National Marine Fisheries Service
Endangered Species Act Section 7

Biological Opinion

Title:	Biological Opinion on EPA Approval of Water Quality Criteria Proposed for Adoption by Massachusetts and New Hampshire
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1 INTRODUCTION

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

Section 7(b)(3) of the ESA requires that at the conclusion of consultation NMFS provides an opinion stating whether the Federal agency's action is likely to jeopardize ESA-listed species or destroy or adversely modify designated and proposed critical habitat. If NMFS determines that the action is likely to jeopardize listed species or destroy or adversely modify critical habitat, NMFS provides a reasonable and prudent alternative that allows the action to proceed in compliance with section 7(a)(2) of the ESA. If the action (or a reasonable and prudent alternative) is expected to cause incidental take without violating section 7(a)(2), section 7(b)(4), as implemented by 50 CFR §402.14(i), requires NMFS to provide an incidental take statement (ITS), which specifies: the impact (i.e., amount or extent of take) of incidental take; reasonable and prudent measures (RPMs) determined necessary or appropriate to minimize such impacts and terms and conditions to implement the RPMs; and, procedures to be used to handle or dispose of any individual species actually taken. Incidental take must also be monitored and reported as the action proceeds and consultation must be immediately reinitiated should the amount or extent of incidental take specified in the ITS be exceeded. Any incidental take which occurs in compliance with the ITS is exempted from the ESA's prohibition on take. The protection from the prohibition on take may lapse if the action agency fails to comply with the RPMs or terms and conditions included in the ITS.

The Federal action agency for this consultation is the U.S. Environmental Protection Agency Region 1 (EPA). The EPA requested ESA section 7 consultation for the approval of certain aquatic life-based Water Quality Criteria for the States of Massachusetts and New Hampshire under Section 303(c) of the Clean Water Act.

This consultation, Opinion, and associated ITS were completed in accordance with ESA section 7, associated implementing regulations (50 CFR §§402.01-402.16),[1] and agency policy and guidance (NMFS/USFWS 1998). On July 5, 2022, the United States District Court for the Northern District of California issued an order vacating the 2019 regulations adopting changes to 50 CFR part 402 (84 FR 44976, August 27, 2019). This consultation was initiated when the 2019 regulations were still in effect. As reflected in this document, we are now applying the section 7 regulations that governed prior to adoption of the 2019 regulations (https://www.govinfo.gov/content/pkg/CFR-2018-title50-vol11/pdf/CFR-2018-title50-vol11-part402.pdf). For purposes of this consultation, we considered whether the substantive analysis and its conclusions regarding the effects of the proposed actions articulated in the biological opinion and incidental take statement would be any different under the 2019 regulations. We have determined that our analysis and conclusions would not be any different.

The NMFS Office of Protected Resources (OPR) Endangered Species Act Interagency Cooperation Division (hereafter referred to as "we" or "our") conducted this consultation.

This document represents NMFS' Opinion on the effects of these actions on Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*, Gulf of Maine Distinct Population Segment [DPS], New York Bight DPS, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs); shortnose sturgeon (*Acipenser brevirostrum*); Kemp's ridley (*Lepidochelys kempii*), leatherback (*Dermochelys coriacea*), hawksbill (*Eretmochelys imbricata*), and loggerhead sea turtles (*Caretta caretta*, Northwest Atlantic Ocean DPS); North Atlantic right (*Eubalaena glacialis*) blue (*Balaenoptera musculus*), fin (*Balaenoptera physalus*), and sei (*Balaenoptera borealis*) whales; and critical habitat designated for Gulf of Maine and New York Bight DPSs of Atlantic sturgeon and North Atlantic right whale.

A complete record of this consultation was filed electronically by the NMFS Office of Protected Resources in Silver Spring, Maryland.

1.1 Background

Under the ESA, it is the policy of Congress that all federal agencies shall seek to conserve threatened and endangered species, use their authorities in furtherance of the ESA, and cooperate with state and local agencies to resolve water resource issues in concert with conserving endangered species (16 U.S.C.§1531. Congressional findings and declaration of purposes and

[1] On July 5, 2022, the United States District Court for the Northern District of California issued an order vacating the 2019 regulations adopting changes to 50 CFR part 402 (84 FR 44976, August 27, 2019). This consultation was initiated when the 2019 regulations were still in effect. As reflected in this document, we are now applying the section 7 regulations that governed prior to adoption of the 2019 regulations (https://www.govinfo.gov/content/pkg/CFR-2018-title50-vol11/pdf/CFR-2018-title50-vol11-part402.pdf). For purposes of this consultation, we considered whether the substantive analysis and its conclusions regarding the effects of the proposed actions articulated in the Opinion and its incidental take statement would be any different under the 2019 regulations. We have determined that our analysis and conclusions would not be any different.

policy Pervaze et al. 2021). Water quality standards are regulations established under the Clean Water Act that are intended to: protect public health and welfare; enhance the quality of water; restore and maintain the chemical, physical, and biological integrity of state waters; and provide water quality protection and propagation of fish, shellfish, and wildlife, and recreation in and on the water. A state's Water Quality Standards include designated uses and narrative or numeric criteria[2] to protect those uses. Narrative water quality criteria describe the desired conditions of a water body as being "free from" certain negative conditions. Numeric water quality criteria are maximum allowable concentrations of toxic pollutants or acceptable aquatic chemistry conditions (e.g., pH or temperature range, nutrients). The uses designated for state waters inform the narrative and numeric water quality criteria that will apply for each use.

Water quality criteria are also used to determine use attainment of waters through monitoring, permitting limits for point and nonpoint source discharges to waters, and in setting loading limits to restore pollution-impaired waters. Because the criteria set the exposure conditions for each stressor, each analysis determines whether adverse effects may result from exposure to the stressor within the limits of its acute and chronic criteria. Facilities are required to monitor for components of their discharge that have a reasonable potential to cause an aquatic impairment. Section 303(c)(2)(B) of the Clean Water Act requires states adopt numeric criteria for all toxic pollutants for National Recommended Water Quality Guidelines (National Criteria) that have been published under Section 304(a). Most of National Criteria were developed by EPA under the 1985 EPA Guidelines for Deriving Numerical National Water Quality Criteria (EPA Guidelines Stephen et al. 1985). Some National Criteria are calculated using models that account for the effects of site-specific aquatic chemistry on biological availability and thus toxicity.

Section 303(c) of the Clean Water Act requires that, at least once every three years, states, tribes, and territories review and, when necessary, modify their water quality standards or adopt new water quality standards to protect waters under their jurisdiction. Implementation of a state's water quality standards can also affect water quality in neighboring states when rivers cross borders or delineate state boundaries. As required by Section 303(c) of the Clean Water Act and 40 CFR Part 131, EPA reviews state and territorial water quality standards, which cannot be implemented until approved by EPA.

In terms of ESA section 7 consultations for Clean Water Act-related actions, the goal of the 2001 Memorandum of Agreement between the EPA, NMFS, and the U.S. Fish and Wildlife Service is to enhance coordination under the Clean Water Act and the ESA for section 7 consultations. EPA consults with the Services on newly proposed and/or revised state aquatic life criteria to ensure that any state or territorial-adopted aquatic life criteria are protective of ESA-listed species and designated critical habitats in waters under that state or territory's jurisdiction and

[2] This Opinion uses the term "criteria" when discussing the numeric water quality criteria EPA proposes to approve to distinguish these from the broader term "water quality standards" that describe the desired condition of water bodies and the means by which conditions will be protected or achieved.

have a Water Quality Standard description that includes the protection and propagation of fish, shellfish, and wildlife.

1.2 Preconsultation

On June 23, 2021, NMFS received a draft Biological Evaluation (BE) from EPA for the effects of EPA approval for Massachusetts' revised aluminum, ammonia, copper, cadmium, and nitrogen site-specific water quality criteria on ESA-listed species under NMFS' jurisdiction. Between that date and January 19, 2022, EPA and NMFS discussed the consultation process, documentation, and other information needed for consultation.

1.3 Consultation History

On January 19, 2022, the National Marine Fisheries Service (NMFS) received a request for written concurrence that the U.S. Environmental Protection Agency Region 1's (EPA) approval of proposed numeric water quality criteria for Massachusetts and New Hampshire "may affect, but is not likely to adversely affect" threatened or endangered species or designated critical habitats under section 7 of the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. 1531 et seq.). The request was accompanied by two BEs: "An evaluation of the potential effects of EPA's approval of state-adopted freshwater aluminum, ammonia, and copper, freshwater and saltwater cadmium, and site-specific saltwater nitrogen aquatic life criteria for Massachusetts" (USEPA 2022) and "An evaluation of the potential effects of state-adopted acrolein, carbaryl, and nonylphenol aquatic life criteria for Massachusetts and New Hampshire" (USEPA 2022).

On February 15, 2022, EPA notified NMFS that they were required to issue an action letter to Massachusetts Department of Environmental Protection (MassDEP) on March 8, 2022 as EPA received the standards package from the state on January 7, 2022. NMFS responded with draft Project Design Criteria to consider should EPA agree to a programmatic approach.

On a March 3, 2022 conference call, NMFS and EPA discussed concerns with cadmium criteria and urban runoff. The effort was refocused from informal consultation on a subset of chemicals toward formal consultation to expedite determinations on metal criteria.

On May 12, 2022, NMFS provided EPA with draft strawman RPMs and information from NMFS' copper biotic ligand modeling effort; e-mail traffic through May 17, 2022 discussed this analysis.

On June 1, 2022, NMFS transmitted an initiation letter to EPA. The original intent was to write a letter of concurrence on those chemicals for which NMFS' concurred (i.e., those chemicals where the EPA's proposed approval of a water quality standard was not likely to adversely affect any listed species or designated critical habitat), then initiate formal consultation for the remaining chemicals.

On June 14, 2022, NMFS notified EPA that Massachusetts' implementation plan for aluminum did not include the Deerfield River as a waterway where ESA-listed sturgeon could occur, but the Greater Atlantic Region Species Presence Tables indicated the shortnose sturgeon larvae spawned in Connecticut River may be present during certain flow conditions.

On June 24, 2022, EPA transmitted comments on the draft RPMs and, on June 30, 2022, NMFS responded with a revised version and included language for the incidental take statement in the body of the e-mail.

On July 5, 2022, EPA's attorney requested a meeting with NMFS' attorney before scheduling a call to discuss RPMs further.

Between July 20 and August 10, 2022, EPA and NMFS collaborated on and ultimately finalized RPMs via e-mail.

2 The Assessment Framework

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

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

"Destruction or adverse modification" means a direct or indirect alteration that appreciably diminishes the value of critical habitat for the conservation of a listed species. Such alterations may include, but are not limited to, those that alter the physical or biological features essential to the conservation of a species or that preclude or significantly delay development of such features (50 CFR §402.02).

This ESA section 7 consultation involves the following steps:

Description of the Action (Section 3): We describe the numeric water quality criteria EPA proposes to approve and their expected implementation.

Action Area (Section 4): We describe the action and those aspects (or stressors) of the action that may alter the physical, chemical, and biotic environment. We describe the action area with the spatial extent of the stressors from those actions.

Status of Species and Designated Critical Habitat (Section 5): We identify the ESA-listed species and designated critical habitat that are likely to co-occur with the stressors from the action in space and time and evaluate the status of those species and habitat. Specifically, we identify those Species and Designated Critical Habitat Not Likely to be Adversely Affected and provide our effects analysis for these species and critical habitats (Section 5.1), and then identify the status of the remaining Species and Designated Critical Habitat Likely to be Adversely Affected (Section 5.2).

Environmental Baseline (Section 6): We describe the environmental baseline as the condition of the listed species or its designated critical habitat in the action area, without the consequences to the listed species or designated critical habitat caused by the proposed action. The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process.

Stressors Associated with the Action (Section 7): We discuss the potential stressors we expect to result from the action. In this Opinion the stressors of the action are the substances for which numeric water quality criteria are proposed.

Effects of the Action (Section 8): refers to the direct and indirect effects of an action on the species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action that will be added to the environmental baseline. The environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process. 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). In this consultation, if EPA approves state adoption of a National Recommended Water Quality Guideline as a numeric water quality criterion, an interdependent effect of that approval is the state's implementation of that criterion.

Because implementation of the criteria is an effect of EPA's approval, NMFS' exposure assessment evaluates monitoring and regulatory data to identify the pollutant's sources, determine whether the criteria are likely to be implemented, and whether implementation is expected to be successful (e.g., monitoring occurs and uses sufficiently sensitive analytical methods as defined in the 122.44(i)(1)(iv) of the Clean Water Act, existing sources submit discharge monitoring reports, etc.). Examples of criteria that are not likely to be implemented include those for non-persistent pesticides with no registered uses in the state adopting the criteria and substances that are not expected to occur in the water column because they are no longer in domestic or industrial use. Because the criteria set the exposure conditions for each stressor, each analysis determines whether adverse effects may result from exposure to the stressor within the limits of its acute and chronic criteria. If NMFS' analysis determines that exposures and/or responses to a stressor within a criterion's limits are insignificant or extremely unlikely to occur for ESA-listed species under NMFS' jurisdiction, NMFS may make a not likely to adversely affect determination for EPA's approval of the state's adoption and implementation of that criterion. If exposure is reasonably certain to occur and adverse effects are expected in individuals of ESA-listed species under NMFS' jurisdiction exposed within criteria limits, NMFS proceeds with a risk analysis to estimate the implications for the population of affected individuals.

Because this action involves criteria for eight stressors, comprised of seven toxicants and one aquatic chemistry parameter, the analysis is structured as a series of independent effects analyses in order to maintain focus on one determination at a time. The structure of the effects of the action subsections for a given stressor "X" in this Opinion is as follows:

Section 8.x Stressor X: Introduces stressor "X" (i.e., acrolein, ammonia, aluminum, cadmium, copper, carbaryl, nonylphenol, or Total Nitrogen [TN]), summarizing uses, sources, environmental fate, mechanism(s) of effect, the BE analysis, and the criteria.

Section 8.x.1 Exposure to Stressor X within the Action Area: Identifies sources within the action area and evaluates monitoring and permitting data for stressor X to characterize current and future implementation of the criteria. This section also identifies the life stages of ESA-listed individuals that are likely to be exposed to stressor X.

Section 8.x.2 Responses to Stressor X within Criteria Limits: Analyzes the available evidence, using data from surrogate species when necessary and appropriate, to determine how individuals of ESA-listed species are likely to respond to exposures to X within criterion limits. This section also evaluates responses of forage species exposed within criteria limits.

Section 8.x.3 Risk Analysis or Basis for a "Not Likely to Adversely Affect" Determination: The risk analysis for a "likely to adversely affect determination" lays out the evidence supporting the determination then evaluates the consequences of effects in individuals to the populations those individuals represent, and the species those populations comprise. Where effects to critical habitat are expected, the risk analysis also considers the impacts of the proposed action on the physical or biological features and conservation value of designated critical habitat. If adverse effects are not expected, this section would lay out the evidence supporting the "not likely to adversely affect" determination.

Risk hypotheses are statements that organize an analysis by describing the relationships among stressor, exposure, and the environmental values to be protected. Generally speaking, the values to be protected are the survival and fitness of individuals and the value of designated critical habitat for conservation of an ESA-listed species. The applicable risk hypotheses for direct stressors like toxic substances are straightforward, EPA's approval will be likely to adversely affect an ESA-listed species if exposures to the toxic pollutant within criteria limits will result in:

- Reduced survival of individuals through direct mortality or effects favoring predation (e.g., immobility, reduced predator detection)
- Reduced growth of individuals through direct effects of toxicity or effects impairing foraging (e.g., swimming, prey detection, strike success)

- Reduced fecundity through direct effects of toxicity (e.g., reduced hatch, egg mass, egg counts) or effects impairing reproduction (e.g., impaired nest tending, gonads mass)
- Reduced survival, growth, and/or fecundity due to reduced quantity or quality of forage due to toxic effects on forage species abundance or toxic effects of body burdens of the stressor in forage species

The applicable risk hypotheses for water chemistry parameters such as TN are related to the direct stressors resulting from eutrophication. These include ammonia, algal toxins, disrupted dissolved oxygen regime, pathogens, light penetration, and algal smothering/altered substrate (Camargo and Alonso 2006, Schlenger et al. 2013).

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

Integration and Synthesis (Section 10): In this section, we integrate the analyses of Effects of the Action (Section 8), the Environmental Baseline (Section 6), and the Cumulative Effects (Section 9) and place this in context of the Status of Species and Designated Critical Habitat (Section 5) to formulate the agency's biological opinion as to whether the action is likely to appreciably reduce the likelihood of survival and recovery of an ESA-listed species in the wild or reduce the conservation value of designated critical habitat for the conservation of a listed species.

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

- Reduce appreciably the likelihood of both the survival and recovery of ESA-listed species in the wild by reducing its numbers, reproduction, or distribution, and state our conclusion as to whether the action is likely to jeopardize the continued existence of such species; or

- Appreciably diminish the value of designated critical habitat for the conservation of an ESA-listed species, and state our conclusion as to whether the action is likely to destroy or adversely modify designated critical habitat.

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

In addition, we include an Incidental Take Statement (Section 12) that specifies the impact of the take, reasonable and prudent measures to minimize the impact of the take, and terms and conditions to implement the reasonable and prudent measures. ESA section 7(b)(4); 50 CFR §402.14 (i). We also provide discretionary Conservation Recommendations (Section 13) that may be implemented by the action agency; 50 CFR §402.14(j). Finally, we identify the circumstances in which Reinitiation of Consultation is required (Section 15); 50 CFR §402.16.

Note: Discovery of toxicity data, either found or newly generated, indicating ESA-listed species may respond to exposures within criterion limits or information that indicates a previously unexpected stressor assessed in this consultation is present or will be discharged where ESA-listed species occur could be considered "new information" and could trigger reinitiation of consultation (Section 14) for EPA's approval of that criterion.

2.1 Numeric Criteria for the Protection of Aquatic Life

It is important to understand the limitations inherent in EPA's criteria derivation process. The criteria were derived with the objective of protecting aquatic life from short and long-term adverse effects. They are derived from laboratory toxicity test data following EPA's Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and their Uses (hereinafter referred to as (hereafter "Guidelines" Stephen et al. 1985). Because the EPA Guidelines are fundamental to the criteria, the assumptions and procedures implemented in the EPA Guidelines are fundamental to the evaluation of the protectiveness of criteria for ESA-listed species and critical habitats.

Relying on laboratory tests for our understanding of toxicity requires us to assume that laboratory conditions are representative of environmentally relevant conditions and that "domesticated" cultures of test animals will produce similar effects, as would exposure to the same substance on the same, or closely related, wild species. The assumption that effects in laboratory tests are reasonable predictors of effects to individuals in the wild is dependent upon the specific factor being considered. While it is generally reasonable to interpret effects from laboratory tests as being applicable to field situations where a water quality criterion is applied to a particular waterbody, there is risk that laboratory tests underpredict effects in wild animals under natural conditions. In nature, the abundance and quality of food and aquatic chemistry (e.g., pH, dissolved oxygen, temperature, organic matter, ion composition) are variable, individuals are subject to predation, competition, parasitism and disease, and vulnerabilities differ among life stages and during life history events (e.g., migration, spawning). Considering this, arriving at a firm conclusion based on extrapolations from the lab to the field is challenging. It may be that the best overall conclusion is the same as that reached by Chapman (1983) that "when appropriate test parameters are chosen, the response of laboratory organisms is a reasonable index of the response of naturally occurring organisms." His conclusion in turn contributed to one of the most fundamental assumptions of EPA Guidelines, that is, "these National Guidelines have been developed on the theory that effects which occur on a species in appropriate laboratory tests will generally occur on the same species in comparable field situations."

Most criteria are developed consistent with Guidelines using endpoints identified through toxicity tests exposing laboratory-reared organisms to toxicants over a range of concentrations. The endpoints that may be used in criteria derivation include the following:

- concentration at which half of the exposed organisms die (lethal concentration for 50% of organisms, LC50);
- lowest exposure at which a given effect did not differ from controls (no observed effects concentration, NOEC);
- lowest exposure at which the effect differed significantly from controls (lowest observed effects concentration, LOEC);
- effect concentration (EC) at which a certain proportion of an effect was observed (EC##, such as EC10 = concentration at which 10% of test organisms show an adverse response); and
- maximum acceptable toxic concentration (MATC), which is typically the geometric mean of the LOEC and NOEC, but other calculations have been used.

Endpoints are sometimes reported with "<" and ">" to indicate studies in which only one exposure concentration was used or responses for that effect either occurred below the lowest exposure concentration, the less than sign "<", or above the highest exposure concentration used in that study, the greater than sign ">".

For stressors that cause toxic effects, the concentration, duration, and frequency of exposure determines whether effects occur and, if so, the severity of the effects. For this reason, the EPA Guidelines for each criterion are expressed as exposure concentrations over a specified duration and frequency at and below which ecologically relevant effects are not expected to occur. The criterion maximum concentration (CMC) is the highest acceptable aquatic exposure concentration of a chemical in water that is not expected to cause severe effects in aquatic organisms during short-term (i.e., acute) exposure. A CMC is defined as one-half the concentration that is hazardous to five precent of species (HC5) using LC50s from acute toxicity tests lasting four days or less. The CMC is intended to protect aquatic life from acute adverse effects on survival. It is not intended to protect aquatic life from the sublethal effects such as growth and reproduction resulting from chronic exposures. The criterion continuous concentration (CCC) is the highest acceptable aquatic exposure concentration of a chemical in water that is not expected to cause adverse effects on survival, growth, and reproduction over indefinite (i.e., chronic) exposures. A CCC is based on longer exposure periods, sublethal effects, or NOECs or LOECs from LC50 tests. The CMC is typically expressed as a one-hour average, and the CCC is expressed as a four-day average not to be exceeded more than once in three years, on average.

The one-hour and four-day duration and averaging periods for the chronic and acute criteria, respectively, were based upon judgments by the EPA Guidelines' authors that included considerations of the relative toxicity of chemicals in fluctuating or constant exposures. The EPA Guidelines considered an averaging period of one hour most appropriate to use with the CMC because high concentrations of some materials could cause death in one to three hours. The few known studies that tested for latent toxicity following short-term exposures have demonstrated delayed mortality following exposures on the order of three to six hours (Marr et al. 1995, Zhao and Newman 2004, Diamond et al. 2006, Zhao and Newman 2006, Meyer et al. 2007). Observations or predictions of appreciable mortality resulting from metals exposures on the order of only three to six hours supports the Guideline recommendation that the appropriate averaging periods for the CMC is on the order of one hour.

The Guideline specifies a four-day averaging period for chronic criteria was selected for use by EPA with the CCC for two reasons. First, "chronic" responses with some substances and species may not really be due to long-term stress or accumulation, but rather the test was simply long enough that a briefly occurring sensitive stage of development was included in the exposure (e.g., Chapman 1978a, Barata and Baird 2000, De Schamphelaere and Janssen 2004, Grosell et al. 2006, Mebane et al. 2008). Second, a much longer averaging period, such as one month would allow for substantial fluctuations above the CCC.

The Guideline's once-per-three-years allowable exceedance policy was based on a review of case studies of recovery times of aquatic populations and communities from locally severe disturbances such as spills, fish eradication attempts, or habitat disturbances (Yount and Niemi 1990, Detenbeck et al. 1992). In most cases, once the cause of the disturbance was lifted, recovery of populations and communities occurred on a timeframe of less than three years. The EPA has further evaluated the issue of allowable frequency of exceedances through extensive mathematical simulations of chemical exposures and population recovery. Unlike the case studies, these simulations addressed mostly less severe disturbances that were considered more likely to occur without violating criteria (Delos 2008). Unless the magnitude of disturbance was extreme or persistent, this three-year period seemed reasonably supported or at least was not contradicted by the information NMFS reviewed (NMFS 2012b, 2014).

The EPA applies restrictions to the types of data that may be used in deriving criteria. The data must meet very specific and stringent requirements; thus, laboratory conditions are tightly controlled, which is quite different from the natural waters criteria are expected to protect. This level of control is necessary to attribute the response to the exposure. Data requirements also limit the types of responses and how those responses are reported. This ensures consistency among data to allow aggregation of information on the responses of multiple species from multiple studies (i.e., meta-analysis) to derive criteria. Data that are not acceptable for criteria derivation include tests that: lack a control, have too few exposure concentrations, have unacceptable mortality or disease in controls, report atypical responses (e.g., behavior) or measures of response (e.g., time to death), have exposures of the wrong duration, or used species that do not have reproducing wild populations in North American waters.

The EPA Guidelines are designed to arrive at criteria that, when applied as discharge limits, monitoring thresholds, and restoration goals, will achieve water quality that provides for the protection and propagation of fish, shellfish, and wildlife and provide for recreation in and on the water. As stated in Section 1.1 of the document:

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

2.1.1 Evaluating Criteria for the Protection of ESA-listed Species

Because the criteria developed using the EPA Guidelines are not expected to protect all species under all circumstances, waters compliant with the criteria may result in pollutant exposures that cause adverse effects in threatened and endangered species. When assessing risk to an ESA-listed species, the vulnerability of an imperiled population of that species to the loss of an individual, or key individuals, amplifies the fundamental threat posed by a toxic pollutant. The underlying assumptions in the methods used to arrive at criteria affect how well ESA-listed species and designated critical habitat are protected. These assumptions include:

- Effects that occur on a species exposed to a toxicant in laboratory tests will generally be the same for the same species exposed to that toxicant under field conditions (i.e., effects are not influenced by predation, competition, disease, exposure to other stressors in the field, and fluctuations in natural water quality parameters).
- Collections of single-species laboratory toxicity test data used to derive criteria reflect communities in natural ecosystems.
- Data on severely toxic effects from short-term "acute" toxicity tests used to derive CMC can be extrapolated to less severe effects that would be expected to occur in long-term "chronic" exposures to derive CCC.
- Loss of a small number of species will not affect the propagation of fish, shellfish, and wildlife.
- Loss of a small number of species will not result in incidental loss of any "economically or recreationally valuable species" for which data were not available.
- Sensitive species and life stages are adequately represented such that criteria are not biased.
- Derivation of criterion for a single chemical in isolation without regard to the potential for additive toxicity or other chemical or biological interactions is acceptable despite chemicals typically occurring in mixtures in the environment.

- When applied to National Pollution Discharge Elimination System (NPDES) permits, unless the waters are already identified as impaired by a particular pollutant, the waters are free of that pollutant (i.e., the baseline concentration of that pollutant in the receiving water is zero).
- Accumulation of chemicals in tissues and along the food web does not result in ecologically significant latent toxicity or toxic exposures for predators.

There are also concerns about the underlying data used in the derivation of criteria including:

- Data sets for sublethal responses are usually small and have gaps such that sensitive species and life stages are under-represented.
- Variability within and among species used in calculating a hazardous concentration to five precent of species (i.e., HC5) may be substantial, but this variability is not reflected in the final HC5 estimate used to derive a CMC.

For an ESA section 7 consultation, NMFS is required to use "the best scientific and commercial data available" (ESA section 7 (a)(2); 50 CFR §402.14(d)). It is important to note that EPA's use of data for criteria derivation and associated regulatory actions is not the same as NMFS' use of data for this consultation. For example, the requirement that EPA only use data for species that are native to waters of the United States means data on effects to sturgeon of the same genus as ESA-listed sturgeon that occur only in foreign waters would be excluded. This consultation is vetting the criteria. It is not necessary to create reference values or extrapolation factors, as these would require restricting data to allow meta-analysis. NMFS considers all data meeting the screening criteria discussed in the following section. This is consistent with the EPA Guidelines, as it discussed use of "Other Data" as follows:

> *"Pertinent information that could not be used in earlier sections might be available concerning adverse effects on aquatic organisms and their uses. The most important of these are data on cumulative and delayed toxicity, flavor impairment, reduction in survival, growth, or reproduction, or any other adverse effect that has been shown to be biologically important. Especially important are data for species for which no other data are available. Data from behavioral, biochemical, physiological, microcosm, and field studies might also be available. Data might be available from tests conducted in unusual dilution water (see IV.D and VI.D), from chronic tests in which the concentrations were not measured (see VI.B), from tests with previously exposed organisms (see II.F), and from tests on formulated mixtures or emulsifiable concentrates (see II.D). Such data might affect a criterion if the data were obtained with an important species, the test concentrations were measured, and the endpoint was biologically important."*

2.1.2 Screening Data for Use in this Opinion

The screened datasets for NMFS' analysis include data that were not used in criteria derivation (e.g., LC10, IC50) because our purpose is to determine whether there is any indication that ESA-listed species under NMFS' jurisdiction are likely to be affected by exposures within criteria limits. In light of that purpose, risk quotients for all available organism-level endpoint effect data are considered. This includes important but less commonly studied effects, such as altered behavior (e.g., prey strikes) or responses that affect behavior (e.g., acetylcholinesterase inhibition). Data for species that do not have reproducing populations in the United States were also included among data considered in this evaluation. When multiple effects were reported for a single endpoint, the effect was reassigned to a single type of response, favoring reproduction over growth, and growth over survival (i.e., effective mortality, morality) when those options are among the effects reported.

In addition to extracting data from EPA's Ecotoxicology Knowledgebase (ECOTOX), the analysis examined original sources for that data to verify critical datum and identify any important details not included in ECOTOX. Information from recently published literature in the Web of Science and Google Scholar was also collected. Queries of EPA's ECOTOX excluded records identified as having unacceptable controls. Data reported as formulations (e.g., acrolein plus another active ingredient) were excluded to ensure the response was the result of exposure to the active ingredient. Data were excluded if test organisms were pre-exposed (i.e., acclimation studies) or if test organisms were collected from polluted waters. Endpoints with effect magnitudes greater than 50% (e.g., EC75, LC90) were excluded because there is no way to place these in context of a criterion's protectiveness (see section 2.1.2.1 below). Only records reporting mean exposure concentrations or concentration ranges where there was less than two-fold difference between the minimum and maximum reported exposures were retained because a definitive effect threshold (i.e., the exposure concentration at which a response is altered) is needed for assessing the protectiveness of a criterion. When an effect threshold was reported as a range, NMFS' analysis used the minimum reported concentration.

Studies reporting nominal rather than measured exposure concentrations were retained when this aspect did not influence the overall consistency among records. For example, data for static and renewal toxicity tests of highly volatile toxicants, such as acrolein, were carefully considered, if not excluded, when concentrations were not analytically confirmed (i.e., reported as nominal exposures). Actual exposure concentrations of highly volatile substances likely change over the exposure period, and therefore are likely lower than the reported nominal concentrations by the end of the test.

Criteria for ammonia, aluminum, cadmium, and copper are calculated using data for aquatic chemistry parameters that influence their biological availability. Consequently, data lacking the minimum necessary chemistry measurements could not be included in the evaluation unless a surrogate value was available. Where necessary, reported metal concentrations were corrected to dissolved form using EPA's recommended conversion factors. We also excluded data for metals toxicity where only the free ion (i.e., labile) concentration was reported because the metals criteria are based on the dissolved fraction of the metal (i.e., the sample fraction that will pass through a 0.45-micron filter) and there is no standard approach to converting labile metal to dissolved metal.

2.1.2.1 Considering Flow-Through, Renewal, or Static Exposure Test Designs

Test organisms are typically exposed to test solutions through one of three methods. In “static” tests, organisms are in the same test solution for the duration of the test. In “renewal” tests, fresh test solution is replaced once every 24 or 48 hours. In “flow-through” tests, steady state exposure is achieved by continuously providing fresh test solution throughout the test (ASTM 1997). A flow-through test does not create a current; it just means that test solution is introduced as a one-through, nearly continuous delivery of test solution. Historically, flow-through toxicity tests were thought to provide a better estimate of toxicity than static or renewal toxicity tests because they provide a greater control of toxicant concentrations, minimize changes in water quality, and reduce accumulation of waste products in test exposure waters (Rand et al. 1995).

While EPA Guidelines instruct that when there are data for flow-through tests, any static or renewal tests data for that species are to be discounted (Stephan et al. 1985), an important consideration is that natural flowing waters should not be assumed to be in chemical equilibria. Tributary inputs, hyporheic exchanges, stormwater and snowmelt, and daily and seasonal fluxes in pH, carbon, light penetration, and temperature cycles will influence the bioavailability of aquatic pollutants (Stumm and Morgan 1996) and the physiology of aquatic organisms (Heath 1995, McCormick and Leino 1999).

Static exposure studies can yield LC50 values substantially higher than values obtained with flow-through tests or tests in which actual concentrations of contaminants in the system during the experiment are measured. For example, for DDT, LC50 values for static tests have been determined to be approximately 20 times higher than LC50s from flow-through tests (Earnest and Benville 1972). Mercury toxicity testing of trout embryos has indicated that effects concentration-based endpoints (e.g., EC_x, or the effects concentration that cause a specified percent reduction in a particular response) could be as much as one to two orders of magnitude[3] lower in flow-through than static tests (Birge et al. 1979, Birge et al. 1981). Static tests also resulted in higher endpoint estimates for endosulfan when compared with data from flow-through tests (Naqvi and Vaishnavi 1993). Several additional studies with a variety of compounds report static exposures under estimating toxicity (i.e., providing higher endpoint estimates. (e.g., Burke and Ferguson 1969, Vernberg et al. 1977, Hedtke and Puglisi 1982, Randall et al. 1983, Erickson et al. 1998). There are a number of reasons static conditions can underestimate the true exposure concentration in a test. Fish will deplete the concentration in solution over time, causing a lack of steady-state exposure. Some toxicants may transform during the test or volatilize from the test chamber. Other toxicants can adsorb to the walls of the exposure chamber or to accumulating organic matter within the exposure chamber.

[3] An order of magnitude expresses data in terms of factors of 10. For example, 78 is an order of magnitude larger than 7.8. It is calculated as Log_{10}([endpoint]/[criterion]) rounded to 0 decimal places.

With metals, renewal tests can also produce higher EC50 concentrations than flow-through tests (i.e., metals were less toxic). This has been attributed to the adsorption to accumulated organic matter (Erickson et al. 1996, Erickson et al. 1998, Welsh et al. 2008). However, in contrast to earlier EPA and American Society for Testing and Materials (ASTM) recommendations favoring flow-through testing, Santore et al. (2001) suggested that flow-through tests were biased low because typical flow-through exposure systems allowed insufficient hydraulic residence time for complete copper-organic carbon complexation to occur. Copper complexation with organic carbon reduces acute toxicity, but is not instantaneous. Davies and Brinkman (1994) similarly found that cadmium and carbonate complexation was incomplete in typical flow-through designs, although they reported the opposite effect of copper studies, with cadmium in aged, equilibrated waters being more toxic.

When comparing data across different tests, it appears that other factors, such as testing the most sensitive sized organisms or number of organisms per liter of test water, may be much more important than flow-through or renewal techniques. For instance, a Pickering and Gast (1972) study with fathead minnows and cadmium produced flow-through LC50 concentrations that were lower than comparable static LC50 values (~ 4,500 to 11,000 micrograms per liter [µg/L] for flow-through tests vs. ~30,000 µg/L for static tests). The fish used in the static tests were described as "immature," weighing about two grams. The size of the fish used in their flow-through acute tests were not given, but is assumed to have been similar. By contrast, using modern protocols and newly hatched fry weighing about 1/1000th of the fish used by Pickering and Gast (1972), cadmium LC50 concentrations for fathead minnows tend to be around 50 µg/L, with no obvious bias for test exposure (USEPA 2002). Studies examining exposure of brook trout to cadmium report dramatically different results using flow-through and static exposures on different life stages. NMFS identified two brook trout studies, one using flow-through and one using static acute tests, both conducted in waters of similar hardness (41 to 47 milligrams per liter). The LC50 of the static test which used fry was <1.5 µg/L whereas the LC50 of the flow-through test using yearlings was >5,000 µg/L (Carroll et al. 1979, Holcombe et al. 1983).

When all other factors are equal, it appears that renewal tests may indicate chemicals are somewhat less toxic (e.g., higher LC50 values), but there is no clear consensus whether this indicates that renewal tests are biased toward lower toxicity than is "accurate" or whether conventional flow-through tests are biased toward higher toxicity. Comparisons with data across studies suggest that other factors, in particular the life stage of exposures (e.g., Pickering and Gast 1972, Carroll et al. 1979, Holcombe et al. 1983), can dwarf the influence of flow-through or renewal methods for the acute toxicity of at least metals. For this reason, data were not excluded on the basis of test design.

2.1.1 Extrapolating Data from Other Species to Shortnose and Atlantic Sturgeon

Ideally, quantitative exposure-response data for shortnose and Atlantic sturgeon or taxonomically-related surrogates would be available for exposures at the applicable criterion concentrations. Toxicity tests are rarely conducted on threatened and endangered species or species that are not easily cultured in the lab. Those data that are available for shortnose and Atlantic sturgeon demonstrate that taxonomic relatedness is not always a good predictor for toxicity and that rainbow trout, which have abundant data in the screened ECOTOX set, are not "excessively sensitive" to toxicants relative to shortnose and Atlantic sturgeon, and can be suitable surrogate when data for sturgeon are absent.

Dwyer et al. (2005) compared the relative toxicity of five chemicals to 18 fish species, including shortnose sturgeon, Atlantic sturgeon, and rainbow trout. Responses for all three species were similar for copper, suggesting rainbow trout are a good surrogate for metal exposures. A copper LC50 of 80 µg/L was reported for both shortnose sturgeon and rainbow trout while the LC50 for Atlantic sturgeon was only slightly lower, at 60 milligrams per liter (mg/L). Information supporting rainbow trout suitability as a surrogate for exposure to organic chemicals is mixed. Sturgeon were sometimes more sensitive. Shortnose sturgeon, Atlantic sturgeon and rainbow trout 4-nonylphenol LC50s were 80, 50, and 190 µg/L respectively. The pentachlorophenol LC50 was less than 40 µg/L for Atlantic sturgeon and the LC50 for shortnose sturgeon was 70 µg/L while the rainbow trout LC50 was more than twice that, at 160 µg/L. Permethrin LC50s for both shortnose and Atlantic sturgeon were less than 1.2 µg/L while the LC50 for rainbow trout was 3.31 µg/L. The shortnose sturgeon LC50 for carbaryl was comparable to that of rainbow trout, at 1810 and 1880 µg/L, respectively while carbaryl LC50 for Atlantic sturgeon was less than 800 µg/L. In this case, taxonomic relatedness did not ensure similar sensitivity. Chambers et al. (2012) reported a four-fold within-genus difference in sensitivity for early-life-stage effects of polychlorinated biphenyl-126 in Atlantic sturgeon in comparison with shortnose sturgeon. The Chambers et al. (2012) study did not evaluate effects in rainbow trout.

To summarize, rainbow trout had similar sensitivity to copper as shortnose and Atlantic sturgeon, was less sensitive than either sturgeon to 4-nonylphenol, pentachlorophenol, and permethrin, was similarly sensitive to carbaryl as shortnose sturgeon, but not Atlantic sturgeon. Finally, shortnose sturgeon were less sensitive to PCB-126 than Atlantic sturgeon. Taken together, in terms of sensitivity to toxicants, these data suggest that rainbow trout are just as suitable a surrogate species for shortnose and Atlantic sturgeon as species within the same genus or family. The similarity in sensitivity to copper of rainbow trout, shortnose sturgeon, and Atlantic sturgeon suggests they are particularly good surrogates for metal toxicity.

Allometric differences (e.g., body size, membrane area, organ size) are factors to be considered when evaluating data. A smaller individual generally succumbs to toxic effects more rapidly than a larger individual does because it takes a longer time for exposures to reach critical concentrations within the tissues of the larger individual. Therefore, higher exposure concentrations would be needed to elicit the same response over a similar exposure period. While adult sturgeon are much larger than adult rainbow trout, a species commonly used in toxicity tests, one year old sturgeon captured in the Connecticut River ranged in length from 9 to 25 inches (Savoy et al. 2017) while a one year old rainbow trout is about seven to nine inches

(Kebus et al. 1992). Rainbow trout hatchlings are reported to be 10 to 18 mm long (Réalis-Doyelle et al. 2016) while shortnose and Atlantic sturgeon hatchlings are 7 to 11 mm long (Smith et al. 1980, COSEWIC 2005). While not identically sized, this similarity suggests greater confidence when using data for rainbow trout as a surrogate species to assess impacts on early-life-stage sturgeon.

In the absence of data for shortnose and Atlantic sturgeon, this Opinion prioritizes data from surrogate species as follows: other sturgeon species and rainbow trout > other salmonids > other fish species. Where the analysis must rely on other fish species, this Opinion applies a comprehensive perspective that considers all fish data in context of differences reported among sturgeon sensitivities to other toxicants, and the need to be protective of ESA-listed sturgeon. This perspective is based on the expectation that mechanisms of effect in tested fish species are generally similar to mechanisms in the ESA-listed fish species based on fundamental physiological functions (e.g., osmoregulation, ion exchange, antioxidant defense, nerve function). This approach uses a high-level review of ECOTOX data, data from government reports, and peer-reviewed literature, to focus on observations suggesting whether adverse effects could occur within criteria limits are reviewed more closely. This review takes into consideration dataset characteristics, such as the diversity of species represented, outliers, life stage effects, allometric influences, how responses were documented by researchers, the number and quality of the available toxicity studies, and the magnitude and types of effects reported.

2.1.2 Interpreting Toxicity Data

Using the available data to assess the implications of exposures within criteria limits will not mirror how data are used for deriving criteria. Deriving criteria is a very different goal from evaluating criteria for protectiveness of imperiled species. Interpreting toxicity test data is made challenging by the tremendous amount of diversity in the available data. The most abundant toxicity data are LC50s, followed by NOECs and LOECs and EC50s. Other fractional endpoint responses (e.g., EC10, LC20) and response endpoints (e.g. inhibition concentration: IC10) are less abundant. Data are typically not available for exposures of ESA-listed species under NMFS' jurisdiction. Saltwater exposures are particularly sparse. Limiting the data to a narrow set of toxicity test types for the sake of consistency would only result in loss of information that is otherwise useful for evaluating the protectiveness of the criteria.

Considering the slope of exposure-response relationships reported for a vast majority of toxicants, we expect that ESA-listed species are extremely unlikely to respond to exposures within criterion limits if the criterion concentration is orders of magnitude lower (i.e., by ten or 100-fold or more) than the lowest reported acute lethal effect (e.g., LC50s or EC50s) or the lowest chronic exposure-response threshold (e.g., LOEC). Interpreting criteria when the minimum exposures resulting in toxic response (i.e., LC50s, LOECs, and MATCs) are not one or more orders of magnitude greater than the criteria is somewhat more complicated. The magnitude of response at the applicable criterion concentration may be at some lower, but still unacceptable, level from the standpoint of effects to ESA-listed species.

This Opinion considered a number of factors when interpreting LOEC and NOEC data. These endpoints are influenced by study design (e.g., distribution and number of concentrations tested). Depending on exposures tested and underlying variability in responses, the LOEC may actually result in a 30% difference in response from controls, as is first demonstrated in Section 8.1.2.2 of this Opinion. Data are not equally available for all types of endpoints or responses and can vary widely due to differences in the life stages of the organisms used and the study design (e.g., exposure duration, flow through versus static exposures). In addition, the same exposure concentration may be reported as the NOEC for one type of response, such as growth, and as the LOEC for another, such as reproduction.

Although LC50 data are abundant, an exposure in which half of exposed organisms die or are otherwise affected (e.g. an EC50 for immobilization) is clearly not an insignificant effect. The same is true for an EC20 for growth or reproductive effects. Comparisons of fractional endpoint responses (e.g., LC10, EC10, EC25 etc.) to a proposed criterion must consider whether, and to what extent, effects could still result from exposures within criterion limits. When the original toxicity test data[4] are not provided, it is not possible to calculate the magnitude of response at the criterion concentration. In such cases, a comparison metric in some form is necessary to place endpoint data in context of the criterion and its implications for ESA-listed species. Comparison metrics used for this purpose are ratio approaches and include safety factors, risk or hazard quotients, and adjustment factors, which are described in more detail below.

Safety factors: Studies comparing the sensitivities of threatened and endangered species relative to species commonly used in laboratory toxicity tests suggest that multiplying the National Recommended Water Quality Guidelines, LC50s, or chronic response thresholds like inhibition concentrations (ICxx) by a generic safety factor of about 0.5 provides an exposure concentration that would protect ESA-listed species. Safety factors for the protection of ESA-listed species proposed in the open literature range from 0.3 to 0.63 (Dwyer et al. 2000, Sappington et al. 2001, Besser et al. 2005, Dwyer et al. 2005). The method for deriving a CMC relies on the assumption that a concentration that is half the LC50 (i.e., a safety factor of 0.5) would be a no effect or LC01 (Stephen et al. 1985). EPA's BE analyses for acrolein, carbaryl, and nonylphenol interpreted one half an LC50 as the "low effect" threshold to which criteria concentrations were compared.

Risk Quotient: A risk or hazard quotient is the ratio of an anticipated exposure concentration to a standard effect threshold concentration, such as the LC50 or EC20. For this consultation, the anticipated exposure concentrations are the acute and chronic criteria concentrations. When

[4] Toxicity test data consist of the response magnitude at each exposure concentration used in the test. Response magnitude at an exposure of interest, in this case the criterion concentration, may be calculated through regression for continuous responses like growth and number of eggs produced, or prohibit analysis for binary responses such as dead or alive and gravid or not gravid.

evaluating acute exposures for nontarget aquatic animals using LC50 data, EPA's Office of Pesticides Programs (OPP) considers a risk quotient greater than 0.5 as warranting concern. That is to say, exposures that are less than half an LC50 are expected to be safe. This is consistent with the assumption that half the LC50 is conceptually equivalent to an LC01 or to a no effect. However, for threatened and endangered aquatic animals OPP's level of concern for acute exposures is an order of magnitude more protective, with risk quotients greater than 0.05 posing a concern (USEPA 2004). For chronic exposures, OPP bases risk on the lowest early life-stage or full life cycle NOEC for freshwater fish and invertebrates and estuarine/saltwater fish and invertebrates, where a risk quotient greater than one is of concern for all aquatic species, regardless of ESA-listing status.

Adjustment factors: Ideally, under the BE's protocol, a Taxonomic Adjustment Factor (TAF) could be calculated using ratios from studies exposing species from the same taxonomic order as the ESA-listed species of interest. Calculation of a TAF ensures adequate representation among taxa by first calculating the geometric mean of ratios within species (SMAV), then the geometric mean of all SMAVs within each genus (GMAV), followed by the geometric mean of all GMAVs within each family, and finally as the geometric mean of all families within the order. If data for test species within the same taxonomic order as the ESA-listed species of interest were not available, a TAF is calculated as the geometric mean of GMAVs for vertebrates, if the species of interest is a vertebrate, or the geometric mean of GMAVs for invertebrates, if the species of interest is an invertebrate. In cases where a TAF within the same order as the species of interest could not be calculated and the vertebrate and invertebrate TAF do not differ significantly (via a two-sample t-test assuming unequal variances, $\alpha = 0.05$), a Mean Adjustment Factor (MAF) would be applied, which is the geometric mean of all GMAVs for both vertebrates and invertebrates.

Many acute LC50s and chronic EC20s are reported without the toxicity test exposure-response data used to calculate them. As a result, the theoretical "low effect" thresholds, such as LC05 or EC05 concentrations, cannot be estimated. For some of its analyses, EPA derived adjustment factors from those exposure-response data that were available and used those to adjust LC50 and EC20 concentrations to LC05 and EC05 concentrations in the BE.

2.1.2.1 Utility of "Bright Line" Approaches

While safety factor, level of concern, and adjustment factor approaches to interpreting ecotoxicology data offer straightforward, "bright line" approaches, using this "bright line" to determine whether a water quality criterion is likely to adversely affect a listed species does not capture the variation around individual LC50 estimates, the overarching depth and quality of available data, or the implications on survival rates for exposures that occur at the criterion concentration in cases where the exposure response relationship is very shallow.

As demonstrated in prior NMFS' Opinions for EPA approval of Oregon and Idaho water quality standards (NMFS 2012a, 2014, 2020b), the validity of the assumption that one half an LC50 is a safe exposure is reliant on the slope of the exposure-response relationship (Figure 1), with shallow exposure-response curves indicating up to 20% mortality at one-half the reported LC50. A more common pattern with metals data analyzed for a previous water quality consultation was that half an LC50 concentration would probably result in about a five precent death rate in salmon (NMFS 2012a). Testing with cutthroat trout and cadmium, lead, and zinc singly and in mixtures, Dillon and Mebane (2002) found that the LC50/2 concentration corresponded with death rates of 0% to 15%.

A study by Spehar and Fiandt (1986) included effect-by-concentration information on the acute toxicity of chemical mixtures. Rainbow trout and *Ceriodaphnia dubia* were exposed for 96 and 48 hours, respectively, to a mixture of six metals, each at their presumptively "safe" acute CMC. In combination, the CMC concentrations killed 100% of rainbow trout and *Ceriodaphnia*, but 50% of the CMC concentrations killed none (Spehar and Fiandt 1986). This gives support to the assumption that dividing a lethal exposure by two would usually kill few if any fish, although it conflicts with arguments that criteria concentrations in mixtures are protective.

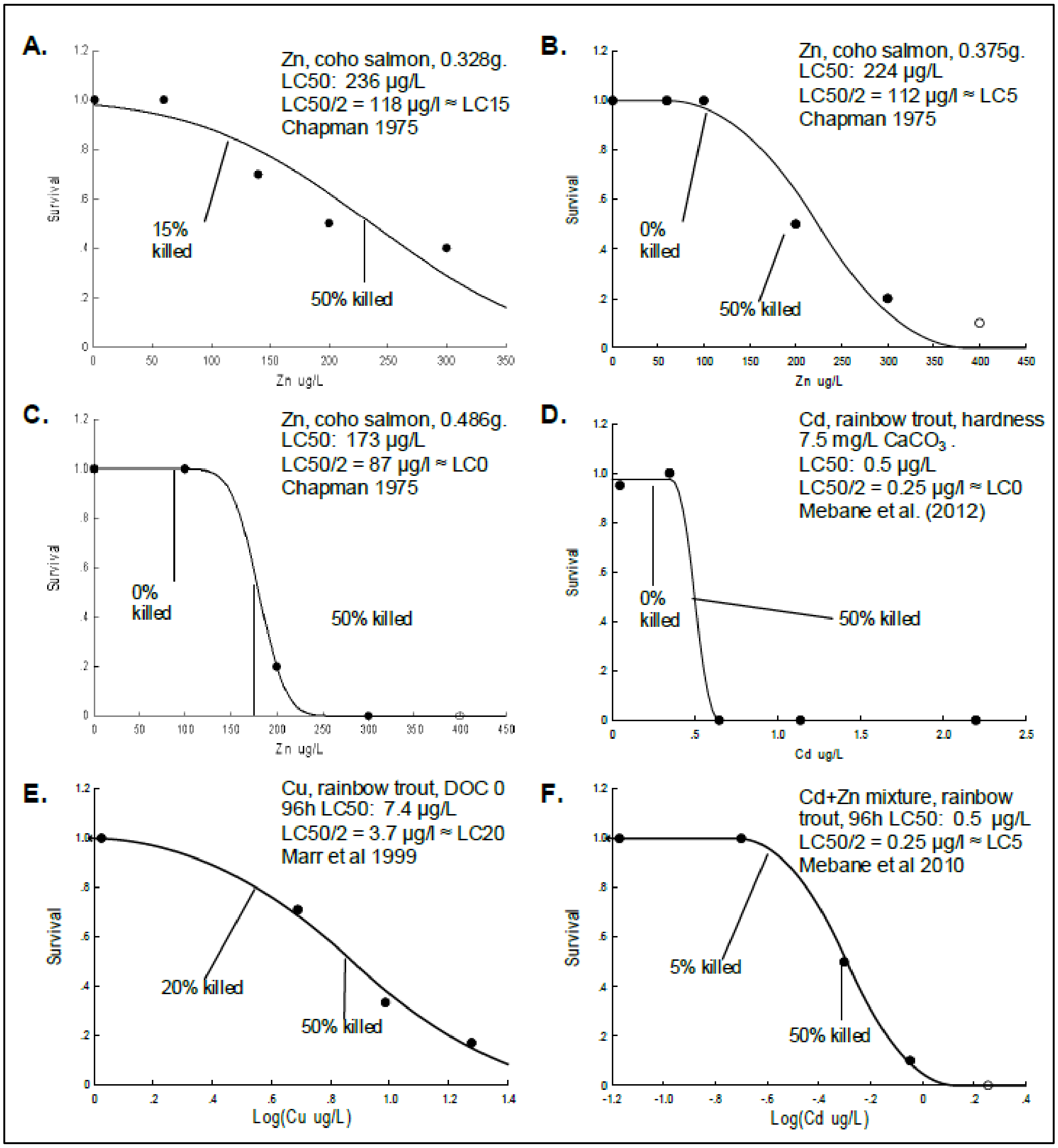

Figure 1. Plots showing proportion of coho salmon or rainbow trout killed at one-half their LC50 concentrations with cadmium, copper, and zinc (NMFS 2014).

The approaches described above are applied to lab data that are based on differences in the physiological sensitivity under optimized, controlled exposures. For this reason, lab tests cannot provide a complete picture of important behavioral effects, exposures under natural conditions and other types of stressors like sediment or dissolved oxygen (DO). Even when test species and ESA-listed species have comparable sensitivities, the loss of an individual from an imperiled population has greater consequences than the loss of an individual from healthy populations.

With respect to the BE's analysis, NMFS appreciates that estimating a "low effect" threshold at five precent is mathematically necessary and, given the variance around any point estimate, the concentration one standard deviation below an LC05 or EC05 could conceptually encompass an

LC00 or EC00, but larger response magnitudes would be possible at the LC05 plus one standard deviation. As such, NMFS does not consider LC05s or EC05s to be bright line decision points that, above and below which, determines “safe” from “not safe.” Rather, these estimates provide context for the potential effects to ESA-listed species.

2.1.2.2 NMFS’ Analysis

For this Opinion, NMFS evaluates monitoring and toxicity data in terms risk quotients because quotients place the data directly in context of the applicable criterion. This Opinion uses the term “applicable criterion” to refer to a criterion calculated to match the aquatic chemistry reported for a monitoring event or toxicity test. When discussing toxicity tests, the term “test-specific criterion” is also used to identify a criterion calculated to match aquatic chemistry conditions of the test. Conversion to “standard conditions” is not how the criteria will be applied in regulatory practice and discussing data in terms of concentrations suggests a level of precision and certainty that is not translatable to real-world exposures. Risk quotients transparently identify those responses that occurred at concentrations one or more orders of magnitude above or below the criterion (i.e., factors of ten), at concentrations that are multiples of the criterion (e.g., twice, four times) or within a “gray area” that demands more careful consideration.

For the analysis of the total ammonia nitrogen, aluminum, and copper CMC and CCC water quality standards, NMFS used the equations published in EPA’s ammonia criteria document (USEPA 2013), EPA’s Aluminum Criteria Calculator[5] and the Biotic Ligand Modeling Software[6] to derive toxicity test-specific criteria. NMFS then used the test-specific criteria to calculate risk quotients: the test-specific criterion, as the presumed exposure concentration, divided by the endpoint effect concentration (e.g., LOEC, NOEC, EC50, LC50 etc.). NMFS also calculated risk quotients using available acrolein, carbaryl, and nonylphenol data. Considering the scale of uncertainty associated with lab-to-field extrapolation discussed in Section 2.1.1, NMFS conservatively applied a CMC, which is implemented as a one-hour average, for toxicity test exposures that were four days or less and applied the CCC, which is implemented as a four-day average, to longer exposures.

The use of risk quotients allows simultaneous presentation of the entirety of the data landscape. As stated previously, our purpose is to determine whether there is any indication that ESA-listed species under NMFS’ jurisdiction are likely to be affected by exposures within criteria limits. In light of that purpose, risk quotients for all available endpoint effect data from the screened datasets are plotted in context of reference values representing the applicable criterion concentration and one-half that criterion concentration. The toxicity data figures in this Opinion present test-specific risk quotients plotted in context of reference lines representing a risk quotient of one (purple) for exposures at the criterion concentration and a risk quotient of 0.5 (orange) representing exposures at one-half the criterion concentration. Risk quotients plotted to

[5] https://www.epa.gov/sites/default/files/2018-12/aluminum-criteria-calculator-v20.xlsm
[6] https://www.windwardenv.com/biotic-ligand-model/

the right of the purple reference line indicate responses occurring at an exposure concentration below the applicable criterion (i.e., higher risk). Risk quotients are plotted on a log scale to enhance resolution.

2.2 Best Scientific and Commercial Data Available for the Consultation

To comply with our obligation to use the best scientific and commercial data available, we collected information identified through searches of Google Scholar, Web of Science, the literature cited sections of peer reviewed articles identified in these searches, reports published by government and private entities, and species listing documentation. The BE provided by EPA includes summaries of toxicity data that EPA used to evaluate whether proposed criteria may result in harm to ESA-listed species and designated critical habitat. Our assessment considers these summaries, but also considers other data found in EPA's ECOTOX database, particularly data that were not available or considered suitable for the derivation of criteria. Use of additional data when vetting the criteria for effects to ESA-listed species is consistent with EPA's Guidelines and the requirement under the ESA that determinations be made based on the best available data. This Opinion is based on our review of this information and various other information sources, including:

- Two Biological Evaluations submitted by EPA:
    - Biological Evaluation for Federally Endangered and Threatened Atlantic and Shortnose Sturgeon; Leatherback, Loggerhead, Green, And Kemp's Ridley Sea Turtles; and North Atlantic Right Whale, Fin Whale, and Sei Whales in Massachusetts And New Hampshire: An Evaluation of the Potential Effects of State-Adopted Acrolein, Carbaryl, and Nonylphenol Aquatic Life Criteria (hereafter, MA-NH BE); and
    - Biological Evaluation for Federally Endangered and Threatened Atlantic and Shortnose Sturgeon; Leatherback, Loggerhead, Green, And Kemp's Ridley Sea Turtles; and North Atlantic Right Whale, Fin Whale, and Sei Whales in Massachusetts: An Evaluation of the Potential Effects Of EPA's Approval Of State-Adopted Freshwater Aluminum, Ammonia, and Copper, Freshwater and Saltwater Cadmium, and Site-Specific Saltwater Nitrogen Aquatic Life Criteria (hereafter, MA BE);
- Water quality monitoring data from the National Water Quality Monitoring Council's Water Quality Portal;
- Government databases, including EPA's Ecotoxicology Knowledgebase (ECOTOX), Enforcement and Compliance History Online Database (ECHO) and the National Water Quality Monitoring Council's Water Quality Portal were frequently consulted interactively during the preparation of this Opinion;
- Government reports, including NMFS biological opinions and stock assessment reports;

- National Oceanic and Atmospheric Administration (NOAA) technical memoranda; and
- Peer-reviewed literature.

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

3 Description of the Action

"Action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by federal agencies (50 CFR §402.02). The action is EPA Region 1's approval of the water quality criteria proposed for adoption by the States of Massachusetts and New Hampshire under Section 303(c) of the Clean Water Act. EPA proposes to approve the adoption of National Recommended Water Quality Guidelines as numeric water quality criteria for implementation of the Clean Water Act in the states of Massachusetts and New Hampshire (Table 1). For both Massachusetts and New Hampshire, EPA proposes to approve freshwater CMC and CCC for acrolein, carbaryl, and nonylphenol and saltwater carbaryl CMC and nonylphenol CMC and CCC. For Massachusetts, EPA proposes to approve aquatic chemistry-specific freshwater CMC and CCC for aluminum, ammonia, cadmium, and copper, saltwater CMC and CCC for cadmium, and site-specific targets for TN. New Hampshire is revising its acrolein criteria and adopting criteria for carbaryl and nonylphenol for the first time. Massachusetts is adopting criteria for acrolein, carbaryl, and nonylphenol for the first time, is revising its aluminum, ammonia, copper, and cadmium criteria, and adopting Total Maximum Daily Load (TMDL) based limits for site specific nitrogen criteria. The purpose of the criteria is to maintain or restore water quality conditions that support aquatic life.

Table 1. National Recommended Water Quality Guidelines, in µg/L, for the protection of aquatic life proposed to be approved by EPA.

	Freshwater		Saltwater		Year
	CMC	**CCC**	**CMC**	**CCC**	**Issued**
Massachusetts and New Hampshire					
Acrolein	3	3	--	--	2009
Carbaryl	2.1	2.1	1.6	--	2012
Nonylphenol	28	6.6	7	1.7	2005
Massachusetts only					
Aluminum	pH, hardness, and Dissolved Organic Carbon (DOC) dependent		--	--	2018
Ammonia	pH, temperature, and life stage dependent		--	--	2013

Cadmium	Hardness dependent		33	7.9	2016
Copper	Biotic Ligand Model		--	--	2007
Total Nitrogen	--	--	Site-specific targets		2008, 2007

The CMC limits for Massachusetts and New Hampshire are one-hour averages not to be exceeded more than once in three years. With the exception of the CCC limits for ammonia and carbaryl, the CCC limits are four-day averages not to be exceeded more than once in three years. The ammonia CCC limit is a 30-day rolling average with the additional restriction that the highest four-day average within the 30 days be no greater than 2.5 times the CCC magnitude that is not to be exceeded more than once in three years. The carbaryl CCC limit is a 30-day rolling average that is not to be exceeded more than once in three years.

Because the ESA requires that we look at all the potential effects of the proposed action on ESA-listed species and designated critical habitat, we consider the act of approving water quality criteria and implementation of the water quality criteria. Once criteria are approved by EPA, EPA may issue NPDES permits for discharges of these pollutants and the state may use criteria to assess and identify aquatic impairments under sections 305(b) and 303(d) of the Clean Water Act, respectively, based on the presence of pollutants above criterion limits. The states of Massachusetts and New Hampshire are unique in that they among the only three states that have not had permitting authority delegated to them. The EPA also remains the permitting authority for New Mexico, Washington D.C., Puerto Rico, and the Pacific Territories (American Samoa, Guam, and the Northern Marianna Islands). Authorization for states, tribes, and territories to issue NPDES permits is achieved through a process defined by Clean Water Act at Section 402 (b) and in 40 CFR Part 123.

4 ACTION AREA

The action area is defined by regulation 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 action area, as described in the BEs and repeated below, is generally consistent with this definition as it is not explicitly limited to state waters in Figure 2, but includes waters that may be affected by water quality conditions within state waters and where the action may affect ESA-listed species.

- Massachusetts’ and New Hampshire’s coastal zone, including coastal bays and waters in Massachusetts and New Hampshire; Narragansett Bay, and the Merrimack, Taunton, North, Westfield, and Connecticut Rivers and their contributing tributaries in Massachusetts, and the Great Bay, its tributaries, Piscataqua River, its tributaries and estuary in New Hampshire for the Atlantic sturgeon, including the critical habitat for the Atlantic sturgeon in both states (see Figure 1-1, of the MA-NH BE);

- Massachusetts' and New Hampshire's coastal zone, including coastal bays and waters, and Narragansett Bay; the Coastal bays and waters in Massachusetts and New Hampshire; Narragansett Bay, and the Merrimack, Deerfield, Westfield, and Connecticut Rivers and their contributing tributaries in Massachusetts, and Great Bay and its tributaries, the Piscataqua River and its tributaries and estuary in New Hampshire for the shortnose sturgeon;
- Massachusetts and New Hampshire and surrounding coastal waters and coastal estuaries and their contributing tributaries, where consequences of the action may be experienced by the green, Kemp's Ridley, leatherback, and loggerhead sea turtles; and
- Massachusetts and New Hampshire coastal waters and any surrounding areas, and their contributing tributaries, where consequences of the action may be experienced by the North Atlantic right whale and fin whale, and Massachusetts coastal waters south of Nantucket for sei whales, including the critical habitat for the North Atlantic right whale in both states (see Figure 1-2, MA-NH BE).

NMFS notes that the definition for the Connecticut River portion of the action area in the BE is limited to waters within Massachusetts. Waters of the Connecticut River in the state of Connecticut are potentially affected by discharges to the river within Massachusetts and are considered part of the action area. All waters where ESA-listed Atlantic and shortnose sturgeon occur are referred to as "Sturgeon Waters" in this Opinion.

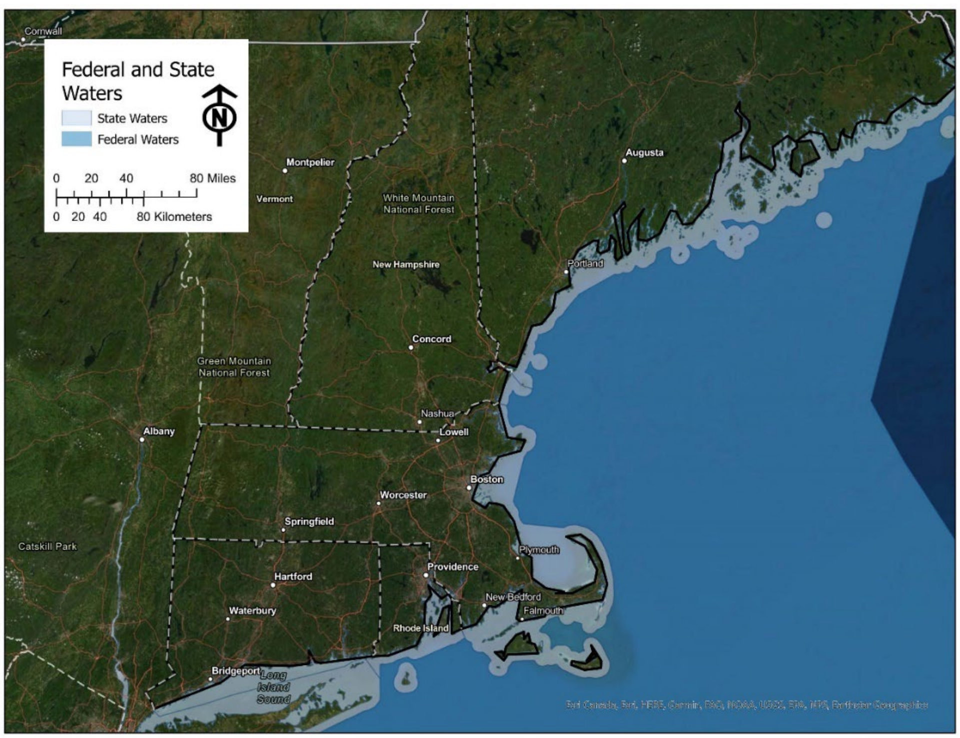

Figure 2. Extent of the saltwater portion of the action area: state waters (light blue) within EPA Region 1.

5 ESA-Listed Species and Designated Critical Habitat

Table 2 identifies the ESA-listed DPSs that occur in the action area and are under NMFS' jurisdiction.

Table 2. Endangered and threatened species and designated critical habitat within the action area under NMFS' jurisdiction.

Species	Federal Register Listing	Designated Critical Habitat
Fin Whale (endangered, *Balaenoptera physalus*)	35 FR 18319	--
North Atlantic Right Whale (endangered, *Eubalaena glacialis*)	73 FR 12024	81 FR 4837
Sei Whale (endangered, *Balaenoptera borealis*)	35 FR 18319	--
Green Turtle (threatened, *Chelonia mydas*), North Atlantic DPS	81 FR 20057	Does not occur in action area
Kemp's Ridley Turtle (endangered, *Lepidochelys kempii*)	35 FR 18319	--
Leatherback Turtle (endangered, *Dermochelys coriacea*)	35 FR 8491	Does not occur in action area
Loggerhead Turtle (threatened, *Caretta caretta*), Northwest Atlantic Ocean DPS	76 FR 58868	Does not occur in action area
Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*) Gulf of Maine DPS (threatened), and New York Bight DPS and migrating Chesapeake Bay, Carolina and South Atlantic DPSs (all endangered)[a]	77 FR 5879 77 FR 5913	82 FR 39160
Shortnose Sturgeon (endangered, *Acipenser brevirostrum*)	32 FR 4001	--

[a]The BEs provided by EPA identified the New York Bight DPS of the Atlantic sturgeon for the Connecticut River, but data indicate that spawning Atlantic sturgeon in those waters are more closely related to the Carolina and South Atlantic DPSs of Atlantic sturgeon (Savoy et al. 2017).

5.1 ESA-Listed Species Not Likely To Be Adversely Affected by EPA's Approval of Water Quality Criteria Proposed for Massachusetts and New Hampshire

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

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

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

Insignificant effects relate to the size or severity of the impact and include those effects that are undetectable, not measurable, or so minor that they cannot be meaningfully evaluated. Insignificant is the appropriate effect conclusion when species or critical habitat will be exposed to stressors, but the response will not be detectable outside of normal behaviors.

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

Prior consultations determined that implementation of EPA's Water Quality Guidelines for aquatic toxicants is not likely to adversely affect ESA-listed sea turtles and baleen whales because their exposures to aquatic pollutants are expected to be far less than that of the fish and aquatic invertebrates the criteria were derived to protect (NMFS 2015, 2018a, 2020a). Fish and aquatic invertebrates are exposed to aquatic toxicants as water continuously passes over their gill filaments where mineral and gas exchange regulates ion balance and oxygenates blood. The folded, feather-like structure of gills maximizes contact between water and respiratory epithelia for this exchange but also maximizes exposure to aquatic toxicants. Saltwater and estuarine fish exposures also occur through ingestion because saltwater fish "osmoregulate" by continuously drinking seawater and excreting solute in order to maintain a lower concentration of solutes in their body fluids than saltwater (Larsen et al. 2014).

5.1.1 Whales

Fin and sei whales are highly migratory species and are associated with deep offshore habitats. Sei whales prefer deep waters off the continental slope (Horwood 1987) but can occur in the Great South Channel and southern Gulf of Maine in spring and early summer. Fin whales are centered along the 100-meter isobath (Figure 3, dark blue) but with sightings well spread out over shallower and deeper water, including sub saltwater canyons along the shelf break (Kenney and Winn 1987, Hain et al. 1992). Two feeding areas in the late 1970s and early 1980s were identified between the Great South Channel and Jeffrey's Ledge (Hain et al. 1992).

> *NMFS determines that EPA's approval of water quality criteria proposed for Massachusetts and New Hampshire may affect, but is not likely to adversely affect fin and sei whales because their response to infrequent and short duration exposures to waters affected by implementation of the criteria are expected to be insignificant, due to their long migrations and affinity for deeper offshore waters (Figure 3).*

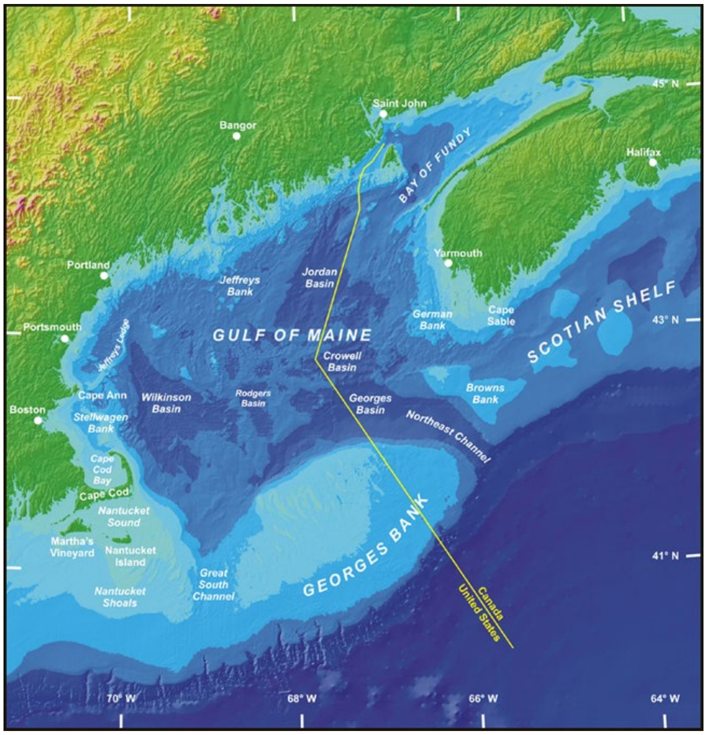

Figure 3. New England coastal waters.[7]

In contrast to fin and sei whales, the North Atlantic right whale will frequent nearshore waters. Cape Cod Bay and the Great South Channel east of Nantucket are important forging areas for this species in the spring after calving season (mid-November and through mid-April) in waters of the southeastern United States. Most individuals migrate northward to Canada during the summer and fall months. Aquatic toxicants are not readily absorbed through mammalian skin, so any exposure of these whales is primarily direct uptake from the water column through membranes that are in contact with ambient water or indirect uptake through ingesting organisms that have accumulated pollutants. The pathway for direct exposure, and subsequent response, of whales to aquatic pollutants is further limited because whales do not drink seawater. Whale osmoregulation employs physiological and allometric adaptations such as increased filtration rates, urine volume, and kidney size along with tolerance of high solute levels in urine and plasma (Kjeld 2003, Birukawa 2005).

NMFS determines that EPA's approval of the water quality criteria for Massachusetts and New Hampshire may affect, but is not likely to adversely affect North Atlantic right whales because their response to infrequent and short duration exposures to waters affected by implementation of the criteria are expected to be insignificant due to their limited exposures to aquatic pollutants because they breathe air, do not drink seawater, and are highly migratory.

[7] https://celebrating200years.noaa.gov/magazine/globec/map_gulfofmaine.html

5.1.2 Sea Turtles

Because ESA-listed North Atlantic DPS of green turtle, Kemp's ridley turtle, leatherback turtle, and the Northwest Atlantic Ocean DPS of loggerhead turtle breathe air and do not have gills, their only direct exposures would be through drinking seawater and limited absorption through exposed membranes. While metals and persistent organic pollutants can accumulate in sea turtles through their diet, turtles are unlikely to accumulate a significant amount of persistent pollutants because they primarily consume lower trophic-level food species (Figgener et al. 2019). The presence of a contaminant in tissues does not necessarily indicate adverse effects on survival, reproduction, or growth. Contaminant burdens in tissues reflect exposures integrated over the lifetime and entire foraging area of these highly migratory species and cannot be directly attributable to exposures within an action area that comprises only a fraction of an individual's range.

Sea turtles are temporary residents to New England waters, undergoing long migrations between breeding and foraging habitats. In general, juvenile and adult green sea turtles migrate north in the spring as water temperatures warm, arriving in mid-Atlantic waters in May. As the waters cool in the fall, the trend reverses, with most sea turtles leaving the area by the end of November to migrate to the southeastern United States, Caribbean, and Gulf of Mexico (NMFS and USFWS 1991, 1992, Shoop and Kenney 1992, NMFS 2007, NMFS and USFWS 2015).

NMFS determines that EPA's approval of the water quality criteria proposed for Massachusetts and New Hampshire may affect, but is not likely to adversely affect the North Atlantic DPS of green turtle, Kemp's ridley turtle, leatherback turtle, and the Northwest Atlantic Ocean DPS of loggerhead turtle because their response to infrequent and short duration exposures to waters affected by implementation of the criteria are expected to be insignificant because they breathe air and are highly migratory.

5.1.3 Critical Habitat Designated for Atlantic Sturgeon

The critical habitat designation for the Gulf of Maine and New York Bight DPSs of Atlantic sturgeon physical and biological features (PBFs) do not include biological features such as prey or vegetative cover that could be affected by exposures to toxicants. However, the designation includes optimal dissolved oxygen conditions. Dissolved oxygen is influenced by eutrophication resulting from excess nutrients. Unless a system is already eutrophic or highly polluted, ammonia nitrogen is a very small fraction of total nitrogen in natural systems, so ammonia concentrations within criteria limits would not be expected to contribute to eutrophic conditions and disruption of a waterbody's dissolved oxygen regime.

NMFS determines that EPA's approval of water quality criteria proposed for Massachusetts and New Hampshire may affect, but is not likely to adversely affect designated critical habitat for the Gulf of Maine or New York Bight DPSs of Atlantic sturgeon because there are no biological features under the designation to respond to *toxicants and the presence of ammonia within criterion limits is not expected to contribute to eutrophic conditions, and is therefore discountable.*

5.1.4 Critical Habitat Designated for North Atlantic Right Whale

The PBFs of critical habitat designated for the North Atlantic Right whale include a PBF that may respond to toxicant exposures: the presence of late stage and diapausing *Calanus finmarchicus* in dense aggregations in the Gulf of Maine and Georges Bank Region. These waters are far from shore and are not state waters: (see Figure 2 and Figure 3).

> *NMFS determines that EPA's approval of water quality criteria proposed for Massachusetts and New Hampshire are not likely to adversely affect designated critical habitat for the North Atlantic right whale because the PBF that may respond to toxicant exposures is located far from waters that would be affected by implementation of the criteria and, due to dilution in the vast ocean waters, exposures would be discountable.*

5.2 Status of Species Likely to be Adversely Affected

Section 5.1 above set forth the rationale for determining that EPA's approval of water quality criteria proposed for Massachusetts and New Hampshire are not likely to adversely affect ESA-listed whales (Section 5.1.1), sea turtles (Section 5.1.2), and designated critical habitat for the Gulf of Maine and New York Bight DPSs of Atlantic sturgeon (Section 5.1.3) and North Atlantic right whale (Section 5.1.4). The ESA-listed species that may be adversely affected by EPA approval of water quality criteria proposed for Massachusetts and New Hampshire are the shortnose sturgeon and the Gulf of Maine and New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon.

This Opinion examines the status of each species and critical habitat that may be adversely affected by the action. The evaluation of adverse effects in this Opinion begins by summarizing the biology and ecology of those species that are likely to be adversely affected and what is known about their life histories in the action area and the condition of designated critical habitat within the applicable critical habitat unit and in the action area. The status is determined by the level of risk that the ESA-listed species and designated critical habitat face based on parameters considered in documents such as recovery plans, status reviews, and listing decisions. This helps to inform the description of the species' current "reproduction, numbers or distribution" that is part of the jeopardy determination as described in 50 C.F.R. §402.02. This section also examines the condition of critical habitat throughout the action area, and discusses the condition and current function of designated critical habitat, including the PBFs that contribute to that conservation value of the critical habitat. More detailed information on the status and trends of these ESA-listed species, and their biology and ecology can be found in the listing regulations and critical habitat designations published in the Federal Register, status reviews, recovery plans, and on the NMFS Web site: [https://www.fisheries.noaa.gov/species-directory/threatened-endangered.

5.2.1 Threats Common to Shortnose and Atlantic Sturgeon

The viability of sturgeon populations is highly sensitive to juvenile mortality resulting in lower numbers of sub-adults recruiting into the adult breeding population. The significant threats to ESA-listed sturgeon include dams that block access to spawning areas or lower parts of rivers, poor water quality, dredging, vessel strikes, water withdrawals from rivers, and unintended catch in some commercial fisheries. Recent reviews also identify climate change as a threat to ESA-listed sturgeon, with the Gulf of Maine as one of the fastest warming areas of the world (SSSRT 2010, NMFS 2022a, b).

5.2.1.1 Dams

Dams impede fish passage, fragmenting populations through eliminating or impeding access to historic habitat. Hydropower turbines, spillways, and fish passage devices can injure or kill fish attempting to migrate or are entrained in turbines. Dams also modify natural hydrology, altering downstream flows and water temperatures, affecting dissolved oxygen, channel morphology, nutrient cycling, stratification, community structure, and sediment regime, which can include redistribution of sediment-associated toxicants(Jager et al. 2001, Secor et al. 2002, Cooke and Leach 2004). Short-term negative impacts of dam removal include the influx of sediments into the stream flow, which can embed spawning substrates and negatively affect water, habitat and food quality. These effects are usually temporary. Several studies have demonstrated that after dam removal, sediments were flushed from river channels, natural sediment transport conditions resumed (American Rivers 2002).

5.2.1.2 Impingement and Entrainment

Depending on life stage and size, sturgeon are susceptible to impingement on or entrainment through cooling water intake screens at power plants. Impingement and entrainment is also a risk during dredging operations. Other effects of dredging include burial of benthic communities, turbidity, siltation of spawning habitats, redistribution of sediment-associated toxicants, noise/disturbance, modified hydrology, and overall loss of habitat (Chytalo 1996, Smith and Clugston 1997, NMFS 1998b, Winger et al. 2000, NMFS 2018b).

5.2.1.3 Bycatch

At this time, Atlantic sturgeon bycatch mortality is now considered a primary threat affecting the recovery of all five DPSs of Atlantic sturgeon (NMFS 2022a, b). The level of bycatch and poaching of shortnose sturgeon is mostly unknown, but modeling suggests that bycatch could have a substantial impact on the status of shortnose sturgeon, especially in populations of small numbers (SSSRT 2010).

5.2.1.4 Contaminants

The 2010 status review for shortnose sturgeon reviewed contaminant risks applicable to all sturgeon species. The life history characteristics of amphidromous sturgeon (i.e., long lifespan, extended residence in estuarine habitats, benthic foraging) predispose these species to long-term and repeated exposure to environmental contamination and potential bioaccumulation of heavy metals and other toxicants (Dadswell 1979, NMFS 1998a). Chemicals and metals such as chlordane, dichlorodiphenyl dichloroethylene (DDE), DDT, dieldrin, PCBs, cadmium, mercury, and selenium settle to the river bottom and are later consumed by benthic feeders, such as macroinvertebrates, and then work their way higher into the food web (e.g., to sturgeon). Some of these compounds may affect physiological processes and impede a fish's ability to withstand stress, while simultaneously increasing the stress of the surrounding environment by reducing DO, altering pH, and altering other physical properties of the water body.

General Effects of Contaminant Exposures and Tissue Burdens in Fish: Pesticide exposure in fishes may affect anti-predator and homing behavior, reproductive function, physiological development, and swimming speed and distance (Beauvais et al. 2000, Scholz et al. 2000, Moore and Waring 2001, Waring and Moore 2004). Sensitivity to environmental contaminants also varies across life stage. Early-life-stages of fishes appear to be more susceptible to environmental and pollutant stress than older life stages (Rosenthal and Alderdice 1976). The presence of a contaminant in the tissues of an organism indicates exposure, but does not always mean these tissues residues are causing adverse effects. Elevated levels of contaminants in fish have been associated with reproductive impairment (Giesy et al. 1986, Mac and Edsall 1991, Cameron et al. 1992, Longwell et al. 1992, Matta et al. 1997, Billsson 1998, Hammerschmidt et al. 2002)), reduced larval survival (Berlin et al. 1981, Giesy et al. 1986), delayed maturity (Jørgensen et al. 2004) and posterior malformations (Billsson 1998).

Tissue Burdens Reported in Sturgeon: Shortnose sturgeon collected from the Delaware and Kennebec Rivers had total toxicity equivalent concentrations of polychlorinated dibenzo-p-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs), PCBs, DDE, aluminum, cadmium, and copper above adverse effect concentration levels reported in the literature (ERC 2002, 2003). Dioxin and furans were detected in ovarian tissue from shortnose sturgeon caught in the Sampit River/Winyah Bay ecosystem, South Carolina. Results showed that 4 out of 7 fish tissues analyzed contained tetrachlorodibenzo-p-dioxin (TCDD) concentrations > 50 ppt, a level which can adversely affect the development of sturgeon fry (NOAA, Damage Assessment Center, Silver Spring, MD, unpublished data).

Dadswell (1975) reported mercury concentrations averaging 0.29 (0.06 – 1.38) milligrams per kilogram (mg/kg) wet weight in 30 juvenile Atlantic sturgeon collected in the Saint John River estuary, New Brunswick. Rehwoldt et al. (1978) analyzed cadmium, mercury, and lead in tissues from some freshly captured Atlantic sturgeon from the Hudson River in 1976 and 1977 and found no chronological relationship when compared to preserved reference samples collected between 1924 and 1953. The 1976-1977 average cadmium, mercury, and lead tissue concentrations were 0.02, 0.09, and 0.16 μg/g wet weight, respectively.

Twenty juvenile Gulf sturgeon, a subspecies of Atlantic sturgeon, exhibited an increase in metal body burdens with an increase in fish length (Alam et al. 2000). Gulf sturgeon collected from a number of rivers between 1985 and 1991 had arsenic, mercury, DDT metabolites, toxaphene, polycyclic aromatic hydrocarbons (PAHs), and aliphatic hydrocarbons at concentrations that were sufficiently high to warrant concern (Bateman and Brim 1994). Kootenai River white sturgeon exhibited organochlorine levels that could potentially affect reproduction or other physiological functions (Kruse and Scarnecchia 2002). Aldrin, 4,4-DDE, α-HCH, copper, and selenium were the most frequently detected contaminants in the plasma of green sturgeon from Washington coastal estuaries, with the highest concentrations in fish collected from more urbanized areas, but few fish had plasma contaminant levels at toxic thresholds (Layshock et al. 2022). The liver and gonads of white sturgeon from the San Francisco Bay Estuary had high concentrations of arsenic, barium, cadmium, copper, chromium, lead, mercury, nickel, selenium, and zinc. The concentrations of arsenic, cadmium, copper, selenium, mercury and selenium were at levels known to impair fish health (Gundersen et al. 2017). Selenium in the ovaries and liver of vitellogenic San Francisco Bay Delta white sturgeon was measured at levels demonstrated to cause reproductive impairment in laboratory studies (Linares-Casenave et al. 2015).

Effects Associated with Exposure in the Wild: Male Columbia River white sturgeon growth and reproductive impacts were observed along with a negative correlation between plasma androgens and gonad size with DDT, pesticides, and PCBs (Feist et al. 2005). Mercury concentrations of white sturgeon captured from the Columbia River was correlated with suppressed circulating sex steroids, decreased condition factor and relative weight, and a lower gonadosomatic index in immature males. A significant positive linear relationship was observed between age and liver mercury concentrations (Webb et al. 2006). Poly- and perfluorinated compounds accumulated in the tissues and eggs of wild 17 to 25 year old female Chinese sturgeon, but accumulations in eggs did not reach estimated concentrations that would impair reproduction (Peng et al. 2010). The condition of wild caught stellate sturgeon was negatively correlated with liver and muscle concentrations of cadmium and lead (Heydari et al. 2011).

5.2.2 Shortnose Sturgeon

Shortnose sturgeon were first listed under the Endangered Species Preservation Act on October 15, 1966 (32 FR 4001). When the ESA was signed into law, replacing the Endangered Species Preservation Act, shortnose sturgeon remained listed as endangered. No critical habitat has ever been designated. Shortnose sturgeon occur along the Atlantic Coast of North America from the Saint John River in Canada to the Saint Johns River in Florida. While shortnose sturgeon spawning has been documented in several rivers across its range, status for many other rivers remain unknown. Within Massachusetts, shortnose sturgeon are reported to occur in Narragansett Bay, the Merrimack River, the Connecticut River and its Deerfield River and

Westfield River tributaries. The species may also be present in other Massachusetts rivers and coastal areas.

In the Merrimack River, shortnose sturgeon occur up to the Essex Dam, at river kilometer (rkm) 46 and spawn near Haverhill at rkm 30 to 32 (Kieffer and Kynard 1996). Larvae begin moving downstream four weeks after the spawning and continue to develop in the freshwater reach of the river (rkm 16 to 32, Kieffer and Kynard 1993). Foraging concentrations are reported near Amesbury and the lower islands (rkm 6 to 24, Kieffer and Kynard 1993, Kynard et al. 2000). Merrimack River sturgeon overwinter from late fall to early spring above the salt wedge at rkm 15 to 29 (Kieffer and Kynard 1993, Wippelhauser et al. 2015).

The 2010 status review indicates that the Connecticut River shortnose sturgeon population is impeded, but not isolated, by the Holyoke dam. Connecticut River shortnose sturgeon occur within the mainstem up to Turners Falls Dam (rkm 198) within the Westfield River and Deerfield River tributaries. Spawning occurs below Turners Falls Dam/Cabot Station at rkm 193 to 194 or, when conditions are favorable, below the Holyoke Dam at rkm 139 to 140 (Kynard et al. 2012a). Offspring drift downstream for up to 20 kilometers (km) such that early-life-stages would be present in downstream freshwater reaches from rkm 13 to 194 (Buckley and Kynard 1981, Kynard et al. 2012b). Foraging and overwintering concentrations are reported from above the Holyoke Dam in Deerfield Concentration Area at rkm 144 to 192 (Kynard et al. 2012b), Agawam at rkm 114 to 119 (Buckley and Kynard 1985b), and the lower Connecticut from rkm 0 to 110 (Kynard et al. 2012b).

Adults may occur in the Deerfield River up to Shelburne Falls at rkm 22.5 and larvae can drift into the Deerfield River under certain flow conditions (Kieffer and Kynard 1993, Kynard et al. 2012b). Foraging may occur from spring through fall in lower Deerfield River from rkm 0 to 3.5 (Kynard et al. 2012b). The Deerfield River also can be used for overwintering potentially for pre to spawning staging for adults (Kynard et al. 2016). Adults are also assumed to forage in the Westfield River up to the Decorative Specialties International Dam at rkm 9.5 (SSSRT 2010).

LIFE HISTORY

The shortnose sturgeon is a relatively slow growing, late maturing, and long-lived fish species. Shortnose sturgeon are amphidromous, inhabiting large coastal rivers or nearshore estuaries within river systems (Buckley and Kynard 1985a, Kieffer and Kynard 1993). Sturgeon spawn in upper freshwater areas, and feed and overwinter in both fresh and saline habitats. Adult shortnose sturgeon typically prefer deep downstream areas with vegetated bottoms and soft substrates. During the summer and winter months, adults occur primarily in freshwater tidally influenced river reaches; therefore, they often occupy only a few short reaches of a river's entire length (Buckley and Kynard 1985a). Older juveniles or sub adults tend to move downstream in the fall and winter as water temperatures decline and the salt wedge recedes. In the spring and summer, they move upstream and feed mostly in freshwater reaches; however, these movements usually occur above the saltwater/freshwater river interface (Dadswell et al. 1984, Hall et al.

1991). Young-of-the-year shortnose sturgeon are believed to move downstream after hatching (Bain 1997) but remain within freshwater habitats.

While shortnose sturgeon do not undertake the long saltwater migrations documented for Atlantic sturgeon, telemetry data indicate that shortnose sturgeon do make localized coastal migrations (Dionne et al. 2013). Inter-basin movements have been documented among rivers within the Gulf of Maine, between the Gulf of Maine and the Merrimack, between the Connecticut and Hudson rivers, between the Delaware River and Chesapeake Bay, and among the rivers in the Southeast region (Welsh et al. 2002, Finney et al. 2006, Fernandes et al. 2010, Dionne et al. 2013). Non-spawning movements include rapid, directed post-spawning movements to downstream feeding areas in the spring, and localized, wandering movements in the summer and winter (Dadswell et al. 1984, Buckley and Kynard 1985a). In the northern extent of their range, shortnose sturgeon exhibit three distinct movement patterns. These migratory movements are associated with spawning, feeding and overwintering activities. In the spring, as water temperatures reach between 7.0 and 9.7 ºC, pre-spawning shortnose sturgeon move from overwintering grounds to spawning areas.

Spawning in northern rivers occurs from mid to late spring depending upon location and water temperature. Shortnose sturgeon spawning migrations are characterized by rapid, directed and often extensive upstream movement (NMFS 1998b). Once males begin spawning, one to two years after reaching sexual maturity, they will spawn every other year or annually depending on the river they inhabit (Dadswell 1979, NMFS 1998b). Age at first spawning for females is around five years post-maturation, with spawning occurring approximately every three to five years (Dadswell 1979). Spawning is estimated to last from a few days to several weeks.

Shortnose sturgeon are believed to spawn at discrete sites within their natal river (Kieffer and Kynard 1996), typically at the farthest upstream reach of the river, if access is not obstructed by dams (NMFS 1998b). In the Merrimack River, males continually returned to only one reach during a four-year telemetry study (Kieffer and Kynard 1996). Spawning occurs over channel habitats containing gravel, rubble, or rock-cobble substrates (Dadswell 1979, NMFS 1998b). Additional environmental conditions associated with spawning activity include decreasing river discharge following the peak spring freshet, water temperatures ranging from 6.5 to 18ºC, and bottom water velocities of 0.4 to 0.8 m/sec (Dadswell 1979, Hall et al. 1991, Kieffer and Kynard 1996, NMFS 1998b). Adult shortnose sturgeon typically leave the spawning grounds shortly after spawning.

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

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

POPULATION DYNAMICS

Historically, shortnose sturgeon are believed to have inhabited nearly all major rivers and estuaries along the entire east coast of North America. NMFS' Shortnose Sturgeon Recovery Plan identifies 19 populations based on the fish's strong fidelity to natal rivers and the premise that populations in adjacent river systems did not interbreed with any regularity (NMFS 1998). Both mtDNA and nDNA analyses indicate effective (with spawning) coastal migrations are occurring between adjacent rivers in some areas, particularly within the Gulf of Maine and the Southeast (King et al. 2014).

The distribution of shortnose sturgeon is disjointed across their range, with northern populations separated from southern populations by a distance of about 400 km near their geographic center in Virginia. Genetic components of sturgeon in rivers separated by more than 400 km appear to be connected by very little migration, while rivers separated by less than 20 km would experience high migration rates. At the northern end of the species' distribution, the highest rate of gene flow (which suggests migration) occurs between the Kennebec, Penobscot, and Androscoggin Rivers (Wirgin et al. 2005).

STATUS

According to the 2010 status review (SSSRT 2010), water quality represents a major threat to one shortnose sturgeon population (Potomac River), a moderately high threat to six populations, a moderate threat to 13 populations, and a moderately low threat to one population. Specific sources of water quality degradation affecting shortnose sturgeon include coal tar, (a potential source of metal exposure, Gao et al. 2016), wastewater treatment plants, fish hatcheries, industrial waste, pulp mills, sewage outflows, industrial farms, water withdrawals, and nonpoint sources. These sources contribute to the following conditions that may have adverse effects on shortnose sturgeon: nutrient loading, low DO, algal blooms, increased sedimentation, elevated contaminant levels (mercury, polychlorinated biphenyl [PCBs], dioxin, polycyclic aromatic hydrocarbons [PAHs], endocrine disrupting chemicals, cadmium), and low pH levels. Impingement/entrainment at power plants and treatment plants was rated as a moderate threat to two shortnose sturgeon populations (Delaware and Potomac).

The shortnose sturgeon status review team (SSSRT 2010) reported results of an age-structured population model using the RAMAS software (Akçakaya and Root 2007) to estimate shortnose sturgeon extinction probabilities for three river systems: Hudson, Cooper, and Altamaha. The estimated probability of extinction was zero for all three populations under the default assumptions, despite the long (100-year) horizon and the relatively high year-to-year variability in fertility and survival rates. The estimated probability of a 50% decline was relatively high (Hudson 0.65, Cooper 0.32, Altamaha 0.73), whereas the probability of an 80% decline was low (Hudson 0.09, Cooper 0.01, Altamaha 0.23 SSSRT 2010).

The largest shortnose sturgeon adult populations are found in the Northeastern rivers: Hudson 56,708 adults (Bain et al. 2007); Delaware 12,047 (ERC 2006); and Saint Johns > 18,000 adults (Dadswell 1979). Shortnose sturgeon populations in southern rivers are considerably smaller by comparison. Peterson and Bednarski (2013) documented a three-fold variation in adult abundance (707 to 2,122 individuals) over a 7-year period in the Altamaha River. Bahr and Peterson (2017) estimated the adult shortnose population in the Savannah River was 1,865 in 2013, 1,564 in 2014, and 940 in 2015. Their estimates of juvenile shortnose sturgeon ranged from 81-270 age 1 fish and 123-486 age 2+ fish over the course of the three-year (2013-2015) study period. This study suggests that the Savannah River population is likely the second largest within the South Atlantic (Bahr and Peterson 2017).

Status within the Action Area

The most recent status review for shortnose sturgeon was written in 2010 (SSSRT 2010). This review developed cumulative shortnose sturgeon population health scores, ranked stressors occurring to shortnose sturgeon within each river, and compared population health to stressors. Population health scores were based on number of individuals (one to five), demographics (three points per life stage present) and abundance trends (zero for unknown or no estimate to three for increasing trend). Stressor impact scores were ranked from one (low or no risk) to five (high risk, SSSRT 2010).

Within the Piscataqua River, the shortnose sturgeon population health score is one; with few historical records, their status is unknown. The 5.75 stressor score in the Piscataqua is due to dredging as a moderate stressor. The river's navigation channel is maintained with in-river disposal and dredging north of the Public Service Company of New Hampshire every five to six years.

Population estimate sampling of the Merrimack River in the winter of 2009 resulted in extremely broad confidence intervals, but suggested significantly higher estimates than 20 years previously. Within the Merrimack River, the population health score is 5.65. Shortnose sturgeon spawning has been confirmed in the Merrimack, but the population size is estimated to be less than 100 adults. The stressor score of 4.50 is due to poor water quality as a moderate stressor with periodic industrial and sewage releases during flood conditions and dissolved oxygen concentrations declining to below minimum thresholds during periods of drought or low flow.

The small (1,242-1,580 adults) but stable population of shortnose sturgeon Connecticut River have a population health score is 8.35. Spawning is confirmed in this river and all life stages are present, potentially serving as source of recruits to other nearby rivers. The stressor score for the Connecticut River population is 7.65 mainly due to impeded mobility and disrupted water flow due to the Holyoke Dam. Water quality is also a source of stress with high PCBs known to occur in fish tissues and coal tar deposits present below the Holyoke Dam are a potential source of metal exposure (Gao et al. 2016).

CRITICAL HABITAT

Critical habitat has not been designated for this species.

RECOVERY GOALS

The recovery plan identifies 19 population segments within their range with a goal of each segment maintaining a minimum population size to maintain genetic diversity and avoid extinction (NMFS 1998a). The actions needed are:

1. Establish listing criteria for shortnose sturgeon population segments;
2. Protect shortnose sturgeon and their habitats;
3. Rehabilitate shortnose sturgeon populations and habitats; and
4. Implement recovery tasks.

The recovery tasks for the Merrimack and Connecticut Rivers that are relevant to the impacts of the proposed action include analyzing contaminant loads in sturgeon tissue and habitat, determining effects of contaminants on sturgeon fitness, and identifying contaminant sources and reducing contaminant loading. These are classified as Priority 2 tasks, which are "that must be taken to prevent a significant decline in population numbers, habitat quality, or other significant negative impacts short of extinction." Tasks are not identified for the Piscataqua River due to the extremely limited population estimate at the time the recovery plan was established.

5.2.1 Atlantic Sturgeon

Five DPSs of Atlantic sturgeon were listed under the ESA in 2012. The Gulf of Maine DPS is listed as threatened while the New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs are listed as endangered (50 CFR §224.101). Sturgeon are among the most primitive of the bony fishes. The Atlantic sturgeon is a long-lived (approximately 60 years), late-maturing, iteroparous, estuarine dependent species (Dadswell 2006, ASSRT 2007). Atlantic sturgeon are anadromous, spawning in freshwater but spending most of their subadult and adult life in the saltwater environment. They can grow to approximately 4.3 meters [m]) long and can weigh up to 370 kg. Atlantic sturgeon are bluish-black or olive brown dorsally (on their back) with paler sides, a white belly, and have five major rows of dermal "scutes."

The Gulf of Maine DPS of Atlantic sturgeon occurs in the Piscataqua River Watershed, including the Salmon Falls and Cocheco tributaries up to the confluence with the Salmon Falls and Cocheco Rivers (rkm 15). This includes Great Bay, Salmon Falls River up to the Route 4 at the South Berwick Dam at rkm 7, and the Cocheco River up to the Cocheco Falls Dam at rkm 6. Spawning potentially occurs in the Salmon Falls and Cocheco Rivers based on habitat features necessary to support reproduction and recruitment and the capture of an adult female Atlantic sturgeon in spawning condition in 1990. Juveniles are potentially present throughout the rivers year-round with adults using these waters for foraging and resting during spring and fall migrations (82 FR 39160 ASSRT 2007). Atlantic sturgeon occur in the Merrimack River up to the Essex Dam at rkm 46 and are often found foraging around the lower islands reach at rkm 3-12 and the mouth of the river (Kieffer and Kynard 1993, Kynard et al. 2000). Spawning potentially occurs due to the presence of features necessary to support reproduction and recruitment, and data suggest these waters are used as a nursery for juveniles (82 FR 39160 ASSRT 2007). Based on reported sightings, adult and juvenile Atlantic sturgeon may occur within Boston Metro area waters, foraging up to Charles River Locks at rkm 5.5 and up to Dam #1 on the North River to Indian Head Reservoir at Luddam's Ford at rkm 21. Subadult and adult Atlantic sturgeon also forage in Narragansett Bay and the Taunton River up to the convergence of the Town River and Matfield River (Burkett and Kynard 1993, ASSRT 2007).

The New York Bight DPS of Atlantic sturgeon ranges from the Hudson River to the Delaware River, including the Connecticut River. The Connecticut River is designated critical habitat for this DPS of Atlantic Sturgeon. Atlantic sturgeon may occur in the Connecticut River up to the Holyoke Dam in Massachusetts at rkm 140, but mainly stay in the summer range of the salt wedge at RKM 0-26 within Connecticut (Savoy and Shake 1992, Savoy and Pacileo 2003). The capture of 45 pre-migratory juvenile Atlantic sturgeon in the lower Connecticut River provides strong evidence that natural reproduction occurs in the upper reaches of the river. The DPS designation for this population is complicated because genetic analysis indicates that these individuals were more genetically similar to the South Atlantic, Chesapeake Bay, and Carolina DPSs than the nearby New York Bight or Gulf of Maine DPSs (Savoy et al. 2017).

Migrating sturgeon within the action area include the Chesapeake Bay, Carolina, and South Atlantic DPSs. The Chesapeake Bay DPS includes all rivers of the Chesapeake Bay. Historically, Atlantic sturgeon were common throughout the Chesapeake Bay and its tributaries. The Carolina DPS includes rivers from the Albemarle Sound drainage that originate in southern Virginia, south to rivers of the Charleston Harbor area north of the Edisto River. Spawning has been confirmed in the Roanoke, Tar-Pamlico, Cape Fear, Waccamaw, and Great Pee Dee Rivers. The South Atlantic DPS includes rivers from the estuary at the mouths of the Ashepoo, Combahee, and Edisto Rivers south to the St. Johns River in Florida. Spawning has been confirmed in the Combahee, Edisto, Savannah, Ogeechee, Altamaha, and Satilla Rivers (ASSRT 2007).

LIFE HISTORY

The general life history pattern of Atlantic sturgeon is that of a long lived, late-maturing, iteroparous, anadromous species. Spawning intervals range from once every one to five years for males (Smith 1985, Bain 1997, Collins et al. 2000, Schueller and Peterson 2010) and three to five years for females (Vladykov and Greeley 1963, Bain 1997, Stevenson and Secor 1999, Schueller and Peterson 2010). Fecundity increases with age and body size (ranging from 400,000 – 8 million eggs) (Smith et al. 1982, Van Eenennaam and Doroshov 1998, Dadswell 2006). The average age at which 50% of maximum lifetime egg production is achieved is estimated to be 29 years, approximately 3-10 times longer than for other bony fish species examined (Boreman 1997).

Sturgeon eggs are highly adhesive and are deposited in freshwater or tidal freshwater reaches of rivers on the bottom substrate, usually on hard surfaces (e.g., cobble) (Gilbert 1989, Smith and Clugston 1997). Hatching occurs approximately 94-140 hours after egg deposition, and larvae assume a bottom-dwelling existence (Smith et al. 1980). The yolk sac larval stage is completed in about 8-12 days, during which time larvae move downstream to rearing grounds over a 6 – 12 day period (Kynard and Horgan 2002). During the daytime, larvae use benthic structure (e.g., gravel matrix) as refugia (Kynard and Horgan 2002). Juvenile sturgeon continue to move further downstream into waters ranging from zero to up to 10 parts per thousand salinity. Older juveniles are more tolerant of higher salinities as juveniles typically spend at least two years and sometimes as many as five years in freshwater before eventually becoming coastal residents as sub-adults (Smith 1985, Boreman 1997, Schueller and Peterson 2010).

Atlantic sturgeon feed primarily on soft-bodied benthic invertebrates like polychaetes, isopods, and amphipods in the saltwater environment, while in fresh water, they feed on oligochaetes, gammarids, mollusks, insects, and chironomids (Moser and Ross 1995, Johnson et al. 1997, Haley 1998, Haley 1999, Brosse et al. 2002, Guilbard et al. 2007, Savoy 2007, Collins et al. 2008). Diets vary latitudinally and seasonally, though universally researchers have found that polychaetes constitute a major portion of Atlantic sturgeon diets. Brosse et al. (2002) reported that over 90% of Atlantic sturgeon diet was polychaetes during spring, summer, and winter in Canada. Savoy (2007) found Atlantic sturgeon diets consisted of approximately 66% polychaetes and 27% decapods in Long Island Sound while at the mouth of the Connecticut River, individuals fed almost exclusively on polychaetes. At the mouth of the Hudson River, Haley (1999) found that sturgeon fed on 47% polychaetes, 27% amphipods, and 22% isopods. In North Carolina, Moser and Ross (1995) determined Atlantic sturgeon fed on 32% polychaetes, 28% isopods, 12% mollusks, and then other items. In South Carolina, Collins et al. (2008) identified the proportion of the sampled Atlantic sturgeon with each species in their guts and most guts contained polychaetes (over 50% of the fish that had been feeding had polychaetes in their guts).

The Gulf of Maine DPS is comprised of all Atlantic sturgeon that are spawned in the watersheds that drain into the Gulf of Maine from the Maine/Canadian border and extending southward to

Chatham, Massachusetts (77 FR 5880; February 6, 2012). Within this range, Atlantic sturgeon historically spawned in the Penobscot, Kennebec, Androscoggin, Sheepscot, and Merrimack Rivers (ASSRT 2007). Of these rivers, there was evidence of current spawning only in the Kennebec River when the Gulf of Maine DPS was listed as threatened.

The spawning area for the Gulf of Maine DPS was broadly identified in the listing rule as occurring within the tidal freshwater reach of the Kennebec River upriver of the former Edwards Dam site at rkm 74 up to the Ticonic Falls (approximately rkm 103). From 1837 to 1999, the Edwards Dam was an impassable barrier to Atlantic sturgeon and prevented them from accessing the full extent of their historical habitat in the river. Atlantic sturgeon were found in the newly accessible area after the dam was removed (Wippelhauser and Squiers 2015). Atlantic sturgeon spend two to three years in the natal estuary, using and moving within the brackish waters of the natal estuary that are most suitable for their growth and development, before emigrating to the saltwater environment. NMFS did not have information at the time of listing for the specific location of juvenile rearing habitat although the best available information supported NMFS' determination that suitable habitat was likely present in Merrymeeting Bay as well as other brackish waters of the Kennebec Estuary.

The directed movement of subadult and adult Atlantic sturgeon in the spring is from saltwater waters to river estuaries. River estuaries provide foraging opportunities for subadult and adult Atlantic sturgeon in addition to providing access to spawning habitat. Brackish waters of the Kennebec River as well as of other Gulf of Maine rivers including the Penobscot, Sheepscot, Saco, Presumpscott, and Merrimack Rivers are used by subadults, non-spawning adults, and post-spawned adults during the spring through fall. These include sub adults and adults that are not natal to the Gulf of Maine DPS. The directed movement of subadult and adult Atlantic sturgeon reverses in the fall as the fish move back into saltwater waters for the winter.

In the saltwater environment, sub adults and adults typically occur within the 50-m depth contour. Genetic analyses indicated the presence of Atlantic sturgeon belonging to the Gulf of Maine DPS in many parts of the saltwater range including the Gulf of Maine, the New York Bight, and the Bay of Fundy (77 FR 5880; February 6, 2012).

POPULATION DYNAMICS

The Gulf of Maine and New York Bight DPSs of Atlantic sturgeon was listed as threatened under the ESA on February 6, 2012 (77 FR 5880). The Gulf of Maine DPS has one known spawning population in the Kennebec River. The geomorphology of most small coastal rivers in Maine is not sufficient to support Atlantic sturgeon spawning populations, except for the Penobscot and the estuarial complex of the Kennebec, Androscoggin, and Sheepscot Rivers. During the summer months, the salt wedge intrudes almost to the site of impassable falls in these systems: St. Croix River (rkm 16), Machias River (rkm 10), and the Saco River (rkm 10). Although surveys have not been conducted to document Atlantic sturgeon presence, sub adults may use the estuaries of these smaller coastal drainages during the summer months (ASMFC 2017).

Prior to any commercial fishing in 1843, population estimates based on commercial landings from the previously unfished population indicated that approximately 10,240 adult sturgeon existed then (Kennebec River Resource Management Plan 1993). There are no modern census population estimates of the Kennebec River population. The effective population size, determined using the linkage disequilibrium method of genetic analysis of 52 adult fish, suggests between 63 and 110 adult sturgeon contributed genetically to produce the heterogeneity seen in the Kennebec River population (Waldman et al. 2019). Because most of those samples were obtained from commercial fisheries or scientific research conducted between 1978 and 2000, there are no more recent estimates of effective population size.

The Connecticut River is designated critical habitat for the New York Bight DPS of Atlantic sturgeon. Sampling within the Connecticut River (primarily with gill nets) resulted in the collection of 112 Atlantic sturgeon from 1988 through 2004. The Connecticut Department of Environmental Protection Marine Fisheries Division has finfish monitoring with a stratified random trawl survey (three bottom types, four depth intervals) collected a total of 355 Atlantic sturgeon between 1984 and 2004 (ASSRT 2007).

Status

Information on the status of Atlantic sturgeon populations is not as detailed as that for shortnose sturgeon. There is not sufficient information on the status of Atlantic sturgeon DPSs within the action area rivers to place these populations in context of the range-wide status of the species. With limited data available to establish quantitative metrics to determine stock status, it was necessary for the Atlantic States Saltwater Fisheries Commission to consider qualitative criteria such as the appearance of Atlantic sturgeon in rivers where they had not been documented in recent years, discovery of spawning adults in rivers they had not been documented before, and increases in anecdotal interactions. In some cases, qualitative metrics may be the result of increased research and attention, not a true increase in abundance (ASMFC 2017). All DPSs of Atlantic sturgeon are considered depleted. All DPSs of Atlantic sturgeon are highly vulnerable to climate change due to their low likelihood to change distribution in response to current global climate change will also expose them to effects of climate change on estuarine habitat such as changes in the occurrence and abundance of prey species in currently identified key foraging areas (NMFS 2022b, a).

The 2017 stock assessment compared the 1998 and 2015 relative abundance index values and found that the Gulf of Maine and Chesapeake Bay DPSs were below their 1998 values while the New York Bight and Carolina DPSs, as well as the coastwise stock, were above their 1998 values. The South Atlantic DPS could not be evaluated due to lack of adequate data to estimate a relative abundance index. All of the DPSs showed qualitative signs of improving populations such as increased presence of Atlantic sturgeon, including in rivers where species interactions had not been reported in recent years, and the discovery of spawning in rivers where it had not been previously documented (ASMFC 2017).

Status Within the Action Area

The NMFS 2022 status assessment for the Gulf of Maine DPS reports that this DPS has low abundance, and that the current numbers of spawning adults are one to two orders of magnitude smaller than historical levels. The status of the DPS has likely neither improved nor declined from what it was when we listed the DPS in 2012. The Kennebec River remains the only known spawning population for the Gulf of Maine DPS despite the availability of suitable spawning and rearing habitat in other Gulf of Maine rivers. The estimated effective population size is less than 70 adults, which suggests a relatively small spawning population. It is currently the only DPS with only one known spawning population (NMFS 2022a). Based on the Stock Assessment, there is a 51% probability that abundance of the Gulf of Maine DPS has increased since implementation of the 1998 fishing moratorium but also a relatively high likelihood (74% probability) that mortality for the Gulf of Maine DPS exceeds the mortality threshold used for the Stock Assessment (ASMFC 2017). However, Atlantic sturgeon are data poor, in general, and among the DPSs, the Gulf of Maine DPS is very data poor.

The NMFS 2022 status assessment for the New York Bight DPS reports that this DPS has low abundance, and that the current numbers of spawning adults are one to two orders of magnitude smaller than historical levels (NMFS 2022b). There is a relatively high probability (75%) that the New York Bight DPS abundance has increased since the implementation of the 1998 fishing moratorium, and a relatively high probability (69%) that mortality for the New York Bight DPS does not exceed the mortality threshold used for the Stock Assessment (ASMFC 2017). However, these conclusions primarily reflect the status and trend of only the Hudson River spawning population and not the Connecticut River population. Critical Habitat

In 2017 critical habitat was designated for the threatened Gulf of Maine DPS of Atlantic sturgeon, the endangered New York Bight DPS of Atlantic sturgeon, the endangered Chesapeake Bay DPS of Atlantic sturgeon, the endangered Carolina DPS of Atlantic sturgeon, and the endangered South Atlantic DPS of Atlantic sturgeon pursuant to the ESA. Specific occupied areas designated as critical habitat for the Gulf of Maine DPS of Atlantic sturgeon contain approximately 244 km (152 miles) of aquatic habitat in the following rivers of Maine, New Hampshire, and Massachusetts: Penobscot, Kennebec, Androscoggin, Piscataqua, Cocheco, Salmon Falls, and Merrimack. Specific occupied areas designated as critical habitat for the New York Bight DPS of Atlantic sturgeon contain approximately 547 km (340 miles) of aquatic habitat in the following rivers of Connecticut, Massachusetts, New York, New Jersey, Pennsylvania, and Delaware: Connecticut, Housatonic, Hudson, and Delaware (82 FR 39160). The PBFs for this designation include substrate characteristics, absence of barriers to movement, and water depth, flow, salinity, temperature, and dissolved oxygen conditions supportive of sturgeon life stages.

RECOVERY GOALS

A recovery plan has not been completed for the listed Atlantic sturgeon DPSs. However, a recovery outline has been prepared. A recovery outline is an interim guidance to guide recovery efforts until a full recovery plan is developed and approved. NMFS' vision, stated in the recovery outline, is that subpopulations of all five Atlantic sturgeon DPSs must be present across the historical range. These subpopulations must be of sufficient size and genetic diversity to support successful reproduction and recovery from mortality events. The recruitment of juveniles to the sub-adult and adult life stages must also increase and that increased recruitment must be maintained over many years. Recovery of these DPSs will require conservation of the riverine and marine habitats used for spawning, development, foraging, and growth by abating threats to ensure a high probability of survival into the future. The outline includes a recovery action to implement region-wide initiatives to improve water quality in sturgeon spawning rivers, with specific focus on eliminating or minimizing human-caused anoxic zones.

6 ENVIRONMENTAL BASELINE

The "environmental baseline" within the regulatory definition of "effects of the action," includes: "the past and present impacts of all Federal, State, or private actions and other human activities in an action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process." (50 CFR §402.02). This includes discharges and activities authorized by the EPA's administratively continued 2017 Construction General Permit, and other activities authorized by the EPA (e.g., NPDES permits, cooling water intake, air emissions, and the cleanup and management of hazardous waste) that have undergone or are in the process of completing ESA section 7 consultations. The purpose of the environmental baseline is to describe the condition of the ESA-listed species in the action area without the consequences caused by the proposed action.

The scope of the environmental baseline is largely focused on the rivers of concern within the action area, as identified on NMFS Greater Atlantic Region section 7 mapper (Figure 4)[8]. The rivers of concern include the:

- Piscataqua River in New Hampshire, including critical habitat 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;
- Cocheco River in New Hampshire, including critical habitat from its confluence with the Piscataqua River and upstream to the Cocheco Falls Dam;
- Salmon Falls River in New Hampshire, including critical habitat from its confluence with the Piscataqua River and upstream to the Route 4 Dam;

[8] https://noaa.maps.arcgis.com/apps/webappviewer/index.html?id=1bc332edc5204e03b250ac11f9914a27

- Merrimack River in Massachusetts, including critical habitat from the Essex Dam (also known as the Lawrence Dam) downstream to where the main stem river discharges at its mouth into the Atlantic Ocean;
- North River in Massachusetts;
- Taunton River of Massachusetts; and
- Connecticut River in Massachusetts, including the tributary rivers the Deerfield River and the Westfield River.

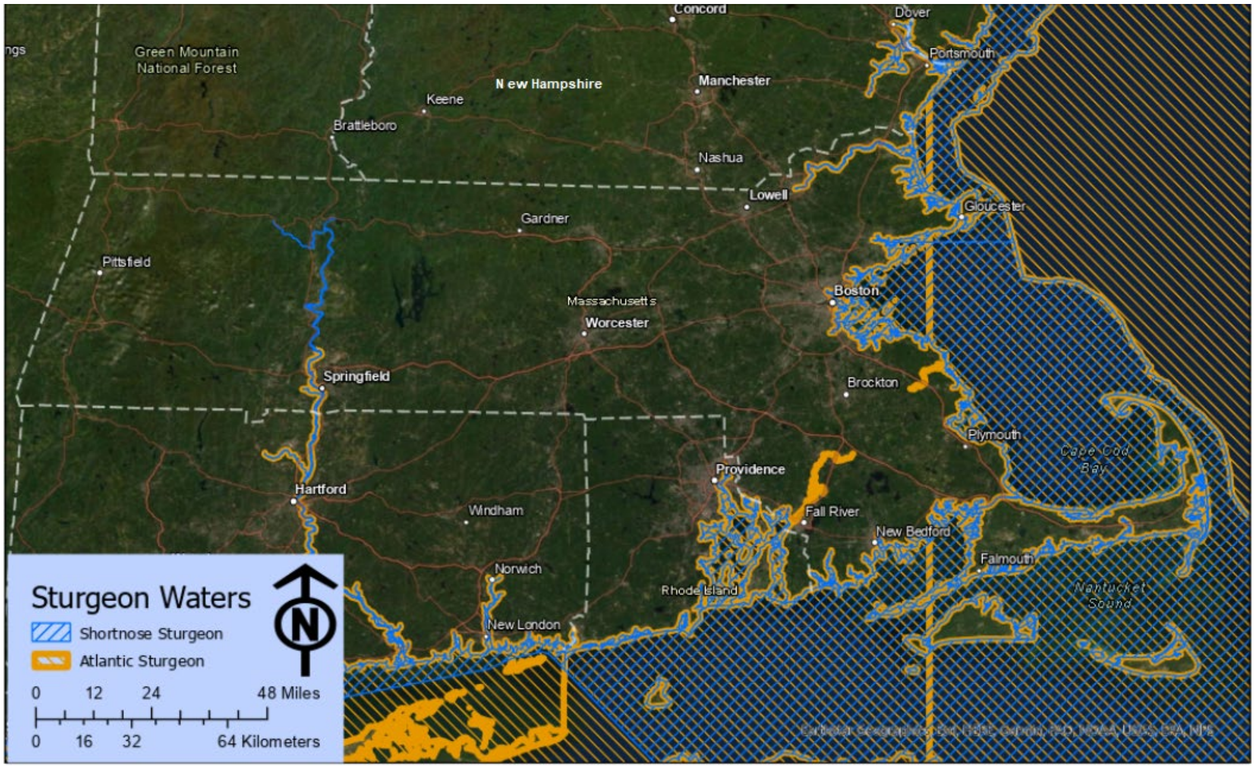

Figure 4. Waters where ESA-listed shortnose sturgeon and Atlantic sturgeon occur within southern New England.

Table 3 describes the sturgeon life stages and their behaviors in these waters. For more details, see the NMFS' Greater Atlantic Region Fisheries Office species presence tables.[9]

[9] https://www.fisheries.noaa.gov/new-england-mid-atlantic/consultations/section-7-species-presence-table-atlantic-sturgeon-greater and https://www.fisheries.noaa.gov/new-england-mid-atlantic/consultations/section-7-species-presence-table-shortnose-sturgeon-greater

Table 3. Life stages and behaviors of Atlantic sturgeon and shortnose sturgeon in the waters of New Hampshire and Masschusetts.

Body of Water (State)	Life Stages Present	Use of the Watershed
Atlantic Sturgeon		
Piscataqua River Watershed including Salmon Falls and Cocheco tributaries (NH)	sub adults and adults (eggs, larvae, young of year, and juveniles possible)	spawning, rearing, foraging
Merrimack River (MA)	sub adults and adults (potentially eggs, larvae, young of year, and juveniles)	spawning, rearing, foraging
Connecticut River (CT/MA)	eggs, larvae, young of year, juveniles, sub adults, and adults	spawning, rearing, foraging
Charles River (MA)	sub adults and adults	foraging
North River (MA)	sub adults and adults	foraging
Taunton River (MA)	sub adults and adults	foraging
Shortnose Sturgeon		
Piscataqua River (NH)	Adults	foraging
Merrimack River (MA)	eggs, larvae, young of year, juveniles, and adults	spawning, rearing, foraging, overwintering
Narragansett Bay (RI)	Adults	foraging
Thames River (CT)	adults undocumented, but assumed based on documented occurrences of Atlantic sturgeon in the river	foraging
Connecticut River (CT/MA)	eggs, larvae, young of year, juveniles, and adults	spawning, rearing, foraging, overwintering
Deerfield River (MA), **tributary of the Connecticut River**	adults documented in lower 3 km; larvae spawned in Connecticut River may be present during certain flow conditions	rearing, foraging, overwintering
Westfield River (MA), **tributary of the Connecticut River**	Adults	foraging

6.1 Existing Permitted Sources

Under the Clean Water Act, NPDES permits are renewed every five years. In both Massachusetts and New Hampshire, there are no permitted discharges for the pollutants with proposed criteria: acrolein, carbaryl, or nonylphenol. In Massachusetts, there are just under 500 facilities discharging one or more of the pollutants considered in this Opinion under a current NPDES permit and about 120 are discharging under an "Administratively Continued" permit, meaning the facility is operating under a permit that was issued more than five years ago. The National Recommended Water Quality Criteria these discharges are subject to are listed in Table 4. The locations of these facilities are illustrated in Figure 5.

Table 4. Existing water quality criteria compared with updated criteria.

Pollutant	Fresh Water		Salt Water	
	Acute	Chronic	Acute	Chronic
Aluminum (µg/L)			Not Applicable	
1988	750	87		
2018[a]	1-4,800	0.63-3,200		
Ammonia (mg/L)[b]				
1999/2009	24[c]/19[d]	4.5[c]/9.1		
2013	17[c]	1.9		
Cadmium[e] (µg/L)				
2001	2.0	0.25	40	8.8
2016	1.8	0.72	33	7.9
Copper (µg/L)			Not Applicable	
2002	13	9		
2016[a]	0.08-3479	0.05-2163		

[a] Aquatic chemistry based
[b] At pH 7 and temperature of 20 °C
[c] Salmonids present
[d] Unionid mussels present
[e] Hardness at 100 mg/L calcium carbonate

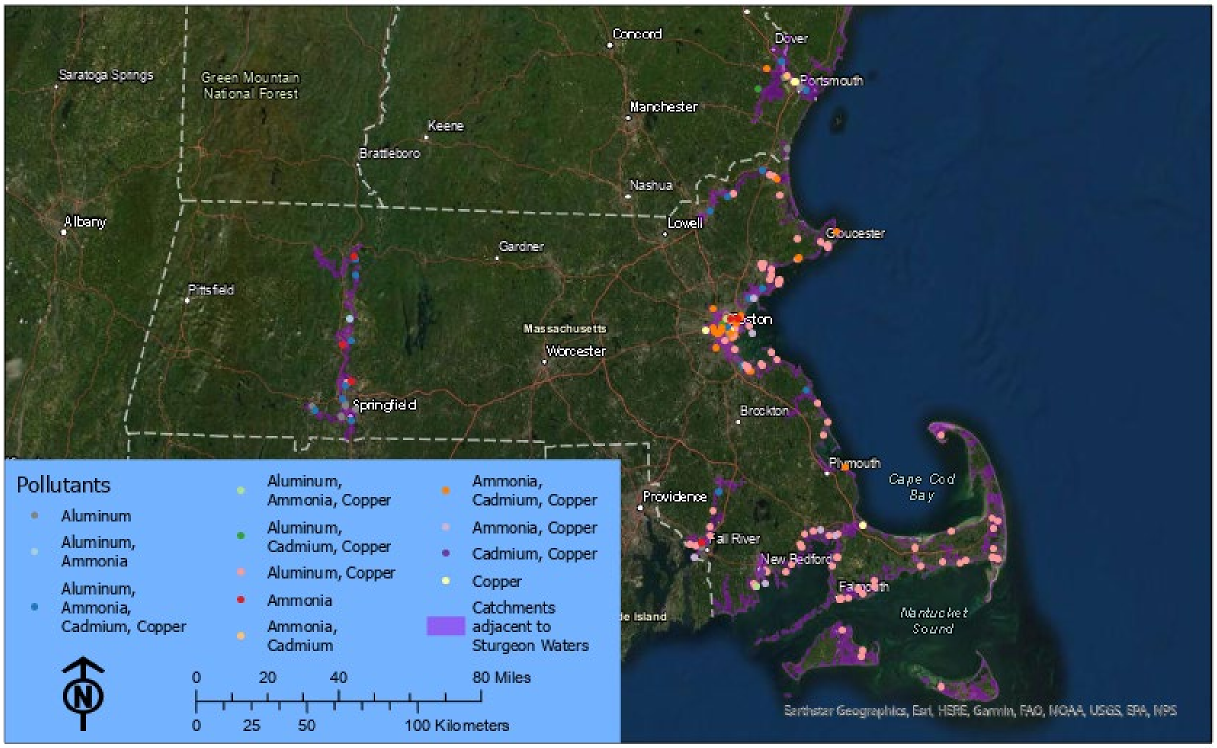

Figure 5. Locations of permitted discharges with permit limits for pollutants considered in this Opinion.

6.2 Mixture Toxicity

In point or nonpoint source pollution, chemicals occur together in mixtures, but criteria for those chemicals are developed in isolation, without consideration of additive toxicity or other chemical or biological interactions. Whether the toxicity of chemicals in mixtures is likely greater or less than that expected of the same concentrations of the chemicals singly is a complex and difficult problem. While long recognized, the "mixture toxicity" problem is far from being resolved. Even the terminology for describing mixture toxicity is dense and inconsistently used (e.g., Sprague 1970, Marking 1985, Vijver et al. 2010). One scheme for describing the toxicity of chemicals in mixtures is whether the substances show additive, less than additive, or more than additive toxicity. The latter terms are roughly similar to the terms "antagonism" and "synergism" that are commonly, but inconsistently used in the technical literature.

Relatively few toxicity studies have addressed this issue, and some studies have indicated conflicting results due to complex interactions that vary with the combination(s) and concentrations involved (Sorenson 1991). However, a number of studies have determined conclusively that adverse effects due to additive or synergistic toxicity mechanisms occur when several criteria are near or equal to acute criteria concentrations (e.g., EIFAC 1969, Alabaster and Lloyd 1982, Spehar and Fiandt 1986, Enserink et al. 1991, Sorensen 1991). Spehar and Fiandt (1986) exposed rainbow trout and *Ceriodaphnia dubia*, simultaneously, to a mixture of arsenic, cadmium, chromium, copper, mercury, and lead, each at their acute criterion, which by definition were intended to be protective. Nearly 100% of all the organisms died. In chronic tests, the authors determined that rainbow trout embryo survival and growth were not reduced when exposed to combinations of these metals at their chronic criteria concentrations. However, adverse effects were observed at mixture concentrations of one-half to one-third the approximate chronic toxicity threshold of fathead minnows and daphnids, respectively, suggesting that components of mixtures at or below NOEC concentrations may contribute significantly to the toxicity of a mixture on a chronic basis (Spehar and Fiandt 1986). Combinations of organic pollutants also have been shown to result in different toxic responses, as have combinations of organic and metals contaminants.

For both metals and organic contaminants that have similar mechanisms of toxicity (e.g., different metals, different chlorinated phenols), assuming chemical mixtures to have additive toxicity has been considered reasonable and usually protective (Alabaster and Lloyd 1982, Norwood et al. 2003, Meador 2006). The criteria evaluated in this Opinion were developed as if that pollutant was the only chemical present. However, in the real world, chemicals always occur in mixtures. As result, criteria and discharge permits based upon them may afford less protection than intended. Measures to address this potential under protection need to be included in discharge permits.

6.3 Water Quality Impairments

The Clean Water Act requires states and territories to assess water quality every two years under 305(b) and identify waters that are impaired under 303(d) and in need of restoration. Restoration is achieved by establishing the maximum amount of an impairing pollutant allowed in a waterbody, or TMDL. These assessments are sent as an integrated report every even numbered year to EPA, which must approve of each impaired waters' listing. As a result, many recent state assessments are not finalized until the following year or later.

The EPA approved New Hampshire's 2020-2022 303(d) list for freshwaters in March of 2022. The Cocheco and associated tributaries remain impaired by PAHs, legacy organochlorine pesticides, lead, aluminum, iron, pH, low dissolved oxygen, and other stressors contributing to the impairment of the biological community (e.g., nutrients, flashiness). At this time, there is insufficient information for listing the current status of the Piscataqua River, but it is likely still impaired. Approved TMDLs for fecal coliform and enterococcus are now in place for these Piscataqua River impairments. For the Salmon Falls River, impairments include impaired biological communities, indicators of eutrophication (chlorophyll-a, dissolved oxygen and oxygen saturation, and TN), dioxin, mercury, PCBs, and pH. Approved TMDLs for mercury and dissolved oxygen are now in place for certain segments of the Salmon Falls River. Approved TMDLs are also in place for enterococcus, *Escherichia coli*, fecal coliform, and non-native aquatic plant impairments. The 2018 assessment did not include saltwater waters, but the draft 2022 303(d) list adds assessment zones located in Great Bay impaired by eutrophication indicators chlorophyll-a and TN.

The EPA approved Massachusetts's 2018-2020 303(d) list in February of 2022. No changes to the listing status of the Connecticut River, the Deerfield River and Westfield River tributaries to the Connecticut River, the Merrimack River, or the Taunton River were noted in EPA's approval letter. The 2016 assessment identified additional *Escherichia coli* impairments in segments of the Merrimack, Taunton, and Connecticut Rivers. Indicators of sewage and eutrophication impairments were also identified for 13 harbor and bay segments. New enterococcus, nitrogen, and estuarine community impairments were identified for 14 waters with existing TMDLs and these impairments were incorporated into the existing TMDL. New TMDLs were established for 12 harbor and bay segments: eight for fecal coliform and four for Enterococcus. Restoration activities resulted in use attainment for the temperature impairments in Mount Hope Bay.

6.4 Municipal Separate Storm Sewer Systems

Municipal Separate Storm Sewer Systems (MS4s) are conveyances or a system of conveyances that are:

- owned by a state, city, town, village, or other public entity that discharges to Waters of the United States,
- designed or used to collect or convey stormwater (e.g., storm drains, pipes, ditches),
- not a combined sewer, and
- not part of a sewage treatment plant, or publicly owned treatment works

The Clean Water Act Section 402(p)(3)(B) states that permits for MS4 discharges may be issued on a system or jurisdiction-wide basis, and must effectively prohibit non-stormwater discharges into the sewer system. Stormwater discharges regulated under an MS4 permit represent a baseline stormwater impact to which other regulated discharges are added. In 2016, EPA Region 1 issued an MS4 General Permit for stormwater discharges within urbanized areas of Massachusetts and New Hampshire. Recent modifications clarifying requirements of permit holders become effective in January 6, 2021. In August of 2016, NMFS Greater Atlantic Region Field Office completed informal consultation, concurring with the conclusion made by EPA Region 1 that the proposed MS4 General Permit for Massachusetts and New Hampshire may affect, but is not likely to adversely affect NMFS ESA-listed species and/or designated/proposed critical habitat within the action area of the permit.

6.5 Climate Change

Climate change is discussed both here in the environmental baseline section of this Opinion and in the cumulative effects section (Section 9), because it is a current and ongoing circumstance that, for the most part, is not subject to consultation, yet influences environmental quality and the effects of the action, currently and in the future. NMFS' policy guidance with respect to climate change when evaluating an agency's action is to project climate effects over the timeframe of the action's consequences. The EPA's approval and subsequent implementation of water quality criteria is an example of an action that will be in effect indefinitely.

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

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

A set of four scenarios was developed by the Intergovernmental Panel on Climate Change (IPCC) to ensure that starting conditions, historical data, and projections are employed consistently across the various branches of climate science. The scenarios are referred to as representative concentration pathways (RCPs), which capture a range of potential greenhouse gas emissions pathways and associated atmospheric concentration levels through 2100 (IPCC 2014). The RCP scenarios drive climate model projections for temperature, precipitation, sea level, and other variables: RCP2.6 is a stringent mitigation scenario; RCP4.5 and RCP6.0 are intermediate scenarios; and RCP8.5 is a scenario with no mitigation or reduction in the use of fossil fuels. IPCC future global climate predictions (IPCC 2014, 2018) and national and regional climate predictions included in the Fourth National Climate Assessment for U.S. states and territories (USGCRP 2018) use the RCP scenarios.

The IPCC is currently in its Sixth Assessment cycle, the AR6 Climate Change 2022: Impacts, Adaptation and Vulnerability report is still in final draft form and are subject to approval and revisions, so quantitative estimates are not reviewed here (IPCC 2022). The report summary states that observed increases in the frequency and intensity of climate and weather extremes attributed to human induced climate change, includes hot extremes on land and in the ocean, heavy precipitation events, drought and fire weather has led to widespread, pervasive impacts to ecosystems. The recent assessment concluded that the substantial damages, and increasingly irreversible losses, in terrestrial, freshwater and coastal and open ocean marine ecosystems are larger than estimated in previous assessments. The report indicates that widespread deterioration of ecosystem structure and function, resilience and natural adaptive capacity, and shifts in seasonal timing is occurring under climate change. Approximately half of the species assessed globally have shifted poleward or, on land, also to higher elevations. Hydrological changes resulting from the retreat of glaciers, or the changes in ecosystems driven by permafrost thaw are approaching irreversibility.

The increase of global mean surface temperature change by 2100 is projected to be 0.3 to 1.7°C under RCP2.6, 1.1 to 2.6°C under RCP4.5, 1.4 to 3.1°C under RCP6.0, and 2.6 to 4.8°C under

RCP8.5 with the Arctic region warming more rapidly than the global mean under all scenarios (IPCC 2014). The Paris Agreement aims to limit the future rise in global average temperature to 2°C, but the observed acceleration in carbon emissions over the last 15 to 20 years, even with a lower trend in 2016, has been consistent with higher future scenarios such as RCP8.5 (Hayhoe et al. 2018).

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

The Atlantic Ocean appears to be warming faster than all other ocean basins except perhaps the southern oceans (Cheng et al. 2017). In the western North Atlantic Ocean, surface temperatures have been unusually warm in recent years (Blunden and Arndt 2016). A study by Polyakov et al. (2010) suggests that the North Atlantic Ocean overall has been experiencing a general warming trend over the last 80 years of 0.031±0.0006 ºC per decade in the upper 2,000 m of the ocean. Additional consequences of climate change include increased ocean stratification, decreased sea-ice extent, altered patterns of ocean circulation, and decreased ocean oxygen levels (Doney et al. 2012). Since the early 1980s, the annual minimum sea ice extent (observed in September each year) in the Arctic Ocean has decreased at a rate of 11 to 16% per decade (Jay et al. 2018). Further, ocean acidity has increased by 26% since the beginning of the industrial era (IPCC 2014) and this rise has been linked to climate change. Climate change is also expected to increase the frequency of extreme weather and climate events including, but not limited to, cyclones, tropical storms, heat waves, and droughts (IPCC 2014).

Climate change has the potential to influence species abundance, geographic distribution, migration patterns, and susceptibility to disease and contaminants, as well as the timing of seasonal activities and community composition and structure (Kintisch and Buckheit 2006, Mcmahon and Hays 2006, Robinson et al. 2008, Macleod 2009, Evans and Bjørge 2013, IPCC 2014). The loss of habitat because of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents (Antonelis et al. 2006, Baker et al. 2006).

Changes in the saltwater ecosystem caused by global climate change (e.g., ocean acidification, salinity, oceanic currents, DO levels, nutrient distribution) could influence the distribution and abundance of lower trophic levels (e.g., phytoplankton, zooplankton, submerged aquatic vegetation, crustaceans, mollusks, forage fish), ultimately affecting primary foraging areas of ESA-listed species. Saltwater species ranges are expected to shift as they align their distributions to match their physiological tolerances under changing environmental conditions (Doney et al. 2012). Similarly, climate-related changes in important prey species populations are likely to affect predator populations. Changes in core habitat area means some species are predicted to experience gains in available core habitat and some are predicted to experience losses (Hazen et al. 2012).

6.6 Impervious Cover

The oldest available impervious cover data from the National Land Cover Dataset is from 2001 and the most recent is from 2019. Table 5 summarizes the change in impervious cover between 2001 and 2019 for catchments immediately adjacent to Sturgeon Waters and catchments abutting water-adjacent catchments. Data for Massachusetts are divided into regions within the state to allow comparison of highly urbanized areas of the state (e.g., Plymouth to Essex) with relatively less developed areas such as the Connecticut River Valley (Figure 6). For example, Figure 7 illustrates the incremental spread of impervious cover along the Connecticut River in Massachusetts. According to Arnold and Gibbons (1996), runoff doubles in forested catchments that are 10 to 20% impervious, triples between 35 and 50% and increases more than five-fold at above 75% impervious. Catchments that shifted from below 10% impervious cover in 2001 to greater than 10% impervious in 2019 are typically adjacent to existing areas of increased impervious cover. These are highlighted in Figure 7 using an aqua-to-fuchsia color scale to illustrate the degree of impervious cover change. For example, impervious cover at 5% in 2001 and 6.5% in 2019 is a 30% increase in impervious cover.

Table 5. Summary of impervious cover and proportion of region, for catchments adjacent to waters where ESA-listed species under NMFS' jurisdiction occur.

Region	Catchment area (km2)	2001 catchment area already >10% impervious cover	Catchment area increased to >10% impervious cover by 2019	2019 catchment area still <10% impervious cover
Connecticut River (MA)	825.25	340.89 (41.3%)	16.07 (1.9%)	468.28 (56.7%)
Buzzards Bay/Taunton River (MA)	1097.19	494.87 (45.1%)	57.78 (5.3%)	544.54 (49.6%)
Cape Cod and Islands (MA)	1389.27	761.97 (54.8%)	40.95 (2.9%)	586.36 (42.2%)
Sandwich to Hingham (MA)	529.25	243.42 (46.0%)	54.89 (10.4%)	230.94 (43.6%)
Hingham to Essex (MA)	687.37	621.16 (90.4%)	9.87 (1.4%)	56.34 (8.2%)
Essex to Lowell (Merrimack River, MA)	450.18	264.87 (58.8%)	11.50 (2.6%)	173.81 (38.6%)
New Hampshire	349.90	169.26 (48.4%)	15.74 (4.5%)	164.91 (47.1%)

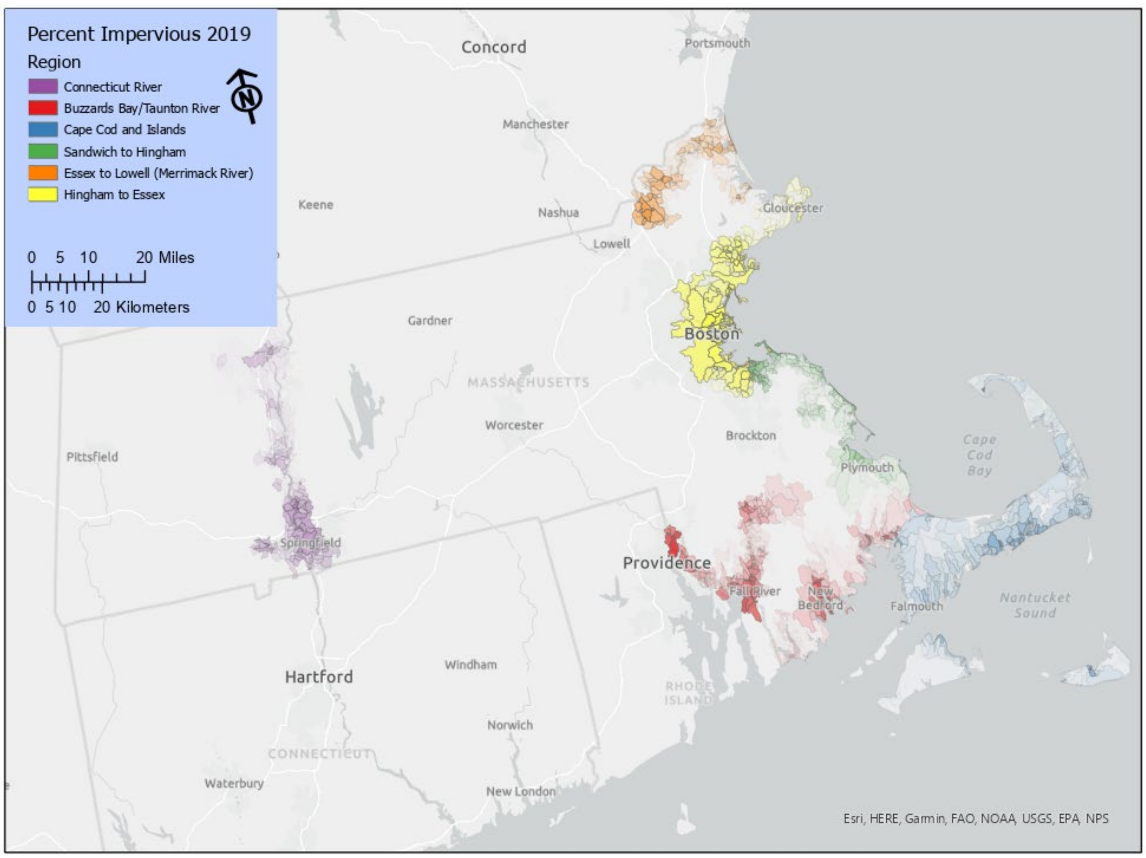

Figure 6. Relative impervious cover for Massachusetts catchments adjacent to waters where ESA-listed species under NMFS' jurisdiction occur (opaque = highly impervious).

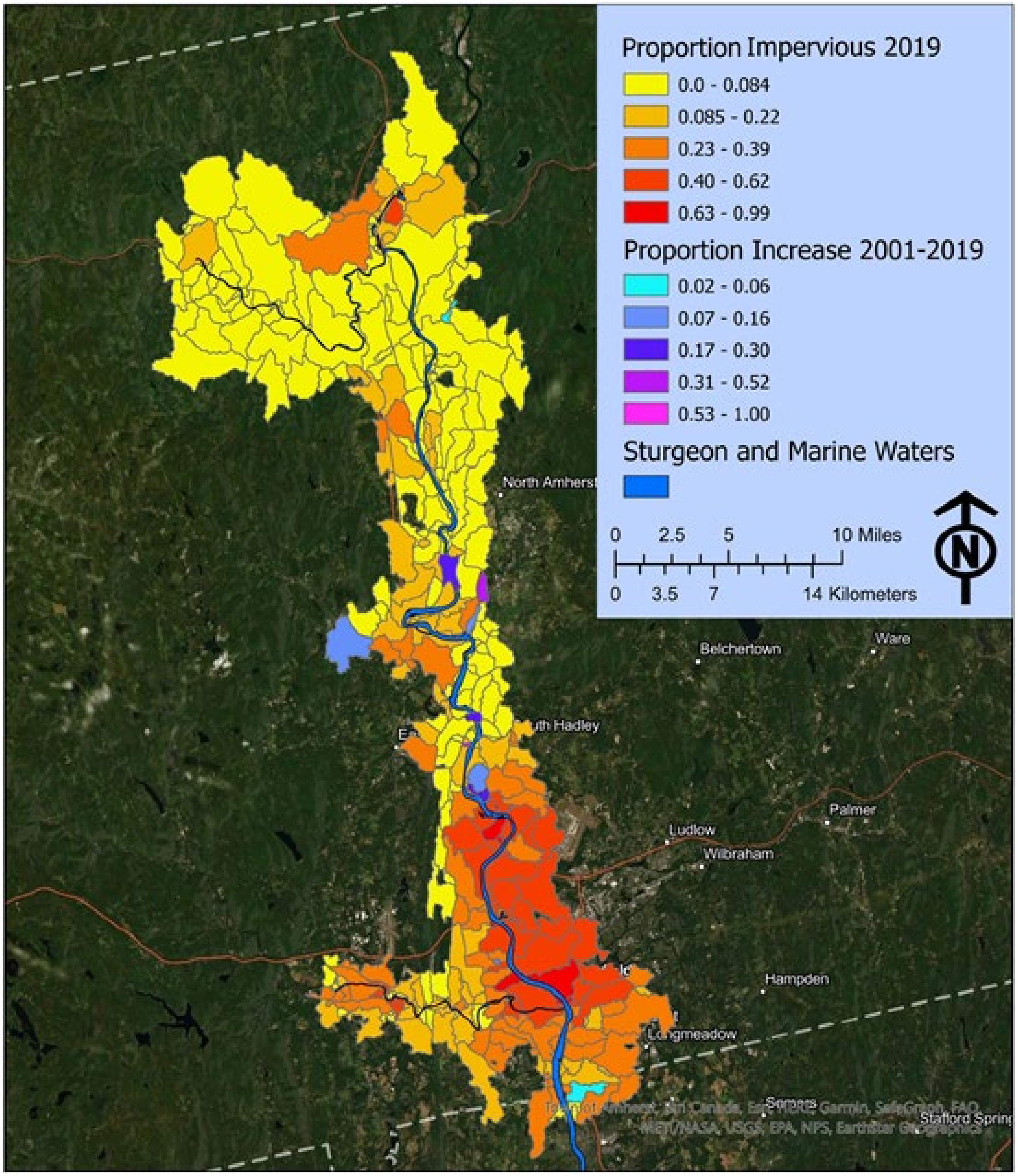

Figure 7. Proportion impervious cover (yellow-to-red) within catchments along the Connecticut River with catchments that increased to greater than 10% impervious cover (aqua-to-fuscia) between 2001 and 2019.

6.7 Climate Change and the Built Environment

The aggregate effects of an increasingly built environment affecting watersheds where species and designated critical habitat under NMFS' jurisdiction occur interacts with climate change-driven shifts in precipitation to result in a continually shifting baseline. Aggregate impacts include:

- time-crowded perturbations (i.e., repeated occurrence of one type of impact in the same area) or perturbations that are so close in time that the effects of one perturbation do not dissipate before a subsequent perturbation occurs;
- space-crowded perturbations (i.e., a concentration of a number of different impacts in the same area) or perturbations that are so close in space that their effects overlap;
- interactions or perturbations that have qualitatively and quantitatively different consequences for the ecosystems, ecological communities, populations, or individuals exposed to them because of synergism (when stressors produce fundamentally different effects in combination than they do individually), additivity, magnification (when a combination of stressors have effects that are more than additive), or antagonism (i.e., when two or more stressors have less effect in combination than they do individually); and
- nibbling (e.g., the gradual disturbance and loss of land and habitat) or incremental and decremental effects are often, but not always, involved in each of the preceding three categories (NRC 1986).

Climate change influences on precipitation frequency and intensity interacting with increasing impervious cover intensifies risk to surface water quality through increased pollutant transport and erosive flow. Further, changes in plant cover and soil structure under climate change will influence infiltration potential (Lal 2015). Annual precipitation in the state of New Hampshire has increased by an average of 6.8 inches over the 1895-2004 average (Runkle et al. 2022). Records for Massachusetts indicate average annual precipitation increased by 4.7% over the 1895-1969 average (Runkle 2022). Both states are projected to have significant increases in spring precipitation of between 5 and 15%. Climate change models indicate a five to 10% increase in annual precipitation. (Frankson et al. 2022a, Frankson et al. 2022b, Frankson et al. 2022c).

The extent to which existing stormwater control technologies and best management practices will be effective under increasingly challenging stormwater conditions has yet to be proven. The increasing impervious area taken with anticipated increases in annual and seasonal precipitation is expected to result in more frequent and extreme uncontrolled stormwater discharges that, in turn, will likely to adversely affect water quality and aquatic life through erosive waters and contribution of land-sourced pollutants.

7 STRESSORS ASSOCIATED WITH THE ACTION

Stressors are any physical, chemical, or biological entity that may induce an adverse response in either an ESA-listed species or their designated critical habitat. Typically, this section of an opinion would disaggregate a proposed action to identify and describe the specific stressors expected to result from the action, including the sources and fate and transport of chemicals. In this case, the stressors of the action are the toxicants and the naturally occurring aquatic characteristic, TN, for which criteria are being proposed (see Section 3, Description of the Action). Because this action involves criteria for eight parameters, the analysis is structured on a parameter-by-parameter basis under the Effects of the Action section in order to maintain focus on one parameter at a time.

8 EFFECTS OF THE ACTION

"Effects of the action" refers to the direct and indirect effects of an action on the species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action that will be added to the environmental baseline. Indirect effects are those that are caused by the proposed action and are later in time, but still are reasonably certain to occur. Interrelated actions are those that are part of a larger action and depend on the larger action for their justification. Interdependent actions are those that have no independent utility apart from the action under consideration. This analysis focuses on any data that indicate exposures within criterion limits may result in short or long-term adverse effects to ESA listed shortnose and Atlantic sturgeon or result in reduction in the quantity or quality of available prey, as described through risk hypotheses identified in the Assessment Framework of this opinion (Section 2) repeated below:

- Reduced survival of individuals through direct mortality or effects favoring predation (e.g., immobility, reduced predator detection)
- Reduced growth of individuals through direct effects of toxicity or effects impairing foraging (e.g., swimming, prey detection, strike success)
- Reduced fecundity through direct effects of toxicity (e.g., reduced hatch, egg mass, egg counts) or effects impairing reproduction (e.g., impaired nest tending, gonads mass)
- Reduced survival, growth, and/or fecundity due to reduced quantity or quality of forage due to toxic effects on forage species abundance or toxic effects of body burdens of the stressor in forage species

8.1 Criteria that are Not Likely to Adversely Affect ESA-listed Species Under NMFS'

Jurisdiction

8.1.1 Total Nitrogen Criteria for the Prevention of Eutrophication

EPA proposes to approve site-specific nitrogen targets to prevent eutrophication in six specific embayments of Cape Cod. NMFS' Greater Atlantic ESA Section 7 Mapper indicates that ESA-listed shortnose sturgeon and Gulf of Maine and New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon would use three of these embayments: Oyster Pond, Mill Pond, and Ryders Cove.

The Oyster Pond and Mill Pond nitrogen criteria are TMDL targets calculated to correct eutrophication-related impairments attributed to excessive nitrogen (N) originating primarily from septic systems resulting in significant decreases in the "environmental quality" of coastal rivers, ponds, and harbors in many communities in southeastern Massachusetts. The specific indicators of eutrophic conditions at this time include reduction of eelgrass beds, increased macroalgae, periodic extreme decreases in dissolved oxygen concentrations, reduced benthic biological diversity, and periodic algae blooms (MassDEP 2007, 2008). If left unchecked, MassDEP anticipates fish kills, unpleasant odors and scum, and depleted benthic diversity.

Under natural conditions, nitrogen contributes to the proper structure and function of healthy ecosystems. However, in excessive quantities, nitrogen can have adverse effects on ecosystems and often ranks as one of the top causes of water resource impairment (Bricker et al. 2008, USEPA 2014). It typically takes years for nutrient load reductions to shift nutrient regimes away from eutrophic conditions or a trajectory towards eutrophy due to internal (e.g., sediment, biota) inputs to a system (Greening and Janicki 2006, Bell et al. 2008, Bell et al. 2014, Greening et al. 2014, Riemann et al. 2016, Staehr et al. 2017). Riemann et al. (2016) reviewed recovery of Danish coastal waters following substantial reductions in nitrogen and phosphorus loading in the 1990s. Trends between 1990s and 2013 include an overall decline in chlorophyll-a of −0.057 micromole per liter per year ($p< <0.0001$), change in water depth for eelgrass growth of −0.006 m/year ($p=0.0080$), and increased macroalgae cover of 0.69%/year ($p= 0.0007$). Taken together, these studies show that individual waterbodies respond differently to changes in nutrient loading. The presence of legacy nutrients, external unmanaged sources, and hydrological variability can result in short term spikes or declines that are not directly attributable to managed loading reductions.

At this time, it is not possible to determine whether or when implementation of the site-specific nitrogen criteria EPA proposes to approve will succeed in preventing or reversing eutrophication in these embayments. NMFS looked for, but did not find, monitoring data or the monitoring plan required under the 2007 TMDL.

Although the effectiveness of the proposed TN criteria cannot be known at this time, NMFS determines that EPA's approval of Massachusetts' site-specific TN criteria proposed for Oyster Pond, Mill Pond, and Ryders Cove is not likely to adversely affect the Gulf of Maine and New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon, and shortnose sturgeon because these species only use Oyster Pond, Mill Pond, and Ryders Cove opportunistically, such that any exposure times and effects would be too small to be detected and thus insignificant.

8.1.2 Acrolein Criteria for the Protection of Aquatic Life in Freshwater

Acrolein is a contact herbicide that binds to organic material and degrades cellular structure by cross-linking proteins. It is used to control aquatic weeds and algae, as a biocide in recirculating industrial water systems, and in some industrial processes (USEPA 2008). Sources of acrolein in the environment include emissions and effluents from its manufacturing and use facilities, emissions from combustion processes, direct application to water and waste water as a slimicide or aquatic herbicide, as a photooxidation product of various hydrocarbon pollutants found in air, and from land disposal of some organic waste materials (Faroon et al. 2008). Acrolein is a reactive compound and is unstable in the environment; most National Pollutant Discharge Elimination System permit post-treatment monitoring data are non-detects (USEPA 2008). However, EPA (2008) reports one extreme case from NPDES monitoring after biocide treatment at the maximum rate of 15 mg/L for eight hours. The monitoring showed that acrolein can be detected at a concentration of 67 μg/L up to 61 miles away and 54 hours after treatment. This exposure concentration is five-fold EPA's level of concern based on toxicity data for fathead minnow (Bartley and Hattrup 1975).

Reported half-lives for acrolein range from two to 20 hours. A half-life of approximately seven hours was observed for acrolein in freshwater by (Nordone et al. 1998), but the authors noted that the dissipation rate was both concentration and temperature dependent. The presence of microbial populations also heavily influences the acrolein degradation rates in freshwater systems (Smith et al. 1995). A half-life of approximated 4.3 hours was reported for acrolein in flowing water, dissipating rapidly through volatilization, degradation, and absorption (Bowmer and Sainty 1977).

Both the acute and chronic freshwater guideline concentration for acrolein is 3 μg/L applied as the CMC limit for one-hour average exposures and also as the CCC for four-day average exposures (USEPA 2009). Typically, acute criteria are higher concentrations than chronic criteria, but in the case of acrolein, the data available for calculating a recommended chronic water quality guideline had similar threshold concentrations for lethal and sublethal responses. In such cases, instructions in EPA's 1985 guidance for calculating recommended chronic guideline concentrations result in identical concentrations for both acute and chronic exposures (Stephen et al. 1985). Since the acute and chronic criteria for acrolein are identical, our evaluation addresses the exposure concentration irrespective of exposure duration.

The BE evaluated the protectiveness of the acrolein criterion concentration using an adjustment factor for acute exposures and comparison with "low effect" thresholds reported chronic exposures. Since data are not available for acrolein toxicity to shortnose sturgeon, Atlantic sturgeon, or species within the taxonomic order Acipenseriformes, the BE calculated acute adjustment factors using the GMAVs of all species within the same taxonomic class, Actinopterygii. The adjusted LC05s ranged from seven to 80 µg/L, two or more times the criterion concentration of three µg/L.

8.1.2.1 Exposure to Acrolein in the Action Area

There are no permitted sources of acrolein in Massachusetts or New Hampshire. No acrolein was applied in New Hampshire by registered applicators from 2018 – 2020 (Rousseau 2022) and the use of acrolein in the New England region is considered unlikely (USEPA 2022). At this time, there are no waters listed as impaired by acrolein in Massachusetts or New Hampshire (MDEP 2021, NHDES 2022). Monitoring data for Massachusetts and New Hampshire indicate that acrolein was only monitored for in 1998 in groundwater near Orleans on Cape Cod. Acrolein was not detected, but the detection limit was 250 µg/L (National Water Quality Monitoring Council Accessed April 5, 2022). The single acrolein-containing pesticide product registered for use in Massachusetts and New Hampshire is registered for terrestrial uses only[10]. The label includes the warning:

> *"Do not apply directly to water, or to areas where surface water is present or to intertidal areas below the mean high-water mark. Do not contaminate water when disposing of equipment wash water or rinsate."*

Taken together, the absence of products registered for use in the presence of water or as an industrial biocide, along with the absence of permitting limits and reported applications and routine monitoring suggests that exposure of ESA-listed shortnose and Atlantic sturgeon to acrolein are extremely unlikely to occur in the waters of Massachusetts or New Hampshire.

8.1.2.2 Responses to Acrolein Exposures Within Criteria Limits

The screened ECOTOX data included data for ten fish species from seven taxonomic families and seven aquatic invertebrate species from seven taxonomic families, to represent sturgeon forage species. There are no data for effects of acrolein on fish or aquatic invertebrate reproduction and only a single study reporting the effects of acrolein on growth in fish (Figure 8: available data in context of reference lines representing the applicable criterion and one-half the applicable criterion). Data for exposures of invertebrates are most abundant, and are one or more orders of magnitude greater than the acrolein criterion concentration of 3 µg/L. While endpoint thresholds found in the screened toxicity data for fish exceed the acrolein criterion concentration of three µg/L by at least threefold, interpreting these thresholds and determining what they mean for this Opinion must consider whether a lower, but still unacceptable level of response, may occur at the criteria acute and chronic exposure concentration of 3 µg/L. The ECOTOX database does not provide information on response magnitude at endpoint concentrations and study documentation often did not provide this information.

[10] A single product, Seican (91473-2-88783) containing 22.5 percent acrolein is registered for use in the states Massachusetts and New Hampshire. (Kelly Solutions Access May 9, 2022)

Those few studies reporting response magnitudes at endpoint concentrations illustrate how reliance on NOECs and LOECs can lead to less-than-protective decisions. For acrolein, two fathead minnow studies reported response magnitudes at toxicity test exposure concentrations, but all test exposures in the study exceeded the acrolein criterion concentration of 3 µg/L. In the absence of exposure concentrations similar to the criterion concentration, mortality LOECs for fathead minnows were observed at exposures ranging from 22.4 µg/L (20% mortality at 27 hours) to 19.2 µg/L (45% mortality at 96 hours, Geiger et al. 1988, 1990). Both studies exposed fish for a total four days and it was necessary to refresh the exposure media daily due to acrolein degradation. A single study comparing acrolein toxicity in several species reported mortality ranging from 1% at the LOEC of 1,620 µg/L for *Chironomus spp*. to 20% at the LOEC of 25 µg/L for rainbow trout. The response magnitudes at the NOECs in this study ranged from <1% at 1,000 µg/L for *Chironomus spp*. to 0.5% at 10 µg/L for rainbow trout (Venturino et al. 2007).

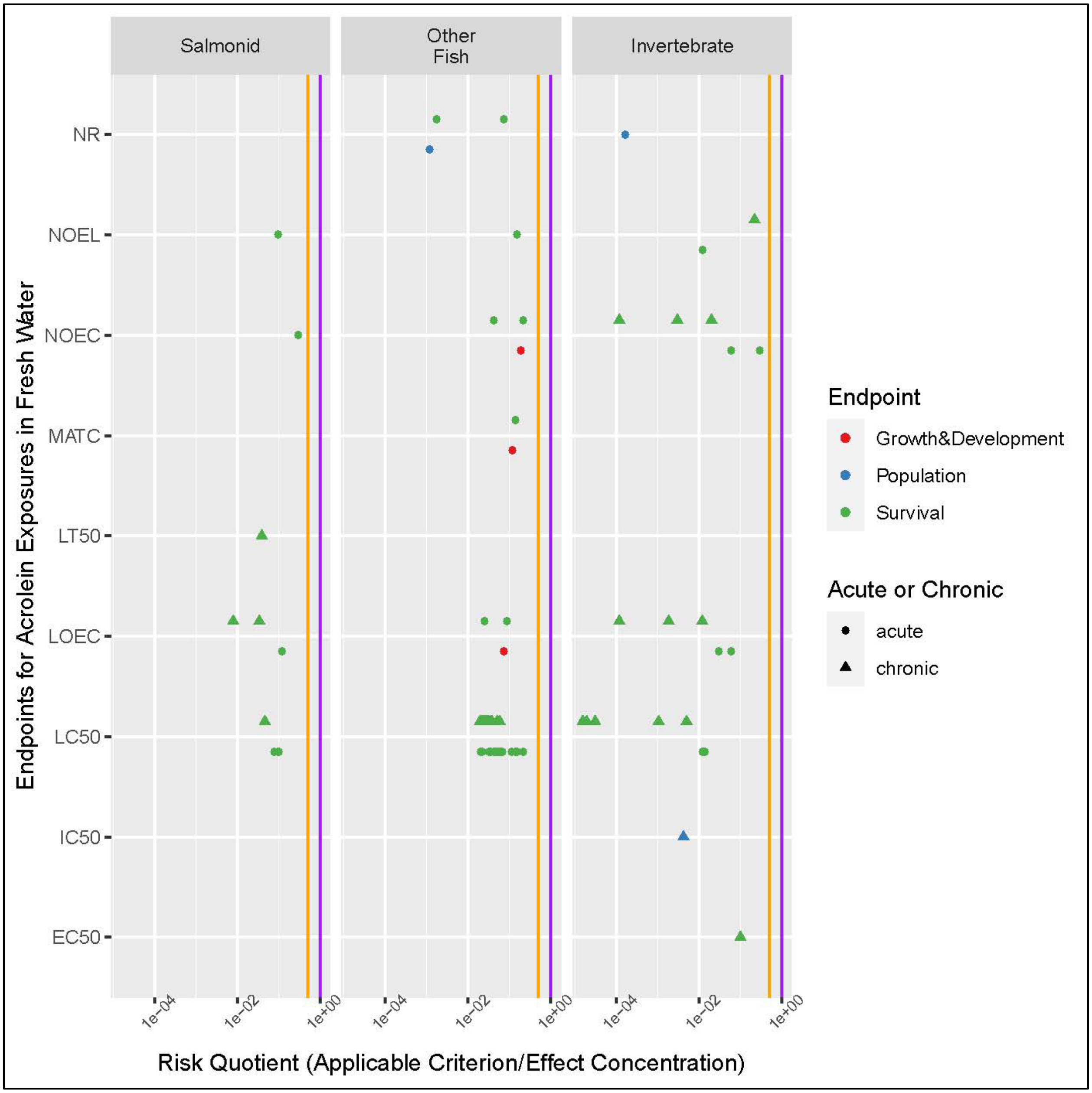

Figure 8. Distribution of risk quotients for freshwater exposures to acrolein in context of reference lines representing the applicable criterion (purple) and one-half the applicable criterion (orange).

There were few data for chronic exposures of Actinopterygii. The BE reported the lowest chronic value at 11.4+/-8.3 µg/L based on survival of newly hatched fry during the continuous exposure of fathead minnows (Macek et al. 1976). While this reported "low effect" threshold is nearly four-fold the criterion concentration of 3 µg/L, the wide confidence interval accompanying this data indicates a range from 3 to 32 µg/L (Table 10 in Macek et al. 1976).

In the case of the acrolein data, the average adverse effect risk quotients (i.e., excluding NOECs) for fathead minnow (n=12) and rainbow trout (n=4) LC50s were 0.07 and 0.08, respectively. Six of the fathead minnow LC50s and three of the rainbow trout LC50s exceeded the EPA OPP's level of concern for ESA-listed species (Table 6). The data suggest that brief exposures at the proposed criterion are more likely to meet the EPA's protective level for ESA-listed species than longer exposures. The fathead minnow LC50 risk quotients above 0.05 were four-day exposures while exposure durations for those at or less than 0.05 ranged from one to six days with five exposures at three days or less. The only rainbow trout LC50 risk quotient below 0.05 was for a one-day exposure. The remaining data are for four-day exposures.

Table 6. Risk quotients and LC50s for fathead minnows and rainbow trout exposed to acrolein.

Species	Risk Quotient	LC50 µg/L	Exposure Duration (days)	Source
Fathead Minnow	0.214	14	4	(Geiger et al. 1988)
	0.154	19.5	4	(Geiger et al. 1990)
	0.111	27	4	(Spehar 1989)
	0.111	27	4	
	0.067	45	4	(Birge et al. 1982)
	0.049	61	4	
	0.026	115	2	(Louder and McCoy 1962)
	0.02	150	1	
	0.02	150	4	(Turner 1982)
	0.02	150	2	
	0.02	150	3	
	0.02	150	1	
Rainbow Trout	0.103	29	4	(Mckim et al. 1987)
	0.079	38	4	(Venturino et al. 2007)
	0.046	65	1	(Bond et al. 1960)

The risk quotients for invertebrate species exposed to acrolein within criterion limits indicate that the criteria are least an order of magnitude lower than concentrations at which invertebrate prey species for sturgeon would respond.

8.1.2.3 Not likely to Adversely Affect Determination for Acrolein Exposures Within Criteria Limits

NMFS' determination for acrolein does not distinguish between acute and chronic exposures because the criterion concentrations are identical regardless of the exposure duration and frequency. The average acute risk quotient for rainbow trout, a surrogate species for sturgeon, exposed to acrolein at the criterion concentration is greater than the OPP level of concern risk quotient of 0.05 for ESA-listed aquatic species. Meanwhile all LOEC risk quotients were below the OPP risk quotient level of concern of one for both fish and aquatic invertebrates.

The best available data suggest that adverse effects may occur in ESA-listed shortnose and Atlantic sturgeon for sustained exposures to acrolein at the criterion concentration of 3 μg/L. The screened data lacked information on the effects of acrolein on reproduction and behavior and only had a single observation for effects on growth. With such data limitations, NMFS gives the species the benefit of the doubt. However, the available data also suggest that exposures are extremely unlikely to occur in Massachusetts or New Hampshire waters, given:

1) The absence of any acrolein-containing product registered for use as a biocide or as a pesticide approved for use in the presence of water,
2) The absence of permitted discharges required to monitor for acrolein, suggesting it is not expected to be present, is an insignificant discharge component, or is not expected to contribute to toxicity for those discharges conducting whole effluent toxicity testing, and
3) The half-life of acrolein in natural flowing waters where ESA-listed sturgeon would occur is on the order of hours, making sustained exposures, if they occur, extremely unlikely.

NMFS concludes that EPA's approval of Massachusetts and New Hampshire adoption of the National Water Quality Criteria for acrolein may affect, but is not likely to adversely affect shortnose sturgeon or the Gulf of Maine and New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPS' of Atlantic sturgeon or reduce the quantity or quality of their prey because exposures to acrolein in waters regulated by MassDEP and NHDES are extremely unlikely to occur and are therefore discountable.

8.1.3 Ammonia Criteria for the Protection of Aquatic Life in Freshwater

Sources of ammonia in surface waters include domestic and industrial wastes, land management, and agricultural practices. Excess stormwater flow may be diverted from municipal waste treatment plants and public-owned treatment works (POTWs) into combined sewer overflows that deposit untreated municipal waste directly into streams and lakes. As wastewater treatment infrastructure ages, increasingly frequent treatment plant failures also may result in high ammonia releases to streams (Boulos 2017). Significant amounts of ammonia may leach into surface waters from failing septic tanks or their leach fields. Ammonia is also a manufacturing byproduct that is permitted as end-of-pipe discharges, but inadequate design or improper operation can result in effluent discharge violations. Fertilizer in runoff from golf courses, recreational fields, residences, cropland, and livestock operations, transports ammonia to surface waters, as does land application of manure and grazing livestock, which spread urine and manure on pastures and even directly into streams if they have access (Constable et al. 2003, Camargo and Alonso 2006). Improperly managed aquaculture systems can release high levels of ammonia. Atmospheric sources include agricultural practices and nitrogen oxide emissions from automobiles and industry (NOAA 2000). Elevated ammonia contributes to depressed oxygen levels when oxidizing microbes convert ammonia into nitrite and nitrate. The resulting dissolved oxygen reductions can decrease species diversity and even cause fish kills (Constable et al. 2003).

Total ammonia nitrogen in water includes both the ionized form (ammonium) and the un-ionized form (ammonia). The two species exist in water in dynamic equilibrium. It is the un-ionized form of ammonia that is highly toxic: damaging gill tissue and disrupting ion balance and blood pH (Thurston and Russo 1981, Ip et al. 2001). The ratio of un-ionized ammonia to ammonium ion depends upon both pH and temperature, generally increases by 10-fold for each rise of a single pH unit and by approximately 2-fold for each 10°C rise in temperature over the 0-30°C range (Erickson 1985). The toxicity of ammonia is best expressed as total ammonia nitrogen as a function of pH and temperature. This has been the basis for calculating criteria since 1999 (USEPA 1999). The 2013 recommended criteria for total ammonia nitrogen incorporates data for several previously untested sensitive freshwater mussel species in the Family Unionidae. The criteria are a set of calculations applicable for waters where species of the genus *Oncorhynchus* occur and waters where they are absent. The CMC is expressed as a one-hour average not to be exceeded more than once in three years on average and the CCC is not to be to exceed 2.5 times CCC as a 4-day average within the 30-days, more than once in three years on average.

The BE evaluated the protectiveness of the ammonia criteria using the adjustment factor approach described in Section 2.1.2.1 using a four-day total ammonia nitrogen LC50 for fingerling shortnose sturgeon reported by Fontenot et al. (1998) at 149.86 +/- 55.20 mg/L at 17.9 +/- 0.62 ºC and a pH between 6.8 and 7.3. While there are no toxicity data available for ammonia effects on Atlantic sturgeon, shortnose sturgeon serves as a genus-level surrogate. The Ammonia Guideline document normalized this LC50 to 156.7 mg/L under standard conditions of 20 ºC and a pH of 7 (USEPA 2013). The BE calculated a vertebrate TAF used to arrive at a quantitative low effect acute threshold, LC5, 109.9 mg/L total ammonia nitrogen under standard conditions.

The Ammonia Guideline derived the total ammonia nitrogen CCC using a species sensitivity distribution of EC20s that included data for early-life-stage rainbow trout, a species considered a suitable surrogate for ESA-listed sturgeon in the absence of more closely related species (see section 2.1.1). The Ammonia Guideline also recommended acute to chronic ratios for predicting chronic sensitivity of untested species. The BE applied the more conservative vertebrate acute to chronic ratio of 8.973 in place of the rainbow trout-specific acute to chronic value of 5.945 to arrive at an EC20 estimate of 17.46. The BE then calculated a MAF of 1.412 and used that to convert the EC20 estimate to an EC05 of 12.37 mg/L total ammonia nitrogen.

8.1.3.1 Exposure to Total Ammonia Nitrogen in the Action Area

There are 43 permitted dischargers with discharge limits for ammonia within catchments adjacent to waters where shortnose and Atlantic sturgeon occur (Figure 9). Two dischargers have had effluent exceedances over their current total ammonia nitrogen limits in the past three years, one of which was subject to a formal compliance action. Ten of the facilities have failed to submit their discharge monitoring reports, which can mask serious deficiencies. Impairments in the receiving waters for these facilities include ammonia, as well as impairments ammonia could contribute to: algal growth and impaired biological communities. However, among 39 monitoring stations within these catchments, 229 total ammonia nitrogen observations ranged from non-detect to 1.4 mg/L and all observations were within calculated criteria limits. The dataset contains measures for unpaired ionized and un-ionized ammonia as well. Taken together, shortnose and Atlantic sturgeon will likely be exposed to ammonia in waters affected by implementation of the ammonia criteria.

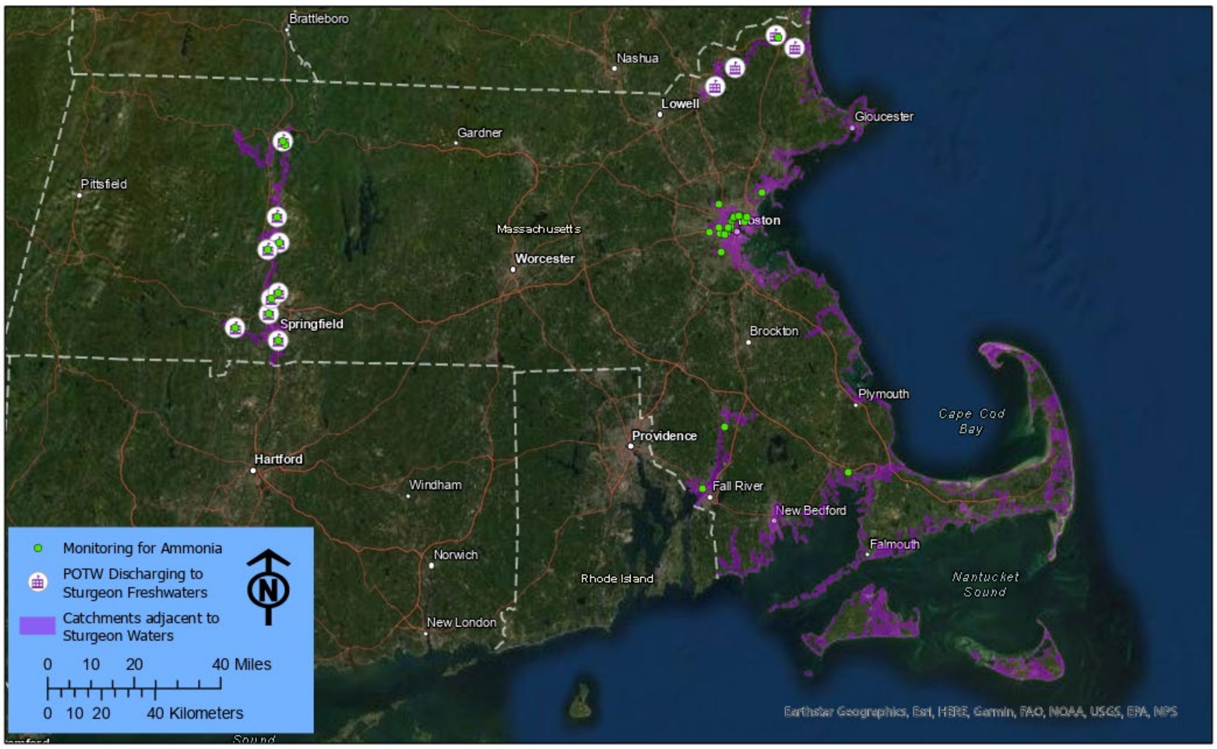

Figure 9. Locations of facilities monitoring for ammonia in their discharges and the locations of public-owned treatment works (POTW).

8.1.3.2 Responses to Total Ammonia Nitrogen Within Criteria Limits

The NMFS-screened ECOTOX dataset included 1,064 entries for 48 fish species and 32 invertebrate families for which pH and temperature data were reported, allowing test specific criteria to be calculated (Figure 10). The fish data included responses for survival, behavior, growth, and development, but no data classified as a reproduction endpoint. However, the CCC was derived using fathead minnow hatchability data tagged as an LC50 (Thurston et al. 1983) and, although not found in ECOTOX, there are EC50s from a study by the same authors for a five-year life cycle for rainbow trout (Thurston et al. 1984b).

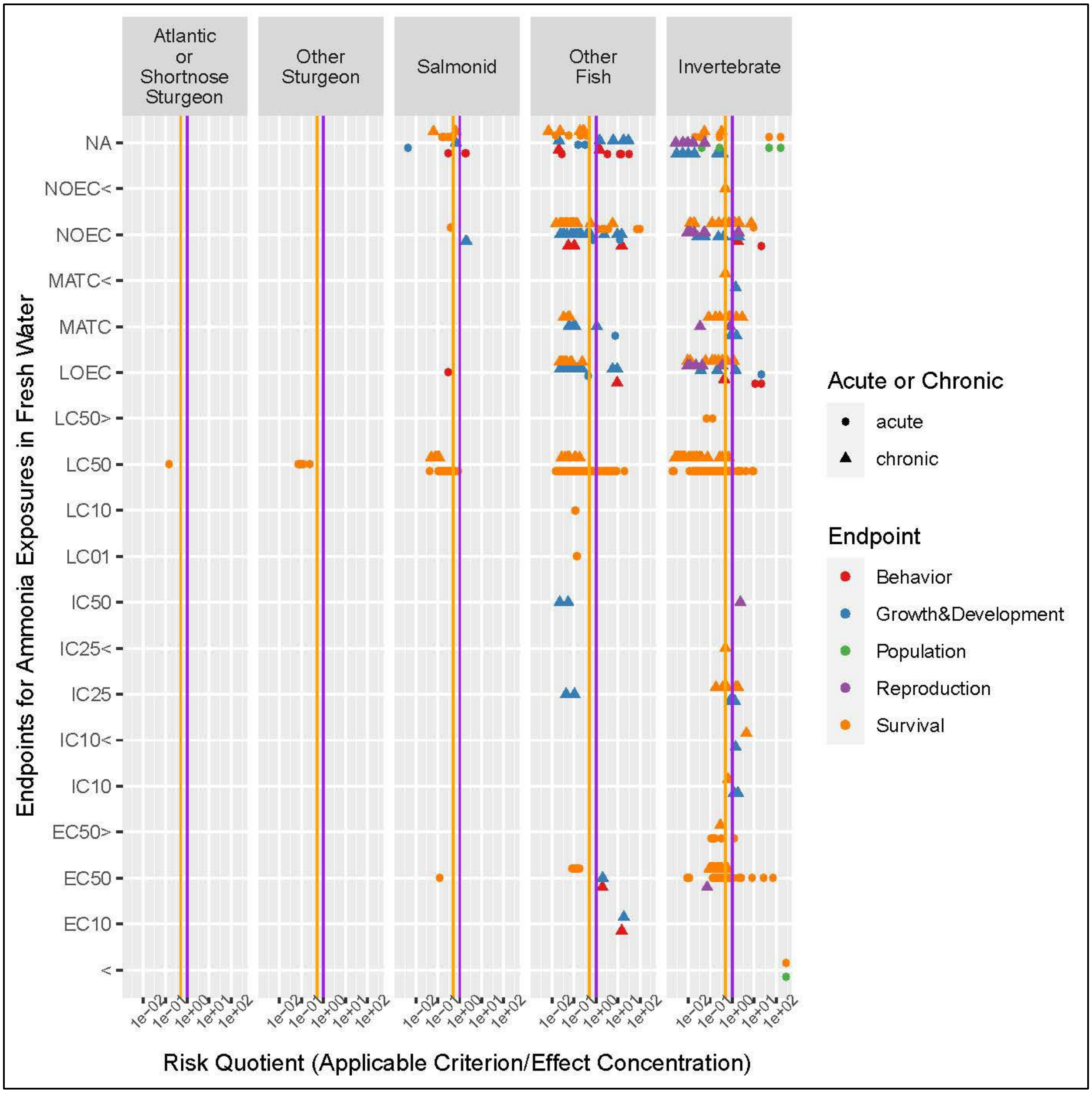

Figure 10. Distribution of risk quotients for freshwater exposures to ammonia in context of reference lines representing the applicable criterion (purple) and one-half the applicable criterion (orange).

The availability of sturgeon data and their application in the BE simplifies NMFS' evaluation. The BE analysis demonstrated that the total ammonia nitrogen CMC of 17 mg/L and the CCC of 1.9 mg/L (under standard conditions) are below the LC05 of 109.9 mg/L and EC05 of 12.37 mg/L estimated in the BE for ESA-listed sturgeon. Yet the analysis in the BE does not take into account the sizable variance around that reported mean. The confidence interval around the reported LC50 of 149.86 +/- 55.20 mg/L indicates a coefficient of variation (mean/standard deviation) of about 37%. This suggests EPA's normalized value would fall somewhere between

99 and 214 mg/L total ammonia nitrogen. Under standard conditions, the total ammonia nitrogen CMC of 17 mg/L is still nearly six-fold lower than the estimated LC50 for shortnose sturgeon fingerlings. Using the coefficient of variation in the original data indicates an LC05 range of about 69 to 150 mg/L total ammonia nitrogen, which is four to eight-fold higher than the CMC of 17 mg/L. If we assume the coefficient of variation for EPAs EC05 threshold estimate mirrors variation among the original data, the EC05 range would be 7.8 to 16.9 mg/L total ammonia nitrogen, which is about four to nine-fold higher than the CCC of 1.9 mg/L.

About 17% of the data indicate adverse effects to invertebrate species. The plotted risk quotients for the effects of ammonia on invertebrates include growth and development, reproduction, behavior, population productivity, and mortality responses. The bulk of the invertebrate data indicate responses occurring above criterion limits. Risk quotients with a reported endpoint (n=34) indicated effects occurring for exposures within criteria limits in species likely to serve as forage for early life stage fish: mayflies, amphipods, rotifers, and *Daphnia* (Kaniewska-Prus 1982, Snell and Persoone 1989, Ankley et al. 1995, Whiteman et al. 1996, Hickey et al. 1999, Khangarot and Das 2009, Liang et al. 2018). There were also risk quotients (n=60) indicating effects would not occur within criterion limits for these same species groups (Buikema et al. 1974, Mount 1982, Reinbold and Pescitelli 1982, Thurston et al. 1984a, Cowgill and Milazzo 1991, Borgmann 1994, Ankley et al. 1995, Whiteman et al. 1996, McDonald et al. 1997, Besser et al. 1998, Hyne and Everett 1998, Hickey et al. 1999, De Rosemond and Liber 2004, Diamond et al. 2006). Given the greater abundance of data indicating effects to prey species would not result from exposures within criteria limits, the criteria are likely to be sufficiently protective of the quantity and quality of prey for sturgeon.

8.1.3.3 Not likely to Adversely Affect Determination for Total Ammonia Nitrogen Exposures Within Criteria Limits

The best available data indicate that it is reasonably certain that shortnose sturgeon and Atlantic sturgeon will be exposed to waters subject to implementation of the ammonia criteria and that the state will use the criteria in the regulation of discharges and identification and restoration of impaired waters. Given current data, effluent exceedances of permit limits and failures to submit discharge monitoring reports will likely continue to occur. Although there are permitted dischargers with ammonia limits within catchments adjacent to nutrient-impaired waters where ESA-listed sturgeon occur (e.g., Mystic River), monitoring data do not provide evidence for elevated total ammonia nitrogen.

The best available data indicate the CMC and CCC for total ammonia nitrogen are expected to be between four-fold and an order of magnitude lower than effect thresholds for ESA-listed sturgeon. While adverse effects may occur in some invertebrate species in Sturgeon Waters, the implications of any effects on the abundance and quality of forage species for shortnose and Atlantic sturgeon will be attenuated by the wide variety of forage species sturgeon consume. A reduction in the abundance of one benthic species is likely to be compensated for by an increase in other species (Wesolek et al. 2010). Therefore, NMFS does not expect that ammonia exposures within CCC or CMC limits will reduce the abundance or quality of forage for shortnose sturgeon and the Gulf of Maine and New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon.

NMFS concludes that the EPA's approval of MassDEP adoption of the National Recommended Water Quality Criteria for total ammonia nitrogen criteria may affect, but is not likely to adversely affect shortnose sturgeon or Gulf of Maine and New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon because the effects of exposures to ammonia within criterion limits are expected to be insignificant for both sturgeon and the abundance and quality of food.

8.1.4 Carbaryl Criteria for the Protection of Aquatic Life in Freshwater

Carbaryl is a carbamate insecticide that acts by inhibiting breakdown of the neurotransmitter acetylcholine by the enzyme acetylcholinesterase. As a consequence, acetylcholine accumulates at nerve synapses causing repeated nerve pulses that result in uncontrolled movement, paralysis, convulsions, tetany, and possible death (Gunasekara et al. 2008). In aquatic systems, the half-life for microbially-mediated degradation (metabolism) of carbaryl is 4.9 days. A half-life of one day was reported for hydrolysis in filtered saltwater (Armbrust and Crosby 1991). Aqueous photolysis of carbaryl may occur in the upper water column of an aquatic system clear enough to allow light penetration. The half-life from aquatic photolysis of carbaryl is reported at 1.8 days (USEPA 2010b). Both the acute and chronic carbaryl freshwater guideline is 2.1 µg/L. The acute saltwater guideline is 1.6 µg/L, but there is no saltwater CCC due to insufficient data. The National Recommended Water Quality Criteria from carbaryl are applied as the CMC limit for one-hour average exposures and as the CCC for four-day average exposures, each not to be exceeded on average more than once in three years.

EPA's BE applied a simplified analysis using the LC50/2 approach to assess the freshwater and saltwater CMC against toxicity data for shortnose sturgeon (for freshwater Dwyer et al. 2000, Dwyer et al. 2005), threespine stickleback, and sheepshead minnow (saltwater Katz 1961, Korn and Earnest 1974). The freshwater invertebrate LC50s used to assess the saltwater CMC were for several species of stonefly and the paper pondshell mussel (Mayer and Ellersieck 1986, Johnson et al. 1993). The freshwater CCC was assessed using data for Colorado pikeminnow, bonytail chub, (Beyers et al. 1994), and fathead minnow (Carlson 1972, Norberg-King 1989).

8.1.4.1 Exposure to Carbaryl in the Action Area

There are many carbaryl-containing products registered for domestic, landscaping and agricultural use in both Massachusetts and New Hampshire.[11] The BE reported that a search for carbaryl in United States Geological Survey National Water Information System returned surface

[11] https://www.mass.gov/pesticide-product-registration and chrome-extension://efaidnbmnnnibpcajpcglclefindmkaj/https://www.agriculture.nh.gov/publications-forms/documents/registered-pesticide-products.pdf accessed June 27, 2022

water and ground water sampling reports for carbaryl at 74 stations in Massachusetts and 22 stations in New Hampshire. Carbaryl was reported at or below analytical limits ranging from 0.2-0.005 µg/L, depending on the analytical method. While there were no data for carbaryl in saltwater, concentrations are expected to be low, given data for inland surface waters. The Water Quality Portal included carbaryl data for 82 freshwater monitoring stations in Massachusetts and New Hampshire, but the most recent data are from 2016 and only three of the stations are in waters where shortnose and/or Atlantic sturgeon occur. The stations are located in the Connecticut River at Montague City, in the Deerfield River near West Deerfield, and in the Westfield River near Westfield Massachusetts. Samples from these stations were analyzed for carbaryl in 1994, but the pesticide was not detected using a method with a detection limit of 0.0056.[12] There are no facilities with permitted discharges within catchments adjacent to waters where shortnose and/or Atlantic sturgeon occur that are required to monitor for carbaryl.

8.1.4.2 Responses to Carbaryl Exposures Within Criteria Limits

The screened carbaryl data from ECOTOX included 966 records from 74 sources exposing 73 species of fish. Data for invertebrates, representing forage species, were provided by 133 studies that conducted 783 toxicity tests evaluating the effects of carbaryl on 79 invertebrate species. Figure 11 illustrates the distribution of risk quotients for freshwater exposures and Figure 12 illustrates risk quotients for saltwater exposures.

While there are only mortality data for Atlantic sturgeon, other species of sturgeon, and salmonids, the location of the risk quotient points for these species is well to the left of the reference lines on the plot, indicating that the LC50s for these species are orders of magnitude higher than both the CMC (circles) and CCC (triangles). Data available for other fish species include effects on reproduction and growth and responses not typically applied to criteria derivation: behavior, acetylcholinesterase inhibition and combined responses (e.g., mortality and morbidity, growth and development). The lowest reported effect threshold among all freshwater fish exposures is an acute MATC of 43.65 µg/L (risk quotient of 0.048) for acetylcholinesterase inhibition in hawk fish larva (Verma et al. 1984). Growth and reproduction risk quotients also indicate responses occurring at concentrations that are one or more orders of magnitude higher than the carbaryl CCC. The reproductive effects risk quotients are obscured by other data on Figure 11. These data are zebra danio hatch LOEC concentrations at concentrations two orders of magnitude higher than the CCC (Mensah et al. 2012). The actual effect threshold for this response in zebra danio is likely lower. The study objective was to assess gonad development, not identify a threshold for carbaryl's effects on hatching, so this study only exposed fish to one concentration of carbaryl. The other available reproductive risk quotient was a 1972 long-term study of fathead minnow reporting adverse effects on survival and spawning, mortality of larvae 30 days of hatching, impaired egg development and release (Carlson 1972).

[12] National Water Quality Monitoring Council Water Quality Portal (https://www.waterqualitydata.us/) 19 observations, accessed June 27, 2022

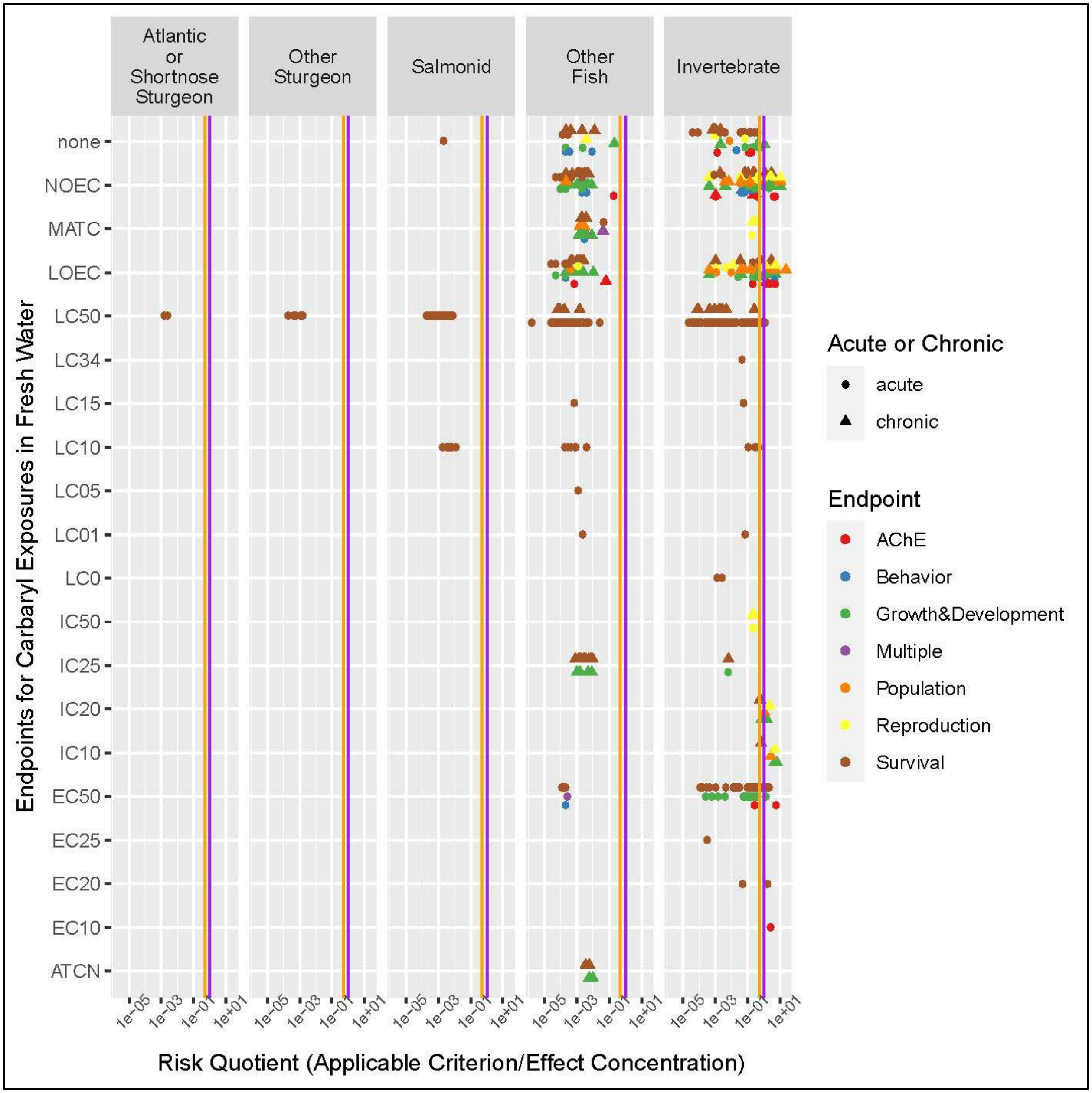

Figure 11. Distribution of risk quotients for freshwater exposures to carbaryl in context of reference lines representing the applicable criterion (purple) and one-half the applicable criterion (orange).

While the bulk of the data for invertebrates indicate endpoint effect concentrations that are well above the criteria, 12% of the endpoint effect concentrations are NOECs, which may underestimate effects. About 10% are LOECS indicating effects may occur at concentrations within an order of magnitude below the criterion concentration. Important responses among the invertebrate data include IC10 and IC20 reproduction, growth, and development inhibition at concentrations six fold lower than the CCC. These risk quotients represent effects on the number of progeny counts, growth rate, lifespan, and maturation of *Daphnia magna* (Toumi et al. 2016).

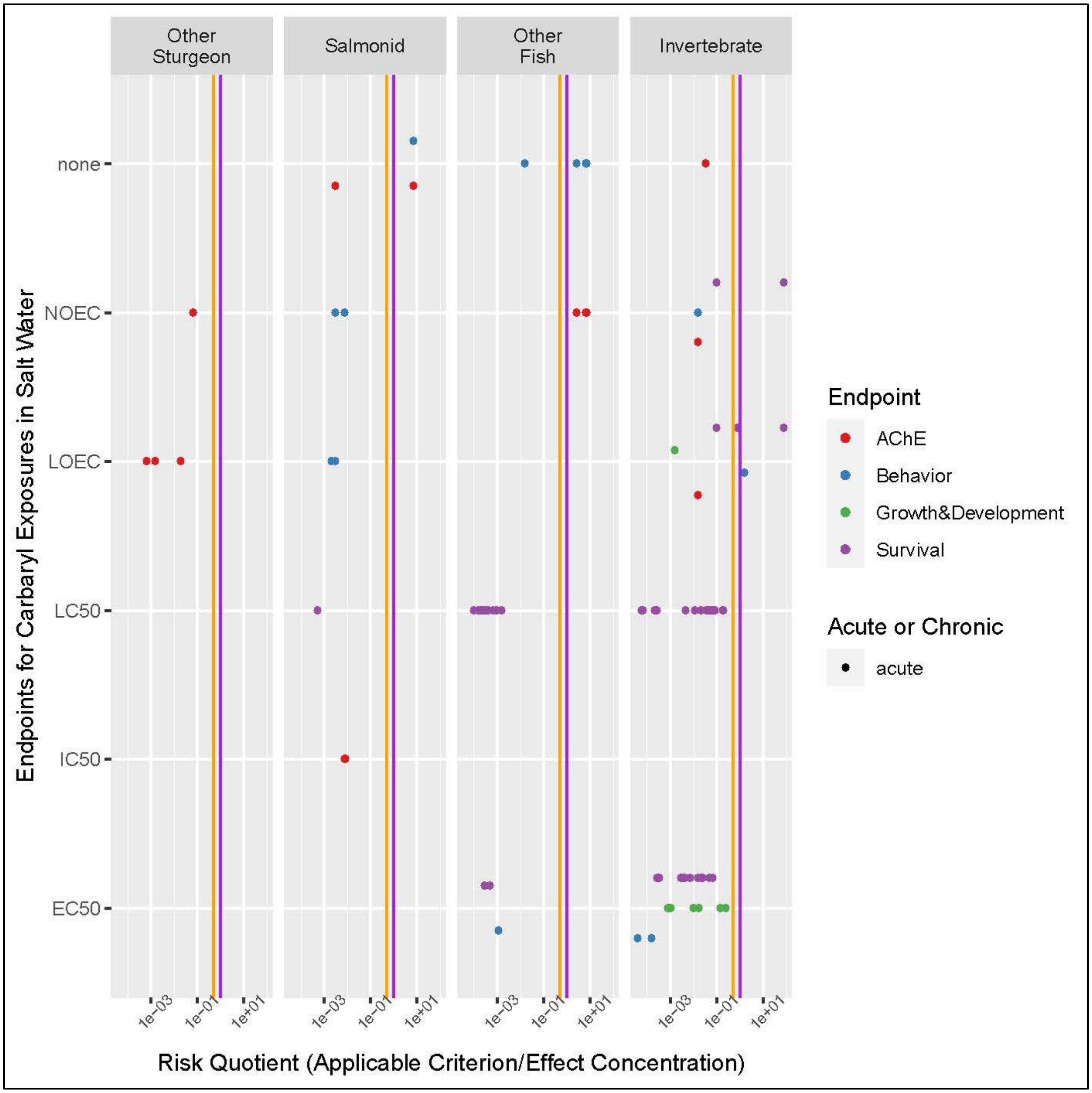

Figure 12. Distribution of risk quotients for saltwater exposures to carbaryl in context of reference lines representing the applicable criterion (purple) and one-half the applicable criterion (orange).

Although sparse and lacking data for chronic exposures and reproductive effects, data for saltwater exposures to carbaryl exhibit a similar pattern (Figure 12), with invertebrates being more likely to experience toxic effects at concentrations below the saltwater criterion. Salmonid and “Other Fish” data identifying h NOECs and effects on feeding behavior at concentrations below the CCC are data collected from the same targeted field study and are based on post application concentrations (Troiano et al. 2013). With respect to reported effects on feeding behavior, stomach contents of shiner perch varied among locations but not exposure, and all Chinook salmon had similar amounts of food in their stomachs in all locations and sampling times. These data do not suggest an adverse effect on feeding behavior attributable to the carbaryl exposure.

8.1.4.3 Not Likely to Adversely Affect Determination for Carbaryl Exposures Within Criteria Limits

NMFS concludes that EPA's approval of MassDEP and NHDES adoption and implementation of the recommended National Recommended Water Quality Criteria for carbaryl is not likely to adversely affect shortnose sturgeon or the Gulf of Maine and New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon because:

1) the freshwater CMC and CCC and the saltwater CMC are orders of magnitude lower than the lowest endpoint indicating adverse effect thresholds for freshwater and saltwater fish, so responses to exposure are expected to be insignificant; and

2) Sturgeon consume a wide range of invertebrate taxa and the criteria were derived to protect aquatic life based on the fifth percentile of sensitive genera and implemented under conservative exposure durations and frequencies (i.e., the CMC is a one hour average derived from four day tests and the CCC is a four day average). Therefore, NMFS expects that the effects of exposures of forage species to carbaryl within criteria limits on the abundance and quality of prey available to ESA-listed sturgeon will be insignificant.

8.1.5 Nonylphenol Criteria for the Protection of Aquatic Life

Nonylphenol (4-nonylphenol) is used in the manufacture of, and is a degradation product of, nonylphenol ethoxylate surfactants that were once commonly used in household products like laundry detergents. EPA and the detergent manufacturers have cooperated to eliminate this use. In addition, nonylphenol ethoxylate use was voluntarily phased out in 2013 from liquid industrial laundry detergents and in 2014 from industrial powder detergents. Discharges of 4-nonylphenol from publically owned treatment works are therefore not expected. Other uses of nonylphenol ethoxylate surfactants, such as dust-control agents and deicers, lead to direct release to the environment. Though less toxic and persistent than 4-nonylphenol, nonylphenol ethoxylates are still highly toxic to aquatic organisms (USEPA 2017).

In the environment, 4-nonylphenol is persistent and accumulates in sediment to concentrations several orders of magnitude greater than concentrations in water. Bottom-feeding fish can be significantly exposed to these persistent and toxic compounds (Brooke 1993b, USEPA 2010a). Half-life in water and sediment is determined by ambient conditions. Nonylphenol accumulates in sediment. Half-lives have been reported to range from 1.1 to 99 days in sediment (Reviewed by Mao et al. 2012) and from 28 to 104 days (Maguire 1999) both reports indicated that persistence was reduced by increased light intensity and the presence of microorganisms (Reviewed by Mao et al. 2012). Concentrations within saltwater sediment cores aged to 30 years using 210Pb dating and plotted as a fraction of the surface concentration showed limited degradation that was directly proportional to nonylphenol concentrations at a rate indicating a half-life of 60 years (Shang et al. 1999).

Accumulation rates vary, depending on exposure duration, concentration, species, and lipid content (Hecht 2002, Hu et al. 2005). Dietary exposures result in accumulation of 4-nonylphenol, but trophodynamic studies indicate that 4-nonylphenol is metabolized and does not biomagnify (i.e., increase in concentration from prey to predator) in the food web (Hu et al. 2005, Diehl et al. 2012, Korsman et al. 2015). The EPA's 2005 water quality criteria document reported bioconcentration factors ranging from 4.7 to 344 (Ward and Boeri 1991, Brooke 1994, after EPA 2005) in freshwater and 78.5 to 2,168 in salt water (Ekelund et al. 1990). Accumulated 4-nonylphenol may be transferred to offspring (Thibaut et al. 2002) with concentrations in eggs increased over maternal levels 30-100-fold (Ishibashi et al. 2006). Persistence and global distribution is indicated by the presence of 4-nonylphenol in organisms living among saltwater debris. Plastic debris contains 4-nonylphenol, but also absorbs 4-nonylphenol from ambient water. The presence of debris can result in enhanced exposures through the creation of 4-nonylphenol-concentrated microhabitats (e.g., poorly flushed areas, relatively sheltered areas of reefs and rocky substrates) or incidental ingestion (Ishibashi et al. 2006, Gassel et al. 2013, Guerranti et al. 2014, Hamlin et al. 2015, Staniszewska et al. 2016). While the proposed criteria are intended to limit exposure of aquatic organisms to harmful levels of 4-nonylphenol, the dynamic flux between ambient water, sediment, and debris may result in unregulable fluctuating microhabitat exposures to concentrations above the proposed criteria in otherwise 4-nonylphenol-compliant waters.

Toxicity tests show that 4-nonylphenol disrupts endocrine systems by mimicking the female hormone 17β-estradiol. Exposure of aquatic animals resulted in abnormal gonad development, changes in reproductive behavior, altered sex ratio of offspring, and the production of yolk proteins (vitellogenin) by immature male fish. Vitellogenin induction in fish by 4-nonylphenol at ambient fresh and salt water occurred concentrations ranging from 5-100 µg/L (Hemmer et al. 2002, Zhang et al. 2005, Ishibashi et al. 2006, Arukwe and Roe 2008) and resulted in altered sex ratios after dietary exposures as low as 1 milligrams per kilogram feed (Demska-Zakes and Zakes 2006). Vitellogenin is an egg yolk protein produced by mature females in response to 17-β estradiol.

Vitellogenin is a robust biomarker of 4-nonylphenol exposure potentially affecting fitness, but without concurrent indicators of exposure and response magnitudes for fitness, a linkage between the intensity of the response and consequences to the survival and fecundity of individuals is not estimable. Ishibashi et al. (2006) reported vitellogenin induction and reduced egg production and fertility after exposure of medaka to 100 micrograms 4-nonylphenol per liter for 21 days. Tilapia gonad development and, sperm abnormalities, and intersex (the presence of oocytes in the testes) after two months of exposure to the same concentration (Ali et al. 2014). A retrospective analysis of an Atlantic salmon population crash implicated 4-nonylphenol, applied as an adjuvant in a series of pesticide applications in Canada as the causal agent (Fairchild et al.

1999, Brown et al. 2003). Additionally, processes involved in sea water adaptation of salmonid smolts are impaired by 4-nonylphenol (Madsen et al. 2004, Jardine et al. 2005, Luo et al. 2005, McCormick et al. 2005, Lerner et al. 2007a, Lerner et al. 2007b). While these data are not for vertebrate species that are present in Massachusetts or New Hampshire, they establish 4-nonylphenol as a persistent pollutant with endocrine disrupting properties, providing a plausible mechanism for fitness effects and survival in the wild, while providing a broad sense of its potency in causing such effects.

The nonylphenol criteria proposed for adoption by MassDEP and NHDES are straightforward pollutant concentrations of 28 and 6.6 µg/L for acute and chronic freshwater exposures, respectively, and 7 and 1.7 µg/L for acute and chronic saltwater exposures, respectively. The CMC duration for nonylphenol is a one-hour average, and the CCC is a four-day average. The frequency of these values is not to be exceeded more than once in three years on average. The BE's assessment of the protectiveness of these criteria used both data from the National Recommended Water Quality Criteria document for nonylphenol and data from ECOTOX. Their screen of the ECOTOX data excluded records for tests that were not within the Guideline limits for deriving criteria. The BE applied a simplified analysis using the LC50/2 approach applied to data for shortnose sturgeon (Dwyer et al. 2005) to assess the freshwater CMC and converted LC50 from the same study to a chronic value using an acute-to-chronic ratio for opossum shrimp.

8.1.5.1 Exposure to Nonylphenol in the Action Area

The Water Quality Portal[13] reports monitoring data for nonylphenol in surface water for 11 stations in Massachusetts. None of these stations are in waters where ESA-listed Atlantic or shortnose sturgeon occur. Of 98 sampling events, nonylphenol was detected at five stations sampled between 2003 and 2010. Concentrations reported for ten sampling events were below criterion limits, ranging from one to 3.1 µg/L. Detection/quantitation limits for the remaining 88 sampling events ranged from 1.6 to 5 µg/L, so NMFS expects any monitoring for nonylphenol would be able to detect its presence within the CCC criterion limit.

A search of EPA's ECHO database did not identify any permitted facilities required to monitor for nonylphenol or discharge monitoring reports with data for nonylphenol submitted between 2007 and 2022. Legacy nonylphenol is likely resident in sediment contaminated by discharges from wastewater treatment plants, airports that formerly used deicing fluids containing nonylphenol ethoxylates, and industrial operations that formerly used nonylphenol ethoxylates in manufacturing processes. While nonylphenol ethoxylates have been phased out from domestic products, they are still in use by some industries. For example, nonylphenol ethoxylates are in some hydraulic fracturing fluids used to extract oil and gas. Treatment and disposal of wastewater from these activities have contaminated surface waters in Western Pennsylvania

[13] National Water Quality Monitoring Council Water Quality Portal (https://www.waterqualitydata.us/) Accessed May 3 through July 5, 2022

(Burgos et al. 2017). Oil and gas extraction activities do not occur in Massachusetts or New Hampshire.

On September 25, 2014, EPA proposed a Significant New Use Rule to require Agency review before a manufacturer starts or resumes use of 15 nonylphenols and nonylphenol ethoxylates (79 FR 59186). This rule provides EPA the opportunity to review and evaluate any intended new or resumed uses of these chemicals and, if necessary, take action to limit those uses. On June 7, 2018, EPA finalized a different rule to include nonylphenol ethoxylates on the Toxics Release Inventory list of reportable chemicals (81 FR 80624). In the rulemaking, EPA estimated that 178 facilities would be expected to submit reporting forms for nonylphenol ethoxylates. Nevertheless, a search of the two most recent years of available Toxics Release Inventory data, 2019 and 2020, did not identify any discharges of nonylphenol ethoxylates (ECHO, accessed June 16, 2022).

8.1.5.2 Responses to Nonylphenol Exposures Within Criteria Limits

The screened nonylphenol data from ECOTOX included 619 records from 54 sources exposing 33 species of fish. Data for invertebrates, representing forage species, were provided by 41 studies that conducted 666 toxicity tests evaluating the effects of nonylphenol on 48 invertebrate species. Risk quotients for all available endpoint effect data are aggregated in Figure 13 for freshwater exposures and Figure 14 for saltwater exposures.

The availability of data for ESA-listed sturgeon simplifies this analysis. The four freshwater LC50s for shortnose sturgeon and Atlantic sturgeon in the proximity of the reference lines are from two studies from the same research group (Figure 13). Corrected for the precent of carbaryl in the formulation, the shortnose sturgeon LC50 was 68 µg/L (risk quotient 0.4) while the Atlantic sturgeon LC50s were 42.5 µg/L (risk quotient 0.65) and 68 µg/L (Dwyer et al. 2000) to 42.5 µg/L (risk quotient 0.4, Dwyer et al. 2005). Confidence intervals and original exposure-response data were not provided with these estimates. Interpretation of the Atlantic sturgeon results in Dwyer et al. (2005) is complicated by mortality in one replicate of the solvent control, and, if a few sturgeon died in either a control or exposure replicate, the water quickly fouled and most or all of the fish then died in that replicate. The risk quotients for other sturgeon indicate that the LC50s were generally an order of magnitude higher than the CMC concentration (Dwyer et al. 1999, Dwyer et al. 2005) as were quotients for salmonids from the same research group (USEPA 1995, Sappington et al. 2001) and other investigators (Calamari et al. 1979, Ernst et al. 1980, Brooke 1993a, Spehar et al. 2010). The LC50s reported for rainbow trout in other studies ranged from 119 µg/L over four days for fry (Dwyer et al. 1999) to 920 µg/L over two days for embryos (Ernst et al. 1980). It is not surprising for an embryo LC50 to be higher than that of older life stages because the vitelline membrane and chorion of the egg are protective (Finn 2007).

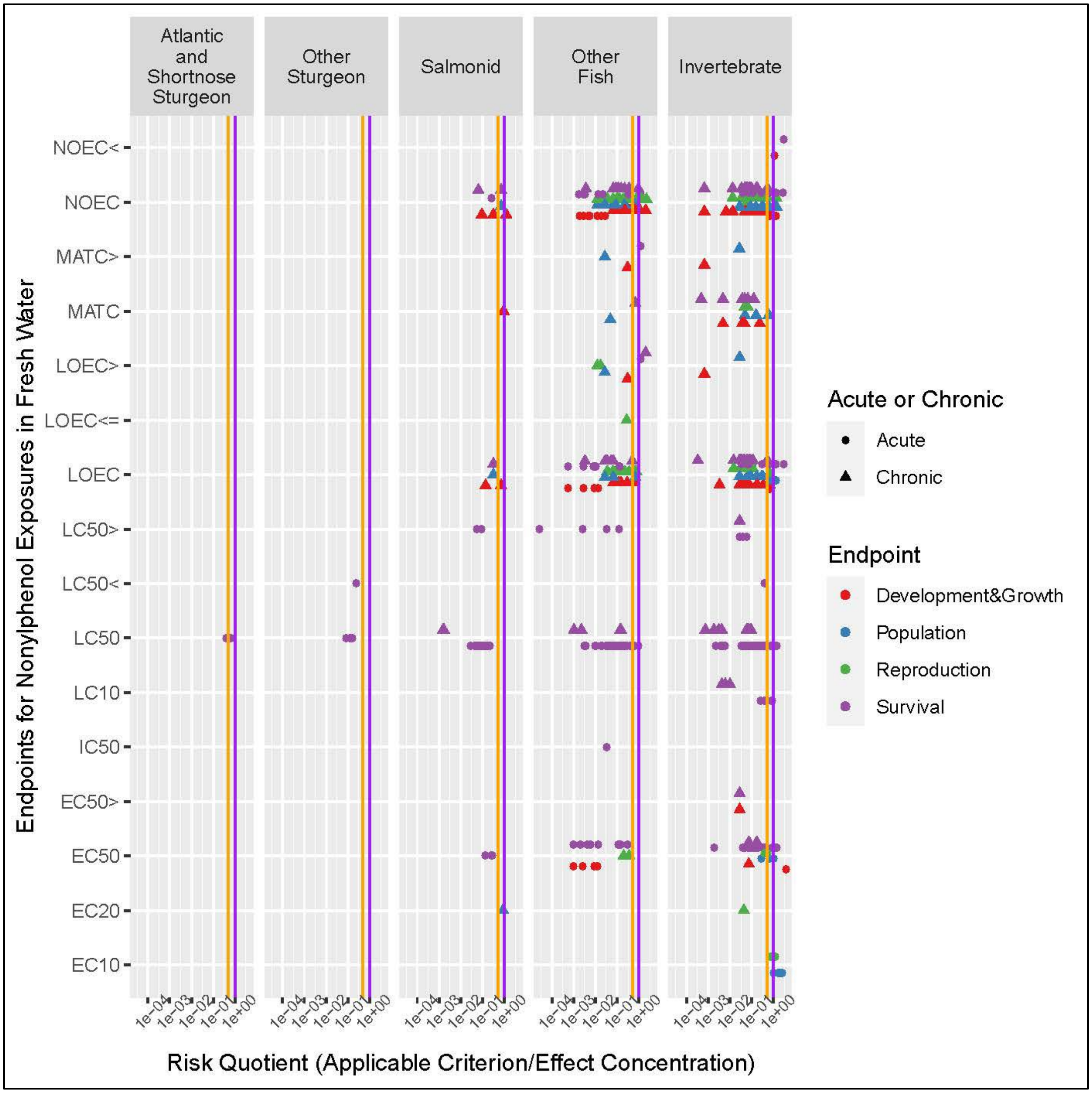

Figure 13. Distribution of risk quotients for freshwater exposures to nonylphenol in context of reference lines representing the applicable criterion (purple) and one-half the applicable criterion (orange).

Among the sublethal data, salmonid development and growth LOEC risk quotients ranged from 0.13 to 0.71 appear to be from two sources. However, the brook trout studies reported in Spehar et al. (2010) appear to be a peer reviewed publication of an earlier Brooke (1993a) government report. Response magnitudes at the LOEC for this work included ~30% reduction in weight (risk quotient 0.71) and 60% reduction in mean percent post-hatch survival (risk quotient 0.32) at 51 days. The study also reported an LC50 at 221 µg/L, and reported an EC50 of 109 µg/L for loss of equilibrium, immobility, and morbidity at 51 days. The LC50 is more than twice LC50s reported for ESA-listed sturgeon in either of the Dwyer et al. studies (2000, 2005) suggesting that growth and development effects would be expected to occur in shortnose and Atlantic sturgeon at lower exposure concentrations than reported in Spehar et al. (2010).

Several of the freshwater invertebrate acute LC50 and LOEC risk quotients indicate effects occurring at and below the CMC. These include data for paper pondshell (Black 2003), scud (Brooke 1993a, Spehar et al. 2010), and *Daphnia magna* (Hong and Li 2007, Campos et al. 2012, Campos et al. 2016). The Campos et al. (2016) was a multigenerational study indicating changed sensitivity over three generations of exposed organisms. The EC10s reported for population growth were 14+/- 2.4 µg/L in the parental generation and 25.5 +/-2.6 µg/L in the third generation but fecundity was 35.7+/- 11.4 µg/L in the parental generation and 27.36+/- 4.9 in the third generation. The freshwater toxicity data did not suggest adverse effects for invertebrate exposures within the nonylphenol CCC.

The available toxicity data for saltwater exposures are sparse but indicate that invertebrates are more sensitive to nonylphenol than fish (Figure 14). The risk quotient adjacent to the 0.5 reference line represents an LC50 for winter flounder larva. This test was part of a study collecting information on nonylphenol in order to form a database of acute toxicity specifically for calculating a national CMC (Lussier et al. 2000). The chronic LOEC is for increased weight in juvenile turbot. At an exposure of 30 mg/L nonylphenol over three weeks, the fish increased significantly in size, but plasma testosterone and beta-estradiol declined (Martin-Skilton et al. 2006). The authors discussed other hormonal and physiological changes, but did not address morphometric effects on plasma hormone levels like changes in blood volume, edema or somatic indices. Data for saltwater invertebrates indicate that adverse effects are expected, but there were no data for adverse effects on reproduction.

The population-level risk quotients for saltwater invertebrates represent approximately 20% inhibition of barnacle larvae settlement at 0.059 +/-0.001 µg/L nonylphenol (Billinghurst et al. 1998) and enhanced intrinsic rate of increase (births minus deaths) for a marine copepod in a study using exposure concentrations ranging from 31 to 500 µg/L (Bechmann 1999). The Billinghurst et al. (1998) study also contributed several of the development and growth risk quotients reflecting delayed maturation at exposure concentrations below the saltwater CMC. Other effects reported to occur at concentrations below the CMC include decreased size and disrupted molting cycles in opossum shrimp (Hirano et al. 2009) and delayed maturation persisting into the next generation of harpacticoid copepods with the parental exposure initiated at the nauplii stage (Marcial et al. 2003).

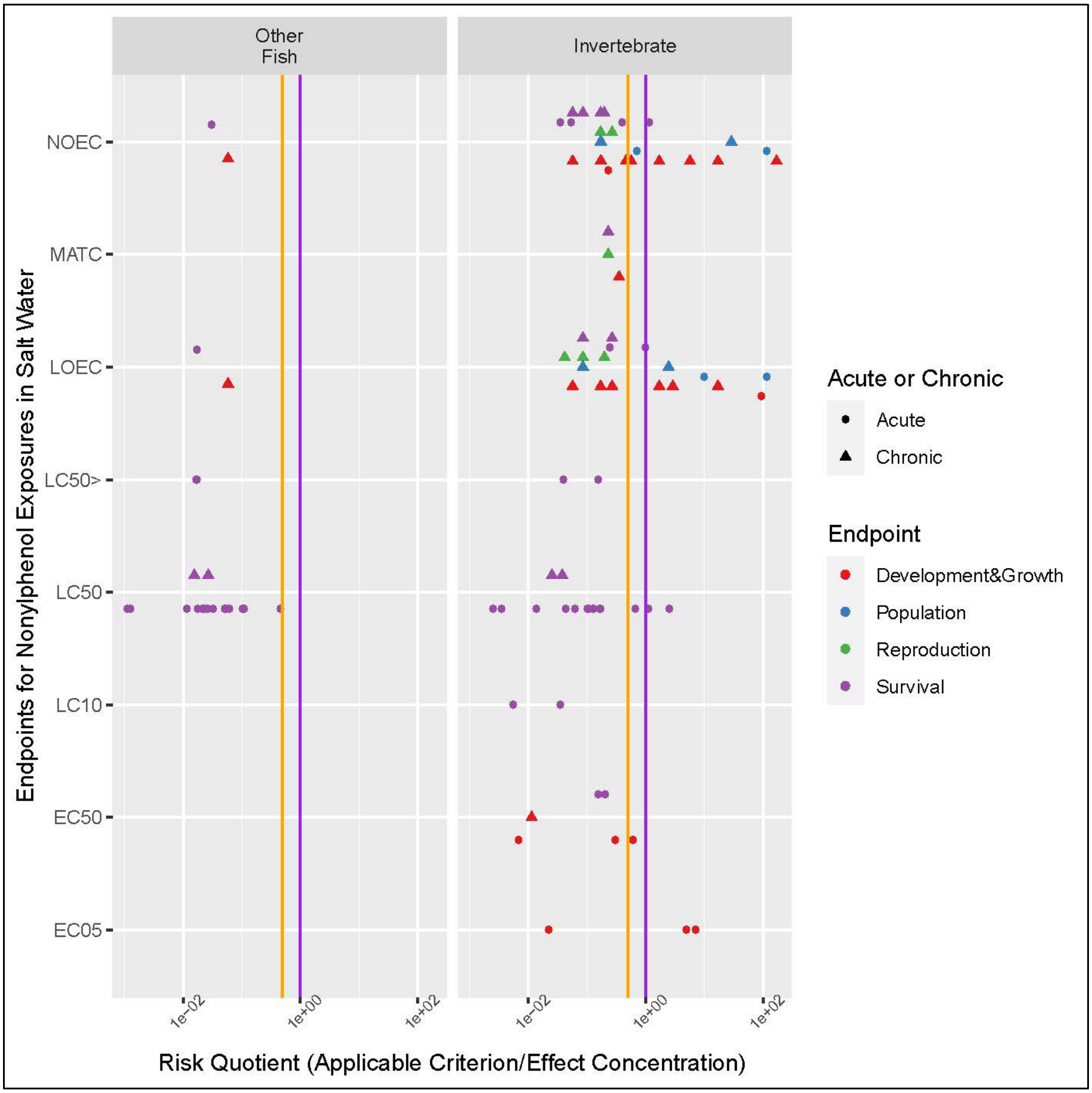

Figure 14. Distribution of risk quotients for saltwater exposures to nonylphenol in context of reference lines representing the applicable criterion (purple) and one-half the applicable criterion (orange).

8.1.5.3 Not Likely to Adversely Affect Determination for Nonylphenol Exposures Within Criteria Limits

NMFS concludes that EPA's approval of MassDEP and NHDES adoption and implementation of the recommended National Recommended Water Quality Criteria for nonylphenol is not likely to adversely affect shortnose sturgeon or the Gulf of Maine and New York Bight, and migrating

Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon, or prey species because exposures to nonylphenol are extremely unlikely to occur and are therefore discountable for the following reasons:

1) domestic and industrial use of nonylphenol has been phased out,
2) there are no regulable sources of nonylphenol within catchments adjacent to waters where ESA-listed sturgeon occur (Sturgeon Waters), and
3) monitoring data provide no indication that legacy contamination is circulating in Sturgeon Waters at this time.

8.2 Criteria that are Likely to Adversely Affect ESA-listed Species Under NMFS' Jurisdiction

The previous section concluded that EPA approval of MassDEP adoption and implementation of site-specific TN criteria and criteria for acrolein, ammonia, carbaryl, and nonylphenol may affect, but are not likely to adversely affect shortnose sturgeon or the Gulf of Maine and New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon. As discussed in Section 2.1.1 of this Opinion, the criteria developed using the EPA Guidelines are not expected to protect all species under all circumstances, so waters compliant with the criteria may result in pollutant exposures that cause adverse effects in some species. When assessing risk to an ESA-listed species, the vulnerability of an imperiled population of that species to the loss of an individual, or key individuals, amplifies the fundamental threat posed by a toxic pollutant. The underlying assumptions in the methods used to arrive at criteria affect how well ESA-listed species and designated critical habitat are protected. Paramount among these are assumptions that:

- Effects that occur on a species exposed to a toxicant in laboratory tests will generally be the same for the same species exposed to that toxicant under field conditions (i.e., effects are not influenced by predation, competition, disease, exposure to other stressors in the field, and fluctuations in natural water quality parameters).
- Collections of single-species laboratory toxicity test data used to derive criteria reflect communities in natural ecosystems.
- Data on severely toxic effects from short-term "acute" toxicity tests used to derive CMC can be extrapolated to less severe effects that would be expected to occur in long-term "chronic" exposures to derive CCC.
- Loss of a small number of species will not affect the propagation of fish, shellfish, and wildlife.
- Loss of a small number of species will not result in incidental loss of any “economically or recreationally valuable species” for which data were not available.
- Sensitive species and life stages are adequately represented such that criteria are not biased.

- Derivation of criterion for a single chemical in isolation without regard to the potential for additive toxicity or other chemical or biological interactions is acceptable despite chemicals typically occurring in mixtures in the environment.
- When applied to NPDES permits, unless the waters are already identified as impaired by a pollutant, the waters are free from that pollutant.
- Accumulation of chemicals in tissues and along the food web does not result in ecologically significant latent toxicity or toxic exposures for predators.

There are also concerns about the underlying data used in the derivation of criteria including:

- Data sets for sublethal responses are usually small and have gaps such that sensitive species and life stages are under-represented.
- Variability within and among species used in calculating a hazardous concentration to 5% of species (i.e., HC5) may be substantial, but this variability is not reflected in the final HC5 estimate used to derive a CMC.

These assumptions are repeated here to underscore the importance of the scale of uncertainty that accompanies lab-to-field extrapolation and the methods used to synthesize data for criteria derivation. Further, aluminum, cadmium, and copper do not exist alone in effluents or natural waters. The toxicity of mixtures is dependent upon many factors, such as which chemicals are most abundant, their concentration ratios, differing factors affecting bioavailability, and organism differences. Because of this complexity, accurate predictions of the combined effects of chemicals in mixtures in every case where the criteria assessed in this Opinion are applied is not current practice. The work of Spehar and Fiandt (1986) showed 100% mortality in rainbow trout and *Ceriodaphnia dubia* exposed to a mixture of six metals at their CMC concentrations suggests severe effects result from exposure to compliant discharges and within "unimpaired" waters.

8.2.1 Aluminum Criteria for the Protection of Aquatic Life in Freshwater

Aluminum is the third most abundant element and the most common metal in the Earth's crust. It is typically found complexed with oxygen (as oxides) and silica (as silicates). It is present in both industrial and nonpoint discharges associated with the manufacturing process and the environmental wear and tear of aluminum-containing objects (e.g., boats, vehicles, trash) and use and disposal of aluminum containing products (e.g., kitchenware, household and personal care products). Non-point sources also include atmospheric deposition, acid mine drainage, forestry, urban stormwater, and agriculture. Point sources relevant to New England include manufacturing, recycling, and drinking water and sewage treatment facilities, where alum (potassium aluminum sulfate) is used to as a coagulant. Dredging and disposal operations can result in substantial suspension and resuspension of particulates in the water column, including those contaminated with aluminum.

Aluminum speciation and solubility is strongly correlated with pH (Cardwell et al. 2018). The toxicity of aluminum appears to be lowest at neutral pH, with toxicity generally increasing with either an increase or decrease in pH. Below a pH value of 5, ionoregulatory effects dominate due to blockage of sodium uptake (Playle and Wood 1989). In moderately acidic water, with pH values less than 6.5, aluminum can accumulate on the gill surface, physically coating the gill surface and reducing gas exchange (Gensemer and Playle 1999). In alkaline conditions (pH > 8), the negatively charged aluminate ion dominates, and although it does not bind to the negatively charged gill surface, it can cause necrosis of the epithelial cells.

Aquatic organisms can accumulate metals from both aqueous and dietary exposure routes. Aluminum adsorbs rapidly to gill surface from the surrounding water, but cellular uptake is slow and accumulation by the internal organs is gradual (Dussault 2001). Total uptake generally depends on the environmental aluminum concentration, exposure route and the duration of exposure (McGeer et al. 2003). Bioaccumulation and toxicity via the diet are considered highly unlikely based on studies by Handy (1993) and Poston (1991), and also supported by the lack of any biomagnification within freshwater invertebrates that are likely to be prey of fish in acidic, aluminum-rich rivers (Otto and Svensson 1983, Wren and Stephenson 1991, Herrmann and Frick 1995). The opposite phenomena, trophic dilution up the food chain, has been suggested based on the lowest aluminum accumulation exhibited by fish predators (perch) and highest by the phytoplankton that their zooplankton prey were consuming (King et al. 1992).

Aluminum sorbs to organic matter, thus aluminum is less bioavailable in waters with higher concentrations of dissolved organic carbon (Wilson 2012). Gensemer and Playle (1999) provide review of studies demonstrating how dissolved organic carbon (DOC) reduces aluminum toxicity. The ameliorating effect of DOC may be more pronounced in higher pH waters, than in low pH where hydrogen ions compete for binding sites (Parkhurst et al. 1990).

Hardness also has an effect on the toxicity of aluminum. Gundersen et al. (1994) demonstrated that increased water hardness (i.e., calcium concentrations) increased the survival of rainbow trout in both short (96-hour) and longer (16-day) exposures. However, at elevated pH conditions (e.g., pH 8) the protectiveness of hardness is reduced (Deforest et al. 2018, Gensemer et al. 2018).

The EPA Aquatic Life Ambient Water Quality Criteria for Aluminum (Aluminum Guideline USEPA 2018) are for total recoverable aluminum based on multiple linear regression modeling of the three major aquatic chemistry determinants of aluminum bioavailability: pH, dissolved organic carbon, and total hardness (USEPA 2018). If aluminum criteria were based on dissolved concentrations, toxicity will be underestimated, because the contribution of aluminum hydroxide precipitates to toxicity would not be measured (USEPA 2018).

The two models, one for vertebrate data and one for invertebrate data, are based on studies characterizing the bioavailability and toxicity of aluminum to fathead minnow or *Ceriodaphnia dubia* in aquatic systems under varying pH, total dissolved organic carbon, and total hardness

(Deforest et al. 2018, OSU 2018a, c, b, Deforest et al. 2020). The models were used to convert LC50 concentrations from toxicity tests reporting pH, dissolved organic carbon, and total hardness to standard conditions: pH of 7, dissolved organic carbon of 1 mg/L, and total hardness of 100 mg/L calcium carbonate. Once standardized, toxicity test results could be used to derive acute and chronic water quality criteria for total recoverable aluminum per the 1985 Guidelines.

A site-specific aluminum criterion is expected to be the aluminum concentration at which the amount of biologically available aluminum is the same as the amount of biologically available aluminum at the criteria concentration under standard conditions. For example, if aluminum was twice as biologically available under site conditions relative to standard conditions, the calculated site-specific criterion should be one-half the criterion concentration under standard conditions.

For discharge permitting, MassDEP is proposing a choice of either calculating site-specific criteria or using default aluminum criteria calculated for individual watersheds. If water quality data for pH, dissolved organic carbon, and total hardness are already available, acute and chronic calculated site-specific criteria supersede the watershed default criteria. The calculation of site-specific criteria requires pH, dissolved organic carbon, and total hardness data for each quarter collected over five years, providing 20 criterion estimates from 20 sets of data. For waters where ESA-listed species occur, the concentration at the fifth percentile among these would be the criterion. This conceptually protects aquatic life from aluminum toxicity at that location 95% of the time.

Existing monitoring data for pH, organic carbon, and hardness collected concurrently (i.e., a sampling event) from sampling stations within a watershed were used to calculate the default CMC and CCC criteria for that specific watershed. For watersheds where ESA-listed species occur, the default watershed criteria are the fifth percentile of criterion concentrations within the watershed. Since monitoring stations are not evenly distributed within a watershed, it is difficult to say whether default watershed criteria protect aquatic life from aluminum toxicity within 95% of the watershed or 95% of the time.

Shortnose and Atlantic sturgeon occur in the Connecticut River, Deerfield River, Merrimack River, Taunton River, and Farmington-Westfield (Westfield) River watersheds. Sampling events used to calculate watershed criteria dated from 1990 to 2017. According to the BE, the sampling events had similar aquatic chemistry variability over time, indicating that default aluminum criteria should be representative of current water quality conditions. The default CMC for the Connecticut, Merrimack-Shawsheen, Taunton, and Westfield watersheds are 600, 460, 300, and 299 µg/L, respectively. The default CCC for the Connecticut, Merrimack-Shawsheen, Westfield, and Taunton watersheds are 290, 249, 190, and 169 µg/L, respectively. While shortnose sturgeon occur in the Deerfield River, the MassDEP technical Methodology for Deriving Watershed Default Criteria did not include fifth percentile criteria for the Deerfield River. The EPA provided NMFS with the database used to generate the watershed-based aluminum criteria for these rivers. The criteria that would be appropriate for the protection of ESA-listed species in the Deerfield River are a CMC of 335 µg/L and CCC of 170 µg/L.

The BE evaluated the protectiveness of these criteria by comparing them to sampling event chemistry-adjusted LC05 concentration estimates for acute exposures and EC05 concentration estimates for chronic exposures for ESA-listed shortnose sturgeon and Atlantic sturgeon. The adjustment factors used to derive the LC05 could not exactly follow EPA's a priori protocol stated in the MA BE Section 1.4 *Overarching Approach and Methodology* because the data meeting the data quality criteria for the derivation of an acute adjustment factor, as described in Section 2.1.2.1, were only available for invertebrate species. Ultimately, EPA applied taxonomic class-level adjustment factors estimated using rainbow trout-to-shortnose sturgeon and rainbow trout-to-Acipenser models from the Interspecies Correlation Estimation program.

Depending on the watershed, between 2.1 and 4.8% of the invertebrate-based LC05 estimates, adjusted for sampling event chemistry, were below the applicable default watershed criterion. That is to say, the default acute watershed criteria were insufficiently protective for a fraction of the sampling events. Further exploration revealed that the aquatic chemistry conditions in a majority of these events had pH values below 6.5, a value that has been suggested as the lower limit of the physiological range for sturgeon (Chebanov et al. 2018). Massachusetts aquatic life pH criteria state that the lower pH limit for all classes of waters is 6.5 and that there shall be no change from natural background conditions (314 CMR 4: The Massachusetts Surface Water Quality Standards).

Meanwhile, 0.16 to 0.51% of the taxonomic class-level based LC05s, adjusted for sampling event chemistry, were below the applicable default watershed criterion. The EPA's not likely to adversely affect determination relied on the taxonomic class-level analysis because shortnose and Atlantic sturgeon are vertebrates, not invertebrates, and the priority for the MA BE methodology is for adjustment factors to optimize taxonomic relatedness.

Since chronic aluminum toxicity data were not available for Atlantic sturgeon or shortnose sturgeon, the BE analysis transformed the taxonomic class-level acute adjustment factors to chronic adjustment factors using a vertebrate-specific acute-to-chronic ratio for aluminum (Kimball 1978). Depending on the watershed, between 3.6% and 5% of the EC05s adjusted for sampling event aquatic chemistry were below their respective default watershed criterion. Again, a majority of these were for sampling events with pH values below 6.5. In Massachusetts, the CMC duration for aluminum is a one-hour average, and the CCC is a four-day average. The frequency of these values is not to be exceeded more than once in three years, on average.

8.2.1.1 Exposure to Aluminum in the Action Area

Current monitoring and permitting data indicate that all life stages of ESA-listed shortnose and Atlantic sturgeon are certain to be exposed to waters where the aluminum criteria will be implemented. Data for stations within rivers occupied by ESA-listed sturgeon from the National Water Quality Monitoring Council Water Quality Portal identify dissolved or total aluminum concentrations in all rivers occupied by shortnose and Atlantic sturgeon. Sixty percent of the available data are for dissolved aluminum and cannot be used to evaluate the criteria, which are calculated for total recoverable aluminum. In natural waters, dissolved metals are a fraction of the total recoverable metal. Metal solubility is influenced by aquatic chemistry measures such as pH, major ions, and suspended solids (Stumm and Morgan 1996). Total recoverable aluminum[14] detections from ten sampling stations within the Merrimack River numbered 11 observations between 1975 and 2021 ranged from 69 to 800 µg/L. Ten observations between 1974 and 1976 from ten sampling stations in the Connecticut River ranged from 100 to 700 µg/L total recoverable aluminum. A single measurement of 100 µg/L was reported for a Deerfield River station in 1969. The CCC for the Deerfield River was calculated at the tenth percentile of 220 µg/L. The CCC at the fifth percentile would be 170 µg/L.

Half of the reported total recoverable aluminum concentrations exceeded the proposed default aluminum watershed chronic criteria of 290 and 249 µg/L for the Connecticut and Merrimack rivers, respectively. The only recent aluminum data are from the Merrimack River. Samples taken at Bates Bridge near Haverhill, Massachusetts in August and September of 2021 exceeded the watershed criterion or 290 µg/L, with concentrations of 328 and 356 µg/L total recoverable aluminum. Samples taken in June, July, and August from Shad Creek within the Merrimack River Estuary were below the criterion with concentrations between 92 and 163 µg/L total recoverable aluminum.

To identify sources of aluminum, NMFS collected information on permits with limits for aluminum discharges to Sturgeon Waters. Among NPDES permits within catchments adjacent to Sturgeon Waters that are required to monitor for aluminum, one has exceeded its current discharge limits for the three out of the past 12 quarters. The permit was issued in 2022 and will not expire until 2027. This facility is discharging to the Chicopee River at its confluence with the Connecticut River. All life stages of shortnose and Atlantic sturgeon are expected to use these waters and the receiving water segment is within designated critical habitat for the New York Bight DPS of Atlantic sturgeon. Recent data indicate that the individuals spawning in the river are actually more closely related to the Carolina and South Atlantic DPSs of Atlantic sturgeon than either the Gulf of Maine or New York Bight DPSs (Savoy et al. 2017).

When this permit is renewed, the discharge limit will actually increase under the revised aluminum criteria due to normalization to aquatic chemistry conditions. Because normalizing for pH, DOC and total hardness is expected to provide a more precise indicator of bioavailability, and therefore toxicity, we would expect the CV for normalized within-species mean LC50s and EC20s would be smaller than the CV for within-species LC50s and EC20s that were not normalized. This provides a more precise estimation of toxicity, but the resulting criterion may be higher or lower than the 1988 aluminum criteria. NMFS tested this expectation using data from the Aluminum Criteria Calculator. More often than not, within species mean LC50s and

[14] Identified in the Water Quality Portal as the total recoverable, recoverable or unfiltered sample fraction.

EC20s normalized for pH, DOC and total hardness had lower CVs. The exceptions for acute toxicity tests were for two datasets including tests conducted under pH conditions that "pushed" the limits of the model and one dataset where all LC50 values were flagged as ">". The pH limit for the model was five and the datasets included tests that were conducted at ~neutral pH and pH values of ~5.5. An LC50> endpoint means the values are not actually LC50s and could represent LC45s, LC48s, etc. Among chronic test data, those for biomass (n=3 species) had higher CVs among normalized toxicity test results. One of the three datasets had tests conducted at pH values of 6.5 and 5.65. Although the criteria may be higher for some permitted discharges under the standard EPA proposes to approve, NMFS views the adjustment to provide a more precise estimate of toxicity. The question at hand is whether that more precise estimate of toxicity is sufficiently protective of ESA-listed species.

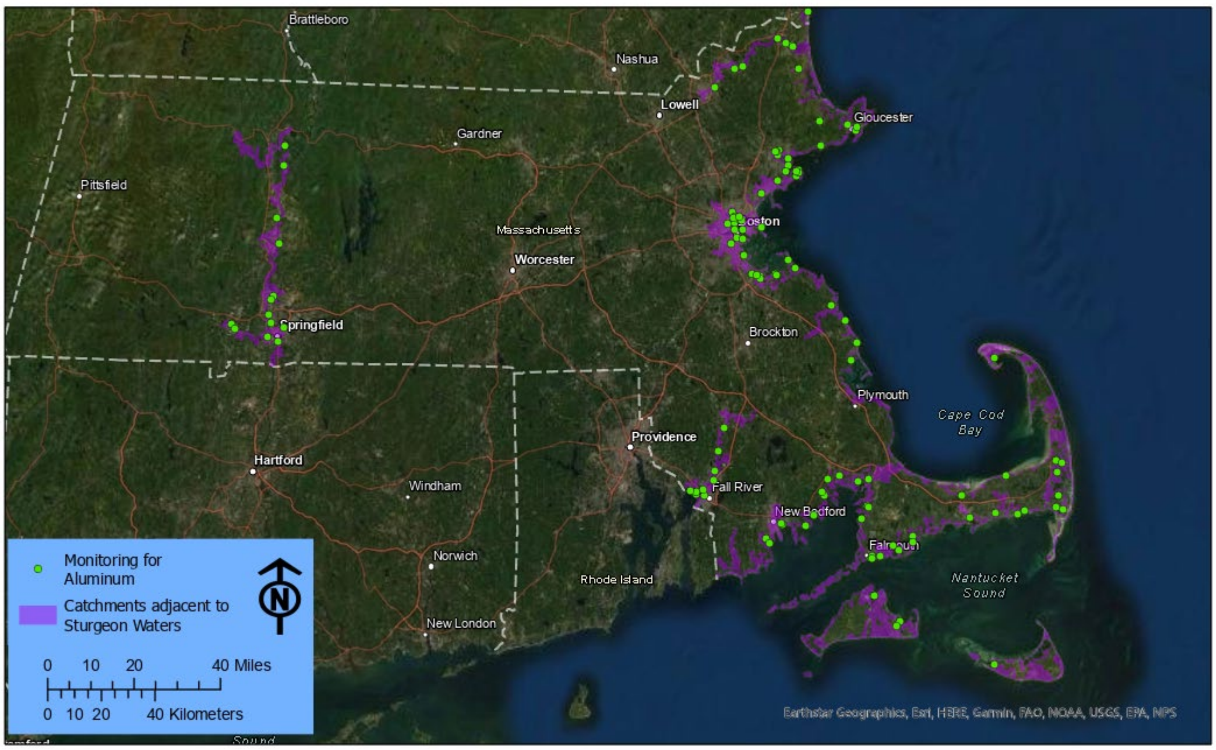

Figure 15. Locations of NPDES Facilities within catchments adjacent to Sturgeon Waters and are required to monitor for aluminum.

At this time, only the Herring River in the Wellfleet region of Cape Cod is identified as impaired by aluminum in the state of Massachusetts. The impairment is attributed to the hydrological modification resulting in release of sulfur from decaying peat into water, lowering the pH such that aluminum is leached from naturally occurring clay substrate (Town of Wellfleet 2019).

8.2.1.2 Responses to Aluminum Exposures Within Criteria Limits

NMFS' screened ECOTOX dataset for aluminum had fewer records that included data for pH, organic carbon, and total hardness than the dataset used by EPA for its BE. The BE relied on data provided in the 2018 Aluminum Guideline (USEPA 2018). In preparing the Aluminum Guideline, EPA obtained the unpublished water chemistry data for toxicity test conditions from the researchers or used values reported by other studies using the same or similar water for the toxicity tests. Where only data for dissolved organic carbon were lacking, default values from several different dilution waters were applied using a methodology documented in the 2007 freshwater Copper Guideline document (USEPA 2007). These values were determined from empirical data obtained for each source water. The data in EPA's document includes observations on behavior that were not used in criteria development, but are important to NMFS' analysis. The unpublished water chemistry data in the Aluminum Guideline are thus the best available data. The dataset for 13 species of fish exposed to aluminum includes responses for behavior, development, growth, and survival. There are also data for 18 species of invertebrates describing aluminum effects on development, growth, population, reproduction, and survival.

EPA used the aquatic chemistry to normalize toxicity test data to standard conditions to allow meta-analysis of SMAVs and GMAVs. NMFS' analysis used EPA's Aluminum Criteria Calculator to obtain toxicity test-specific criteria in order to calculate test-specific risk quotients for the reported endpoint concentrations (e.g., LOEC, NOEC, EC50, LC50 etc.). Risk quotients for all available endpoint effect data are aggregated in Figure 15.

The only acute exposure toxicity data proximate to the orange and purple reference lines are behavioral responses indicating that fish would avoid waters with aluminum within the CMC limits. Gunn and Noakes (1986) reported an EC50 for avoidance behavior by brook trout exposed to a steep aluminum gradient concentration. The test design counted the number of individuals that moved to uncontaminated water within 15 minutes after introduction of aluminum-contaminated water. The sudden, sharp exposure gradient represented by this study would be more similar to a discharge pulse than a mixing zone interface. At a risk quotient of 0.57, this response represents an exposure that is above the test-specific CMC, but we cannot be certain that exposure at the CMC would not result in an EC20 or EC35. The confidence intervals for this EC50 estimate are broad, representing about 25% of the mean. In the control, fish spent 47 +/- 15.7% of the 15-minute observation period in the un-dosed side of the tank. At 100 µg/L fish spent 62.8+/- 15.5% of that 15 minutes in the un-dosed side of the tank, but avoidance behavior was not statistically significant until, at 500 µg,/L fish spent 80.7+/-13.1% of the time in the un-dosed side of the tank.

The rainbow trout behavior LOEC risk quotient of 1.62 is for increased frequency of gill flushing (i.e. "cough") over a 24-hour exposure period (Ogilvie and Stechey 1983). While this response is typically associated with clearance of particulate matter, it is not an unexpected response to aluminum exposure because hydroxide precipitates contribute to toxicity (USEPA 2018). The magnitude of response at the LOEC was twice that of the control and the NOEC. Interpreting the ecological significance of this response is complex. In the wild, this may result in avoidance if there are refugia. In the absence of refugia, an increased cough rate might interfere with feeding, predator avoidance, and be associated other stress responses like mucous production. Relocation to refugia also has implications. Relocation requires energy expenditure and can increase visibility to predators (Nunes et al. 2019). Refugia may be otherwise suboptimal habitat or be occupied by competitors (reviewed by Magoulick and Kobza 2003).

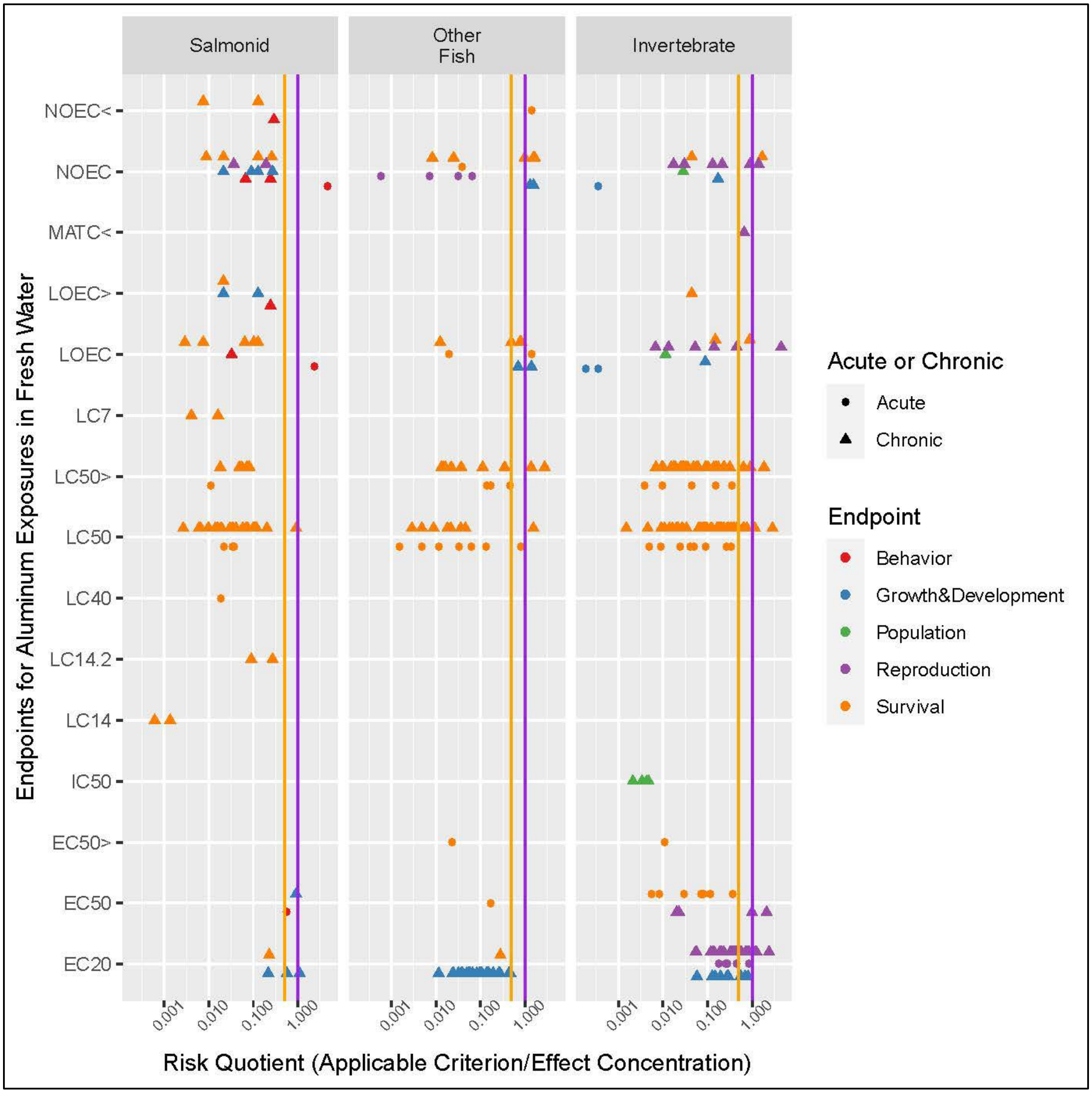

Figure 16. Distribution of risk quotients for freshwater exposures to aluminum in context of reference lines representing the applicable criterion (purple) and one-half the applicable criterion (orange).

Most concerning among the data for chronic exposures are the risk quotients representing a 16-day rainbow trout mortality LC50 and 28-day EC20s for growth and development (Birge 1978, Birge et al. 1978, Birge et al. 2000). These studies were not included in criteria derivation due to the duration of the exposures. The risk quotient indicating an LC50 below its test-specific CCC is for embryo-larval exposures of rainbow trout (Birge et al. 2000). Although not found in ECOTOX, the study also reported an LC10 indicating early-life-stage mortality could occur at nearly one-third the test-specific CCC concentration. The EC20s and EC50 reported in (Birge et al. 1978) represent gross morphological impairments to the vertebrae in rainbow trout surviving the test. A brook trout growth EC20 used in criterion derivation had a risk quotient of 1.09, indicating frank effects occurring at the test-specific CCC and suggesting detrimental effects occurring at and below the CCC limits (Cleveland et al. 1986). Original data from the study providing an Atlantic salmon EC20 (McKee et al. 1989) that was also used in CCC derivation suggested a exposure-response relationship with reductions in growth below the test-specific CCC, but the effect was not statistically significant until growth was reduced, on average, by 36%.

The plotted risk quotients for invertebrates include data for growth, reproduction, ecosystem productivity, and mortality among 29 species. While the bulk of the data indicate responses occurring above criterion limits, the plots draw attention to risk quotients representing chronic reproduction EC50s and EC20s for *Ceriodaphnia dubia* ranging from 0.12 to 2.4 (McCauley et al. 1986, ENSR Consulting and Engineering 1992, European Al Association 2010, Gensemer et al. 2018, OSU 2018a) and risk quotients representing chronic LC50s below their test-specific CCCs for *Ceriodaphnia dubia* and *Daphnia magna* (European Al Association 2009, 2010).

8.2.1.3 Risk of Aluminum Exposures within criteria limits in Waters Regulated by MassDEP

Section 2.1.1 of this Opinion establishes rainbow trout as a suitable surrogate species in the absence of data for effects on sturgeon. The attendant uncertainties when extrapolating across species can lead to underestimation or over estimation of effects. Taken with the discussion of lab-to-field extrapolation in Sections 2.1.1 and 2.1.2.1 of this Opinion and response magnitudes associated with the endpoints used in deriving the aluminum CCC, NMFS gives the species the benefit of the doubt.

SURVIVAL

NMFS' 2020 Opinion on EPA's promulgation of freshwater aquatic life criteria for aluminum in Oregon (NMFS 2020b) relied on data reported by Gundersen et al. (1994) for its likely to adversely affect determination. NMFS' 2020 Opinion concluded that the application of EPA's "low effect" adjustment factor to the lower LC50 estimates reported by Gundersen et al. (1994) for rainbow trout provided an LC05 estimate that was less than the CMC, indicating mortality in fish is likely to occur due to exposures within CMC limits. The lower normalized LC50s ranged from 1,680 to 2,180 µg/L and were for exposures with pH values at 8.3 while the normalized LC50s that were reported at >5,164 to >7,216 µg/L in the same study were for exposures at pH 7.6. This is an important distinction because New England waters trend towards more neutral to acidic conditions whereas the Columbia River Basin, one of the principle waterways within the action area for NMFS' 2020 Opinion, is relatively alkaline, with pH values around 8.3 (Little et al. 2012). This Opinion, therefore, does not replicate the basis for the determination of NMFS' 2020 Opinion because the exposure conditions within the state of Massachusetts are not expected to result in mortality at or below the CMC limits.

The acute aluminum exposure data drawing our attention in this Opinion are not direct effects on survival. Behavioral studies for avoidance and doubling of cough frequency suggest acute behavioral effects may occur within the CMC concentration limits (Ogilvie and Stechey 1983, Gunn and Noakes 1986). For such responses to be considered take under the ESA, it would need to be found to significantly impair or disrupt normal behavioral patterns. To place this response in context, Hughes (1975) reported that rainbow trout cough frequency generally doubled at 100 mg/L total suspended sediment and another study reported rainbow trout avoided waters with 100 mg/L suspended sediment (Suchanek et al. 1984, after Newcombe and Jensen, 1996). Taken together these studies provide evidence that a doubling of cough frequency would result in avoidance by rainbow trout. Aluminum exposures within the CMC limits may not result in mortality, but exposures would potentially cause fish to leave otherwise suitable habitat. The CMC is implemented as a one-hour average and the cough and avoidance tests were conducted within 15-minute intervals. In the absence of data indicating fish would return to an area one hour after adverse conditions abate, or whether gill damage, delayed mortality, or increased predation vulnerability would occur subsequent to the avoidance response, NMFS gives the species the benefit of the doubt.

While EPA's assessment methodology suggests that the CCC is generally protective against mortality, the rainbow trout embryo-larval LC50 represented by a risk quotient of 0.91 (Birge et al. 2000) indicates that early-life-stage mortality would occur at and below the CCC limits. The other rainbow trout LC50s were for exposures of alevins (Holtze 1983, Hickie et al. 1993), fingerlings (Call et al. 1984), and juveniles (Gundersen et al. 1994).

Taken together, these data suggest adverse effects on survival are likely to occur in ESA-listed shortnose sturgeon and the Gulf of Maine and New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon due to exposures within the aluminum CMC and CCC limits. While there are no data for population viability analysis, the viability of ESA-listed sturgeon populations in Massachusetts' waters is highly sensitive to juvenile mortality resulting in lower numbers of sub-adults recruiting into the adult breeding population (NMFS 1998a, ASSRT 2007).

GROWTH

Growth is an important determinant of survival and maturation, and thus recruitment (Anderson 1988, Poletto et al. 2018). Early-life-stage studies for salmonids, including an EC50 risk quotient or 0.91 for embryo-larval rainbow trout, indicate that adverse effects on growth could result from exposures within the aluminum CCC limits (Birge 1978, Birge et al. 1978, Cleveland et al. 1986, McKee et al. 1989, Birge et al. 2000). These data suggest adverse effects on growth are likely to occur in ESA-listed shortnose sturgeon and the Gulf of Maine and New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon due to exposures within the aluminum CCC limits.

Reproduction

Available data for the effects of aluminum on reproduction in fish were NOECs for fertilization (Everhart and Freeman 1973) and hatch success (Buckler et al. 1995). The Everhart and Freeman (1973) study reported no effects on successful fertilization, but the study apparatus intended to mimic a natural redd prevented the removal of dead eggs, so the test was terminated before hatch success could be evaluated. The Buckler et al. (1995) reported hatch success to be unaffected by aluminum exposures ranging from 38 to 300 µg/L and pH value of 5.5.

Abundance and quality of forage species

Examination of the data behind the risk quotients presented in Figure 16 indicates that adverse effects will occur in invertebrates exposed to aluminum within the CCC, but not CMC limits. While the diets of larval shortnose and Atlantic sturgeon have not yet been characterized, there are studies of larval green sturgeon (Zarri and Palkovacs 2019) and larval white sturgeon (Muir et al. 2000) diets. Diets are likely location-specific based on availability; larval stages of both green and white sturgeon were reported to rely on zooplankton species such as copepods, amphipods, and dipterans. An assessment of effects for listed species must address any evidence indicating adverse effects may occur to an individual of that species, but when evaluating effects to forage species it is the abundance and quality of forage species that is of concern. With respect to the quality of forage species, NMFS does not expect that EPA's approval of the aluminum CMC and CCC will affect the quality of forage species because, as discussed previously, aluminum does not bioaccumulate in the food chain (see Section 8.2.1).

Among the 44 zooplankton risk quotients representing LC50s (0.30+/-0.46), five indicated adverse effects on survival within criterion limits (ENSR Consulting and Engineering 1992, European Al Association 2009). Among 36 zooplankton risk quotients representing EC20s for reproduction, four indicate adverse effects within criterion limits (ENSR Consulting and Engineering 1992, European Al Association 2009, CECM 2014, Gensemer et al. 2018). Risk quotients for the types of species more likely to occur in the diet of adult sturgeon, worms and mollusks, ranged from 0.0045 representing an LC50 for the red-rimmed melania snail (foreign Shuhaimi-Othman et al. 2013) to 0.55 representing an EC20 for fat mucket mussel growth (Wang et al. 2016). The implications of these effects on the abundance and quality of forage species for shortnose and Atlantic sturgeon is attenuated by the majority of risk quotients representing LC50s and EC20s indicating adverse effects would not occur within criterion limits and the wide variety of forage species sturgeon consume. A reduction in the abundance of one benthic species is likely to be compensated for by an increase in other species (Wesolek et al. 2010). NMFS does not expect that aluminum exposures within CCC or CMC limits are likely to affect the abundance or quality of forage for shortnose sturgeon and the Gulf of Maine and New

York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon.

8.2.1.4 Likely to Adversely Affect Determination for EPA Approval of MassDEP Adoption of Freshwater Aluminum Criteria

NMFS concludes that EPA's approval of MassDEP adoption and implementation of the recommended National Recommended Water Quality Criteria for aluminum is likely to adversely affect shortnose sturgeon and the Gulf of Maine, New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon because:

1) Permitting and monitoring of MassDEP-regulated waters indicate that exposures to aluminum in Sturgeon Waters will occur,
2) The toxicity of aluminum in surrogate species indicate that exposures within criteria limits will likely result in adverse effects on the survival and growth of early-life-stage fish, and
3) The viability of ESA-listed sturgeon populations in Massachusetts' waters is highly sensitive to juvenile mortality resulting in lower numbers of sub-adults recruiting into the adult breeding population (NMFS 1998a, ASSRT 2007).

8.2.2 Cadmium Criteria for the Protection of Aquatic Life in Fresh and Salt Waters

Cadmium is used in batteries and pigments and in the manufacture of electronics and plastics. It is a component of fossil fuels, alloys, cement, and some fertilizers (ATSDR 2012). Given its abundant usage, cadmium is a common pollutant in stormwater. Shaver et al. (2007) reported the median cadmium concentration in urban runoff at 1.0 +/- 4.42 μg/L with highway runoff ranging from 0-40 μg/L and parking lot runoff ranging from 0.5-3.3 μg/L. Median dissolved cadmium concentrations in stormwater from commercial, industrial, and freeway land use areas were reported at 0.3, 0.6, and 0.7 μg/L, respectively.

The biological availability of cadmium in water is strongly influenced by aquatic chemistry: the abundance of ligand ions, organic acids, organic matter, and clay particles. While complexation with substances in the water column results in precipitation and incorporation in bed sediments, bed sediment is not a static sink. Cadmium can return into the water column and become biologically available when sediments are disturbed and conditions, such as low pH, favor cadmium release in the free ion form (Cadmium Guideline USEPA 2016). Scenarios in which this might occur include storm events (Krein and Bierl 1999, Paus et al. 2014, Vidal-Dura et al. 2018) and, in particular, re-inundation of exposed sediments after drought (Mosley et al. 2014).

Cadmium is a calcium analog that competes with calcium receptors at the gill. This disrupts calcium and ionic homeostasis in both freshwater and saltwater species (Adiele et al. 2010, Garcia-Santos et al. 2011, Onukwufor et al. 2015, Tang et al. 2016). Cadmium can accumulate at the gill, but is also transported throughout the body, accumulating to the highest extent in the organs with important roles in filtration and detoxification, the liver and anterior kidneys for fish and the hepatopancreas of arthropods and mollusks (Kouba et al. 2010, Paschoalini and Bazzoli 2021, Rodrigues et al. 2022). At the cellular level, cadmium induces oxidative stress, interfering with mitochondrial function and cellular repair that can lead to organ-level effects. If cellular injury is extensive, consequences for organ function will influence the survival and health of individuals (Paschoalini and Bazzoli 2021, Sun et al. 2022). A study by Mebane (2006) included a review of other data for cadmium dietary exposures and body burdens. Although there were not adequate data to establish acceptable tissue effect concentrations for aquatic life, Mebane (2006) concluded that cadmium is unlikely to accumulate in tissue to levels that would result in adverse effects to aquatic invertebrates or fish at the calculated CCC. In the Cadmium Guideline, EPA concluded that the evaluation of direct exposure effects to organisms via water is more applicable to the development of criteria for aquatic life than dietary exposure.

The EPA proposes to approve MassDEP adoption and implementation of the freshwater and saltwater cadmium National Recommended Criteria for the Protection of Aquatic Life. For fresh waters, these criteria are hardness-based values for the CMC and CCC. The saltwater CMC and CCC concentrations are 33 and 7.9 µg/L, respectively (USEPA 2016). The CMC is a one-hour average and the CCC is a four-day average not to be exceeded more than once in three years on average.

The EPA's BE used raw data from the 2016 Cadmium Guideline to estimate an acute MAF, then obtained chronic MAF using the final acute-to chronic ratio provided in the guideline document. These MAFs were then used to normalize and aggregate the available data for the toxic effects of cadmium in aquatic organisms.

8.2.2.1 Exposure to Cadmium in the Action Area

The mineral resources of Massachusetts do not include ores that would be associated with cadmium (e.g., zinc ore). Because cadmium is not naturally enriched in Massachusetts soils, we would not expect it to be concentrated or redistributed to aquatic habitats due to soil disturbing activities. Cadmium is expected to reach Sturgeon Waters through stormwater runoff and snowmelt from highways and urbanized areas and through discharges from permitted outfalls. A review of EPA Enforcement and Compliance History Online database identified 62 facilities within catchments adjacent to Sturgeon Waters that are required to monitor for cadmium in their discharges. Three of these dischargers have records of exceeding discharge limits and eight failed to report required discharge monitoring data. Two of these dischargers are in the Connecticut River watershed and the remaining are in the Boston area (Figure 17). The BE reports, and NMFS has confirmed, that the available monitoring data for sturgeon rivers are at or below CCC criterion limits at ambient water hardness. However, nearly all of the available data for Sturgeon Waters was collected prior to 2005. The most recent monitoring data from the Water Quality Portal is a single observation of 0.033 µg/L cadmium in 2021 from a station in North Agawam. While hardness data were not available to assess whether this exceeded the applicable criterion concentration, a hardness of 1.5 mg/L calcium carbonate would require a

CCC concentration of 0.3 µg/L cadmium. Meanwhile total hardness reported for Sturgeon Waters in the Water Quality Portal range from 23 to 350 mg/L calcium carbonate.

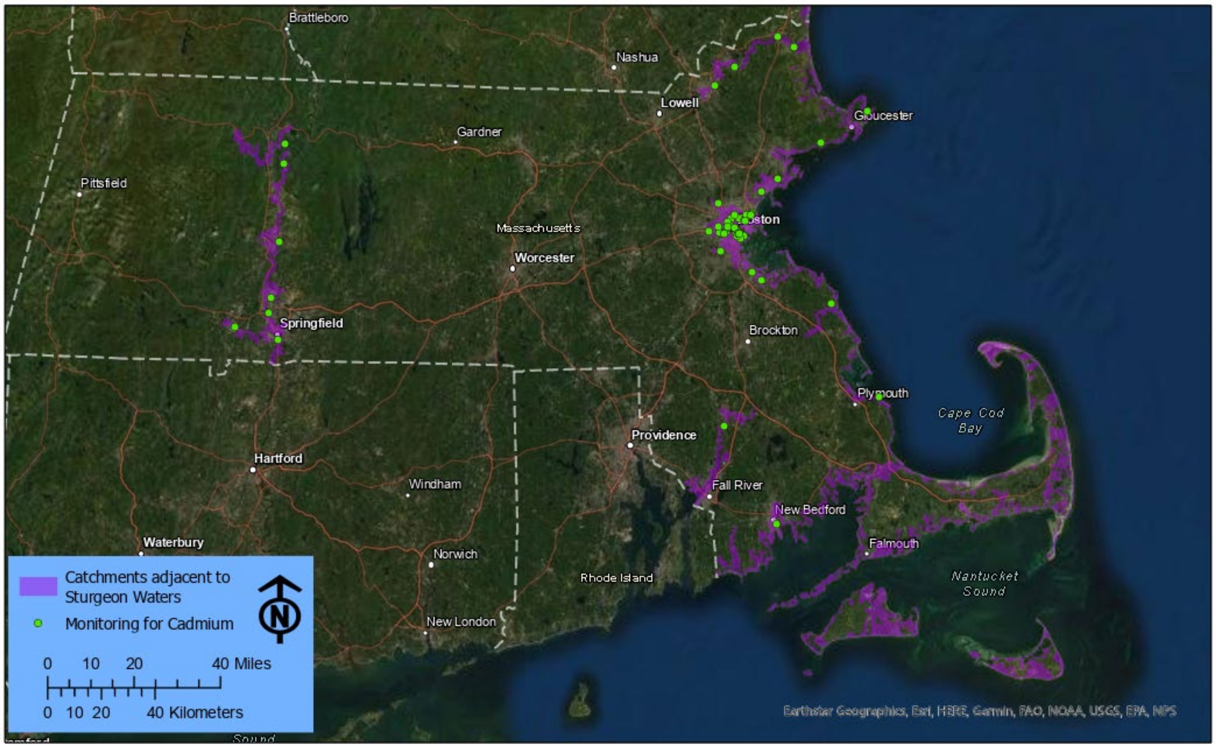

Figure 17. Locations of NPDES facilities within catchments adjacent to sturgeon waters and are required to monitor for cadmium.

8.2.2.2 Responses to Cadmium Within Criteria Limits

The screened cadmium data from ECOTOX included 824 records from 101 sources exposing 63 species of fish. Data for invertebrates, representing forage species, were provided by 299 studies that conducted 2007 toxicity tests evaluating the effects of cadmium on 160 invertebrate species.

While the LC50s for white sturgeon are generally an order of magnitude lower than the test-specific criteria, the magnitude of responses at the LOECs from the same tests suggest adverse effects would occur within CMC limits. Twenty percentof individuals exhibited loss of equilibrium and immobilization at the LOEC of 3.125 µg/L for an acute test reported by Calfee et al. (2014). With a test-specific criterion of 2.12 µg/L cadmium, the risk quotient for this LOEC is 0.67. The four-day survival LOECs for fish exposed at age two days post hatch, 30 days post hatch, 44 days post hatch, 61 days post hatch, 72 days post hatch, and 89 days post hatch had risk quotients ranging from 0.08 to 0.69, but the response magnitudes ranged from 20+/-11.55% to 95 +/-10% (Ingersoll et al. 2014). Ingersoll et al. (2014) also reported a biomass LOEC for white sturgeon at 5.29 µg/L for a chronic exposure that reduced fish mass by 25%, the EC10 calculated for this exposure was 2.4 µg/L with a range from 1.5 to 4.0 µg/L.

The risk quotients for salmonids showing LC10, LC20, LC50, LOEC, and MATC mortality endpoints for acute (rainbow trout fry and larva Davies et al. 1993, Stratus Consulting Inc. 1999, Ingersoll et al. 2014) and chronic (rainbow trout egg, fry, and juveniles Davies et al. 1993, Stratus Consulting Inc. 1999, Mebane et al. 2008) exposures within criterion limits are concerning. Mebane et al. (2008) reported a chronic growth and development LOEC with a risk quotient of 1.8 representing a 40+/- 10% decrease in wet weight of rainbow trout exposed from egg stage to 62 days.

The four-day survival LOECs for rainbow trout exposed at 18 days post hatch, 32 days post hatch, 46 days post hatch, 60 days post hatch, 74 days post hatch, and 95 days post hatch had risk quotients ranging from 0.36 to 1, averaging 0.56 with response magnitudes ranging from 20+/-14.14% to 92.5 +/-9.57% (Ingersoll et al. 2014). Risk quotients for the LC10s reported for these exposures by Ingersoll et al. (2014) ranged from 0.49 to 2.37, averaging 0.94, which is essentially within criterion limits. The LC50 risk quotients reported for rainbow trout in 19 separate studies[15] ranged from less than 0.001 to 2, averaging 0.54+/-0.45 (n=91). Among these, 38 tests exceeded a risk quotient of 0.5.

Hansen et al. (2002) reported five-day LC50s that ranged from 0.36 to 2.07 µg/L for rainbow trout exposed to cadmium under differing temperature and water hardness conditions. Risk quotients for these data, which apply criteria calculated using EPA’s hardness adjustment equation, ranged from 0.84 to 0.86 for exposures under an average water hardness of 30.4 mg/L calcium carbonate and temperature of 9.4 ºC. At a mean temperature of 7.8, risk quotients ranged from 0.35 to 0.58 with water hardness values of 30 and 90 mg/L calcium carbonate. Risk quotients for LC50s reported by a larger study with a similar design ranged from 0.24 to 0.81 (Stratus Consulting Inc. 1999).

The risk quotients representing growth and development of rainbow trout ranged from 0.03 to 1.78 (Besser et al. 2007, Mebane et al. 2008, Adiele et al. 2011, Ingersoll et al. 2014). The LOECs reported by Mebane et al. (2008) underscore the influence of temperature on cadmium toxicity. The risk quotient of 1.78 represents response magnitudes of five precent reduction in length and 17% reduction in weight at 12.5 ºC and hardness of 29.4 mg/L calcium carbonate. At 9.8 ºC and hardness of 19.7 mg/L calcium carbonate, risk quotients were 0.03 and 0.07 Mebane et al. (2008).

The plotted risk quotients for the effects of cadmium on freshwater invertebrates include growth and development, reproduction, behavior, population, and mortality responses. While the bulk of the invertebrate data indicate responses occurring above criteria limits, the plots draw attention to risk quotients representing acute and chronic LC50s for *Daphnia magna* and scud (Gale et al. 1992, Borgmann et al. 2005), chronic reproduction and mortality LOECs (Zuiderveen and Birge

[15] (Davies 1975, Chapman 1978b, Goettl and Davies 1978, Goettl et al. 1978, Call et al. 1981, Daoust 1981, Birge et al. 1983, Phipps and Holcombe 1985, Cusimano et al. 1986, Pascoe et al. 1986, Davies et al. 1993, Davies and Brinkman 1994, Hollis et al. 1999, Stratus Consulting Inc. 1999, Niyogi et al. 2004, Besser et al. 2007, Calfee et al. 2014, Ingersoll et al. 2014, Naddy et al. 2015)

1997), an acute reproduction IC10s and chronic IC20 quotients for growth, development and mortality (Chadwick Ecological Consultants Inc. 2003).

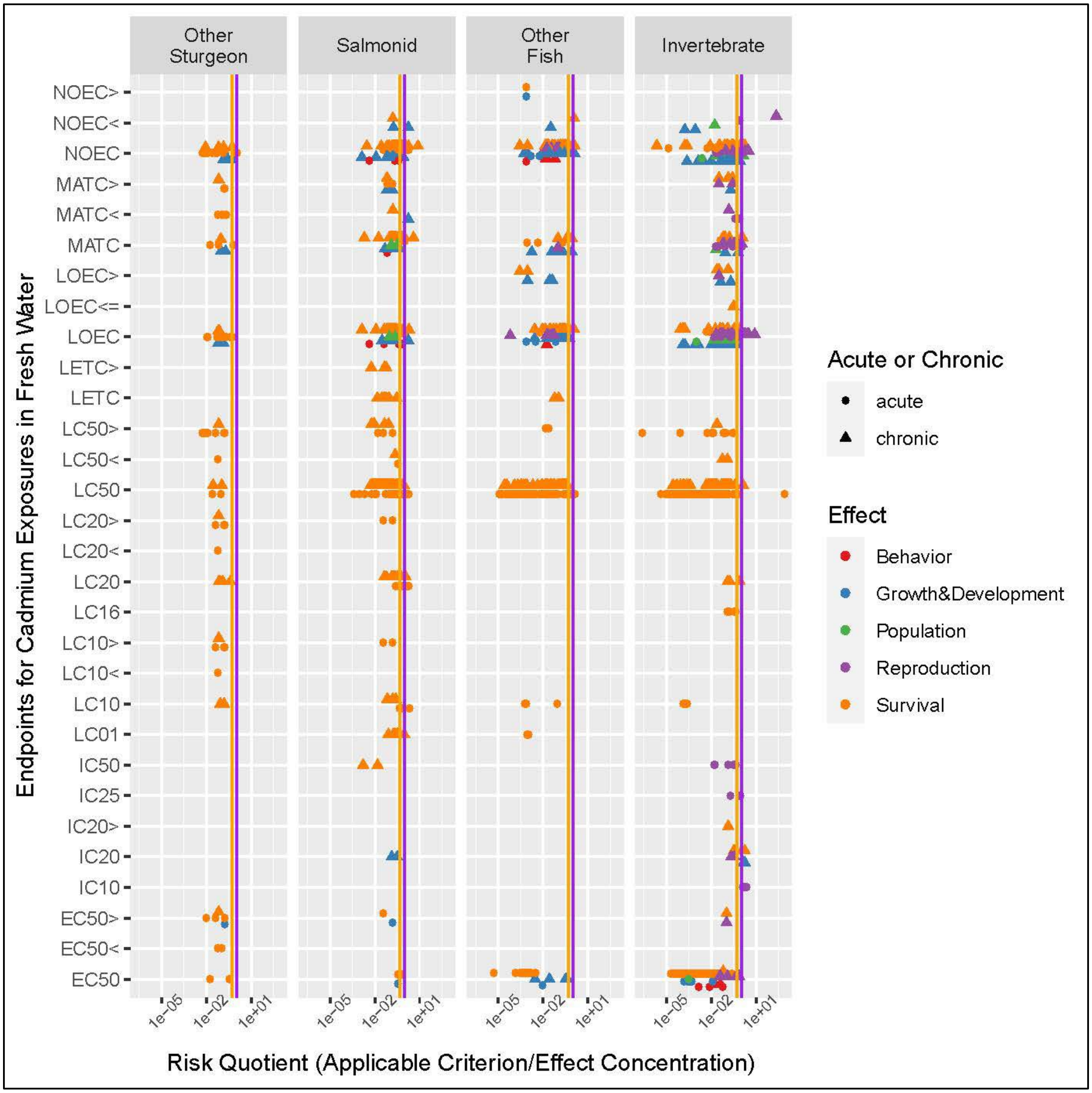

Figure 18. Distribution of risk quotients for freshwater exposures to cadmium in context of reference lines representing the applicable criterion (purple) and one-half the applicable criterion (orange).

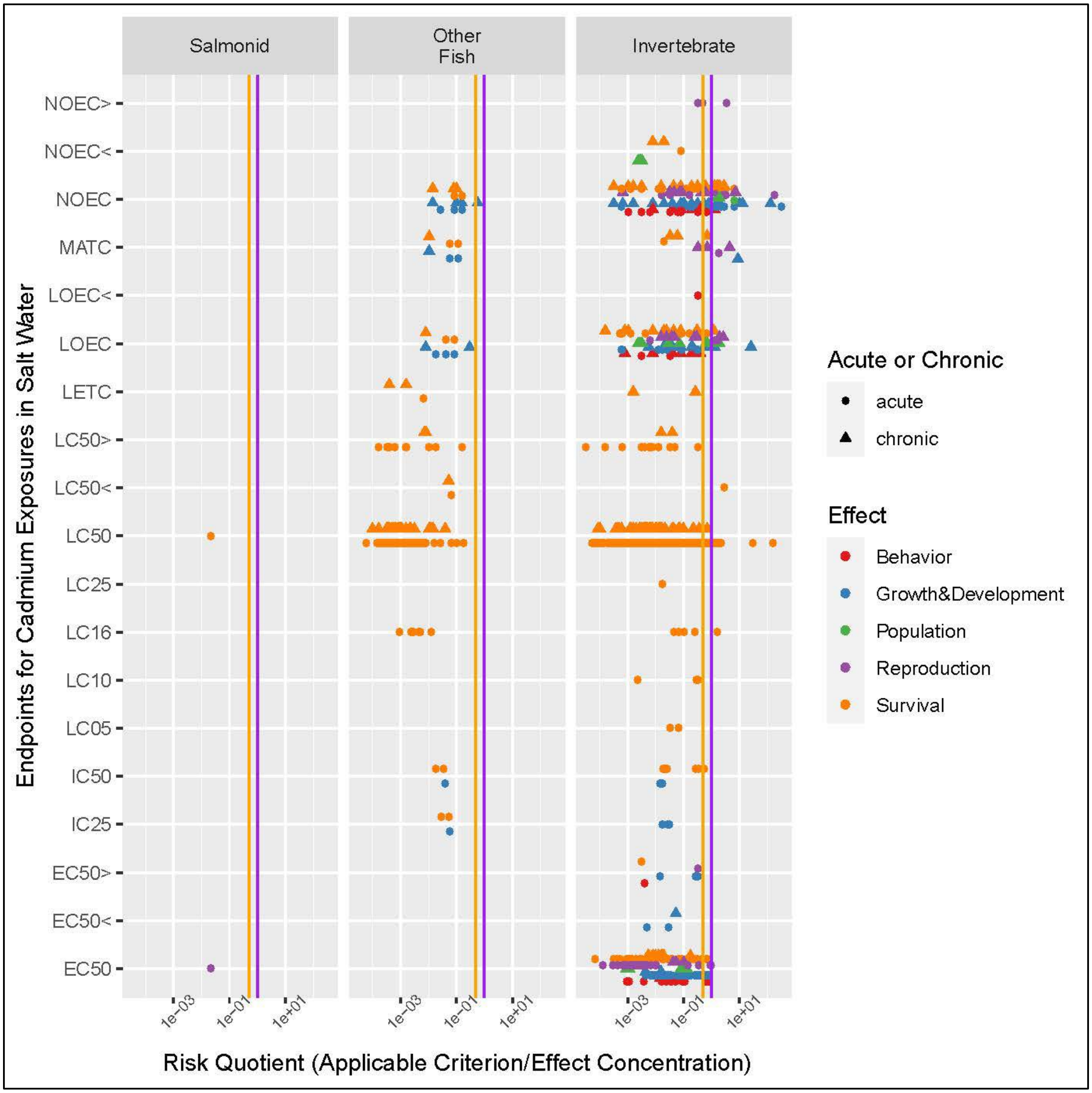

Figure 19. Distribution of risk quotients for saltwater exposures to cadmium in context of reference lines representing the applicable criterion (purple) and one-half the applicable criterion (orange).

Data for exposures of saltwater fish species in Figure 19 do not indicate that increased mortality would be expected to occur within the cadmium saltwater CMC limits. Given that mortality, growth and development LOECs, inhibition concentrations, and lethal thresholds (ICxx and LETC in Figure 19, respectively) are at concentrations close to an order of magnitude higher than the CCC and CMC, it is reasonable to expect that reproduction and other effects would not occur within the saltwater CMC or CCC limits either.

The plotted risk quotients for the effects of cadmium on saltwater invertebrates include growth and development, reproduction, behavior, population productivity, and mortality responses. While the bulk of the invertebrate data indicate responses occurring above criterion limits, risk quotients representing LC50s for amphipod (Meador 1993), daggerblade grass shrimp and mud crab (Thorpe 1990), harpacticoid copepod (Forget et al. 1998), opossum shrimp (Nimmo et al. , Roberts et al. 1982, Ward 1989, Voyer and Modica 1990), and rock crab (Johns and Gentile 1981) indicate that mortality will occur at concentrations below the saltwater CMC. Effects within the CCC limits are also indicated by risk quotients representing reproduction LOECs for sea urchin (Jonczyk et al. 1991, Arizza et al. 2009), growth and development of cuttlefish (Lacoue-Labarthe et al. 2010) and daggerblade grass shrimp (Manyin and Rowe 2009) and reproduction and population stability of *Moina monogolica* (Wang et al. 2009).

However, early life stage shortnose and Atlantic sturgeon do not live in marine waters, so larger prey items that would be consumed by adult and juvenile sturgeon are of interest: mollusks, gastropods, polychaetes, crabs, oysters, and mussels (excluding larval stages). The risk quotients for effects (i.e., excluding NOECs) in species likely to be consumed by adult and juvenile sturgeon ranged from less than 0.001 to 0.44, with 85% of risk quotients below 0.05.

NMFS concludes that EPA's approval of Massachusetts adoption of the saltwater National Water Quality Criteria for cadmium may affect, but is not likely to adversely affect shortnose sturgeon or the Gulf of Maine, New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon because responses in surrogate species are extremely unlikely to occur and are therefore discountable. Further, NMFS also concludes that the exposure of forage species to cadmium within criteria limits may affect, but is not likely to adversely affect ESA-listed sturgeon because responses of saltwater forage species is extremely unlikely to occur and thus discountable.

8.2.2.3 Risk of Cadmium Exposures Within Criteria Limits in Waters Regulated by MassDEP

The best available data do not indicate any adverse effects to fish exposed to cadmium in salt water within the saltwater CMC or CCC. The following discussion addresses freshwater exposures of fish to cadmium within criterion limits and fresh and saltwater exposures of invertebrates to cadmium within criterion limits.

SURVIVAL

Although the risk quotients for white sturgeon survival, growth and development LOECs and MATCs indicate responses at exposures above cadmium CMC and CCC limits, the magnitude of the responses at the MATCs and LOECs suggest that exposures of shortnose sturgeon and Atlantic sturgeon to cadmium within the CMC and CCC limits would result in mortality and reduced growth. In addition, within genus comparability of sensitivity to toxicants is not always consistent (see discussion in Section 2.1.1). NMFS considers rainbow trout to be a suitable surrogate and data from multiple sources indicate mortality in early-life-stage fish exposed to cadmium within CCC limits. While data are not available to perform a population viability analysis for ESA-listed sturgeon populations in Massachusetts waters, these data are important because the viability of these populations are highly sensitive to juvenile mortality resulting in lower numbers of sub-adults recruiting into the adult breeding population (NMFS 1998a, ASSRT 2007).

GROWTH

Growth is an important determinant of survival, and thus recruitment (Anderson 1988, Poletto et al. 2018). Significant effects of cadmium on growth was reported to occur within criterion limits, but was temperature dependent (Mebane et al. 2008). The white sturgeon studies did not evaluate the effect of temperature on cadmium toxicity. The studies comparing white sturgeon to rainbow trout ran toxicity tests at each of species' optima, 15+/-1 ºC for sturgeon and 12+/-1 ºC for trout (Calfee et al. 2014, Ingersoll et al. 2014, Wang et al. 2014). The test in the Mebane et al. (2008) study reporting rainbow trout growth effects within criteria limits was conducted at 12.5 degrees while the tests conducted at 9.8 ºC had LOECs resulting in risk quotients of 0.03 and 0.07. With increasing temperatures expected under climate change (IPCC 2021), it is reasonable to expect that cadmium exposures within CCC limits is likely to affect growth of shortnose sturgeon and the Gulf of Maine, New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon.

REPRODUCTION

Data for the effects of cadmium on reproduction in sturgeon and salmonid species are not available. Data for other fish species do not indicate effects on reproduction within cadmium criteria limits. While reproduction is critical to population persistence, fish must first survive and grow in order to reproduce. Given that cadmium exposures within criteria limits are expected to adversely affect early-life-stage survival and growth, it is reasonable to expect that these effects will, in turn reduce recruitment of reproductive fish.

ABUNDANCE AND QUALITY OF FORAGE SPECIES

Examination of the data behind the risk quotients presented in Figure 15 indicates that adverse effects will occur in invertebrates exposed to cadmium within the CCC criteria limits. While the diets of larval shortnose and Atlantic sturgeon have not yet been characterized, there are studies of larval green sturgeon (Zarri and Palkovacs 2019) and larval white sturgeon (Muir et al. 2000) diets. Although diets are likely to be location-specific based on availability, larval stages of both green and white sturgeon were reported to rely on zooplankton species such as copepods, amphipods, and dipterans. An assessment of effects for listed species must address any evidence indicating adverse effects may occur to an individual of that species, but when evaluating effects to forage species it is the abundance and quality of forage species that is of concern. NMFS does not expect that EPA's approval of the cadmium CMC and CCC will affect the quality of forage species because, as discussed previously, Mebane (2006) concluded that exposures to cadmium within criterion limits is unlikely to result in accumulation in tissues to levels that would result in adverse effects to aquatic invertebrates or fish (see Section 8.2.2).

While adverse effects may occur in invertebrates exposed to cadmium within both the freshwater and saltwater CMC and CCC limits, the bulk of the data indicate effects occurring above criterion limits. Early-life-stage sturgeon rely on zooplankton. Excluding extreme risk quotient values greater than 100, risk quotients for freshwater planktonic species ranged from less than 0.001 to 7.9 in 26 species. Data indicating adverse effects within criteria limits are for *Hyalella*, *Daphnia,* and *Ceriodaphnia* species. About half of the risk quotients in Figure 18 are from toxicity tests of *Daphnia* and *Ceriodaphnia* species, which are used in toxicity tests because they are extremely sensitive to aquatic pollutants. Among food items consumed by larger sturgeon, including mollusks, worms, and larger crustaceans like crayfish or crab, risk quotients ranged from less than 0.001 to 0.45 in 26 freshwater species.

The implications of any effects on the abundance and quality of forage species for shortnose and Atlantic sturgeon is attenuated by the wide variety of forage species sturgeon consume. A reduction in the abundance of one benthic species is likely to be compensated for by an increase in other species (Wesolek et al. 2010). NMFS does not expect that cadmium exposures within CCC or CMC limits are likely to affect the abundance or quality of forage for shortnose sturgeon and the Gulf of Maine, New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon.

8.2.2.4 Likely to Adversely Affect Determination for EPA Approval of MassDEP Adoption of Freshwater Cadmium Criteria

NMFS concludes that EPA's approval of MassDEP adoption and implementation of the recommended National Recommended Water Quality Criteria for cadmium in freshwater is likely to adversely affect shortnose sturgeon and the Gulf of Maine, New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon because:

1) Permitting and monitoring of MassDEP-regulated waters indicate that exposures to cadmium will occur,
2) The toxicity of cadmium in surrogate species indicate that exposures within criteria limits will likely result in adverse effects to the survival of early-life-stage fish,
3) With increasing temperatures under climate change (IPCC 2021), temperature-dependent effects of cadmium exposure on growth in surrogate species indicates that exposures within criteria limits are likely to affect growth of shortnose sturgeon and the Gulf of Maine, New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon, and
4) The viability of ESA-listed sturgeon populations in Massachusetts' waters is highly sensitive to juvenile mortality resulting in lower numbers of sub-adults recruiting into the adult breeding population (NMFS 1998a, ASSRT 2007).

8.2.3 Copper Criteria for the Protection of Aquatic Life

Copper is a ubiquitous metal in the built environment. Copper is used in pipes, wire, roof flashing, brake pads, cookware, and antimicrobial products. Copper is a common pollutant in stormwater. Shaver et al. (2007) reported the median copper concentration in urban runoff at 16 +/- 2.24 µg/L with highway runoff ranging from 22 to 7033 µg/L and parking lot runoff ranging from 8.9 to 78 µg/L. Median dissolved cadmium concentrations in stormwater from residential, commercial, industrial, and freeway land use areas were reported at 7, 8, 8, and 11 µg/L, respectively.

The biological availability of copper in water is strongly influenced by aquatic chemistry: pH, and the abundance of ligand ions, organic acids, organic matter, and clay particles. While complexation with substances in the water column can result in precipitation and incorporation in bed sediments, bed sediment is not a static sink. Copper can return to the water column and become biologically available when sediments are disturbed and conditions, such as low pH, favor cadmium release in the free ion form (USEPA 2007). Scenarios in which this might occur include storm events (Krein and Bierl 1999, Paus et al. 2014, Vidal-Dura et al. 2018) and, in particular, re-inundation of exposed sediments after drought (Mosley et al. 2014).

Copper is an essential trace element found in metal-containing enzymes such copper-zinc superoxide dismutase. Copper uptake at the gill is dependent on ambient concentrations of the free ion form (Taylor et al. 2003). Excess copper induces oxidative stress, olfactory impairment, increased plasma ammonia and disturbed acid–base balance (Eyckmans et al. 2011, Grosell 2011). The balance between uptake, distribution throughout the body, detoxification and removal of excess copper is modulated by the liver (Cousins 1985, Grosell and Wood 2002). Oxidative stress results in gill damage through either disruption of branchial structure, mucus secretion and occlusion, inhibition of respiratory enzymes and damage to gill oxygen receptors (Grosell 2011). The generation of free radicals caused by excess copper also damages internal organs. Specifically, the liver, kidney, heart, brain, and reproductive organs are adversely affected by excess copper. Impairments of physiological and biochemical functions in turn can affect the performance of the whole organism (McDonald and Wood 1993, Beaumont et al. 2000). Copper generally does not biomagnify in food chains consisting of primary producers, macro invertebrate consumers, and fish occupying trophic level and higher (Cardwell et al. 2013).

The EPA proposes to approve MassDEP adoption and implementation of the 2007 National Recommended Copper Criteria for the Protection of Aquatic Life for freshwater (Copper Guideline USEPA 2007), hereafter "Copper Guideline." MassDEP is adopting duration and frequency metrics for acute and chronic criteria of one-hour average copper concentration must not exceed the acute value more than once in three years on average and the 4-day average copper concentration must not exceed the chronic value more than once in three years on average. As described in section 5.1 of the BE, the Copper Guideline uses the copper biotic ligand model (BLM) to calculate acute and chronic values based on site-specific aquatic chemistry for ten parameters. This integrates the influence of aquatic chemistry on biological availability, and thus the toxicity, of copper in the water column. EPA's BE used data curated for Copper Guideline "because such data have been (1) reviewed following strict data quality requirements (Stephen et al. 1985) to ensure data are relevant and of high quality and (2) normalized to reference conditions using the BLM so toxicity data can be compared across species and to copper criteria in identical water chemistries." Data from the Copper Guideline for the shovelnose sturgeon were used to infer effects of exposures shortnose and Atlantic sturgeon within criteria limits. The BE used the LC50/2 approach to assess the freshwater CMC. The CCC was estimated using acute toxicity data transformed by an Acute to Chronic Ratio[16] reported in the Copper Guideline to calculate a chronic effect EC20 representative of the shortnose and Atlantic sturgeon.

Massachusetts' implementation plan (MassDEP 2021) is a guidance that includes specific requirements on the timing and conditions under which data are to be collected for the calculation of copper criteria that are consistent with EPA's Copper Guideline. With sufficient data, the tenth percentile of acute and chronic instantaneous criteria values generated from the BLM software will be used to calculate the final site-dependent acute and chronic copper criteria values, respectively. For watersheds where endangered species occur, the criteria will be the fifth percentile of the instantaneous criteria values (MassDEP 2021).

The following sections provide NMFS' evaluation of BLM-based acute and chronic copper criteria using monitoring data from the Water Quality Portal and the aquatic chemistry reported for toxicity tests from ECOTOX. Monitoring data were used to explore, to the extent possible, the influence of seasonal/temporal variability in aquatic chemistry on BLM-calculated criteria. Toxicity data were used to evaluate the BLM-calculated criteria for protectiveness. These evaluations consider criteria calculated using both the full water chemistry (Full) and simplified site chemistry (Simplified) options of the BLM software, Research Version 3.41.2.45. This is the best available tool for placing copper data in context of EPA's BLM-based criteria. The full water chemistry option requires data for temperature, pH, dissolved organic carbon (DOC), Major cations (calcium, magnesium, sodium, and potassium), major anions (sulfate and chloride), alkalinity, and sulfide. The simplified site chemistry requires data for temperature, pH, metal concentration, DOC, and hardness to estimate the major cations and anions.

Both the monitoring data and toxicity data lacked measurements for humic acid and sulfide. Humic acid and sulfide data are less critical to copper biological availability relative to dissolved organic carbon content. The BLM User's Guide acknowledges that humic acid is not routinely measured. Dissolved organic carbon is composed of multiple potential ligands for copper: humic and fulvic acids as well as low-molecular-weight carboxylic acids and phenols. The analysis of monitoring data in this Opinion applied the User's Guide recommendation to use a value of 10% humic acid for most natural waters (WindWard Environmental LLC 2019). While the software

[16] An acute to chronic ratio is the LC50 divided by the MATC

includes a column for sulfide, the User's Guide indicates that metal-sulfide reactions have not yet been incorporated into the BLM, but will likely to be added to subsequent versions of the model. Sulfide concentrations added in that column will not affect the BLM calculation. Sulfide concentrations are expected to be negligible in aerated waters, but waters affected by wastewater treatment plant effluents can have elevated sulfide concentrations. There are a number of research questions that need to be addressed before sulfide can be included as a required datum for the BLM. The analyses in this Opinion applied the User's Guide recommendation that a near-zero value be used in the absence of sulfide data. The guide also states that when sulfide reactions are omitted from model simulations, the BLM will always predict a lower (i.e., more protective) estimate of a concentration associated with toxicity.

8.2.3.1 Exposure to Copper in the Action Area

Water quality portal data from within Sturgeon Waters were available for 33 stations dated from 1952 (n=2 observations) to 2021 (n=11 observations). The most abundant data were collected in 1999 and 2000 (n=70 and 75, respectively). Fewer than 15 samples were taken in more recent years: 2019, 2020, and 2021. The data indicated a consistent pattern of seasonal influence on calculated CCC, with values for winter and spring consistently lower than for summer and fall (Figure 20). This is not surprising considering the natural carbon cycle's accumulation of organic matter over the growing season and eventual decomposition (e.g. Barth and Veizer 1999, Dodds 2002). The results of a Kruskal-Wallace Rank Sum test indicate statistical significance in calculated CCC among seasons for Taunton, Deerfield, and Westfield watersheds. Data for the Merrimack watershed are consistent with the winter-spring, summer-fall but, due to the smaller sample size, returned a p-value of 0.11.

Monitoring data within these waters indicate copper is present at low levels, at about one third or less the calculated criterion. The detection limits reported in the water quality portal range from 0.2 to 0.5 µg/L while the calculated CCC in Figure 20 are greater than 1 µg/L, so, if monitored, copper impairments would be detected under current monitoring practices. Data from ECHO identifies 162 facilities required to monitor for copper in their discharges. While the majority of these discharges are compliant with their permits, seven facilities have exceeded their copper discharge limits in the past three years, one is currently exceeding copper discharge limits, and 36 have failed to report required discharge monitoring data (Figure 21).

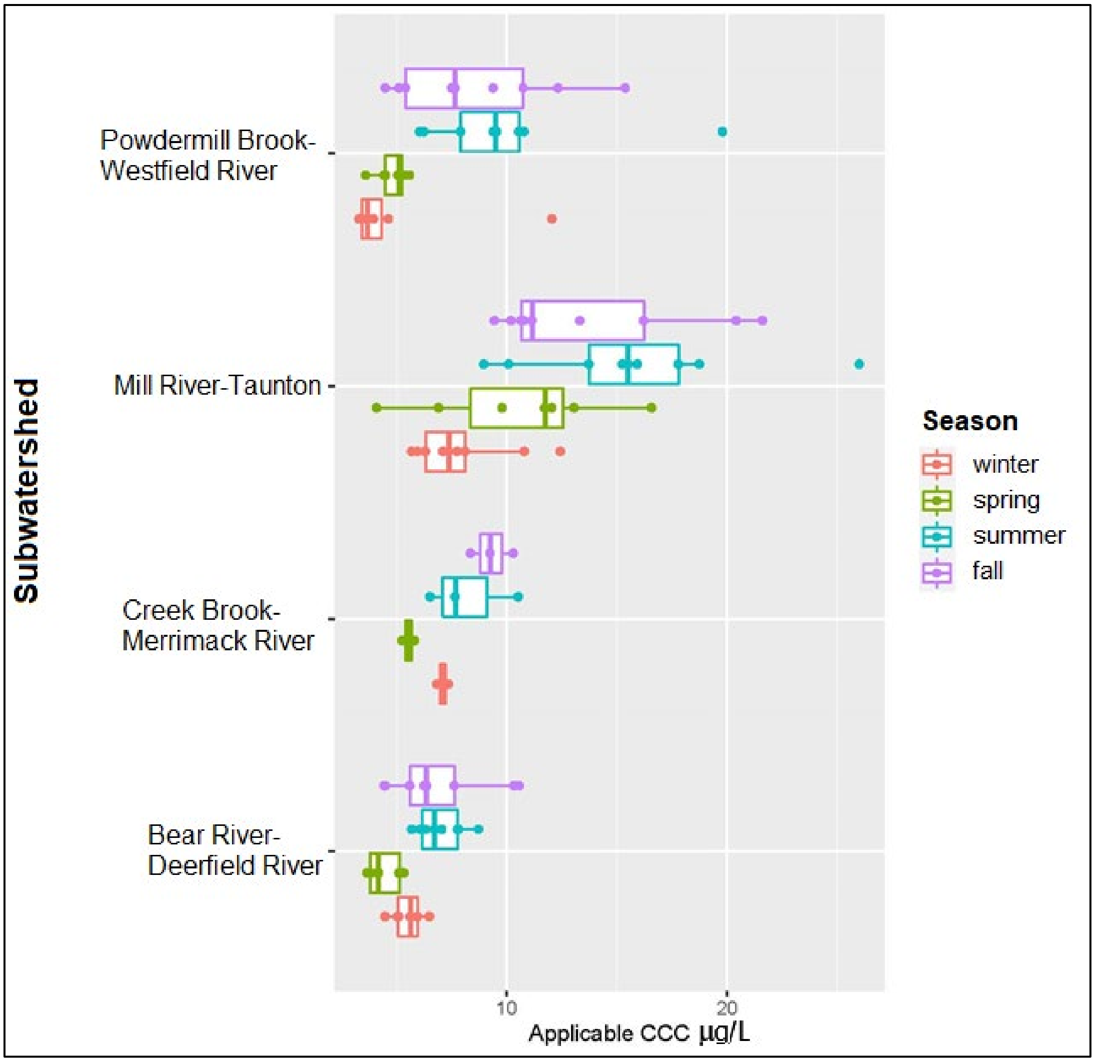

Figure 20. Seasonal distribution of CCC calculated using available data from sub-watersheds where shortnose and Atlantic sturgeon occur.

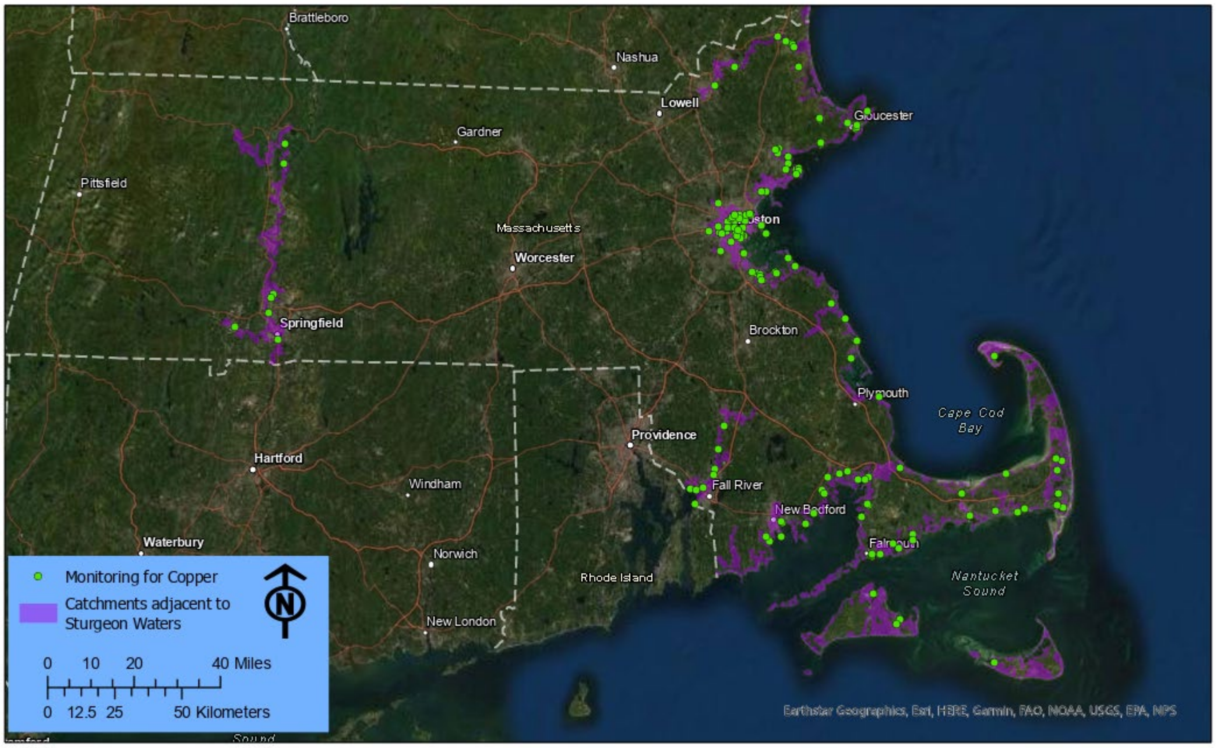

Figure 21. Locations of facilities required to monitor for copper in discharges within catchments adjacent to Sturgeon Waters and are required to monitor for copper.

8.2.3.2 Responses to Copper within Criteria Limits

Copper toxicity data for responses that are relatable to survival, growth, and reproduction, as well as behavioral responses that are important to survival in the wild, were collected from the ECOTOX June 15, 2022 update. Screening identified 66 references for toxicity tests with acceptable controls and meeting at least the simplified site chemistry data requirements of the BLM. Using the BLM simplified site chemistry option, test-specific criteria (Simple Criteria) were modeled for 824 chronic (>4 day) response observations, 859 acute response observations, and 246 non-response observations (e.g., LC0, NOECs) covering 19 fish species from eight taxonomic families and 38 invertebrate species from 20 taxonomic orders. A subset of these observations also included data for the major ions calcium, magnesium, sodium, potassium, sulfate, and chloride. This allowed calculation of test-specific criteria using the BLM full site chemistry option (Full Criteria), assuming a humic acid content of 10% and sulfur content of 0.001 mg/L per manual recommendations. Each resulting criterion was compared to its corresponding Simple Criterion to determine whether criteria generated with simplified chemistry data may bias interpretation. These comparisons were possible for 535 unique exposure chemistries for 806 responses reported in 14 studies. Overall, the Simple Criteria averaged 6% lower than the Full Criteria.

The BLM User's Manual recommends a value of 10% humic acid for most natural waters, stating that: *The variability of the dissolved organic matter content in diverse water sources has* *not been found to be an especially critical parameter, and little benefit is achieved by characterizing natural organic matter beyond DOC concentrations.* However, the diluent used by toxicity tests in the screened dataset included de-chlorinated tap water, well water diluted with deionized water, or artificially generated water composed of distilled deionized water and a mixture of reagent grade salts. These are not natural waters. About half of the observations in the screened dataset did not include test diluent information.

To assess the impact of the default humic acid value in the BLM and reliance on Simple Criteria in the analysis to maximize available data, test-specific criteria were recalculated using a humic acid content of 1% (Full Criteria HA1). In most cases, the resulting risk quotients were not altered enough to influence their interpretation as applied in this Opinion (see section 2.1.2.2). For example, a Simple Criterion of 2.54 µg/L is about 50% higher than the corresponding Full Criterion HA1 of 1.68 µg/L, but for the *Daphnia magna* LC50 at 7.7 µg/L (Villavicencio et al. 2011), the corresponding risk quotients of 0.22 and 0.33 would not influence NMFS' interpretation of the data. Among the 806 observations that could be evaluated, about 2% (n=16) of the Simple Criterion risk quotients exceeded a reference value of 0.5 while the Full Criterion HA1 risk quotient for that observation did not. These records are from two studies, one reporting LC50s for rainbow trout (n=13 Marr et al. 1999) and the other reporting reproduction EC50s for *Daphnia magna* (n=3 Villavicencio et al. 2011). Risk quotients for all available endpoint effect data are aggregated in Figure 18. The data for "Other Sturgeon" come from seven studies examining copper toxicity to white sturgeon that were published after the Copper Guideline was released.

Risk quotients representing acute LC50s for white sturgeon ranged from 0.28 to 0.96 (n=6) with four of these above the risk quotient of 0.5. (Little et al. 2012, Vardy et al. 2013, Calfee et al. 2014, Wang et al. 2014). Ingersoll et al. (2014) reported LC10s represented in Figure 17 by risk quotients ranging from 0.04 to 1.3 (n=6), with values of 1.1 and 1.3 for the youngest age groups exposed: two and 16 days post hatch. The LC20s from the same study ranged from 0.04 to 0.87, with the two and 16-day post-hatch fish having risk quotients greater than 0.8. Ingersoll et al. (2014) and Calfee et al. (2014) also reported acute "effective mortality" EC10s, EC20s, EC50s, and LOECs. Effective mortality in these studies included fish that had died, were presumably moribund because they were immobile, or would not seek cover, making them vulnerable to predation. The effective mortality risk quotients for EC50s for exposures beginning at two, 16, and 30 days post hatch were 1.5, 0.8, and 0.8, respectively (Calfee et al. 2014). The risk quotient for the effective mortality EC50 reported by Ingersoll et al. (2014) was 0.6 for two day post hatch white sturgeon. The EC10s and EC202 from this study indicated risk quotients above one for the two and 16-day post hatch fish. The 30-day post hatch fish EC10 and EC20 risk quotients were 0.8 and 0.7, respectively.

The chronic LC50s were for eight, 25, and 64 day exposures of white sturgeon and had risk quotients of 0.35, 0.92, and 0.74, respectively (Vardy et al. 2011, Ingersoll et al. 2014). Ingersoll et al. (2014) reported LC10s for eight, 14, 24, 28, and 53-day exposures. These are represented in

Figure 17 by risk quotients ranging from 0.5 to 1.1 (n=5), with values of greater than 0.8 for the 1.1 and 1.3 for the eight, 14, 24-day exposures. The chronic LC20s are from several studies (Vardy et al. 2011, Ingersoll et al. 2014, Wang et al. 2014) and are represented by risk quotients ranging from 0.4 to 2.7 (n=7) with three risk quotients greater than 0.8. Risk quotients representing chronic values Ingersoll et al. (2014) reported for effective mortality LOEC, EC10, EC20, and EC50 were 1.2, 1.3, 1.2, and 0.8, respectively.

There were much fewer data for effects on growth and development in white sturgeon. Risk quotients for biomass, length, and weight EC10s ranged from 0.4 to 3.1 (Ingersoll et al. 2014). Exposures of juvenile fish within criterion limits for 28 days affected weight but not length. Weight also decreased in larval fish exposed within criterion limits for 24 days. Risk quotients for length and weight of larval fish exposed for 53 days were 1.7 and 3.1, respectively. The EC20s reported for larval length and weight resulted in risk quotients ranging from 1.2 to 2.4 (Ingersoll et al. 2014, Wang et al. 2014) while risk quotients for juvenile EC20s ranged from 0.4 to 0.6 (Ingersoll et al. 2014, Wang et al. 2014).

The single observation for the effects of copper on reproduction in fish was LOEC providing a risk quotient of 0.4 for fathead minnow egg production (Ouellet et al. 2013). The behavior data for salmonids and other fish include a LOEC for rainbow trout food selection and an IC50 for stimulus avoidance by Chinook salmon. The LOEC did not indicate exposures within criterion limits would influence food consumption (Niyogi et al. 2006). After four and 14 day exposures to copper and varying concentrations of dissolved organic carbon, Chinook salmon tested for their ability to detect and avoid the odorant L-histidine, which is considered an indicator of predator avoidance (Kennedy et al. 2012). In the absence of dissolved organic carbon, the risk quotients representing the IC50s for this response were 2.9 and 2.3 for the 4 and 14-day exposures, respectively.

The screened data for the effects of copper on freshwater invertebrates include growth and development, reproduction, population, and mortality responses from 30 studies reporting data for 405 toxicity tests. Risk quotients for these data appear to be more or less equally distributed above and below the reference lines in Figure 22. The 160 invertebrate acute LC50s provide risk quotients ranging from less than 0.001 to 1,208. The overwhelming majority of these were exposures of *Ceriodaphnia* and *Daphnia* species.

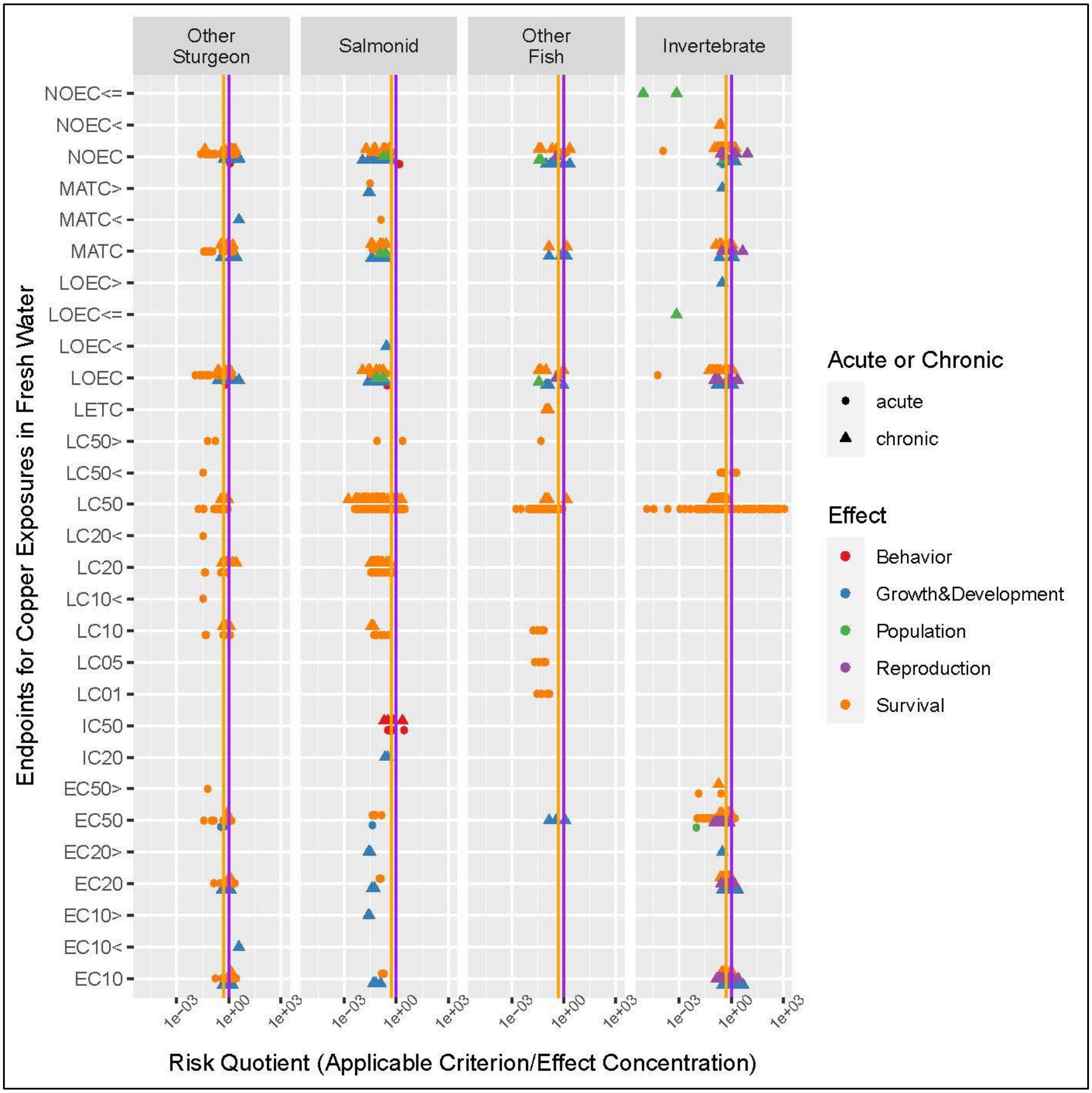

Figure 22. Distribution of risk quotients for freshwater exposures to copper in context of reference lines representing the applicable criterion (purple) and one-half the applicable criterion (orange).

8.2.3.3 Risk of Copper Exposures Within Criteria Limits in Waters Regulated by MassDEP

SURVIVAL

Risk quotients for endpoints representing mortality and effective mortality of early-life-stage of a surrogate species, white sturgeon, would occur within the CMC and CCC limits (Little et al. 2012, Vardy et al. 2013, Calfee et al. 2014, Ingersoll et al. 2014, Wang et al. 2014). While data are not available to perform a population viability analysis for ESA-listed sturgeon populations in Massachusetts waters, these data are important because the viability of these populations are highly sensitive to juvenile mortality resulting in lower numbers of sub-adults recruiting into the adult breeding population (NMFS 1998a, ASSRT 2007).

GROWTH

Growth is an important determinant of survival, and thus recruitment (Anderson 1988, Poletto et al. 2018). While there were much fewer data for effects on growth and development in white sturgeon, data did indicate reduced growth of early-life-stage fish for exposures within the copper CCC limits (Ingersoll et al. 2014, Wang et al. 2014)

The single observation for the effects of copper on reproduction in fish was LOEC providing a risk quotient of 0.4 for fathead minnow egg production (Ouellet et al. 2013). The behavior data for salmonids and other fish include a LOEC for rainbow trout food selection and an IC50 for stimulus avoidance by Chinook salmon. The LOEC did not indicate exposures within criterion limits would influence food consumption (Niyogi et al. 2006). After four and 14-day exposures to copper and varying concentrations of dissolved organic carbon, Chinook salmon tested for their ability to detect and avoid the odorant L-histidine, which is considered an indicator of predator avoidance (Kennedy et al. 2012). In the absence of dissolved organic carbon, the risk quotients representing the IC50s for this response were 2.9 and 2.3 for the four and 14-day exposures, respectively.

REPRODUCTION

The screened data only included one observation for the effects of copper on reproduction in fish. While reproduction is critical to population persistence, fish must first survive and grow in order to reproduce. Given that copper exposures within criteria limits are expected to adversely affect early-life-stage survival and growth, it is reasonable to expect that these effects will, in turn, reduce recruitment of reproductive fish.

ABUNDANCE AND QUALITY OF FORAGE SPECIES

Examination of the data behind the risk quotients presented in Figure 22**.** indicates that adverse effects will occur in invertebrates exposed to copper within the CCC and CMC limits. While the diets of larval shortnose and Atlantic sturgeon have not yet been characterized, there are studies of larval green sturgeon (Zarri and Palkovacs 2019) and larval white sturgeon (Muir et al. 2000) diets. Although diets are likely to be location-specific based on availability, larval stages of both green and white sturgeon were reported to rely on zooplankton species such as copepods, amphipods, and dipterans. An assessment of effects for listed species must address any evidence indicating adverse effects may occur to an individual of that species, but when evaluating effects to forage species, it is the abundance and quality of forage species that is of concern. With respect to the quality of forage species, NMFS does not expect that EPA's approval of the copper CMC and CCC will affect the quality of forage species because, as stated previously, copper generally does not biomagnify in food chains consisting of primary producers, macroinvertebrate consumers, and fish occupying trophic level and higher (Cardwell et al. 2013).

While adverse effects may occur in invertebrates exposed to cadmium within both the freshwater and saltwater CMC and CCC limits, the bulk of the data indicate effects occurring above criteria limits. Early-life-stage sturgeon rely on zooplankton. Risk quotients for freshwater planktonic species ranged from less than 0.001 to 1,208, but these data are dominated by *Ceriodaphnia* and *Daphnia* species, which are used in toxicity tests because they are extremely sensitive to aquatic pollutants. The seven risk quotients for non-*Ceriodaphnia* or *Daphnia* species ranged from 0.002 to 0.13 (Ewell et al. 1986, Markich and Camilleri 1997, Borgmann et al. 2005, Clearwater et al. 2011, Ouellet et al. 2013, Shuhaimi-Othman et al. 2013). Among food items consumed by larger sturgeon, including mollusks, worms, and larger crustaceans like crayfish or crab, risk quotients ranged from less than 0.001 to 4.9 in 24 freshwater species. NMFS does not expect that copper exposures within CCC or CMC limits are likely to affect the abundance or quality of forage for shortnose sturgeon and the Gulf of Maine, New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon.

8.2.3.4 Likely to Adversely Affect Determination for EPA Approval of MassDEP Adoption of Freshwater and Saltwater Cadmium Criteria

NMFS concludes that EPA's approval of MassDEP adoption and implementation of the recommended National Recommended Water Quality Criteria for copper in freshwater is likely to adversely affect shortnose sturgeon and the Gulf of Maine, New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon because:

1) Permitting and monitoring of MassDEP-regulated waters indicate that exposures to copper will occur,
2) The toxicity of copper in surrogate species indicate that exposures within criteria limits will likely result in adverse effects on the survival and growth of early-life-stage fish, and
3) The viability of ESA-listed sturgeon populations in Massachusetts' waters is highly sensitive to juvenile mortality resulting in lower numbers of sub-adults recruiting into the adult breeding population (NMFS 1998a, ASSRT 2007).

9 CUMULATIVE EFFECTS

"Cumulative effects" are those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation (50 CFR §402.02). Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

The future intensity of specific non-Federal activities in the action area is influenced by the difficult-to-predict future economy, funding levels for restoration activities, and individual investment decisions. In addition, the need for communities to adapt to climate change and recover from severe climatic events will influence how wetlands, inland surface waters, and coastal areas are managed. Due to their additive and long-lasting nature, the adverse effects of non-Federal activities that are stimulated by general resource demands and driven by changes in human population density and standards of living, are likely to compound in the future. Specific human activities that may contribute to declines in the abundance, range, and habitats of ESA-listed species in the action area include the following: urban and suburban development, shipping, infrastructure development, water withdrawals and diversion, recreation (including off-road vehicles and boating), and expansion of agricultural and grazing activities (including alteration or clearing of native habitats for domestic animals or crops), and introduction of non-native species which can alter native habitats, out-compete or prey upon native species.

Activities that degrade water quality will continue into the future. These include conversion of natural lands, land use changes from low impact to high impact activities, increases in impervious cover (e.g., Section 6.6), water withdrawals, effluent discharges, the progression of climate change, the introduction of nonnative invasive species, and the introduction of contaminants and pesticides. In particular, many nonpoint sources of pollution, which are not subject to Clean Water Act NPDES permit and regulatory requirements, have proven difficult for states to monitor and regulate. Nonpoint source pollution has been linked to loss of aquatic species' diversity and abundance, fish kills, seagrass bed declines, and toxic algal blooms (Gittings et al. 2013). Nonpoint sources of pollution are expected to increase as the human population continues to grow. Given the challenges of monitoring and controlling nonpoint source pollution and accounting for all the potential stressors and effects on listed species, chronic stormwater discharges will continue to result in aggregate impacts.

9.1 Climate Change

Climate change is discussed in both the environmental baseline section of this Opinion and in the cumulative effects because it is a current and ongoing circumstance that, for the most part, is not subject to consultation, yet influences environmental quality in the action area currently and in the future. As climate change proceeds, precipitation rates will change (Figure 23), and the frequency of heavy rainfall events will increase (Figure 24), leading to increased flooding and erosive flows resulting in unmanaged pollutant discharges and redistribution of legacy pollutants in sediments.

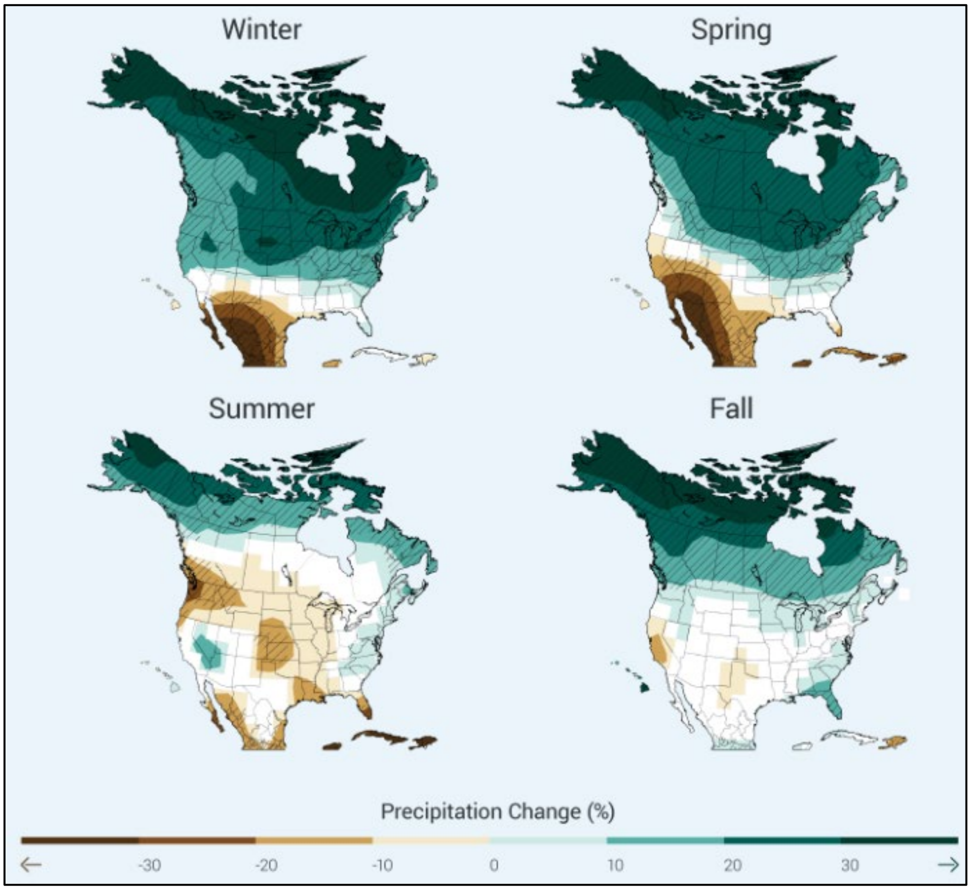

Figure 23. Seasonal precipitation change for 2071-2099 (compared to 1970-1999).[17]

[17] Assumes existing emissions rate increases. Hatched areas are projected changes that are significant and consistent among models, unhatched areas indicate projected changes do not differ from natural variability. (Figure source: NOAA NCDC / CICS-NC). http://nca2014.globalchange.gov/report/our-changing-climate/precipitation-change

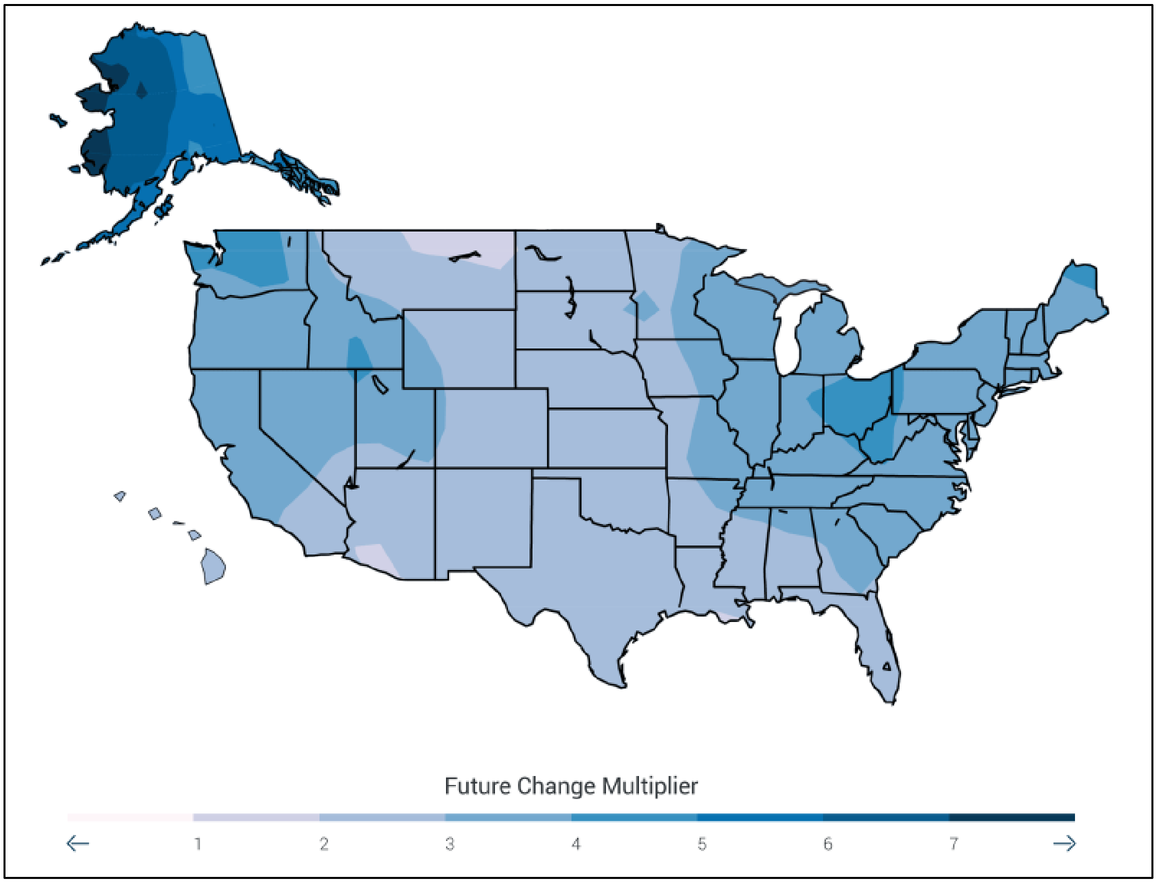

Figure 24. Increase in frequency of extreme daily precipitation events for 2081-2100 (compared to 1981-2000).[18]

[18] http://nca2014.globalchange.gov/report/our-changing-climate/precipitation-change

10 INTEGRATION AND SYNTHESIS

The *Integration and Synthesis* section is the final step in our assessment of the risk posed to species and critical habitat because of implementing the action. In this section, we add the *Effects of the Action* (Section 8) to the *Environmental Baseline* (Section 6) and the *Cumulative Effects* (Section 9) to formulate the agency's biological opinion as to whether the proposed action is likely to reduce appreciably the likelihood of both the survival and recovery of a ESA-listed species in the wild by reducing its numbers, reproduction, or distribution. This assessment is made in full consideration of the *Status of the Species Likely to be Adversely Affected by the Action* (Section 5.2). Populations that occur in the Merrimack and Connecticut Rivers are of primary concern for this action because the criteria determined likely to adversely affect ESA-listed sturgeon are proposed only for Massachusetts.

Some ESA-listed species and designated critical habitat are located within the action area but the effects of the action on these ESA resources were determined to be insignificant or discountable and thus not likely to adversely affect these resources. Some activities evaluated individually were determined to have insignificant effects or discountable effects and thus to be not likely to adversely affect some ESA-listed species and designated critical habitat (Sections 5.1 and 8.1).

The following discussions provide an overview of the findings of this opinion and a Jeopardy Analysis that summarizes the probable risks the proposed action poses to shortnose surgeon and the Atlantic sturgeon Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs. These summaries integrate the exposure profiles presented previously with the results of our response and risk analyses (Section 8) for each of the water quality criteria considered further in this Opinion.

10.1 Overview

This Opinion concluded that EPA approval of MassDEP adoption and implementation of Nationally Recommended Freshwater Criteria for aluminum, cadmium, and copper is likely toadversely affect early life stage and young of year shortnose sturgeon and the Gulf of Maine, New York Bight/migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon that spawn within Massachusetts' rivers. The viability of ESA-listed sturgeon populations in Massachusetts' waters is highly sensitive to juvenile mortality resulting in lower numbers of sub-adults recruiting into the adult breeding population (NMFS 1998a, ASSRT 2007).

Poor water quality in these rivers contributes to the stressor scores for shortnose sturgeon (Section 5.2.2). If current monitoring data on aluminum, cadmium, and copper in Sturgeon Waters were available, it could indicate whether baseline conditions attenuate the concern that the criteria concentrations are not sufficiently protective. When revised criteria are more protective than those currently applied to discharge permits, and solid monitoring information indicates that baseline instream concentrations are below effects thresholds, then it is reasonable to expect more stringent criteria applied to permits would not result in exposures above those thresholds. In the absence of that information, NMFS gives the species the benefit of the doubt.

For aluminum, historical data indicate concentrations exceeding the proposed default aluminum watershed chronic criteria for the Connecticut and Merrimack rivers. The only current aluminum data is from the Merrimack River and the value exceeded that river's criterion. The available monitoring data for cadmium in Sturgeon Rivers are at or below CCC criterion limits at ambient water hardness. However, nearly all of the available data for Sturgeon Waters were collected before 2005. Copper monitoring data are also dominated by older sampling. Copper data indicate a distinct seasonal influence on copper bioavailability, influencing calculated copper criteria as shown in Figure 20. Lower criteria values occur during spring, just as spawning begins. Since the criteria will be derived using systematically collected data to identify criteria at the tenth percentile for most waters and fifth percentile for waters where ESA-listed aquatic species occur, the criterion may be sufficiently protective in the summer and fall, when aquatic carbon levels are higher. Harmful effects become likely as aquatic chemistry conditions converge on the fifth percentile criterion conditions at about the same time sturgeon are spawning and hatching.

In all cases, monitoring for aluminum, cadmium, and copper was conducted using sufficiently sensitive analytical methods. It is necessary to verify this because, under the Clean Water Act, the use of sufficiently sensitive analytical methods is a requirement for NPDES permitting, but not for 305(b) monitoring. If and when monitoring occurs for the purposes of identifying impairments, NMFS expects that aquatic impairments by aluminum, cadmium, and copper would be detected.

Current water quality impairments in Sturgeon Waters are attributed to nutrients and indicator bacteria (Section 6.3). Exposures of shortnose and Atlantic sturgeon to aluminum, cadmium, and copper are likely to occur through stormwater and snowmelt runoff and discharges from facilities that use either these metals or treat waste containing these metals (Figure 14, Figure 16, and Figure 20). Under section 402 of the Clean Water Act, an NPDES permit will require monitoring for substances if there is a reasonable potential that the discharge would result in pollutant levels that would impair the designated use of the receiving water (40 CFR 122.44(d)(1)).

Massachusetts has not been delegated permitting authority under the Clean Water Act so, when EPA issues NPDES permits, NMFS consults on them individually. Criteria implemented by NPDES programs are in place indefinitely and are applied to multiple sources within a watershed, thus there is an aggregate impact to EPA's approval of the criteria that is not addressed when permits are reviewed on an individual basis. For example, NMFS' concurrence with EPA's not likely to adversely affect determination for a discharger discharging aluminum above permit limits to the Chicopee River was based on the degree to which the discharge would be rapidly diluted. That specific discharge, and its effects, were determined to be insignificant. Yet, there are also 14 other discharges to catchments adjacent to the Connecticut River that are required to monitor for aluminum. Meanwhile, a majority of available historical data for ambient aluminum data in the Connecticut River indicates that half of the total recoverable aluminum concentrations exceeded the proposed default aluminum watershed CCC of 290 µg/L.

In the absence of solid monitoring information, data collected *after* implementation of revised criteria for *all contributing sources* may or may not indicate actual instream concentrations below effects thresholds. Often the constituents monitored for are selected based on what is likely to be present given local land usage and industries. For example, if sampling in the Everglades, one might monitor for nutrients and sugarcane pesticides, but not industrial chemicals. Sources for aluminum, cadmium, and copper exist along sturgeon waters. The paucity of recent monitoring data is not reassuring given that a number of permitted dischargers fail to submit discharge monitoring reports, which can mask significant problems such as the inability to meet permit limits.

The analyses in section 8.2 establish that early-life-stage shortnose sturgeon and Atlantic sturgeon are likely to be exposed to aluminum, cadmium, and copper in Massachusetts Sturgeon Waters and that adverse effects are expected to occur in early-life-stage exposed to these metals within their respective criteria limits. The majority of the monitoring data are historical and may not reflect current conditions.

10.2 Jeopardy Analysis

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

10.2.1 Shortnose Sturgeon

Whether the potential effects to reproductive output would appreciably reduce the likelihood of survival of shortnose sturgeon in the wild depends on the probable effect the changes in reproductive output would have relative to current population sizes and trends. The most recent population estimates available for the species indicate that the largest shortnose sturgeon adult populations are found in the Northeastern rivers: the Hudson with 56,708 adults (Bain et al. 2007); the Delaware, 12,047 (ERC 2006); and the Saint Johns, > 18,000 adults (Dadswell 1979). Age-structured population modeling for the Hudson, Cooper, and Altamaha Rivers indicated an extinction risk of zero, but estimated probability of a 50% decline was relatively high and the probability of an 80% decline was low (Hudson 0.09, Cooper 0.01, Altamaha 0.23 SSSRT 2010).

Shortnose sturgeon spawning has been confirmed in the Merrimack, but the population size is estimated to be less than 100 adults, which is considerably higher than the early 1990s. This population is subject to periodic industrial and sewage releases during flood conditions and dissolved oxygen concentrations declining to below minimum thresholds during periods of drought or low flow. A small (1,242-1,580 adults) but stable spawning population of shortnose sturgeon occurs in the Connecticut River and all life stages are present, potentially serving as a source of recruits to other nearby rivers. Water quality is also a source of stress in the Connecticut River, with high PCBs known to occur in fish tissues, and coal tar deposits present below the Holyoke Dam are a potential source of metal exposure (Gao et al. 2016).

The overall trend in shortnose sturgeon populations in Massachusetts is not considered a major concern because the Connecticut River population is thought to be stable and the population estimate sampling of the Merrimack River in the winter of 2009 suggested significantly higher estimates than 20 years previously (SSSRT 2010). Taking existing pollutant exposures within these rivers into consideration, we believe the proposed action is not reasonably expected to cause an appreciable reduction in the likelihood of survival of shortnose sturgeon in the wild. The reduction in fitness of shortnose sturgeon that could occur because of effects to early life stage and young of year individuals is not expected to appreciably affect the overall reproductive output of shortnose sturgeon populations in Massachusetts.

The 1998 recovery plan identifies 19 population segments within their range with a goal of each segment maintaining a minimum population size to maintain genetic diversity and avoid extinction (NMFS 1998a). The recovery tasks for the Merrimack and Connecticut River that are relevant to the impacts of the proposed action include analyzing contaminant loads in sturgeon tissue and habitat, determining effects of contaminants on sturgeon fitness, and identifying contaminant sources and reducing contaminant loading. These are classified as Priority 2 tasks, which are actions "that must be taken to prevent a significant decline in population numbers, habitat quality, or other significant negative impacts short of extinction."

In the Merrimack River, shortnose sturgeon occur up to the Essex Dam, at rkm 46 and spawn near Haverhill at rkm 30 to 32 (Kieffer and Kynard 1996). Larvae begin moving downstream four weeks after the spawning and continue to develop in the freshwater reach of the river (rkm 16 to 32, Kieffer and Kynard 1993). Foraging concentrations are reported near Amesbury and the lower islands (rkm 6 to 24, Kieffer and Kynard 1993, Kynard et al. 2000). Merrimack River sturgeon overwinter from late fall to early spring above the salt wedge at rkm 15 to 29 (Kieffer and Kynard 1993, Wippelhauser et al. 2015).

The 2010 status review indicates that the Connecticut River shortnose sturgeon population is impeded, but not isolated, by the Holyoke dam. Connecticut River shortnose sturgeon occur within the mainstem up to Turners Falls Dam (rkm 198) within the Westfield River and Deerfield River tributaries. Spawning occurs below Turners Falls Dam/Cabot Station at rkm 193 to 194 or, when conditions are favorable, below the Holyoke Dam at rkm 139 to 140 (Kynard et al. 2012a). Offspring drift downstream for up to 20 km such that early-life-stages would be present in downstream freshwater reaches from rkm 13 to 194 (Buckley and Kynard 1981, Kynard et al. 2012b). Foraging and overwintering concentrations are reported from above the Holyoke Dam in Deerfield Concentration Area at rkm 144 to 192 (Kynard et al. 2012b), Agawam at rkm 114 to 119 (Buckley and Kynard 1985b), and the lower Connecticut from rkm 0 to 110 (Kynard et al. 2012b). In addition adults may occur in the Deerfield River up to Shelburne Falls at rkm 22.5 and larvae can drift into the Deerfield River under certain flow conditions (Kieffer and Kynard 1993, Kynard et al. 2012b). Foraging may occur from spring through fall in lower Deerfield River from rkm 0 to 3.5 (Kynard et al. 2012b). The Deerfield River also can be used for overwintering potentially for pre to spawning staging for adults (Kynard et al. 2016). Adults are also assumed to forage in the Westfield River up to the Decorative Specialties International Dam at rkm 9.5 (SSSRT 2010).

The anticipated take of shortnose sturgeon from the effects of implementing the water quality criteria for aluminum, cadmium, and copper is not likely to reduce population numbers of the species over time given current populations sizes and expected recruitment. Thus, the proposed action is not likely to impede the applicable recovery objective for shortnose sturgeon and will not result in an appreciable reduction in the likelihood of the recovery of this species in the wild. We conclude the proposed action is not likely to jeopardize the continued existence of shortnose sturgeon in the wild.

10.2.2 Atlantic Sturgeon

Whether the potential effects to reproductive output would appreciably reduce the likelihood of survival of Gulf of Maine and New York Bight DPSs of Atlantic sturgeon with designated critical habitat in the action area, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon in the wild depends on the probable effect the changes in reproductive output would have relative to current population sizes and trends.

In the absence of quantitative population estimates of Atlantic sturgeon DPSs, the Atlantic States Saltwater Fisheries Commission considers qualitative criteria such as the appearance of Atlantic sturgeon in rivers where they were not documented in recent years, discovery of spawning adults in rivers they had not been documented in before, and increases in anecdotal interactions. However, qualitative metrics may be the result of increased research and attention, not a true increase in abundance (ASMFC 2017). All DPSs of Atlantic sturgeon are considered depleted. All DPSs of Atlantic sturgeon are highly vulnerable to climate change due to their low likelihood to change distribution in response to current global climate change will also expose them to effects of climate change on estuarine habitat such as changes in the occurrence and abundance of prey species in currently identified key foraging areas (NMFS 2022b, a).

Populations that occur in the Merrimack and Connecticut Rivers are of primary concern for this action because the criteria determined likely to adversely affect Atlantic sturgeon are proposed only for Massachusetts. Recent data indicate that spawners in the Connecticut River are more closely related to the Chesapeake Bay, Carolina, and South Atlantic DPSs, so the action potentially affects the persistence and recovery of these DPSs as well.

The NMFS 2022 status assessment for the Gulf of Maine DPS reports that this DPS has low abundance, and that the current numbers of spawning adults are one to two orders of magnitude smaller than historical levels. The status of the DPS has likely neither improved nor declined from what it was when we listed the DPS in 2012. The Kennebec River remains the only known spawning population for the Gulf of Maine DPS despite the availability of suitable spawning and rearing habitat in other Gulf of Maine rivers. The estimated effective population size is less than 70 adults, which suggests a relatively small spawning population. It is currently the DPS with only one known spawning population (NMFS 2022a). Based on the Stock Assessment, there is a 51% probability that abundance of the Gulf of Maine DPS has increased since implementation of the 1998 fishing moratorium but also a relatively high likelihood (74% probability) that mortality for the Gulf of Maine DPS exceeds the mortality threshold used for the Stock Assessment (ASMFC 2017). However, Atlantic sturgeon are data poor, in general, and among the DPSs, the Gulf of Maine DPS is very data poor.

The NMFS 2022 status assessment for the New York Bight DPS reports that this DPS has low abundance, and that the current numbers of spawning adults are one to two orders of magnitude smaller than historical levels (NMFS 2022b). There is a relatively high probability (75%) that the New York Bight DPS abundance has increased since the implementation of the 1998 fishing moratorium, and a relatively high probability (69%) that mortality for the New York Bight DPS does not exceed the mortality threshold used for the Stock Assessment (ASMFC 2017). However, these conclusions primarily reflect the status and trend of only the Hudson River spawning population and not the Connecticut River population. The 2017 stock assessment compared the 1998 and 2015 relative abundance index values and found that the Gulf of Maine and Chesapeake Bay DPSs were below their 1998 values while the New York Bight and Carolina DPS, as well as the coastwise stock, were above their 1998 values. The South Atlantic DPS could not be evaluated due to lack of adequate data to estimate a relative abundance index. All of the DPSs showed qualitative signs of improving populations such as increased presence of Atlantic sturgeon, including in rivers where species interactions had not been reported in recent years, and the discovery of spawning in rivers where it had not been previously documented (ASMFC 2017).

The overall trend in the Gulf of Maine DPS of Atlantic sturgeon in Massachusetts is not considered a major concern because the status has likely neither improved nor declined from what it was when listed in 2012 and there is a 51% probability that abundance of the Gulf of Maine DPS has increased since implementation of the 1998 fishing moratorium. The overall trend in the New York Bight DPS of Atlantic sturgeon in Massachusetts is not considered a major concern because the status has likely neither improved nor declined from what it was when listed in 2012 and there is a 75% probability that abundance of the New York Bight DPS has increased since implementation of the 1998 fishing moratorium. Because data indicate that individuals related to the Chesapeake Bay, Carolina, and South Atlantic DPSs are spawning in the Connecticut River, we consider implications of the action for these DPS not to be a major concern because all Atlantic sturgeon DPSs showed qualitative signs of improving populations.

Therefore, we conclude the effects of the proposed action are not likely to impede the survival of Atlantic sturgeon DPSs in the wild.

A recovery plan has not been completed for the listed Atlantic sturgeon DPSs. However, a recovery outline has been prepared to guide recovery efforts until a full recovery plan is developed and approved. The stated goal of the recovery outline is that subpopulations of all five Atlantic sturgeon DPSs must be present across the historical range at sufficient size and genetic diversity to support successful reproduction and recovery from mortality events, with increases in the recruitment of juveniles to the sub-adult and adult life stages to be maintained over many years. The outline includes a recovery action to implement region-wide initiatives to improve water quality in sturgeon spawning rivers, with specific focus on eliminating or minimizing human-caused anoxic zones.

The Gulf of Maine DPS of Atlantic sturgeon occurs in the Piscataqua River Watershed, including the Salmon Falls and Cocheco tributaries up to the confluence with the Salmon Falls and Cocheco Rivers (rkm 15). This includes Great Bay, Salmon Falls River up to the Route 4 at the South Berwick Dam at rkm 7, and the Cocheco River up to the Cocheco Falls Dam at rkm 6. Spawning potentially occurs in the Salmon Falls and Cocheco Rivers based on habitat features necessary to support reproduction and recruitment and the capture of an adult female Atlantic sturgeon in spawning condition in 1990. Juveniles are potentially present throughout the rivers year-round with adults using these waters for foraging and resting during spring and fall migrations (82 FR 39160 ASSRT 2007). Atlantic sturgeon occur in the Merrimack River up to the Essex Dam at rkm 46 and are often found foraging around the lower islands reach at rkm 3-12 and the mouth of the river (Kieffer and Kynard 1993, Kynard et al. 2000). Spawning potentially occurs due to the presence of features necessary to support reproduction and recruitment, and data suggest these waters are used as a nursery for juveniles (82 FR 39160 ASSRT 2007). Based on reported sightings, adult and juvenile Atlantic sturgeon may occur within Boston Metro area waters, foraging up to Charles River Locks at rkm 5.5 and up to Dam #1 on the North River to Indian Head Reservoir at Luddam's Ford at rkm 21. Subadult and adult Atlantic sturgeon also forage in Narragansett Bay and the Taunton River up to the convergence of the Town River and Matfield River (Burkett and Kynard 1993, ASSRT 2007).

The New York Bight DPS of Atlantic sturgeon ranges from the Hudson River to the Delaware River, including the Connecticut River. The Connecticut River is designated critical habitat for this DPS of Atlantic Sturgeon. Atlantic sturgeon may occur in the Connecticut River up to the Holyoke Dam in Massachusetts at rkm 140, but mainly stay in the summer range of the salt wedge at rkm 0-26 within Connecticut (Savoy and Shake 1992, Savoy and Pacileo 2003). The capture of 45 pre-migratory juvenile Atlantic sturgeon in the lower Connecticut River provides strong evidence that natural reproduction occurs in the upper reaches of the river. The DPS designation for this population is complicated because genetic analysis indicates that these individuals were more genetically similar to the Chesapeake Bay, Carolina, and South Atlantic DPSs than the nearby New York Bight or Gulf of Maine DPSs (Savoy et al. 2017).The anticipated take of ESA-listed Atlantic sturgeon from the effects of implementing the water quality criteria for aluminum, cadmium, and copper is not likely to reduce population numbers of the species over time given current populations sizes and expected recruitment. Thus, the proposed action is not likely to impede the applicable recovery objectives for the Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon and will not result in an appreciable reduction in the likelihood of the recovery of this species in the wild. We conclude the proposed action is not likely to jeopardize the continued existence of Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon in the wild.

11 CONCLUSION

After reviewing the current status of the ESA-listed species, the environmental baseline within the action area, the effects of the proposed action, and cumulative effects, it is NMFS' biological opinion that the proposed action is likely to adversely affect, but is not likely to jeopardize the continued existence of shortnose sturgeon or the Gulf of Maine, New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon in Massachusetts.

For the states of New Hampshire and Massachusetts, the proposed action is not likely to adversely affect fin whale, North Atlantic right whale, sei whale, green turtle (North Atlantic DPS), Kemp's ridley turtle, leatherback turtle, or loggerhead turtle (Northwest Atlantic Ocean DPS) or critical habitats designated for the North Atlantic right whale or Gulf of Maine DPS of Atlantic sturgeon. For those criteria proposed for the state of New Hampshire, the action is also not likely to adversely affect shortnose sturgeon, Gulf of Maine, New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon.

12 Incidental Take Statement

Section 9 of the ESA and Federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. "Take" is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by regulation to include significant habitat modification or degradation that results in death or injury to ESA-listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering (see 50 CFR §222.102).

Incidental take is defined as take that results from, but is not the purpose of, carrying out an otherwise lawful activity (see 50 CFR §402.02). Section 7(b)(4) and section 7(o)(2) of the ESA provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA if that action is performed in compliance with the terms and conditions of this incidental take statement.

Exposures of shortnose sturgeon and Gulf of Maine, New York Bight, and migrating Chesapeake Bay, Carolina, and South Atlantic DPSs of Atlantic sturgeon to aluminum, cadmium, and copper within criteria limits in the action area is likely to result in incidental take due to the reductions in survival of early life stage fish and fitness of these species.

12.1 Amount or Extent of Take

Section 7 regulations require NMFS to specify the impact of any incidental take of endangered or threatened species; that is, the amount or extent, of such incidental taking on the species (50 CFR §402.14(i)(1)(i)). The amount of take represents the number of individuals that are expected to be taken by actions while the extent of take specifies the impact, i.e., the amount or extent of such incidental taking on the species, which may be used if we cannot assign numerical limits for animals that could be incidentally taken during the course of an action (see 80 FR 26832).

Where it is not practical to quantify the number of individuals that are expected to be taken by the action, a surrogate (e.g. similarly affected species or habitat or ecological conditions) may be used to express the amount or extent of anticipated take (50 CFR §402.14(i) (1) (i)). To use a surrogate we must describe the causal link between the surrogate and take of the listed species, explain why it is not practical to express the amount or extent of anticipated take or to monitor take-related impacts in terms of individuals of the listed species, and set a clear standard for determining when the level of anticipated take has been exceeded

Incidental take under the proposed aluminum, cadmium, or copper criteria cannot be accurately quantified or monitored as a number of individuals because the action area includes all waters of Massachusetts and data do not exist that would allow us to quantify how many individuals of each species and life stage exist in affected waters, especially considering that the numbers of individuals vary with environmental conditions, and changes in population size due to recruitment and mortality. In addition, currently we have no means to detect or determine which impairments to reproduction, development, and growth are due to the water quality within criteria limits versus other natural and anthropogenic environmental stressors. Because we cannot quantify the amount of take, we will use the regulatory application of the criteria as a measure reflecting the potential for harmful exposures to aluminum, cadmium, and copper for the extent of authorized take as a surrogate for the amount of authorized take.

Further, NMFS cannot precisely predict the number of shortnose sturgeon and Atlantic sturgeon that are reasonably certain to demonstrate behavioral and injurious effects due to the presence of aluminum, cadmium, and copper within criteria limits. Also, there is no feasible way to count, observe, or determine the number of individuals of each species that would be affected by exposures because the effects of the action will occur over a large geographic area and effects may occur in areas where animals are not likely to be observed due to water depth. Even if affected animals are observed, it is unlikely that the exact cause of injury, mortality or behavioral effects could be determined.

For the reasons discussed above, the specified amount or extent of incidental take of ESA-listed shortnose and Atlantic sturgeon species requires that MassDEP's intended level of protection is met, as confirmed through the terms and conditions specified in this incidental take statement. The amount or extent of incidental take applies only to exposures when waters are monitored using sufficiently sensitive analytical methodology as defined in the 122.44(i)(1)(iv) of the Clean Water Act. Effects of the proposed action could manifest later in time and those discharges for which reasonable potential, monitoring requirements, and discharge limits are determined using sufficiently sensitive analytical methodology. If sufficiently sensitive analytical methodology is not applied, it will not be not possible to confirm whether MassDEP's intended level of protection is met. NMFS expects that, upon identification, MassDEP and EPA will address any noncompliance with 40 CFR Part 136. This reflects MassDEP's and EPA's intended level of protection for aquatic life and ensures that exceedances will be detected and addressed, thereby minimizing take.

12.2 Reasonable and Prudent Measures

"Reasonable and prudent measures" are measures that are necessary or appropriate to minimize the impact of the amount or extent of incidental take. (50 CFR 402.02). Section 7(b)(4) of the ESA requires that when a proposed agency action is found to be consistent with section 7(a)(2) of the ESA and the proposed action may incidentally take individuals of ESA-listed species, NMFS will issue a statement that specifies the impact of any incidental taking of endangered or threatened species. To minimize such impacts, reasonable and prudent measures, and term and conditions to implement the measures, must be provided. Only incidental take resulting from the agency actions and any specified reasonable and prudent measures and terms and conditions identified in the incidental take statement are exempt from the taking prohibition of section 9(a), pursuant to section 7(o) of the ESA.

RPMs are defined by regulation as: "those actions the Director believes necessary or appropriate to minimize the impacts, i.e., amount or extent, of incidental take" (50 CFR 402.02). NMFS believes the RPMs described below are necessary and appropriate to minimize the impacts of incidental take on threatened and endangered species resulting from exposure to aluminum, cadmium, or copper within criteria limits:

1) EPA Region 1, Water Division will work within its authorities to ensure that the implementation of water quality standards for aluminum, cadmium, and copper adopted by Massachusetts minimizes aggregate adverse effects to ESA-listed species and designated critical habitat under NMFS' jurisdiction.
2) EPA Region 1 will ensure that persons applying EPA-approved standards in regulatory actions and those who are subject to regulations applying EPA-approved standards are aware of the prohibition of take of ESA-listed species under section 9 of the ESA and where ESA-listed species under NMFS' jurisdiction occur.

Terms and Conditions for RPM 1:

In order to be exempt from the prohibitions of section 9 of the ESA, the Federal action agency must comply with the following terms and conditions. The EPA Region 1, Water Division shall achieve RPM 1 through use of the revised criteria in NPDES permits for new sources and existing NPDES permits upon renewal, providing guidance to MassDEP, and participating in sustained attention to water quality within waters where Atlantic and shortnose sturgeon occur. Specifically:

1) The EPA Water Division will notify the MassDEP and EPA-Region 1 NPDES Permit Branch of: 1) updated water quality criteria for copper, cadmium, and aluminum, and 2) the importance of compliance with permit limits based on such criteria in all NPDES permits, including general permits, to protect threatened and endangered species, including the Atlantic and shortnose sturgeon.
2) EPA Guidance to MassDEP:
    a) EPA will inform the state that the fifth percentile shall be used when calculating facility-specific aluminum criteria for discharges to Deerfield River. This is consistent with MassDEP's "Fresh Water Aquatic Life Water Quality Criteria for Aluminum: Application of the Aluminum Criteria Calculator for National Pollutant Discharge Elimination System (NPDES) and Massachusetts Surface Water Discharge (SWD) Permits (December, 2021)" which states "If there are endangered species (as defined in the federal Endangered Species Act or Massachusetts Endangered Species Act) within the watershed, the 5th percentile shall be used." (Section 9.0, Site-Dependent Criteria Development).
    b) EPA will strongly encourage MassDEP to monitor aluminum, cadmium, and copper in areas where ESA-listed Atlantic and shortnose sturgeon occur.

c) If EPA becomes aware of new information that indicates revisions to criteria subject to this consultation may be necessary to protect threatened and endangered species, EPA will work with Massachusetts regulatory authorities to revise water quality standards or take other actions, as appropriate.

3) Baseline Water Quality Review

a) Within 6 months of the signature of the Biological Opinion, EPA will collaborate with NMFS on the development of a baseline water quality condition tool for those stressors addressed in this consultation in waters where Atlantic and shortnose sturgeon occur.

b) Thereafter, EPA will meet with NMFS at least annually to review water quality conditions for those stressors addressed in this consultation potentially affecting Atlantic and shortnose sturgeon and discuss changes in water quality, gaps in information regarding water quality, and approaches to resolving those gaps.

Terms and conditions for RPM 2:

1) EPA Region 1 Water Division will support other EPA Region 1 branches applying EPA-approved criteria subject to this consultation in providing notice of EPA's obligations under the ESA in its communications, as appropriate, including, but not limited to, 303(c) decision letters, NPDES permit development and decisions, permit application materials, training, and/or informational websites. Such notice shall contain the following:

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

b) Take of ESA-listed endangered species is prohibited under section 9 of the ESA, and these prohibitions apply to all individuals, organizations, and agencies subject to United States jurisdiction. These take prohibitions have also been extended to the Gulf of Maine DPS of Atlantic Sturgeon under section 4(d) of the ESA (50 CFR §223.211).

c) "Take" is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct 16 U.S.C. 1532(19). "Harm" for purposes of the ESA is further defined by regulation to mean "an act which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation which actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns, including, breeding, spawning, rearing, migrating, feeding or sheltering" 50 CFR §222.102.

d) Endangered shortnose sturgeon, threatened Gulf of Maine Atlantic sturgeon, and the endangered New York Bight, Chesapeake Bay, South Atlantic, and Carolina DPSs of Atlantic sturgeon may spawn, migrate, and forage within accessible inland rivers, estuaries, and coastal waters from Canada to Florida. Poor water quality is among the most significant threats to the species due to harm to offspring development. Sensitive early life stages may occur in the following waters of Massachusetts: Merrimack River, Connecticut River, Westfield River, and Deerfield River.

13 CONSERVATION RECOMMENDATIONS

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

The following conservation recommendations are discretionary measures that NMFS believes are consistent with this obligation and therefore should be carried out by the EPA:

1. EPA should encourage MassDEP and NHDES to conduct water quality monitoring for pollutants of emerging concern, particularly endocrine disruptors and PFAS to determine whether the MassDEP and NHDES should include criteria to protect ESA resources from exposures to these pollutants.
2. EPA should courage MassDEP and NHDES to monitor for legacy nonylphenol contamination the sediments of likely sources.
3. EPA should encourage its EPA Region 1 Enforcement and Compliance Assurance Division to consider notifying NMFS' Northeast Division Office of Law Enforcement of chronic permit limit violations for discharges to Sturgeon Waters.

In order for NMFS' Office of Protected Resources Interagency Cooperation Division to be kept informed of actions minimizing or avoiding adverse effects on, or benefiting, ESA-listed species or their critical habitat, the EPA should notify the Interagency Cooperation Division of any conservation recommendations they implement whether in their final action or separate from the proposed action of approving the water quality criteria evaluated during this consultation.

14 REINITIATION NOTICE

This concludes consultation on EPA approval of water quality criteria for the States of New Hampshire and Massachusetts. Consistent with 50 CFR §402.16(a), reinitiation of formal consultation is required and shall be requested by the Federal agency, where discretionary Federal involvement or control over the action has been retained or is authorized by law and:

(1) The amount or extent of taking specified in the incidental take statement is exceeded;

(2) New information reveals effects of the agency action that may affect ESA-listed species or critical habitat in a manner or to an extent not previously considered;

(3) The identified action is subsequently modified in a manner that causes an effect to the ESA-listed species or critical habitat that was not considered in this Opinion; or

(4) A new species is listed or critical habitat designated under the ESA that may be affected by the action.

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