covered by this general permit will help minimize the discharge of pollutants which are adsorbed to particulate matter. In addition, control of TSS will ensure proper operation of treatment units widely used to meet effluent limitations in this general permit (e.g., carbon adsorption treatment systems can be clogged by TSS).

The RGP establishes effluent limits for TSS that can be reasonably achieved. As indicated above and as included in the RGP, all discharges must meet a monthly average limit for TSS of 30 mg/L. This is sufficiently stringent to achieve the water quality standards of Massachusetts and New Hampshire. The RGP also includes non-numeric limitations based on the Massachusetts narrative water quality standard for solids that require waters to be free from floating, suspended and settleable solids in concentrations that would impair any use assigned to the class or would impair the benthic biota and New Hampshire's narrative standard in Env-Wq 1703.03.

TSS can either affect aquatic life directly by killing them or reducing growth rate or resistance to disease, by preventing the successful development of fish eggs and larvae, by modifying natural movements and migration, and by reducing the abundance of available food (USEPA, 1976). For example, the Biological Assessment for the shortnose sturgeon stated that elevated turbidity, from events including construction, or erosion, can be lethal by clogging the gills of (juvenile) fish (Ross, 1996). It can also impair the ability of juvenile and adult sturgeon when foraging for prey (Peterson, et al., 2000). It should be noted that eggs and larvae are less tolerant of sediment levels than juveniles and adults because successful spawning for both shortnose and Atlantic sturgeon is dependent upon the availability of relatively clean, hard substrate upon which the

eggs can adhere (McCord, n.d.). In addition, as described in Section 4.c.2, below, one of the four essential and biological features for the proposed habitat of the Atlantic sturgeon, specifically in the Piscataqua, Cocheco, and Salmon Falls Rivers, requires hard bottom substrate (e.g., rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (i.e., 0.0 to 0.5 parts per thousand range).

Studies of the effects of turbid water (high sediment concentrations) on fish suggest that concentrations of suspended sediments can reach the thousands of milligrams per liter before an acute toxic reaction is expected (Burton, 1993). The RGP monthly average TSS discharge limit of 30 mg/L is significantly below such a threshold. Based on all of these factors, EPA concludes that the impact of TSS from discharges under the RGP on ESA listed species, including the shortnose sturgeon and the Atlantic sturgeon, will be insignificant and/or discountable and not likely to adversely affect any of the ESA-listed species in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River, or the coastal embayments/nearshore marine waters of Massachusetts and New Hampshire.

Effects Determination for TSS

Based on the best available information, EPA has made the determination that the water quality effects from TSS on shortnose sturgeon or Atlantic sturgeon will be insignificant and/or discountable because:

1) Water quality standards are met at the point of discharge: The ranges of the effects data do not exceed the maximum allowable TSS discharge concentration, 30 mg/L, suggesting that, if this numeric limit is taken to represent surrogate instream constituent exposure concentration, any effects are extremely unlikely to occur and are therefore discountable; and
2) Any increase in turbidity/suspended sediment is minor and temporary such that there is no impairment of movement of individual animals or any other effect that can be meaningfully measured, detected, or evaluated, and effects are therefore insignificant.

With respect to prey quantity/quality, EPA has made the determination that the proposed action will be insignificant because if TSS is present in a discharge authorized under the RGP:

1) Discharges are likely to cause only a minor and temporary reduction in available prey, such that any effects on individual listed species are not capable of being meaningfully measured, detected, or evaluated: Discharges eligible for coverage under this general permit are expected to occur with low frequency (intermittent), small magnitude (small volume limited to no more than 1.0 MGD), and short duration (temporary or short-term) and as such are likely to cause effects minor and temporary in nature. However, if TSS is present in remediation activity discharges, the discharge must meet non-numeric limitations and a numeric technology-based limitation lower than levels that are toxic to benthic communities. Given the high available dilution in the waterbodies in the Action Area, the effect from individual remediation activity discharges, even if at the maximum allowable concentration, 30 mg/L, is not expected to change the instream solids concentration. Fully mixed effluent is likely to cause only a minor and temporary change in the abundance, distribution, quality and availability of only the most sensitive prey

species. This minor and temporary loss or alteration is not expected to affect the way that individual animals use the Action Area or result in behavior change (e.g., foraging) in individual animals that can be meaningfully measured, detected, or evaluated and is therefore insignificant.

Consequently, the proposed action is not likely to adversely affect the shortnose sturgeon, the Atlantic sturgeon or their prey in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River, or the coastal embayments/nearshore marine waters of Massachusetts and New Hampshire.

Cyanide and Metals: Antimony, Arsenic, Cadmium, Chromium III, Chromium VI, Copper, Iron, Lead, Mercury, Nickel, Selenium, Silver, and Zinc

All sites authorized under the RGP are subject to end-of-pipe effluent limitations for all metals included in the RGP. However, the individual metals present in a given remediation activity discharge can vary widely depending on the types of contamination at a site, the activities occurring at a site, and the surficial and bedrock geology present. Petroleum-related sources can contain small quantities of antimony, arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, and zinc, depending upon the type of fuel. Residual metals may also be present at sites with a use history of coal storage, transport or combustion, as antimony, arsenic, cadmium, chromium, lead, mercury, nickel and selenium are constituents of coal, depending upon the source.[26] Potable water used for remediation or dewatering activities may also contain residual metals, depending upon the source water and the treatment processes used (e.g., iron used for coagulation, silver used for disinfection). Water supply piping may also leach metals such as copper or lead into the source. Metals such as copper and nickel can also leach from treatment system piping that contains the metal or alloy (e.g., plumbing pipes, sheet metal, and stainless steel). Operators may also use compounds containing metals, such as copper and iron in treatment systems (e.g., algaecide, and coagulation, respectively). Metals are also common trace impurities in treatment chemicals.

The fate and transport of metals in aquatic systems is highly dependent upon partitioning between soluble and solid phases. Adsorption, precipitation, co-precipitation, and complexation are processes that affect partitioning and adsorption. For example, hydrous metal oxides of iron, aluminum and manganese can remove cations and anions from solution by ion exchange, specific adsorption and surface precipitation. These processes can be highly site-specific, varying by oxygen-reduction potential, the concentration of complexing ions, and the species and concentration of the metal(s) present.[27] Water quality parameters such as hardness, pH, salinity, alkalinity, other metals, and organic carbon can alter the toxicity of metals to aquatic organisms. For example, in saltwater, the acute toxicity of cadmium generally increases as salinity decreases. Also, according to NMFS, metals are more toxic to invertebrate and fish species in water with low hardness than in water with high hardness. Decreasing metal toxicity to fish with increasing water hardness has been shown throughout the literature.[28] All hardness-based metals

[26] See Table 3-4 and 3-5 in EPA 745-B-00-004, 2000: pages 3-11 through 3-28.
[27] Evanko, C.R., et.al. *Remediation of Metals-Contaminated Soils and Groundwater*. Technology Evaluation Report TE-97-01. EPA Technology Innovation and Field Services Division Contaminated Site Clean-Up Information.
[28] See footnote 9, above.

effluent limitations must be calculated based on site-specific hardness in accordance with State water quality standards and applied as end-of-pipe effluent limitations.

In general, metals, such as copper, lead, and zinc, can be directly toxic to aquatic life. Metals can also accumulate in the metabolically-active tissues of aquatic organisms, particularly in benthic feeders such as shortnose and Atlantic sturgeon, which may lead to lethal and sublethal effects including reduced fecundity, body malformation, inability to avoid predation, and susceptibility to infectious organisms (Post, 1987, Alam et al., 2000). Accumulation of metals in living organisms can lead to biomagnification within a food chain. Data suggest that the uptake of contaminants in benthic feeders like sturgeon, and subsequent accumulation in tissues, could occur through water, sediments or food sources (Alam et al., 2000). Exposure to environmentally persistent pollutants such as metals can cause lesions, retard the growth, or impair the reproductive capabilities of aquatic life (Cooper, 1989); (Sindermann, 1994). As stated in the recovery plan for the shortnose sturgeon and the status review for the Atlantic sturgeon, the life history of these species (which includes a long lifespan and benthic foraging habit) predispose the sturgeon to long-term and repeated exposure to environmental contamination (NMFS, 1998); (Atlantic Sturgeon Status Review Team, 2007). Although metals are known to accumulate in the fat tissues of sturgeon, the long term effects are not yet fully known (Ruelle & Henry, 1992).

The metals parameters potentially present in remediation activity discharges, the applicable limitations, and the expected water quality effects on shortnose, Atlantic sturgeon, or their prey, if known, are discussed further with respect to each individual metal, below. Unless otherwise noted, only effects data for surrogate species, potential prey species, or unrelated, but sensitive species were available for EPA's analysis.

Antimony is limited to a maximum of 206 µg/L, regardless of dilution, which will always be the more stringent effluent limitation. Antimony is also limited to a WQBEL of 640 µg/L in Massachusetts and 4.3 mg/L (draft) and 640 µg/L (final) in New Hampshire (due to revision of New Hampshire water quality regulations), equivalent to EPA's organisms-only human health NRWQC, which was retained to meet anti-backsliding requirements. The WQBEL is not expected to apply to discharges, given that the TBEL is more stringent. Antimony often occurs with other metals at sites, particularly lead and zinc. Antimony forms complex ions with organic and inorganic acids, adsorbing strongly to particles that contain iron, manganese, or aluminum.[29] Antimony may also partition to sediment, but low levels are typically found in surface water.

According to NMFS, antimony has low likelihood of bioaccumulation, and an estimated toxic concentration to shortnose sturgeon at 21,900 µg/L. Since Atlantic sturgeon are closely related to shortnose sturgeon, the effects of the proposed action on Atlantic sturgeon are likely similar to effects of the proposed action on shortnose sturgeon. Therefore, EPA considers this estimated toxic concentration to shortnose sturgeon an acceptable surrogate for effects to Atlantic sturgeon. The best available effects information for other sensitive species include a 96-h LC_{50} for fathead minnow reported at 21,900 µg/L (Kimble) and for sheepshead minnow reported at >6200 µg/L.[30]

[29] *Toxicological Profile for Antimony and Compounds*. Agency for Toxic Substances and Disease Registry: September, 1992.
[30] See footnote 9, above.

Additional effects data available in EPA's *Quality Criteria for Water* indicates that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 9,000 µg/L and 1,600 µg/L, respectively.[31] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 1,600 µg/L and 5,400 in fish and daphnid, respectively.[32]

Arsenic is limited to 10 µg/L in freshwater, equivalent to State groundwater quality standards, which is more stringent than EPA's chronic NRWQC for the protection of aquatic life, 150 µg/L. This limitation is imposed near or below analytical minimum levels of detection. Therefore, while human health and/or risk-based effluent limitations are not specifically derived for the protection of aquatic life, such limitations are an appropriate proxy because any potential effects to aquatic life at concentrations that could potentially occur near or below analytical levels of detection cannot be meaningfully measured, detected or evaluated. Arsenic is limited to 36 µg/L in saltwater, equivalent to EPA's chronic aquatic life NRWQC. Arsenic is also limited to a maximum of 104 µg/L, regardless of dilution. Inorganic arsenic occurs primarily in two oxidation states, arsenic III and arsenic V. Arsenic V is more common under oxidizing conditions, while Arsenic III is most common under reducing conditions.[33] Arsenic can adsorb to particulate matter and sediment. Where arsenic forms insoluble complexes with iron, aluminum, and magnesium oxides, and is relatively immobile. Arsenic is more water soluble under reducing conditions.[34] Arsenic can be present in groundwater in New England, including groundwater where the arsenic levels are naturally occurring. In addition, most potable water supplies (i.e., freshwater and occasionally source waters in RGP discharges) have arsenic levels between 2 and 10 µg/L.[35] Discharges to freshwater are limited to 10 µg/L. This suggests that discharges that contain arsenic, even if naturally occurring, will contain concentrations far below the freshwater water quality criterion for the protection of aquatic life.

According to NMFS, arsenic accumulates in organisms, and has an estimated toxic concentration to shortnose sturgeon at 1,921 µg/L. Since Atlantic sturgeon are closely related to shortnose sturgeon, the effects of the proposed action on Atlantic sturgeon are likely similar to effects of the proposed action on shortnose sturgeon. Therefore, EPA considers this estimated toxic concentration to shortnose sturgeon an acceptable surrogate for effects to Atlantic sturgeon. The best available effects information for other, potential prey or sensitive species include: Johnson and Finley (1980) reported a 96-h LC_{50} value of 1,921 µg/L for bluegill exposed to arsenic and Cardin (1980) reported a 96-h LC_{50} value of 14,953 µg/L arsenic for the fourspine stickleback.[36] Additional effects data available in EPA's *Quality Criteria for Water* indicates that for inorganic arsenic(III), acute toxicity values for 16 freshwater species ranged from 812 µg/L for a cladoceran to 97,000 µg/L for a midge, with inorganic arsenic(V) covering about the same range.

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[31] See footnote 25, above.
[32] See footnote 19, above.
[33] Colman, J. *Arsenic and Uranium in Water from Private Wells Completed in Bedrock of East-Central Massachusetts—Concentrations, Correlations with Bedrock Units, and Estimated Probability Maps*. U.S. Geological Survey Scientific Investigations Report 2011–5013: 2011; and Ayotte, J.D., et. al. *Relation of Arsenic, Iron, and Manganese in Ground Water to Aquifer Type, Bedrock Lithogeochemistry, and Land Use in the New England Coastal Basins*.
[34] *Toxicological Profile for Arsenic*. Agency for Toxic Substances and Disease Registry: August, 2007.
[35] Drinking Water Treatment Plant Residuals Management Technical Report. EPA 820-R-11-003, September 2011
[36] See footnote 9, above.

Tests with early life stages appeared to be the most sensitive indicator of arsenic toxicity. Twelve species of saltwater animals have acute values for inorganic arsenic(III) from 232 µg/L to 16,030 µg/L, and two invertebrate values available for inorganic arsenic(V) between 2,000 µg/L and 3,000 µg/L.[37] EPA's *1995 Updates* indicate that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 1,269 µg/L and 3,300 µg/L, respectively for the most sensitive species tested.[38] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 2,962 µg/L and 914.1 µg/L in fish and daphnid, respectively, for arsenic III and 892 µg/L and 450 µg/L in fish and daphnid, respectively, for arsenic V.[39]

Cadmium is limited to 0.25 µg/L for freshwater for sites in Massachusetts and New Hampshire, 8.8 µg/L for saltwater for sites in Massachusetts, and 9.3 µg/L for saltwater for sites in New Hampshire, which are equivalent to the chronic aquatic life water quality criteria in Massachusetts and New Hampshire (at assumed hardness values, which will be adjusted for site-specific hardness, once determined by an applicant). Cadmium is also limited to a maximum of 10.2 µg/L, regardless of dilution.

In surface water, cadmium becomes strongly adsorbed to clays, muds, humic and organic materials and some hydrous oxides (Watson 1973). This complexation tends to remove cadmium from the water column by precipitation (Lawrence et al. 1996), where it is generally not bioavailable except to benthic feeders and bottom dwellers (Callahan et al. 1979; Kramer et al. 1997). Cadmium can occur as a hydrated ion or as ionic complexes with other inorganic or organic substances. Toxic effects are thought to result from the free ionic form of cadmium (Goyer et al. 1989), which causes acute and chronic toxicity in aquatic organisms primarily by disrupting calcium homeostasis and causing oxidative damage. Soluble forms of cadmium migrate in water.[40] In one study comparing the acute toxicity of all 63 atomically stable heavy metals in the periodic table, cadmium was found to be the most acutely toxic metal to the amphipod, *Hyalella azteca*, based on the results of seven-day acute aquatic toxicity tests (Borgmann et al. 2005). Chronic exposure leads to adverse effects on growth, reproduction, immune and endocrine systems, development, and behavior in aquatic organisms (McGeer et al. 2012). According to NMFS, cadmium accumulates at all levels of the food chain, in plants and animals, and an estimated toxic concentration to shortnose sturgeon at 0.38 µg/L. Since Atlantic sturgeon are closely related to shortnose sturgeon, the effects of the proposed action on Atlantic sturgeon are likely similar to effects of the proposed action on shortnose sturgeon. Therefore, EPA considers this estimated toxic concentration to shortnose sturgeon an acceptable surrogate for effects to Atlantic sturgeon. The best available effects information for other surrogate, potential prey, or sensitive species include: Stratus 1999 (in (USEPA 2001) reported a 96-h LC_{50} value of 0.38 µg/L cadmium for the rainbow trout, Cardin (1980) reported a 96-h LC50 value of 577 µg/L cadmium for the Atlantic silverside, and Choi and Kinae (1994) reported 96-h LC_{50} for *Sebastes sp.* exposure to cadmium chloride of approximately 30,000 µg/L.[41] G.W. Suter II and C.

[37] See footnote 25, above.
[38] *1995 Updates: Water Quality Criteria Documents for the Protection of Aquatic Life in Ambient Water*. Environmental Protection Agency: EPA-820-B-96-001; September, 1996.
[39] See footnote 19, above.
[40] *Toxicological Profile for Cadmium*. Agency for Toxic Substances and Disease Registry: September, 2012.
[41] See footnote 9, above.

L. Tsao (1996) reported that the lowest chronic values were 1.7 µg/L and 0.15 µg/L in fish and daphnid, respectively.[42]

Additional effects data available in EPA's *Quality Criteria for Water* indicates that acute toxicity values for 44 freshwater species ranged from 1.0 µg/L for a rainbow trout to 28,000 µg/L for a mayfly, and chronic toxicity values for 12 freshwater fish species and 4 freshwater invertebrate species ranged from 0.15 µg/L for *Daphnia magna* to 156 µg/L for the Atlantic salmon. The antagonistic effect of hardness on acute toxicity has been demonstrated with five species. Saltwater acute values for cadmium and five species of fishes range from 577 µg/L for larval Atlantic silverside to 114,000 µg/L for juvenile mummichog. Acute values for 30 species of invertebrates range from 15.5 µg/L for a mysid to 135,000 µg/L for an oligochaete worm. Two life-cycle tests with *Mysidopsis bahia* under different test conditions resulted in similar chronic values of 8.2 and 7.1 µg/L. A life-cycle test with *Mysidopsis bigelowi* also resulted in a chronic value of 7.1 µg/L [43]

EPA's *1995 Updates* indicates that three chronic toxicity tests have been conducted with the estuarine/marine invertebrate, *Americamysis bahia*, formerly classified as *Mysidopsis bahia*, and one acceptable study was conducted with *Americamysis bigelowi*, formerly classified as *Mysidopsis bigelowi*. No unacceptable effects were observed at cadmium concentrations < 6.4 µg/L and the 96-hr LC_{50} was 15.5 µg/L. Another life-cycle test was conducted with *Americamysis bahia* at a constant temperature of 21°C and salinity of 30 g/kg (Gentile et al. 1982; Lussier et al. 1985). All organisms died in 28 days at 23 µg/L cadmium. A third *Americamysis bahia* chronic study was conducted by Carr et al. (1985) at a salinity of 30 g/kg, but the temperature varied from 14 to 26°C over the 33-day study. At test termination, >50 percent of the organisms had died in cadmium exposures ≥8 µg/L. Gentile et al. (1982) also conducted a life-cycle test with the mysid, *Americamysis bigelowi*, and the results were very similar to those for *Americamysis bahia*.[44]

Chromium III is limited to 74 µg/L in freshwater, which is equivalent to EPA's chronic aquatic life NRWQC (at an assumed hardness value, which will be adjusted for site-specific hardness, once determined by an applicant). Chromium III is limited to 100 µg/L in saltwater, equivalent to State groundwater quality standards. Chromium III is also limited to a maximum of 323 µg/L, regardless of dilution. Chromium III is the most commonly occurring form of chromium in the environment and is largely naturally occurring. Chromium III has very low solubility and low reactivity, resulting in low mobility. Chromium III is insoluble in water. Acid-soluble chromium III complexes in soil may migrate to surface water. Chromium III can also be present as suspended solids adsorbed onto clays, organic matter, or iron oxides.[45]

According to NMFS, chromium has a low likelihood of bioaccumulation, and an estimated toxic concentration to shortnose sturgeon at 3,300 µg/L. Since Atlantic sturgeon are closely related to shortnose sturgeon, the effects of the proposed action on Atlantic sturgeon are likely similar to

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[42] See footnote 19, above.
[43] See footnote 25, above.
[44] See footnote 38, above.
[45] *Toxicological Profile for Chromium.* Agency for Toxic Substances and Disease Registry: September, 2012.

effects of the proposed action on shortnose sturgeon. Therefore, EPA considers this estimated toxic concentration to shortnose sturgeon an acceptable surrogate for effects to Atlantic sturgeon. The best available effects information for other, potential prey or sensitive species includes a 96-h LC_{50} value of 3,330 µg/L for the guppy, reported by Pickering and Henderson (1966). EPA's criteria document reported a 96-h LC_{50} value of 12,400 µg/L for the Atlantic silverside (USEPA 1980a).[46] Additional effects data available in EPA's *Quality Criteria for Water* indicates that for inorganic chromium III, acute toxicity values for 20 freshwater species ranged from 2,221 µg/L for a mayfly to 71,060 µg/L for a caddisfly. In a life-cycle test with Daphnia magna with low hardness, the chronic value was 66 µg/L. In a life-cycle test with the fathead minnow with high hardness, the chronic value was 1,025 µg/L. Two acute values available for chromium (III) in saltwater indicate acute toxicity values of 10,300 µg/L for the eastern oyster and 31,500 µg/L for the mummichog.[47,48] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 68.63 µg/L and <44 µg/L in fish and daphnid, respectively.[49]

Chromium VI is limited to 11 µg/L in freshwater and 50 µg/L in saltwater, equivalent to EPA's chronic aquatic life NRWQC. Chromium VI is also limited to a maximum of 323 µg/L, regardless of dilution. Chromium VI is generally produced by industrial processes and is highly toxic. Available data indicate that the acute toxicity of chromium VI decreases as hardness and pH increase. Common compounds of chromium VI are relatively soluble and mobile. Chromium VI can occur in the soluble state or as suspended solids adsorbed onto clays, organic matter, or iron oxides. Chromium VI is reduced to chromium III by organic matter or other reducing agents in water, and can be reduced through treatment.[50]

Fish exposed to chromium may experience chromosomal aberrations, reduced disease resistance, and morphological changes.[51] Additional effects data available in EPA's *Quality Criteria for Water* indicates that for chromium VI, acute toxicity values for 27 freshwater genera ranged from 23.07 µg/L for a cladoceran to 1,870,000 µg/L for a stonefly. All five tested species of daphnids are especially sensitive. The chronic value indicated for both rainbow trout and brook trout is 264.6 µg/L, while six chronic tests with five species of daphnids indicated chronic values that ranged from <2.5 to 40 µg/L. Twenty-three saltwater vertebrate and invertebrate species had acute values ranging from 2,000 µg/L for a polychaete worm and a mysid to 105,000 µg/L for the mud snail. and The chronic values indicated ranged from <13 to 36.74 µg/L for a polychaete, and 132 µg/L for a mysid.[52,53] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 73.18 µg/L and 6.132 µg/L in fish and daphnid, respectively.[54]

Copper is limited to 9 µg/L in freshwater and 3.1 µg/L in saltwater, which are equivalent to the chronic aquatic life water quality criteria in Massachusetts and New Hampshire (at assumed

[46] See footnote 9, above.
[47] See footnote 25, above.
[48] See footnote 38, above.
[49] See footnote 19, above.
[50] See footnote 45, above.
[51] See footnote 9, above.
[52] See footnote 25, above.
[53] See footnote 38, above.
[54] See footnote 19, above.

hardness values, which will be adjusted for site-specific hardness, once determined by an applicant). Copper is also limited to a maximum of 242 µg/L, regardless of dilution. Copper readily adsorbs to organic matter, clay, soil, or sand and does not easily breakdown. In aerobic sediments, copper can bind to iron oxides. In anaerobic sediments, copper can be reduced to form insoluble salts. Water-soluble copper compounds migrate to groundwater. In water, copper predominantly occurs in the copper II oxidation state, most of which is likely complexed or tightly bound to organic matter. In freshwater, most dissolved copper II occurs as carbonate complexes. Copper II forms compounds or complexes with both inorganic and organic ligands, including ammonia and chloride. Copper also forms stable complexes with organic ligands such as humic acids (e.g., compounds of nitrogen, sulfur or oxygen and hydrogen).[55] In freshwater species, acute toxicity decreases as hardness increases and data for several species indicate that toxicity also decreases with increased alkalinity and total organic carbon.

In high doses, copper contamination can be lethal to shortnose sturgeon, acting as a fish neurotoxin (Gross et al., 2003). Exposure to dissolved copper may impair sensory organs, and contribute to predator avoidance in juvenile fish (Hecht et al., 2007, Sandahl et al., 2007). Flynn and Benfey (2007) identified mortality in their test individuals as a result of copper contamination at 110 µg/L. Besser et al. (2005) identified chronic copper toxicity (i.e. sublethal effects) in rainbow trout and fathead minnows (sturgeon surrogates) at concentrations of 11 to 23 µg/L, respectively. For fathead minnows, growth was inhibited at concentrations of 4.4 µg/L and for rainbow trout growth was inhibited at 12 µg/L. Cardin (1980) reported a 96-h LC50 value of 11.9 µg/L for the summer flounder. According to NMFS, copper has a low likelihood of bioaccumulation in fish, but a higher likelihood in mollusks, and an estimated toxic concentration to shortnose sturgeon at 80 µg/L.[56] Since Atlantic sturgeon are closely related to shortnose sturgeon, the effects of the proposed action on Atlantic sturgeon are likely similar to effects of the proposed action on shortnose sturgeon. Therefore, EPA considers this estimated toxic concentration to shortnose sturgeon an acceptable surrogate for effects to Atlantic sturgeon.

The best available effects information for other, surrogate or sensitive species available in EPA's *Quality Criteria for Water* indicates acute toxicity data available for 41 genera of freshwater species at a hardness of 50 mg/L ranged from 16.74 µg/L for Ptvchocheilus to 10,240 µg/L for Acroneuria. Chronic values available for 15 freshwater species at a hardness of 50 mg/L ranged from 3.873 µg/L for brook trout to 60.36 µg/L for northern pike. Acute values available for saltwater species ranged from 5.8 µg/L for the blue mussel to 600 µg/L for the green crab. A chronic life-cycle test conducted with a mysid indicated that adverse effects were observed at 77 µg/L. EPA's *1995 Updates* indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 2.8 µg/L for the most sensitive species tested at low hardness, a rainbow trout and 23 µg/L for the most sensitive species tested at high hardness, a cladoceran. Chronic toxicity to freshwater aquatic life occurred at 6.2 µg/L for the species tested, a fathead minnow.[57] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 3.8 µg/L and 0.23 µg/L in fish and daphnid, respectively.[58]

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[55] *Toxicological Profile for Copper*. Agency for Toxic Substances and Disease Registry: September, 2004.
[56] See footnote 9, above.
[57] See footnote 38, above.
[58] See footnote 19, above.

Iron is limited to 1,000 µg/L in freshwater, equivalent to EPA's chronic aquatic life NRWQC. Iron is also limited to a maximum of 5,000 µg/L, regardless of dilution. Iron-bearing minerals in rocks and soils (e.g., clays) contribute elevated levels of iron to remediation activity discharges composed of groundwater.[59] Iron most commonly occurs as the ferrous (Fe^{2+}) and ferric (Fe^{3}) iron ions. These ions readily combine with oxygen- and sulfur-containing compounds to form oxides, hydroxides, carbonates, and sulfides. Fe^{2+} (iron salts) are relatively unstable and precipitate as insoluble Fe^{3+} (iron hydroxide). Fe^{3+} settles out of the water column as a rust-colored silt.

The smothering effects of settled iron precipitates may be detrimental to fish eggs and bottom-dwelling fish prey organisms. Fe^{2+} can persist in waters absent dissolved oxygen, but can precipitate when exposed to adequate oxygen (i.e., clear water iron).[60] Elevated levels of iron can promote undesirable bacterial growth, which produce a filamentous, slimy deposit that can clog treatment and plumbing components (i.e., fouling).[61] Iron bacteria include a number of organisms that obtain carbon from the carbon dioxide (CO_2) in the air and obtain energy from dissolved iron or manganese. Iron bacteria are small, approximately 12 microns (i.e., one millionth of a meter) wide and 315 microns long. Species of iron bacteria include: *Sphaertilus*, *Clonothrix*, *Crenothrix*, and *Leptothrix*. Iron bacteria occur naturally in the soil and thrive when there is adequate food (i.e., iron and/or manganese) and CO_2.[62] Alam et al. (2000) indicate that Gulf sturgeon from the Suwannee River (a threatened species) tend to accumulate iron in their blood, although the direct toxicity of iron is unknown (Vuorinen, 1999).

Additional effects data available in EPA's *Quality Criteria for Water* indicates trout and other fish were not observed in the field until iron in a contaminated stream precipitated to effect a concentration of less than 1.0 mg/L even though other water quality constituents measured were suitable for the presence of trout. Warnick and Bell (1969) obtained 96-hour LC_{50} values of 0.32 mg/L for mayflies, stonef1ies, and caddisflies. Brandt (1948) found iron toxic to carp, *Cyprinus carpio*, at 0.9 mg/L when the pH of the water was 5.5. Pike, *Esox 1ucius*, and trout (species not known) experienced mortality at iron concentrations of 1 to 2 mg/L (Doudoroff and Katz, 1953).[63] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 1,300 µg/L and 158 µg/L in fish and daphnid, respectively.[64]

Lead is limited to 2.5 µg/L in freshwater and 8.1 µg/L in saltwater, which are equivalent to EPA's chronic aquatic life NRWQC (at assumed hardness values, which will be adjusted for site-specific hardness, once determined by an applicant). Lead is also limited to a maximum of 160 µg/L, regardless of dilution. Lead most commonly occurs in the oxidation state Pb^{2+}. Lead does not breakdown, but may transform to other lead compounds. When lead is exposed to air

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[59] DeSimone, L.A., et. al. *Quality of Water from Domestic Wells in Principal Aquifers of the United States, 1991–2004*. U.S. Geological Survey Scientific Investigations Report 2008–5227: 2009.
[60] See footnote 25, above.
[61] *Health criteria and other supporting information.* World Health Organization; Guidelines for Drinking-Water Quality Second ed. Vol. 2: 1996.
[62] *Iron Bacteria in Drinking Water*. NHDES Environmental Fact Sheet WD-DWGB-3-21: 2010.
[63] *Quality Criteria for Water.* Environmental Protection Agency: EPA 440-9-76-023; 1976. (EPA's "Red Book")
[64] See footnote 19, above.

and water, films of lead sulfate, lead oxides, and lead carbonates form, creating a protective barrier that slows or halts corrosion. Lead also strongly adsorbs to sediment. As a result, lead is most commonly found in the upper layers of sediment. The solubility of lead compounds in water is a function of pH, hardness, salinity, and the presence of humic material. Solubility is highest in low hardness, low pH water. The acute toxicity of lead to several species of freshwater animals has been shown to decrease as the hardness of water increases.[65]

Fish exposed to high levels of lead exhibit a wide-range of effects, including muscular and neurological degeneration and destruction, growth inhibition, mortality, reproductive problems, and paralysis (USEPA 1980b, Eisler 1988b). Alam et al. (2000) indicate that Gulf sturgeon from the Suwannee River (a threatened species) tend to accumulate lead in their blood (Vuorinen, 1999). Holcombe et al. (1976) exposed brook trout (a commonly used surrogate species for shortnose sturgeon in whole effluent toxicity testing) to 235 µg/L of lead for 20 weeks. Results indicate that metal accumulation occurred mostly in the gills, liver and kidneys and may reduce survival and impair reproduction and growth. Lead may also accumulate in hard tissues such as bones, skin and scales (Patterson and Settle, 1976). According to NMFS, lead accumulates at all levels of the food chain, in plants and animals, but does not biomagnify, and has an estimated toxic concentration to shortnose sturgeon at 1,170 µg/L. Since Atlantic sturgeon are closely related to shortnose sturgeon, the effects of the proposed action on Atlantic sturgeon are likely similar to effects of the proposed action on shortnose sturgeon. Therefore, EPA considers this estimated toxic concentration to shortnose sturgeon an acceptable surrogate for effects to Atlantic sturgeon. The best available effects information for other surrogate or sensitive species includes a 96-h LC_{50} value of 1,170 µg/L reported for the rainbow trout (Goettl 1972, Davies and Everhart 1973, Davies and al. 1976), and a 96-h LC_{50} value of 315 µg/L for the mummichog, reported by Dorfman (1977). For rockcod exposed to lead over 24, 48, 72, and 96 hour exposures, the LC_{50}s reported were 42 µg/L,500 µg/L, 22,500 µg/L, 19,000 µg/L and 17,000 µg/L, respectively (Siammai and Chiayvareesajja 1988).[66]

Additional effects data available in EPA's *Quality Criteria for Water* indicates that at a hardness of 50 mg/L the acute sensitivities of 10 freshwater species range from 142.5 µg/L for an amphipod to 235,900 µg/L for a midge. Available chronic effects data ranged from 12.26 µg/L to 128.l µg/L, both for a cladoceran, but in water with low hardness and high hardness, respectively. Acute values available for 13 saltwater species ranged from 315 µg/L for the mummichog to 27,000 µg/L for the soft shell clam. A chronic toxicity test was conducted with a mysid; unacceptable effects were observed at 37 µg/L.[67] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 18.88 µg/L, 12.26 µg/L, and 25.46 in fish, daphnid, and non-daphnid invertebrates, respectively.[68]

Mercury is limited to 0.77 µg/L in freshwater and 0.94 µg/L in saltwater, which are equivalent to EPA's chronic aquatic life NRWQC (at assumed hardness values, which will be adjusted for site-specific hardness, once determined by an applicant). Mercury is also limited to a maximum

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[65] *Toxicological Profile for Lead*. Agency for Toxic Substances and Disease Registry: August, 2007.
[66] See footnote 9, above.
[67] See footnote 25, above.
[68] See footnote 19, above.

of 0.739 µg/L, regardless of dilution. Mercury can occur in several forms, including elemental mercury, inorganic mercury, and organic mercury. Inorganic mercury compounds form with elements such as chlorine, sulfur, or oxygen (i.e., mercury salts). Organic mercury compounds form with carbon. The most common organic mercury compound is methylmercury, produced mainly by microorganisms in water and soil that convert inorganic mercury compounds.[69]

Mercury toxicity is greatly influenced by mercury form, with organic forms (i.e., methyl mercury, phenyl mercury) being more toxic than inorganic mercury due to the greater biological availability of organic forms (Sorensen 1991). Multigenerational exposures of early life stage brook trout to methyl mercuric chloride at concentrations as low as 0.96 µg/L resulted in absence of spawning in second generation fish. Other reported effects include deformities and expression of neurological effects as muscle twitching (McKim et al. 1976). Exposure to inorganic mercury at concentrations as low as 20 µg/L resulted in reduced hatchability, increased deformities and embryo death (Heisinger and Green 1975, Weis and Weis 1977). Mercury also adversely effects growth, behavior, metabolism, blood chemistry, osmoregulation, and oxygen exchange (Weis and Khan 1990, Sorensen 1991). Juveniles are more susceptible than adults. Larval or juvenile fish exposed to elevated concentrations of mercury show larval mortality, developmental abnormalities, and reduced larval growth. In saltwater, fishes tend to be more resistant and mollusks and crustaceans tend to be more sensitive to the acute toxic effects of mercury II. Mercury also exhibits a high potential for bioaccumulation and biomagnification, with reported mercury concentrations in fish up to 100,000 times the ambient water concentrations (Sorensen 1991). According to NMFS, mercury accumulates and magnifies at all levels of the aquatic food chain. The best available effects information available were for other sensitive species. Hansen (1983) reported a 96-h LC_{50} value of 36 µg/L mercury for the juvenile spot. *Sebastes schlegeli* was exposed to mercury chloride for up to 96-h and resulted in 48-h, 72-h, and 96-h LC_{50}s of less than 100 µg/L (Choi and Kinae 1994).[70] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were <0.23 µg/L and 0.96 µg/L in fish and daphnid, respectively for inorganic or total mercury.[71]

Additional effects data for other potential prey, or sensitive species were available in EPA's *Quality Criteria for Water*, which indicates available acute toxicity data for mercury II to 28 freshwater genera of freshwater animals. For invertebrate species, acute toxicity ranged from 2.2 µg/L for *Daphnia pulex* to 2,000 µg/L for three insects. For fishes, acute toxicity ranged from 30 µg/L for the guppy to 1,000 µg/L for the Mozambique tilapia. Available chronic effects data indicate that methylmercury is the most chronically toxic of the tested mercury compounds. Chronic values for methylmercury with *Daphnia magna* and brook trout were less than 0.07 µg/L. A chronic value for mercury II with *Daphnia magna* was approximately 1.1 µg/L. In both a life-cycle test and an early life-stage test for mercuric chloride with the fathead minnow, the chronic value was less than 0.26 µg/L. Acute toxicity effects of mercuric chloride available for 29 genera of saltwater animals, including annelids, mollusks, crustaceans, echinoderms, and fishes ranged from 3.5 µg/L for a mysid to 1,678 µg/L for winter flounder. Concentrations of mercury that affected growth and photosynthetic activity of one saltwater diatom and six species

[69] *Toxicological Profile for Mercury*. Agency for Toxic Substances and Disease Registry: March, 1999.
[70] See footnote 9, above.
[71] See footnote 19, above.

of brown algae ranged from 10 µg/L to 160 µg/L.[72] EPA's *1995 Updates* indicate that acute toxicity to freshwater aquatic life occurs at concentrations at a species mean low of 2.9 µg/L for the most sensitive species tested, a cladoceran.[73]

Nickel is limited to 52 µg/L in freshwater and 8.2 µg/L in saltwater, which are equivalent to EPA's chronic aquatic life NRWQC (at assumed hardness values, which will be adjusted for site-specific hardness, once determined by an applicant). Nickel is also limited to a maximum of 1,450 µg/L, regardless of dilution. Nickel typically combines with sulfur to form sulfides under anaerobic conditions. In soil, nickel typically combines with oxygen to form oxides. Nickel strongly adsorbs to solids containing iron or manganese to form amorphous oxides. Nickel also adsorbs onto suspended particles, particulate matter and dissolved organic matter. Nickel compounds containing chlorine, sulfur, and oxygen are relatively water-soluble. Under acidic conditions, nickel is mobile in soil and will leach to groundwater.[74]

According to NMFS, nickel has a low likelihood of bioaccumulation, and an estimated toxic concentration to shortnose sturgeon at 2,480 µg/L. Since Atlantic sturgeon are closely related to shortnose sturgeon, the effects of the proposed action on Atlantic sturgeon are likely similar to effects of the proposed action on shortnose sturgeon. Therefore, EPA considers this estimated toxic concentration to shortnose sturgeon an acceptable surrogate for effects to Atlantic sturgeon. The best available effects information for other sensitive species includes a 96-h LC_{50} value of 2,480 µg/L nickel for the rock bass, reported by Lind et al. (1986).[75] Additional effects data available in EPA's *1995 Updates* indicate that acute toxicity to freshwater aquatic life occurs at concentrations as low as 239 µg/L for the most sensitive species tested at low hardness (26 mg/L), a snail. The species mean acute value reported for the fathead minnow was 6,707 µg/L at 50 mg/L hardness.[76] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were <35 µg/L, <5 µg/L, and 128.4 in fish, daphnid, and non-daphnid invertebrates, respectively.[77]

Selenium is limited to 5.0 µg/L in freshwater and 71 µg/L in saltwater, which are equivalent to the chronic aquatic life water quality criteria in Massachusetts and New Hampshire. Selenium is also limited to a maximum of 235.8 µg/L, regardless of dilution. Selenium exists in four oxidation states (VI, IV, 0, -II) and in a wide range of chemical and physical species across these oxidation states (Doblin et al. 2006; Maher et al. 2010; Meseck and Cutter 2006). Selenium generally occurs in combination with sulfide or with silver, copper, lead, and nickel minerals. The occurrence of selenium is influenced by its oxidation state. The forms of selenium generally found in surface water and the water contained in soils are the salts of selenic and selenious acids. Soluble and mobile forms of selenium (e.g., selenite and selenate) are dominant under aerobic and alkaline conditions. Insoluble forms of selenium can settle to the bottom as solids.[78]

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[72] See footnote 25, above.
[73] See footnote 38, above.
[74] *Toxicological Profile for Nickel*. Agency for Toxic Substances and Disease Registry: August, 2005.
[75] See footnote 9, above.
[76] See footnote 38, above.
[77] See footnote 19, above.
[78] *Toxicological Profile for Selenium.* Agency for Toxic Substances and Disease Registry: September, 2003.

The bioavailability and toxicity of selenium depend on both its concentration and speciation (Cutter and Cutter 2004; Meseck and Cutter 2006; Reidel et al. 1996).

Excess concentrations of selenium that are only an order of magnitude greater than the required level have been shown to be toxic to fish, apparently due to generation of reactive oxidized species, resulting in oxidative stress (Palace et.al. 2004). Dietary requirements in fish have been reported to range from 0.05 to 1.0 mg/kg (Watanabe et al. 1997). A variety of lethal and sublethal deformities can occur in the developing fish exposed to selenium, affecting both hard and soft tissues (Lemly 1993b). Because the most sensitive adverse effects of selenium are reproductive effects (larval deformities and mortality) on the offspring of exposed fish, chronic effects from long term exposure are possible.[79] According to NMFS, selenium accumulates in the aquatic food chain, and has an estimated toxic concentration to shortnose sturgeon at 1,325 µg/L. Since Atlantic sturgeon are closely related to shortnose sturgeon, the effects of the proposed action on Atlantic sturgeon are likely similar to effects of the proposed action on shortnose sturgeon. Therefore, EPA considers this estimated toxic concentration to shortnose sturgeon an acceptable surrogate for effects to Atlantic sturgeon. De Riu et al. (2014) suggests that white sturgeon are less sensitive to selenium than the threatened green sturgeon. The best available effects information for other sensitive species includes a 96-h LC_{50} value of 1,325 µg/L selenium for the striped bass, reported by Palawski et al. (1985). EPA aquatic life water quality criteria documents report a 96-h LC_{50} values of 599 µg/L selenium for the haddock (USEPA 2004).[80] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 88.32 µg/L and 91.65 µg/L in fish and daphnid, respectively.[81]

Silver is limited to 3.2 µg/L in freshwater and 1.9 µg/L in saltwater, which are equivalent to EPA's chronic aquatic life NRWQC (at assumed hardness values, which will be adjusted for site-specific hardness, once determined by an applicant). Silver is also limited to a maximum of 35.1 µg/L, regardless of dilution. Silver can occur as the monovalent ion (e.g., sulfide, bicarbonate, or sulfate salts), or as part of more complex ions with chlorides and sulfates. Silver occurs primarily as sulfides, in association with metals such as iron and lead. Silver also combines with chloride and nitrate. Silver adsorbs onto particulate matter, the dominant process controlling partitioning in water.[82]

According to NMFS, silver accumulates to a limited extent in algae, mussels, clams, and other aquatic organisms, and has an estimated toxic concentration to shortnose sturgeon at 4 µg/L. Since Atlantic sturgeon are closely related to shortnose sturgeon, the effects of the proposed action on Atlantic sturgeon are likely similar to effects of the proposed action on shortnose sturgeon. Therefore, EPA considers this estimated toxic concentration to shortnose sturgeon an acceptable surrogate for effects to Atlantic sturgeon. The best available effects information for other potential prey, or sensitive species includes a 96-h LC_{50} value of 4.7 µg/L silver (USEPA

[79] *Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater 2016*. Environmental Protection Agency: EPA 822-R-16-006; June 2016.
[80] See footnote 9, above.
[81] See footnote 19, above.
[82] *Toxicological Profile for Silver*. Agency for Toxic Substances and Disease Registry: December, 1990.

1987) for the haddock, reported by Goettl and Davies (1978).[83] EPA's *Quality Criteria for Water* indicates that a concentration of 70 µg/L was 1ethally toxic to bass (Coleman and Clearly, 1974). Data compiled by Doudoroff and Katz (1953) show that lethality to sticklebacks occurred at 20 µg/L of silver nitrate in two days. Anderson (1948) reported that the toxic threshold of silver nitrate for the stickleback, *Gasterosteus aculeatus* was 3.0 µg/L as the ion silver. In saltwater, Calabrese et al. (1973) reported a 48-hour LC_{50} of 5.8 µg/L as the silver ion for oyster larvae, *Crassostrea virginica*, and a 48-hour LC_{50} of 21.0 µg/L as the silver ion for larvae of the hard-shell clam, *Mercenaria mercenaria*.[84] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 0.12 µg/L and 2.6 µg/L in fish and daphnid, respectively.[85]

Zinc is limited to 120 µg/L in freshwater and 81 µg/L in saltwater, which are equivalent to EPA's chronic aquatic life NRWQC (at assumed hardness values, which will be adjusted for site-specific hardness, once determined by an applicant). Zinc is also limited to a maximum of 420 µg/L, regardless of dilution. Zinc occurs mainly as a free ion (i.e., Zn^{2+}) and can occur in both suspended and dissolved forms. Suspended zinc can dissolve and can readily adsorb onto suspended solids. Dissolved zinc generally increases as pH decreases and may occur as the free ion or as dissolved complexes and compounds. Under aerobic conditions and at high pH, zinc readily adsorbs onto hydrous iron and manganese oxides, clay minerals, and organic material. Zinc compounds commonly found at contaminated or formerly contaminated sites include zinc chloride, zinc oxide, zinc sulfate, and zinc sulfide.[86]

Zinc may contribute to endocrine disruption, and specifically to reproductive alterations in fish including decreased vitellogenin levels, delayed spawning, decreased egg viability, impaired spermatogenesis, increased oocyte atresia, reduced egg size and larval deformities at elevated levels (Gross et al., 2003). According to NMFS, zinc has a low likelihood of bioaccumulation. The best available effects information available were for other surrogate, potential prey, or sensitive species. Choi and Kinae (1994) reported a 72-h LC_{50} for exposure to zinc of greater than 10,000 µg/L.[87] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 36.41 µg/L, 46.73 µg/L, and >5,243 in fish, daphnid, and non-daphnid invertebrates, respectively.[88]

Additional effects data available in EPA's *Quality Criteria for Water* indicates a 54 percent mortality of rainbow trout fry in a zinc concentration of 10 µg/L for 28 days using dilution water with a calcium concentration of 1.7 mg/L and a magnesium concentration of 1.0 mg/L (Affleck (1952)). Pickering and Henderson (1966) determined a 96-hour LC_{50} of zinc for fathead minnows, *Pimephales Prome1as*, of 870 µg/L at 20 mg/L $CaCO_3$ and 33,000 µg/L at 360 mg/L $CaCO_3$. The Atlantic salmon, *Salmo salar*, was tested in a 168-hour continuous-flow bioassay at 17°C in water with a total hardness of 14 mg/L $CaCO_3$. The incipient lethal level, the level beyond which the organism can no longer survive, was 420 µg/L of zinc (Sprague and Ramsay,

[83] See footnote 9, above.
[84] See footnote 63, above.
[85] See footnote 19, above.
[86] *Toxicological Profile for Zinc*. Agency for Toxic Substances and Disease Registry: August, 2005.
[87] See footnote 9, above.
[88] See footnote 19, above.

1965). Wurtz (1962) determined a 96-hour LC_{50} for young pond snails, *Physa heterostropha*, of 434 µg/L at 100 mg/L $CaCO_3$ and of 303 µg/L at 20 mg/L $CaCO_3$. The LC_{50} of a zinc sulfate solution to a mayfly, *Ephemerella subvaria*, in a 10-day test to was 16,000 µg/L at 44 mg/L $CaCO_3$ (Warnick and Bell, 1969). A 48-hour LC_{50} for *Daphnia magna* was found to be 100 µg/L at a hardness of 45 mg/L $CaCO_3$ and an alkalinity of 42 mg/L (Biesinger and Christensen, 1972).[89]

Cyanide

Cyanide is limited to 5.2 µg/L in freshwater and 1.0 µg/L in saltwater, equivalent to EPA's chronic National Recommended Water Quality Criteria for the protection of aquatic life. EPA's cyanide criteria are stated in terms of free cyanide, defined as the sum of the cyanide present as HCN and CN^-. Free cyanide is considered a more reliable index of toxicity to aquatic life than total cyanide because total cyanides can include organic cyanides (e.g., nitriles) and relatively stable metallocyanide complexes. However, current EPA approved test methods are only available for total cyanide and available cyanide in water and not sufficiently sensitive to measure concentrations of cyanide as low as 1.0 µg/L. As a result, the draft RGP specifies that the WQBEL is shown as free cyanide per liter. However, total cyanide must be reported. The compliance level for total cyanide is 5 µg/L. Cyanide is also limited to a maximum of 178 mg/L, regardless of dilution (as total cyanide).

Cyanide is strongly associated with metals at contaminated or formerly contaminated sites because it readily forms complexes with transition metals, particularly iron. Cyanide occurs in water in many forms, including hydrogen cyanide (HCN), the cyanide ion (CN^-), simple cyanides, metallocyanide complexes, and as organic compounds. The relative concentrations of these forms depend mainly on pH and temperature. Both HCN and CN^- are toxic to aquatic life. The cyanide ion readily converts to hydrogen cyanide at pH values less than 7.0. As a result, when present, cyanide occurs more commonly as the more toxic hydrogen cyanide. Certain bacteria, fungi, and algae can also produce cyanide, and cyanide is found naturally in several species of plants.[90] Cyanide is soluble in water. Sensitive fish species are highly susceptible to cyanide exposure, exhibiting lethal effects at concentrations as low as 20 µg/L to 76 µg/L (Eisler 1991). Sub-lethal effects include reduced reproductive capacity (decreased egg number and viability, and reduced embryo and larval survival), impaired swimming ability, altered growth, and hepatic necrosis (dead liver tissue) (Eisler 1991). According to NMFS, cyanide has no bioaccumulation potential, and an estimated toxic concentration to shortnose sturgeon at 40 µg/L. Since Atlantic sturgeon are closely related to shortnose sturgeon, the effects of the proposed action on Atlantic sturgeon are likely similar to effects of the proposed action on shortnose sturgeon. Therefore, EPA considers this estimated toxic concentration to shortnose sturgeon an acceptable surrogate for effects to Atlantic sturgeon. The best available effects information for other surrogate, or sensitive species include a 96-h LC_{50} of 40 µg/L cyanide for the rainbow trout, reported by Kovacs (1979), and Kovacs and Leduc (1982) (1982b), and a 96-h LC_{50} value of 59 µg/L cyanide for the Atlantic silverside, reported by Gardner and Berry (1981).[91]

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[89] See footnote 63, above.
[90] *Toxicological Profile for Cyanide*. Agency for Toxic Substances and Disease Registry: July, 2006.
[91] See footnote 9, above.

Additional effects data available in EPA's *Quality Criteria for Water* indicates the acute toxicity of free cyanide ranged from 44.73 µg/L for a rainbow trout to 2,490 µg/L for a midge, but all of the species with acute sensitivities above 400 µg/L were invertebrates. A long-term survival, and a partial and complete life-cycle test with fish yielded chronic values of 13.57 µg/L, 7.849 µg/L, and 16.39 µg/L, respectively. Chronic values for two freshwater invertebrate species were 18.33 µg/L and 34.06 µg/L. Chronic values for two freshwater invertebrate species were 18.33 µg/L and 34.06 µg/L. The acute toxicity of free cyanide to saltwater species ranged from 4.893 µg/L to >10,000 µg/L. Long-term survival in an early life-stage test with the sheepshead minnow yielded a chronic value of 36.12 µg/L. Long-term survival in a mysid life-cycle test resulted in a chronic value of 69.71 µg/L.[92]

Effects Determination for Cyanide and Metals

Based on the best available information, EPA has made the determination that the water quality effects from the thirteen aforementioned metals and cyanide on shortnose sturgeon or Atlantic sturgeon will be insignificant and/or discountable because if one or more of these metals is present in a discharge authorized under the RGP:

1) Water quality standards are met at the point of discharge: The numeric water quality-based limits for cyanide and metals have been established at the chronic aquatic life criteria adopted in each state, are more stringent than available estimated effects data to the listed species, and will be adjusted for site-specific hardness in freshwater. The cyanide WQBELs are imposed at concentrations near or below the minimum levels of detection. Further, given high dilution in the Action Area waterbodies, the maximum allowable discharge concentrations will result in the use of only a small portion of the available assimilative capacity of the nearshore marine waters of Massachusetts and New Hampshire or in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River such that cumulative effects from the environmental persistence of metals are extremely unlikely to occur. Therefore, the numeric limits will not adversely affect listed species because effects are extremely unlikely to occur and are therefore discountable.

With respect to prey quantity/quality, EPA has made the determination that the proposed action will be insignificant because if one or more of these metals is present in a discharge authorized under the RGP:

1) Discharges are likely to cause only a minor and temporary reduction in available prey, if any, such that any effects on individual animals are not capable of being meaningfully measured, detected, or evaluated: Discharges eligible for coverage under this general permit are expected to occur with low frequency (intermittent), small magnitude (small volume limited to no more than 1.0 MGD), and short duration (temporary or short-term) and as such are likely to cause effects minor and temporary in nature. However, if any of these pollutants are present in remediation activity discharges, the discharge must meet numeric water quality-based limits established at the chronic aquatic life criteria adopted in each state, at concentrations below available estimated effects concentrations at end-

[92] See footnote 63, above.

of-pipe. In addition, these pollutants are expected to undergo rapid, full mixing because of the high dilution in the receiving waters to concentrations less than the minimum level of detection such that effects are likely to cause only a minor and temporary change in the abundance, distribution, quality and availability of only the most sensitive prey species. This minor and temporary loss or alteration is not expected to affect the way that individual animals use the Action Area or result in behavior change (e.g., foraging) in individual animals that can be meaningfully measured, detected, or evaluated and is therefore insignificant.

Consequently, the proposed action is not likely to adversely affect the shortnose sturgeon, the Atlantic sturgeon or their prey in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River, or the coastal embayments/nearshore marine waters of Massachusetts and New Hampshire.

Non-Halogenated and Halogenated Volatile Organic Compounds (VOCs)

VOCs are organic compounds that participate in atmospheric photochemical reactions except those designated by EPA as having negligible photochemical reactivity. A halogenated compound is one that has a halogen (e.g., fluorine, chlorine, bromine, or iodine) attached to its chemical structure. In general, VOCs undergo rapid volatilization to the atmosphere when released to surface water, attributable to the relatively high vapor pressure and relatively low aqueous solubility of low molecular weight organic compounds (Dilling 1977; Dilling et al. 1975). The less halogenated the compound (i.e., the lower the number of halogens attached to its chemical structure), the more rapidly the compound degrades.[93]

The non-halogenated and halogenated VOC parameters potentially present in remediation activity discharges and the expected water quality effects on shortnose, Atlantic sturgeon, or their prey, if known, are discussed in this section. EPA's determination follows this information and is made with respect to all of the non-halogenated and halogenated VOC parameters potentially present in remediation activity discharges. Numeric effluent limitations for the majority of these parameters are equivalent to human health- or risk-based water quality criteria such as EPA's human health NRWQC and State-adopted groundwater quality standards, which are imposed near or below analytical minimum levels of detection. Therefore, while human health and/or risk-based effluent limitations are not specifically derived for the protection of aquatic life, such limitations are an appropriate proxy because any potential effects to aquatic life at concentrations that could potentially occur near or below analytical levels of detection cannot be meaningfully measured, detected or evaluated.

BETX: Benzene, Ethylbenzene, Toluene and Total Xylenes (BETX)

Total BETX is the sum of the four alkyl benzenes: benzene, toluene, ethylbenzene, and total xylenes (i.e., the sum of the ortho, para, and meta isomers of xylene). Total BETX is limited to a maximum of 100 µg/L. EPA NRWQC are not available for this parameter. One pollutant that comprises this parameter, benzene, is also limited as an individual parameter in the RGP.

[93] *Remediation Technologies Screening Matrix and Reference Guide, Version 4.0, Section 2.4.1: Properties and Behavior of Halogenated VOCs* (2007).

These VOCs have relatively high vapor pressures and high Henry's law constants such that they have a strong tendency to partition from water into air (Mackay 1979; Masten et al. 1994) and volatilization is expected to be the dominant transport mechanism for xylenes in surface water. do not significantly bioaccumulate in aquatic food chains. For example, the volatilization half-life of ethylbenzene has been measured from approximately 40 to 200 hours (Keefe et al. 2004),[94] toluene with a half-life on the order of a few hours at 25°C,[95] and o-xylene is reported to be 5.6 hours from a surface water depth of 1 meter (Mackay and Leinonen 1975). These volatilization rates vary with conditions in the surface water, such as current/turbulence, water depth and surface conditions (e.g., wind).[96] Under aerobic conditions, when mixtures of BETX are present, toluene usually degrades first, followed by xylene, and lastly benzene and ethylbenzene, if they are degraded at all.[97] BETX compounds are present at relatively high concentrations in light distillates (e.g., approximately 2% ethylbenzene, 5% benzene, and 11-12% toluene and xylenes). However, the composition of petroleum products that contain BETX is highly variable, and for some petroleum products, any one of the four BETX compounds could be the dominant COC. BETX concentrations decrease in the heavier grades of petroleum distillate products such as fuel oils.[98]

The best available effects information available were for other sensitive species. Marchini et al. (1992) reported a 96-h LC_{50} of 24.6 mg/L for benzene in juvenile fathead minnow, and 15.6 mg/L in larvae. Geiger et al. (1986) reported a 96-h LC_{50} of 9.09 mg/L for ethylbenzene in juvenile fathead minnow. Marchini et al. (1992) reported a 96-h LC_{50} of 36.2 mg/L for toluene in juvenile fathead minnow, and 17.0 mg/L in larvae. Geiger et al. (1986) reported a 96-h LC_{50} of 16.0 mg/L for juvenile fathead minnow for xylenes.[99] Additional effects data available in EPA's *Quality Criteria for Water* indicates acute toxicity to freshwater aquatic life occurs at concentrations as low as 32,000 µg/L and acute toxicity to saltwater aquatic life occurs at concentrations as low as 430 µg/L for ethylbenzene. [100] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 134 µg/L in fish for benzene, >440 µg/L and 12,922 µg/L in fish and daphnid, respectively, for ethylbenzene, 1,269 µg/L and 25,229 µg/L in fish and daphnid, respectively, for toluene, and 62,308 µg/L in fish for xylenes.[101]

Benzene is limited to a maximum of 5.0 µg/L. EPA aquatic life NRWQC are not available for this parameter. This effluent limitation is equivalent to State groundwater quality standards. This effluent limitation is also more stringent than the most current human health NRWQC for the consumption of organism-only ("organism-only"), 16 to 58 µg/L. Benzene is frequently found at petroleum-related remediation sites because benzene is present at relatively high concentrations

[94] *Toxicological Profile for Ethylbenzene*. Agency for Toxic Substances and Disease Registry: November, 2010.
[95] *Draft Toxicological Profile for Toluene*. Agency for Toxic Substances and Disease Registry: September, 2015.
[96] *Toxicological Profile for Xylene*. Agency for Toxic Substances and Disease Registry: August, 2007.
[97] *Toxicological Profile for Benzene*. Agency for Toxic Substances and Disease Registry: August 2007.
[98] *Composition of Petroleum Mixtures*, Total Petroleum Hydrocarbon Criteria Working Group Series, T.L. Potter and K.E. Simmons, Vol. 2, p. 52, May 1998.
[99] See footnote 9, above.
[100] See footnote 63, above.
[101] See footnote 19, above.

in light distillates (e.g., approximately 20,000 parts per million (ppm) in gasoline and approximately 300 ppm in diesel fuel).[102]

The high volatility of benzene is the controlling physical property in its environmental fate and transport. Benzene is considered highly volatile with a vapor pressure of 95.2 mm Hg at 25 °C. Benzene has moderate solubility in water, 1,780 mg/L at 25°C. The Henry's law constant for benzene, 5.5x10-3 atm-m3/mole at 25 °C, indicates that benzene partitions readily to the atmosphere from surface water (Mackay and Leinonen 1975). Benzene can adsorb to solids, which tends to occur with greater organic matter content. Benzene can be degraded in water, mostly through aerobic biodegradation. Benzene can be resistant to aerobic biodegradation in the presence of nutrients or when present as a mixture with other aromatic hydrocarbons. Benzene biodegradation under anaerobic conditions does not readily occur.[103]

The best available effects information available were for other sensitive species. EPA's *Quality Criteria for Water* indicates that acute toxicity to freshwater aquatic life occurs at concentrations as low as 5,300 µg/L and acute toxicity to saltwater aquatic life occurs at concentrations as low as 5,100 µg/L. Chronic toxicity of benzene to sensitive saltwater aquatic life has been found to occur at concentrations as low as 700 µg/L with a sensitive fish species.[104]

1,4-dioxane

1,4-dioxane is limited to a maximum of 200 µg/L. EPA NRWQC are not available for this parameter. This effluent limitation is equivalent to EPA's lifetime health advisory under the Safe Drinking Water Act. 1,4-dioxane is a synthetic cyclic ether generally released during its production and use, the processing of other chemicals, and with its unintentional formation during the manufacture of ethoxylated surfactants (EC 2002). Historically, 1,4-dioxane was released used as a stabilizer for 1,1,1-trichloroethane (TCA). As a result, 1,4-dioxane is frequently found at sites in association with releases of chlorinated solvents, especially1,1,1-TCA.[105] The potential for bioconcentration in aquatic organisms is low (Franke et al. 1994). 1,4-dioxane adsorbs weakly to suspended sediments and is relatively resistant to biodegradation (Kawasaki 1980; Lyman et al. 1982). [106] 1,4-dioxane is highly miscible in water, mixing with water so readily that it can be found in groundwater plumes far in advance of any solvents with which it was originally released. [107]

Given its estimated Henry's law constant of 4.88 x10-6 atm·m3 mol-1 (Howard 1990), 1,4-dioxane is expected to be moderately volatile from water surfaces, as well as moist soils (Park et al., 1987; Thomas, 1990; EU, 2002). The volatilization half-life from a model river was estimated to be five days, while the volatilization half-life from a model lake was estimated to be 56 days (U.S. EPA, 2005). The best available effects information available were for other

[102] See footnote 98, above.
[103] See footnote 97, above.
[104] See footnote 25, above.
[105] *Technical Fact Sheet – 1,4-Dioxane.* U.S. EPA, Federal Facilities Restoration and Reuse Office. EPA 505-F-14-011: January, 2014.
[106] *Toxicological Profile for 1,4-Dioxane*. April, 2012; Agency for Toxic Substances and Disease Registry.
[107] *1,4-Dioxane Fact Sheet: Support Document*. EPA Office of Pollution Prevention and Toxics (OPPT) Chemical Fact Sheet. EPA 749-F-95-010a: February, 1995.

sensitive species. In freshwaters, acute toxicity concentrations ranged from 4,269 mg/L for the bluegill sunfish (*Lepomis macrochirus*) (Brooke, 1987) to 13,000 mg/L for the fathead minnow (*Pimephales promelas*) (GDCH, 1991b). In invertebrates, a 96h LC_{50} of 2,274 mg/L was determined for the scud (*Gammarus pseudolimnaeus*) (Brooke, 1987) and a 24h LC_{50} of 4,700 mg/L was determined for the water flea (*Daphnia magna*) (Bringmann and Kuhn, 1977). The lowest acute effect concentration for invertebrates was 163 mg/L for the water flea (*Ceriodaphnia dubia*) (GDCH, 1991b). Chronic effect concentrations ranged from a No Observed Effect Concentration (NOEC) of 145 mg/L for embryo-larval fathead minnow (*Pimephales promelas*) over 32 days (GDCH, 1991b) to an Observed Effect Concentration (LOEC) of 6,933 mg/L for medaka (*Oryzias latipes*) over 28 days (Johnson et al., 1993). In the invertebrate water flea, *Ceriodaphnia dubia*, the NOEC and LOEC were 635 mg/L and 1,250 mg/L, respectively, over 7 days (Dow, 1995).[108]

Acetone

Acetone is limited to a maximum of 7.97 mg/L. EPA NRWQC are not available for this parameter. Acetone is miscible in water and soluble in benzene and ethanol. Acetone is highly volatile and will volatilize rapidly from surface water. Acetone does not readily adsorb to sediment but may be consumed by microorganisms when present in surface water, which can lead to oxygen depletion in aquatic systems. Acetone produces detectable odors in air and water, with an odor threshold in water of 20 mg/L.

The environmental half-life in a river is estimated at six (6) days. Aqueous biodegradation has been estimated as less than one (1) day. The best available effects information available were for other surrogate, or sensitive species. Acute toxicity to fish ranges from an LC_{50} of 6,070 mg/L for brook trout to 15,000 mg/L for fathead minnow. The lowest LC_{50} for aquatic invertebrates is 2,100 mg/L, ranging to 16,700 mg/L. The chronic NOEC for daphnia is 1,660 mg/L.[109] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 507,640 µg/L and 1,560 µg/L in fish and daphnid, respectively.[110]

Total Phenol

Total phenol is limited to 300 µg/L in freshwater and saltwater, equivalent to EPA's organoleptic effects NRWQC. EPA aquatic life NRWQC are not available for this parameter. This effluent limitation is also more stringent than the most current human health organism-only NRWQC, 300 mg/L. Phenol is also limited to a maximum of 1,080 µg/L. Phenol is a widely used chemical intermediate. Residual phenol can also occur as a byproduct of the combustion of wood, petroleum products, and coal gas, and the degradation of organic matter and organic wastes, especially benzene. Phenol degrades rapidly in air and will generally biodegrade rapidly in soil at lower concentrations in the presence of microorganisms capable of degrading phenol. When biodegradation is sufficiently slow, phenol in soil will leach to groundwater.[111]

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[108] Canadian Council of Ministers of the Environment. 2008. Canadian water quality guidelines for the protection of aquatic life: 1,4-Dioxane. In: Canadian environmental quality guidelines, 1999, Canadian Council of Ministers of the Environment, Winnipeg.

[109] Toxicological Review of Acetone. U.S. Environmental Protection Agency: EPA/635/R-03/004, May, 2003; and *Toxicological Profile for Acetone*. Agency for Toxic Substances and Disease Registry: May, 1994.

[110] See footnote 19, above.

[111] *Acetone, CAS No: 67-64-1*, SIDS Initial Assessment Report (SIAR) for the 9th SIAM. U.S. Environmental

The best available effects information available were for other sensitive species. EPA's *Quality Criteria for Water* indicates that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 10,200 µg/L and 2,560 µg/L, respectively, and acute toxicity to saltwater aquatic life occurs at concentrations as low as 5,800 µg/L.[112] According to NMFS, phenol is not expected to bioaccumulate in fish. Pink salmon (*Oncorhynchus gorbuscha*) exhibited a 96-h LC_{50} of 3,730 µg/L for phenol (Korn et al. 1979, Korn et al. 1985).[113] Additional effects data available in EPA's *Quality Criteria for Water* indicates acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 10,200 µg/L and 2,560 µg/L, respectively, and acute toxicity to saltwater aquatic life occurs at concentrations as low as 5,800 µg/L. [114] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were <200 µg/L and 2,005 µg/L in fish and daphnid, respectively.[115]

Chlorinated Halogenated VOCs: Carbon Tetrachloride, Total dichlorobenzene, 1,2-dichlorobenzene, 1,3-dichlorobenzene, 1,4-dichlorobenzene, 1,1 Dichloroethane (1,1-DCA) 1,2 Dichloroethane (1,2-DCA), 1,1 Dichloroethylene (1,1-DCE), Methylene Chloride, 1,1,1 Trichloroethane (1,1,1-TCA) 1,1,2 Trichloroethane (1,1,2-TCA), Tetrachloroethylene (PCE), Trichloroethylene (TCE), cis-1,2 Dichloroethylene (cis-1,2-DCE), Vinyl Chloride

Carbon Tetrachloride is limited to 1.6 µg/L in freshwater and saltwater for sites in Massachusetts, which is equivalent to the human health organisms-only water quality criteria in Massachusetts. Carbon tetrachloride is also limited to a maximum of 4.4 µg/L. EPA aquatic life NRWQC are not available for this parameter. The best available effects information available were for other sensitive species. EPA's *Quality Criteria for Water* indicates that acute toxicity to freshwater aquatic life occurs at concentrations as low as 35,200 µg/L and acute toxicity to saltwater aquatic life occurs at concentrations as low as 50,000 µg/L.[116] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 1,970 µg/L and 5,580 µg/L in fish and daphnid, respectively.[117]

This VOC volatilizes rapidly when released surface water. In the presence of free or available sulfide and ferrous ions, carbon tetrachloride can also degrade through reductive dechlorination (Kriegman-King and Reinhard 1991). Limited data indicate that carbon tetrachloride has a low tendency to bioconcentrate in the food chain, mainly due to volatility (Neely et al. 1974; Pearson and McConnell 1975).[118]

Protection Agency Office of Pollution Prevention and Toxics: 1999.
[112] See footnote 25, above.
[113] See footnote 9, above.
[114] See footnote 63, above.
[115] See footnote 19, above.
[116] See footnote 25, above.
[117] See footnote 19, above.
[118] *Toxicological Profile for Carbon Tetrachloride*. Agency for Toxic Substances and Disease Registry: August, 2005.

Total dichlorobenzene is the sum of three isomers: 1,2-dichlorobenzene, 1,3-dichlorobenzene, and 1,4-dichlorobenzene. Total dichlorobenzene is limited to a maximum of 763 µg/L in freshwater and saltwater for sites in New Hampshire, equivalent to New Hampshire's water quality criterion for this parameter. EPA NRWQC are not available for this parameter. WQC are also not available for this parameter for Massachusetts. However, the three isomers of total dichlorobenzene are limited individually in both Massachusetts and New Hampshire. The best available effects information available were for other sensitive species. EPA's *Quality Criteria for Water* indicates that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 1,120 µg/L and 763 µg/L, respectively, and acute toxicity to saltwater aquatic life occurs at concentrations as low as 1,970 µg/L.[119] Individual isomers are not specified.

These VOCs volatilize rapidly when released surface water. Dichlorobenzenes (DCBs) are not known to occur naturally. Biodegradation of DCBs may occur in water under aerobic, but not anaerobic, conditions. A study of chlorobenzenes in sediments, water, and selected fish from the Great Lakes indicated that many chlorobenzenes are bioconcentrated by fish, but that DCBs are concentrated to a smaller extent than some of the more highly chlorinated chlorobenzene compounds such as pentachlorobenzene and hexachlorobenzene (Oliver and Niimi 1982a).[120]

1,2 Dichlorobenzene (1,2-DCB) is limited to a maximum of 600 µg/L, equivalent to State groundwater quality standards. This effluent limitation is more stringent than the most current human health organism-only NRWQC, 3,000 µg/L. EPA aquatic life NRWQC are not available for this parameter. EPA's *Quality Criteria for Water* indicates toxicity information for dichlorobenzenes, as noted above. Individual isomers are not specified.

1,2-DCB is one of the three DCBs isomers described with respect to total DCBs, above. 1,2-DCB is a liquid at room temperature. 1,2-DCB may be produced as a by-product in the manufacture of 1,4-DCB.[121]

1,3 Dichlorobenzene (1,3-DCB) is limited to a maximum of 320 µg/L, which is equivalent to the human health water quality criteria for the consumption of water and organisms ("water + organisms") in Massachusetts and New Hampshire. EPA aquatic life NRWQC are not available for this parameter. EPA's *Quality Criteria for Water* indicates toxicity information for dichlorobenzenes, as noted above. Individual isomers are not specified.

1,3-DCB is one of the three DCBs isomers described with respect to total DCBs, above. 1,3-DCB is a liquid at room temperature.[122]

1,4 Dichlorobenzene (1,4-DCB) is limited to a maximum of 5.0 µg/L, equivalent to State groundwater quality standards. This effluent limitation is more stringent than the most current human health organism-only NRWQC, 900 µg/L. EPA aquatic life NRWQC are not available

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[119] See footnote 25, above.
[120] *Toxicological Profile for Dichlorobenzenes*. Agency for Toxic Substances and Disease Registry: August, 2006.
[121] See footnote 120, above.
[122] See footnote 120, above.

for this parameter. EPA's *Quality Criteria for Water* indicates toxicity information for dichlorobenzenes, as noted above. Individual isomers are not specified.

1,4-DCB is one of the three DCBs isomers described with respect to total DCBs, above. 1,4-DCB is a widely used deodorizer/repellant and is generally the more widely used of the DCBs. Whereas 1,2- and 1,3-DCB are liquids at room temperature, 1,4-DCB is a solid that sublimes readily.[123]

1,1 Dichloroethane (1,1-DCA) is limited to a maximum of 70 µg/L, equivalent to State groundwater quality standards. EPA aquatic life NRWQC are not available for this parameter. The best available effects information available were for other sensitive species. G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic value was 14,680 µg/L in fish.[124]

This VOC volatilizes rapidly when released surface water. In the absence of oxygen and in the presence of anaerobic, methane-producing bacteria in groundwater, 1,1-DCA is produced by biodegradation of 1,1,1-TCA. Further degradation to chloroethane can also occur.[125]

1,2 Dichloroethane (1,2-DCA) is limited to a maximum of 5.0 mg/L, equivalent to State groundwater quality standards. This effluent limitation is more stringent than the most current human health organism-only NRWQC, 650 µg/L. EPA aquatic life NRWQC are not available for this parameter. The best available effects information available were for other sensitive species. EPA's *Quality Criteria for Water* indicates that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 118,000 µg/L and 20,000 µg/L, respectively, and acute toxicity to saltwater aquatic life occurs at concentrations as low as 113,00 µg/L for fish and invertebrates.[126] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 41,364 µg/L and 15,200 µg/L in fish and daphnid, respectively.[127]

This VOC volatilizes rapidly when released surface water. 1,2-DCA is not known to occur naturally. However, 1,2-DCA may be present from the anaerobic biodegradation of other chlorinated alkanes such as 1,1,2,2-tetrachloroethane. Biodegradation occurs slowly in water. 1,2-DCA generally does not adsorb to suspended solids and sediment in the water column.[128]

1,1 Dichloroethylene (1,1-DCE) is limited to a maximum of 3.2 µg/L, which is equivalent to the human health organism-only water quality criteria in Massachusetts and New Hampshire. EPA aquatic life NRWQC are not available for this parameter. The best available effects information available were for other sensitive species. EPA's *Quality Criteria for Water* indicates that acute toxicity to freshwater aquatic life occurs at concentrations as low as 11,600 µg/L and chronic toxicity to saltwater aquatic life occurs at concentrations as low as 224,000 µg/L. The toxicity

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[123] See footnote 120, above.
[124] See footnote 19, above.
[125] *Toxicological Profile for 1,1-Dichlorethane*. Agency for Toxic Substances and Disease Registry: August, 2006.
[126] See footnote 25, above.
[127] See footnote 19, above.
[128] See footnote 125, above.

information as noted is specified for dichloroethylenes. Individual isomers are not specified.[129] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were >4,800 µg/L and 4,720 µg/L in fish and daphnid, respectively.[130]

This VOC generally volatilizes rapidly when released surface water. However, 1,1-DCE also has high water solubility. 1,1-DCE reduces to vinyl chloride under methanogenic conditions and through reductive chlorination by microorganisms.[131]

Methylene Chloride, also known as dichloromethane (DCM), is limited to a maximum of 4.6 µg/L, which is equivalent to the human health water + organisms water quality criteria in Massachusetts and New Hampshire. EPA aquatic life NRWQC are not available for this parameter. The best available effects information available were for other sensitive species. G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 108,000 µg/L and 42,667 µg/L in fish and daphnid, respectively.[132]

This VOC volatilizes rapidly when released surface water. Both aerobic and anaerobic biodegradation of methylene chloride can occur in water. The biodegradation of methylene chloride may increase in the presence of elevated levels of organic carbon. Methylene chloride does not strongly adsorb to sediments.[133]

1,1,1 Trichloroethane (1,1,1-TCA) is limited to a maximum of 200 µg/L, equivalent to State groundwater quality standards. This effluent limitation is more stringent than the most current human health organism-only NRWQC, 200,000 µg/L. EPA aquatic life NRWQC are not available for this parameter. The best available effects information available were for other sensitive species. EPA's *Quality Criteria for Water* indicates that acute toxicity to saltwater aquatic life occurs at concentrations as low as 31,200 µg/L for fish and invertebrates.[134] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic value was 3,493 µg/L in fish.[135]

This VOC volatilizes rapidly when released surface water. In surface waters, 1,1,1-TCA also does not readily adsorb to sediment or suspended organic material. Slow biodegradation of 1,1,1-TCA can occur under both anaerobic and aerobic conditions. Under anaerobic conditions, 1,1,1-TCA degrades to 1,1-dichloroethane through reductive dechlorination by methane-producing bacteria and by sulfate-reducing organisms, which can further degrade to chloroethane.[136]

1,1,2 Trichloroethane (1,1,2-TCA) is limited to a maximum of 5.0 µg/L, equivalent to State groundwater quality standards. This effluent limitation is more stringent than the most current human health organism-only NRWQC, 8.9 µg/L. The best available effects information available

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[129] See footnote 25, above.
[130] See footnote 19, above.
[131] *Toxicological Profile for 1,2,-Dichlorethene*. Agency for Toxic Substances and Disease Registry: May, 1994.
[132] See footnote 19, above.
[133] *Toxicological Profile for Methylene Chloride*. Agency for Toxic Substances and Disease Registry: September, 2000.
[134] See footnote 25, above.
[135] See footnote 19, above.
[136] *Toxicological Profile for 1,1,1-Trichloroethane*. Agency for Toxic Substances and Disease Registry: July, 2006.

were for other sensitive species. EPA aquatic life NRWQC are not available for this parameter. EPA's *Quality Criteria for Water* indicates that chronic toxicity to freshwater aquatic life occurs at concentrations as low as 9,400 µg/L.[137] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 9,400 µg/L and 18,400 µg/L in fish and daphnid, respectively.[138]

This VOC volatilizes rapidly when released surface water. In surface waters, 1,1,2-TCA also does not readily adsorb to sediment or suspended organic material. While aerobic biodegradation does not generally occur, 1,1,2-TCA can be formed during the anaerobic biodegradation of 1,1,2,2-tetrachloroethane and 1,1,2-TCA can further degrade to form vinyl chloride.[139]

Tetrachloroethylene (PCE) is limited to 3.3 µg/L in freshwater and saltwater for sites in Massachusetts, which is equivalent to the human health organisms-only water quality criterion in Massachusetts. PCE is also limited to a maximum of 5.0 µg/L, equivalent to State groundwater quality standards. EPA aquatic life NRWQC are not available for this parameter. The best available effects information available were for other sensitive species. EPA's *Quality Criteria for Water* indicates that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 5,280 µg/L and 840 µg/L, respectively and acute and chronic toxicity to saltwater aquatic life occurs at concentrations as low as 10,200 µg/L and 450 µg/L, respectively.[140] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 840 µg/L and 750 µg/L in fish and daphnid, respectively.[141]

This VOC generally volatilizes rapidly when released surface water. PCE can biodegrade to DCE, trichloroethylene, vinyl chloride and ethene through reductive dechlorination, but is generally slow to break down in water. PCE has low water solubility, but is miscible with alcohol, ether, benzene, and most fixed and volatile oils. PCE also has a density higher than water, which causes PCE that is not immediately volatilized to submerge below water.[142]

Trichloroethylene (TCE) is limited to a maximum of 5.0 µg/L, equivalent to State groundwater quality standards. This effluent limitation is more stringent than the most current human health organism-only NRWQC, 7 µg/L. The best available effects information available were for other sensitive species. EPA aquatic life NRWQC are not available for this parameter. EPA's *Quality Criteria for Water* indicates that acute toxicity to freshwater aquatic life occurs at concentrations as low as 45,000 µg/L and acute toxicity to saltwater aquatic life occurs at concentrations as low as 2,000 µg/L.[143] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 11,100 µg/L and 7,257 µg/L in fish and daphnid, respectively.[144]

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[137] See footnote 25, above.
[138] See footnote 19, above.
[139] *Toxicological Profile for 1,1,2-Trichloroethane*. Agency for Toxic Substances and Disease Registry: December, 1989.
[140] See footnote 25, above.
[141] See footnote 19, above.
[142] *Draft Toxicological Profile for Tetrachloroethylene*: October 2014; Agency for Toxic Substances and Disease Registry.
[143] See footnote 25, above.
[144] See footnote 19, above.

This VOC generally volatilizes rapidly when released surface water. TCE also has a density higher than water, which causes TCE that is not immediately volatilized to submerge below water. Anaerobic degradation of TCE in water can produce DCE, vinyl chloride and ethylene. TCE may adsorb onto organic and inorganic solids (e.g., fats, waxes, and resins). [145]

cis-1,2 Dichloroethylene (cis-1,2-DCE) is limited to a maximum of 70 µg/L, equivalent to State groundwater quality standards. EPA aquatic life NRWQC are not available for this parameter. This VOC volatilizes rapidly when released surface water. cis-1,2-DCE often occurs as a mixture with the trans- isomer of DCE. cis-1,2-DCE can be formed when other solvents such as PCE, TCE, and vinyl chloride degrade. Multiple anaerobic degradation processes can occur in water.[146]

The best available effects information available were for other sensitive species. EPA's *Quality Criteria for Water* indicates that acute toxicity to freshwater aquatic life occurs at concentrations as low as 11,600 µg/L and chronic toxicity to saltwater aquatic life occurs at concentrations as low as 224,000 µg/L. The toxicity information as noted is specified for dichloroethylenes. Individual isomers are not specified.[147] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic value for 1,2-DCE (isomer unspecified) was 9,538 µg/L in fish.[148]

Vinyl Chloride is limited to a maximum of 2.0 µg/L, equivalent to State groundwater quality standards. EPA aquatic life NRWQC are not available for this parameter.

This VOC volatilizes rapidly when released surface water. However, vinyl chloride also has high water solubility. The persistence of vinyl chloride in water can be affected by turbidity and the presence of salts, which form complexes with vinyl chloride that increase its water solubility. Vinyl chloride can also occur from of anaerobic reductive dehalogenation of PCE, TCE, and 1,1,1-TCA, which generally occurs relatively slowly [149] The best available effects information available were for other sensitive species. EPA's *Quality Criteria for Water* indicates that acute toxicity to freshwater aquatic life occurs at concentrations as low as 35,200 µg/L and acute toxicity to saltwater aquatic life occurs at concentrations as low as 50,000 µg/L.[150]

Ethylene Dibromide (EDB)
EDB is limited to a maximum of 0.05 µg/L, equivalent to State groundwater quality standards. EPA aquatic life NRWQC are not available for this parameter.

EDB is an aliphatic hydrocarbon. EDB has high water solubility. A fraction of EDB is relatively immobile and resistant to mobilization, chemical transformation and biodegradation when bound to micropores. EDB can leach slowly from micropore sites, especially if disturbed or crushed,

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[145] *Draft Toxicological Profile for Trichloroethylene*. Agency for Toxic Substances and Disease Registry: October, 2014.
[146] *Toxicological Profile for cis-1,2-dichloroethene*. Agency for Toxic Substances and Disease Registry: August, 1996.
[147] See footnote 25, above.
[148] See footnote 19, above.
[149] *Toxicological Profile for Vinyl Chloride*. Agency for Toxic Substances and Disease Registry: July, 2006.
[150] See footnote 25, above.

contaminating water over longer periods.[151] The best available effects information available were for other sensitive species. EPA's *Quality Criteria for Water* indicates that acute toxicity to freshwater aquatic life occurs at concentrations as low as 35,200 µg/L and acute toxicity to saltwater aquatic life occurs at concentrations as low as 50,000 µg/L.[152]

Effects Determination for Non-Halogenated and Halogenated VOCs

Based on the best available information, EPA has made the determination that the effects from the fifteen aforementioned non-halogenated and halogenated VOC parameters on shortnose sturgeon and Atlantic sturgeon will be insignificant and/or discountable because if these parameters are present in a discharge authorized under the RGP:

1) Water quality standards are met at the point of discharge: These parameters are expected to volatilize rapidly before undergoing any significant chemical or biological degradation such that effects are extremely unlikely to occur and are therefore discountable. Further, available effects data for most parameters do not exceed the maximum allowable discharge concentrations, suggesting that, if the numeric limits for these parameters are taken to represent the surrogate instream constituent exposure concentrations, the numeric limits are an acceptable proxy and any effects are extremely unlikely to occur and are therefore discountable.

With respect to prey quantity/quality, EPA has made the determination that the proposed action will be insignificant because if one or more of these VOC parameters is present in a discharge authorized under the RGP:

1) Discharges are likely to cause only a minor and temporary reduction in available prey, if any, such that any effects on individual animals are not capable of being meaningfully measured, detected, or evaluated: Discharges eligible for coverage under this general permit are expected to occur with low frequency (intermittent), small magnitude (small volume limited to no more than 1.0 MGD), and short duration (temporary or short-term) and as such are likely to cause effects minor and temporary in nature. The discharge must meet numeric limits established at concentrations below available estimated effects concentrations for most parameters at end-of-pipe. If any of these parameters are present in remediation activity discharges, they are expected to volatilize rapidly before causing any significant chemical or biological effect. These pollutants are then expected to undergo rapid, full mixing because of the high dilution in the receiving waters to concentrations less than the minimum level of detection such that effects are likely to cause only a minor and temporary change in the abundance, distribution, quality and availability of only the most sensitive prey species. This minor and temporary loss or alteration is not expected to affect the way that individual animals use the Action Area or result in behavior change (e.g., foraging) in individual animals that can be meaningfully measured, detected, or evaluated and is therefore insignificant.

[151] *Toxicological Profile for 1,2-dibromoethane*. Agency for Toxic Substances and Disease Registry: July, 1992.
[152] See footnote 25, above.

Consequently, the proposed action is not likely to adversely affect the shortnose sturgeon, the Atlantic sturgeon or their prey in the shoreline areas of Massachusetts and New Hampshire or the Connecticut River, Merrimack River, Taunton River, or Piscataqua River.

Non-Halogenated and Halogenated Semi-Volatile Organic Compounds (SVOCs)

SVOCs are organic compounds that volatilize slowly at standard temperature and pressure (i.e., 20 degrees Celsius and 1 atmosphere). A halogenated compound is one that has a halogen (e.g., fluorine, chlorine, bromine, or iodine) attached to its chemical structure. Aquatic organisms can be expected to experience greater exposure to more soluble substances. Other factors also affect the likelihood of an organism's exposure to SVOCs, including environmental degradation and biodegradation. It can be inferred, based on observed effects in other non-salmonid fish, that organic pollutants could lead to decreased growth, alterations of metabolic functions, and reduced recruitment.[153]

Numeric effluent limitations for the majority of these parameters are equivalent to human health- or risk-based water quality criteria such as EPA's human health NRWQC and State-adopted groundwater quality standards, which are imposed near or below analytical minimum levels of detection. Therefore, while human health and/or risk-based effluent limitations are not specifically derived for the protection of aquatic life, such limitations are an appropriate proxy because any potential effects to aquatic life at concentrations that could potentially occur near or below analytical levels of detection cannot be meaningfully measured, detected or evaluated. The non-halogenated and halogenated SVOC parameters potentially present in remediation activity discharges and the expected water quality effects on shortnose, Atlantic sturgeon, or their prey, if known, are discussed in this section. EPA's determination follows this information and is made with respect to all of the non-halogenated and halogenated SVOC parameters potentially present in remediation activity discharges.

Phthalates

Total Phthalates is limited to 3.0 µg/L in freshwater and 3.4 µg/L in saltwater for sites in New Hampshire, equivalent to New Hampshire's water quality criteria for total phthalate esters. Total phthalates is also limited to a maximum of 190 µg/L. EPA NRWQC are not available for this parameter. However, one pollutant that comprises this parameter, diethylhexyl phthalate, is also limited in the draft RGP.

Phthalates are a group of compounds that contain a phenyl ring with two attached acetate groups. They are often referred to as plasticizers. Because phthalates are not a part of the polymers that make up plastics, they can be released from these materials fairly easily. The use of plastics and materials containing plasticizers is widespread. Total phthalates is the sum of: diethylhexyl phthalate (DEHP), benzyl butyl phthalate, di-n-butyl phthalate, diethyl phthalate, dimethyl phthalate and di-n-octyl phthalate. The best available effects information available were for other sensitive species. EPA's *Quality Criteria for Water* indicates that acute and chronic toxicity to

[153] See footnote 9, above.

freshwater aquatic life occurs at concentrations as low as 940 µg/L and 3 µg/L, respectively and chronic toxicity to saltwater aquatic life occurs at concentrations as low as 2,944 µg/L. The toxicity information as noted is specified for phthalate esters. Individual phthalates are not specified.[154] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 912 µg/L in daphnid for DEHP, 717 µg/L and 697 µg/L in fish and daphnid, respectively, for di-n-butyl phthalate, and 3,822 µg/L 708 µg/L in fish and daphnid, respectively, for di-n-octyl phthalate.[155]

Diethylhexyl Phthalate (DEHP) is limited to 2.2 µg/L in Massachusetts and 5.9 µg/L (draft) and 2.2 µg/L (final) in New Hampshire (due to revision of New Hampshire water quality regulations), equivalent to the human health organisms-only water quality criteria in Massachusetts and New Hampshire. DEHP is also limited to a maximum of 101 µg/L. EPA aquatic life NRWQC are not available for this parameter.

DEHP is one of the six phthalates described with respect to total phthalates, above. As noted, G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 912 µg/L in daphnid for DEHP.

Polycyclic Aromatic Hydrocarbons (PAHs): Total Group I PAHs, Benzo(a)anthracene, Benzo(a)pyrene, Benzo(b)fluoranthene, Benzo(k)fluoranthene, Chrysene, Dibenzo(a,h)anthracene and Indeno(1,2,3-cd)pyrene, Total Group II PAHs, Naphthalene
PAHs are a group of organic compounds that form through the incomplete combustion of organic materials. PAHs are also present in fossil fuels, petroleum derivatives and residuals (e.g., asphalt, coal, crude oil, heavy distillates, and tars). PAHs consist of two or more aromatic rings. In general, physical and chemical characteristics of PAHs vary with the number of aromatic rings comprising their chemical structure (i.e., molecular weight). Two- and three-ring PAH compounds mainly occur in the vapor phase. PAHs that have five or more aromatic rings mainly occur in the particulate phase. Four-ring PAH compounds occur in both phases.[156] In surface water, PAHs can volatilize, oxidize, biodegrade and bind to suspended particles or sediments. Several PAHs are known animal carcinogens, while others can enhance the response of the carcinogenic PAHs.[157] Acute toxicity to saltwater aquatic life has been documented at concentrations as low as 300 µg/L.[158] The best available effects information available were for other sensitive species. EPA's *Quality Criteria for Water* indicates that the most susceptible category of organisms, the marine larvae, appear to be intolerant of the water soluble petroleum-related compounds, at concentrations as low as 0.1 mg/L.[159] The PAH indicator parameters included in the draft RGP are described in this section, below.

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[154] See footnote 25, above.
[155] See footnote 19, above.
[156] *Remediation Technologies Screening Matrix and Reference Guide, Version 4.0, Section 2.5.1: Properties and Behavior of Non-Halogenated SVOCs* (2007).
[157] *Toxicological Profile for Polycyclic Aromatic Hydrocarbons*. Agency for Toxic Substances and Disease Registry: August, 1995.
[158] *Ambient Water Quality Criteria for Polynuclear Aromatic Hydrocarbons.* EPA 440/5-80-069: October 1980.
[159] See footnote 63, above.

Exposure to environmentally persistent pollutants such as PAHs can cause lesions, retard the growth, or impair the reproductive capabilities of aquatic life (Cooper, 1989); (Sindermann, 1994). As stated in the recovery plan for the shortnose sturgeon and the status review for the Atlantic sturgeon, the life history of these species (which includes a long lifespan and benthic foraging habit) predispose the sturgeon to long-term and repeated exposure to environmental contamination (NMFS, 1998); (Atlantic Sturgeon Status Review Team, 2007). EPA's *Quality Criteria for Water* indicates that acute toxicity to saltwater aquatic life occurs at concentrations as low as 300 µg/L. The toxicity information as noted is specified for polynuclear aromatic hydrocarbons. Individual PAHs are not specified.[160]

The PAH parameters potentially present in remediation activity discharges and the expected water quality effects on shortnose, Atlantic sturgeon, or their prey, if known, are discussed further with respect to each individual metal, below.

Total Group I PAHs is limited to a maximum of 1.0 µg/L. Total Group I PAHs is the sum of: benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene and indeno(1,2,3-cd)pyrene. The draft RGP requires that analysis of individual Group I PAH compounds achieve a minimum level of analysis (ML) of 0.1 µg/L or less (i.e., at the approximate minimum level of detection using the most sensitive test method currently approved in 40 CFR Part 136). The sum of Group I PAH compound MLs in compliance with this requirement is 0.7 µg/L. The effluent limitation reflects the sum of the compliance levels for individual Group I PAH compounds, adjusted upward to 1.0 µg/L to account for variation in analytical MLs expected to be achieved. EPA NRWQC are not available for this parameter. However, the pollutants that comprise this parameter are also limited individually in the draft RGP.

Group I PAHs have higher molecular weights (i.e., contain four to seven aromatic rings). As a result, Group I PAHs are more resistant to oxidation, reduction, and vaporization, are less water-soluble and are generally persistent (i.e., less degradable). Group I PAHs are generally less toxic to aquatic organisms but are carcinogenic. Higher molecular weight PAHs more strongly bind to organic carbon in soil and sediment. Because of their low solubility and high affinity for organic carbon, Group I PAHs in surface water typically occur adsorbed to particles that either have settled to the bottom or are suspended in the water column.

Several characteristics of short-nose sturgeon (i.e., long lifespan, extended residence in estuarine habitats, benthic predator) predispose the species to long-term and repeated exposure to surface water and sediment contamination from oil and oil-related derivatives (PAHs). Kocan et al. (1996) investigated the survival of sturgeon eggs and larvae exposed to sediment. Coal-tar contaminated sediment produced approximately 95 percent embryo-larval mortality after 18 days of exposure. Toxicity appeared to be via direct contact of the embryos with contaminated whole sediment, as opposed to water soluble extracts of the sediment (elutriate). For example, the concentration of low molecular weight PAHs (LPAHs; water soluble) that resulted in embryo and larval mortality was ≥ 0.47 mg/L, which is higher than would occur naturally.[161]

[160] See footnote 25, above.
[161] See footnote 9, above.

Benzo(a)anthracene is limited to 0.0038 µg/L in freshwater and saltwater, which is equivalent to the human health water + organisms water quality criteria in Massachusetts and New Hampshire. EPA aquatic life NRWQC are not available for this parameter. Benzo(a)anthracene is also limited to a maximum of 1.0 mg/L as total group I PAHs.

Benzo(a)anthracene is one of the seven Group I PAHs described with respect to total Group I PAHs, above. The best available effects information available were for other sensitive species. G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic value was 0.65 µg/L in daphnid.[162]

Benzo(a)pyrene is limited to 0.0038 µg/L in freshwater and saltwater, which is equivalent to the human health water + organisms water quality criteria in Massachusetts and New Hampshire. EPA aquatic life NRWQC are not available for this parameter. The most current human health organism-only NRWQC is 0.00013 µg/L, which Massachusetts and New Hampshire have not adopted into their standards. Benzo(a)pyrene is also limited to a maximum of 1.0 mg/L as total group I PAHs.

Benzo(a)pyrene is one of the seven Group I PAHs described with respect to total Group I PAHs, above. The best available effects information available were for other sensitive species. Effects have been observed from oil-related derivatives such as PAHs. For example, Oris and Giesy (1987) reported a LT_{50} (lethal time to 50 percent mortality) of 5.6 µg/L for benzo(a) pyrene (BaP) in larval fathead minnow.[163] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic value was 0.30 µg/L in daphnid.[164]

Benzo(b)fluoranthene is limited to 0.0038 µg/L in freshwater and saltwater, which is equivalent to the human health water + organisms water quality criteria in Massachusetts and New Hampshire. EPA aquatic life NRWQC are not available for this parameter. The most current human health organism-only NRWQC is 0.0013 µg/L, which Massachusetts and New Hampshire have not adopted into their standards. Benzo(b)fluoranthene is also limited to a maximum of 1.0 mg/L as total group I PAHs.

Benzo(b)fluoranthene is one of the seven Group I PAHs described with respect to total Group I PAHs, above. No specific effects information for this pollutant was identified.

Benzo(k)fluoranthene is limited to 0.0038 µg/L in freshwater and saltwater, which is equivalent to the human health water + organisms water quality criteria in Massachusetts and New Hampshire (retained in New Hampshire to meet anti-backsliding requirements). EPA aquatic life NRWQC are not available for this parameter. This effluent limitation is more stringent than the most current human health organism-only NRWQC,0.013 µg/L, which Massachusetts and New Hampshire have not adopted into their standards.

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[162] See footnote 19, above.
[163] See footnote 9, above.
[164] See footnote 19, above.

Benzo(k)fluoranthene is one of the seven Group I PAHs described with respect to total Group I PAHs, above. No specific effects information for this pollutant was identified.

Chrysene is limited to 0.0038 µg/L in freshwater and saltwater, which is equivalent to the human health water + organisms water quality criteria in Massachusetts and New Hampshire (retained in New Hampshire to meet anti-backsliding requirements). EPA aquatic life NRWQC are not available for this parameter. This effluent limitation is more stringent than the most current human health organism-only NRWQC, 0.13 µg/L, which Massachusetts and New Hampshire have not adopted into their standards. Chrysene is also limited to a maximum of 1.0 mg/L as total group I PAHs.

Chrysene is one of the seven Group I PAHs described with respect to total Group I PAHs, above. No specific effects information for this pollutant was identified.

Dibenzo(a,h)anthracene is limited to 0.0038 µg/L in freshwater and saltwater, which is equivalent to the human health water + organisms water quality criteria in Massachusetts and New Hampshire. EPA aquatic life NRWQC are not available for this parameter. The most current human health organism-only NRWQC is 0.00013 µg/L, which Massachusetts and New Hampshire have not adopted into their standards. Dibenzo(a,h)anthracene is also limited to a maximum of 1.0 mg/L as total group I PAHs.

Dibenzo(a,h)anthracene is one of the seven Group I PAHs described with respect to total Group I PAHs, above. No specific effects information for this pollutant was identified.

Indeno(1,2,3-cd)pyrene is limited to 0.0038 µg/L in freshwater and saltwater, which is equivalent to the human health water + organisms water quality criteria in Massachusetts and New Hampshire. EPA aquatic life NRWQC are not available for this parameter. The most current human health organism-only NRWQC is 0.0013 µg/L, which Massachusetts and New Hampshire have not adopted into their standards. Indeno(1,2,3-cd)pyrene is also limited to a maximum of 1.0 mg/L as total group I PAHs.

Indeno(1,2,3-cd)pyrene is one of the seven Group I PAHs described with respect to total Group I PAHs, above. No specific effects information for this pollutant was identified.

Total Group II PAHs is limited to a maximum of 100 µg/L (0.1 mg/L). Total Group II PAHs is the sum of: acenaphthene, acenaphthylene, anthracene, benzo(g,h,i)perylene, fluoranthene, fluorene, phenanthrene and pyrene. This effluent limitation is more stringent than the State aquatic life criterion for this parameter in New Hampshire and is equivalent to EPA's lifetime health advisory value for one of the nine Group II PAHs, naphthalene. EPA NRWQC are not available for this parameter. However, one pollutant that comprises this parameter, naphthalene, is also limited in the draft RGP.

Group II PAHs have lower molecular weights (i.e., contain two or three aromatic rings). Naphthalene has the lowest molecular weight of all PAHs. As a result, Group II PAHs are more water-soluble and transform more quickly than higher molecular weight PAHs, mainly through volatilization and biodegradation. Group II PAHs are not generally considered carcinogenic.

However, Group II PAHs can enhance or inhibit the response of the carcinogenic Group I PAHs and have significant acute toxicity to aquatic organisms. The best available effects information available were for other sensitive species. G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 74 µg/L 6,646 µg/L in fish and daphnid, respectively, for acenaphthene, 0.09 µg/L and <2.1 µg/L in fish and daphnid, respectively, for anthracene, 30 µg/L and 15 µg/L in fish and daphnid, respectively, for fluoranthene, and 200 µg/L in daphnid for phenanthrene.[165] EPA's *Quality Criteria for Water* indicates that acute toxicity to freshwater aquatic life occurs at concentrations as low as 1,700 µg/L and acute and chronic toxicity to saltwater aquatic life occurs at concentrations as low as 970 µg/L and 710 µg/L, respectively, for acenaphthene.[166]

Naphthalene is limited to a maximum of 20 µg/L, which is equivalent to State groundwater quality standards. Naphthalene is also limited to a maximum of 100 µg/L as total group II PAHs, equivalent to EPA's lifetime health advisory value for this parameter. EPA NRWQC are not available for this parameter.

Naphthalene is one of the nine Group II PAHs described with respect to total Group II PAHs, above. The best available effects information available were for other sensitive species. Acute and chronic toxicity to freshwater aquatic life has been documented at concentrations as low as 2,300 µg/L and 620 µg/L, respectively and acute toxicity to saltwater aquatic life at concentrations as low as 2,350 µg/L.[167] The PubChem Compound Database provides estimates for the amount of time required for each chemical to volatilize completely from a modeled river or lake. Naphthalene is expected to volatilize from a modeled river in four hours or from a modeled lake in five days. Naphthalene is the least water soluble of the Group II PAHs. According to NMFS, naphthalene is generally more likely to bioaccumulate, with reported bioconcentration factors ranging from 23-168 in fish. DeGraeve et al. (1982) reported a 96-h LC50 of 7.9 mg/L for naphthalene.[168]

Total PCBs

Total PCBs is limited to 0.000064 µg/L, equivalent to EPA's human health organisms-only NRWQC. This effluent limitation is more stringent than the chronic freshwater and saltwater aquatic life NRWQC, 0.014 µg/L and 0.03 µg/L, respectively. Total PCBs is the sum of the full list for Chemical Abstracts Service (CAS) Registry number 1336-36-3A.

PCBs encompass a class of compounds with a dual ring chemical structure that is formed by the addition of chlorine (C_{12}) to biphenyl ($C_{12}H_{10}$). PCBs include up to 209 variations, or congeners, with different physical and chemical characteristics, bioavailability and toxicity. PCBs were commonly used as mixtures called aroclors, typically found in oils associated with electrical transformers or gas pipelines. PCBs alone are not usually very mobile in water. PCBs are only slightly soluble in water, bind strongly to sediments, and are resistant to degradation. As a result,

[165] See footnote 19, above.
[166] See footnote 25, above.
[167] *Ambient Water Quality Criteria for Naphthalene.* EPA 440/5-80-059: October 1980.
[168] See footnote 9, above.

PCBs tend to persist in the environment[169] and can be transported by disturbance.[170] Exposure to environmentally persistent pollutants such as PCBs can cause lesions, retard the growth, or impair the reproductive capabilities of aquatic life (Cooper, 1989); (Sindermann, 1994). As stated in the recovery plan for the shortnose sturgeon and the status review for the Atlantic sturgeon, the life history of these species (which includes a long lifespan and benthic foraging habit) predispose the sturgeon to long-term and repeated exposure to environmental contamination (NMFS, 1998); (Atlantic Sturgeon Status Review Team, 2007).

The best available effects information available were for other sensitive species. EPA's *Quality Criteria for Water* indicates that acute toxicity to freshwater aquatic life probably occurs at concentrations above 2.0 µg/L and acute toxicity to saltwater aquatic life probably occurs at concentrations above 10 µg/L. The toxicity information as noted is specified for PCBs.[171] G.W. Suter II and C. L. Tsao (1996) reported that the lowest chronic values were 0.2 µg/L, 2.1 µg/L, and 0.8 µg/L in fish, daphnid and non-daphnid invertebrates, respectively, for total PCBs. Additional lowest chronic values for individual Aroclors included 60 µg/L in fish for Aroclor 1221, 124 µg/L in fish for Aroclor 1232, 9.00 µg/L and 2.9 µg/L in fish and non-daphnid invertebrates, respectively, for Aroclor 1242, 4.9 µg/L in daphnid for Aroclor 1254, and <1.3 µg/L in fish for Aroclor 1260.[172]

Pentachlorophenol (PCP)

PCP is limited to 1.0 µg/L, which is equivalent to State groundwater quality standards. This effluent limitation is more stringent than the chronic freshwater and saltwater aquatic life NRWQC, 15 µg/L and 7.9 µg/L, respectively.

PCP has a chlorinated ring structure that tends to increase its stability. However, its polar hydroxyl group can facilitate biodegradation. Metal salts of PCP are very soluble in water. The phenolic form is less soluble. PCP is denser than water, but the commonly used form is a solution of PCP and petroleum solvents in a mixture less dense than water. Therefore, PCP tends to occur on water surfaces.

The best available effects information available were for other sensitive species. EPA's *Quality Criteria for Water* indicates that acute and chronic toxicity to freshwater aquatic life occurs at concentrations as low as 55 µg/L and 3.2 µg/L, respectively and acute and chronic toxicity to saltwater aquatic life occurs at concentrations as low as 53 µg/L and 34 µg/L, respectively. The toxicity information as noted is specified for phthalate esters. Individual phthalates are not specified.[173]

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[169] *Toxicological Profile for Polychlorinated Biphenyls (PCBs)*. Agency for Toxic Substances and Disease Registry: November, 2000.
[170] *Remediation Technologies Screening Matrix and Reference Guide, Version 4.0, Section 2.6.1: Properties and Behavior of Halogenated SVOCs* (2007).
[171] See footnote 25, above.
[172] See footnote 19, above.
[173] See footnote 25, above.

Effects Determination for Non-Halogenated and Halogenated SVOCs
Based on the best available information, EPA has made the determination that the effects from the fourteen aforementioned non-halogenated and halogenated SVOC on shortnose sturgeon, Atlantic sturgeon, or their prey, will be insignificant and/or discountable because if one or more of these parameters is present in a discharge authorized under the RGP:

1) Water quality standards are met at the point of discharge: The numeric limits have been established near or below the minimum levels of detection and do not exceed available effects data, suggesting that, if the numeric limits are taken to represent the surrogate instream constituent exposure concentrations, the numeric limits are an acceptable proxy and any effects are extremely unlikely to occur and are therefore discountable. Further, given high dilution in the Action Area waterbodies, the maximum allowable discharge concentrations will result in the use of only a small portion of the available assimilative capacity of the nearshore marine waters of Massachusetts and New Hampshire or in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River such that cumulative effects from the environmental persistence of SVOCs are extremely unlikely to occur and are therefore discountable.

With respect to prey quantity/quality, EPA has made the determination that the proposed action will be insignificant because if one or more of these SVOC parameters is present in a discharge authorized under the RGP:

1) Discharges are likely to cause only a minor and temporary reduction in available prey, if any, such that any effects on individual animals are not capable of being meaningfully measured, detected, or evaluated: Discharges eligible for coverage under this general permit are expected to occur with low frequency (intermittent), small magnitude (small volume limited to no more than 1.0 MGD), and short duration (temporary or short-term) and as such are likely to cause effects minor and temporary in nature. The discharge must meet numeric limits established at concentrations below available estimated effects concentrations at end-of-pipe, near or below minimum levels of detection. If any of these pollutants are present in remediation activity discharges, these pollutants are expected to undergo rapid, full mixing because of the high dilution in the receiving waters to concentrations less than the minimum level of detection such that effects are likely to cause only a minor and temporary change in the abundance, distribution, quality and availability of only the most sensitive prey species. This minor and temporary loss or alteration is not expected to affect the way that individual animals use the Action Area or result in behavior change (e.g., foraging) in individual animals that can be meaningfully measured, detected, or evaluated and is therefore insignificant.

Consequently, the proposed action is not likely to adversely affect the shortnose sturgeon, the Atlantic sturgeon or their prey in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River, or the coastal embayments/nearshore marine waters of Massachusetts and New Hampshire.

Fuels Parameters

Fuels parameters are generally non-halogenated and may be both VOCs and SVOCs. Fuels are complex mixtures of many hydrocarbon compounds, additives and impurities. The exact composition of fuels varies depending upon: 1) the source of the crude oil; and 2) the refining practices used to produce the fuel. Fuels can contain a variable number of VOCs, SVOCs, additives such as oxygenates and/or metals.[174] While many VOCs, SVOCs and metals potentially present in fuels have already been evaluated under other contaminant type subcategories, several indicator parameters included in the RGP are specific to fuels contamination. Gasoline and fuel oils are the most commonly encountered sources of fuels parameters at sites covered under this general permit.

Aquatic organisms can be expected to experience greater exposure to more soluble substances. Other factors also affect the likelihood of an organism's exposure to the organic compounds of concern, including environmental degradation and biodegradation. It can be inferred, based on observed effects in other non-salmonid fish, that fuel pollutants could lead to decreased growth, alterations of metabolic functions, and reduced recruitment. The long-term sublethal effects of oil pollution include interferences with cellular and physiological processes such as feeding and reproduction and do not lead to immediate death of the organism.[175]

The fuels parameters potentially present in remediation activity discharges and the expected water quality effects on shortnose, Atlantic sturgeon, or their prey, if known, are discussed in this section. Numeric effluent limitations for the majority of these parameters are equivalent to human health- or risk-based water quality criteria such as EPA's human health NRWQC and State-adopted groundwater quality standards, which are imposed near or below analytical minimum levels of detection. Therefore, while human health and/or risk-based effluent limitations are not specifically derived for the protection of aquatic life, such limitations are an appropriate proxy because any potential effects to aquatic life at concentrations that could potentially occur near or below analytical levels of detection cannot be meaningfully measured, detected or evaluated.

Total Petroleum Hydrocarbons (TPH)

TPH is limited to a maximum of 5.0 mg/L. EPA NRWQC are not available for this parameter. However multiple individual petroleum hydrocarbon compounds are also limited in the draft RGP, which have been discussed in previous sections (e.g., BETX, PAHs). EPA's *Quality Criteria for Water* indicates that to prevent deleterious effects in aquatic organisms, surface waters should be virtually free from floating oils. A concentration of 15 mg/L is recognized as the level at which many oils produce a visible sheen and/or cause an undesirable taste in fish.[176] Therefore, in addition to the numeric limit of 5.0 mg/L for TPH, the RGP includes non-numeric limits that require treatment to ensure discharges remain free from pollutants in concentrations or combinations that float as foam, debris, scum, form a visible sheen.

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[174] *Toxicological Profile for Total Petroleum Hydrocarbons (TPH).* September, 1999; Agency for Toxic Substances and Disease Registry.
[175] See footnote 25, above.
[176] See footnote 63, above.

TPH generally refers to gasoline range, diesel range and/or oil range hydrocarbon compounds. Measurement of all individual hydrocarbon compounds in a petroleum product released to the environment is generally not practical, cost-effective or necessary to attain and maintain WQSs. Fuels and other petroleum products are complex mixtures of hydrocarbon compounds. When released to the environment, these compounds migrate by one or both of two general pathways: 1) as bulk product that migrates under the forces of gravity and capillary action; and 2) as individual compounds which separate from the bulk product by entering the aqueous phase in water or the vapor phase in air. In addition, the longer a petroleum product is exposed to environmental processes, the greater the change in chemical character (i.e., weathering). After extensive weathering, sampling is generally better informed by a more focused set of hydrocarbon compounds that includes ranges (e.g., TPH) and typical individual compounds (e.g., target analytes like BETX and/or PAHs).[177] Because of the wide range of compounds included in the category of oils and greases, EPA determined in the *Quality Criteria for Water* that it would be impossible to establish meaningful 96-hour LC_{50} values for oil and grease, which includes the total petroleum hydrocarbon fractions, without specifying the product involved.[178] Consequently, the RGP contains numeric limits for individual BETX and PAH parameters.

Nevertheless, oil constituents can be highly toxic to aquatic life, and may inhibit reproduction and cause organ damage or mortality (Howarth 1989). The effects of petroleum oils in fish include: impaired reproduction and growth, blood disorders, liver disorders, kidney disorders, malformations, altered respiration or heart rate, altered endocrine function in fish, altered behavior, increased gill cells, fin erosion, and death. Oils can also act on the epithelial surfaces of fish, accumulate on gills, and prevent respiration (Howarth 1989, USEPA 1999). In addition, secondary effects have been observed. Oil coating surface waters can interfere with natural processes of re-aeration and increase BOD, depleting the water of oxygen. Exposures to oil and grease fractions in remediation activity discharges would be through surface coating or direct ingestion of material with food items or ingestion of oil and grease constituents incorporated into food items through the food web. The best available effects information available were for other sensitive species. One study available examined the toxic action of the water accommodated fraction (WAF) and chemically-dispersed fraction (CEWAF) of crude oil on smolts of Chinook salmon (Tjeerdema et al. 2007). The results of this study showed that, based on total hydrocarbon content (THC), the mean LC_{50} of the WAF tests (LC_{50} = 7.46 mg/L THC) was approximately 20 fold lower than that of the CEWAF tests (LC_{50} = 155.93 mg/L THC). This suggests that although there were much higher concentrations of total hydrocarbons present in the CEWAF solutions, hydrocarbon bioavailability to salmon smolts was lower under dispersed conditions. [179]

Ethanol (EtOH)

EtOH is subject to a monitor-only requirement.

EtOH is a fuel oxygenate blended with gasoline to replace more toxic oxygenates, and has been used increasingly in the northeast since approximately 2006. EtOH is miscible with water, as

[177] See footnote 174, above.
[178] See footnote 25, above.
[179] See footnote 9, above.

well as many organic solvents. When released into surface water, it will volatilize or biodegrade rapidly and is not expected to adsorb to sediment. However, large releases of ethanol may deplete dissolved oxygen concentrations resulting in levels unable to support aquatic life.[180]

The best available effects information available were for other sensitive species. EtOH has relatively short residence time in the environment and low toxicity, with lethal effects to aquatic life occurring at concentrations between approximately 11,000 mg/L to 34,000 mg/L. Also, available benchmark monitoring levels for EtOH are 13 mg/L for depletion of in stream dissolved oxygen in a large river (most conservative), and 564 mg/L and 63 mg/L for acute and chronic effect concentrations, respectively.[181] These represent the concentrations at which EtOH would be expected to deplete dissolved oxygen levels below those necessary to sustain aquatic life or cause acute and chronic effects, conditions would violate Massachusetts WQSs and New Hampshire WQRs. Further, NHDES used standard risk assessment procedures to derive a comparison value of 0.4 mg/l of ethanol in drinking water as an exposure likely to be without adverse health effects.[182] Cowgill and Milazzo (1991) reported an LC_{50} of 454 mg/L for *Hyallela azteca* and Bowman, et. al (1981) reported an LC_{50} 8,210 mg/L for *Daphnia magna*.[183]

EPA does not currently have information regarding EtOH in discharges covered under this general permit. However, monitoring data available for sites with remediation and/or dewatering discharges covered under *individual* permits in Region 1 indicate that EtOH may be present in similar discharges.[184] In order to determine the extent of this *potential* pollutant in remediation activity discharges and to determine the frequency with which remediation activity discharges may contain EtOH, the draft RGP includes monitoring for EtOH. Unless monitoring data indicate EtOH is present in remediation activity discharges, EPA assumes EtOH is not present in remediation activity discharges, such that effects are extremely unlikely to occur and are therefore discountable. EPA NRWQC are not available for this parameter. If monitoring data indicate EtOH is present in remediation activity discharges, EPA will evaluate whether water quality standards will be met at the point of discharge and that any concentrations prior to full dilution will rapidly dissipate because of high dilution to concentrations less than the minimum level of detection such that effects cannot be meaningfully measured, detected, or evaluated and are therefore insignificant. Further, EPA will evaluate whether discharges cause only a minor and temporary reduction in available prey such that any effects on individual animals are not capable of being meaningfully measured, detected, or evaluated or where the proposed action could potentially cause a permanent reduction in the abundance, availability, accessibility, and quality of prey, it is so small that any effect on listed species cannot be meaningfully measured, detected, or evaluated and are therefore insignificant.

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[180] Large Volume Ethanol Spills – Environmental Impacts and Response Options. MassDEP: July, 2011.
[181] Developed by the New England Interstate Water Pollution Control Commission (NEIWPCC) using guidance included in EPA's *Final Water Quality Guidance for the Great Lakes System* (1995), referred to as Tier II procedures.
[182] New England Interstate Water Pollution Control Commission, *Health, Environmental, and Economic Impacts of Adding Ethanol to Gasoline in the Northeast States, Volume 3, Water Resources and Associated Health Impacts*. July 2001, 129 pp.
[183] Summarized in Gibbons, J.H. *Interagency Assessment of Oxygenated Fuels*. October 1, 1999; 260 pp.
[184] See, for example, Discharge Monitoring Reports for MA0000825, MA0001929, MA0003280, MA0003298, MA0003425 and MA0004006.

Fuel Additives: Methyl-tert-Butyl Ether (MtBE), tert-Amyl Methyl Ether (tAME), and tert-Butyl Alcohol (tBA)
Potential effects of fuel oxygenates include behavioral, growth, immobilization, reproductive and equilibrium effects.[185]

Methyl *tert*-Butyl Ether (MtBE) is limited to 20 µg/L in freshwater and saltwater, equivalent to the final drinking water advisory for MtBE issued by EPA in 1998, based on the odor threshold for MtBE.[186] EPA NRWQC are not available for this parameter. MtBE is also limited to a maximum of 70 µg/L, which is equivalent to State groundwater quality standards.

MtBE is a synthetic compound used as a replacement for lead-containing compounds in fuels. MtBE was typically added in concentrations less than 1% by volume in regular gasoline, and 2% to 9% by volume in premium gasoline, but increased to 11% to 15% by volume following the 1990 Clean Air Act oxygen content requirements. MtBE has a small molecular size and a high solubility in water.[187] MtBE is also persistent in the environment, and can exhibit high resistance to biological degradation. [188] Where biodegradation does occur, toxic degradation products such as acetone, tBA and *tert*-Butyl formate can be generated.[189] The best available effects information available were for other sensitive species. LC_{50} exposure concentrations have been reported for fathead minnow by Veith et. al. (1983a, 1983b) at 706 mg/L and by Geiger et. al. (1988) at 672 mg/L.[190] Under review of EPA's NRWQC in 1999, the approximate freshwater acute and chronic exposure concentrations for MtBE were determined to be 151 mg/L and 51 mg/L, respectively. For saltwater, the acute and chronic exposure concentrations for MtBE were determined to be 53 mg/L and 18 mg/L.[191]

***tert*-Amyl Methyl Ether (tAME)** is limited to 120 µg/L in Massachusetts, equivalent to the State drinking water guideline, and 40 µg/L in New Hampshire, equivalent to the State drinking water standard, derived using available toxicological data with a 10-fold reduction. EPA NRWQC are not available for this parameter.

tAME is an ether and fuel oxygenate. Oxygenates tend to leach to groundwater because they do not strongly adsorb to soil and are fairly water soluble. The best available effects information available were for other surrogate, potential prey, or sensitive species. LC_{50} exposure

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[185] See footnote 183, above.
[186] Drinking Water Advisory Table in *2012 Edition of Drinking Water Standards and Health Advisories*. U.S. EPA, 2012: p 12.
[187] *MtBE in Drinking Water*. NHDES Environmental Fact Sheet WD-DWGB-3-19: 2014.
[188] *Chapter 13: MTBE* in *Regulatory Determinations Support Document for Selected Contaminants from the Second Drinking Water Contaminant Candidate List (CCL 2*). EPA Report 815-R-08-012: June 2008.
[189] *Technologies for Treating MTBE and Other Fuel Oxygenates*. EPA 542-R-04-009: 2004, 106 pp. EPA Technology Innovation and Field Services Division Contaminated Site Clean-Up Information.
[190] See footnote 183, above.
[191]Mancini, E.R. et. al. *"*MTBE Ambient Water Quality Criteria Development: A Public/Private Partnership*". Environmental Science and Technology (2002, Volume 36, pages 125-129: 2002.* See also, EPA-822-F-06-002: March 2006.

concentrations have been reported for rainbow trout (API, 1995d) at 580 mg/L and for shrimp, *Mysidopsis bahia* (API, 1995c) at 14 mg/L.[192]

***tert*-Butyl Alcohol (tBA)** is limited to 90 µg/L in Massachusetts, equivalent to the State drinking water guideline, and 140 µg/L in New Hampshire, equivalent to the State drinking water standard, derived using available toxicological data with a 10-fold reduction. EPA NRWQC are not available for this parameter.

tBA is a fuel additive, chemical additive, solvent and intermediate. tBA is also a major breakdown product of EtBE and MtBE in the environment. Some tBA may occur naturally as a product of fermentation. tBA will rapidly volatilize when released to surface water. tBA is soluble in water and is also miscible with alcohol, ether, and other organic solvents.[193] The best available effects information available were for other sensitive species. LC_{50} exposure concentrations have been reported for fathead minnow by Geiger et. al. (1988) at 6,410 mg/L and for midge larvae, *Chironomus riparius,* by Roghair et. al. (1994) at 5,800 mg/L.[194]

Effects Determination for Fuels Parameters

Based on the best available information, EPA has made the determination that the effects from the five aforementioned fuels parameters on shortnose sturgeon, Atlantic sturgeon, or their prey, will be insignificant and/or discountable because if one or more of these parameters is present in a discharge authorized under the RGP:

1) Water quality standards are met at the point of discharge: The numeric limits do not exceed available effects data, suggesting that, if the numeric limits are taken to represent the surrogate instream constituent exposure concentrations, the numeric limits are an acceptable proxy and any effects are extremely unlikely to occur and are therefore discountable. Further, given high dilution in the Action Area waterbodies, the discharge concentrations will result in the use of only a small portion of the available assimilative capacity of the nearshore marine waters of Massachusetts and New Hampshire or in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River such that cumulative effects from the environmental persistence of fuels parameters are extremely unlikely to occur and are therefore discountable.

With respect to prey quantity/quality, EPA has made the determination that the proposed action will be insignificant because if one or more of these fuels parameters are present in a discharge authorized under the RGP:

1) Discharges are likely to cause only a minor and temporary reduction in available prey, if any, such that any effects on individual animals are not capable of being meaningfully measured, detected, or evaluated: Discharges eligible for coverage under this general permit are expected to occur with low frequency (intermittent), small magnitude (small

[192] See footnote 183, above.
[193] *IRIS Toxicological Review of tert-Butyl Alcohol (tert-Butanol)(Public Comment Draft).* EPA/635/R-16/079a: April 2016.
[194] See footnote 183, above.

volume limited to no more than 1.0 MGD), and short duration (temporary or short-term) as such are likely to cause effects minor and temporary in nature. The discharge must meet numeric limits established at concentrations below available estimated effects concentrations at end-of-pipe. If any of these pollutants are present in remediation activity discharges, these pollutants are expected to undergo rapid, full mixing because of the high dilution in the receiving waters to concentrations less than the minimum level of detection such that effects are likely to cause only a minor and temporary change in the abundance, distribution, quality and availability of only the most sensitive prey species. This minor and temporary loss or alteration is not expected to affect the way that individual animals use the Action Area or result in behavior change (e.g., foraging) in individual animals that can be meaningfully measured, detected, or evaluated and is therefore insignificant.

Consequently, the proposed action is not likely to adversely affect the shortnose sturgeon, the Atlantic sturgeon or their prey in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River, or the coastal embayments/nearshore marine waters of Massachusetts and New Hampshire.

pH

The hydrogen-ion (H^-) concentration in an aqueous solution is represented by the pH using a logarithmic scale of 0 to 14 standard units (SU). Solutions with pH 7.0 SU are neutral, while those with pH less than 7.0 SU are acidic and those with pH greater than 7.0 SU are basic. Of note, although basic solutions are alkaline, basicity and alkalinity are not exactly the same. Basicity refers to the ratio of hydrogen and hydroxyl (OH^-) ions in solution, and is directly related to pH. Alkalinity is related to the acid-neutralizing capacity of a solution. In aquatic ecosystems, biological processes (e.g., decomposition) that increase the amount of dissolved carbon dioxide or dissolved organic carbon decrease pH but have no effect on acid-neutralizing capacity.[195]

Effluent with pH values markedly different from the receiving water pH can have a detrimental effect on the environment. Sudden pH changes can kill aquatic life. According to NMFS, "the pH of water affects the normal physiological functions of aquatic organisms. [These] processes operate normally in most aquatic biota under a relatively wide pH range (e.g., 6 – 8.5 pH units). There is no definitive range within which all freshwater aquatic life is unharmed and outside which adverse impacts occur."[196] As NMFS indicated in a November 4, 2013 ESA concurrence letter to EPA regarding the Lawrence Hydroelectric Project under the NPDES Hydroelectric Generating Facility General Permit, a pH range of 6.0 to 9.0 is harmless to most marine/aquatic organisms, including the ESA listed species of shortnose and Atlantic sturgeon

As summarized in Table 1 of Section 4.A.1, the pH range designated by the Massachusetts Water Quality Standards for Class A and B Inland waters is from 6.5 to 8.3 SU while the pH range for

[195] Summarized from U.S. Environmental Protection Agency, Entry: Causal Analysis/Diagnosis Decision Information System, Volume 2: Sources, Stressors & Responses, pH. Available at http://www.epa.gov/caddis/index.html.
[196] See footnote 9, above.

Class SA and Class SB waters is 6.5 to 8.5 SU. According to the Surface Water Quality Regulations for the State of New Hampshire, the pH range shall be 6.5 to 8.0 SU, unless due to natural causes. As previously mentioned, New Hampshire does not allow discharge into Class A waters.

The effluent limits for pH in the Draft RGP are established to be consistent with the aforementioned water quality standards in Massachusetts and New Hampshire. Based on these water-quality standards, the Draft Permit contains the following limits for the indicated waterbody classifications.

1) Massachusetts Class B: 6.5 – 8.3 standard units
2) Massachusetts Class SB: 6.5 – 8.5 standard units
3) New Hampshire Class B: 6.5 – 8.0 standard units

Also, the permit indicates there shall be no change from natural conditions that would impair any uses assigned to the receiving water. EPA, with State approval, may expand the pH range to the federal standard of 6.0 to 9.0 SU, where the more restrictive pH limits cannot be consistently achieved by the treatment system, and where receiving water quality and dilution characteristics allow state water quality standards to be achieved. Refer to Part 2.1.1 of the General Permit for Massachusetts sites and Part 3.1.1 of the General Permit for New Hampshire sites.

pH data collected by the USGS station 01172010 indicate that the median pH in the Connecticut River since 2005 (beginning of water quality dataset) is 7.0 SU with a minimum recorded value of 6.6 SU and a maximum recorded value of 7.6 SU. pH data collected by the USGS station 01100000 indicate that the median pH in the Merrimack River since 2003 is 7.0 SU with a minimum recorded value of 6.3 SU and a maximum recorded value of 7.4 SU. These data are within the threshold values for pH that are harmless to aquatic organisms. pH data collected by the USGS station 01108000 indicate that the median pH in the Taunton River is 6.5 SU with a minimum recorded value of 5.7 SU and a maximum recorded value of 7.6 SU. The majority of pH measurements are within the threshold values for pH that are harmless to aquatic organisms with only 6 of the 53 measurements below a pH of 6.0 SU. pH data provided by EPA's STORET database for the Piscataqua River Basin indicate that the median pH in waters within the Piscataqua River Basin from 1984-2015 is 6.55 SU.

Given the high available dilution in the waterbodies in the Action Area, the effect from individual remediation activity discharges on pH is extremely unlikely to cause a change in the instream pH concentration. Further, since the pH effluent limit for the draft RGP falls within the 6.0 to 9.0 SU range, EPA has made the determination that any effects from the pH range from remediation activity discharges on the shortnose sturgeon and the Atlantic sturgeon are extremely unlikely to occur and are therefore discountable. Consequently, the proposed action is not likely to adversely affect the ESA-listed species in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River, including the coastal embayments/nearshore marine waters of Massachusetts and New Hampshire.

Temperature

Section 502(6) of the Clean Water Act defines heat as a "pollutant." 33 U.S.C. § 1362(6). Therefore, thermal effluent, such as cooling water or boiler blowdown, is considered a pollutant, and such discharges require a NPDES permit. Changes in the ambient thermal profile can alter the toxic effect of certain pollutants. The draft RGP contains effluent limitations for temperature based on State WQSs. To evaluate any potential thermal impacts that would exceed applicable water quality criteria for authorized discharges, EPA requires all applicants for the RGP to report discharge and ambient temperature conditions during in the NOI submitted to EPA. If EPA determines that a measurable thermal effluent may be discharged, an applicant will be subject to end of pipe temperature limitations. Thermal effluents are not typical under this general permit.

Of the protected species included in this document, the two protected sturgeon species may be the most sensitive to a potential temperature elevation from an RGP discharge. This is because adult sturgeon have a relatively smaller body size (compared to marine mammals and reptiles) and there is the potential for early life stages of sturgeon to be present in the rivers where an RGP authorized discharges may occur. The RGP does not allow for the discharges of effluent relating to cooling water intake structures. The temperature of the discharge is not generally a pollutant of concern for this general permit. However, effluent limitations for temperature have been established in the RGP as a conservative measure, and may be applied on a case-by-case basis in the event such limitations are necessary. The current water quality of Class B waters must be maintained. Under Env-Wq 17013.13, New Hampshire provides a narrative (not a numeric) standard for water temperature. Unlike Class A waters, a temperature change is allowed in Class B waters. However, any stream temperature increase of Class B waters shall not be such as to "appreciably interfere with the uses assigned" to the class of water (RSA 485-A:8.II). In Massachusetts Class B waters, thermal discharges shall not exceed 83°F (28.3°C) in warm water fisheries.

The temperature preference for shortnose sturgeon is not known (Dadswell et al. 1984) but shortnose sturgeon have been found in waters with temperatures as low as 2 to 3°C (Dadswell et al. 1984) and as high as 34°C (Heidt and Gilbert 1978). However, temperatures above 28°C are thought to adversely affect shortnose sturgeon. NMFS has indicated that shortnose sturgeon (and presumably Atlantic sturgeon) may be adversely affected by moderate to long term exposure to temperatures above 29°C (approximately 84°F) and are likely to display avoidance behaviors of waters of this temperature (NMFS correspondence for Cabot, August, 2012 and Lawrence Hydroelectric Project, November, 2013 under EPA's NPDES Hydroelectric Generating Facility General Permit). Since discharges under the RGP are expected to be of short duration and small volume, any thermal discharge associated with the outfalls will not have the potential to create moderate to long term thermal exposure conditions to protected species.

More importantly, the RGP does not allow discharges greater than 83°F (28.3°C) at the end-of-pipe, before mixing (in Massachusetts waters), and does not allow any thermal discharge to "appreciably interfere with the uses assigned" (in New Hampshire waters). Because of these requirements, EPA has made the determination that the impact of thermal pollution, if any, on the shortnose sturgeon and the Atlantic sturgeon, if present in the vicinity of discharges authorized under the RGP, will be so small they cannot be detected and are therefore

insignificant in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River, or the coastal embayments/nearshore marine waters of Massachusetts and New Hampshire.

Dissolved Oxygen

While the discharge of pollutants/parameters potentially present in remediation activity discharges have the potential to impact dissolved oxygen concentrations in the Connecticut, Merrimack, Taunton, and Piscataqua Rivers, and the shoreline areas in Massachusetts and New Hampshire, it is beyond the capability of EPA to estimate specific impacts on dissolved oxygen from individual discharges. However, data provided by EPA's STORET database indicate that the median dissolved oxygen concentration is 7.6 mg/L for waters in the Piscataqua River Basin, is 10.55 mg/L in the Connecticut River since 2005, with a minimum recorded value of 8.0 mg/L (at USGS station 01100000), is 9.0 mg/L in the Merrimack River since 2003, with a minimum recorded value of 6.8mg/L (at USGS station 01100000), and is 7.0 mg/L in the Taunton River (at USGS station 01108000). These median concentrations are above the threshold value for dissolved oxygen that could affect the listed species included in EPA's effects determination.

EPA has made the determination that the water quality effects from the pollutants in discharges authorized under the RGP, if present, on the shortnose sturgeon and the Atlantic sturgeon, will be discountable and/or insignificant in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River, including the coastal embayments/nearshore marine waters of Massachusetts and New Hampshire, because:

1) Water quality standards are met at the point of discharge: In Massachusetts, Class B waterbodies must attain a minimum DO of 5.0 mg/L in warm water fisheries and 6.0 mg/L in cold water fisheries. Massachusetts Class SB waterbodies must attain a minimum DO of 5.0 mg/L. In New Hampshire, Class B waterbodies must attain a minimum DO of 5.0 mg/L. Non-numeric permit conditions require all remediation activity discharges meet the relevant water quality standards. Available data do not exceed water quality standards, suggesting that, if the water quality criteria for DO are taken to represent the allowable discharge concentrations, and the non-numeric limit requires these criteria be met, the limit is an acceptable proxy and any effects to listed species are extremely unlikely to occur and are therefore discountable;
2) Any exposure to the discharge prior to full dilution would have insignificant effects: If a remediation activity discharge consisting of a combination of pollutants that through additive, cumulative and/or synergistic effects contained impaired dissolved oxygen concentrations, even if the effluent is discharged to surface water at the maximum allowable effluent flow, the discharge will rapidly dissipate because of high dilution such that effects that differ from the ambient DO concentrations and as such any potential effects on listed species cannot be meaningfully measured, detected, or evaluated and are therefore insignificant.

With respect to prey quantity/quality, EPA has made the determination that the water quality effects from the pollutants in discharges authorized under the RGP, if present, on the prey of shortnose sturgeon and the Atlantic sturgeon, will be insignificant in the shoreline areas of Massachusetts and New Hampshire or the Connecticut River, Merrimack River, Taunton River, or Piscataqua River because:

1) discharges will potentially cause only a minor and temporary reduction in available prey such that any effects on individual animals are not capable of being meaningfully measured, detected, or evaluated and are therefore insignificant.

Consequently, the proposed action is not likely to adversely affect the shortnose sturgeon, the Atlantic sturgeon or their prey in the Connecticut River, Merrimack River, Taunton River, or Piscataqua River, or the coastal embayments/nearshore marine waters of Massachusetts and New Hampshire.

c. Potential Effects of the Action on Essential Elements of Proposed Critical Habitat

In EPA's opinion, the requirements proposed in the draft RGP for eligible discharges will cause no adverse modification to proposed/designated critical habitat of ESA-listed species for several detailed, specific reasons.

First, the proposed limits will cause no adverse modification to critical habitat because the discharges must meet the stringent requirements specified in the draft RGP. As previously discussed, the draft RGP contains numeric and non-numeric effluent limitations and special conditions. These include the prohibition of discharges of toxic substances in toxic amounts, influent and effluent monitoring and reporting, and require whole effluent toxicity testing of certain discharges to ensure discharges meet State WQSs for a wide variety of potential pollutants. All potential pollutants included the draft RGP are not generally expected in a single discharge; rather, discharges typically contain a very small subset of pollutants, depending on the source of contamination at a site. Because this general permit is designed for a variety of potential situations, the effluent limitations in the draft RGP, excepting a small number of parameters (e.g., total recoverable metals), have been established conservatively as TBELs equivalent to human health- or risk-based water quality criteria, imposed near or below analytical minimum levels of detection, with no allowable dilution. These effluent limitations are as stringent as or more stringent than water quality criteria for the protection of aquatic life. Therefore, while human health and/or risk-based effluent limitations are not specifically derived for the protection of aquatic life or critical habitat, such limitations are an appropriate proxy because any potential effects at concentrations that could potentially occur near or below analytical levels of detection cannot be meaningfully measured, detected or evaluated. Thusly, the water quality of the critical habitat will experience no adverse modifications from the proposed action.

Second, although the RGP does not require the use of specific treatment technologies, treatment technologies must be employed at these sites if necessary to meet effluent limitations. See Part 2.5 of the 2016 RGP for treatment technology requirements. The types of treatment technology employed routinely produce high quality effluent, typically at concentrations below laboratory minimum levels of detection (i.e., "non-detect). The types of treatment include, but are not limited to: 1) adsorption/absorption; 2) advanced oxidation processes; 3) air stripping; 4) granulated activated carbon/liquid phase carbon adsorption; 5) ion exchange; 6) precipitation/coagulation/flocculation; and 7) separation/filtration. Further, the RGP requires

operators to implement BMPs, including the basic requirements listed in Part 2.5 of the RGP. EPA has judged that discharges treated by these technologies will cause no adverse modification to critical habitat.

Third, the majority of discharges to be covered under this general permit are generated through batch operations. A batch operation occurs with low frequency (intermittent), consists of a small magnitude (low volumes not to exceed 1.0 MGD), and continues for a short duration (temporary and short-term). The design flow of the discharges covered by this general permit typically range from a few gallons per minute (GPM) to approximately 50 GPM. Approximately half of the remediation and/or dewatering activities covered by the 2005 and 2010 RGP have lasted less than one year in duration, many lasting only a few days or weeks. The discharges themselves are not continuous. EPA expects that these characteristics will further support the judgment that the discharges will result in no adverse modification to critical habitat.

Fourth, the draft RGP allows States to add additional requirements for CWA §401 certification. The 2016 RGP also allows EPA to require toxicity testing if necessary to ensure that a discharge is not having a toxic effect on sensitive species. EPA can revoke coverage under this general permit at any time if an adverse modification to critical habitat occurs, either because of non-compliance or from unanticipated effects from a discharge. Similarly, using the Notice of Intent review process, EPA shall require an individual permit for a remediation site and associated discharge location that may not meet the "no adverse modification to critical habitat" threshold.

In conclusion, discharges eligible for coverage under the RGP will cause no adverse modification to proposed/designated critical habitat of ESA-listed species for the following reasons:

1) This general permit action does not constitute a new source of pollutants; it is the reissuance of an existing NPDES general permit. No adverse modification to critical habitat was documented under the previous issuance of this general permit;
2) The RGP prohibits the addition of materials or chemicals in amounts that would be toxic to aquatic life. This prohibition results in no adverse modification to critical habitat;
3) The effluent limitations proposed in the RGP ensure protection of aquatic life and maintenance of the receiving waters as aquatic habitat. This protection of aquatic life and aquatic habitat results in no adverse modification to critical habitat;
4) Discharges eligible for coverage under this general permit are primarily a result of site remediation (i.e., treatment to regulatory clean up levels) or dewatering of formerly contaminated sites (i.e., former remediation sites that achieved regulatory clean up levels). These "clean up levels" contribute to the judgment that no adverse modification to critical habitat is expected;
5) Discharges eligible for coverage under this general permit are generally expected to occur with low frequency (intermittent), small magnitude (small volume limited to no more than 1.0 MGD), and short duration (temporary or short-term); therefore, any potential effects of the discharges on receiving waters are expected to be proportionately small and subject to a large dilution factor when discharged to the receiving water. These characteristics of the discharges contribute to the judgment that no adverse modification to critical habitat is expected;

6) The proposed effluent limitations in the RGP are sufficiently stringent to ensure that State and Federal water quality standards will be met. Meeting protective water quality standards results in no adverse modification to critical habitat;
7) For the majority of limited pollutants, effluent limitations are applied at or below water quality criteria, with no allowable dilution (i.e., “end-of-pipe”). This conservative regulatory approach further contributes to the judgment that no adverse modification to critical habitat is expected;
8) If any pollutant is present at a site at a level that does not meet the effluent limitation for that pollutant, the operator at that site is required to utilize pollution control technologies that will, at a minimum, reduce the level of that pollutant to the effluent limitation. The use of pollution control technologies to achieve prescribed limitations contributes to the judgment that no adverse modification to critical habitat is expected; and
9) Discharges that have the potential to result in the adverse modification or destruction of habitat that is designated as critical under ESA are expressly prohibited and will not be authorized under this general permit. EPA will use the Notice of Intent screening process to determine whether an applicant would be prohibited from authorization to discharge under this general permit due to a potential adverse modification to critical habitat.

i. Proposed Critical Habitat for Atlantic Sturgeon

On June 3, 2016, NMFS issued two proposed rules to designate critical habitat for the five listed distinct population segments (DPSs) of Atlantic sturgeon found in U.S. waters (Gulf of Maine, New York Bight, and Chesapeake Bay DPSs: 81 FR 35701; Carolina and South Atlantic DPSs: 81 FR 36078). Federal agencies are required to confer with NFMS on any action that is likely to jeopardize the continued existence of any species proposed for listing or result in destruction or adverse modification of proposed critical habitat (50 CFR §402.10). "Destruction or adverse modification" is defined as a direct or indirect alteration that appreciably diminishes the value of critical habitat for the conservation of a listed species (50 CFR § 402.02).

The proposed rules identified the following four essential physical and biological features (PBFs) necessary for the conservation of the species. The term “physical and biological features” is defined as the features that support the life-history needs of the species, including, but not limited to, water characteristics, soil type, geological features, sites, prey, vegetation, symbiotic species or other features. For example, physical features essential for Atlantic sturgeon reproduction and recruitment are:

1) Hard bottom substrate (e.g., rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (i.e., 0.0 to 0.5 parts per thousand range) for settlement of fertilized eggs, refuge, growth, and development of early life stages;
2) Aquatic habitat with a gradual downstream salinity gradient of 0.5 to 30 parts per thousand and soft substrate (e.g., sand, mud) downstream of spawning sites for juvenile foraging and physiological development;
3) Water of appropriate depth and absent physical barriers to passage (e.g., locks, dams, reservoirs, gear, etc.) between the river mouth and

spawning sites necessary to support:
(1) Unimpeded movement of adults to and from spawning sites; (2) seasonal and physiologically dependent movement of juvenile Atlantic sturgeon to appropriate salinity zones within the river estuary; and (3) staging, resting, or holding of subadults or spawning condition adults. Water depths in main river channels must also be deep enough (e.g.,2:1.2 m) to ensure continuous flow in the main channel at all times when any sturgeon life stage would be in the river; and

4) 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 interannual adult, subadult, larval, and juvenile survival; and (3) larval, juvenile, and subadult growth, development, and recruitment (e.g., l3°C to 26°C for spawning habitat and no more than 30°C for juvenile rearing habitat, and 6 mg/L dissolved oxygen for juvenile rearing habitat).

NFMS has proposed to designate Atlantic sturgeon critical habitat for the Gulf of Maine DPS in the Piscataqua River from its confluence with the Salmon Falls and Cocheco rivers downstream to where the main stem river discharges at its mouth into the Atlantic Ocean, as well as the waters of the Cocheco River from its confluence with the Piscataqua River and upstream to the Cocheco Falls Dam, and waters of the Salmon Falls River from its confluence with the Piscataqua River and upstream to the Route 4 dam. The proposed action *could* authorize remediation activity discharges into these rivers.

As previously described, discharges eligible for coverage under this general permit are generally expected to occur with low frequency (intermittent), small magnitude (small volume limited to no more than 1.0 MGD), short duration (temporary or short-term), and are expected to experience high dilution factors and immediate and complete mixing with the receiving water.

Section 4.b. discusses how the effluent limitations for pollutants such as TSS will support PBF #1, which includes the need for Atlantic sturgeon to have clean, hard substrate. Section 4.b. also discusses the potential effects from pollutants found in remediation activity discharges and the steps taken (e.g., effluent limits set or Best Management Practices (BMPs) required) to regulate such discharges. This includes a BMP Plan with a focus on pollutant controls through use of treatment technologies whenever necessary and adequate site controls to ensure that any construction or land/water disturbing activities do not result in the discharge of pollutants to receiving waters. Based on this information, EPA does not believe remediation activity discharges will lead to destruction or adverse modification of proposed critical habitat, especially PBF #3 which requires a water column that is absent from physical barriers to passage (e.g., locks, dams, reservoirs, gear, etc.)

The discharges covered under this General Permit primarily consist of freshwater, except where groundwater and brackish/saltwater interact along the immediate shoreline. The draft RGP also contains effluent limitations for chloride, if a discharge is expected to exceed water quality standards. This effluent limitation has generally applied to a limited number of sites. However, sites do not utilize membrane desalination operations (which desalinate seawater/saltwater into

freshwater for treatment purposes). Therefore, EPA does not have any specific data on the salinity of remediation activity discharges since it is not a parameter of concern for these types of facilities. Since remediation activity discharges are small, localized, and primarily consist of freshwater, EPA does not believe they will lead to destruction or adverse modification of proposed critical habitat, especially PBFs #1, 2, and 4 which include specific salient gradients and zones.

Similarly, the RGP does not allow for the discharges of effluent relating to cooling water intake structures. The temperature of the discharge is not generally a pollutant of concern for this general permit. However, effluent limitations for temperature have been established in the RGP as a conservative measure and may be applied on a case-by-case basis in the event such limitations are necessary. The current water quality of the Piscataqua River, which is classified by the State as Class B, must be maintained. Under Env-Wq 17013.13, New Hampshire provides a narrative (not a numeric) standard for water temperature. Unlike Class A waters, a temperature change is allowed in Class B waters. However, any stream temperature increase of Class B waters shall not be such as to "appreciably interfere with the uses assigned" to the class of water (RSA 485-A:8.II). One such use for surface waters (Class A and Class B waters) includes Biological and Aquatic Community Integrity. This supports PBF #4.

EPA has determined that the discharges authorized under this general permit are not likely to adversely affect the four PBFs identified as essential for the proposed critical habitat of Atlantic sturgeon in the Piscataqua, Cocheco, or Salmon Falls Rivers, based on the following:

1) The effluent limitations and requirements established in the draft RGP, including numeric limitations for TSS, pH, temperature, and toxic pollutants that could impact dissolved oxygen, ensure that state and federal water quality standards will be met, including water quality standards in accordance with Env-Wq 1703.19 of New Hampshire's surface water quality regulations that require surface waters to "be free from toxic substances or chemical constituents that injure or are inimical to plants, animals, humans or aquatic life." This supports PBF #4.
2) The effluent limitations and requirements established in the draft RGP ensure the protection of aquatic life and maintenance of the receiving water(s) as an aquatic habitat, including water quality standards in accordance with Env-Wq1703.21 of New Hampshire's surface water quality regulations that require surface waters to "support and maintain a balanced, integrated, and adaptive community of organisms". This support PBFs #1, #3 and #4.

Based on the aforementioned reasons, EPA believes that no destruction or adverse modification of proposed critical habitat (which includes clean, hard substrate) for Atlantic sturgeon in the Piscataqua, Cocheco, or Salmon Falls Rivers will occur and no conference is necessary.

d. Indirect Effects

Indirect effects are those that are caused by the proposed action and are later in time, but still are reasonably certain to occur. The RGP requires operators to comply with effluent limitations and requirements by using appropriate BMPs, when necessary, and to meet Federal and State water

quality standards for all receiving waters. The BMP requirements include a written plan designed to reduce or eliminate the discharge of pollutants in discharges from a site as well as a special conditions pertaining to pollutant minimization, including prohibitions pertaining to the discharge of chemicals and additives. These requirements include operational and preventative maintenance activities, an examination of control measures to minimize pollutant discharges, and procedures and schedules for the management and removal of waste. The proposed action does not *authorize or require* any structural disturbance on adjacent land or in/near the waterbodies within the defined Action Area. Further, discharges under this general permit are authorized only during the remediation activity. Discharges eligible for coverage under this general permit are expected to occur with low frequency (intermittent), small magnitude (low volume limited to no more than 1.0 MGD, typically approximately 0.0072 MGD to 0.072 MGD), and short duration (temporary or short-term, typically from 24 hours up to 12 months in duration). Following completion of the remediation activity, discharges and the authorization to discharge are terminated. As a result, any exposure to discharges, if at all, would be brief. Therefore, indirect effects to the shortnose sturgeon and Atlantic sturgeon are extremely unlikely to occur later in time and are therefore discountable.

e. Effects from Interdependent and Related Actions

Interdependent actions are defined as actions with no independent use apart from the proposed action. Interrelated actions include those that are part of a larger action and depend on the larger action for justification. No interdependent/interrelated actions are expected to result from the reissuance of the NPDES permit for remediation activity discharges within the states of Massachusetts and New Hampshire.

5. Conclusions

Based on the analysis that all effects of the proposed action will be insignificant and/or discountable, EPA has determined that the proposed reissuance of the RGP is **not likely to adversely affect (NLAA)** the shortnose sturgeon, Atlantic sturgeon (or its proposed Critical Habitat). EPA certifies that we have used the best scientific and commercial data available to complete this analysis. EPA requests your concurrence with this determination.

Sincerely,

David M. Webster, Chief
Water Permits Branch

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