UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
West Coast Region
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Refer to NMFS No: WCRO-2020-00007

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June 30, 2020

Hanh Shaw
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U.S. Environmental Protection Agency
Region 10
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Re: Endangered Species Act Section 7(a)(2) Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Response for the proposed EPA promulgation of freshwater aquatic life criteria for aluminum in Oregon

Dear Ms. Shaw:

Thank you for your letter of January 2, 2020, requesting initiation of consultation with NOAA's National Marine Fisheries Service (NMFS) pursuant to section 7 of the Endangered Species Act of 1973 (ESA) (16 U.S.C. 1531 et seq.) for the U.S. Environmental Protection Agency (EPA) proposal to promulgate freshwater aquatic life criteria for aluminum in Oregon. This consultation was conducted in accordance with the 2019 revised regulations that implement section 7 of the ESA (50 CFR 402, 84 FR 45016). Thank you, also, for your request for consultation pursuant to the essential fish habitat (EFH) provisions in Section 305(b) of the Magnuson-Stevens Fishery Conservation and Management Act (MSA)(16 U.S.C. 1855(b)) for this action.

In your request, EPA asked NMFS to concur with their determination that the proposed action is not likely to adversely affect the Southern Distinct Population Segment (sDPS) of green sturgeon (*Acipenser medirostris*). NMFS did not concur with the EPA's determination. Rather, NMFS determined the proposed action may affect, and is likely to adversely affect the sDPS green sturgeon. Accordingly, we address sDPS of green sturgeon and our rationale for our non-concurrence in the biological opinion (Opinion) portion of the document.

In this Opinion, NMFS concludes that the action, as proposed, is not likely to jeopardize the continued existence of the following 18 species or result in the destruction or modification of their critical habitats:

1. Lower Columbia River Chinook salmon (*Oncorhynchus tshawytscha*)

2. Upper Willamette River Chinook salmon
3. Upper Columbia River Spring-run Chinook salmon
4. Snake River Spring/summer-run Chinook salmon
5. Snake River Fall Chinook salmon
6. Columbia River Chum salmon (*O. keta*)
7. Lower Columbia River Coho salmon (*O. kisutch*)
8. Oregon Coast Coho salmon
9. Southern Oregon/Northern California Coast Coho salmon
10. Snake River Sockeye salmon (*O. nerka*)
11. Lower Columbia River Steelhead (*O. mykiss*)
12. Upper Willamette River Steelhead
13. Middle Columbia River Steelhead
14. Upper Columbia River Steelhead
15. Snake River Basin Steelhead
16. sDPS Eulachon (*Thaleichthys pacificus*)
17. sDPS Green Sturgeon
18. Southern Resident Killer Whale (*Orcinus orca*)

As required by section 7 of the ESA, NMFS provides an incidental take statement (ITS) with the Opinion. The ITS describes reasonable and prudent measures (RPM) NMFS considers necessary or appropriate to minimize the impact of incidental take associated with this action. The take statement sets forth nondiscretionary terms and conditions, including reporting requirements, that the EPA and any permittee who performs any portion of the action must comply with to carry out the RPM. Incidental take from actions that meet these terms and conditions will be exempt from the ESA take prohibition.

This document also includes the results of our analysis of the action's effects on essential fish habitat (EFH) pursuant to section 305(b) of the Magnuson-Stevens Fishery Conservation and Management Act (MSA), and includes three Conservation Recommendations to avoid, minimize, or otherwise offset potential adverse effects on EFH. These Conservation Recommendations include a subset of the ESA Terms and Conditions. Section 305(b)(4)(B) of the MSA requires federal agencies provide a detailed written response to NMFS within 30 days after receiving these recommendations.

If the response is inconsistent with the EFH Conservation Recommendations, the EPA must explain why the recommendations will not be followed, including the justification for any disagreements over the effects of the action and the recommendations. In response to increased oversight of overall EFH program effectiveness by the Office of Management and Budget, NMFS established a quarterly reporting requirement to determine how many Conservation Recommendations are provided as part of each EFH consultation and how many are adopted by the action agency. Therefore, in your statutory reply to the EFH portion of this consultation, NMFS asks that you clearly identify the number of Conservation Recommendations accepted.

Please contact Johnna Sandow, Fish Biologist in the Southern Snake Branch, at (208) 378-5737 or at johnna.sandow@noaa.gov if you have any questions concerning this consultation, or if you require additional information.

Sincerely,

Michael Tehan
Assistant Regional Administrator
Interior Columbia Basin Office

Enclosure

cc: M. Jankowski – EPA
R. Labiosa – EPA
P. Henson – USFWS
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Endangered Species Act (ESA) Section 7(a)(2) Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Response for the

Federal Promulgation of Freshwater Aluminum Aquatic Life Criteria in Oregon

NMFS Consultation Number: 2020-00007

Action Agency: U.S. Environmental Protection Agency

Affected Species and NMFS' Determinations:

ESA-Listed Species	Status	Is Action Likely to Adversely Affect Species?	Is Action Likely To Jeopardize the Species?	Is Action Likely to Adversely Affect Critical Habitat?	Is Action Likely To Destroy or Adversely Modify Critical Habitat?
Lower Columbia River Chinook salmon (*Oncorhynchus tshawytscha*)	Threatened	Yes	No	Yes	No
Upper Willamette River Chinook salmon (*O. tshawytscha*)	Threatened	Yes	No	Yes	No
Upper Columbia River spring Chinook salmon (*O. tshawytscha*)	Endangered	Yes	No	Yes	No
Snake River spring/summer Chinook salmon (*O. tshawytscha*)	Threatened	Yes	No	Yes	No
Snake River fall Chinook salmon (*O. tshawytscha*)	Threatened	Yes	No	Yes	No
Columbia River chum salmon (*O. keta*)	Threatened	Yes	No	Yes	No
Lower Columbia River coho salmon (*O. kisutch*)	Threatened	Yes	No	Yes	No
Oregon Coast coho salmon (*O. kisutch*)	Threatened	Yes	No	Yes	No
Southern Oregon/Northern California Coasts coho salmon (*O. kisutch*)	Threatened	Yes	No	Yes	No
Snake River sockeye salmon (*O. nerka*)	Endangered	Yes	No	Yes	No
Lower Columbia River steelhead (*O. mykiss*)	Threatened	Yes	No	Yes	No
Upper Willamette River steelhead (*O. mykiss*)	Threatened	Yes	No	Yes	No
Middle Columbia River steelhead (*O. mykiss*)	Threatened	Yes	No	Yes	No
Upper Columbia River steelhead (*O. mykiss*)	Threatened	Yes	No	Yes	No
Snake River Basin steelhead (*O. mykiss*)	Threatened	Yes	No	Yes	No
Green sturgeon, Southern *(Acipencer medirostris)*	Threatened	Yes	No	Yes	No
Eulachon, Southern *(Thaleichthys pacificus)*	Threatened	Yes	No	Yes	No
Southern Resident Killer Whale *(Orcinus orca)*	Endangered	Yes	No	Yes	No

Fishery Management Plan That Identifies EFH in the Project Area	Does Action Have an Adverse Effect on EFH?	Are EFH Conservation Recommendations Provided?
Pacific Coast Salmon	Yes	Yes
Pacific Coast groundfish	Yes	Yes
Coastal pelagic species	Yes	Yes

Consultation Conducted By: National Marine Fisheries Service, West Coast Region

Issued By: Michael Tehan
Assistant Regional Administrator

Date: June 30, 2020

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ACRONYM	DEFINITION
ASTM	American Society for Testing and Materials
AWQMS	Ambient Water Quality Monitoring System
BCF	Bioconcentration Factor
BE	Biological Evaluation
BIA	Bureau of Indian Affairs
BLM	Bureau of Land Management
BOR	Bureau of Reclamation
BPA	Bonneville Power Administration
CCC	Criterion Chronic Concentration
CHARTs	Critical Habitat Analytical Review Teams
CHRT	Critical Habitat Review Team
CMC	Criterion Maximum Concentration
COE	U.S. Army Corps of Engineers
CR	Columbia River
CWA	Clean Water Act
DDT	Dichlorodiphenyltrichloroethane
DOC	Dissolved Organic Carbon
DPS	Distinct Population Segment
DQA	Data Quality Act
EC	Effect Concentration
EFH	Essential Fish Habitat
ELS	Early Life Stages
EPA	Environmental Protection Agency
ESA	Endangered Species Act
ESU	Evolutionarily Significant Unit
FCRPS	Federal Columbia River Power System
g	Grams
GMAV	Genus Mean Acute Value
h	Hours
HAPCs	Habitat Areas of Particular Concern
HC	Hazard Concentration
HUC	Hydrologic Unit Code
ICIS	Integrated Compliance Information System
ICTRT	Interior Columbia Technical Recovery Team
ISAB	Independent Science Advisory Board
ITS	Incidental Take Statement
IWQC	Instantaneous Water Quality Criterion Concentration
LASAR	Laboratory Analytical Storage and Retrieval

ACRONYM	DEFINITION
LC	Lethal Concentration
LCR	Lower Columbia River
LCRE	Lower Columbia River Estuary
LOEC	Lowest Observed Effect Concentration
LT	Lethal Time
MCR	Middle Columbia River
mg/L	Milligrams Per Liter
MLR	Multiple Linear Regression
MPGs	Major Population Groups
MSA	Magnuson-Stevens Fishery Conservation and Management Act
NAWQA	National Water-Quality Assessment
NFIP	National Flood Insurance Program
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOEC	No Observed Effect Concentration
NPDES	National Pollutant Discharge Elimination System
NWIS	National Water Information System
OPINION	Biological Opinion
ORC	Oregon Coast
ODEQ	Oregon Department of Environmental Quality
ODFW	Oregon Department of Fish and Wildlife
OSU	Oregon State University
OWEB	Oregon Watershed Enhancement Board
OWQI	Oregon Water Quality Index
PAHs	Polycyclic Aromatic Hydrocarbons
PBFs	Physical or Biological Features
PCE	Primary Constituent Elements
PCBs	Polychlorinated Biphenyls
PFMC	Pacific Fishery Management Council
PIT	Passive Integrated Transponder
RM	River Mile
RPA	Reasonable and Prudent Alternative
RPMs	Reasonable and Prudent Measures
SD	Standard Deviation
sDPS	Southern Distinct Population Segment
SMAV	Species Mean Acute Value
SMCV	Species Mean Chronic Value
SONCC	Southern Oregon Northern California Coast
SR	Snake River
SRB	Snake River Basin

ACRONYM	DEFINITION
SRKW	Southern Resident Killer Whale
SSD	Species Sensitivity Distribution
STEP	Salmon-Trout Enhancement Program
TAFs	Taxonomic Adjustment Factors
TMDLs	Total Maximum Daily Loads
TOC	Total Organic Carbon
UCR	Upper Columbia River
UCSRB	Upper Columbia River Salmon Recovery Board
µg/L	Micrograms Per Liter
USFS	U.S. Forest Service
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
UWR	Upper Willamette River
VSP	Viable Salmonid Population
WDFW	Washington Department of Fish and Wildlife
WEB-ICE	Web-Based Interspecies Correlation Estimation
WET	Whole Effluent Toxicity
WQS	Water Quality Standards
YOY	Young of the Year

1. INTRODUCTION

This Introduction section provides information relevant to the other sections of this document and is incorporated by reference into Sections 2 and 3 below.

1.1 Background

The National Marine Fisheries Service (NMFS) prepared the biological opinion (Opinion) and incidental take statement (ITS) portions of this document in accordance with section 7(b) of the Endangered Species Act (ESA) of 1973 (16 USC 1531 et seq.), and implementing regulations at 50 CFR 402, as amended. We also completed an essential fish habitat (EFH) consultation on the proposed action, in accordance with section 305(b)(2) of the Magnuson-Stevens Fishery Conservation and Management Act (MSA) (16 U.S.C. 1801 et seq.) and implementing regulations at 50 CFR 600.

We completed pre-dissemination review of this document using standards for utility, integrity, and objectivity in compliance with applicable guidelines issued under the Data Quality Act (DQA) (section 515 of the Treasury and General Government Appropriations Act for Fiscal Year 2001, Public Law 106-554). The document will be available within 2 weeks at the NOAA Library Institutional Repository [https://repository.library.noaa.gov/welcome]. A complete record of this consultation is on file at the Snake Basin office in Boise, Idaho.

1.2 Consultation History

On July 8, 2004, Oregon submitted revised acute and chronic aquatic life criteria for aluminum (and numerous other toxic pollutants) to the U.S. Environmental Protection Agency (EPA) for approval. The EPA initially considered approving the aluminum criteria and requested initiation of formal consultation with NMFS on January 14, 2008. Before receiving a biological opinion from NMFS, the EPA realized Oregon's aluminum criteria were not consistent with the Clean Water Action (CWA) section 304(a) nationally recommended criteria. As a result, EPA prepared to disapprove of Oregon's aluminum criteria. As part of this effort EPA sent a letter to NMFS identifying this change in their proposed action, that is, to remove aluminum from the list of pollutants considered in the consultation. However, this request was made as NMFS was in the process of finalizing the jeopardy and adverse modification biological opinion. Rather than excising portions of the biological opinion specific to the acute and chronic aluminum criteria, NMFS noted EPA's request to withdraw the aluminum criteria from consultation in the biological opinion. On January 31, 2013, the EPA disapproved of seven of Oregon's aquatic life criteria, including the acute and chronic aluminum criteria.

On April 20, 2015, the EPA was sued for failing to promptly prepare and publish replacement criteria for seven of the aquatic life criteria disapproved in its January 31, 2013 action (Northwest Environmental Advocates v. U.S. EPA, 3:15-cv-00663-BR (D. Or. 2015)). This lawsuit was resolved in a consent decree entered by the District Court on June 9, 2016 which established deadlines for the EPA to address the disapproved aquatic life criteria by either approving replacement criteria submitted by Oregon or by proposing and promulgating federal criteria. For the freshwater aluminum criteria, the consent decree originally established deadlines for the EPA to propose federal criteria by December 15, 2017, and to take final action by September 28, 2018. On December 5, 2017, the District Court granted an extension of the applicable deadlines for EPA's proposal and final action. At that time, the consent decree required the EPA to propose federal criteria for Oregon by March 15, 2018, and to take final action on the proposal by March 27, 2019. On March 1, 2018, the District Court again granted an extension of the consent decree deadlines for EPA's proposed and final actions. The consent decree required that by March 15, 2019, the EPA will either approve aluminum criteria submitted by Oregon or EPA will sign a notice of federal rulemaking proposing aluminum criteria for Oregon. The consent decree also established a deadline of March 27, 2020 for EPA to sign a notice of final rulemaking, absent any state submittal of aluminum criteria.

The EPA notified NMFS of their intentions to promulgate freshwater aluminum aquatic life criteria in Oregon by letter dated February 24, 2017. The EPA hosted a conference call with NMFS and the U.S. Fish and Wildlife Service (USFWS) on September 22, 2017 to introduce the criteria and federal rule as well as to discuss the consultation process and timing. Between November 1, 2017 and May 22, 2019, additional discussions about the proposed action, effects analysis methodology, and consultation schedule occurred during a series of an in-person meeting and three conference calls.

On May 14, 2019, NMFS received EPA's request for confirmation of the list of species to include in the consultation. NMFS confirmed the list and provided additional recommendations by letter dated May 30, 2019 and by email on December 17, 2019. NMFS recommended EPA include designated critical habitat for Snake River spring/summer Chinook salmon. In addition, NMFS informed EPA that we intended to proposed an expansion to the critical habitat designation for Southern Resident killer whales (SRKW) in the fall of 2019. Considering the timing of consultation for the proposed action, we advised EPA to request a conference on the forthcoming proposed revisions to SRKW designated critical habitat. Finally, NMFS informed EPA of the presence of designated critical habitat for Southern Oregon/Northern California Coast coho salmon within the action area by email. The species and designated critical habitats included in the consultation are listed in Table 1. In addition, EPA requested EFH consultation for Pacific salmon (Chinook and coho salmon).

EPA shared an interim draft of the biological evaluation (BE) with NMFS on August 12, 2019. The EPA, NMFS, and USFWS discussed the interim draft BE during a meeting on August 19, 2019. EPA submitted a request to initiate formal consultation, along with the final BE to NMFS on September 20, 2019. On October 15, 2019, NMFS sent a letter to EPA conveying initial concerns we had with the proposed chronic criterion. In that letter, NMFS expressed a desire for EPA to reconsider their proposed action to utilize a lower effects concentration to derive more protective criteria and minimize potential adverse effects. Upon completing our sufficiency review, NMFS sent a 30-day letter, dated October 19, 2019, to EPA informing them the submission was insufficient to initiate consultation. In our letter, NMFS requested additional information about the proposed action, action area, effects analysis, relevant information regarding state implementation of the criteria, and best available science specific to aluminum toxicity. We requested receipt of the additional information by December 3, 2019.

Table 1. List of species, their listing status, and designated critical habitats included in this consultation.

Species Name	Status	Designated Critical Habitat
Chinook salmon *(Oncorhynchus tshawytscha)*		
Lower Columbia River	T	Y
Upper Willamette River	T	Y
Upper Columbia River, spring-run	E	Y
Snake River spring/summer-run	T	Y
Snake River fall-run	T	Y
Chum salmon *(O. keta)*		
Columbia River	T	Y
Coho salmon *(O. kisutch)*		
Southern Oregon/Northern California Coast	T	Y
Oregon Coast	T	Y
Lower Columbia River	T	Y
Sockeye salmon *(O. nerka)*		
Snake River	E	Y
Steelhead *(O. mykiss)*		
Lower Columbia River	T	Y
Upper Willamette River	T	Y
Middle Columbia River	T	Y
Upper Columbia River	T	Y
Snake River Basin	T	Y
Eulachon *(Thaleichthys pacificus)*		
Southern	T	Y
Green sturgeon *(Acipenser medirostris)*		
Southern	T	Y
Killer whale *(Orcinus orca)*		
Southern Resident	E	Y*

T = Threatened; E = Endangered; Y* = Proposed expansion of critical habitat designation, conference requested.

EPA and NMFS participated in four conference calls (October 31, November 1, November 13, and November 15, 2019) to discuss potential resolutions to the additional information requests. On December 4, 2019, NMFS sent a letter informing EPA of our closure of the consultation record associated with their initial request to initiate consultation. In that letter, we acknowledged additional information was forthcoming and recognized EPA's intention to submit a new request to initiate consultation. NMFS received EPA's new request to initiate consultation, along with a revised BE, on January 2, 2020. On January 14, 2020, NMFS sent a letter to EPA informing them the submittal package was complete and that formal consultation was initiated on January 2, 2020. On May 4, 2020, EPA granted NMFS a 45-day extension to the consultation.

On June 12, 2020, NMFS provide a copy of the proposed action and terms and conditions sections of the draft Opinion to the EPA; Nez Perce Tribe; Confederated Tribes of the Siletz Indians of Oregon; Confederated Tribes of Coos, Lower Umpqua, and Siuslaw Indians; Confederated Tribes of the Grand Ronde Community of Oregon; Coquille Indian Tribe; Cow Creek Band of Umpqua Tribe of Indians; Klamath Tribe; Confederated Tribes of the Umatilla Reservation; Burns Paiute Tribe; and Confederated Tribes of the Warm Springs Reservation. NMFS received a supportive comment from the Nez Perce Tribe and did not receive comments from the other Tribes. On June 22, 2020, NMFS received EPA's comments on the draft Opinion excerpts, and the agencies participated in two conference calls to discuss potential revisions to the proposed terms and conditions. As a result of EPA's input, NMFS revised the terms and conditions to improve clarity and better reflect EPA's permit oversight authority and capabilities.

In preparing this opinion, NMFS relied on information from the following sources:

- Biological Evaluation and supporting documentation;
- ECOTOX database;
- Published scientific literature;
- Other scientific literature (government reports);
- Available biological and chemical surface water monitoring data; and
- Salmonid population models.

The above information provided the basis for our determinations as to whether the EPA can ensure that its promulgation of freshwater aquatic life criteria for aluminum in Oregon is not likely to jeopardize the continued existence of ESA-listed species, and is not likely to result in the destruction or adverse modification of designated critical habitat.

1.3 Proposed Federal Action

Under the ESA, "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). Under the MSA, a "Federal action" means any action authorized, funded, or undertaken, or proposed to be authorized, funded or undertaken by a Federal agency (50 CFR 600.910). The proposed Federal action that is the subject of this consultation is EPA's proposed promulgation of fresh water aluminum aquatic life criteria for the state of Oregon.

The CWA requires all states to adopt water quality standards (WQS) to restore and maintain the physical, chemical, and biological integrity of the Nation's waters (33 U.S.C. §§ 1251 et seq.). At a minimum, state WQS must include beneficial use designations (e.g., fish and aquatic life, recreation, water supply, etc.), narrative and numeric criteria to protect beneficial uses, and an antidegradation policy. Beneficial uses are those purposes or benefits that are to be derived from a water body. Oregon has adopted a number of beneficial uses specific to fish including, but not limited to, salmon and steelhead migration corridors, salmon and steelhead spawning through fry emergence, and shad and sturgeon spawning and rearing. Numeric water quality criteria establish levels of individual pollutants (e.g., metals, organic pollutants, chlorine, ammonia, etc.) or parameters (e.g., dissolved oxygen, temperature, dissolved gas, etc.) that will protect the designated use of the waterbody. Any water quality standards adopted or revised after May 30, 2000, must be approved by EPA before being used as the basis for any CWA-related actions. Once approved by EPA, a water quality standard is considered "effective for CWA purposes." The WQS are implemented through various regulatory programs under the CWA, including permitting of point source discharges (Section 402), permitting of discharges of dredge and fill material (Section 404), issuing water quality certifications (Section 401), and developing and implementing total maximum daily loads (TMDLs) (Section 303(d)).

The EPA proposes to establish federal CWA criteria for fresh waters under the state of Oregon's jurisdiction[1] to protect aquatic life from the effects of exposure to harmful levels of aluminum. Although the criteria were developed for fresh water, the Oregon Department of Environmental Quality (ODEQ) may choose to apply the criteria in estuarine waters (where there are currently no EPA-approved saltwater criteria for aluminum) if the pH, dissolved organic carbon (DOC), and hardness values are within the bounds of the criteria model (ODEQ 2019). The proposed aluminum criteria for Oregon are based on the EPA's 2018 final CWA section 304(a) national recommended freshwater aquatic life criteria for aluminum (EPA 2018). The 304(a) criteria recommendations were developed consistent with EPA's guidelines for deriving water quality criteria for the protection of aquatic life (Stephan et al. 1985). The final 2018 recommended national criteria are based upon Multiple Linear Regression (MLR) models for fish and invertebrate species.

The MLR models use pH, DOC, and total hardness (as calcium carbonate [$CaCO_3$]) to quantify the effects of these water chemistry parameters on the toxicity (a function of bioavailability) of aluminum to aquatic organisms. The MLR models are then used to normalize the available toxicity data to accurately reflect the effects of the water chemistry (i.e., pH, DOC, and total hardness) on the toxicity of aluminum to tested species. The normalized toxicity test data are then used in a criteria calculator to generate the criterion maximum concentration ((CMC); also known as the acute criterion concentration) and the criterion continuous concentration ((CCC); also known as the chronic criterion concentration) outputs for specific water chemistry conditions. Thus, the acute and chronic criteria concentrations are not fixed numbers, but instead their values depend on the specific pH, DOC, and total hardness entered into the MLR models. Because the criteria concentrations are not explicit numeric values, but rather are specific to the water chemistry at a given time and place, EPA refers to the acute or chronic criteria concentration as calculator outputs. For purposes of this Opinion, we use the phrase "instantaneous water quality concentration" (IWQC) as a synonym for "calculator output", although that phrase is not used by EPA. More specifically, we will use the phrases "acute IWQC" and "chronic IWQC" interchangeably with CMC and CCC, respectively. The calculator outputs (i.e., CMC and CCC) are numeric values that EPA considers to be protective of designated aquatic life beneficial uses for that site-specific set of input conditions.

Table 2 lists the aquatic life criteria for aluminum that EPA is proposing to promulgate. These aquatic life criteria apply to fresh waters under the jurisdiction of Oregon and account for water chemistry characteristics that affect aluminum bioavailability and toxicity. The EPA published this proposed federal rule in the Federal Register on May 1, 2019 (84 FR 18454).

Table 2. Proposed Federal freshwater aluminum aquatic life criteria for Oregon.

Metal	CAS Number	Criterion Maximum Concentration (CMC)[2] (µg/L)	Criterion Continuous Concentration (CCC)[3] (µg/L)
Aluminum[1]	7429905	Acute (CMC) and chronic (CCC) freshwater aluminum criteria values for a site shall be calculated using the 2018 Aluminum Criteria Calculator (*Aluminum Criteria Calculator V.2.0.xlsx, or a calculator in R or other software package using the same 1985 Guidelines calculation approach and underlying model equations as in the Aluminum Criteria Calculator*	

[1] The State of Oregon does not have jurisdiction over surface waters that are located within the boundaries of Federally-recognized Indian Tribes.

Metal	CAS Number	Criterion Maximum Concentration (CMC)[2] (µg/L)	Criterion Continuous Concentration (CCC)[3] (µg/L)
		V.2.0.xlsx) as established in the EPA's Final Aquatic Life Ambient Water Quality Criteria for Aluminum 2018 (EPA 822-R-18-001, USEPA (2018)). Calculator outputs shall be used to calculate criteria values for a site that protect aquatic life throughout the site under the full range of ambient conditions, including when aluminum is most toxic given the spatial and temporal variability of the water chemistry at the site.	

[1]The criteria for aluminum are expressed as total recoverable metal concentrations.
[2]The CMC is the highest allowable one-hour average instream concentration of aluminum. The CMC is not to be exceeded more than once every three years. The CMC is rounded to two significant figures.
[3]The CCC is the highest allowable four-day average instream concentration of aluminum. The CCC is not to be exceeded more than once every three years. The CCC is rounded to two significant figures.

The proposed rule provides that the criteria calculator, which incorporates pH, DOC, and total hardness as input parameters, be used to calculate protective CMC and CCC aluminum criteria values for a site as set forth in the final 2018 recommended national criteria. These calculated criteria values would protect aquatic life under the full range of ambient conditions found at each site, including conditions when aluminum is most toxic given the spatial and temporal variability of the water chemistry at the site. The empirical toxicity test data used to develop the MLR models were developed under a range of water chemistry conditions. The MLR models were then used to normalize all of the toxicity data used in the criteria calculations. The EPA criteria calculator is designed to allow for extrapolation beyond the input parameter values used to general the MLR models; however, caution should be used when extrapolating. Table 3 summarizes the range of water quality represented by the toxicity tests underlying the MLR model along with the allowable extrapolation range. If water chemistry outside of these bounds are entered into the criteria calculator, the magnitude of the criteria will reflect that calculated for the model bounds.

Table 3. Range of values for the MLR mode=l input parameters for the toxicity tests and for the MLR model application.

Input Parameter	Empirical Testing Bounds	MLR Model Bounds
pH	6.0 – 8.7	5.0 – 10.5
Total hardness (mg/L)	9.8 – 428	0.01 – 430
DOC	0.08 – 12.3	0.08 – 12.0

Characterization of the parameters that affect the bioavailability, and associated toxicity, of aluminum is the primary feature to determine protectiveness of aquatic life at a site at any given time. Oregon will need to use ambient water chemistry data (i.e., pH, DOC, total hardness) as inputs to the model in order to determine protective aluminum criteria values for specific sites, unless the State develops default values to be used in implementation. Oregon has the discretion to select the appropriate method to reconcile model outputs and calculate the final criteria values for each circumstance as long as the resulting calculated criteria values shall protect aquatic life throughout the site and throughout the range of spatial and temporal variability, including when aluminum is most toxic consistent with regulatory language that is proposed to be promulgated by EPA.

The EPA recognized that the proposed criteria will vary based on site-specific water chemistry and specified that aquatic life shall be protected when aluminum is most toxic. However, EPA did not prescribe implementation methodologies that would assure aquatic life are protected under the most toxic conditions. Instead, EPA identified several possible approaches to reconciling multiple outputs of the criteria calculator in the preamble to the Oregon proposed rule (84 FR 18454). Those approaches are excerpted below. The appropriate method for each circumstance will depend primarily on data availability.

- **Method 1**: Users identify protective criteria values by selecting one or more individual model outputs based upon spatially and temporally representative site-specific measured values for model inputs. Method one can be used where input datasets are complete and inputs are measured frequently enough to statistically represent changes in the toxicity of aluminum, including conditions under which aluminum is most toxic. In this case, the criteria values are determined by selecting one or more individual outputs that will be protective of aquatic life under the full range of ambient conditions, including conditions of high aluminum toxicity. Method one could be used to also establish criteria values to apply on a seasonal basis where the data are sufficient.

- **Method 2**: Users calculate protective criteria values from the lowest 10th percentile of the distribution of individual model outputs, based upon spatially and temporally representative site-specific measured model input values. While the 10th percentile of outputs should be protective in a majority of cases, certain circumstances may warrant use of a more stringent model output (e.g., consideration of listed species). Sufficient data to characterize the appropriate distribution of model outputs are necessary to derive a protective percentile so that the site is protected under conditions of high aluminum toxicity.

- **Method 3**: Users select the lowest model outputs (the lowest CMC and the lowest CCC) calculated from spatially and temporally representative input datasets that capture the most toxic conditions at a site as the criteria values. Method three should be used where ten or fewer individual model outputs are available.

Because the criteria values vary based on site-specific chemistry and because the rule does not prescribe how the most toxic conditions are to be assigned, the potential consequences of promulgating these criteria depend, in part, on how the criteria are implemented in CWA programs (i.e., point source discharge permits, 303(d) listing determinations, and TMDL development). The ODEQ is responsible for implementing CWA programs in the state, and has substantial flexibility in establishing procedures for characterizing the most toxic conditions and implementing the criteria in TMDLs, discharge permits, and other programs. This flexibility introduces uncertainty into our analysis of potential consequences of the proposed action. Our analysis assumes that the most toxic conditions will be adequately characterized and the aluminum criteria will be implemented in a manner that is adequately protective when conditions are most toxic.

2. Endangered Species Act: Biological Opinion And Incidental Take Statement

The ESA establishes a national program for conserving threatened and endangered species of fish, wildlife, plants, and the habitat upon which they depend. As required by section 7(a)(2) of the ESA, each federal agency must ensure that its actions are not likely to jeopardize the continued existence of endangered or threatened species, or adversely modify or destroy their designated critical habitat. Per the requirements of the ESA, federal action agencies consult with NMFS and section 7(b)(3) requires that, at the conclusion of consultation, NMFS provide an Opinion stating how the agency's actions would affect listed species and their critical habitats. If incidental take is reasonably certain to occur, section 7(b)(4) requires NMFS to provide an ITS that specifies the impact of any incidental taking and includes non-discretionary RPMs and terms and conditions to minimize such impacts.

The EPA determined the proposed action is not likely to adversely affect the Southern Distinct Population Segment (sDPS) of green sturgeon and its critical habitat. We do not concur with this determination, and our rationale is provided in Section 2.5.4.

2.1 Analytical Approach

This Opinion includes both a jeopardy analysis and an adverse modification analysis. The jeopardy analysis relies upon the regulatory definition of "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 CFR402.02). Therefore, the jeopardy analysis considers both survival and recovery of the species. This Opinion relies on the definition of "destruction or adverse modification," which "means a direct or indirect alteration that appreciably diminishes the value of critical habitat as a whole for the conservation of a listed species" (50 CFR 402.02).

The designations of critical habitat for green sturgeon (74 FR 52300), salmon (58 FR 68543, 64 FR 24049, 70 FR 52630, 78 FR 7816, and 81 FR 9252), steelhead (70 FR 52630), and SRKW (71 FR 69054) use the term primary constituent element (PCE) or essential features. The 2016 critical habitat regulations (50 CFR 424.12) replaced this term with physical or biological features (PBFs). The shift in terminology does not change the approach used in conducting a ''destruction or adverse modification'' analysis, which is the same regardless of whether the original designation identified PCEs, PBFs, or essential features. In this Opinion, we use the term PBF to mean PCE or essential feature, as appropriate for the specific critical habitat.

The 2019 regulations define effects of the action using the term "consequences" (50 CFR 402.02). As explained in the preamble to the regulations (84 FR 44977), that definition does not change the scope of our analysis and in this Opinion we use the terms "effects" and "consequences" interchangeably.

We use the following approach to determine whether a proposed action is likely to jeopardize listed species or destroy or adversely modify critical habitat:

- Evaluate the rangewide status of the species and critical habitat expected to be adversely affected by the proposed action.

- Evaluate the environmental baseline of the species and critical habitat.

- Evaluate the effects of the proposed action on species and their habitat using an exposure-response approach.

- Evaluate cumulative effects.

- In the integration and synthesis, add the effects of the action and cumulative effects to the environmental baseline, and, in light of the status of the species and critical habitat, analyze whether the proposed action is likely to: (1) Directly or indirectly 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; or (2) directly or indirectly result in an alteration that appreciably diminishes the value of critical habitat as a whole for the conservation of a listed species.

- If necessary, suggest a reasonable and prudent alternative (RPA) to the proposed action.

The EPA's promulgation of aluminum criteria does not authorize specific actions implementing the criteria. Instead, the rule establishes the allowable concentrations of aluminum, considering site-specific water chemistry characteristics, and requires that aquatic life be protected at all times, including when aluminum is most toxic. These criteria will remain in place and will be applied in CWA programs until the criteria are repealed and replaced. Thus, our analysis of effects for species and their designated critical habitat extends from the date of this Opinion for as long as the criteria remain effective. Regarding criteria implementation, we have employed the following assumptions as part of our analysis:

- Sufficient information to characterize the most toxic conditions will be collected to inform point source discharge permit development. Where sufficient information is not available, EPA will ensure the most conservative assumptions are utilized to characterize the most toxic conditions.

- Implementation of the criteria will be done in a manner that ensures aluminum concentrations are adequately protective under the most toxic conditions.

- Mixture toxicity will be regulated and sufficiently minimized in point source discharge permits.

- Authorized mixing zones for aluminum will be as small as possible.

- When waters are listed as impaired for aluminum, TMDLs will be prepared and implemented to bring the criteria back into compliance.

2.2 Rangewide Status of the Species and Critical Habitat

This Opinion examines the status of each species that would be adversely affected by the proposed action. The status is determined by the level of extinction risk that the listed species face, based on parameters considered in documents such as recovery plans, status reviews, and listing decisions. This informs the description of the species' likelihood of both survival and recovery. The species status section also helps to inform the description of the species' "reproduction, numbers, or distribution" as described in 50 CFR 402.02. The Opinion also examines the condition of critical habitat throughout the designated area, evaluates the conservation value of the various watersheds and coastal and marine environments that make up the designated area, and discusses the function of the essential PBFs that help to form that conservation value. Table 4 describes the Federal Register notices and notice dates for the species and critical habitats under consideration in this Opinion.

The status of each species and designated critical habitats are described further in Sections 2.2.1 and 2.2.3, respectively. One factor affecting the status of ESA-listed species considered in this Opinion, and aquatic habitat at large, is climate change. The impact of climate change on species and their designated critical habitat is discussed on Section 2.2.3.

Table 4. Listing status, status of critical habitat designations and protective regulation, and relevant Federal Register decision notices for ESA-listed species considered in this Opinion

Species	Listing Status	Critical Habitat	Protective Regulations
Chinook salmon *(Oncorhynchus tshawytscha)*			
Lower Columbia River	T 6/28/05; 70 FR 37160	9/02/05; 70 FR 52630	6/28/05; 70 FR 37160
Upper Columbia River, spring-run	E 6/28/05; 70 FR 37160	9/02/05; 70 FR 52630	ESA section 9 applies
Upper Willamette River	T 6/28/05; 70 FR 37160	9/02/05; 70 FR 52630	6/28/05; 70 FR 37160
Snake River spring/summer-run	T 6/28/05; 70 FR 37160	10/25/99; 64 FR 57399	6/28/05; 70 FR 37160
Snake River fall-run	T 6/28/05; 70 FR 37160	12/28/93; 58 FR 68543	6/28/05; 70 FR 37160
Chum salmon *(O. keta)*			
Columbia River	T 6/28/05; 70 FR 37160	9/02/05; 70 FR 52630	6/28/05; 70 FR 37160
Coho salmon *(O. kisutch)*			
Lower Columbia River	T 6/28/05; 70 FR 37160	2/24/16; 81 FR 9252	6/28/05; 70 FR 37160
Southern Oregon/Northern California Coast	T 6/28/05; 70 FR 37160	5/5/99; 64 FR 24049	6/28/05; 70 FR 37160
Oregon Coast	T 6/20/11; 76 FR 35755	2/11/08; 73 FR 7816	2/11/08; 73 FR 7816
Sockeye salmon *(O. nerka)*			
Snake River	E 6/28/05; 70 FR 37160	12/28/93; 58 FR 68543	ESA section 9 applies
Steelhead (*O. mykiss*)			
Lower Columbia River	T 1/05/06; 71 FR 834	9/02/05; 70 FR 52630	6/28/05; 70 FR 37160
Middle Columbia River	T 1/05/06; 71 FR 834	9/02/05; 70 FR 52630	6/28/05; 70 FR 37160
Upper Columbia River	T 8/24/09; 74 FR 42605	9/02/05; 70 FR 52630	2/1/06; 71 FR 5178
Upper Willamette River	T 1/05/06; 71 FR 834	9/02/05; 70 FR 52630	6/28/05; 70 FR 37160
Snake River Basin	T 1/05/06; 71 FR 834	9/02/05; 70 FR 52630	6/28/05; 70 FR 37160
Killer whale (*Orcinus orca*)			
Southern Resident	E 11/18/05; 70 FR 69903	11/29/06; 71 FR 69054[1]	ESA section 9 applies
Eulachon (*Thaleichthys pacificus*)			
Southern	T 3/18/10; 75 FR 13012	10/20/11; 76 FR 65324	None at this time
Green sturgeon (*Acipenser medirostris*)			
Southern	T 4/7/06; 71 FR 17757	10/9/09; 74 FR 52300	6/2/10; 75 FR 30714

Note: Listing status: 'T' means listed as threatened under the ESA; 'E' means listed as endangered.
[1]NMFS proposed to expand the designated critical habitat to include six new areas along the West Coast on September 19, 2019 (84 FR 49214).

2.2.1 Status of the Species

For Pacific salmon and steelhead, we commonly use the four "viable salmonid population" (VSP) criteria (McElhany et al. 2000) to assess the viability of the populations that, together, constitute the species. These four attributes (abundance, productivity, spatial structure, and diversity) encompass the species' "reproduction, numbers, or distribution" as described in 50 CFR 402.02. A brief explanation of each attribute is provided below. For non-salmonid species (e.g., eulachon, green sturgeon, and SRKW), we apply these same principles and approach to describe their viability, referring to these attributes as "viable population" criteria.

"**Abundance**" generally refers to the number of naturally-produced adults (i.e., the progeny of naturally-spawning parents) in the natural environment (e.g., on spawning grounds).

"**Productivity**" refers to the entire life cycle (i.e., the number of naturally-spawning adults produced per parent). When progeny replace or exceed the number of parents, a

> population is stable or increasing. When progeny fail to replace the number of parents, the population is declining.
>
> "**Spatial structure**" refers both to the spatial distributions of individuals in the population and the processes that generate that distribution. A population's spatial structure depends on habitat quality and spatial configuration, and the dynamics and dispersal characteristics of individuals in the population.
>
> "**Diversity**" refers to the distribution of traits within and among populations. These range in scale from DNA (deoxyribonucleic acid) sequence variation in single genes to complex life history traits (McElhany et al. 2000).

A viable salmonid population (or viable population for non-salmonids) needs sufficient levels of these attributes in order to: safeguard the genetic diversity of the listed Evolutionarily Significant Unit (ESU) or Distinct Population Segment (DPS); enhance its capacity to adapt to various environmental conditions; and allow it to become self-sustaining in the natural environment (ICTRT 2007). These viability attributes are influenced by survival, behavior, and experiences throughout the entire life cycle, characteristics that are influenced in turn by habitat and other environmental and anthropogenic conditions.

The condition of these four attributes informs NMFS's determination of a species status. NMFS expresses the status of an ESU or DPS in terms of likelihood of persistence over 100 years (or risk of extinction over 100 years). NMFS uses McElhany et al.'s (2000) description of a VSP that defines "viable" as less than a 5 percent risk of extinction within 100 years (low risk of extinction) and "highly viable" as less than a 1 percent risk of extinction within 100 years (very low risk of extinction). A third category, "maintained," represents a less than 25 percent risk within 100 years (moderate risk of extinction). For salmonids and eulachon to be considered viable, an ESU/DPS should have multiple viable populations so that a single catastrophic event is less likely to cause the ESU/DPS to become extinct, and so that the ESU/DPS may function as a metapopulation that can sustain population-level extinction and recolonization processes (ICTRT 2007). The risk level of the ESU/DPS is based upon the aggregate risk levels of its component individual populations and major population groups (MPGs).

Information regarding the listing status, population structure, life history, current status, recovery strategy, and limiting factors for each species addressed in this Opinion is summarized in the following subsections and in Table 5 at the end of this section. More detailed information about these species can be found in their respective recovery plans, status reviews, and 5-year reviews. These documents are available on the NMFS West Coast Region website (http://www.westcoast.fisheries.noaa.gov/).

2.2.1.1 Lower Columbia River Chinook Salmon

The Lower Columbia River (LCR) Chinook salmon were originally listed in 1999 as threatened (64 FR 14308), and the listing was reaffirmed in 2005 (70 FR 37160) and updated in 2014 (79 FR 20802). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as threatened (NMFS 2016a; 81 FR 33468).

The LCR Chinook salmon ESU includes all naturally spawned populations from the mouth of the Columbia River upstream to and including the White Salmon River in Washington and the Hood River in Oregon. This ESU also includes the Willamette River upstream to Willamette Falls (exclusive of spring-run Chinook salmon in the Clackamas River), and 15 artificial propagation programs. The ESU spans three distinct ecological regions (Coast, Cascade, and Gorge) and includes three distinct life-history types (spring-run, fall-run, and late-fall-run). This ESU is comprised of 32 independent populations: 9 spring-run; 21 fall-run, and 2 late fall-run.

A recovery plan for the ESU was completed in 2013 (NMFS 2013), and the most recent status review was completed in 2016 (NMFS 2016a). Recovery of this species will require very large improvements in abundance in most populations. A number of notable efforts to restore migratory access to areas upstream of dams have occurred; however, additional restoration actions and improvements in other limiting factors are required in order to achieve ESA recovery. Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.2 Upper Columbia River Spring-run Chinook Salmon

The Upper Columbia River (UCR) spring-run Chinook salmon ESU was originally listed as endangered under the ESA in 1998 (64 FR 14308), and the status was reaffirmed in 2005 (70 FR 37160) and updated in 2014 (79 FR 20802). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as endangered (NMFS 2016b; 81 FR 33468).

The UCR spring-run Chinook salmon ESU includes naturally spawned populations in the major tributaries of the Columbia River upstream of Rock Island Dam and downstream of Chief Joseph Dam, excluding the Okanogan River. It also includes six artificial propagation programs (Twisp River, Chewuch River, Methow, Winthrop National Fish Hatchery, Chiwawa River, and White River). The ESU is comprised of a single MPG with three extant populations (the Wenatchee River, Methow River, and Entiat River) and one functionally extirpated population (Okanogan River).

A recovery plan was completed in 2007 (UCSRB 2007), and the most recent status review was completed in 2015 (NMFS 2016b). Substantial improvements in survival and/or natural production capacity are required in order to achieve recovery. Improvements have been made in operations and fish passage at tributary dams and at the Columbia River dams, numerous habitat restoration projects have been completed in many tributaries, and many regulatory mechanisms have been improved and updated. However, a substantial amount of work remains to address the factors that are limiting recovery of the species. Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.3 Upper Willamette River Chinook Salmon

The Upper Willamette River (UWR) Chinook salmon ESU was originally listed in 1999 (64 FR 14308), and the listing was reaffirmed in 2005 (70 FR 37160) and updated in 2014 (79 FR 20802). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as threatened (NMFS 2016c; 81 FR 33468).

The UWR Chinook salmon ESU includes all naturally spawning populations of spring-run Chinook salmon in the Clackamas River and the Willamette River, and its tributaries, above Willamette Falls. It also includes six artificial propagation programs (McKenzie River Hatchery, Marion Forks Hatchery/North Fork Santiam River, South Santiam Hatchery in the South Fork Santiam River and Mollala River; Willamette Hatchery, and the Clackamas Hatchery). The ESU is comprised of seven populations: Clackamas, Molalla, North Santiam, South Santiam, Calapooia, McKenzie, and the Middle Fork Willamette.

A recovery plan was completed in 2011 (ODFW and NMFS), and the most recent status review was completed in 2016 (NMFS 2016c). Substantial improvements in survival and/or natural production capacity are required in order to achieve recovery. Many populations have very high pre-spawning mortality. It is critical to address factors that contribute to prespawn mortality so that biological responses to improvements in other limiting factors can be fully realized. Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.4 Snake River Spring/summer Chinook Salmon

The Snake River (SR) spring/summer Chinook salmon ESU was listed as threatened on April 22, 1992 (57 FR 14653). The listing was reaffirmed in 2005 (70 FR 37160) and updated in 2014 (79 FR 20802). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as threatened (NMFS 2016d; 81 FR 33468).

This ESU includes all naturally spawning populations of spring/summer Chinook in the mainstem SR (below Hells Canyon Dam) and in the Tucannon River, Grande Ronde River, Imnaha River, and Salmon River subbasins (57 FR 23458), as well as the progeny of 15 artificial propagation programs (70 FR 37160). The hatchery programs include the South Fork Salmon River (McCall Hatchery), Johnson Creek, Lemhi River, Pahsimeroi River, East Fork Salmon River, West Fork Yankee Fork Salmon River, Upper Salmon River (Sawtooth Hatchery), Tucannon River (conventional and captive broodstock programs), Lostine River, Catherine Creek, Lookingglass Creek, Upper Grande Ronde River, Imnaha River, and Big Sheep Creek programs. The historical SR spring/summer Chinook salmon ESU likely also included populations in the Clearwater River drainage and extended above the Hells Canyon Dam complex. The ESU is comprised of five MPGs: Lower Snake River, Grande Ronde/Imnaha Rivers, South Fork Salmon River, Middle Fork Salmon River, and Upper Salmon River.

A recovery plan was completed in 2017 (NMFS 2017a), and the most recent status review was completed in 2016 (NMFS 2016d). Substantial improvements in abundance and productivity are required in order to achieve recovery. Improvements have been made in operations and fish passage at tributary dams and at the Columbia River dams, numerous habitat restoration projects have been completed in many tributaries, and many regulatory mechanisms have been improved and updated. However, a substantial amount of work remains to address the factors that are limiting recovery of the species. Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.5 Snake River Fall Chinook Salmon

The SR fall Chinook salmon ESU was listed as threatened on April 22, 1992 (57 FR 14653), and its status was reaffirmed in 2005 (70 FR 37160) and updated in 2014 (79 FR 20802). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as threatened (81 FR 33468).

This ESU includes one extant population of fish spawning in the mainstem of the SR and the lower reaches of several of the associated major tributaries including the Tucannon, Grande Ronde, Clearwater, Salmon, and Imnaha Rivers. The ESU also includes four artificial propagation programs: the Lyons Ferry Hatchery and the Fall Chinook Acclimation Ponds Program in Washington; the Nez Perce Tribal Hatchery in Idaho; and the Oxbow Hatchery in Oregon and Idaho (70 FR 37160). Historically, this ESU included one large additional population spawning in the mainstem of the SR upstream of the Hells Canyon Dam complex, an impassable migration barrier (NWFSC 2015).

A recovery plan was completed in 2017 (NMFS 2017b), and the most recent status review was completed in 2016 (NMFS 2016d). In order for the single population in this ESU to achieve a highly viable status, improvements in productivity (or a decrease in the year-to-year variability associated with the estimated productivity) is required, assuming natural-origin abundance remains high. An increase in productivity could occur with further reductions in mortalities across all life stages. Since the species was originally listed, a variety of management actions (operational changes and fish passage improvements at the Columbia River dams, harvest reductions, increased natural production through hatchery supplementation) have been made to aid the recovery of the species. Yet, a substantial amount of work remains to ensure this population is highly viable with a high degree of certainty. Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.6 Columbia River Chum Salmon

The Columbia River (CR) chum ESU was first listed as threatened under the ESA in 1999 (64 FR 14508), and its status was reaffirmed in 2005 (70 FR 37160) and updated in 2014 (79 FR 20802). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as threatened (81 FR 33468).

The CR chum ESU includes all naturally spawned populations of chum salmon in the Columbia River and its tributaries in Oregon and Washington. The historical upstream boundary for this species is generally considered to be about where The Dalles Dam is now located (NMFS 2013). It also includes chum salmon from three artificial propagation programs: Grays River; Washougal River Hatchery/Duncan Creek; and Big Creek Hatchery. The species is comprised of three MPGs (Coast, Cascade, and Gorge) and 17 populations.

A recovery plan was finalized in 2013 (NMFS 2013) and the most recent status review was completed in 2016 (NMFS 2016a). The recovery strategy focuses on improving tributary and estuarine habitat conditions, reducing or mitigating hydropower impacts, and reestablishing chum salmon populations where they may have been extirpated. Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.7 Lower Columbia River Coho Salmon

The LCR coho salmon ESU was first listed as threatened under the ESA on June 28, 2005 (70 FR 37160). The ESU was updated on April 14, 2014 (79 FR 20802). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as threatened (81 FR 33468).

The LCR coho salmon ESU includes all naturally spawned coho salmon originating from the Columbia River and its tributaries downstream from the Big White Salmon and Hood Rivers (inclusive) and the Willamette River and its tributaries below Willamette Falls. The ESU also includes the following artificial propagation programs: The Grays River Program; Peterson Coho Project; Big Creek Hatchery Program; Astoria High School Salmon-Trout Enhancement Program (STEP) Coho Program; Warrenton High School STEP Coho Program; Cowlitz Type-N Coho Program in the Upper and Lower Cowlitz Rivers; Cowlitz Game and Anglers Coho Program; Friends of the Cowlitz Coho Program; North Fork Toutle River Hatchery Program; Kalama River Type-N Coho Program; Kalama River Type-S Coho Program; Lewis River Type-N Coho Program; Lewis River Type-S Coho Program; Fish First Wild Coho Program; Fish First Type-N Coho Program; Syverson Project Type-N Coho Program; Washougal River Type-N Coho Program; Eagle Creek National Fish Hatchery Program; Sandy Hatchery Program; Bonneville/Cascade/Oxbow Complex Hatchery Program; Clatsop County Fisheries Net Pen Program; and the Clatsop County Fisheries/Klaskanine Hatchery Program. The ESU is comprised of three MPGs (Coastal, Cascade, and Gorge), containing 24 populations.

A recovery plan was finalized in 2013 (NMFS 2013) and the most recent status review was completed in 2016 (NMFS 2016a). While some improvements in status were reported in the 2015 status review, this improvement may be mostly due to improved level of monitoring rather than a true change in status. Most populations in the ESU remain at high risk, with low abundances. The recovery strategy focuses on protecting existing high-functioning habitat, improving tributary habitat (particularly overwintering habitat), reducing hatchery and harvest impacts, and reestablishing naturally spawning populations above tributary dams on the Cowlitz and North Fork Lewis Rivers. Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.8 Oregon Coast Coho Salmon

The Oregon Coast (OCR) coho salmon ESU was originally listed as threatened under the ESA on February 11, 2008 (73 FR 7816), and its status was reaffirmed in 2011 (76 FR 35755) and updated in 2014 (79 FR 20802). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as threatened (81 FR 33468).

The OCR coho salmon ESU is comprised of all naturally spawned coho salmon originating from coastal rivers south of the Columbia River and north of Cape Blanco. The ESU also includes coho salmon from the Cow Creek Hatchery Program. The ESU is comprised of five MPGs (North Coast, Mid Coast, Mid-South Coast, Umpqua, and Lakes) and 21 independent populations that were historically self-sustaining and likely had relatively little demographic influence from neighboring populations.

NMFS completed a recovery plan for this ESU in 2016 (NMFS 2016e) and the most recent status review was completed in 2016 (NMFS 2016f). NMFS' overall recovery strategy focuses on restoring degraded habitats and the ecosystem processes and functions that affect those habitats, and protecting those existing high-functioning habitats through effective regulatory backstops. The highest priorities are for the strategies and actions related to freshwater and estuarine rearing habitats in order to improve egg-to-smolt survival. Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.9 Southern Oregon/Northern California Coast Coho Salmon

The Southern Oregon/Northern California Coast (SONCC) coho salmon ESU was originally listed as threatened under the ESA on May 6, 1997 (62 FR 24589), and its status was reaffirmed in 2005 (70 FR 37160) and updated in 2014 (79 FR 20802). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as threatened (81 FR 33468).

The SONCC coho salmon ESU includes all naturally spawned populations of coho salmon originating from coastal streams and rivers between Cape Blanco, Oregon, and Punta Gorda, California. The ESU also includes coho salmon from the following artificial propagation programs: The Cole Rivers Hatchery Program; Trinity River Hatchery Program; and the Iron Gate Hatchery Program. This ESU is comprised of 30 independent and 10 dependent populations, grouped into seven diversity strata.

NMFS completed a recovery plan for this ESU in 2014 (NMFS 2014a), and the most recent status review was completed in 2016 (NMFS 2016g). NMFS' overall recovery strategy consists of two phases. The first phase is geared toward rebuilding spawner numbers to above depensation in core and non-core 1 populations and to build capacity to support strays by restoring habitat to support all life stages in non-core 2 and dependent populations. The second phase is geared to rebuild the number of spawners and juvenile occupancy to levels necessary for recovery. Actions to aid recovery are focused on addressing limiting factors in areas where coho currently persist and in unoccupied areas of suitable habitat. Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.10 Lower Columbia River Steelhead

The LCR steelhead were originally listed in 1998 as threatened (63 FR 13347), and the listing was reaffirmed in 2006 (71 FR 834) and updated in 2014 (79 FR 20802). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as threatened (NMFS 2016a; 81 FR 33468).

The LCR steelhead DPS includes all naturally spawned anadromous *O. mykiss* originating below natural and manmade impassable barriers from rivers between the Cowlitz and Wind Rivers (inclusive) and the Willamette (below Willamette Falls) and Hood Rivers (inclusive). This ESU also includes steelhead from seven artificial propagation programs: Cowlitz Trout Hatchery Late Winter-run, Kalama River wild winter-run and summer-run, Clackamas Hatchery late winter-run, Sandy Hatchery Late winter-run, Hood River winter-run, and Lewis River wild late-run winter steelhead. This DPS is comprised of 17 winter-run and 6 summer-run populations grouped into four strata.

A recovery plan for the ESU was completed in 2013 (NMFS 2013), and the most recent status review was completed in 2016 (NMFS 2016a). Recovery of this species will require very large improvements in abundance in most populations. Loss of tributary habitat, hatchery effects, and predation are pervasive threats that affect most populations, but the types of recovery actions that will be most beneficial will vary among the populations (NMFS 2013). Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.11 Middle Columbia River Steelhead

The Middle Columbia River (MCR) steelhead were originally listed in 1999 as threatened (64 FR 14517), and the listing was reaffirmed in 2006 (71 FR 834) and updated in 2014 (79 FR 20802). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as threatened (NMFS 2016g; 81 FR 33468).

The MCR steelhead DPS includes all naturally spawned anadromous winter-run *O. mykiss* originating below natural and manmade impassable barriers in streams from above the Wind River and the Hood River (exclusive), upstream to, and including the Yakima River. Seven artificial propagation programs are also included in the DPS: Touchet River Endemic, Yakima River Kelt Reconditioning (in Satus Creek, Toppenish Creek, Naches River, and Upper Yakima River), Umatilla River, and the Deschutes River. This DPS is comprised of seventeen extant populations, grouped into the following four MPGs: Cascades Eastern Slope Tributaries, John Day River, Umatilla/Walla Walla, and Yakima Basin. Steelhead that are designated as part of an experimental population above the Pelton Round Butte Hydroelectric Project in the Deschutes Basin are not included in the DPS.

A recovery plan was completed in 2009 (NMFS 2009), and the most recent status review was completed in 2016 (NMFS 2016h). The Oregon Steelhead Recovery Plan (Appendix A, NMFS 2009) sets a higher standard for Oregon populations than the minimum criteria established by the Interior Columbia Technical Recovery Team (ICTRT). The majority of populations are at a moderate risk for abundance and productivity, but low to moderate risk for spatial structure and diversity. Improvements in survival are needed to meet recovery criteria for abundance and productivity. Some of the key broader recovery actions identified in the plan are to restore important tributary habitat functions in areas that likely supported substantial production, including summer rearing and overwintering habitat; improve hatchery management to reduce straying; and improve mainstem and estuary survival. Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.12 Upper Columbia River Steelhead

The UCR steelhead were originally listed in 1997 as endangered (62 FR 43937), and their listing was reclassified to threatened in 2006 (71 FR 834), and updated in 2014 (79 FR 20802). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as threatened (NMFS 2016a; 81 FR 33468).

UCR steelhead DPS includes all naturally spawned anadromous winter-run *O. mykiss* originating below natural and manmade impassable barriers in streams in the Columbia River basin upstream from the Yakima River to the U.S.-Canada border. Six artificial propagation programs are also included in the DPS: Wenatchee River, Wells Hatchery (in the Methow and Okanogan Rivers), Winthrop National Fish Hatchery, Omak Creek, and the Ringold steelhead hatchery program. This DPS is comprised of four extant populations: Wenatchee, Entiat, Methow, and Okanogan. Several smaller tributaries of the Columbia River (e.g., Squilchuck, Stemilt, Colockum, Tarpiscan Creeks, and probably Crab Creek) potentially produced steelhead, but never in great numbers.

A recovery plan was completed in 2007 (UCSRB 2007), and the most recent status review was completed in 2016 (NMFS 2016b). Improvements in natural origin abundance and productivity, considerable improvements in habitat condition, among other things, are necessary to recover this species. Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.13 Upper Willamette River Steelhead

The UWR steelhead were originally listed in 1999 as threatened (64 FR 14517), and the listing was reaffirmed in 2006 (71 FR 834) and updated in 2014 (79 FR 20802). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as threatened (NMFS 2016c; 81 FR 33468).

The UWR steelhead DPS includes all naturally spawned anadromous winter-run *O. mykiss* originating below natural and manmade impassable barriers from the Willamette River and its tributaries upstream of Willamette Falls to and including the Calapooia River. No hatchery programs are included in the DPS. This DPS has four independent populations, all of which are located geographically to the east of the Willamette River: North Santiam, South Santiam, Calapooia, and Molalla.

A recovery plan was completed in 2011 (ODFW and NMFS 2011a), and the most recent status review was completed in 2016 (NMFS 2016c). According to the recovery plan, it is assumed that the threats criteria are what remain to be addressed in order to achieve recovery. Actions undertaken to address limiting factors for UWR Chinook will also increase the viable abundance, productivity, and spatial structure parameters for UWR steelhead. Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.14 Snake River Basin Steelhead

The Snake River Basin (SRB) steelhead was listed as a threatened ESU on August 18, 1997 (62 FR 43937), with a revised listing as a DPS on January 5, 2006 (71 FR 834). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as threatened (81 FR 33468).

This species includes all naturally-spawning anadromous *O. mykiss* populations below natural and manmade impassable barriers in streams in the SRB of southeast Washington, northeast Oregon, and Idaho, as well as the progeny of six artificial propagation programs (71 FR 834). The hatchery programs include Dworshak National Fish Hatchery, Lolo Creek, North Fork Clearwater River, East Fork Salmon River, Tucannon River, and the Little Sheep Creek/Imnaha River steelhead hatchery programs. The ICTRT (2003) identified 24 extant populations within this DPS, organized into five MPGs. The five MPGs with extant populations are the Clearwater River, Salmon River, Grande Ronde River, Imnaha River, and Lower Snake River. In the Clearwater River, the historic North Fork population was blocked from accessing spawning and rearing habitat by Dworshak Dam.

A recovery plan was completed in 2017 (NMFS 2017a), and the most recent status review was completed in 2016 (NMFS 2016d). Substantial improvements in abundance and productivity are required in order to achieve recovery. Improvements have been made in operations and fish passage at tributary dams and at the Columbia River dams, numerous habitat restoration projects have been completed in many tributaries, and many regulatory mechanisms have been improved and updated. However, a substantial amount of work remains to address the factors that are limiting recovery of the species. Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.15 Snake River Sockeye Salmon

The SR sockeye salmon ESU was first listed as endangered under the ESA in 1991 (56 FR 58619), and the listing was reaffirmed in 2005 (70 FR 37160). On May 26, 2016, in the agency's most recent 5-year review for Pacific salmon and steelhead, NMFS concluded that the species should remain listed as endangered (81 FR 33468).

The SR sockeye salmon ESU includes all anadromous and residual sockeye salmon from the SRB in Idaho, as well as artificially propagated sockeye salmon from the Redfish Lake captive propagation program. The ICTRT identified historical sockeye salmon production in five Sawtooth Valley lakes, in addition to Warm Lake and the Payette Lakes in Idaho and Wallowa Lake in Oregon (ICTRT 2003). The sockeye runs to Warm, Payette, and Wallowa Lakes are now extinct, and the Sawtooth Valley lakes are identified as a single MPG for this ESU. The MPG consists of the Redfish, Alturas, Stanley, Yellowbelly, and Pettit Lake populations (ICTRT 2007). The only extant population is Redfish Lake, supported by a captive broodstock program.

NMFS completed a recovery plan in 2015 (NMFS 2015a), and the most recent status review was completed in 2016 (NMFS 2016d). In terms of natural production, this ESU remains at extremely high risk. Substantial improvements in survival and natural production capacity are required in order to achieve recovery. Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.16 Southern Eulachon

The Southern DPS (sDPS) of eulachon was listed as threatened in 2010 (75 FR 13012). On May 26, 2016, in the agency's most recent 5-year review for the sDPS of eulachon, NMFS concluded that the species should remain listed as threatened (81 FR 33468).

This species includes all naturally-spawned populations that occur in rivers south of the Nass River in British Columbia to the Mad River in California. Until additional information is available, the minimum set of subpopulations considered for this species include the Fraser River, Columbia River, British Columbia coastal rivers, and the Klamath River. However, other rivers that may have (or had) important contributions to the overall productivity, spatial distribution, and genetic and life history diversity of the species include the Elwha, Naselle, Umpqua, and Smith Rivers.

A recovery plan was completed in 2017 (NMFS 2017c), and the most recent status review was completed in 2016 (Gustafson 2016, NMFS 2016i). Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.17 Southern Green Sturgeon

The sDPS of green sturgeon was listed as threatened in 2006 (71 FR 17757) and updated in 2014 (79 FR 20802). On August 11, 2014, in the agency's most recent 5-year review for the sDPS of green sturgeon, NMFS concluded that the species should remain listed as threatened (NMFS 2015b).

This species includes all naturally-spawned populations that occur in rivers south of the Nass River in British Columbia to the Mad River in California. Until additional information is available, the minimum set of subpopulations considered for this species include the Fraser River, Columbia River, British Columbia coastal rivers, and the Klamath River. However, other rivers that may have (or had) important contributions to the overall productivity, spatial distribution, and genetic and life history diversity of the species include the Elwha, Naselle, Umpqua, and Smith Rivers.

A recovery plan was completed in 2018 (NMFS 2018), and the most recent status review was completed in 2015 (NMFS 2015b). Information about the life history strategies, recovery objectives, latest status summary, and limiting factors are summarized in Table 5.

2.2.1.18 Southern Resident Killer Whale

The Southern Resident Killer Whales (SRKW) DPS comprised of the J, K, and L pods, was listed as endangered under the ESA on November 18, 2005 (70 FR 69903). In the 5-year review completed in 2016, NMFS concluded that SRKWs should remain listed as endangered, based on recent information on the population, threats, and new research results and publications (NMFS 2016j). NMFS considers SRKWs to be currently among eight of the most at-risk species as part of the Species in the Spotlight[2] initiative because of their endangered status, and declining population trend. They are high priority for recovery based on conflict with human activities and recovery programs in place to address threats. The population has relatively high mortality and low reproduction unlike other resident killer whale populations that have generally been increasing since the 1970s (Carretta et al. 2019).

Killer whales are a long-lived species and sexual maturity can occur at age 10 (review in NMFS (2008)). Females produce a small number of surviving calves (n < 10, but generally fewer) over the course of their reproductive life span (Bain 1990; Olesiuk et al. 1990). Compared to Northern Resident killer whales, SRKW females appear to have reduced fecundity (Ward et al. 2013; Vélez-Espino et al. 2014), and all age classes of SRKWs have reduced survival compared to other fish-eating populations of killer whales in the Northeast Pacific (Ward et al. 2013).

The limiting factors described in the final recovery plan include reduced prey availability and quality, toxic chemicals that accumulate in top predators, and disturbances from vessels and sound (NMFS 2008). Oil spills and disease as well as the small population size are also risk factors. It is likely that multiple threats are acting together to impact SRKWs. Table 5 summarizes the status of SRKWs throughout their range, information from the recovery plan (NMFS 2008), and the most recent 5-year review (NMFS 2016j). In addition, it includes recent information from the Biological Opinion and Conference Opinion on Implementation of the Pacific Fishery Management Council (PFMC) Salmon Fishery Management Plan for Southern Resident Killer Whales and their Current and Proposed Critical Habitat (NMFS 2020).

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[2] https://www.fisheries.noaa.gov/resource/document/species-spotlight-priority-actions-2016-2020-southern-resident-killer-whale

Table 5. Summarized life history, recovery plan, status review, and limiting factor information for each species considered in this Opinion

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Lower Columbia River Chinook Salmon	Stream-type (spring; late-fall) Ocean-type (fall) The spring-run populations have a longer freshwater residency, migrating to the ocean as yearlings. Typically, they rear in their natal streams for a full year; however, some juveniles migrate downstream in the fall or winter as subyearlings to overwinter in larger systems such as the Columbia River. Spring Chinook usually move quickly through the estuary. Adults enter the lower Columbia River from March through June, and spawning in tributaries occurs in August and September. The fall-run populations typically migrate to the ocean within the first three months of life. Ocean-type juveniles typically spend a substantial amount of time in the estuary. Adults enter the lower Columbia River from August to September and spawn from late September to November. Late-fall runs exhibit stream-type life history. Adults return later and spawn later than the fall runs and utilize the Lewis and Sandy Rivers.	NMFS (2013) identified biological criteria for recovery and recognized there are many scenarios under which recovery can be achieved. One possible recovery scenario that meets these criteria and reflects the goals in the management unit plans includes: 16 populations targeted to be viable or highly viable; 12 populations targeted to achieve a maintained status; and 4 populations targeted to retain their baseline status. Currently, only two populations (North Fork Lewis and Sandy late fall run populations) are considered viable. The vast majority of populations (27) are at a very high risk of extinction, two populations are at high risk, and one population is at moderate risk. The 2015 status review reflected information through 2014, and concluded that overall, there was little change in the biological status of this ESU as a whole since the previous review by Ford et al. (2011). Increases in abundance were noted in about 70 percent of the fall-run populations and decreases in hatchery contribution were noted for several populations. Spring-run Chinook populations were generally unchanged; although the Sandy River spring-run population exhibited a substantial increase in relative abundance. Relative to baseline VSP levels identified in the NMFS 2013 recovery plan, there was an overall improvement in the status of a number of populations through 2014, although most are still far from the recovery plan goals. Furthermore, since the last status review in 2015, observations of coastal ocean conditions suggested that the 2015-2017 outmigrant year classes experienced below average ocean survival during a marine heatwave and its lingering effects. This led researchers to predict a corresponding drop in adult returns through 2019 (Werner et al. 2017).	• Degraded freshwater habitat (reduced channel complexity, loss of side channel and floodplain habitat, reduced water quality, altered hydrologic patters, barriers to spawning/rearing habitat) • Degraded estuary habitat (reduced quantity and accessibility of in-channel, off-channel, and plume habitat; elevated water temperatures; toxic contamination) • Hydropower-related effects • Harvest-related effects • Hatchery-related effects • Predation

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Upper Columbia River Spring-run Chinook Salmon	Stream-type The vast majority of juveniles spend a full year in freshwater and migrate as yearlings in the spring to the mainstem Columbia River and out to sea. It takes about 15 days to migrate from McNary Dam to Bonneville Dam. About 50 percent of the Entiat population juveniles migrate out from their natal areas as subyearlings and overwinter in the mainstem reservoirs of the Columbia River (ISAB 2018). The lower Columbia river (below Bonneville) is a feeding area for juveniles (ISAB 2018). Adults begin returning from the ocean in April and May, with the run into the Columbia River peaking in mid-May. They enter the Upper Columbia River tributaries from April through July and spawning peaks in mid-to-late August.	Recovery of this ESU requires: (1) abundance/productivity criteria for all three populations be categorized as viable (i.e., <5% risk of extinction); (2) distribution of naturally produced spring Chinook salmon be restored to previously occupied areas where practical; and (3) expression of natural patterns of genetic and phenotypic diversity be allowed for. Currently, three populations are at a high risk of extinction and one is functionally extirpated (NWFSC 2015). The latest estimates of natural origin spawner abundance increased relative to the levels observed in the prior review for all three extant populations. Productivities were higher for the Wenatchee and Entiat populations and unchanged for the Methow population. However, abundance and productivity remain well below the viable thresholds called for in the recovery plan for all three populations. Furthermore, since the last status review in 2015, observations of coastal ocean conditions suggested that the 2015-2017 outmigrant year classes experienced below average ocean survival during a marine heatwave and its lingering effects. This led researchers to predict a corresponding drop in adult returns through 2019 (Werner et al. 2017). The negative impacts of on juvenile salmonids had subsided by spring 2018, but other aspects of the ecosystem (e.g., temperatures below the 50-meter surface layer) had not returned to normal (Harvey et al. 2019).	• Degraded freshwater habitat (instream complexity, bed and channel form, riparian condition, increased sediment) • Degraded estuarine and nearshore marine habitat • Hydropower-related effects • Hatchery-related effects • Persistence of non-native fish species • Harvest in Columbia River fisheries

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Upper Willamette River Chinook Salmon	Stream-type UWR spring Chinook exhibit three migration strategies – fry and early fingerling migration, subyearling migration, and yearling migration. Fry and fingerling migrants rear in lower reaches of spawning tributaries and the mainstem Willamette River. Those that grow quickly may continue migrating to the lower Columbia River and spend time in the estuary. Subyearling migrants move downstream from natal tributaries in the fall and early winter. Some of these fish move past Willamette Falls, and presumably rear in the lower Columbia River. The vast majority of juveniles spend a full year in freshwater and migrate fairly quickly as yearlings in the spring to the mainstem Columbia River and out to sea. Adult UWR spring Chinook appear in the lower Willamette River in January, with fish entering the Clackamas River as early as March. The majority of the run ascends Willamette Falls from late April through May. Peak spawning typically occurs in September.	NMFS (2011) identified biological criteria for recovery and recognized there are many scenarios under which recovery can be achieved. One possible recovery scenario that meets these criteria is to recover the McKenzie and Clackamas populations to an extinction risk status of very low, the North Santiam and Middle Fork Willamette populations to a low extinction risk, the South Santiam population to a moderate extinction risk, and the remaining populations to a high risk. Currently, five populations are at very high risk of extinction, the Clackamas River population is at moderate risk, and the McKenzie River population is at low risk. Abundance levels for five of the seven populations remain well below their recovery goals. Of these, the Calapooia River may be functionally extinct and the Molalla River remains critically low. Abundances in the North and South Santiam Rivers have increased, but still range only in the high hundreds of fish. The Clackamas and McKenzie populations have previously been viewed as natural population strongholds, but have both experienced declines in abundance despite having access to much of their historical spawning habitat. The fraction of hatchery origin fish in all populations remains high (even in Clackamas and McKenzie populations). Overall, the ESU is considered to be at a moderate risk of extinction, and the recent trend is considered to be declining (NWFSC 2015).	• Degraded quality/quantity of freshwater habitat (e.g., floodplain connectivity, riparian condition, sediment, channel structure and complexity) • Reduced access to spawning and rearing habitats • Degraded water quality (e.g., temperature, toxics, dissolved oxygen) • Increased predation, competition, and disease incidence • Altered stream flows • Altered food web due to reduced inputs of microdetritus • Predation by native and non-native species, including hatchery fish • Competition related to introduced salmon and steelhead • Altered population traits due to hatchery influences, harvest, and altered environmental conditions

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Snake River Spring/summer-run Chinook Salmon	Stream-type Juvenile SR spring/summer Chinook salmon typically spend a full year in spawning habitat and migrate downstream in early to mid-spring as age-1 smolts. Depending on tributary and site-specific habitat conditions, juveniles may migrate extensively from natal reaches into alternative summer-rearing or overwintering areas. Adult SR spring/summer Chinook salmon typically spend 2 to 3 years in the ocean. They return to the Columbia River and pass Bonneville Dam between April and August (50% of the run typically passes before the end of June).	There are a variety of scenarios under which recovery can be achieved and the possible recovery scenarios are outlined the recovery plan (NMFS 2017a). At a minimum, at least one population in each MPG should achieve a very low risk of extinction. Currently, all except one extant population (Chamberlin Creek) are at high risk of extinction (NWFSC 2015). Most populations will need to see increases in abundance and productivity in order for the ESU to recover. Several populations have a high proportion of hatchery-origin spawners—particularly in the Grande Ronde, Lower Snake, and South Fork Salmon MPGs—and diversity risk will also need to be lowered in multiple populations in order for the ESU to recover (NWFSC 2015). Before 2015, natural origin abundance increased over the levels reported in the 2011 status review for most populations in this ESU. Although the increases were not substantial enough to change viability ratings, the risk trend for the ESU was considered to be stable (NWFSC 2015). However, since the last status review in 2015, observations of coastal ocean conditions suggested that the 2015-2017 outmigrant year classes experienced below average ocean survival during a marine heatwave and its lingering effects. This led researchers to predict a corresponding drop in adult returns through 2019 (Werner et al. 2017). In fact, documented adult returns have remained very low over the past 3 years (Nez Perce Tribe 2018; Nez Perce Tribe 2019), and the trend for the most recent 5 years (2014-2018) has been generally downward (ODFW and WDFW 2019).	• Effects related to the hydropower system in the mainstem Columbia River, • Degraded freshwater habitat • Altered flows and degraded water quality • Harvest-related effects • Predation

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Snake River Fall-run Chinook Salmon	Ocean-type SR fall Chinook exhibit different early life-history strategies. The subyearling life-history strategy is predominant with juveniles entering the ocean within their first year of life. Some fish delay migration and overwinter in the mainstem Columbia River and estuary before entering saltwater as yearlings. Adult SR fall Chinook spend 2 to 5 years in the ocean. They return to the Columbia River and pass Bonneville Dam from mid-August to the end of September.	The recovery plan (NMFS 2017b) identifies three potential recovery scenarios for this species. The recovery strategy focuses on recovery for the extant Lower Snake River population, concurrent with scoping efforts for reintroduction above the Hells Canyon Complex. For a single extant population scenario, the recovery goal is for the population to be "highly viable with high certainty." Overall, the status of Snake River fall Chinook salmon has improved compared to the time of listing and compared to prior status reviews. The one extant population is at moderate risk for both diversity and spatial structure and abundance and productivity. The population is considered viable, but will need to see an increase in productivity combined with a reduction in diversity risk for the ESU to recover (ICTRT 2010; NWFSC 2015). In terms of risk, the NWFSC (2015) considered the trend at the time to be improving. From 2015 through 2018, annual returns steadily decreased (Personal Communication, Bill Young, Nez Perce Tribe Hatchery Evaluations Coordinator, October 17, 2019).	• Degraded floodplain connectivity and function • Harvest-related effects • Loss of access to historical habitat above Hells Canyon and other Snake River dams • Impacts from mainstem Columbia River and Snake River hydropower systems • Hatchery-related effects • Degraded estuarine and nearshore habitat.

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Columbia River Chum Salmon	Ocean-type Juvenile chum are believed to migrate downstream to the estuary promptly after emergence. Emergence typically occurs from March through May. Adults spend 2 to 6 years in the ocean. They return to the Columbia River from mid-October through November and spawn from early November to late December.	The goal of the recovery plan is to improve the VSP parameters such that the Coast and Cascade strata achieve a high probability of persistence and the persistent probability of the two Gorge populations is improved (including achieving a high persistence probability for the Lower Gorge population). Overall, the status of most chum salmon populations is unchanged from the baseline VSP scores estimated in the recovery plan. A total of 3 of 17 populations are at or near their recovery viability goals, and are considered to be stabilization populations. Although not targeted for improvements, recovery actions will still be needed for these three stabilizing populations to maintain their baseline status. The remaining populations generally require a higher level of viability and most require substantial improvements to reach their viability goals. While some improvements in status were reported in the 2015 status review, the majority of populations in this ESU remain at a high or very high risk category and considerable progress remains to be made to achieve recovery. In terms of risk, the recent trend for the ESU is considered to be stable (NWFSC 2015).	• Degraded estuarine and nearshore marine habitat • Degraded freshwater habitat • Degraded stream flow as a result of hydropower and water supply operations • Reduced water quality • Current or potential predation • An altered flow regime and Columbia River plume • Reduced access to off-channel rearing habitat in the lower Columbia River • Reduced productivity resulting from sediment and nutrient-related changes in the estuary • Juvenile fish wake strandings • Contaminants

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Lower Columbia River Coho Salmon	Stream-type Juveniles typically rear in freshwater form more than a year. Most juveniles migrate to downstream in the spring (March through May) and enter the ocean in late spring (April to June). Adult coho return to the Columbia River either early (from mid-August) or late (late September through December). Early returns typically spawn in tributaries from mid-October to early November. Late returns spawn from November to January, but could spawn as late as March.	The recovery strategy is to improve the VSP parameters such that the Coast, Cascade, and Gorge strata achieve a high probability of persistence. Of the 24 populations that make up this ESU, 21 populations are at very high risk, one population is at high risk, and two populations are at moderate risk. Recent recovery efforts may have contributed to observed natural production, but in the absence of longer term data sets it is not possible to parse out these effects. Populations with longer term data sets exhibit stable or slightly positive abundance trends. Some trap and haul programs appear to be operating at or near replacement, although other programs still are far from that threshold and require supplementation with additional hatchery-origin spawners. Initiation of or improvement in the downstream juvenile facilities at Cowlitz Falls, Merwin, and North Fork Dams are likely to further improve the status of the associated upstream populations. While these and other recovery efforts have likely improved the status of a number of coho salmon populations, abundances are still at low levels and the majority of the populations remain at moderate or high risk. For the LCR region land development and increasing human population pressures will likely continue to degrade habitat, especially in lowland areas. Although populations in this ESU have generally improved, especially in the 2013/14 and 2014/15 return years, recent poor ocean conditions suggest that population declines might occur in the upcoming return years. In terms of risk, the recent trend for the ESU is considered to be stable/improving (NWFSC 2015).	• Degraded estuarine and near-shore marine habitat • Fish passage barriers • Degraded freshwater habitat: Hatchery-related effects • Harvest-related effects • An altered flow regime and Columbia River plume • Reduced access to off-channel rearing habitat in the lower Columbia River • Reduced productivity resulting from sediment and nutrient-related changes in the estuary • Juvenile fish wake strandings • Contaminants

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Oregon Coast Coho Salmon	Stream-type Most juvenile coho migrate to the ocean as smolts in the spring from as late as March into June. Outmigrating smolts may rear in lower mainsteam and estuarine habitats for days or weeks before entering the nearshore ocean environment. Some fry also emigrate downstream and rear in lower river habitats, with a fraction of these entering the ocean as subyearlings. Adult coho return to natal tributaries from September to November.	The recovery objective for this species is that there is at least a moderate certainty that the ESU is sustainable. To achieve this, all five strata comprising this species must be considered sustainable, and for a stratum to be considered sustainable, more than half of its independent populations must be sustainable. This ESU comprises 56 populations including 21 independent and 35 dependent populations. The last status review indicated a moderate risk of extinction. Significant improvements in hatchery and harvest practices have been made for this ESU. Most recently, spatial structure conditions have improved in terms of spawner and juvenile distribution in watersheds; none of the geographic area or strata within the ESU appear to have considerably lower abundance or productivity. The ability of the ESU to survive another prolonged period of poor marine survival remains in question. In terms of risk, the recent trend for the ESU is considered to be improving (NWFSC 2015).	• Reduced amount and complexity of habitat including connected floodplain habitat • Degraded water quality • Blocked/impaired fish passage • Inadequate long-term habitat protection • Changes in ocean conditions

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Southern Oregon/ Northern California Coast Coho Salmon	Stream-type The dominant life-history pattern is for juveniles to rear in freshwater for a year before migrating to the ocean. However, a fraction of juveniles may rear in freshwater for two years or may emigrate to estuarine habitats shortly after emergence. Some may rear in the estuary for a number of months and return upstream to overwinter. Downstream migration typically occurs in the spring between April and May and continues into June. This species typically spends about 18 months in the ocean before returning as 3 year olds to freshwater. Upriver migration of adults to spawning areas typically occurs from October to March, with a peak between November and January.	To achieve recovery, all "core" populations should be at a low risk of extinction, all non-core 1 populations should be at least at moderate risk of extinction, and all non-core 2 populations should have demonstrated juvenile occupancy. Additionally, population growth rates should be neutral or positive for all core and non-core 1 populations, populations should be widely distributed with sufficient connectivity, hatchery impacts should be low or moderate, and life history should be attained and retained (NMFS 2016g). Of the 30 independent populations, 24 are at high risk of extinction and 6 are at moderate risk of extinction (NMFS 2014a). The extinction risk of an ESU depends upon the extinction risk of its constituent independent populations; because the population abundance of most independent populations are below their depensation threshold, the SONCC coho salmon ESU is at high risk of extinction and is not viable. Although recent changes in trend/viability of the ESU are considered to be mixed, there has not been an apparent trend toward recovery. While the overall level of concern has increased based on likely effects from increased water withdrawal in many areas and on drought conditions, the available information does not appear to suggest the need for a change in extinction risk at this time (NMFS 2016g).	• Lack of floodplain and channel structure • Impaired water quality • Altered hydrologic function • Impaired estuary/mainstem function • Degraded riparian forest conditions • Altered sediment supply • Increased disease/predation/competition • Barriers to migration • Fishery-related effects • Hatchery-related effects

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Snake River Sockeye Salmon	Juvenile sockeye salmon rear in Sawtooth Valley lakes for 1 to 3 years prior to migrating to the ocean. Juvenile outmigration from the lakes generally occurs from April through May. Juveniles move rapidly downstream and spend little time rearing in migration corridor and Columbia River estuary (NMFS 2015a). Adult sockeye salmon spend 1 to 3 years in the ocean and enter the Columbia River in June and July. Spawning in the Sawtooth Valley of Idaho generally occurs in fall, peaking in October.	The long-term strategy is for a naturally produced population to achieve escapement goals in a manner than is self-sustaining and without the reproductive contribution of hatchery spawners. This single population ESU is at very high risk dues to small population size. There is high risk across all four basic risk measures. Although the captive brood program has been successful in providing substantial numbers of hatchery produced fish for use in supplementation efforts, substantial increases in survival rates across all life history stages must occur to reestablish sustainable natural production. In terms of natural production, the SR Sockeye ESU remains at extremely high risk although there has been substantial progress on the first phase of the proposed recovery approach – developing a hatchery-based program to amplify and conserve the stock to facilitate reintroductions. In terms of risk, the recent trend for the ESU is considered to be improving (NWFSC 2015).	• Effects related to the hydropower system in the mainstem Columbia River • Reduced water quality and elevated temperatures in the Salmon River • Water quantity • Predation
Upper Columbia River Steelhead	Stream-type Most juveniles smolt after two years in freshwater, but can migrate at ages ranging from 1 to 7 years. Outmigration generally occurs from March to June, with peak migration in April or May. Most steelhead spend 1 to 2 years in the ocean. Adults return to the Columbia River between late summer and early fall. A portion of these fish overwinter in mainstream reservoirs, passing over Upper Columbia River dams in April and May of the following year. Spawning occurs in the late spring of the calendar year following re-entry into freshwater.	Recovery of UCR steelhead will require all four populations in the DPS achieve a low extinction risk. To achieve this, increases in both steelhead abundance and productivity is needed as well as increases in the current distribution of naturally produced steelhead (UCSRB 2007). Of the four populations in this DPS, three are at high risk of extinction while one population is at moderate risk. UCR steelhead populations have increased relative to the low levels observed in the 1990s, but natural origin abundance and productivity remain well below viability thresholds for three out of the four populations. The status of the Wenatchee River steelhead population continued to improve based on the additional year's information available for the most recent review. The abundance and productivity viability rating for the Wenatchee River exceeds the minimum threshold for 5 percent extinction risk. However, the overall DPS status remains unchanged from the prior review, remaining at high risk driven by low abundance and productivity relative to viability objectives and diversity concerns. In terms of risk, the recent trend for the DPS is considered to be improving (NWFSC 2015).	• Adverse effects related to the mainstem Columbia River hydropower system • Impaired tributary fish passage • Degraded floodplain connectivity and function, channel structure and complexity, riparian areas, large woody debris recruitment, stream flow, and water quality • Hatchery-related effects • Predation and competition • Harvest-related effects

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Lower Columbia River Steelhead	Stream-type Emergence occurs from March into July. Most juveniles smolt after two years in freshwater, but can migrate at ages ranging from 1 to 4 years. Outmigration generally occurs from March to June, with peak migration in April or May. Most steelhead spend 2 years in the ocean, although ocean residence can range from 1 to 4 years. Summer-run adults return between May and October and require several months to mature before spawning. These fish spawn between late February and early April. Winter-run adults return to freshwater between December and May as sexually mature fish. Peak spawning occurs in late April and early May.	The recovery strategy for this DPS is aimed at restoring all four strata to a high probability of persistence. Crucial recovery elements include: (1) protect favorable tributary habitat and restore degraded but potentially productive habitat; (2) protect and improve the South Fork Toutle, East Fork Lewis, Clackamas, and Hood winter steelhead populations; (3) significantly reduce hatchery impacts on the Hood summer steelhead population; (4) reestablish naturally spawning winter-run populations above tributary dams in the Cowlitz system, improve the status of the Tilton winter-run population, and reintroduce winter-run above dams on the North Fork Lewis River. Currently, 9 populations are at very high risk, 7 are at high risk, 6 are at moderate risk, and 1 is at low risk. The majority of winter-run steelhead populations in this DPS continue to persist at low abundances. Hatchery interactions remain a concern in select basins, but the overall situation is somewhat improved compared to prior reviews. Summer-run steelhead populations were similarly stable, but at low abundance levels. The decline in the Wind River summer-run population is a source of concern, given that this population has been considered one of the healthiest of the summer-runs; however, the most recent abundance estimates suggest that the decline was a single year aberration. Passage programs in the Cowlitz and Lewis basins have the potential to provide considerable improvements in abundance and spatial structure, but have not produced self-sustaining populations to date. Even with modest improvements in the status of several winter-run populations, none of them appear to be at fully viable status, and similarly none of the MPGs meet the criteria for viability. In terms of risk, the recent trend for the DPS is considered to be stable (NWFSC 2015).	• Degraded estuarine and nearshore marine habitat • Degraded freshwater habitat • Reduced access to spawning and rearing habitat • Avian and marine mammal predation • Hatchery-related effects • An altered flow regime and Columbia River plume • Reduced access to off-channel rearing habitat in the lower Columbia River • Reduced productivity resulting from sediment and nutrient-related changes in the estuary • Juvenile fish wake strandings • Contaminants

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Upper Willamette River Steelhead	Stream-type Most juveniles migrate to the ocean after 2 years in freshwater. However, freshwater rearing can range from 1 to 4 years. Smoltification generally occurs in April through May, when juveniles are migrating quickly downstream through the mainstem Willamette River and the lower Columbia River estuary (LCRE)[3]. This species spends 1 to 4 years in the ocean. Adult fish typically return to freshwater in January through April and pass Willamette Falls from mid-February to mid-May. Spawning occurs in March through June, peaking in late April and early May.	The recovery approach for this DPS is to ensure no population has a higher extinction risk than currently, and to maintain or improve all core populations and one non-core population to a viable level. Most current simulation of biological viability criteria indicate this DPS is viable, and addressing threats criteria is assumed to be what remains for delisting purposes. Of the four populations comprising this DPS, three are at low risk and one is at moderate risk. Declines in abundance noted in the last status review continued through the period from 2010-2015. Current rates of decline have increases, and the DPS continues to demonstrate the overall low abundance pattern that was of concern during the last status review. The causes of these declines are not well understood, although much accessible habitat is degraded and under continued development pressure. The elimination of winter-run hatchery release in the basin reduced hatchery threats, but non-native summer steelhead hatchery releases may still be a source of competition. While the collective risk to the persistence of the DPS has not changed significantly in recent years, continued declines and potential negative impacts from climate change may cause increased risk in the near future. In terms of risk, the recent trend for the DPS is considered to be declining (NWFSC 2015).	• Degraded freshwater habitat • Degraded water quality • Increased disease incidence • Altered stream flows • Reduced access to spawning and rearing habitats due to impaired passage at dams • Altered food web due to changes in inputs of microdetritus • Predation by native and non-native species, including hatchery fish and pinnipeds • Competition related to introduced salmon and steelhead • Altered population traits due to interbreeding with hatchery origin fish

[3] The LCRE, for purpose of this Opinion, begins at Bonneville Dam, the head-of-tide in the Columbia River and extends about 134 miles downstream to the Pacific Ocean.

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Middle Columbia River Steelhead	Stream-type Most juveniles migrate to the ocean after 2 years in freshwater; however, freshwater residency may range anywhere from 1 to 5 years. Downstream migration occurs between March and June. Most steelhead spend 2 years in the ocean (range of 1 to 4 years) prior to returning to freshwater. The MCR steelhead are categorized as either summer-run or winter-run, depending on the timing of their return. Adult summer-run steelhead reenter freshwater between May and October and require several months to mature. Adult winter-run steelhead reenter freshwater as sexually mature fish between November and April.	There are a variety of scenarios under which recovery can be achieved and the possible recovery scenarios are outlined the recovery plan (NMFS 2009). Of the 17 populations comprising this DPS, 4 are high risk, 7 are moderate risk, and 6 are low risk of extinction (NWFSC 2015). Returns to the Yakima River basin and to the Umatilla and Walla Walla Rivers have been higher over the most recent brood cycle, while natural origin returns to the John Day River have decreased. There have been improvements in the viability ratings for some of the component populations, but the DPS is not currently meeting the viability criteria in the MCR steelhead recovery plan. In general, the majority of population level viability ratings remained unchanged from prior reviews for each MPG within the DPS. In terms of risk, the recent trend for the DPS is considered to be stable/improving (NWFSC 2015).	• Degraded freshwater habitat • Mainstem Columbia River hydropower-related impacts • Degraded estuarine and nearshore marine habitat • Hatchery-related effects • Harvest-related effects • Effects of predation, competition, and disease

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Snake River Basin Steelhead	Stream-type Most juveniles migrate to the ocean after 2 years in freshwater; however, freshwater residency may range anywhere from 1 to 5 years. Downstream migration occurs from March to mid-June. Most steelhead spend 2 years in the ocean (range of 1 to 4 years) prior to returning to freshwater. Adult steelhead reenter freshwater from late June to October. Adults tend to hold in larger rivers over the winter, and spawn in tributary streams from March through May.	There are a variety of scenarios under which recovery can be achieved and the possible recovery scenarios are outlined the recovery plan (NMFS 2017a). At a minimum, at least one population in each MPG should achieve a very low risk of extinction. Of the 24 populations in this DPS, five pare tentatively rated at high risk of extinction, 17 are rated at moderate risk of extinction, one is viable, and one is highly viable. Four out of the five MPGs are not meeting the population viability goals laid out in the recovery plan (NMFS 2017a). In order for the species to recover, more populations will need to reach viable status through increases in abundance and productivity. Additionally, the relative proportion of hatchery fish spawning in natural spawning areas near major hatchery release sites remains uncertain and may need to be reduced (NWFSC 2015). Since 2015, abundance has declined steadily with only 10,717 natural-origin adult returns counted in 2018 (ODFW & WDFW 2019).	• Adverse effects related to the mainstem Columbia River hydropower system • Impaired tributary fish passage • Degraded freshwater habitat • Increased water temperature • Harvest-related effects, particularly for B-run steelhead • Predation • Genetic diversity effects from out-of-population hatchery releases

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Southern DPS of Green Sturgeon	Green sturgeon spend a substantial portion of their lives in marine waters. After reaching maturity (around 15 years of age), adults will typically spawn every 3 to 4 years. Adults return to the San Francisco Bay in later winter through early spring and spawn in the Sacramento River primarily from April through early July. It is thought that juveniles spend the first several months in freshwater environments prior to entering the estuary. Some individuals may enter the ocean and transition to the subadult life stage in their first year, but typical lengths of fish encountered in the ocean suggest later ocean entry. Adults and subadults migrate along the coast and congregate in estuarine waters.	The objective of the recovery plan is to increase the abundance, distribution, productivity, and diversity of this species by alleviating significant threats. Various demographic recovery criteria and threats-based recovery criteria are outlined in the recovery plan (NMFS 2018). The Sacramento River contains the only known green sturgeon spawning population in this DPS. The current estimate of spawning adult abundance is between 824-1,872 individuals. Telemetry data and genetic analyses suggest that Southern DPS green sturgeon generally occur from Graves Harbor, Alaska to Monterey Bay, California and, within this range, most frequently occur in coastal waters of Washington, Oregon, and Vancouver Island and near San Francisco and Monterey bays. Within the nearshore marine environment, tagging and fisheries data indicate that Northern and Southern DPS green sturgeon prefer marine waters of less than a depth of 110 meters.	• Reduction of its spawning area to a single known population • Lack of water quantity • Poor water quality • Poaching

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Southern DPS of Eulachon	Adult eulachon typically spawn at ages 2 to 5, in the lower portions of rivers that have prominent spring flow events. Adult migrations to spawning grounds typically occur between December and June. Immediately following hatch, larvae are transported rapidly by spring freshets to estuaries, and juveniles disperse onto the continental shelf within the first year of life.	There are many uncertainties about the distribution and abundance of eulachon in the marine environment and about how the species responds to shifts in freshwater and marine environmental conditions. As such, specific quantifiable recovery criteria specific to abundance, productivity, spatial diversity, and genetic diversity have not been developed. Instead, the current recovery approach at this time includes a set of priority actions to reduce the severity of threats to the species and to expand the research necessary to improve our understanding of the species. In the early 1990s, there was an abrupt decline in the abundance of eulachon returning to the Columbia River. Despite a brief period of improved returns in 2001-2003, the returns and associated commercial landings eventually declined to the low levels observed in the mid-1990s. Although eulachon abundance in monitored rivers has generally improved, especially in the 2013-2015 return years, recent poor ocean conditions and the likelihood that these conditions will persist into the near future suggest that population declines may be widespread in the upcoming return years (Gustafson et al. 2016).	• Changes in ocean conditions due to climate change, particularly in the southern portion of the species' range where ocean warming trends may be the most pronounced and may alter prey, spawning, and rearing success. • Climate-induced change to freshwater habitats • Bycatch of eulachon in commercial fisheries • Adverse effects related to dams and water diversions • Water quality, • Shoreline construction • Over harvest • Predation

Species	Life History	Recovery Objectives and Status Summary	Limiting Factors
Southern Resident Killer Whale	SRKWs exhibit advanced vocal communication and live in highly stable social groupings, or pods, led by matriarchal females. The whales use echolocation during foraging and feed primarily on salmonids. The DPS consists of three pods (J, K, and L) which inhabit coastal waters off Washington, Oregon, and Vancouver Island and are known to travel as far south as central California and as far north as Southeast Alaska. SRKWs occur primarily in the in the Salish Sea area of Washington State and British Columbia in the summer and fall. Satellite-linked tag data show K and L pods use the coastal waters along Washington, Oregon, and California during non-summer months. Sightings, tagging results, and acoustic detections of J pod indicate extensive occurrence in inland waters, particularly in the northern Georgia Strait, and limited occurrence along the outer coast.	The SRKWs are currently well below the population growth goals in the Recovery Plan (NMFS 2008). The population totaled 73 individuals as of December, 2019 with one whale now missing and presumed dead (Center for Whale Research, unpublished data), a decline from the 81 whales reported as of September 2013 (NWFSC 2014). Results of an updated population viability analysis project a downward trend in population size over the next 50 years, driven by the changing age and sex structure of the population, but also related to the relatively low fecundity rate observed 2011-2016 (NMFS 2016i). NMFS and WDFW (2018) developed a report identifying Chinook salmon stocks for which actions could be implemented to increase the availability of critical prey for SRKW along the West. Contaminants and pollution also affect the quality of SRKW prey in Puget Sound and in coastal waters of Washington, Oregon, and California. Chemical contamination of prey is a potential threat to SRKW critical habitat. Additional information on direct links between contaminant levels and physiological impacts will support recovery actions to reduce contaminant inputs into SRKW habitat. Federal vessel regulations established in 2011 prohibit vessels from approaching SRKWs and from parking in the path of SRKWs (within 200 and 400 yards respectively). Ferrara et al. (2017) assessed these for effectiveness and noted increasing awareness and enforcement of the regulations would help improve compliance and further reduce biological impacts to the whales.	• Prey availability • Environmental contaminants • Vessel effects and sound • Oil spills It is likely that multiple threats are acting together to impact the whales.

2.2.2 Status of Critical Habitat

In evaluating the condition of designated critical, NMFS examines the condition and trends of PBFs which are essential to the conservation of the ESA-listed species because they support one or more of the life stages of the species. Proper function of these PBFs is necessary to support successful adult and juvenile migration, adult holding, spawning, incubation, rearing, and growth and development of juvenile fish. Modification of these PBFs may affect freshwater spawning, rearing, or migration in the action area. Generally speaking, sites required to support one or more life stages of the ESA-listed species (i.e., sites for spawning, rearing, migration, foraging, estuarine areas, nearshore marine areas, and offshore marine areas) contain PBFs essential to the conservation of the listed species (e.g., spawning gravels, water quality and quantity, side channels, or food).

For most salmon and steelhead, NMFS critical habitat analytical review teams (CHARTs) ranked watersheds within designated critical habitat at the scale of the fifth-field hydrologic unit code (HUC5) in terms of the conservation value they provide to each ESA-listed species that they support (NMFS 2005). The conservation rankings were high, medium, or low. To determine the conservation value of each watershed to species viability, the CHARTs evaluated the quantity and quality of habitat features, the relationship of the area compared to other areas within the species' range, and the significance to the species of the population occupying that area. Even if a location had poor habitat quality, it could be ranked with a high conservation value if it were essential due to factors such as limited availability, a unique contribution of the population it served, or is serving another important role.

For southern DPS green sturgeon, a team similar to the CHARTs — a critical habitat review team (CHRT) — identified and analyzed the conservation value of particular areas occupied by southern green sturgeon, and unoccupied areas necessary to ensure the conservation of the species (74 FR 52300). The CHRT did not identify those particular areas using HUC nomenclature, but did provide geographic place names for those areas, including the names of freshwater rivers, the bypasses, the Sacramento-San Joaquin Delta, coastal bays and estuaries, and coastal marine areas (within 110 m depth) extending from the California/Mexico border north to Monterey Bay, California, and from the Alaska/Canada border northwest to the Bering Strait; and certain coastal bays and estuaries in California, Oregon, and Washington.

For southern DPS eulachon, critical habitat includes portions of 16 rivers and streams in California, Oregon, and Washington (76 FR 65324). We designated all of these areas as migration and spawning habitat for this species.

Critical habitat for the SRKW DPS was designated on November 29, 2006 (71 FR 69054) and includes approximately 2,560 square miles of inland waters of Washington in three specific areas: 1) Summer Core Area in Haro Strait and waters around the San Juan Islands; 2) Puget Sound; and 3) the Strait of Juan de Fuca. On September 19, 2019, NMFS proposed to revise the critical habitat designation for the SRKW DPS by designating six new areas along the U.S. West Coast (84 FR 49214). Specific new areas proposed along the U.S. West Coast include 15,626.6 square miles (mi^2) (40,472.7 square kilometers (km^2)) of marine waters between the 6.1-meter (m) depth contour and the 200-m depth contour from the U.S. international border with Canada south to Point Sur, California. In the proposed rule (84 FR 49214), NMFS states that the "proposed areas are occupied and contain PBFs that are essential to the conservation of the species and that may require special management considerations or protection." The three PBFs essential to conservation in the 2006 designated critical habitat were also identified for the six new areas along the U.S. West Coast. Those PBFs are: (1) Water quality to support growth and development; (2) prey species of sufficient quantity, quality and availability to support individual growth, reproduction and development, as well as overall population growth; and (3) passage conditions to allow for migration, resting, and foraging.

A summary of the status of critical habitats, considered in this opinion, is provided in Table 6 below.

Table 6. Critical habitat, designation date, federal register citation, and status summary for critical habitat considered in this Opinion.

Species	Designation Date and Federal Register Citation	Critical Habitat Status Summary
LCR Chinook Salmon	9/02/05 70 FR 52630	Critical habitat encompasses 10 subbasins in Oregon and Washington containing 47 occupied watersheds, as well as the lower Columbia River rearing/migration corridor. Most HUC5 watersheds with PBFs for salmon are in fair-to-poor or fair-to-good condition (NMFS 2005). However, most of these watersheds have some, or high potential for improvement. We rated conservation value of HUC5 watersheds as high for 30 watersheds, medium for 13 watersheds, and low for four watersheds. Removal of multiple barriers has improved access and allowed the restoration of hydrological processes that may improve downstream habitat conditions. However, the value of PBFs remains impaired by tributary barriers, loss of habitat complexity, toxics and water quality issues, concerns about predation during migration, and inundation of spawning sites by Bonneville Pool.
UCR Spring-run Chinook Salmon	9/02/05 70 FR 52630	Critical habitat encompasses four subbasins in Washington containing 15 occupied watersheds, as well as the Columbia River rearing/migration corridor. Most HUC5 watersheds with PBFs for salmon are in fair-to-poor or fair-to-good condition. However, most of these watersheds have some, or high, potential for improvement. We rated conservation value of HUC5 watersheds as high for 10 watersheds, and medium for five watersheds. Migratory habitat quality in this area has been severely affected by the development and operation of the dams and reservoirs of the Federal Columbia River Power System (FCRPS).
UWR Chinook Salmon	9/02/05 70 FR 52630	Critical habitat encompasses 10 subbasins in Oregon containing 56 occupied watersheds, as well as the lower Willamette/Columbia River rearing/migration corridor. Most HUC5 watersheds with PBFs for salmon are in fair-to-poor or fair-to-good condition. However, most of these watersheds have some, or high, potential for improvement. Watersheds are in good to excellent condition with no potential for improvement only in the upper McKenzie River and its tributaries (NMFS 2005). We rated conservation value of HUC5 watersheds as high for 22 watersheds, medium for 16 watersheds, and low for 18 watersheds. Major water storage and hydroelectric developments in the watershed have significantly reduced access to spawning habitat in four of the most historically productive basins. The lower Willamette/Columbia River rearing/migration corridor is considered to have a high conservation value. This corridor connects every population with the ocean and is used by juveniles and adults.
SRS Chinook Salmon	10/25/99 64 FR 57399	Critical habitat consists of river reaches of the Columbia, Snake, and Salmon Rivers, and all tributaries of the Snake and Salmon Rivers (except the Clearwater River) presently or historically accessible to this ESU (except reaches above impassable natural falls and Hells Canyon Dam). Habitat quality in tributary streams varies from excellent in wilderness and roadless areas, to poor in areas subject to heavy agricultural and urban development (Wissmar et al. 1994). Reduced summer stream flows, impaired water quality, and reduced habitat complexity are common problems. Migratory habitat quality in the lower Snake and Columbia Rivers has been severely affected by the development and operation of the dams and reservoirs of the FCRPS.

Species	Designation Date and Federal Register Citation	Critical Habitat Status Summary
SRF Chinook Salmon	10/25/99 64 FR 57399	Critical habitat consists of all Columbia River estuarine areas, as well as river reaches upstream to the confluence of the Columbia and Snake Rivers, and all Snake River reaches from the confluence of the Columbia River upstream to Hells Canyon Dam. It also includes lower portions of the Palouse, Clearwater, and North Fork Clearwater Rivers. Habitat quality in all reaches is influenced by various land uses, especially irrigated agriculture, in terms of heavy sediment and nutrient loading from irrigation returns (NMFS 2017b). Migratory habitat quality has been severely affected by the development and operation of the dams and reservoirs of the FCRPS.
CR Chum Salmon	9/02/05 70 FR 52630	Critical habitat encompasses six subbasins in Oregon and Washington containing 19 occupied watersheds, as well as the lower Columbia River rearing/migration corridor. Most HUC5 watersheds with PBFs for salmon are in fair-to-poor or fair-to-good condition (NMFS 2005). However, most of these watersheds have some or a high potential for improvement. We rated conservation value of HUC5 watersheds as high for 16 watersheds, and medium for three watersheds. Key habitat concerns for this species include: (1) Reduced habitat complexity, connectivity, quantity, and quality of habitat used for spawning, rearing, foraging, and migration; and (2) toxic contamination of surface water through the production, use, and disposal from anthropogenic activities.
LCR Coho Salmon	2/24/16 81 FR 9252	Critical habitat encompasses 10 subbasins in Oregon and Washington containing 55 occupied watersheds, as well as the LCRE. The LCR rearing/migration corridor downstream of the spawning range is considered to have a high conservation values. Most HUC5 watersheds with PBFs for salmon are in fair-to-poor or fair-to-good condition (NMFS 2005). However, most of these watersheds have some or a high potential for improvement. We rated conservation value of HUC5 watersheds as high for 34 watersheds, medium for 18 watersheds, and low for three watersheds. Key habitat concerns for this species include: (1) Reduced habitat complexity, connectivity, quantity, and quality of habitat used for spawning, rearing, foraging, and migration; and (2) toxic contamination of surface water through the production, use, and disposal from anthropogenic activities.
ORC Coho Salmon	2/11/08 73 FR 7816	Critical habitat encompasses 13 subbasins in Oregon. The long-term decline in Oregon Coast coho salmon productivity reflects deteriorating conditions in freshwater habitat as well as extensive loss of access to habitats in estuaries and tidal freshwater. Many of the habitat changes resulting from land use practices over the last 150 years that contributed to the ESA-listing of OC coho salmon continue to hinder recovery of the populations; changes in the watersheds due to land use practices have weakened natural watershed processes and functions, including loss of connectivity to historical floodplains, wetlands and side channels; reduced riparian area functions (stream temperature regulation, wood recruitment, sediment and nutrient retention); and altered flow and sediment regimes (NMFS 2016e). Several historical and ongoing land uses have reduced stream capacity and complexity in Oregon coastal streams and lakes through disturbance, road building, splash damming, stream cleaning, and other activities. Beaver removal, combined with loss of large wood in streams, has also led to degraded stream habitat conditions for coho salmon (Stout et al. 2012).

Species	Designation Date and Federal Register Citation	Critical Habitat Status Summary
SONCC Coho Salmon	5/5/99 64 FR 24049	Critical habitat includes all areas accessible to any life-stage up to long-standing, natural barriers and adjacent riparian zones. Southern Oregon Northern California Coast (SONCC) coho salmon critical habitat within this geographic area has been degraded from historical conditions by ongoing land management activities. Habitat impairments recognized as factors leading to decline of the species that were included in the original listing notice for SONCC coho salmon include: (1) Channel morphology changes; (2) substrate changes; (3) loss of in-stream roughness; (4) loss of estuarine habitat; (5) loss of wetlands; (6) loss/degradation of riparian areas; (7) declines in water quality; (8) altered stream flows; (9) fish passage impediments; and (10) elimination of habitat.
SR Sockeye Salmon	10/25/99 64 FR 57399	Critical habitat consists of river reaches of the Columbia, Snake, and Salmon Rivers; Alturas Lake Creek; Valley Creek; and Stanley, Redfish, Yellow Belly, Pettit, and Alturas Lakes (including their inlet and outlet creeks). Water quality in all five lakes generally is adequate for juvenile sockeye salmon, although zooplankton numbers vary considerably. Some reaches of the Salmon River and tributaries exhibit temporary elevated water temperatures and sediment loads that could restrict sockeye salmon production and survival (NMFS 2015a). Migratory habitat quality in the lower Snake River and Columbia River has been severely affected by the development and operation of the dams and reservoirs of the FCRPS.
LCR Steelhead	9/02/05 70 FR 52630	Critical habitat encompasses nine subbasins in Oregon and Washington containing 41 occupied watersheds, as well as the LCR rearing/migration corridor. Most HUC5 watersheds with PBFs for salmon are in fair-to-poor or fair-to-good condition (NMFS 2005). However, most of these watersheds have some or a high potential for improvement. We rated conservation value of HUC5 watersheds as high for 28 watersheds, medium for 11 watersheds, and low for two watersheds. Removal of multiple barriers has improved access and allowed the restoration of hydrological processes that may improve downstream habitat conditions. However, the value of PBFs remains impaired by tributary barriers, loss of habitat complexity, toxics and water quality issues, and concerns about predation during migration.
MCR Steelhead	9/02/05 70 FR 52630	Critical habitat encompasses 15 subbasins in Oregon and Washington containing 111 occupied watersheds, as well as the Columbia River rearing/migration corridor. Most HUC5 watersheds with PBFs for salmon are in fair-to-poor or fair-to-good condition (NMFS 2005). However, most of these watersheds have some or a high potential for improvement. We rated conservation value of occupied HUC5 watersheds as high for 80 watersheds, medium for 24 watersheds, and low for 9 watersheds. Habitat quality in tributary streams varies from excellent in wilderness and roadless areas, to poor in areas subject to heavy agricultural and urban development (Wissmar et al. 1994). Lack of stream flows, impaired water quality, and reduction of habitat complexity are common problems. Migratory habitat quality has been severely affected by the development and operation of the dams and reservoirs of the FCRPS.

Species	Designation Date and Federal Register Citation	Critical Habitat Status Summary
UCR Steelhead	9/02/05 70 FR 52630	Critical habitat encompasses 10 subbasins in Washington containing 31 occupied watersheds, as well as the Columbia River rearing/migration corridor. Most HUC5 watersheds with PBFs for salmon are in fair-to-poor or fair-to-good condition (NMFS 2005). However, most of these watersheds have some or a high potential for improvement. We rated conservation value of HUC5 watersheds as high for 20 watersheds, medium for eight watersheds, and low for three watersheds. Migratory habitat quality in this area has been severely affected by the development and operation of the dams and reservoirs of the FCRPS.
UWR Steelhead	9/02/05 70 FR 52630	Critical habitat encompasses seven subbasins in Oregon containing 34 occupied watersheds, as well as the lower Willamette/Columbia River rearing/migration corridor. Most HUC5 watersheds with PBFs for salmon are in fair-to-poor or fair-to-good condition (NMFS 2005). However, most of these watersheds have some or a high potential for improvement. Watersheds are in good to excellent condition with no potential for improvement only in the upper McKenzie River and its tributaries (NMFS 2005). We rated conservation value of HUC5 watersheds as high for 25 watersheds, medium for 6 watersheds, and low for 3 watersheds. Major water storage and hydroelectric developments in the watershed have significantly reduced access to spawning habitat in four of the most historically productive basins. The lower Willamette/Columbia River rearing/migration corridor is considered to have a high conservation value. This corridor connects every population with the ocean and is used by juveniles and adults.
SRB Steelhead	9/02/05 70 FR 52630	Critical habitat encompasses 25 subbasins in Oregon, Washington, and Idaho. Habitat quality in tributary streams varies from excellent in wilderness and roadless areas, to poor in areas subject to heavy agricultural and urban development (Wissmar et al. 1994). Reduced summer stream flows, impaired water quality, and reduced habitat complexity are common problems. Migratory habitat quality in this area has been severely affected by the development and operation of the dams and reservoirs of the FCRPS.

Species	Designation Date and Federal Register Citation	Critical Habitat Status Summary
sDPS Green Sturgeon	10/09/09 74 FR 52300	Critical habitat has been designated in coastal U.S. marine waters within 60 fathoms depth from Monterey Bay, California (including Monterey Bay), north to Cape Flattery, Washington, including the Strait of Juan de Fuca, Washington, to its United States boundary; the Sacramento River, lower Feather River, and lower Yuba River in California; the Sacramento-San Joaquin Delta and Suisun, San Pablo, and San Francisco bays in California; tidally influenced areas of the LCRE from the mouth upstream to river mile 46; and certain coastal bays and estuaries in California (Humboldt Bay), Oregon (Coos Bay, Winchester Bay, Yaquina Bay, and Nehalem Bay), and Washington (Willapa Bay and Grays Harbor), including, but not limited to, areas upstream to the head of tide in various streams that drain into the bays, as listed in Table 1 in USDC (2009). The CHRT identified several activities that threaten the PBFs in coastal bays and estuaries and necessitate the need for special management considerations or protection. The application of pesticides is likely to adversely affect prey resources and water quality within the bays and estuaries, as well as the growth and reproductive health of sDPS green sturgeon through bioaccumulation. Other activities of concern include those that disturb bottom substrates, adversely affect prey resources, or degrade water quality through resuspension of contaminated sediments. Of particular concern are activities that affect prey resources. Prey resources are affected by: commercial shipping and activities generating point source pollution and non-point source pollution that discharge contaminants and result in bioaccumulation of contaminants in green sturgeon; disposal of dredged materials that bury prey resources; and bottom trawl fisheries that disturb the bottom (but result in beneficial or adverse effects on prey resources for green sturgeon).
sDPS Eulachon	10/20/11 76 FR 65324	Critical habitat for eulachon includes portions of 16 rivers and streams in California, Oregon, and Washington. All of these areas are designated as migration and spawning habitat for this species. Dams and water diversions are moderate threats to eulachon in the Columbia and Klamath Rivers where hydropower generation and flood control are major activities. Degraded water quality is common in some areas occupied by southern DPS eulachon. In the Columbia and Klamath River basins, large-scale impoundment of water has increased winter water temperatures, potentially altering the water temperature during eulachon spawning periods. Numerous chemical contaminants are also present in spawning rivers, but the exact effect these compounds have on spawning and egg development is unknown. Dredging is a low to moderate threat to eulachon in the Columbia River. Dredging during eulachon spawning would be particularly detrimental.

Species	Designation Date and Federal Register Citation	Critical Habitat Status Summary
SRKW	11/29/2016 (71 FR 69054) Proposed 9/19/2019 (84 FR 49214)	Critical habitat includes approximately 2,560 square miles of inland waters of Washington in three specific areas: 1) Summer Core Area in Haro Strait and waters around the San Juan Islands; 2) Puget Sound; and 3) Strait of Juan de Fuca. Based on the natural history of SRKWs and their habitat needs, NMFS identified water quality, prey, and passage conditions as the PBFs essential to conservation. In 2006, few data were available on SRKWs distribution and habitat use in coastal waters of the Pacific Ocean. Since the 2006 designation, additional effort has been made to better understand the geographic range and movements of SRKWs, including opportunistic visual sightings, satellite tracking, and passive acoustic research conducted that provide an updated estimate of the whales' coastal range from the Monterey Bay area in California, north to Chatham Strait in southeast Alaska (NMFS 2019a). Six new areas along the U.S. West Coast (84 FR 49214) are included in the proposed revisions to the critical habitat designation. Prey quantity is reduced with some wild salmon stocks throughout the whales' geographic range are at a fraction of historic levels. Past overfishing, habitat losses, and hatchery practices were major causes of decline. Poor ocean conditions over the past two decades have reduced populations already weakened by the degradation and loss of freshwater and estuary habitat, fishing, hydropower system management, and hatchery practices. While wild salmon stocks have declined in many areas, hatchery production has been generally strong. Water quality is especially important in high-use areas where foraging behaviors occur and contaminants can enter the food chain. Water quality varies in coastal waters from Washington to California. Toxicants in Puget Sound persist and build up in SRKWs and their prey resources, despite bans in the 1970s of some harmful substances and cleanup efforts. High levels of DDTs have been found in SRKWs, especially in K and L pods (NMFS 2019a). High-volume spills off the California coast released over 650,000 gallons of crude oil between 2008 and 2016; no crude oil spills were reported off the coasts of Oregon or Washington in the same years (Stephens 2017). Passage is affected by activities that interfere with movements of the whales. In particular, vessels may present obstacles to whale passage, causing the whales to swim further and change direction more often, which can increase energy expenditure for whales and impacts foraging behavior.

2.2.3 Climate Change Implications for ESA-listed Species and their Critical Habitat

Climate change is likely to play an increasingly important role in determining the abundance and distribution of ESA-listed species, and the conservation value of designated critical habitats, in the Pacific Northwest. The U. S. Global Change Research Program reports average warming of about 1.3°F from 1895 to 2011, and projects an increase in average annual temperature of 3.3°F to 9.7°F by 2070 to 2099 in the Pacific Northwest (Mote et al. 2014). According to the Independent Science Advisory Board (ISAB) (ISAB 2007), climate change will cause the following:

- Warmer air temperatures will result in diminished snowpack and a shift to more winter/spring rain and runoff, rather than snow that is stored until the spring/summer melt season;

- With a smaller snowpack, watersheds will see their runoff diminished earlier in the season, resulting in lower flows in the June through September period, while more precipitation falling as rain rather than snow will cause higher flows in winter, and possibly higher peak flows; and,

- Water temperatures are expected to rise, especially during the summer months when lower flows co-occur with warmer air temperatures.

These changes will not be spatially homogeneous across the entire Pacific Northwest. The largest hydrologic responses are expected to occur in basins with significant snow accumulation, where warming decreases snow pack, increases winter flows, and advances the timing of spring melt (Mote et al. 2014; Mote et al. 2016). Rain-dominated watersheds and those with significant contributions from groundwater may be less sensitive to predicted changes in climate (Tague et al. 2013; Mote et al. 2014).

Climate change is predicted to cause a variety of impacts to Pacific salmon (including steelhead) and their ecosystems (Mote et al. 2003; Crozier et al. 2008a; Martins et al. 2012; Wainwright and Weitkamp 2013). Examples of long-term impacts include, but are not limited to: depletion of important cold-water habitat; variation in quality and quantity of tributary rearing habitat; alterations to migration patterns; accelerated embryo development; premature emergence of fry; and increased competition among species. The complex life cycles of anadromous fishes, including salmon, rely on productive freshwater, estuarine, and marine habitats for growth and survival, making them particularly vulnerable to environmental variation. Ultimately, the effects of climate change on fishes across the Pacific Northwest will be determined by the specific nature, level, and rate of change and the synergy between interconnected terrestrial/freshwater, estuarine, nearshore, and ocean environments.

The primary effects of climate change on Pacific Northwest fishes include:

- Direct effects of increased water temperatures on fish physiology;

- Temperature-induced changes to streamflow patterns;

- Alterations to freshwater, estuarine, and marine food webs; and,
- Changes in estuarine and ocean productivity.

While all habitats used by ESA-listed fish will be affected, the impacts and certainty of the change vary by habitat type. Some effects (e.g., increasing temperature) affect all life stages in all habitats, while others are habitat-specific, such as streamflow variation in freshwater, sea-level rise in estuaries, and upwelling in the ocean. How climate change will affect each stock or population of salmon also varies widely depending on the level or extent of change, the rate of change, and the unique life-history characteristics of different natural populations (Crozier et al. 2008b). For example, a few weeks' difference in migration timing can have large differences in the thermal regime experienced by migrating fish (Martins et al. 2011).

Temperature Effects. Salmon, steelhead, eulachon, and green sturgeon are poikilotherms (cold-blooded animals); therefore, increasing temperatures in all habitats can have pronounced effects on their physiology, growth, and development rates (see review by Whitney et al. 2016). Increases in water temperatures beyond their thermal optima will likely be detrimental through a variety of processes, including increased metabolic rates (and therefore food demand), decreased disease resistance, increased physiological stress, and reduced reproductive success. All of these processes are likely to reduce survival (Beechie et al. 2013; Wainwright and Weitkamp 2013; Whitney et al. 2016).

By contrast, increased temperatures at ranges well below thermal optima (i.e., when the water is cold) can increase growth and development rates. Examples of this include accelerated emergence timing during egg incubation stages, or increased growth rates during fry stages (Crozier et al. 2008a; Martins et al. 2011). Temperature is also an important behavioral cue for migration (Sykes et al. 2009), and elevated temperatures may result in earlier-than-normal migration timing. While there are situations or stocks where this acceleration in processes or behaviors is beneficial, there are also others where it is detrimental (Martins et al. 2012; Whitney et al. 2016).

Freshwater Effects. Climate change is predicted to increase the intensity of storms, reduce winter snow pack at low and middle elevations, and increase snowpack at high elevations in northern areas. Middle and lower-elevation streams will have larger fall/winter flood events and lower late-summer flows, while higher elevations may have higher minimum flows. How these changes will affect freshwater ecosystems largely depends on their specific characteristics and location, which vary at fine spatial scales (Crozier et al. 2008b; Martins et al. 2012). Salmon populations inhabiting regions that are already near or exceeding thermal maxima will be most affected by further increases in temperature and, perhaps, the rate of the increases. The effects of altered flow are less clear and likely to be basin-specific (Crozier et al. 2008b; Beechie et al. 2013). However, flow is already becoming more variable in many rivers, and this increased variability is believed to negatively affect anadromous fish survival more than other environmental parameters (Ward et al. 2015). It is likely this increasingly variable flow is detrimental to multiple salmon and steelhead populations, and also to other freshwater fish species in the Columbia River basin.

Stream ecosystems will likely change in response to climate change in ways that are difficult to predict (Lynch et al. 2016). Changes in stream temperature and flow regimes will likely lead to shifts in the distributions of native species and provide "invasion opportunities" for exotic species. This will result in novel species interactions, including predator-prey dynamics, where juvenile native species may be either predators or prey (Lynch et al. 2016; Rehage and Blanchard 2016). How juvenile native species will fare as part of "hybrid food webs," which are constructed from natives, native invaders, and exotic species, is difficult to predict (Naiman et al. 2012).

Estuarine Effects. In estuarine environments, the two big concerns associated with climate change are rates of sea level rise and water temperature warming (Wainwright and Weitkamp 2013; Limburg et al. 2016). Estuaries will be affected directly by sea-level rise: as sea level rises, terrestrial habitats will be flooded and tidal wetlands will be submerged (Kirwan et al. 2010; Wainwright and Weitkamp 2013; Limburg et al. 2016). The net effect on wetland habitats depends on whether rates of sea-level rise are sufficiently slow that the rates of marsh plant growth and sedimentation can compensate (Kirwan et al. 2010).

Due to subsidence, sea-level rise will affect some areas more than others, with the largest effects expected for the lowlands, like southern Vancouver Island and central Washington coastal areas (Verdonck 2006; Lemmen et al. 2016). The widespread presence of dikes in Pacific Northwest estuaries will restrict upward estuary expansion as sea levels rise, likely resulting in a near-term loss of wetland habitats (Wainwright and Weitkamp 2013). Sea-level rise will also result in greater intrusion of marine water into estuaries, resulting in an overall increase in salinity, which will also contribute to changes in estuarine floral and faunal communities (Kennedy 1990). While not all anadromous fish species are highly reliant on estuaries for rearing, extended estuarine use may be important in some populations (Jones et al. 2014), especially if stream habitats are degraded and become less productive. Preliminary data indicate that some Snake River Basin steelhead smolts actively feed and grow as they migrate between Bonneville Dam and the ocean (Beckman 2018), suggesting that estuarine habitat is important for this DPS.

Marine Effects. In marine waters, increasing temperatures are associated with observed and predicted poleward range expansions of fish and invertebrates in both the Atlantic and Pacific Oceans (Lucey and Nye 2010; Asch 2015; Cheung et al. 2015). Rapid poleward species shifts in distribution in response to anomalously warm ocean temperatures have been well documented in recent years, confirming this expectation at short time scales. Range extensions were documented in many species from southern California to Alaska during unusually warm water associated with "the blob" in 2014 and 2015 (Bond et al. 2015; Di Lorenzo and Mantua 2016) and past strong El Niño events (Pearcy 2002; Fisher et al. 2015). For example, recruitment of the introduced European green crab (*Carcinus maenas*) increased in Washington and Oregon waters during winters with warm surface waters, including 2014 (Yamada et al. 2015). Similarly, the Humboldt squid (*Dosidicus gigas*) dramatically expanded its range northward during warm years of 2004–09 (Litz et al. 2011). The frequency of extreme conditions, such as those associated with El Niño events or "blobs" is predicted to increase in the future (Di Lorenzo and Mantua 2016), further altering food webs and ecosystems.

Expected changes to marine ecosystems due to increased temperature, altered productivity, or acidification will have large ecological implications through mismatches of co-evolved species and unpredictable trophic effects (Cheung et al. 2015; Rehage and Blanchard 2016). These effects will certainly occur, but predicting the composition or outcomes of future trophic interactions is not possible with current models.

Wind-driven upwelling is responsible for the extremely high productivity in the California Current ecosystem (Bograd et al. 2009; Peterson et al. 2014). Minor changes to the timing, intensity, or duration of upwelling, or the depth of water-column stratification, can have dramatic effects on the productivity of the ecosystem (Black et al. 2015; Peterson et al. 2014). Current projections for changes to upwelling are mixed: some climate models show upwelling unchanged, but others predict that upwelling will be delayed in spring, and more intense during summer (Rykaczewski et al. 2015). Should the timing and intensity of upwelling change in the future, it may result in a mismatch between the onset of spring ecosystem productivity and the timing of fish entry into the ocean, and a shift toward food webs with a strong sub-tropical component (Bakun et al. 2015).

Columbia River anadromous fishes also use coastal areas of British Columbia and Alaska and midocean marine habitats in the Gulf of Alaska, although their fine-scale distribution and marine ecology during this period are poorly understood (Morris et al. 2007; Pearcy and McKinnell 2007). Increases in temperature in Alaskan marine waters have generally been associated with increases in productivity and salmon survival (Mantua et al. 1997; Martins et al. 2012), thought to result from temperatures that are normally below thermal optima (Gargett 1997). Warm ocean temperatures in the Gulf of Alaska are also associated with intensified downwelling and increased coastal stratification, which may result in increased food availability to juvenile salmon along the coast (Hollowed et al. 2009; Martins et al. 2012). Predicted increases in freshwater discharge in British Columbia and Alaska may influence coastal current patterns (Foreman et al. 2014), but the effects on coastal ecosystems are poorly understood.

In addition to becoming warmer, the world's oceans are becoming more acidic as increased atmospheric carbon dioxide is absorbed by water. The North Pacific is already acidic compared to other oceans, making it particularly susceptible to further increases in acidification (Lemmen et al. 2016). Laboratory and field studies of ocean acidification show that it has the greatest effects on invertebrates with calcium-carbonate shells, and has relatively little direct influence on finfish; see reviews by Haigh et al. (2015) and Mathis et al. (2015). Consequently, the largest impact of ocean acidification on salmon will likely be the influence on marine food webs, especially the effects on lower trophic levels (Haigh et al. 2015; Mathis et al. 2015). Marine invertebrates fill a critical gap between freshwater prey and larval and juvenile marine fishes, supporting juvenile salmon growth during the important early-ocean residence period (Daly et al. 2009, 2014).

The potential impacts of climate and oceanographic change on whales and other marine mammals will likely involve effects on habitat availability and food availability. For species that depend on salmon for prey, such as SRKWs, the fluctuations in salmon survival that occur with these changes in climate conditions can have negative effects. Site selection for migration, feeding, and breeding may be influenced by factors such as ocean currents and water temperature. For example, there is some evidence from Pacific equatorial waters that sperm whale feeding success and, in turn, calf production rates are negatively affected by increases in sea surface temperature (Smith and Whitehead 1993; Whitehead 1997). Different species of marine mammals will likely react to these changes differently. MacLeod (2009) estimated, based on expected shifts in water temperature, 88% of cetaceans would be affected by climate change, with 47% likely to be negatively affected. Range size, location, and whether or not specific range areas are used for different life history activities (e.g., feeding, breeding) are likely to affect how each species responds to climate change (Learmonth et al. 2006). Although no formal predictions of impacts on the SRKWs have yet been made, it seems likely that any changes in weather and oceanographic conditions resulting in effects on salmon populations will have consequences for the whales.

Summary. Considering all of the potential impacts described above, climate change is expected to make recovery targets for ESA-listed fish more difficult to achieve. 01064247 actions can address the adverse impacts of climate change and improve resilience of species as their habitats change. Examples include restoring connections to historical floodplains and freshwater and estuarine habitats to provide fish refugia and areas to store excess floodwaters, protecting and restoring riparian vegetation to ameliorate stream temperature increases, and purchasing or applying easements to lands that provide important cold water or refuge habitat (Battin et al. 2007; ISAB 2007).

2.3 Action Area

"Action area" means 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 for anadromous species considered in this consultation includes the freshwater and estuarine areas[4] subject to the jurisdiction of the State of Oregon, where the criteria apply, as well as areas beyond the state's jurisdiction where the regulated pollutants are likely to be transported. More specifically, the action area includes all inland basins in Oregon that provide access to the species considered in this Opinion, including the Columbia River from the mouth to the Washington-Oregon border (river mile [RM] 292) and the Snake River from RM 169 to RM 247.5. In addition, because the action may reduce the prey base for SRKW, the action area for SRKW includes the Pacific Ocean, limited to the entire coastal range from California to Vancouver, British Columbia, where the marine ranges of prey species subject to this consultation overlap with SRKW.

The species and designated critical habitats occurring within the action area are listed in Table 4. The action area is illustrated in Figures 1 and 2. The life stage and extent to which species use the action area are described in the environmental baseline. The action area, except for areas above natural barriers to fish passage, also contains EFH for Chinook and coho salmon (PFMC 2014) and is in an area where environmental effects of the proposed project may adversely affect EFH for this species. Subbasins in the action area that contain EFH are illustrated in Figure 1.

[4] As described in Section 1.3, the ODEQ may apply the criteria in estuarine waters (where there are currently no EPA-approved saltwater criteria for aluminum) if the pH, DOC, and hardness values are within the bounds of the criteria model. Therefore, the action area includes estuarine areas.

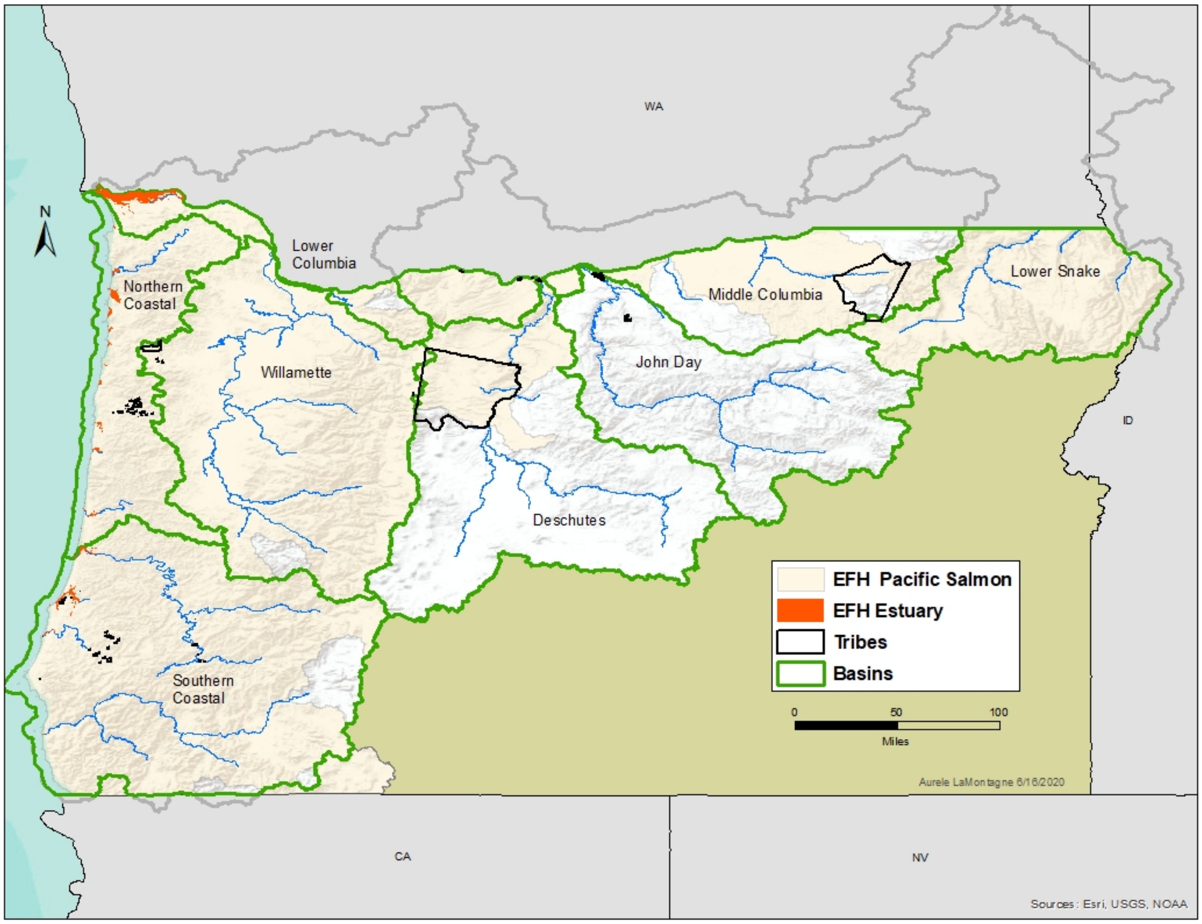

Figure 1. Overview of the action area for anadromous species, including EFH. Boundaries of Federally-recognized Tribes are also shown to illustrate areas that are not subject to this consultation.

Figure 2. Action area (light shading) for SRKW. Reprinted from Wiles (2004).

2.4 Environmental Baseline

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

2.4.1 Presence of ESA-listed Species and Critical Habitat in the Action Area

There are eighteen ESA-listed species in the action area (Table 4). The ESU delineation for three of these species (i.e., UWR Chinook, UWR steelhead, and OC coho) occurs entirely within Oregon. Conversely there are four species (i.e., UCR spring-run Chinook, UCR steelhead, SR sockeye, sDPS green sturgeon) that do not spawn in Oregon waters, but instead use the Columbia River and the Lower Columbia River Estuary (LCRE) for migration and/or rearing. The remaining species include populations in Oregon and its bordering states (e.g., Washington, Oregon, and/or Idaho). Table 7 identifies each fish species' populations that spawn in Oregon and provides a brief summary of how each species typically uses Oregon waters. A brief description about SRKW presence in the action area is provided at the end of this section.

All salmonid species originating from the Columbia River basin considered in this Opinion must move through the LCRE. Residency and habitat use within the LCRE varies among species and stocks and their associated life history characteristics. In general terms, larger-sized juveniles (e.g., yearlings) tend to utilize the main channel and generally migrate through the LCRE relatively rapidly (e.g., two weeks or less) whereas smaller fish generally use off-channel and shallow-waters shorelines and may spend longer periods of time in these areas. Yet, we cannot discount the use of various habitat types by older, larger juveniles nor can we discount the use of the main channel by subyearlings. McComas et al. (2008) implanted yearling and subyearling Chinook salmon at Bonneville Dam with acoustic and passive integrated transponder (PIT) tags. The mean travel time from release at Bonneville Dam through the mouth of the LCRE was 4.1 days for both yearling and subyearling Chinook salmon. McNatt et al. (2016) documented subyearling Chinook salmon residing in wetland and off-channel habitats of the LCRE for 2 to 4 weeks. The authors noted that this may be a conservative estimate of residency given the duration of the study and recognized the possibility that juveniles may reside for longer periods of time in the LCRE. Johnson et al. (2015) documented two life history strategies in the LCRE: (1) Active migration (i.e., short residence times in the LCRE) by upper river Chinook salmon and steelhead during the primary spring and summer migration periods; and (2) overwinter rearing in tidal habitats by coho salmon and natural-origin Chinook from stocks below Bonneville Dam. Recent studies have documented some yearling salmon and steelhead in shallow-water shorelines or off-channel habitats (Hanson et al. 2015; McNatt et al. 2017; Harnish et al. 2012; McMichael et al. 2010; Rose et al. 2015; and Weitkamp et al. 2015). Overall, how the various species use the LCRE is complex and although recent research efforts are improving our understanding, how each species utilize the LCRE is not well understood.

Table 7. Summary of each species' extent of use of waters in the action area, including a list of populations that spawn in the action area.

Species (Total # of Populations)	Populations that Spawn in Oregon	Extent of Use of Oregon Waters
LCR Chinook (32)	Youngs Bay (fall); Big Creek (fall); Clatskanie River (fall); Scappoose Creek (fall); Clackamas River (fall); Sandy River (spring, fall, late fall); Lower Gorge Tribs (fall); Upper Gorge Tribs (fall); and Hood River (spring; fall).	Major Oregon rivers designated as critical habitat for this species include, but are not limited to, the Columbia River (including its estuary), Hood River, Sandy River, Clackamas River, and the Willamette River. Spawning occurs in tributaries to the Columbia River, from its mouth upstream to above Hood River. Populations from within Oregon as well as populations originating outside of Oregon use the LCRE for juvenile rearing and for both adult and juvenile migration. Teel et al. (2009) documented subyearling LCR Chinook in seasonal floodplain wetlands near the confluence of the Willamette and Columbia Rivers. Johnson et al. (2015) found that some fish may overwinter in tidally-influenced freshwater of the LCRE.
UCR Spring-run Chinook (3)	None	Within the action area, critical habitat for this species includes the Columbia River and the LCRE. All populations of this species must migrate past the four lower Columbia River dams. The ten-year (2008-2017) average minimum survival estimates for migrating adult fish between Bonneville and McNary Dams is 91.5 percent (range of 80.4 to 105.1) (NMFS 2019b). The average upstream migration time from Bonneville to McNary ranges from 5 to 10 days (Columbia River DART). Juvenile and adult UCR spring-run Chinook salmon migrate through the Columbia River and its estuary. The average migration time between McNary and Bonneville Dams for juveniles has ranged from 4-6 days (Columbia River DART). Little is known about juvenile use of the LCRE; however, Johnson et al. (2015) reported that the majority of fish from stocks upstream of Bonneville Dam exhibited relatively quick migrations through the LCRE. Adults took an average of 30 to 40 days to pass through the LCRE (Sorel et al. 2017). The ISAB (2018) concluded adults may spend over a month in this part of the river.

Species (Total # of Populations)	Populations that Spawn in Oregon	Extent of Use of Oregon Waters
UWR Chinook (7)	Clackamas Molalla North Santiam South Santiam Calapooia McKenzie Middle Fork Willamette	Major Oregon rivers designated as critical habitat for this species include, but are not limited to, the Columbia River and its estuary, Willamette River, Clackamas River, and Mackenzie River. Adult fish begin to arrive in the lower Willamette River in January, with the majority of the run passing over Willamette Falls from late April through May. Spawning typically occurs in the upper portions of the larger subbasins. Juvenile emigration varies with environmental conditions and includes: (1) Late winter to early spring as fry; (2) fall to early winter as fingerlings; and (3) late winter through spring as yearlings. Most fry and fingerlings migrate to lower reaches of tributaries or the mainstem Willamette River for rearing in late winter and early spring (Schroeder et al. 2005, 2007). Friesen et al. (2007) reported subyearling Chinook in the mainstem river below Willamette Falls soon after hatching. Teel et al. (2009) reported subyearling UWR Chinook rearing in seasonal floodplains near the confluence of the Willamette and Columbia Rivers. Subyearlings that grow quickly migrate out of the Willamette Basin as subyearling smolts. These fish have been detected in the upper reaches of the LCRE in May and in near-shore ocean locations in June. Yearlings are thought to migrate out of the Willamette River basin and the LCRE relatively quickly. However, Rose et al. (2015) found the yearlings may use the tidally influenced Columbia River for extended periods before entering the ocean.
SRS Chinook (28 extant, 3 functionally extirpated)	Wenaha River Minam River Lostine/Wallowa Rivers Catherine Creek Upper Grande Ronde River Imnaha River *Lookingglass Creek* *Big Sheep Creek*	Major Oregon rivers designated as critical habitat for this species include, but are not limited to, the Columbia River (and its estuary), Snake River, Grande Ronde River, and Imnaha River. All populations of this species must migrate past the four lower Columbia River dams. The ten-year (2008-2017) average minimum survival estimates for migrating adult fish between Bonneville and McNary Dams is 88.7 percent (range of 82.8 to 100) (NMFS 2019b). The average upstream migration time from Bonneville to McNary ranges from 6 to 8 days (Columbia River DART). Juveniles migrate from McNary Dam to Bonneville Dam in approximately 4 to 6 days (Columbia River DART). Juveniles migrate as yearlings and the majority of these fish are thought to move through the LCRE fairly quickly – a week or less (NMFS 2017a; Fresh et al. 2014).

Species (Total # of Populations)	Populations that Spawn in Oregon	Extent of Use of Oregon Waters
SR Fall Chinook (1)	Lower Snake River	Within the action area, designated critical habitat for this species includes the Columbia River (including its estuary), Snake River, Grande Ronde River, and Imnaha River. The single extant population must migrate past the four lower Columbia River dams that border Oregon and Washington. Spawning occurs in the Snake River along the Oregon/Idaho border and in the lower reaches of the Imnaha River (NMFS 2017b). This species primarily uses the mainstem Columbia River as a migration corridor; although some subyearlings may rear for extended periods of time in reservoirs. Juvenile use of the LCR and the LCRE is complex. There is little evidence of extended rearing (weeks to months) by yearling SR fall Chinook in the estuary. In contrast, subyearlings may rear for extended periods of time.
CR Chum (17)	Youngs Bay Big Creek Clatskanie Scappoose Clackamas Sandy Lower Gorge Upper Gorge	Critical habitat within the action area includes the Columbia River, its estuary, and Big Creek. Adult chum spawn in the mainstem Columbia River and the lower reaches of larger tributaries. A large spawning aggregate occurs at the Ives/Pierce Island complex in the Bonneville tailrace. Only the Upper Gorge population passes over Bonneville Dam. Juvenile chum are believed to migrate downstream soon after emergence and rear in the LCR and its estuary. Salt marshes, tidal creeks, and intertidal flats are significant rearing areas for juveniles during their estuarine residence (which is thought to last from weeks to months) (NMFS 2013)
SONCC Coho (40)	Elk River Brush Creek Mussel Creek Lower Rogue Hunter Creek Chetco River Pistol River Winchuck River Hubbard Creek Euchre Creek Smith River	The Iron Gate dam prevents up-river migration of SONCC coho salmon across the Oregon-California border. Iron Gate Dam is located on the Klamath River at RM 190.2 in California. Considering their geographic location, NMFS determined that individuals of populations in the Klamath or Trinity strata are not at risk of direct exposure to aluminum in association with this action. Within the action area, critical habitat includes, but is not limited to, the following major rivers: Rogue River, Applegate River, and the Illinois River. Most of the populations in Oregon emigrate through the Rogue River estuary. The majority of juveniles emigrate as yearlings and typically move through the estuaries relatively quickly; however, some life history strategies involve extensive rearing in the estuary.

Species (Total # of Populations)	Populations that Spawn in Oregon	Extent of Use of Oregon Waters
OC Coho (56)	All; Refer to the recovery plan for a list of populations (NMFS 2016e).	Designated critical habitat within the action area includes, but is not limited to, the following major rivers and estuaries: Umpqua River, Coos River, Siuslaw River, Alsea River, Nehalem River, Tillamook Bay, Depoe Bay, and Coos Bay. Adult coho typically spawn in relatively small tributaries. As previously stated, juvenile life history strategies are diverse with a large proportion of juveniles moving quickly through the estuaries while others spend a substantial amount of time rearing in estuarine habitats.
LCR Coho (24)	Youngs Bay Big Creek Clatskanie Scappoose Clackamas Sandy Lower Gorge Upper Gorge/Hood	Major Oregon rivers designated as critical habitat for this species include, but are not limited to, the LCRE, Columbia River, Hood River, Sandy River, Clackamas River, and the lower reach of the Willamette River. Spawning occurs in small to medium sized tributaries. Populations from within Oregon as well as populations originating outside of Oregon migrate through the LCR and the estuary. Generally speaking, most juveniles are not believed to spend extended periods of time rearing in the estuary. However, Teel et al. (2009) documented subyearling LCR Chinook in seasonal floodplain wetlands near the confluence of the Willamette and Columbia Rivers.
SR Sockeye (1)	None	Within the action area, critical habitat for this species includes the Columbia River and its estuary and the Snake River along the Oregon-Idaho border. Adult sockeye also migrate upstream relatively quickly. The average upstream migration time from Bonneville to McNary Dams ranges from 5 to 14 days. Recent (2013-2017) survival rates have averaged about 60 percent in this reach (NMFS 2019b) Juvenile sockeye are believed to migrate rapidly through the LCR and LCRE. Average migration times from McNary Dam to Bonneville Dam have ranged from 3-5 days (Columbia River DART 2020). Peak catches of juvenile sockeye in the estuary occur in early June (Fresh et al. 2014). While sockeye have been captured in the main channel of the LCRE, sockeye juveniles have rarely been observed in emergent wetland and backwater sloughs of the LCRE. This may reflect a true preference for rapid downstream movement, or it may reflect either the difficulties of catching a species of such low abundance or the limitations of the sampling methodologies (Fresh et al. 2014; Johnson et al. 2018).

Species (Total # of Populations)	Populations that Spawn in Oregon	Extent of Use of Oregon Waters
LCR Steelhead (23)	Hood (summer; winter) Clackamas (winter) Sandy (winter) Lower Gorge (winter) Upper Gorge (winter)	Major Oregon rivers designated as critical habitat for this species include, but are not limited to, the LCRE, Columbia River, Hood River, Sandy River, Clackamas River, and the lower reach of the Willamette River. Three of the Oregon LCR steelhead populations must migrate past Bonneville Dam, and all populations from within Oregon as well as populations originating outside of Oregon migrate through the LCRE. Although residency time in the LCRE is believed to be relatively short, it is an important area where growth occurs.
MCR Steelhead (20)	Walla Walla Umatilla River John Day Lower Mainstem North Fork John Day Middle Fork John Day John Day Upper Mainstem South Fork John Day Fifteenmile Creek Deschutes River – westside Deschutes River – eastside	Major Oregon rivers designated as critical habitat for this species include, but are not limited to, the LCRE, Columbia River, Deschutes River (from mouth upstream to near Pelton Round Butte Dam), John Day River, Umatilla River, and the upper reaches of the Walla Walla River. Reintroduction efforts are currently underway in the Deschutes River basin, upstream of Pelton Round Butte Dam. All of the Oregon populations must pass over at least two dams on the Columbia River. For those populations passing McNary Dam, the average upstream migration time from Bonneville to McNary Dams ranges from 45 to 57 days. When the mainstem Columbia River temperatures increase, many MCR steelhead will hold in thermal refugia offered by cool tributaries such as the Deschutes River or deeper/cooler areas within the mainstem (NMFS 2019b). For juveniles produced upstream of McNary Dam, the average travel time between McNary and Bonneville Dams ranges from 3-8 days. Similar to other steelhead species, residency time in the LCRE is believed to be relatively short.

Species (Total # of Populations)	Populations that Spawn in Oregon	Extent of Use of Oregon Waters
UCR Steelhead (4)	None	Within the action area, critical habitat for this species includes the Columbia River and the LCRE. All populations of this species must migrate through the LCRE and migrate past the four LCR dams. The ten-year (2008-2017) average minimum survival estimates for migrating adult fish between Bonneville and McNary Dams is 92.1 percent (range of 87.6 to 96.8) (NMFS 2019b). The average upstream migration time from Bonneville to McNary Dams ranges from 10 to 20 days (Columbia River DART). The average migration time between McNary and Bonneville Dams for juveniles has ranged from 4-6 days (Columbia River DART). Steelhead have rarely been observed in emergent wetland and/or backwater areas of the LCRE. Their residency in the LCRE is believed to be relatively short; however, little information exists (Johnson et al. 2018).
UWR Steelhead (4)	Molalla North Santiam South Santiam Calapooia	Major Oregon rivers designated as critical habitat for this species include, but are not limited to, the LCR and its estuary, Willamette River, Santiam River (including the North and South branches), Mill Creek, Luckiamute River and Pudding River. Although not currently included as an independent population in the DPS, winter steelhead have been reported spawning in the West-side tributaries to the Willamette River above Willamette Falls. Production from these tributaries function as a population sink with the DPS meta-population structure. Juvenile and adult fish migrate through the LCRE on their way to the Willamette basin. Adult fish begin to arrive in the lower Willamette River in January, and pass over Willamette Falls from mid-February to mid-May. Spawning typically occurs in the upper portions of the larger subbasins. Juveniles generally spend one to four years (typically two years) rearing in the upper part of the basin. They migrate quickly downstream through the mainstem Willamette River and LCRE into the ocean.

Species (Total # of Populations)	Populations that Spawn in Oregon	Extent of Use of Oregon Waters
SRB Steelhead (24)	Joseph Creek Wallowa River Upper Grande Ronde River Lower Grande Ronde River Imnaha The *Hells Canyon Tributaries* population is classified as extirpated.	Major Oregon rivers designated as critical habitat for this species include, but are not limited to, the Columbia River (and its estuary), Snake River, Grande Ronde River, and Imnaha River. All populations of this species must migrate past the four lower Columbia River dams. The ten-year (2008-2017) average minimum survival estimates for migrating adult fish between Bonneville and McNary Dams is 94.3 percent (range of 90.1 to 100) (NMFS 2019b). The average upstream migration time from Bonneville to McNary Dams ranges from 18 to 37 days (Columbia River DART). In Oregon, spawning occurs in the Imnaha and Grande Ronde subbasins. Juveniles typically migrate as age-2 or age-3 smolts and move rapidly downstream. Average migration durations from McNary Dam to Bonneville Dam range from approximately 3 to 7 days (Columbia River DART). Available evidence suggests that juveniles continue their rapid downstream migration through the LCR and its estuary. It is believed that these fish generally spend less than a week migrating through the LCRE (NMFS 2017a; Fresh et al. 2014).
sDPS Eulachon (4 subpopulations)	Columbia subpopulation	In the action area, designated critical habitat includes the LCRE, about 24 miles of the lower Umpqua River, about 12.5 miles of the lower Sandy River, and 0.2 miles of Tenmile Creek. Large spawning aggregations of eulachon have been observed in the mainstem Columbia River and Sandy River. Spawning occurs in January through March. Historically, eulachon spawned in tributaries as far upstream as Hood River; since construction of Bonneville Dam, there have only been occasional observations of eulcahon at or upstream of the dam. These observations are limited to years of high abundance. Larvae are rapidly transported downstream soon after they hatch. In the LCRE, juveniles forage on small prey items. The length of eulachon residency in the LCRE is unknown at this time.

Species (Total # of Populations)	Populations that Spawn in Oregon	Extent of Use of Oregon Waters
sDPS Green Sturgeon (1 known spawning population)	None	In the action area, critical habitat includes Coos Bay, Winchester Bay, Yaquina Bay, Nehalem Bay, and the LCRE up to RM46. Within these areas, the designation includes all tidally influenced areas up to the mean higher high water, and includes areas that are up to the head of tide in tributary streams/rivers. It is unlikely that sDPS of green sturgeon would occur in non-natal streams beyond the head of tide (74 FR 52306). Large aggregations of green sturgeon occur in the LCRE. The greatest abundance of green sturgeon occur in the lower portion of the LCRE (i.e., up to ~RM20), with numbers sharply decreasing upstream of RM52. Both juvenile and spermiating adult green sturgeon have been documented in the Columbia River (Schreier et al. 2016); however, these likely belonged to the northern DPS, which is not an ESA-listed species. Adult and subadult sDPS green sturgeon occur from late spring to autumn within the bays and estuaries designated as critical habitat. Peak abundances occur in summer and autumn. Adults and subadults are present year-round in marine habitats along the west coast.
SRKW	None	The SRKW action area includes the whale’s coastal migration route, which overlaps with many of the salmonid prey species listed above, along Northern California, Oregon, and Washington coasts.

2.4.2 General Habitat Conditions in the Action Area

As described above in the Status of the Species and Critical Habitat sections, factors that limit the recovery of species considered in this Opinion vary with the overall condition of aquatic habitats and surrounding lands. Within the action area, many stream and riparian areas have been degraded by the effects of land and water use, including road construction, forest management, agriculture, mining, transportation, urbanization, and water development. Each of these economic activities has contributed to the myriad factors for the decline of species in the action area. Among the most important of these are changes in stream channel morphology, degradation of spawning substrates, reduced instream roughness and cover, loss and degradation of estuarine rearing habitats, loss of wetlands, loss and degradation of riparian areas, degradation of water quantity and quality (e.g., temperature, sediment, dissolved oxygen, contaminants), blocked fish passage, direct take, and loss of habitat refugia. Climate change is likely to play an increasingly important role in determining the abundance of ESA-listed species and the conservation value of designated critical habitats in the Pacific Northwest.

West of the Cascade Mountains in Oregon, stream habitats and riparian areas have been degraded by road construction, timber harvest, splash damming, urbanization, agricultural activities, mining, flood control, filling of estuaries, and construction of dams. East of the Cascade Mountains, aquatic habitats have been degraded by road building, timber harvest, splash damming, livestock grazing, water withdrawal, agricultural activities, mining, urbanization, and construction of reservoirs and dams (FEMAT 1993; Lee et al. 1997; McIntosh et al. 1994; Wissmar et al. 1994).

Anadromous salmonids have been affected by the development and operation of dams. Dams, without adequate fish passage systems, have extirpated anadromous fish from their pre-development spawning and rearing habitats. Dams and reservoirs, within the currently accessible migratory corridor, have greatly altered the river environment and have affected fish passage. Dam operations have altered the natural hydrograph of many rivers. Water impoundment and dam operations also affect downstream water quality characteristics, vital components to anadromous fish survival. In recent years, fish passage has been restored through both improvements to existing fish passage facilities and dam removal.

Within the habitat currently accessible by species considered in this Opinion, dams have negatively affected spawning and rearing habitat. Floodplains have been reduced, off-channel habitat features have been eliminated or disconnected from the main channel, and the amount of large wood in mainstem rivers has been greatly reduced. Remaining habitats often are affected by flow fluctuations associated with reservoir water management for power peaking, flood control, and other operations.

The development of hydropower and water storage projects within the Columbia River basin have resulted in the inundation of many mainstem spawning and shallow-water rearing areas (loss of spawning gravels and access to spawning and rearing areas); altered water quality (reduced spring turbidity levels), water quantity (seasonal changes in flows and consumptive losses resulting from use of stored water for agricultural, industrial, or municipal purposes), water temperature (including generally warmer minimum winter temperatures and cooler maximum summer temperatures), water velocity (reduced spring flows and increased cross-sectional areas of the river channel), food (alteration of food webs, including the type and availability of prey species), and safe passage (increased mortality rates of migrating juveniles) (Ferguson et al. 2005; Williams et al. 2005).

ESA-listed fish species considered in this Opinion are exposed to high rates of predation during all life stages. Fish, birds, and marine mammals (including harbor seals, sea lions, and killer whales) all prey on juvenile and adult salmon. The Columbia River basin has a diverse assemblage of native and introduced fish species, some of which prey on salmon, steelhead, and eulachon. The primary resident fish predators of salmonids in many areas of the State of Oregon inhabited by anadromous salmon are northern pikeminnow (native), smallmouth bass (introduced), and walleye (introduced). Other predatory resident fish include channel catfish (introduced), Pacific lamprey (native), yellow perch (introduced), largemouth bass (introduced), and bull trout (native). Increased predation by non-native predators has and continues to decrease population abundance and productivity.

Avian predation is another factor limiting salmonid recovery in the Columbia River basin. Throughout the basin, piscivorous birds congregate near hydroelectric dams and in the estuary near man-made islands and structures. Avian predation has been exacerbated by environmental changes associated with river developments. Water clarity caused by suspended sediments settling in impoundments increases the vulnerability of migrating smolts. Delay in project reservoirs, particularly immediately upstream from the dams, increases smolt exposure to avian predators, and juvenile bypass systems concentrate smolts, creating potential feeding stations for birds. Dredge spoil islands, associated with maintaining the Columbia River navigation channel, provide habitat for nesting Caspian terns and other piscivorous birds. Caspian terns, double-crested cormorants, glaucous-winged/western gull hybrids, California gulls, and ring-billed gulls are the principal avian predators in the basin. As with piscivorous predators, predation by birds has and continues to decrease population abundance and productivity.

The existing highway system contributes to a poor environmental baseline condition in several ways. Many miles of highway that parallel streams have degraded streambank conditions by armoring the banks with riprap, degraded floodplain connectivity by adding fill to floodplains, and discharge untreated or marginally treated highway runoff to streams. Culvert and bridge stream crossings have similar effects, and create additional problems for fish when they act as physical or hydraulic barriers that prevent fish access to spawning or rearing habitat, or contribute to adverse stream morphological changes upstream and downstream of the crossing itself.

Water quality, as characterized by the Oregon Water Quality Index (OWQI) (Risser, 2000), is typically poor, or very poor, except in the Cascades and Coast Range ecoregions and in the Blue Mountain ecoregion during high flows. The OWQI utilizes measures of temperature, dissolved oxygen, biochemical oxygen demand, pH, ammonia and nitrate, total phosphorous, total solids and fecal coliform that are collected at both high and low flow (Risser 2000). Most of the waterbodies in Oregon are on the CWA section 303(d) list for not meeting temperature standards. Temperature alterations can affect aquatic biota metabolism, growth rate, and disease resistance, as well as the timing of adult salmonid migrations, fry emergence, and smoltification.

Summer temperatures above thermal maxima likely put fish at greater risk of effects that range from effects on the individual organism to effects at the aquatic community level. These effects would impair salmon productivity from the reach scale to the stream network scale by reducing the area of usable habitat and adversely affecting fish growth, behavior, and disease resistance. The loss of vegetative shading is the predominant cause of elevated summer water temperatures. Smaller streams with naturally lower temperatures that are critical to maintaining downstream water temperatures are most vulnerable to this effect. The same factors that elevate summer water temperature can decrease winter water temperatures and put salmon at additional risk.

Contaminants are another reason for degraded habitat conditions in many waters across the state. Aerial deposition, discharges of treated effluents, and stormwater runoff from residential, commercial, industrial, agricultural, recreational, and transportation land uses are all source of these contaminants. For example, 4.7 million pounds of toxic chemicals were discharged into surface waters of the Columbia River basin (a 39 percent decrease from 2003) and another 91.7 million pounds were discharged in the air and on land in 2011 (EPA 2011). This reduction can be attributed, in part, to significant state, local and private efforts to modernize and strengthen tools available to treat and manage stormwater runoff (EPA 2009; 2011).

In a typical year in the U.S., pesticides are applied at a rate of approximately five billion pounds of active ingredients per year (Kiely et al. 2004). Therefore, pesticide contamination in the nation's freshwater habitats is ubiquitous and pesticides usually occur in the environment as mixtures. The U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program conducted studies and monitoring to build on the baseline assessment established during the 1990s to assess trends of pesticides in basins across the Nation, including the Willamette River basin. More than 90 percent of the time, water from streams within agricultural, urban, or mixed-land-use watersheds had detections of two or more pesticides or degradates, and about 20 percent of the time they had detections of 10 or more. Fifty-seven percent of 83 agricultural streams had concentrations of at least one pesticide that exceeded one or more aquatic-life benchmarks at least one time during the year (68 percent of sites sampled during 1993–1994, 43 percent during 1995–1997, and 50 percent during 1998–2000) (Gilliom et al. 2006). Rinella and Janet (1998) reported that 34 herbicides and 16 insecticides were detected in the Willamette basin. Forty-nine of the 50 pesticides were detected in streams draining predominantly agricultural land (Rinella and Janet 1998). In the lower Clackamas River basin, Oregon (2000–2005), USGS detected 63 pesticide compounds, including 33 herbicides. High-use herbicides such as glyphosate, triclopyr, 2,4-D, and metolachlor were frequently detected, particularly in the lower-basin tributaries (Carpenter et al. 2008).

The role of stormwater runoff in degrading water quality has been known for years but reducing that role has been notoriously difficult because the runoff is produced everywhere in the developed landscape, the production and delivery of runoff are episodic and difficult to attenuate, and runoff accumulates and transports much of the collective waste of the developed environment (NRC 2009). In most rivers in Oregon, the full spatial distribution and load of contaminants is not well understood. Hydrologically low-energy areas, where fine-grained sediment and associated contaminants settle, are more likely to have high water temperatures, concentrations of nitrogen and phosphorus that may promote algal blooms, and concentrations of aluminum, iron, copper, and lead that exceed ambient water quality criteria for chronic toxicity to aquatic life (Fuhrer et al. 1996). Even at extremely low levels, contaminants still make their way into salmon tissues at levels that are likely to have sublethal and synergistic effects on individual Pacific salmon, such as immune toxicity, reproductive toxicity, and growth inhibition (Baldwin et al. 2011; Carls and Meador 2009; Hicken et al. 2011; Johnson et al. 2013), that may be sufficient to reduce their survival and therefore the abundance and productivity of some populations (Baldwin *et al.* 2009; Spromberg and Meador 2006). The adverse effect of contaminants on aquatic life often increases with temperature because elevated temperatures accelerate metabolic processes and thus the penetration and harmful action of toxicants.

Decades of industrial use along the Willamette River led EPA to include the Portland Harbor Superfund Site on the National Priority List in 2000. This site includes in-river and upland portions of the lower Willamette River. Water and sediment at the site are contaminated with many hazardous substances, including polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), dioxings/furans, pesticides, and heavy metals. The Columbia River between Portland, Oregon, and Longview, Washington, also appears to be an important source of contaminants for juvenile salmon and is a region where salmon are exposed to toxicants associated with urban development and industrial activity. Johnson et al. (2013) found PCBs and dichlorodiphenyltrichloroethane (DDT) in juvenile salmon and salmon diet samples from the LCR and LCRE at concentrations above estimated thresholds for effects on growth and survival. The highest concentrations of PCBs were found in fall Chinook salmon stocks with subyearling life histories, including populations from the upper Columbia and Snake Rivers, which feed and rear in the tidal freshwater and estuarine portions of the river for extended periods. Spring Chinook salmon stocks with yearling life histories that migrate more rapidly through the estuary generally had low PCB concentrations, but high concentrations of DDTs. Pesticides can be toxic to primary producers and macroinvertebrates, thereby limiting salmon population recovery through adverse, bottom-up impacts on aquatic food webs (Macneale et al. 2010).

The full presence of contaminants throughout the program action area is poorly understood, but the concentration of many increase in downstream reaches (Fuhrer et al. 1996; Johnson et al. 2013; Johnson et al. 2005; Morace 2012). The fate and transport of contaminants varies by type, but are all determined by similar biogeochemical processes (Alpers et al. 2000b; Alpers et al. 2000a; Bricker 1999; Chadwick et al. 2004; Johnson et al. 2005). After deposition, each contaminant typically processes between aqueous and solid phases, sorption and deposition into active or deep sediments, diffusion through interstitial pore space, and resuspension into the water column. Uptake by benthic organisms, plankton, fish, or other species may occur at any stage except deep sediment, although contaminants in deep sediments become available for biotic uptake when resuspended by dredging or other disturbances.

Whenever a contaminant is in an aqueous phase or associated with suspended sediments, it is subject to the processes of advection and dispersion toward the Pacific Ocean. However, once soluble metal releases are reduced or terminated, the solute half-time in Columbia River water is months versus about 20 years for adsorbed metals on surficial (or resuspended) bed sediments. The much slower rate of decline for sediment, as compared to the solute phase, is attributed to resuspension, transport and redeposition of irreversibly bound metals from upstream sedimentary deposits. This implies downstream exposure of benthic or particle-ingesting biota can continue for years following source remediation and/or termination of soluble metal releases (Johnson et al. 2005). Adsorbed contaminants are highest in clay and silt, which can only be deposited in areas of reduced water velocity, such as behind dams and the backwater or off-channel areas preferred as rearing habitat by juveniles of some Pacific salmon (Johnson et al. 2005). Similar estimates for the residence time of contaminants in the freshwater plume are unavailable, although the plume itself has been tracked as a distinct coastal water mass that may extend up to 50 miles beyond the mouth of the Columbia River, where the dynamic interaction of tides, river discharge, and winds can cause significant variability in the plume's location at the interannual, seasonal scale, and even at the event scale of hours (Burla et al. 2010; Kilcher et al. 2012; Thomas and Weatherbee 2006).

The environmental baseline includes the anticipated impacts of all Federal actions in the action area that have already undergone formal consultation. The U.S. Army Corps of Engineers (COE), Bonneville Power Administration (BPA), and Bureau of Reclamation (BOR) have consulted on large water management actions, such as operation of the FCRPS, the Umatilla Basin Project, and the Deschutes Project. The U.S. Bureau of Indian Affairs (BIA), U.S. Bureau of Land Management (BLM), and the U.S. Forest Service (USFS) have consulted on Federal land management throughout Oregon, including restoration actions, forest management, livestock grazing, and special use permits. NMFS issued biological opinions for implementation of the National Flood Insurance Program (NFIP) in Oregon. This NFIP Opinion concluded that implementation of the National Flood Insurance Program would jeopardize the continued existence of ESA-listed species

The BPA, NOAA Restoration Center, and USFWS have also consulted on large restoration programs that consist of actions designed to address species limiting factors or make contributions that would aid in species recovery. Restoration actions may have short-term adverse effects, but generally result in long-term improvements to habitat condition and population abundance, productivity, and spatial structure. After going through consultation, many ongoing actions, such as stormwater facilities, roads, culverts, bridges and utility lines, have less impact on ESA-listed salmon and steelhead.

As noted above, the proposed action establishes water quality criteria for aluminum that is dependent upon site-specific chemistry. The criteria apply to all freshwaters of Oregon, with a few exceptions (e.g., freshwater within jurisdictional boundaries of Federally-recognized Indian Tribes). In addition, ODEQ may choose to apply the criteria to estuaries where appropriate. To evaluate the potential effect of the action on ESA-listed resources, NMFS made the following assumptions regarding the environmental baseline conditions:

1. Existing water quality data (i.e., pH, DOC, total hardness, and total aluminum) provide a representative snapshot of conditions that will be experienced by ESA-listed species.

2. Implementation of the proposed action will not cause new point and non-point sources to contribute aluminum, nor will it cause the allowance of existing point and non-point sources to contribute more aluminum to the receiving water than they already are.

3. Where there appears to be potential exceedances of the proposed aluminum criteria, the state will continue to collect samples to evaluate the potential impairment, and, if the

affected waterbody segment is listed as impaired under CWA section 303(d), a TMDL will be prepared and implemented to bring the water(s) into compliance.

2.4.3 Sources of Aluminum in Oregon

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). Because it is abundant in rocks and minerals, its presence in surface water is dominated by natural sources (e.g., rock and mineral weathering, volcanic activity, or acidic springs). However, anthropogenic activities can exacerbate aluminum concentrations in surface water through point and nonpoint sources. Point source discharges of aluminum include industrial facilities (such as plants that recycle aluminum or mines where bauxite is processed), urban stormwater, and drinking water and sewage treatment facilities where alum (potassium aluminum sulfate) is used to as a coagulant. Non point sources of aluminum include atmospheric deposition, acid mine drainage, forestry, and agriculture. Dredging and disposal operations can result in substantial suspension and resuspension of particulates in the water column, including those contaminated with aluminum. The primary source of aluminum in estuaries and oceans is riverine discharges.

As previously described, human activities can exacerbate aluminum concentrations in streams and rivers. Figures 3 and 4 illustrate various point and non-point sources of aluminum in Oregon. Aluminum may be present in the effluent of industrial facilities, drinking water facilities, stormwater facilities, or sewage treatment plant facilities. EPA searched its Integrated Compliance Information System (ICIS) National Pollutant Discharge Elimination System (NPDES) (ICIS-NPDES) database for permitted point source discharges that had the potential to discharge aluminum in Oregon (Table 8). EPA (2019) identified two industrial dischargers that currently have aluminum effluent limits. EPA also identified 63 drinking water facilities with NPDES permits. Aluminum sulfate (alum) and aluminum chlorohydrate/polyaluminum chloride are often used as coagulants in the treatment of drinking water. Although unknown, there is a possibility that these Oregon facilities discharge aluminum to surface water. In addition, metal salts containing aluminum are commonly used for removing phosphorus from wastewater. EPA found four individually permitted sewage treatment facilities in Oregon that use metals salts containing aluminum in their treatment process to meet phosphorus effluent limits (2019).

Table 8. Potential number of facilities that may utilize aluminum in their operations in Oregon (EPA 2019a).

Facility Type[1]	Potential Number of Individual Facilities
Aluminum anodizing facility (Discharge to sewage treatment plants)	12
Drinking water treatment plants (Often use aluminum sulfate (alum) in treatment processes as a coagulant)	57 (under a general permit)
Wastewater treatment facilities (Have total phosphorus limits, so may use alum in their processes)	4
Totals	***73***

[1]These data were prepared by EPA (2019) for the purposes of the Economic Analysis associated with the proposed action and because of the lack of data for aluminum in point source discharges in the State. The EPA identified potential point source dischargers that utilize aluminum in their operations and, therefore, could potentially be affected by the proposed action. EPA evaluated three types of facilities (aluminum anodizing facilities, drinking water treatment plants, and wastewater treatment facilities) that could incur costs under the proposed action (if aluminum effluent limitations were necessary). Refer to EPA 2019a for a list of the assumptions employed for this analysis.

Aluminum anodizing facilities may generate wastewater containing aluminum. EPA identified twelve aluminum anodizing facilities in Oregon; however, none of these facilities discharged directly to surface water. Instead, these facilities send their wastewater to publically owned sewage treatment systems. It is possible that sewage treatment plants receiving these industrial wastewaters contain aluminum in their effluent, regardless of whether metal salts are used in the treatment process.

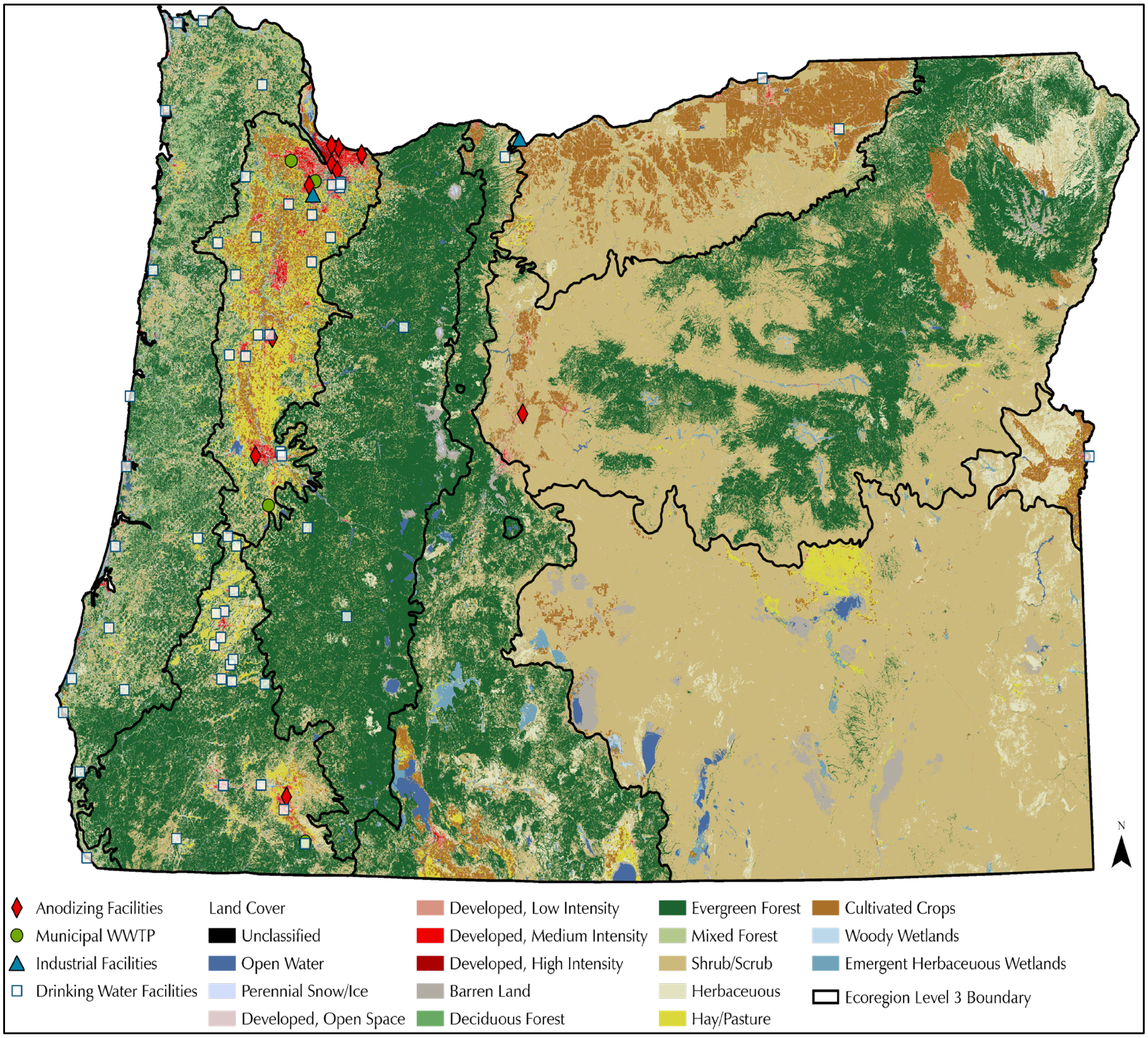

Figure 3. Land cover along with known and potential point sources of aluminum in Oregon, including wastewater treatment plants, indirect anodizing facilities, industrial facilities, and drinking water treatment facilities

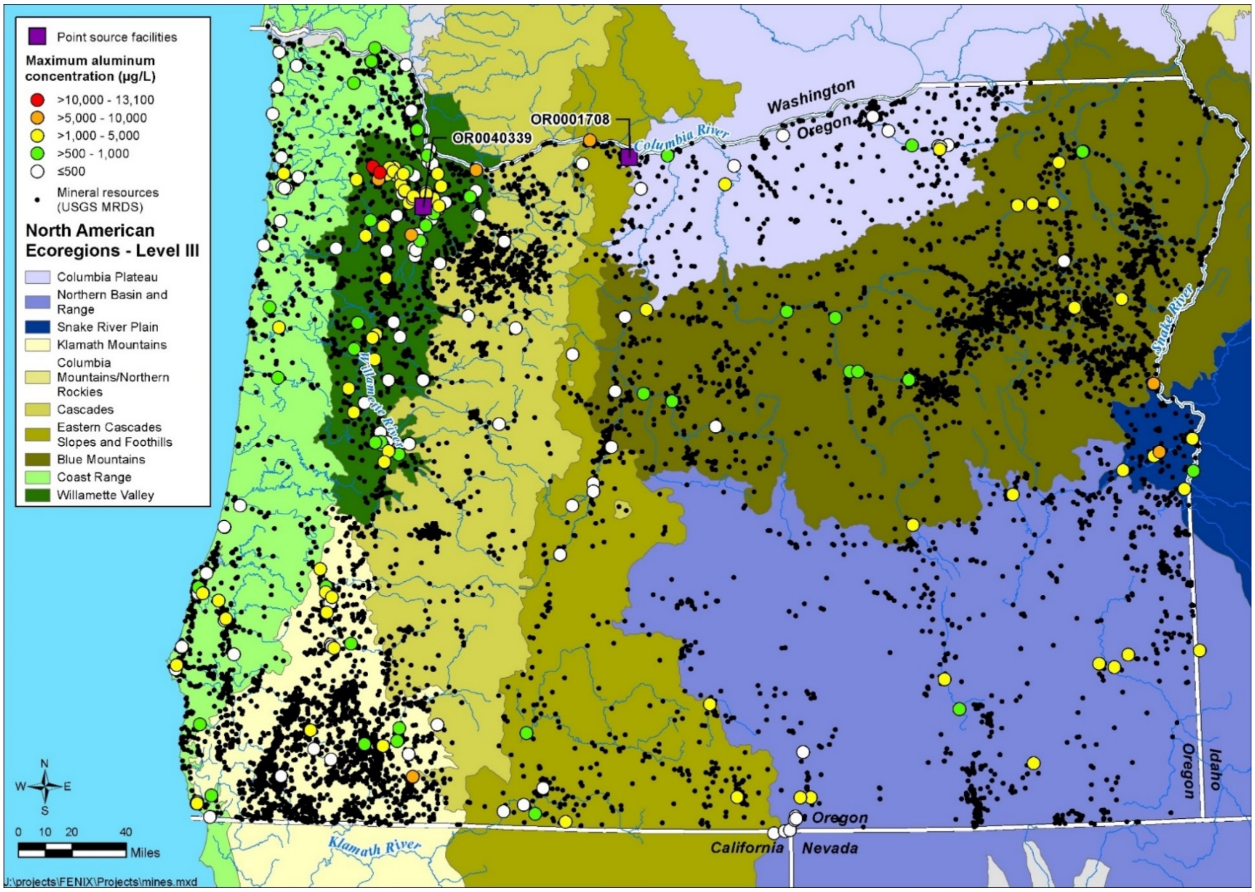

Figure 4. Mine sites, prospects, and known deposits of hard-rock minerals in Oregon (USGS 2011).

Runoff from urban stormwater also has the potential to contribute aluminum to surface waters. Solomon and Natusch (1977) documented aluminum in automobile exhaust. Urban stormwater (rooftops and parking lots) contains high concentrations of aluminum, primarily in the particulate form (Muthukrishnan 2005). The ODEQ issued four Phase I stormwater discharge permits and fifteen Phase II stormwater discharge permits. Stormwater runoff from agricultural areas may also be a significant source of aluminum given aluminum's ubiquitous nature. Aluminum may be present in pesticides used in agriculture. Aluminum phosphide was widely used in Oregon as late as 1996 (USGS 2015); there was little to no use of the pesticide in the state from 1997 through 2014 (EPA 2020).

Similar to agriculture, active or inactive mine sites may be another source of aluminum to surface waters. Abandoned and inactive mines exist throughout Oregon, including mining districts in eastern Oregon, southwest Oregon, and the Willamette Basin. State and federal agencies have identified over 150 mines in Oregon for possible further investigation or cleanup and have initiated assessment of about 95 mines (EPA 2020). For example, at the Formosa Mine Superfund site, acid mine drainage contributes to elevated heavy metal concentrations, negatively affecting downstream water quality and fisheries in Middle Creek (EPA 2008).

2.4.4 Fate and Transport of Aluminum

Aluminum can enter surface water in a dissolved form or in a particulate form attached to organic and inorganic matter. The amount of aluminum in the dissolved versus particulate form in natural waters can vary greatly, but the particulate form is usually found in greater concentrations due to changes in pH and temperature. The chemistry of aluminum in surface water is complex due to its properties (Angel et al. 2016; Campbell et al. 1983; Hem 1968a, b; Hem and Roberson 1967; Hsu 1968; Roberson and Hem 1969; Smith and Hem 1972). Aluminum can flux between different states and forms in an aquatic environment, depending on environmental conditions. These transformations can occur within and between water, sediment, and biota as the cycles of nature change and as certain (e.g., acid-mine drainage) anthropogenic activities occur.

Factors such as pH, temperature, and presence of complexing ions influence the fate and transport of aluminum in the environment. The pH of the water is particularly influential, as it affects aluminum speciation and solubility. At neutral pH, aluminum is nearly insoluble, but its solubility increases exponentially as the pH reaches either acidic (pH<6) or basic (pH>8) conditions (Gensemer and Playle 1999). At pH values between 6.5 and 9.0 in fresh water, aluminum occurs predominantly in solution as monomeric, dimeric, and polymeric hydroxides and as complexes with fulvic and humic acid, chloride, phosphate, sulfate, and less common anions (EPA 2018). Aluminum can sorb to DOC, such as humic and fulvic acids, and form organic aluminum complexes. Aluminum concentrations in marine and estuarine waters are generally lower than levels found in freshwaters (Gensemer and Playle 1999). At the typical ocean pH of 8.0-8.3, aluminum forms complexes with hydroxide ions, primarily as aluminum hydroxide, which precipitates out of solution (EPA 2018). In estuaries, the majority of aluminum is believed to be sorbed to the surface of clay particles in sediments (EPA 2018).

2.4.5 Oregon Water Quality

Because aluminum toxicity is known to be affected by water quality characteristics, this section provides an overview of baseline conditions of pH, DOC, total hardness, and aluminum in the action area. EPA obtained, and subsequently shared with NMFS, water quality data from the following four sources:

- ODEQ's Ambient Water Quality Monitoring System (AWQMS) web portal (https://orwater.deq.state.or.us/Login.aspx);
- ODEQ's Laboratory Analytical Storage and Retrieval (LASAR) database;
- Washington Department of Ecology's Environmental Information Management System (https://fortress.wa.gov/ecy/eimreporting/Default.aspx); and
- USGS' National Water Information System (NWIS); (https://waterdata.usgs.gov/nwis/qwdata).

For the entire state of Oregon, there were 19,274 unique samples from 1,554 stations for the period of 2000 to 2017 that had either measured or estimated values of DOC, pH, and total hardness. Total aluminum was not available for all of these unique samples, rather total aluminum was available for 707 unique samples from 259 stations. Instead of utilizing the analysis performed by EPA (available data was summarized by Level III ecoregion in the state), NMFS considered data from stations summarized the data by ESU/DPS. To accomplish this, NMFS utilized ArcGIS (geographic information system software) to query samples within the geographic range of each ESU/DPS. In some instances, the delineated ESU/DPS geographic range did not include the entire migration corridor along the Columbia or Willamette (where applicable) Rivers. In these instances, NMFS also queried samples within the 10th-level HUCs bordering the migratory corridors downstream of the delineated ESU/DPS geographic range.

2.4.5.1 pH

Within the action area, measured pH data were available for 15,814 unique samples from 1,158 stations. Table 9 summarizes, and Figure 5 illustrates, the ranges of all pH values for each ESU/DPS (excluding SRKW) considered in this Opinion. In general, pH values in the action area tend to fall between 7 and 8.5; however, there is substantial variability as illustrated in Figure 5. Overall, pH conditions throughout the action area appear to be within the bounds that support successful spawning, rearing, and migration of anadromous species.

Table 9. Summary of pH data available for each ESU/DPS considered in this Opinion.

Species	N	Minimum	5th %	Average	95th %	Maximum	Standard Deviation
LCR steelhead	1,775	6.1	7.1	7.7	8.5	10.3	0.45
LCR Chinook	2,286	6.1	6.9	7.6	8.3	10.3	0.48
LCR coho	2,286	6.1	6.9	7.6	8.3	10.3	0.48
COL chum	2,187	6.1	6.9	7.6	8.4	9.7	0.48
UWR steelhead	4,671	5.7	6.9	7.4	7.9	9.7	0.31
UWR Chinook	4,519	5.7	7.1	7.5	8	9.7	0.29
MCR steelhead	2,372	6.6	7.6	8.2	9	10.1	0.41
SNR steelhead	932	6.6	7.4	8.0	8.6	9.6	0.37
SRS Chinook	932	6.6	7.4	8.0	8.6	9.6	0.37
SRF Chinook	369	6.6	7.4	7.9	8.46	8.9	0.31
SONCC coho	1,247	5.9	7.3	7.9	8.6	9.4	0.38
ORC coho	4,378	6.2	7	7.5	8.3	9.7	0.42
Columbia River species[1]	368	6.6	7.4	7.9	8.465	8.9	0.31

Note: In general, pH values in the action area tend to fall between 7 and 8.5.
N = sample size
[1]The "Columbia River species" includes: sDPS green sturgeon, sDPS eulachon, SR sockeye, and UCR spring Chinook.

range

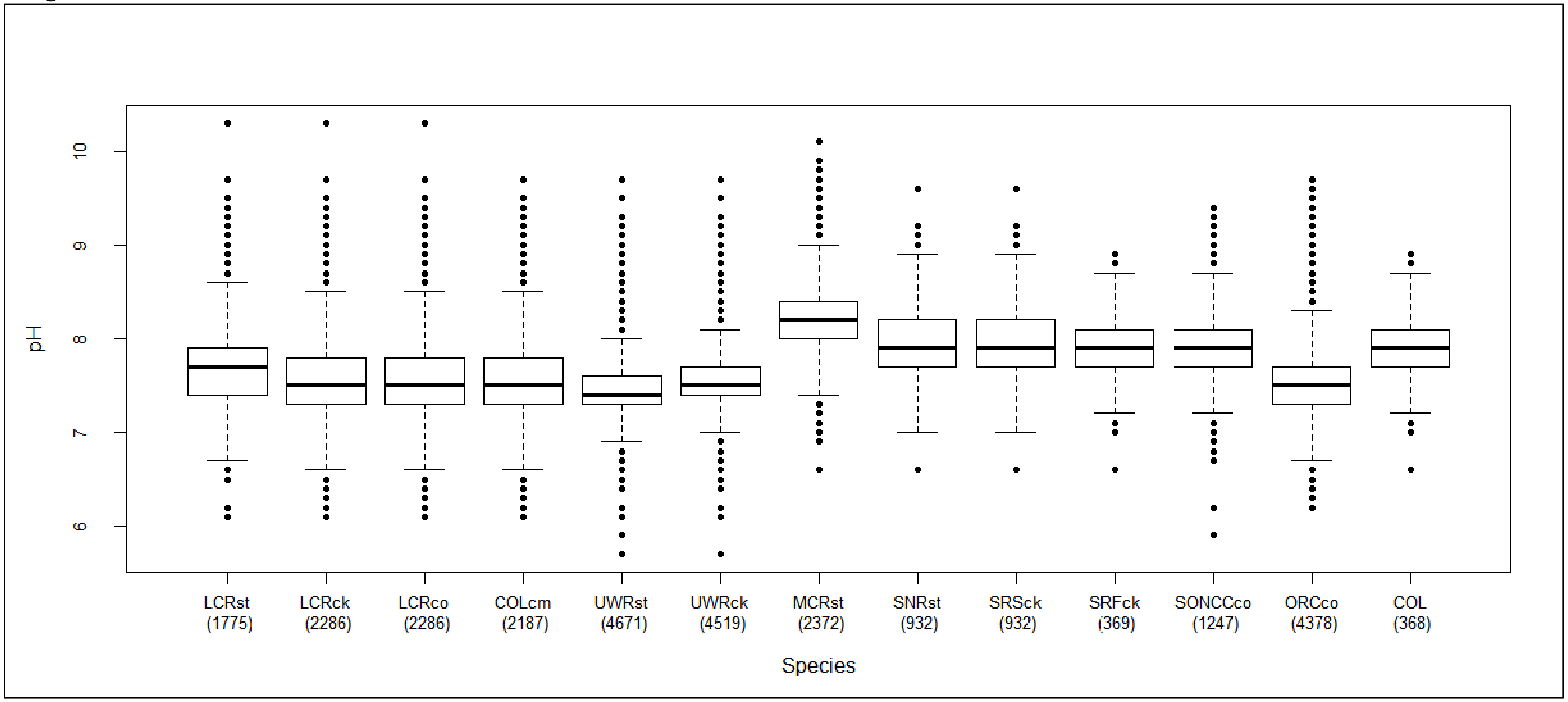

Figure 5 .Box and whiskers plots of pH data available for each ESU/DPS considered in this Opinion.
Note: The box represents the 25th, 50th, and 75th quartiles. The upper and lower bounds represent 1.5 times the interquartile range. Solid black dots are potential outliers. Abbreviations: st = steelhead; ck = Chinook, co = coho, cm = chum, COL = Columbia River.

2.4.5.2 DOC

Within the action area, measured or estimated DOC data were available for 15,814 unique samples from 1,158 stations. EPA estimated DOC values if measured values were not available but total organic carbon (TOC) values were available. In these instances, TOC concentrations were multiplied by 0.83 to estimate DOC concentrations (EPA 2019b). The 0.83 multiplier was the statewide mean value of the DOC-to-TOC ratio calculated by ODEQ (2016). Table 10 summarizes, and Figure 6 illustrates, the ranges of all DOC values for each ESU/DPS considered in this Opinion. In general, DOC values in the action area tend to fall between 1 and 4 mg/L.

Table 10. Summary of DOC data, reported as mg/L, available for each ESU/DPS considered in this Opinion.

Species	N	Minimum	5th %	Average	95th %	Maximum	Standard Deviation
LCR steelhead	1,775	0.3	0.8	1.8	4.2	12.0	1.2
LCR Chinook	2,286	0.3	0.8	2.0	5.4	12.0	1.7
UWR steelhead	4,671	0.2	0.8	2.6	7.0	12.0	2.0
UWR Chinook	4,519	0.1	0.8	1.6	3.5	12.0	1.1
MCR steelhead	2,372	0.3	0.8	2.5	5.1	12.0	1.6
SNR steelhead	932	0.7	0.8	2.0	4.2	8.3	1.1
SRS Chinook	932	0.7	0.8	2.0	4.2	8.3	1.1
SRF Chinook	369	0.8	0.8	1.7	2.5	5.0	0.5
COL chum	2,187	0.5	0.8	2.1	5.7	12.0	1.7
SONCC coho	1,247	0.3	0.8	1.7	5.0	12.0	1.6
ORC coho	4,378	0.2	0.8	1.8	4.0	12.0	1.4
LCR coho	2,286	0.3	0.8	2.0	5.4	12.0	1.7
Columbia River species	368	0.8	0.8	1.7	2.5	5.0	0.5

Note: N = sample size

[1]The "Columbia River species" includes: sDPS green sturgeon, sDPS eulachon, SR sockeye, and UCR spring Chinook.

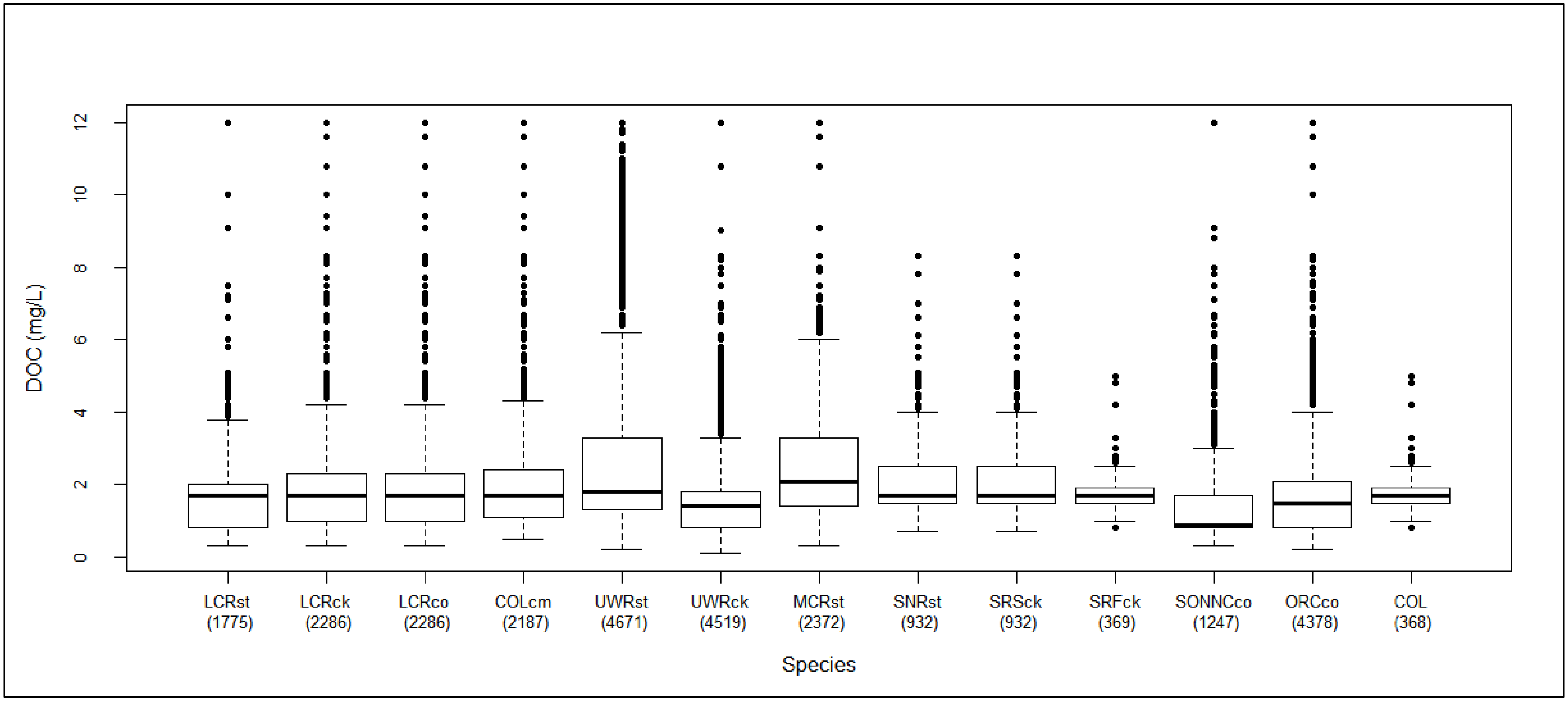

Figure 6. Box and whiskers plots of DOC data available for each ESU/DPS considered in this Opinion.
Note: The box represents the 25th, 50th, and 75th quartiles. The upper and lower bounds represent 1.5 times the interquartile range. Solid black dots are potential outliers. Abbreviations: st = steelhead; ck = Chinook, co = coho, cm = chum, COL = Columbia River.

2.4.5.3 Total Hardness

Within the action area, measured total hardness data[5] were available for 15,814 unique samples from 1,158 stations. If measured values were not available, EPA (2019b) estimated total hardness in one of two ways: (1) Summed measures of calcium (Ca) and magnesium (Mg) ions following a unique equation; or (2) if Ca and Mg ions were not measured, EPA estimated their concentrations from conductivity measurements and then summed the estimated Ca and Mg values. Table 11 summarizes, and Figure 7 illustrates, the ranges of all total hardness values for each ESU considered in this Opinion. In the action area, total hardness values typically fall between 15 and 75 mg/L.

Table 11. Summary of total hardness data, reported as mg/L, available for each ESU/DPS considered in this Opinion.

Species	N	Minimum	5th %	Average	95th %	Maximum	Standard Deviation
LCR steelhead	1,775	0.05	14.4	36.7	71.6	286.9	23.1
LCR Chinook	2,286	0.05	14.8	36.3	72.0	356.6	27.7
UWR steelhead	4,671	0.8	14.8	42.6	100.8	235.0	29.1
UWR Chinook	4,519	0.8	14.1	31.0	72.5	235.0	25.0
MCR steelhead	2,372	9.4	22.6	65.9	146.1	265.0	39.8
SNR steelhead	932	8.7	17.7	42.3	75.1	111.0	17.9
SRS Chinook	932	8.7	17.7	42.3	75.1	111.0	17.9
SRF Chinook	369	30.6	37.14	52.4	67.3	82.4	9.3
COL chum	2,187	0.05	15.4	36.7	71.3	356.6	27.0
SONCC coho	1,247	6.0	16.7	40.9	93.4	430.0	30.6
ORC coho	4,378	2.4	15.4	31.6	58.1	430.0	30.3
LCR coho	2,286	0.05	14.8	36.3	72.0	356.6	27.7
Columbia River Species	368	30.6	37.1	52.4	67.3	82.4	9.3

Note: N = sample size
[1]The "Columbia River species" includes: sDPS green sturgeon, sDPS eulachon, SR sockeye, and UCR spring Chinook.

[5] The ODEQ (2016) reported minimal differences between total and dissolved hardness values. As such, EPA (2019b) included dissolved hardness measurements as total hardness.

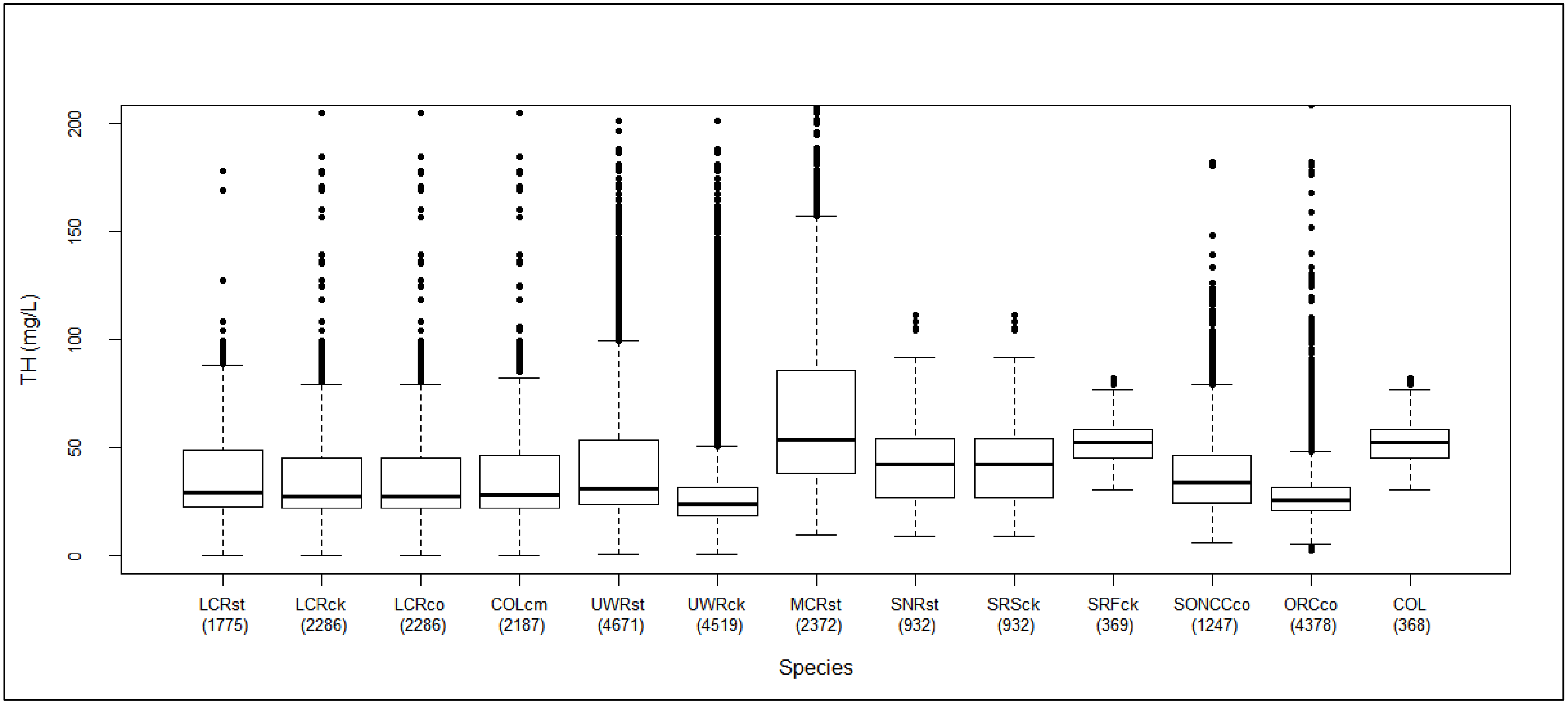

Figure 7. Box and whiskers plots of total hardness data available for each ESU/DPS considered in this Opinion.

Note: Values greater than 200 mg/L are not shown on this plot. The box represents the 25th, 50th, and 75th quartiles. The upper and lower bounds represent 1.5 times the interquartile range. Solid black dots are potential outliers. Abbreviations: st = steelhead; ck = Chinook, co = coho, cm = chum, COL = Columbia River.

2.4.5.4 Aluminum Concentrations

Measured aluminum concentrations in natural waters can vary widely depending on the test method used (EPA 2018). The current EPA-approved test method for measuring total aluminum recommends an unfiltered sample first be digested to a pH of <2. This process dissolves the monomeric and polymeric forms of aluminum as well as the colloidal, particulate, and clay-bound aluminum (EPA 2018). Under natural conditions, not all of these forms of aluminum are bioavailable. As such, detected total aluminum concentrations using current, EPA-approved methodologies could overestimate the bioavailable fraction of aluminum, especially in samples where the total suspended solids are high. Elevated total suspended solids are common during rain or snowmelt. Thus, applying the aluminum criteria as total aluminum concentrations given the current approved test methodologies is a conservative approach. Research on new analytical methods is ongoing to address concerns with including aluminum bound to particulate matter (i.e., the non-bioavailable aluminum) in the measurements of total aluminum (OSU 2018).

Total aluminum concentration data were available for 417 unique freshwater samples, collected at 160 stations within the boundaries of all the species taken together. Table 12 summarize the number of samples and ranges of all total aluminum concentrations within the geographic range of each fish ESU/DPS considered in this Opinion. In general, as described in more detail in the effects section of this Opinion, total aluminum does not appear to be present in concentrations sufficient to exert population-level effects in the action area.

Table 12. Summary statistics for total aluminum concentrations (µg/L) in the action area.

Species	# of sample locations	N	Maximum (µg/L)	Minimum (µg/L)	Average (SD) (µg/L)	5th %	95th %
LCR steelhead	16	33	7,680	20	705.3 (1633.3)	43.2	3,446
LCR Chinook	24	48	7,680	20	548 (1372.7)	38.5	1,629
LCR coho	24	48	7,680	20	548 (1372.7)	38.5	1,629
COL chum	23	46	7,680	35.2	570.7 (1398.4)	53.0	1,675
UWR steelhead	46	108	4,810	25.5	396.8 (580.9)	45.4	1,238
UWR Chinook	56	137	4,810	8.2	425.4 (737.6)	31	1,452
MCR steelhead	23	69	2,060	14.5	331.2 (404.9)	20	1,100.8
SRB steelhead	8	19	1,280	16.6	462.4 (404.4)	18.6	1,199
SRS Chinook	8	19	1,280	16.6	462.4 (404.4)	18.6	1,199
SONCC coho	14	44	2,110	7.8	369.1 (443)	10.2	1,255.5
ORC coho	42	103	3,520	11.9	404.2 (604.8)	21.3	1767

Species	# of sample locations	N	Maximum (µg/L)	Minimum (µg/L)	Average (SD) (µg/L)	5th %	95th %
Columbia River[1]	3	3	420	113	217.7 (175.3)	113.7	390
Green sturgeon	15	32	1,500	15.2	305.4 (357.0)	40.4	1,010.6
Eulachon	17	33	7,680	35.2	513.1 (1,318)	47.9	1,030.6

Abbreviations: N = number of samples; SD = standard deviation
[1]This represents SR sockeye, SRF Chinook, UCR spring Chinook, and UCR steelhead

2.5 Effects of the Action

Under the ESA, "effects of the action" are all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action. A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur. Effects of the action may occur later in time and may include consequences occurring outside the immediate area involved in the action (see 50 CFR 402.17). In our analysis, which describes the effects of the proposed action, we considered 50 CFR 402.17(a) and (b).

As previously described, consequences of this action will arise when the proposed criteria are implemented in CWA programs. Furthermore, the criteria values vary based on site-specific water chemistry, which means that at some times of the year criteria values may be less stringent relative to other times of the year. It is important to understand this variability and properly characterize (i.e., seasonality, distribution of IWQC data, etc.) conditions in the water because the rule stipulates that aquatic life be protected under the full range of ambient conditions found at each site, including conditions when aluminum is most toxic (i.e., when the criteria are most stringent). However, the rule does not prescribe how to characterize the most toxic conditions for aluminum that is representative of a site or of a waterbody, although the preamble to the rule provides recommendations. For these reasons, the potential consequences of promulgating aluminum criteria depend, in part, on how the criteria are implemented in CWA programs (i.e., point source discharge permits, 303(d) listing determinations, and TMDL development). The ODEQ is responsible for implementing CWA programs in the state, and has substantial flexibility in establishing procedures for characterizing the most toxic conditions and implementing the criteria in TMDLs, discharge permits, and other programs. This flexibility introduces uncertainty into our analysis of potential consequences of the proposed action. Our analysis assumes that the most toxic conditions will be adequately characterized and the aluminum criteria will be implemented in a manner that are adequately protective when conditions are most toxic.

2.5.1 Assessment Framework

In their BE, the EPA employed a weight-of-evidence approach to evaluate whether their proposed action was "may affect, not likely to adversely affect" or "may affect, likely to adversely affect." EPA considered the following evidence in its approach: potential for direct toxicity (i.e., mortality or reduced growth) to ESA-listed species, potential for exposure to aluminum, and potential for indirect effects (i.e., toxicity to prey) to ESA-listed species. More details about EPA's approach can be found in Section 5 of the BE (EPA 2020). EPA concluded that the proposed action may affect all of the species and designated critical habitats listed in Table 1. Furthermore, EPA concluded the proposed action was likely to adversely affect all of these species and designated critical habitats with the exception of the sDPS of green sturgeon and its designated critical habitat.

Our assessment approach, described below, was very similar to that used by EPA; however, we deviated in the following ways:

- Rather than using ecoregional boundaries to assess the risk of exposure and toxicity, NMFS utilized the geographic ranges of individual species;

- Rather than characterize land use by characteristics within ¼-mile of a surface water sample location, NMFS characterized land use based on the proportion of developed and agricultural land within the geographic ranges of each species, coupled with best professional judgement about the level of exposure risk anthropogenic development posed to each species.

- Rather than rely on the web-based interspecies correlation estimation (Web-ICE) model to evaluate potential toxicity to sDPS green sturgeon, NMFS utilized salmonid toxicity data as a surrogate to characterize potential toxicity to the sDPS of green sturgeon.

- Rather than compare total aluminum concentrations to ecoregional criteria, NMFS compared concentrations to the paired acute and chronic IWQC values.

- Rather than assigning a quantitative risk score, NMFS characterized the risk of exposure and risk of toxicity qualitatively as low, medium, or high.

To conduct the effects analysis, we follow an ecological risk assessment framework to evaluate whether the proposed action will adversely affect individuals, and if so, whether those adverse effects to individuals will negatively affect populations and the species they comprise. The first step in our assessment framework was to examine the toxicity of aluminum to species and their critical habitats and compare this to the proposed criteria. The results of this analysis identifies the potential effects on individuals and their critical habitats if they were to be exposed to water concentrations that were equal to the proposed criteria. Impacts to individual fitness can occur through direct toxicity of aluminum, including both direct lethality and sublethal effects (e.g., reduced growth, reduced swimming performance, etc.). Impacts may also occur due to impacts to designated critical habitat (e.g., degraded water quality, reduced quantity or quality of the forage base or impacts to substrates). The BE examined the potential for impacts to individuals and designated critical habitat, and much of that information is included in this Opinion.

The next step in our assessment framework is to then examine the potential for exposure of species and designated critical habitats to aluminum. At the individual level, we assume that there is a risk of exposure to aluminum at criteria levels. Where we concluded that reductions in individual fitness would occur, we then examined whether those fitness reductions are likely to be sufficient to reduce the viability of the populations those individuals represent, and if so, whether such reductions in viability would be sufficient to reduce the viability of the species those populations comprise. When evaluating the risk of population-level exposure to aluminum, we consider the current status of the environmental baseline, including existing aluminum concentrations, existing IWQC information, and anthropogenic sources of aluminum relative to the presence of ESA-listed species and designated critical habitats. Our third step in the analysis is to integrate the risk of toxicity with the risk of exposure and assess whether negative impacts will occur at the population and species levels. Each of these steps are further described in Sections 2.5.4 through 2.5.6.

2.5.2 Criteria Derivation

Before describing aluminum toxicity, it is important to understand the limitations and uncertainties inherent in EPA's criteria derivation process. The criteria were developed following EPA's guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses (hereinafter referred to as "Guidelines") (Stephan et al. 1985). Because the Guidelines are fundamental to criteria, they are fundamental to the evaluation of the protectiveness of criteria for ESA-listed species and critical habitats. The assumptions and procedures implemented in the Guidelines influence our evaluation of the protectiveness of the criteria. Some of the key assumptions are summarized below. A more detailed description of these are provided in NMFS' biological opinions for EPA's approval of water quality criteria for toxics in Oregon (NMFS 2012) and Idaho (NMFS 2014b).

2.5.2.1 Laboratory Tests Are Representative

The data used to derive criteria must meet very specific and stringent requirements; thus, laboratory conditions are tightly controlled, which is quite different from the ambient waters criteria are expected to protect. This level of control affords the opportunity to more closely attribute causal relationships. Field studies, on the other hand, have the converse problem of being uncontrolled and difficult to unambiguously attribute apparent effects to causes.

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. If responses are different, then is it possible to identify a consistent bias in laboratory tests and make adjustments for it? There are myriad of factors that may influence the effects of a chemical stressor on aquatic organisms, and this complexity makes the question of bias in sensitivity difficult or even impossible to answer with any certainty. Table 13 summarized a number of reasons why the effects of a chemical could be more- or less-severe on ESA-listed steelhead and salmon in laboratory or in wild settings (NMFS 2012; NMFS 2014b).

Table 13. Reasons why the effects of a chemical substance could be more- or less-severe on ESA-listed fish in laboratory or in wild settings.

Factor	Are effects likely more severe in typical lab settings, or effects uncertain, or effects more severe in the wild?
Environmental	
Nutritional state – acute test exposures	**In the wild.** In acute toxicity tests with fish fry, fish are selected for uniform size, and unusually skinny fish that might be weakened from being in poor nutritional state are culled from tests. For instance, if <90% of control fish survive the 4 days starvation of an acute toxicity test, the test may be rejected from inclusion in the criteria dataset. In the wild, not all fish can be assumed to be in optimal nutritional state. While perhaps counterintuitive, starvation can protect fish against waterborne copper exposure (Kunwar et al. 2009). Fish are routinely starved during acute laboratory tests of the type used in criteria development.
Nutritional state – chronic test exposures	**In the wild.** Fish in the wild must compete for prey and if chemicals impair fish's ability to detect and capture prey because of subtle neurological impairment, this could cause feeding shifts and reduce their competitive fitness (Riddell *et al*. 2005). Fish in chronic lab tests with waterborne chemical exposures are often fed to satiation and food pellets do not actively evade capture like live prey. Perhaps these factors dampen responses in lab settings.
Temperature	**In the wild.** In lab test protocols, nearly optimal test temperatures are recommended, e.g., 12°C (53.6°F) for rainbow trout, the most commonly tested salmonid. Fish may be most resistant to chemical insults when at optimal temperatures. At temperatures well above optimal ranges, increased toxicity from chemicals often results from increased metabolic rates (Sprague 1985). Under colder temperatures fish have been shown to be more susceptible to at least copper, zinc, selenium, and cyanide, although the mechanisms of toxicity are unclear (Hodson and Sprague 1975; Kovacs and Leduc 1982; Dixon and Hilton 1985; Erickson et al. 1987; Lemly 1993; Hansen et al. 2002).
Flow	**In the wild.** Fish expend energy to hold their position in streams and to compete for and defend preferred positions that provide optimal feeding opportunity from the drift for the energy expended. Subordinate fish are forced to less profitable positions and become disadvantaged. Subordinate fish in lab settings still get adequate nutrition from feeding. Chemical exposure can reduce swimming stamina or speeds, as can exposure to soft water. Chemical exposures in soft water can be expected to exacerbate effects (Adams 1975; Kovacs and Leduc 1982; McGeer et al. 2000; De Boeck et al. 2006).
Disease and Parasites	**In the wild.** Disease and parasite burden are common in wild fish, but toxicity tests that used diseased fish are likely to be considered compromised and results would not be used in criteria compilations. Chemical exposure may weaken immune responses and increase morbidity or deaths (Stevens 1977; Arkoosh et al. 1998a, b).

Factor	Are effects likely more severe in typical lab settings, or effects uncertain, or effects more severe in the wild?
Predation	**In the wild.** Fish use chemical cues to detect and evade predators; these can be compromised by some chemical exposures (Berejikian et al. 1999; Phillips 2003; Scott et al. 2003; Labenia et al. 2007).
Exposure	
Variable exposures	**In the lab.** Most toxicity tests used to develop criteria are conducted at nearly constant exposures. Criteria are expressed not just as a concentration but also with an allowed frequency and duration of allowed exceedances. In field settings, most point or non-point pollution scenarios that rarely if ever exceed the criteria concentration (i.e., no more than for one four day interval per 3 years) will have an average concentration that is less than the criteria concentration. For some chemicals, such as copper, fish might detect and avoid harmful concentrations if clean-water refugia were readily available.
Metal form and bioavailability	**Uncertain.** Metals other than mercury and some organics are commonly assumed to be more bioavailable in the lab because DOC, which reduces the bioavailability and toxicity of several metals, is low in laboratory tests that are eligible for use in criteria. The Guidelines call for laboratory tests to have <5 mg/L TOC in order to be used in criteria derivation (Stephan et al. 1985), but probably more often TOC is <2 mg/L in laboratory studies. However, in mountainous streams in Idaho, TOC is often as low (≈1-2 mg/L) during baseflow conditions, so differences in bioavailability between streams and laboratory waters that both have low TOC are not necessarily large. Organic carbon is more often discussed as DOC in this Opinion. TOC includes particulates, which other than during runoff conditions in streams will tend to be low and thus TOC and DOC would be similar during conditions without runoff.
Chemical equilibria	**Uncertain.** While results conflict, metals are usually considered less toxic when in equilibrium with other constituents in water, such as organic carbon, calcium, carbonates and other minerals. In the wild, daily pH cycles prevent full equilibria from being reached (Meyer et al. 2007). Likewise, in conventional laboratory flow-through test designs chemicals may not have long enough contact time to reach equilibria. Static-renewal tests are probably nearly in chemical equilibria although organic carbon accretion can lessen toxicity which may not reflect natural settings (Santore et al. 2001; Welsh et al. 2008).
Prior exposure	**Uncertain.** If fish are exposed to sublethal concentration of a chemical, they could potentially either become weakened or become more tolerant of future exposures. With some metals, normally sensitive life stages of fish may become acclimated and less sensitive during the course of a chronic test if the exposure was started during the resistant egg stage (Chapman 1983, 1985; Sprague 1985; Brinkman and Hansen 2007).

Factor	Are effects likely more severe in typical lab settings, or effects uncertain, or effects more severe in the wild?
Life stage exposed	**In the wild.** Most lab studies are short term; realistically testing all life stages of anadromous fish is probably infeasible. Reproduction is often the most sensitive life stage with fish but most "chronic" studies are much shorter and just test early life stage survival and growth (Suter et al. 1987). At different life stages and sizes, salmonids can have very different susceptibility to some chemicals; even when limited to a narrow window of young-of-the-year (YOY) fry, sensitivity can vary substantially (this review). Unless the most sensitive life stages are tested, lab tests could provide misleadingly high toxicity values for ESA-listed species (further discussion follows in the text).
Chemical mixtures	**In the wild.** In field conditions, organisms never experience exposure to a single pollutant; rather, ambient waters typically have low concentrations of numerous chemicals. The toxic effects of chemicals in mixture can be less than those of the same chemicals singly, greater than, or have no appreciable difference. The best known case of one toxicant reducing the effects of another is probably selenium and mercury (e.g., Belzile et al. 2006). However, strongly antagonistic responses are probably uncommon, and much more common are situations where chemical mixtures have greater toxicity than each singly or little obvious interaction (e.g., Norwood et al. 2003; Borgert 2004; Playle 2004; Scholz et al. 2006; Laetz et al. 2009). In general, it seems prudent to assume that if more than one toxicant were jointly elevated it is likely that lower concentrations of chemicals would be required to produce a given magnitude of effect than would be predicted from their actions separately. However, the magnitude or increased effects at environmentally relevant concentrations is uncertain and for some combinations may be slight or imperceptible.
Dietary exposures	**In the wild.** Toxicity test data used in criteria development have been mostly based solely on waterborne exposures; yet in the wild, organisms would be exposed to contaminants both through dietary and water exposures. With at least some organics (e.g., dioxins, polychlorinated binphenyls) dietary exposures are more important than water exposures as is the case for some inorganics (e.g., arsenic, mercury, and selenium). For some other metals (e.g., cadmium, copper, lead), at environmentally relevant concentrations that would be expected when waterborne concentrations are close to criteria, dietary exposures have not been shown to directly result in appreciable adverse effects to fish (Hansen et al. 2004; Schlekat et al. 2005; Erickson et al. 2010). However, while dietary exposures of metals have not yet been implicated in adverse effects to fish at or below criteria concentrations, they may in fact be both the primary route of exposure and an important source of toxicity for benthic invertebrates (Irving et al. 2003; Poteat and Buchwalter 2014). For instance Besser et al. (2005) found that the effects threshold for lead to the benthic crustacean *Hyalella* was well above the chronic criterion in water exposures, but when lead was added to the diet, effects threshold dropped to near criteria concentrations.

Factor	Are effects likely more severe in typical lab settings, or effects uncertain, or effects more severe in the wild?
Population Dynamics	
Density effects	**In the lab.** Salmonid fishes are highly fecund (~500 to 5,000 eggs per spawning female). When abundant, overcrowding and competition for food and shelter may result in relatively high death rates for some life stages, particularly YOY during their first winter. After many fish die in a density-dependent bottleneck, the survivors have greater resources and improved growth and survival. Conceptually, if an acute contamination episode killed off a significant portion of YOY fish prior to their entering a resource bottleneck, then assuming no residual contaminant effects, the losses to later life stages and to adult spawners would be buffered.
Meta-population dynamics	**In the lab.** If habitats are interconnected, as is the case in intact stream networks, then if pervasive contamination from discharges to a stream were to impair only some endpoints or life-stages, such as reproductive failure or YOY mortalities, immigration from source populations may make detection of population reductions in the affected sink population difficult (Ball et al. 2006; Palace et al. 2007). If an episodic contamination pulse were to kill a large proportion of fish in a stream, the proximity of refugia and donors from source populations affect recovery rates (Detenbeck et al. 1992).

Considering all the reasons why the effects of a given chemical concentration could have more or less severe effects in laboratory settings or the wild, general conclusions are elusive. 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 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.*"

Based on this analysis, 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 criteria are applied, there is risk that laboratory tests underpredict effects in the wild.

2.5.2.2 Protection Afforded By the Species Sensitivity Distribution Approach

EPA's fundamental approach to setting criteria involves compiling reports of laboratory tests for species and genus mean values, rank ordering the genus mean values, and basing criteria concentrations on the 5th percentile distribution of the rank ordered values. A major assumption inherent in this approach is that effects deemed statistically significant in laboratory tests are reasonable predictors of toxic effects in the natural environment. Many authors have noted a number of concerns with this approach (Cairns 1986; Forbes and Forbes 1993; Hopkin 1993; Smith and Cairns 1993; Underwood 1995; Power and McCarty 1997; Aldenberg and Jaworska 2000; Newman et al. 2000; Forbes and Calow 2002; Suter et al. 2002; Duboudin et al. 2004; Brix et al. 2005; Maltby et al. 2005; Forbes et al. 2008). Those concerns include:

- Relevancy of single-species tests to natural ecosystems;

- Bias of small datasets toward more or less sensitive species than what would be expected in natural ecosystems;

- Acceptance of the loss of species from an ecosystem due to a toxic contaminant;

- Appropriateness of the 5th percentile as an acceptable level of protection;

- Bias toward toxicity data specific to mortality rather than other endpoints that are also ecologically meaningful;

- Lack of accounting for variability in the toxicity data; thus apparent differences between species ranks may not be meaningful, especially for species with few datapoints; and

- Uncertainties in the statistical properties of the distributions and appropriate models.

On the other hand, some authors have found reasonably good agreement between effects in the laboratory and field tests (Geckler et al. 1976; de Vlaming and Norberg-King 1999), and lack of pronounced adverse effects in ecosystem tests at criteria-like concentrations below the 5th percentile of a species sensitivity distribution (SSD) (Versteeg *et al*. 1999; Mebane 2010).

No explicit process for the protection of exceptionally vulnerable populations of threatened or endangered species was included in the Guidelines. However, it is clear from subsequent writings by the authors that they thought criteria should specifically protect or be adjusted to protect socially valued special status species, including threatened and endangered species. For instance, the introduction to the Guidelines states that "*to be acceptable to the public and useful in field situations, protection of aquatic organisms and their uses should be defined as prevention of unacceptable long-term and short-term effects on (1) commercially, recreationally, and other important species….*" as well as fish and invertebrate assemblages (Stephan et al. 1985). Other writings and guidance are more explicit about the need to consider protection of species listed under the ESA; suggesting a review of whether the 95% of protected species included ESA-listed species and adequate prey for them (Stephan 1985, 1986; EPA 1994). If not, the criteria should be adjusted to protect these "critical" species. Such reviews and adjustments were recommended to be done on a site-specific basis, where a "site" may be a state, region, watershed, water body, or segment of a water body (EPA 1994).

2.5.2.3 Protectiveness of the Acute Adjustment Factor

One challenge for deriving aquatic life criteria for short-term (acute) exposures is that the great majority of available data is for mortality. Most often, the data is reported as the lethal concentration that kills 50 percent of the test organisms (LC_{50}). A fundamental assumption of EPA's criteria derivation methodology is that the final acute value (i.e., the LC_{50} for a hypothetical species with a sensitivity equal to the 5th percentile of the SSD), may be divided by two in order to extrapolate from a concentration that would likely be extremely harmful to sensitive species in short-term exposures (kill 50% of the population) to a concentration expected to kill few, if any, individuals. This assumption, which must be met for acute criteria to be protective of sensitive species, is difficult to evaluate from published literature because so few studies report the data behind an LC_{50} test statistic. While LC_{50} values are almost universally used in reporting short-term toxicity testing, they are not something that can be "measured" but are statistical model fits. An acute toxicity test is actually usually a series of four to six tests run in parallel in order to test effects at different chemical concentrations. An LC_{50} is estimated by a statistical distribution or regression model which generates an LC_{50} estimate, usually a confidence interval, and then all other information is thrown away. Thus, while the original test data included valuable information on what concentrations resulted in no, low, or severe effects, that information is lost to reviewers unless the unpublished raw lab data are available to them.

The assumption that dividing an LC_{50} by two will result in a no- or very low effects concentration rests on further assumptions of the steepness of the concentration-response slope. NMFS (2014b) evaluated several metals toxicity studies which had a range of response slopes. These studies were selected from data sets that were relevant to salmonid species in Idaho and for which the necessary data to evaluate the range of responses could be located (Chapman 1975, 1978b; Marr et al. 1995b; Marr et al. 1999; Mebane et al. 2010; Mebane et al. 2012). The citations are to reports with detailed enough original data to examine the mortality at the LC_{50} concentration divided by two. The vast majority of published data were inadequate for this comparison, because usually only the LC_{50} values are reported, not the actual responses by concentration. NMFS (2012; 2014c) examined around 100 tests for this comparison, and found a variety of concentration-response slopes, from very shallow to very steep In the shallowest slopes, a concentration equal to ½ of the LC_{50} concentration would still result in 15% to 20% mortality. However, a more common pattern with the metals data was that a concentration equal to one-half the LC_{50} value would probably result in about a 5% death rate, and in many instances, no deaths at all would be expected.

In one of the few additional published sources that gave relevant information, 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 concentration by two would usually kill few if any fish, although it does not bode well for arguments of the overall protectiveness of criteria concentrations in mixtures.

Other reviews include Dwyer et al. (2005) who evaluated the "$LC_{50}/2$" assumption with the results of the acute toxicity testing of 20 species with five chemicals representing a broad range of toxic modes of action. In those data, multiplying the LC_{50} by a factor of 0.56 resulted in a low (10%) or no-acute effect concentration. Testing with cutthroat trout (*O. clarkii*) and cadmium, lead, and zinc singly and in mixtures, Dillon and Mebane (2002) found that the $LC_{50}/2$ concentration corresponded with death rates of 0% to 15%.

In summary, the assumption that one-half of an LC_{50} concentration for a sensitive test (i.e., a concentration near the 5th percentile of the ranked species sensitivities) will result in little or no deaths was supported by several data sets plus two published articles. While up to 20% mortality was calculated, in most cases the expected morality associated with a $LC_{50}/2$ was less than 10% and often zero.

2.5.2.4 Susceptibility of Salmonids to Chemicals at Different Life Stages

EPA's Guidelines recommend that if the available data indicate that some life stages are at least a factor of two more resistant than other life stages, the data for the more resistant life stages should not be used to calculate species mean acute values (Stephan et al. 1985). Smaller, juvenile life stages of fish are commonly expected to be more vulnerable to metals toxicity than larger, older life stages of the same species. For instance, a standard guide for testing the acute toxicity of fish recommends that tests should be conducted with juvenile fish, that is, post-larval or older and actively feeding, usually in the size range from 0.1 and 5.0 grams (g) in weight (ASTM 1997).

A review of several datasets in which salmonids of different sizes were similarly tested shows that even among juvenile fish in the 0.1 to 5.0g size range, differences in sensitivity can approach a factor of 10. This emphasizes the importance of EPA's guidance not to use the more resistant life stages. However, the datasets analyzed indicated that in practice, there were sometimes greater influences of life stage on the sensitivity of salmonids to some substances than was apparent to the authors of the individual criteria documents using the datasets available to them at the time. In these instances, some of the species mean acute values (SMAVs) and genus mean acute values (GMAVs) which were used to rank species sensitivity and set criteria concentrations at levels considerably higher than LC_{50}s with salmonids that were tested at the most sensitive life stages (NMFS 2012; NMFS 2014b). This resulted from the inclusion of toxicity tests using less sensitive life stages in the calculation of the SMAVs and GMAVs.

For three Pacific salmonid species (coho salmon, rainbow trout, and cutthroat trout) for which comparable test data were available for different life stages, the data suggest that swim-up fish weighing around 0.5g to about 1g may be the most sensitive life stage. None of the datasets examined in detail or other published studies reviewed had sufficient resolution to truly define at what weight fish became most sensitive to metals, but along with other data they suggest that larger fish may be less sensitive than fish at 0.4 to 0.5g. For instance with zinc, rainbow trout in the size range of about 0.1 to about 1.5g consistently became more sensitive to zinc in two studies with multiple tests in that size range. All data located for early swim-up stage *Oncorhynchus* in the 0.1 to 0.5g range were consistent with increasing sensitivity with size. With Hansen et al. (1999b) rainbow trout studies, this relationship continued with fish up to about 1.5g. However, with cutthroat trout, the few data available suggests that fish larger than about 0.5g are less sensitive with increasing size (NMFS 2012; NMFS 2014b).

Some studies with older and larger rainbow trout have found that the fish became more resistant to zinc and copper (Chakoumakos et al. 1979; Chapman 1978b; Chapman and Stevens 1978; Howarth and Sprague 1978). Studies with copper all showed this trend, but the strength of size-sensitivity relations varied across studies. Chakoumakos et al. (1979) found that fish between about 1 and 25g in weight varied in their sensitivity to copper by about 8 times, but steelhead that were tested with copper at sizes of 0.2, 7, 70, and 2700g showed little pattern of sensitivity with size (Chapman and Stevens 1978; Chapman 1978b). However, the large differences in sizes may have missed changes at intermediate sizes in the ranges compared. Similarly, with copper and rainbow trout, Anderson and Spear (1980) found that rainbow trout at sizes of 3.9, 29 to 176g had similar sensitivities.

NMFS (2012; 2014c) reviewed several data sets that indicated increasing susceptibility of salmonids to at least metals with increasing size and age as fish progressed from the resistant alevin stage. Salmonids can have profound difference in susceptibility to chemicals at different life stages and in some instances SMAVs used in criteria may be skewed high because insensitive life stages were included. A "U" shaped pattern of sensitivity with life stage was suggested for several datasets with Pacific salmon or trout species (i.e., *Oncorhynchus*) and some metals. Across several good datasets, the most vulnerable life stage and size appeared to be swim-up fry weighing between about 0.5 to 1.5g. However, no consistent pattern was obvious across other species of fish, chemicals, and life stages.

Ultimately, caution is needed when using SMAVs or GMAVs as summary statistics for ranking species sensitivity or setting criteria. Reviews of the protectiveness of chemical concentrations or criteria that rely in large part upon published mean acute values for species of special concern such threatened species, or their surrogates, may be subject to considerable error if the underlying data points are not examined.

2.5.2.5 Effects of Acclimation on Susceptibility to Chemicals

Exposure to sublethal concentrations of organic chemicals and other metals may result in pronounced increases in resistance to later exposures of the organisms. With metals, the increased resistance may be on the order of two to four times for acute exposures, but may be much higher for some organic contaminants (Chapman 1985). However, the increased resistance can be temporary and can be lost in as little as seven days after return to unpolluted waters (Bradley et al. 1985; Sprague 1985; Hollis et al. 1999; Stubblefield et al. 1999). For this reason, EPA's Guidelines specify that test results from organisms that were pre-exposed to toxicants should not be used in criteria derivation (Stephan et al. 1985).

However, there is a less obvious source of acclimation that is not precluded by the Guidelines and influences chronic values and thus chronic criteria. Several tests have shown that life stages typically sensitive to toxins (e.g., fry stage) become more resistant when toxicity tests were initiated during resistant early life stages (ELS, e.g., embryo stage). This suggests that acclimation to toxin(s) during ELS exposure may lead to greater resistance in later life stages in comparison to the same life stages of fish which had no previous exposure (Chapman 1978a; Spehar et al. 1978; Chapman 1994; Brinkman and Hansen 2004, 2007). The Guidelines could actually be interpreted to exclude chronic exposures that did not pre-expose, and acclimate fish to metals as eggs (Stephan et al. 1985).

Chapman (1994) exposed different life stages of steelhead for the same duration (3 months) to the same concentration of copper (13.4 µg/L at a hardness of 24 mg/L as $CaCO_3$). The survival of steelhead which were initially exposed as embryos was no different from that of the unexposed control fish, even though the embryos developed into the usually-sensitive swim-up fry stage during the exposure. In contrast, steelhead which were initially exposed as swim-up fry without the opportunity for acclimation during the embryo state, suffered complete mortality. Brinkman and Hansen (2007) compared the responses of brown trout (*Salmo trutta*) to long-term cadmium exposures that were initiated either at the embryo stage or the swim-up fry stage (i.e., chronic growth and survival tests). In three comparative tests, fish that were initially exposed at the swim-up fry stage were consistently two to three times less resistant than were the fish initially exposed at the embryo stage.

These studies support the counterintuitive conclusion that because of acclimation, longer-term tests or tests that expose fish over their full life cycle are not necessarily more sensitive than shorter-term tests which are initiated at the sensitive fry stage. Conceptually, whether this phenomenon is important depends on the assumed exposure scenario. If it were assumed that spawning habitats would be exposed, then the less-sensitive ELS tests would be relevant. However, for migratory fishes such as ESA-listed salmon and steelhead, their life histories often involve spawning migrations to headwater reaches of streams, followed downstream movements of fry shortly after emerging from the substrates, and followed by further seasonal movements to larger, downstream waters to overwinter (Willson 1997; Baxter 2002; Quinn 2005). These life history patterns often correspond to human development and metals pollution patterns such that headwater reaches likely have the lowest metals concentrations, and downstream increases could occur due to point source discharges or urbanization.

In chronic tests with salmonids and metals, the Guidelines inadvertently favor a test method (ELS tests) that may be inherently biased toward insensitivity because acclimation can occur during the insensitive egg stage of exposure. Thus, Species Mean Chronic Values (SMCV) listed in criteria documents may be also be biased high.

2.5.2.6 Flow-Through, Renewal, or Static Exposure Test Designs

One area of controversy in evaluating toxicity test data or risk assessments, or criteria derived from them, has to do with potential bias in how test organisms are exposed to test solutions. Exposures of test organisms to test solutions are usually conducted using variations on three techniques. In "static" exposures, test solutions and organisms are placed in chambers and kept there for the duration of the test. The "renewal" technique is like the static technique except that test organisms are periodically exposed to fresh test solution of the same composition, usually once every 24 hours or 48 hours, by replacing nearly all the test solution. In the "flow-through" technique, test solution flows through the test chamber on a once-through basis throughout the test, usually with at least five volume replacements/day (ASTM 1997).

The term "flow-through test" is commonly mistaken for a test with flowing water, i.e., to mimic a lotic environment in an artificial stream channel or flume. This is not the case; rather the term refers to the once-through, continuous delivery of test solutions (or frequent delivery in designs using a metering system that cycles every few minutes). Flows on the order of about five volume replacements per 24 hours are insufficient to cause discernable flow velocities. In contrast, even very slow moving streams have velocities of around 0.04 feet per second (ft/sec) or more. At that rate, a parcel of water would pass the length of a standard test aquarium (~2 ft) in about 48 seconds, resulting in about 9,000 volume replacements per day. A more typical stream velocity of about 0.5 ft/sec would produce over 100,000 volume replacements per day.

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). Flow-through exposures have been preferred in the development of standard testing protocols and water quality criteria. The Guidelines first advise that for some highly volatile, hydrolysable, or degradable materials, it is probably appropriate to use only results of flow-through tests. However, this advice is followed by specific instructions that if toxicity test results for a species were available from both flow-through and renewal or static methods, then results from renewal or static tests are to be discounted (Stephan et al. 1985). Thus, depending upon data availability, toxicity results in the criteria databases may be a mixture of data from flow-through, renewal or static tests, raising the question of whether this could result in bias. In the Guidelines, the rationale for the general preference for flow-through exposures was not detailed, but it was probably based upon assumptions that static exposures will result in LC_{50} values that are biased high (apparently less toxic) than comparable flow-through tests, or that flow-through tests have more stable exposure chemistries and will result in more precise LC_{50} estimates.

Static exposures studies often yield LC_{50} 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, with differences in some cases of an order of magnitude lower. For example, LC_{50} values for static tests have been determined to be approximately 20 times higher than those from flow-through tests for DDT (Earnest and Benville 1971). 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 lower in flow-through than static tests (Birge et al. 1979; 1981). Static assays were also found to underestimate the toxicity of endosulfan in comparisons with flow-through systems (Naqvi and Vaishnavi 1993). Several additional studies with a variety of compounds report increased toxicity in flow-through compared to static systems (e.g., Erickson et al. 1998; Hedtke and Puglisi 1982; Vernberg et al. 1977; Randall et al. 1983; Burke and Ferguson 1969). Static conditions may underestimate the true exposure concentration because the fish will deplete the concentration in solution over time, causing a lack of steady-state exposure. Acute LC_{50} concentrations for salmonids that are based on static tests could therefore underestimate toxicity.

With metals, renewal tests can produce higher EC_{50} concentrations (i.e., metals were less toxic), probably because of accretion of DOC (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 copper complexation with organic carbon, which reduces acute toxicity, is not instantaneous, and typical flow-through exposure systems allowed insufficient hydraulic residence time for complete copper-organic carbon complexation to occur. Davies and Brinkman (1994) similarly found that cadmium and carbonate complexation was incomplete in typical flow-through designs, although in their study incomplete complete complexation had the opposite effect of the copper studies, with cadmium in the aged, equilibrium waters being more toxic. A further complication is that it is not at all clear that natural flowing waters should be assumed to be in chemical equilibria because of tributary inputs, hyporheic exchanges and daily pH, inorganic carbon, and temperature cycles. Predicting or even evaluating risk of toxicity through these cycles is complex and seldom attempted (Meyer et al. 2007), in part because pulse exposures cause latent mortality (i.e., fish die after exposure to the contaminant is removed), a phenomenon that is often overlooked or not even recognized in standard acute toxicity testing.

When comparing data across different tests, it appears that other factors such as testing the most sensitive sized organisms or organism loading may be much more important than if the test was conducted by flow through or renewal techniques. For instance, Pickering and Gast's (1972) study with fathead minnows (*Pimephales promelas*) and cadmium produced flow-through LC_{50} concentrations that were lower than comparable static LC_{50} values (~ 4,500 to 11,000 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 2g. The size of the fish used in their flow-through acute tests were not given, but is assumed to have been similar. In contrast, 8 to 9 day old fathead minnow fry usually weigh about 1 mg or less (EPA 2002b). Using newly hatched fry weighing about 1/1000th of the fish used by Pickering and Gast (1972) in the 1960s, and modern protocols, cadmium LC_{50} concentrations for fathead minnows at similar hardnesses tend to be around 50 µg/L, with no obvious bias for test exposure. Similar results have been reported with brook trout (*Salvelinus fontinalis*). One each flow-through and static acute tests with brook trout were located, both conducted in waters of similar hardness (41 to 47 mg/L). The LC_{50} of the static test which used fry was <1.5 µg/L whereas the LC_{50} 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 LC_{50} 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 factors such as the life stage of exposures, can dwarf the influence of flow-through or renewal methods for the acute toxicity of at least metals.

2.5.2.7 Mixture Toxicity

In point or nonpoint source pollution, chemicals occur together in mixtures, but criteria for those chemicals are developed in isolation, without regard to 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 same 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 has been inconsistently used (e.g., Sprague 1970; Marking 1985; Borgert 2004; 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., Alabaster and Lloyd 1982; Spehar and Fiandt 1986; EIFAC 1987; Enserink et al. 1991; Sorenson 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 percent 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 no effect concentrations (NOEC) 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 a reasonable and usually protective (Norwood et al. 2003; Meador 2006; Alabaster and Lloye 1982). The aluminum criteria evaluated in this Opinion was developed as if it 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.

2.5.2.8 Sufficiency of Toxicity Endpoint Selection

For acute criteria, the toxicity endpoint used is mortality. In the case of chronic criteria, the most commonly used toxicity endpoints are survival, growth, and reproduction. Data on other sublethal effects such as swimming performance, predator avoidance, or other altered behaviors are often not included. These sublethal effects, which can come about as a result of either acute or chronic exposures cannot be considered trivial, because they are associated with the potential for increased mortality. Sublethal effects involving alterations in behavior can occur during relatively low concentration, short-term exposure, and can have profound biological implications (e.g., chemical migration barrier, interference with spawning behavior). NMFS recognizes that relevant data may not be available for all toxic substances, and that determination of a repeatable, detectable endpoint may involve a degree of subjectivity. Relatively little data are available to help elucidate these concerns; however, the research that does exist indicates that sublethal effects can be very serious for at least some toxicants.

Based on this analysis, the risks of sublethal effects will exacerbate adverse effects, and are likely to result in sublethal effects, such as interference in physiochemical processes, interruption of ecological interactions, changes in pathological stress, and toxicosis of listed species considered in this opinion.

2.5.2.9 Criteria Frequency, Duration, and Magnitude Exposure Components

For simplicity, much of the discussion of the water quality criteria that are the subject of this consultation treats the criteria as though they were defined solely as a concentration in water. However, the action actually defines aquatic life criteria in three parts: a concentration, a duration of exposure, and an allowable exceedance frequency. The 4-day and 1-hour duration and averaging periods for the chronic and acute criteria, respectively, were based upon judgments by EPA authors that included considerations of the relative toxicity of chemicals in fluctuating or constant exposures. The Guidelines considered an averaging period of 1 hour most appropriate to use with the acute criterion because high concentrations of some materials could cause death in 1 to 3 hours. Also, even when organisms do not die within the first few hours, few toxicity tests attempt to monitor for latent mortality by transferring the test organism into clean water for observation after the chemical exposure period is over. Thus, it was not considered appropriate to allow concentrations above the CMC for more than 1 hour (Stephan et al. 1985).

A review of more recent information supported EPA's judgments from the 1980s that if an averaging period is used with acute criteria for metals, it should be short. Some of the more relevant research relates the rapid accumulation of metals on the gill surfaces of fish to their later dying. When fish are exposed to metals such as cadmium, copper, or zinc, a relatively rapid increase in the amount of metal bound to the gill occurs above background levels. This rapid increase occurs during exposures on the order of minutes to hours, and these brief exposures have been sufficient to predict toxicity at 96 to 120 hours. The half saturation times for cadmium and copper to bind to the gills of rainbow trout may be on the order of 150 to 200 seconds (Reid and McDonald 1991). Several other studies have shown that exposures well under 24 hours are sufficient for accumulation to develop that is sufficient to cause later toxicity (Playle et al. 1992; Playle et al. 1993; Zia and McDonald 1994; Playle 1998; MacRae et al. 1999; Di Toro et al. 2001). Acute exposures of 24 hours might not result in immediate toxicity, but deaths could result over the next few days. Simple examination of the time-to-death in 48- or 96-hour exposures would not detect latent toxicity from early in the exposures. The few known studies that tested for latent toxicity following short-term exposures have demonstrated delayed mortality following exposures on the order of 3 to 6 hours (Marr et al. 1995a; Zhao and Newman 2004, 2005; Diamond et al. 2006; Meyer et al. 2007). Observations or predictions of appreciable mortality resulting from metals exposures on the order of only 3 to 6 hours supports the earlier recommendations by Stephan and others (1985) that the appropriate averaging periods for the CMC is on the order of 1 hour.

The 4-day averaging period for chronic criteria was selected for use by EPA with the CCC for two reasons (Stephan et al. 1985). 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 1 month would allow for substantial fluctuations above the CCC. Whether fluctuating concentrations would result in increased or decreased adverse effects from those expected in constant exposures seems to defy generalization. A comparison of the effects of the same average concentrations of copper on developing steelhead that were exposed either through constant or fluctuating concentrations found that steelhead were about twice as resistant to the constant exposures as they were to the fluctuating exposures (Seim et al. 1984). Similarly, *Daphnia magna* exposed to daily pulses of copper for 6 hours at close to their 48-hour LC_{50} concentrations had more severe effects after 70 days than did comparisons that were exposed to constant copper concentrations that were similar to the average of the daily fluctuations (Ingersoll and Winner 1982). In contrast, cutthroat trout exposed instream to naturally fluctuating zinc concentrations survived better than fish tested under the same average, but constant zinc concentrations (Nimick et al. 2007; Balistrieri et al. 2012). Thus, literature reviewed either supports or at least do not contradict EPA's position on averaging periods.

The third component of criteria, EPA's once-per-3-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 3 years. The EPA has subsequently 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 3-year period seemed reasonably supported or at least was not contradicted by the information NMFS (2012; 2014c) reviewed.

A more difficult evaluation is the exceedance magnitude, which is undefined and thus not limited by the letter of the criteria. Thus, by the definition, a once-per-3-year exceedance that has no defined limits to its magnitude, could be very large, and have large adverse effects on ESA-listed species. However, within the 4-day and 1-hour duration constraints of the criteria definitions, some estimates of the potential magnitude of exceedances that could occur without "tripping" the duration constraints can be calculated. This is because environmental data such as chemical concentrations in water are not unpredictable but can be described with statistical distributions, and statements of exceedance probabilities can be made. Commonly with water chemical data and other environmental data, the statistical distributions do not follow the common bell-curve or normal distribution, but have a skewed distribution with more low than high values. This pattern may be approximated with a log-normal statistical distribution (Blackwood 1992; Limpert et al. 2001; Helsel and Hirsch 2002; Delos 2008).

NMFS (2014b) evaluated three hypothetical scenarios to illustrate contaminant concentrations that could occur without violating the exceedance frequency and duration limitations of the proposed criteria. The scenarios use randomly generated values from a log-normal distribution with different variabilities and serial correlations. Serial correlation refers to the pattern in environmental data where values at time one are often highly correlated with values at time two and so on. For example, a hot day in summer is much more likely to be followed by another hot day than a bitterly cold day, a low chemical concentration during stable low flows on a day in September will most likely be followed by low chemical concentration the next day, a high chemical concentration in a stream during runoff on a day in April will more likely to be repeated by another high concentration, and so on (Helsel and Hirsch 2002; Delos 2008).

The first scenario involved concentrations that were close to the criteria where organisms would have little relief of exposure for recovery. Scenario 2 involved mean concentrations well below the criterion coupled with concentrations that were slightly above the CCC followed by a recovery opportunity. Scenario 3 might be more likely in runoff of nonpoint pollutants from snowmelt or stormwater, where concentrations were generally well below the CCC, but there were instances of concentrations that were substantially greater than the CCC. In these scenarios, sensitive populations could experience effects ranging from appreciable reduction in abundance if the contaminant pulse hit during a sensitive part of their life history, to no effect if it hit during a resistant phase or if the ESA-listed species was less sensitive than the species that drove the criteria calculations.

An actual event that was very similar to Scenario 3 occurred when an upset at a large, industrial mining operation caused elevated cadmium concentrations in Thompson Creek, a tributary to the upper Salmon River in Idaho. In April 1999, a pulse of cadmium about 30 times higher than background, 2.6 times higher the chronic criterion, and equal to the acute criterion was detected. The duration of exceedance was probably greater than a day and less than a week. By August 1999, when a biological survey was conducted, few if any adverse effects could be detected in the benthic community structure. Whether subtle differences between unaffected upstream survey sites were lingering effects of the disturbance or just differences in naturally patchy stream invertebrate communities was unclear. However, it does suggest that benthic communities in similar mountain streams would be either resilient to, or recover quickly from criteria exceedances of this magnitude when it occurs at a small enough spatial scale (Mebane 2006).

These hypothetical scenarios used a simplified, fixed criterion, whereas in actuality, some of EPA's criteria vary and may be positively correlated with the concentrations of metals in water. If the criteria accurately reflect risks from varying environmental conditions, and if ambient conditions co-vary with and are positively correlated with criteria, this will tend to lessen risks resulting from ambient increases in concentration. In cases where the criteria were positively correlated with the contaminants, such as in example for Pine Creek with cadmium or the biotic ligand model-copper example for Panther Creek (NMFS 2014b), the frequency and magnitude of exceedances is expected to be less than if the criteria and contaminant concentrations did not rise and fall together. This is because the contaminant and another water quality parameter that mitigates toxicity have common sources and rise and fall together, such as cadmium and calcium in Pine Creek where the source for both is probably weathering of gangue rock and spring snowmelt and runoff appears to dilute both. In the Panther Creek example, copper and DOC tended to rise and fall together with snowmelt and runoff, similarly mitigating exceedance frequency and magnitude.

While NMFS (2014b) did not locate any plausible examples of negative correlations between contaminants and important factors modifying toxicity, it is likely that such scenarios do occur somewhere because if the event that releases the contaminant, such as a runoff pulse from a storm or snowmelt, caused a contaminant spike from washing accumulations into a stream and at the same time lowered the pH and hardness, then the magnitude of exceedances could be more severe. Such a circumstance could be plausible for metals such as cadmium, lead, or zinc in which hardness is a major modifier of toxicity.

Further, the actual possibility that an extreme exceedance would occur and be "allowed" under the exceedance policy seems unlikely. This is because in natural waters seasonal and hydrologic factors tend to cause concentrations to be serially correlated, that is low concentrations follow low concentrations and high concentrations follow high concentrations (Helsel and Hirsch 2002; Delos 2008). Thus for an extreme exceedance to be allowable under the chronic criteria 4-day average concentration definition, it would also have to not exceed the 1-hour acute criteria definition. A very large exceedance of the sort described in Scenario 3, would likely span across more than one, 1-hour averaging period for acute criteria and "violate" the one exceedance per 3-year recurrence interval term. While there are no regulatory limits on the upper concentration of an exceedance of the 1-hour acute criteria, the idea that a chemical concentration in a natural water could rapidly rise to acutely toxic concentrations and then drop back down to below criteria seems like a remote possibility. Although, in urban watersheds that are characterized by high proportions of impervious surface, runoff is flashier than in forested watersheds, and short-term pulse exposures could occur in those settings (Booth et al. 2002). In the predominately forested areas of the action area, such scenarios seem less likely.

The 1-hour and 4-day exceedance durations for acute and chronic criteria respectively are supported by the science as reasonable and adequately protective. Whether the allowable 1 in 3 year's exceedance frequency is sufficiently protective was difficult to evaluate, in part because the magnitude of allowable exceedances is undefined. However, the likelihood that a runoff pulse could both rise and fall so high within an hour that it could cause acute effects without exceeding the acute criteria seems unlikely. This remains an aspect of uncertainty regarding the protectiveness of criteria.

2.5.3 Effects of Expressing Criteria as a Function of pH, DOC, and Hardness

Aluminum is considered a non-essential metal because it does not have a meaningful biological function. Aluminum generally acts as a surface active toxicant by binding to anionic sites on respiratory surfaces (e.g., fish gills), which in turn leads to both ionoregulatory and respiratory effects (Wood et al. 1997, as cited in EPA 2008). Factors that influence the fate and transport of aluminum (i.e., pH, DOC, and total hardness) also affect its bioavailability, and hence the toxicity, of aluminum.

The most important parameter for aluminum toxicity is pH because aluminum speciation and solubility is strongly correlated with pH (Caldwell et al. 2018). At low pH (e.g., pH <5), ionoregulatory effects dominate and include a mechanism similar to hydrogen ion toxicity alone, i.e., blockage of sodium uptake (Playle and Wood, 1989). In moderately acidic water (e.g., pH <6.5), respiratory effects generally predominate. Under these conditions, 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. The toxicity of aluminum appears to be lowest at neutral pH, with toxicity tending to increase with either an increase or decrease in pH. DeForest et al. (2018a) also found that aluminum toxicity increased as pH decreased from 7 to 6.2, and toxicity decreased as pH increased from 7 to about 8. EPA's 2018 Final Aquatic Life Ambient Water

Quality Criteria for Aluminum provides an overview of the processes related to the toxicity of aluminum in varying pH values and the effect of pH on the derivation of water quality criteria.

Aluminum sorbs to organic matter, thus aluminum is less bioavailable in waters with higher concentrations of DOC (Wilson 2012). A number of authors have demonstrated DOC reducing aluminum toxicity (Neville 1985; Parkhurst et al., 1990; Lacroix et al. 1990; Witters et al. 1990b; Baldigo and Murdoch 1997; Gundersen et al. 1994; Roy and Campbell 1997). Gensemer and Playle (1999) provides a summary of many of these studies. The ameliorating effect of DOC may be more pronounced in higher pH waters, than in low pH where where hydrogen ions compete for binding sites (Parkhurst et al. 1990). DeForest et al. (2018a) reported that aluminum toxicity decreased with increasing DOC concentrations across all pH and hardness conditions.

Hardness also has an effect on the toxicity of aluminum. Gundersen et al. (1994) studied the modifying effects of water hardness against aluminum toxicity to rainbow trout. Increasing hardness (i.e., calcium concentrations) increased survival of trout compared to those in soft waters in studies of both short (96 hour) and longer (16 days) duration exposures. This is likely because the cation Al^{+3} competes with other cations, such as calcium, for uptake (Gensemer and Playle 1999). Typically, an increase in hardness, will decrease aluminum toxicity; however, in elevated pH conditions (e.g., pH 8), the impact of hardness on toxicity is reduced (Gensemer et al. 2018). DeForest et al. (2018a) reported decreasing aluminum toxicity with increasing hardness in low pH waters, but in higher pH waters, aluminum toxicity increased with hardness.

DeForest et al. (2018a, b) developed two models (one for invertebrates and one on for fish) to characterize the bioavailability of aluminum based on the effects of pH, total hardness, and DOC on aluminum toxicity. The authors then evaluated the ability of the two models (one for invertebrates and one for fish) to predict aluminum toxicity based on water chemistry conditions. Data for this effort included 23 *C. dubia* tests and 22 fathead minnow tests from Gensemer et al. (2018), plus an additional nine *C. dubia* and nine fathead minnow tests that were obtained in response to public comments on the draft national criteria recommendations in order to expand the ranges of water chemistry conditions for use in the models. DeForest et al. (2018a) noted that the slope of the dose-response curve varied with different combinations of DOC, pH, and hardness, indicating that interaction among the variables was impacting their effects on aluminum toxicity. As such, they evaluated models that included various terms representing the interaction of water chemistry conditions.

For invertebrates, the final model included all of the individual water chemistry conditions, in addition to a pH:hardness interaction term and a squared pH (pH^2) term (DeForest 2018b, *as cited in* EPA 2018). The negative pH^2 term accounts for the fact that aluminum toxicity varies with pH, as previously described. The negative pH:hardness interaction term accounts for the decreasing influence total hardness has on aluminum toxicity as pH increases. The adjusted R^2 for the final model was 0.88, and predicted toxicities were within a factor of two of observed values used to create the model for 97% of the tests (DeForest et al. 2018b, *as cited in* EPA 2018). The final fish model included all of the water chemistry terms in addition to terms for the pH:hardness and pH:DOC interactions. Using the final MLR model for fish, the predicted EC_{20} values (effects concentration causing a 20 percent reduction in the measured endpoint) were within a factor of two of the observed values used to create the model for 97% of the tests (DeForest et al. 2018b, *as cited in EPA 2018).*

EPA utilized the models developed by DeForest et al. (2018b, *as cited in EPA 2018)* to normalize both acute and chronic toxicity data used in developing sensitivity distributions for criteria derivation. There are uncertainties associated with this approach, as noted by DeForest et al. 2018a). First, there is uncertainty in assuming the model developed for fathead minnow applies to other fish species. Second, there is uncertainty in assuming the model developed for one particular toxicity endpoint at a specific exposure duration (e.g., chronic studies examining biomass) is applicable to other exposure durations and toxicity endpoints (e.g., short-term studies examining mortality). Considering aluminum's mechanism of toxicity, and in absence of sufficient scientific information to assess the ability of the model to predict toxicity for other species, we have concluded that such assumptions are reasonable and the available data represents the best available scientific information.

2.5.4 Risk of Aluminum Toxicity to Aquatic Species

To evaluate the potential for adverse effects resulting from exposure to aluminum, we compared adverse effects indicated from short-term experiments of 4 days or less duration to the acute criteria concentrations that are intended to protect against short-term effects, and compared adverse effects shown in longer-term studies to the proposed chronic criteria concentrations. There were some conflicts in the scientific literature where for the same species and similar types of experiments, one study might find no ill effects from a given concentration and another might find severe effects. Thus, we considered the overall strength of the evidence for or against the protectiveness of criteria.

2.5.4.1 Acute Toxicity to ESA-listed Species

Due to concerns of acid rain most studies of aluminum toxicity have been conducted in acidic conditions (Gensemer and Playle 1999; Sparling and Lowe 1996; Cardwell et al. 2018). Studies involving circumneutral to basic conditions are relatively few in comparison, and there is relatively little toxicity data for pH ranges that are representative of Oregon waters (i.e., 6-8). According to EPA (2018), the specific mechanisms of aluminum toxicity at alkaline pH are not well understood.

For many fish, aluminum toxicity increases with early life stage such that eggs and endogenously-feeding alevins are generally less sensitive than exogenous-feeding swim-up larvae (Buckler et al. 1995; Delonay et al. 1993). Rainbow trout data from Gundersen et al. (1994), Atlantic salmon (*Salmo salar*) data from Hamilton and Haines (1995), and brook trout data from Tandjun (1982) were utilized in the criteria derivation. Hamilton and Haines (1995) utilized sac fry, a life stage thought to be slightly more tolerant that older life stages. As such, the criteria may be biased slightly high under some water chemistry conditions, as exhibited by the 8,642 µg/L SMAV (geometric mean of 20,749 and 3,599 µg/L). Other acute studies utilizing salmonids did not meet the criteria derivation guidelines and were not used for criteria derivation; however, they are utilized here to examine potential effects of the proposed criteria.

Gunderson et al. (1994) did not observe mortality when rainbow trout fry were exposed to aluminum concentrations greater than 8,000 µg/L in near-neutral pH (i.e., pH ~ 7.5) tests;

however, the authors noted that fish secreted noticeable amounts of mucus in the highest aluminum concentration tests. Mortality was observed in weakly alkaline pH (i.e., pH ~ 8.5) tests. One hundred percent of the fish survived 96-hour exposures to aluminum concentrations of 1,680 µg/L in tests waters with a pH of 8.25, low DOC, and moderate hardness.

Table 14 summarizes the short-term toxicity studies for salmonids where water chemistry information could be reasonable ascertained. In order to compare toxicities among studies, each LC_{50} value was normalized to the same reference water chemistry (pH = 7.0; hardness = 100 mg/L; DOC = 1 mg/L). Because there are no data for eulachon or green sturgeon, we have used salmonid toxicity data as a surrogate because salmonid fishes were the closest taxonomic group for which data were available. As such, the discussion below, while it is specific to salmonids, is applied to both eulachon and green sturgeon for purposes of this Opinion. Figure 8 illustrates the normalized aluminum toxicity information.

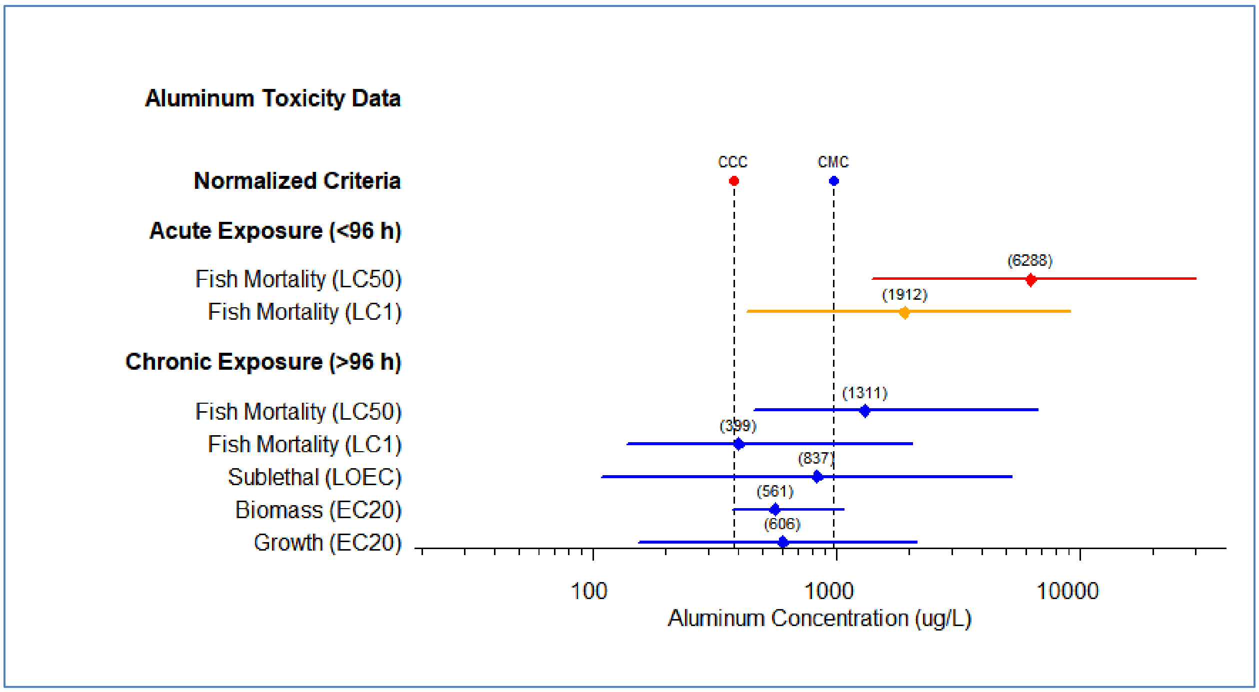

Figure 8. Acute and chronic aluminum criteria concentrations and toxicity data normalized to the following water quality conditions: pH = 7; DOC = 1 mg/L; total hardness = 100 mg/L.
Note: Numbers in parentheses are the geometric means of the data. Sublethal effects include growth and behavioral (e.g., strike frequency) endpoints.

To evaluate the risk of acute toxicity to ESA-listed species, EPA calculated acute taxonomic adjustment factors (TAFs) to estimate LC_5, LC_{10}, and LC_{15} thresholds. Dose-response curves using toxicity data for brook trout (Tandjung 1982) met criteria established by EPA for calculation of a TAF (see Section 5.2.1 of the BE for a more detailed discussion [EPA 2020]). The TAF for the LC_{50}:LC_5, LC_{50}:LC_{10}; and LC_{50}:LC_{15} ratios were 1.967, 1.638, and 1.466, respectively. These TAFs were then used to convert the LC_{50} *Oncorhynchus* GMAV to the following low mortality thresholds: LC_5, LC_{10}, and LC_{15}. Next, EPA normalized these low mortality thresholds to the site-specific chemistry for every sample across the state of Oregon and compared those values to the acute IWQC. EPA (2020) found that the acute IWQCs never exceeded the site-specific LC_5, LC_{10}, or LC_{15} values. A similar process was performed for the sDPS green sturgeon; however, EPA utilized the ICE model to estimate the sturgeon GMAV based on a rainbow trout GMAV. More details about this approach, including nuances regarding use of multiple adjustment factors to scale the LC_{50} down to lower effects thresholds are provided in the BE (EPA 2020). There is substantial uncertainty in the use of the ICE model and because of this uncertainty, NMFS concluded that applying a more conservative salmonid approach was warranted.

While EPA's assessment methodology suggests that the acute criterion is generally protective against acute mortality, the calculated acute criterion is greater than a NOEC reported by Gundersen et al. (1994). If we were to apply the low effect threshold adjustment factor (LC_{50} :LC_5 = 1.967) that EPA estimated to the lower LC_{50} estimates reported by Gunderson et al. (1994), the resultant LC_5 estimate would be less than the CMC, indicating some mortality could occur. Considering this, NMFS concludes there is a risk of individual mortality from exposure to the CMC. We believe this risk applies to Chinook, sockeye, coho, chum, steelhead, eulachon, and sturgeon species, because we have concluded that absent species-specific data, the best surrogate toxicity information is that which is available for salmonids (e.g., rainbow trout, brook trout, cutthroat trout, etc.). To evaluate what the individual risk of mortality could mean for a salmonid population, NMFS conducted population modeling where such population models exist. The population modeling is not applicable to eulachon or green sturgeon.

Population Modeling. Knowing that some mortality to individual fish may occur with exposure to aluminum at acute concentrations, NMFS conducted population modeling in accordance with the Oregon Toxics Biological Opinion (NMFS 2012). The process used here follows the approach used in that Opinion. More details of the model can be found in Appendix 3 of the 2012 Opinion (NMFS 2012).

We reviewed quantified changes in productivity from a population model that addressed impacts on first-year mortality resulting from exposure to the proposed acute criterion concentrations of aluminum. The investigation of population-level responses to chemical exposures uses life-history transition matrix models. Individuals within a population exhibit various growth, reproduction, and survival rates depending on their developmental or life-history stage or age. Changes in these rates can impact a population's intrinsic growth rate (lambda or λ) which is calculated directly from a transition matrix.

Table 14. Summary of acute toxicity data (i.e., test durations of 96-hours or less) for salmonids.

Organism	Life Stage	Test Duration	Endpoint	pH	Total Hardness (mg/L)	DOC (mg/L)	Reported Conc.[a] (µg/L)	Normalized Conc.[b] (µg/L)	Reference
O. mykiss	Juvenile 1-3g	96 h	LC_{50}	7.61	26.35	0.5[c]	>9,840	**>7,216**	Gunderson et al. 1994
O. mykiss	Juvenile 1-3g	96 h	LC_{50}	7.59	45.5	0.5[c]	>8,070	**>5,766**	Gunderson et al. 1994
O. mykiss	Juvenile 1-3g	96 h	LC_{50}	7.6	88.05	0.5[c]	>8,160	**>5,390**	Gunderson et al. 1994
O. mykiss	Juvenile 1-3g	96 h	LC_{50}	7.61	127.6	0.5[c]	>8,200	**>5,164**	Gunderson et al. 1994
O. mykiss	Juvenile 1-3g	96 h	LC_{50}	8.28	23.25	0.5[c]	6,170	**1,685** (1,413-1,922)	Gunderson et al. 1994
O. mykiss	Juvenile 1-3g	96 h	LC_{50}	8.3	35.4	0.5[c]	6,170	**1,680** (1,445-1,959)	Gunderson et al. 1994
O. mykiss	Juvenile 1-3g	96 h	LC_{50}	8.31	83.6	0.5[c]	7,670	**2,180** (1,789-2,764)	Gunderson et al. 1994
O. mykiss	Juvenile 1-3g	96 h	LC_{50}	8.31	128.5	0.5[c]	6,930	**2,026** (NR)	Gunderson et al. 1994
O. mykiss	Juvenile 1-3g	96 h	NOEC	8.25	114.5	0.5[c]	1,680	521	Gunderson et al. 1994
O. mykiss	Alevin	96 h	LC_{50}	5.5	14.3	0.4	310	8,467	Holtze 1983
O. mykiss	Alevin	96 h	LC_{50}	5.0	14.3	0.4	160	10,037	Holtze 1983
O. mykiss	Fingerling 1-2g	96 h	LC_{50}	6.59	47.4	1.1[c]	7,400 (5,800-9,400)	13,495 (10,577-17,142)	Call et al. 1984
O. mykiss	Fingerling 1-2g	96 h	LC_{50}	7.31	47.4	1.1[c]	14,600 (9,300-23,100)	11,879 (7,567-18,795)	Call et al. 1984
O. mykiss	Fingerling 1-2g	96 h	LC_{50}	8.17	47.4	1.1[c]	>24,700	>7,664	Call et al. 1984
O. mykiss	Fingerling 1-2g	96 h	LC_{50}	7.46	47.4	1.1[c]	8,600 (6,200-11,900)	5,915 (4,264-8,184)	Call et al. 1984
O. mykiss	Fingerling 1-2g	24 h	LC_{50}	7.46	44	1.1[c]	13,400 (9,600-18,800)	9,216 (6,602-12,929)	Call et al. 1984
O. mykiss	Fingerling 1-2g	48 h	LC_{50}	7.46	44	1.1[c]	10,500 (7,900-14,000)	7,221 (5,433-9,628)	Call et al. 1984

Organism	Life Stage	Test Duration	Endpoint	pH	Total Hardness (mg/L)	DOC (mg/L)	Reported Conc.[a] (µg/L)	Normalized Conc.[b] (µg/L)	Reference
O. mykiss	Fingerling 1-2g	72 h	LC_{50}	7.46	44	1.1 [c]	9,700 (7,600-12,400)	6,671 (5,227-8528)	Call et al. 1984
S. salar	Alevins	96 h	LC_{50}	6.5	6.8	0.5 [c]	599 (445-772)	**3,599** (2,674-4,639)	Hamilton and Haines 1995
S. salar	Alevins	96 h	LC_{50}	5.5	6.8	0.5 [c]	584 (310-676)	**20,749** (15,775-24,018)	Hamilton and Haines 1995
S. fontinalis	Juvenile	96 h	LC_{50}	5.6	40	1.6	6,530	**30,038**	Tandjung 1982
S. fontinalis	Juvenile	96 h	LC_{50}	5.6	18	1.6	3,400	**24,514**	Tandjung 1982
S. fontinalis	Juvenile	96 h	LC_{50}	5.6	2	1.6	370	**9,187**	Tandjung 1982

Abbreviations: Conc. = concentration; h = hours;

[a]Concentrations reported in the study for the specific pH, total hardness, and DOC concentrations reported in the previous three columns.

[b]Concentrations are normalized to the following chemistry: total hardness of 100 mg/L ($CaCO_3$), pH of 7, and DOC of 1 mg/L. Bold values are used in the SMAV for the criteria calculations. Values in parenthesis are the reported upper and lower 95% confidence interval bounds.

[c]Following EPA's approach, when definitive DOC values were not reported by the authors: a DOC value of 0.5 mg/L was used when dilution water was reconstituted, 1.1 mg/L when dilution water was Lake Superior, MN water, 2.8 mg/L when dilution water was Liberty Lake, WA water, 1.6 mg/L when dilution water was tap or well water, or half the detection limit when the reported value was less than the detection limit, based on recommendations in the 2007 Freshwater Copper Ambient Water Quality Criteria (U.S. EPA 2007b).

Here, the percent change in the intrinsic population growth rate, lambda, resulting from the chemical exposure was compared for the species that would be exposed to acute criteria concentrations during freshwater rearing[6]. Due to differences in the life-history strategies, specifically lifespan, age at reproduction, and first year residence and migration habits, three separate life-history models representing coho, ocean-type Chinook and stream-type Chinook were the used to measure the different potential responses of these species to freshwater chemical exposures and to assess different population-level responses. Steelhead were modeled by the stream-type Chinook parameterization (NMFS 2012, Appendix 3). The conservative model assumption is that all juveniles in the population will be exposed throughout the entire first year to acute criteria concentrations, an extremely unlikely possibility. Exposures are assumed to result in a cumulative reduction in survival as defined by the concentration, and the dose-response curve using the LC_{50} and slope. One model output is the lambda value and the percent change in lambda compared to the unexposed control population's lambda. The percent change in lambda represents changes in population productivity. The acute mortality percent for the first year age class is also calculated across all model runs.

Our input data to the population model were derived from the instantaneous acute water quality criteria, or CMC values calculated using the EPA MLR calculator (EPA 2020). This calculator also provided an *Oncorhynchus* LC_{50} GMAV that was normalized to the site chemistry associated with each IWQC, an input necessary for the population model. For each species known to rear in Oregon water to which the criteria apply, available data for DOC, hardness, and pH were used to calculate acute IWQC, and from this large set of possible criteria concentrations, we calculated the mean and standard deviations. For the population model, four values were then selected from the summary values to represent a wide range of possible exposures, with low values calculated as the mean minus one and two standard deviations, and high values from the mean plus one and two standard deviations.

Due to the criteria concentrations being rounded to two significant figures, many samples had the same calculated IWQC; however, water chemistry may have been different among these samples. In these instances, NMFS selected the site with the most conservative conditions and that could be occupied by rearing juveniles of the ESA-listed species. Because the LC_{50} GMAVs are specific to site chemistry, it is possible that different LC_{50} GMAVs were reported for the same acute IWQC. For example, the IWQC associated with one standard deviation of the mean for LCR Chinook salmon is 2,100 µg/L. The LC_{50} GMAV paired with this IWQC is 11,111. Although the LCR coho salmon shared the same IWQC, the LC_{50} GMAV paired with the coho sample is 8,452. This reflects varying water chemistry during sample collection and the complexity of the proposed criterion.

Table 15 shows the values used, and the resulting output for modeled mortality in year 1. Within the calculated range of acute criteria, model outputs for the percent change in lambda, the intrinsic growth rate, were all zero. This indicates that exposure to the range of acute criteria concentrations would be unlikely to result in population level effects. The year one percent acute mortality output from the model were also quite low, between 0 and 2 percent, and these would be expected to reflect actual effects only when the assumption of population-wide exposure to

[6] Snake River sockeye salmon were not modeled because they primarily use Oregon waters as migration corridors and there is little evidence of extended rearing in areas where the Oregon water quality criteria would apply.

acute criteria for sufficient periods of time to elicit effect was accurate. Given that for most samples in the available water quality data (Table 12; Section 2.4.5.4), the measured values were less than half the acute criteria concentrations, this is very unlikely.

Table 15. Summary of the acute IWQC and acute toxicity data inputs used in the population modeling and the predicted percent mortality of age 1 salmonids for each species.

ESU/DPS	N	Mean CMC (SD)	Mean-2SD/LC_{50}[a]	Mean-1SD/LC_{50}[a]	Mean+1SD/LC_{50}[a]	Mean+2SD/LC_{50}[a]
			% mortality	**% mortality**	**% mortality**	**% mortality**
LCR Steelhead	1,775	1,535 (629)	260/1,186	910/3,316	2,200/7,764	2800/9481
			0%	**1%**	**1%**	**1%**
LCR Chinook	2,286	1,451 (620)	120/591	830/3,744	2,100, 11,111	2,700/15,951
			0%	**0%**	**0%**	**0%**
LCR Coho	2,286	1,451 (620)	120/591	830/3,744	2,100/8,452	2,700/15,951
			0%	**0%**	**1%**	**0%**
UWR Steelhead	4,671	1,548 (630)	280/1296	920/3,879	2,200/7,429	2,800/8,201
			0%	**1%**	**1%**	**2%**
UWR Chinook	4,519	1,309 (472)	370/4,643	840/4,609	1,800/11,607	2,300/8,325
			0%	**0%**	**0%**	**1%**
MCR Steelhead	2,372	2,320 (811)	700/2,868	1,500/6,043	3,100/11,666	3,900/13,188
			1%	**1%**	**1%**	**1%**
SNB Steelhead	932	1,942 (469)	1,000/4,029	1,500/5,669	2,400/8,938	2,900/11,443
			1%	**1%**	**1%**	**1%**
SRS Chinook	932	1,942 (469)	1,000/4,029	1,500/5,669	2,400/8,938	2,900/11,443
			1%	**1%**	**1%**	**1%**
SRF Chinook	369	19,76 (351)	1,300/4,427	1,600/6,115	2,300/10,189	2,700/14,808
			1%	**1%**	**0%**	**0%**
CR Chum	2,187	1,470 (634)	120/591	840/4,104	2,100/11,111	2,700/9,996
			0%	**0%**	**0%**	**1%**
SONCC Coho	1247	1696 (770)	170/893	930/3,889	2,500/10,702	3,300/12,215
			0%	**1%**	**1%**	**1%**
ORC Coho	4378	1314 (576)	150/695	740/2,802	1,900/8,349	2,500/28,281
			0%	**1%**	**0%**	**0%**

Abbreviations: N = number of data points, SD = standard deviation; - = minus; + = plus; % mortality = modeled mortality of age 1 salmonids for each CMC and LC_{50} pair

[a]The statistically-derived CMC values were matched to the nearest actual CMC and the corresponding site chemistry was used to derive the *Oncorhynchus* genus-mean acute LC_{50} value.

Based on the above information, there appears to be little risk of population-level changes as a result of exposure to the proposed CMC.

2.5.4.2 Chronic Toxicity to ESA-listed Species

There were no chronic toxicity studies with *O. mykiss* that met EPA criteria derivation requirements. Only two fish species within the Salmonidae family had acceptable toxicity data for criteria derivation: Atlantic salmon (McKee et al. 1989) and brook trout (Cleveland et al. 1989). Atlantic salmon toxicity data from Buckler et al. (1995) was also deemed acceptable; however, the authors reported information for survival, which was not the most sensitive endpoint. As such, EPA did not include that information in the criteria derivation. If aluminum reduced both the survival and growth of test species in chronic exposures, the product of these test endpoints (biomass) was analyzed (EPA 2018; EPA 2013), rather than analyzing them separately. For criteria derivation purposes, the EPA selected an EC_{20} to estimate a low level of effect for aluminum that would typically be statistically different from control effects, but not severe enough to cause chronic effects at the population level (see U.S. EPA 1999). As previously described, most toxicity studies have been performed in acidic tests waters, and there is relatively little toxicity data for pH ranges that are representative of Oregon waters (i.e., pH of 6 to 8). For many fish, aluminum toxicity increases with early life stages such that eggs and endogenously-feeding alevins are generally less sensitive than exogenous-feeding swim-up larvae (Buckler et al. 1995; Delonay et al. 1993).

Brook trout data from Cleveland et al. (1989) and Atlantic salmon data from McKee et al. (1989), along with fathead minnow and zebrafish data, were used to represent fish taxa in the derivation of chronic criteria. Cleveland et al. (1989) conducted an ELS test, exposing eyed eggs and the resultant larvae and juveniles to various concentrations of aluminum at a pH of approximately 6.5 and 5.5. Exposures continued for 60 days post-hatch and effects were assessed at hatch, 15, 30, 45, and 60 days post-hatch. Mortality after 15 days of exposure ranged from one to four percent (control mortality was one to two percent). Mortality increased as the exposure duration increased; however, significant mortality (~8, 37, and 49 percent for 30, 45, and 60 day exposures, respectively) was not observed until the highest test concentration in both pH test conditions. In these instances, the test concentrations were about two times the test-specific chronic IWQC.

McKee et al. (1989) conducted an ELS tests in acidic water (pH ~ 5.7) using Atlantic salmon. Exposures began before hatch and extended through the alevin stage. By the 60th day of exposure, growth and survival were significantly reduced at the two highest aluminum concentrations. The chronic IWQC is primarily driven by non-fish species in acidic waters; as such, the test-specific IWQC were much lower than the reported NOEC and lowest observed effect concentration (LOEC) values for growth and were roughly one-half the EC_{20} value for the biomass endpoint.

To assess the effects on ESA-listed species, EPA first estimated low effects thresholds (EC_5, EC_{10}, and EC_{15}) for the biomass endpoint in a similar manner as that performed for the acute criterion. Their methodology is described in section 5.2.1.2.2 of the BE (EPA 2020). Ultimately, the EC_{20}:EC5, EC_{20}:EC_{10}, and EC_{20}:EC_{15} TAFs were calculated based on the dose response relationship for one brook trout study Cleveland et al. (1989). The resultant normalized EC_5, EC_{10}, and EC_{15} low effects thresholds used to evaluate the potential effects to ESA-listed salmonids and the sDPS of eulachon from exposure to the CCC were 310.4, 400, and 467.9 µg/L, respectively. For green sturgeon, EPA followed a similar process; however, information from many species (e.g., fathead minnow acute to chronic ratio; fathead minnow and cladoceran adjustment factors) and the ICE model were used to calculate a GMCV and low effects thresholds for the species. More details about this approach, including nuances regarding use of multiple adjustment factors to scale the GMAV down to lower effects thresholds are provided in the BE (EPA 2020).

EPA then calculated the chronic IWQC across all sites in an ecoregion and compared those values to the low effects thresholds that were normalized to sample water chemistry characteristics (see section 5.2.1.2.2 for a more detailed description of the methodology [EPA

2020]). For all fish species, the chronic IWQC were greater than the EC_5 and EC_{10} over 90 and 58 percent of the time, respectively. None of the chronic IWQC exceeded the EC_{15} threshold. Results from EPAs assessment indicate that in most situations, a 5 percent reduction in biomass may occur if individuals are exposed to the CCC for sufficient periods of time.

While EPA's evaluation provides one perspective into the potential effects of the chronic criteria, that methodology was limited to a subset of the available studies that have examined chronic toxicity of aluminum to salmonids. Furthermore, substantial uncertainty exists with respect to the methodology for assessing toxicity risk to green sturgeon (i.e., use of the ICE model and use of toxicity data for invertebrate and vertebrate species). For green sturgeon, NMFS concluded that applying a more conservative approach of treating salmonid toxicity information as a suitable surrogate was warranted.

Although not used in criteria derivation, other studies and/or other toxicity endpoint evaluations provide insight into aluminum toxicity. Tables 16 and 17 summarize available toxicity information for mortality and sublethal effects, respectively. Studies reviewed were limited to those where test pH, DOC, and total hardness was reported or could otherwise be reasonably ascertained. A few notable studies are summarized below. Because there are no data for eulachon or green sturgeon, we have used salmonid toxicity data as a surrogate because salmonid fishes are the closest taxonomic group for which data were available. As such, the discussion below, while it is specific to salmonids, is applied to both eulachon and green sturgeon for purposes of this opinion.

Freeman and Everhart (1971) conducted one of the earliest tests of aluminum toxicity in alkaline waters. They exposed juvenile rainbow trout to four aluminum concentrations in varying pH test waters and recorded changes in mortality, growth, and behavior. Mortality was reported as the time it took to kill (lethal time [LT]) a specific percent of test organisms (LT_x). Death of half the test organisms only occurred in the highest tested aluminum concentration (i.e., 5,200 µg/L), which was about four times the test-specific CCC. The LT_{50} rapidly decreased as pH increased from 7 to 9, although results may be confounded by the use of different aged fish (i.e., 6-month old fish in pH 8, 11-week old fish in pH 7, and 6-week old fish in pH 8.5 and 9.0).

Table 16. Salmonid mortality response data for longer duration exposures (i.e., > 4 days).

Organism	Life Stage	Test Duration	Endpoint	pH	Total Hardness (mg/L)	DOC (mg/L)	Reported Conc.[a] (µg/L)	Test CCC[b] (µg/L)	Norm. Conc.[c] (µg/L)	Reference
O. mykiss	Alevin 23 dph	6 d	LC_{50}	5.8	10.3	2	>1,050	41	>6,738	Hickie et al. 1993
O. mykiss	Juvenile 3.5g	6 d	LC_{50}	5.09-5.31	11.2	1.6[d]	175	9.9	2,837	Orr et al. 1986
O. mykiss	Embryo	7-12 d	LC_{50}	7.4	100	0.5[d]	560	510	460	Birge et al. 2000, *as cited in EPA 2020*
O. mykiss	Yolk sac fry	8 d	NOEC	7.2	14.3	0.4	750	410	1,212	Holtze 1983
O. mykiss	Yolk sac fry	8 d	NOEC	6.5	14.3	0.4	760	170	3,935	Holtze 1983
O. mykiss	Yolk-sac Fry	8 d	>90%	5.5	14.3	0.4	>410	9.2	>11,198	Holtze 1983
O. mykiss	Yolk-sac fry	8 d	NOEC	7.2	14.3	0.4	750	410	1,212	Holtze 1983
O. mykiss	Yolk-sac *suf*	8 d	NOEC	6.5	14.3	0.4	760	170	3,935	Holtze 1983
O. mykiss	Eyed embryo	8 d (12 d rec)	5.8% (56.0%)	7.2	14.3	0.4	330	250	533	Holtze 1983
O. mykiss	Eyed embryo	8 d (12 d rec)	14.5% (52.6%)	7.2	14.3	0.4	750	250	1,212	Holtze 1983
O. mykiss	Eyed embryo	8 d (12 d rec)	4.3% (15.5%)	6.5	14.3	0.4	110	80	570	Holtze 1983
O. mykiss	Eyed embryo	8 d (12 d rec)	10.7% (49.4%)	6.5	14.3	0.4	320	80	1,657	Holtze 1983
O. mykiss	Eyed embryo	8 d (12 d rec)	21.6% (46.4%)	6.5	14.3	0.4	760	80	3,935	Holtze 1983
O. mykiss	Juvenile	10 d	40%	8	25	1.6[d]	50,000	18,009	6,486	Hunter et al. 1980
O. mykiss	Juvenile	10 d	100%	8.5	25	1.6[d]	50,000	10,025	2,010	Hunter et al. 1980
O. mykiss	Juvenile	10 d	100%	9	25	1.6[d]	50,000	5,581	623	Hunter et al. 1980
O. mykiss	Juvenile 1-3g	16 d	LC_{50}	8.14	20.3	0.5[d]	1,940	590	651 (543-798)	Gundersen et al. 1994

Organism	Life Stage	Test Duration	Endpoint	pH	Total Hardness (mg/L)	DOC (mg/L)	Reported Conc.[a] (µg/L)	Test CCC[b] (µg/L)	Norm. Conc.[c] (µg/L)	Reference
O. mykiss	Juvenile 1-3g	16 d	LC_{50}	8.1	103	0.5[d]	8,570	880	1433 (1048-3141)	Gundersen et al. 1994
O. mykiss	Juvenile 6 mo	45 d	NOEC	8.04	46.8	1.6[d]	51.6	**1200**	18	Freeman and Everhart 1971
O. mykiss	Juvenile 6 mo	45 d	2%	8.03	46.8	1.6[d]	514	**1200**	179	Freeman and Everhart 1971
O. mykiss	Juvenile 6 mo	45 d	77%	8.02	46.8	1.6[d]	5200	1200	1839	Freeman and Everhart 1971
O. mykiss	Juvenile 11 wk	45 d	<50%	6.52	46.8	1.6[d]	514	250	843	Freeman and Everhart 1971
O. mykiss	Juvenile 11 wk	45 d	>50%	6.52	46.8	1.6[d]	5135	250	632	Freeman and Everhart 1971
O. aquabonita	Alevin	7 d	NOEC-LOEC	5	4.89	0.5[d]	97-293	3.3	18,359	Delonay 1991; Delonay et al. 1993
O. clarkii	Alevin 2 dph	7 d	NOEC-LOEC	5	42.5	2	50-100	15	482	Woodward et al. 1989
O. mykiss	Embryo / larvae	28 d	EC50	7.4	104	0.5[d]	560	510	457.4	Birge, 1978; Birge, Hudson, Black, and Westerman, 1978
S. salar	Juvenile, 1.4g	5 d	LC_{50}	5.26	10.6	0.5[d]	54	4	2,209	Roy and Campbell 1995
S. salar	Juvenile 1.4g	5 d	LC_{50}	5.24	10.6	0.5[d]	51	3.8	2,170	Roy and Campbell 1995
S. salar	ELS	60 d	LC_{20}	5.7	12.7	1.8[d]	154.2	35	1,088[e]	Buckler et al. 1995; EPA 2018
S. salar	ELS *Newly hatched*	15 d	NOEC	5.7	12.7	1.8[d]	264	35	1,863	McKee et al. 1989
S. salar	ELS *Alevin*	30 d	NOEC (4%)	5.7	12.7	1.8[d]	124	35	875	McKee et al. 1989
S. salar	ELS *Alevin*	30 d	LOEC (8%)	5.7	12.7	1.8[d]	264	35	1,863	McKee et al. 1989

Organism	Life Stage	Test Duration	Endpoint	pH	Total Hardness (mg/L)	DOC (mg/L)	Reported Conc.[a] (µg/L)	Test CCC[b] (µg/L)	Norm. Conc.[c] (µg/L)	Reference
S. salar	ELS *suf*	60 d	NOEC (7%)	5.7	12.7	1.8 [d]	71	35	501	McKee et al. 1989
S. salar	ELS *suf*	60 d	LOEC (12%)	5.7	12.7	1.8 [d]	124	35	875	McKee et al. 1989
S. fontinalis	ELS Eyed egg	15 d	NOEC	7.24	13.4	1.1 [d]	242	**440**	268	Cleveland et al. 1986
S. fontinalis	ELS Alevin	15 dph	NOEC	7.24	13.4	1.1 [d]	242	**440**	268	Cleveland et al. 1986
S. fontinalis	ELS Fry	30 dph	NOEC	7.24	13.4	1.1 [d]	242	**440**	268	Cleveland et al. 1986
S. fontinalis	Juvenile, 37-d	15 d	NOEC	7.35	14.3	1.1 [d]	242	**520**	227	Cleveland et al. 1986
S. fontinalis	Juvenile, 37-d	30 d	NOEC	7.35	14.3	1.1 [d]	242	**520**	227	Cleveland et al. 1986
S. fontinalis	ELS Alevin	15 dph	NOEC	6.6	12.3	1.8	350	190	139	Cleveland et al. 1989
S. fontinalis	ELS Alevin	15 dph	LC_{20}	6.6	12.3	1.8	9,021	190	20,004	Cleveland et al. 1989
S. fontinalis	ELS Fry	30 dph	NOEC	6.6	12.3	1.8	169	**190**	139	Cleveland et al. 1989
S. fontinalis	ELS Fry	30 dph	LOEC (8.5%)	6.6	12.3	1.8	350	190	288	Cleveland et al. 1989
S. fontinalis	ELS Fry	30 dph	LC_{20}	6.6	12.3	1.8	1,405	190	1,248	Cleveland et al. 1989 EPA 2020
S. fontinalis	ELS Fry	45 dph	NOEC	6.6	12.3	1.8	169	**190**	139	Cleveland et al. 1989
S. fontinalis	ELS Fry	45 dph	LOEC (37%)	6.6	12.3	1.8	350	190	288	Cleveland et al. 1989
S. fontinalis	ELS Fry	45 dph	LC_{20}	6.6	12.3	1.8	285.4	190	288	Cleveland et al. 1989 EPA 2020
S. fontinalis	ELS Fry	60 dph	NOEC	6.6	12.3	1.8	169	**190**	139	Cleveland et al. 1989

Organism	Life Stage	Test Duration	Endpoint	pH	Total Hardness (mg/L)	DOC (mg/L)	Reported Conc.[a] (µg/L)	Test CCC [b] (µg/L)	Norm. Conc.[c] (µg/L)	Reference
S. fontinalis	ELS Fry	60 dph	LOEC (48.5%)	6.6	12.3	1.8	350	190	288	Cleveland et al. 1989
S. fontinalis	ELS Fry	60 dph	LC_{20}	6.6	12.3	1.8	305.3	190	677	Cleveland et al. 1989; EPA 2020

Abbreviations: ELS = Early life stage, where exposures began with embryo and continued through specified period of time (test subjects moved from eyed embryo to alevin to swim up fry); *suf* = swim up fry; dph = days post-hatch; rec = recovery; g = grams; mo = month; d = days; h = hours;

[a]Concentrations reported in the study for the specific pH, total hardness, and DOC concentrations reported in the previous three columns.

[b]The chronic IWQC for the test conditions. Bolded values indicate a test CCC that is greater than the reported concentration.

[c]Concentrations are normalized to the following chemistry: total hardness of 100 mg/L ($CaCO_3$), pH of 7, and DOC of 1 mg/L. These may be compared to the normalized CCC, which is 380 µg/L.

[d]Following EPA's approach, when definitive DOC values were not reported by the authors: a DOC value of 0.5 mg/L was used when dilution water was reconstituted, 1.1 mg/L when dilution water was Lake Superior, MN water, 2.8 mg/L when dilution water was Liberty Lake, WA water, 1.6 mg/L when dilution water was tap or well water, or half the detection limit when the reported value was less than the detection limit, based on recommendations in the 2007 Freshwater Copper AWQC (U.S. EPA 2007b).

[e]Buckler et al. (1995) appears to be a republication of McKee et al. (1989), but does not report the most sensitive endpoint and therefore only the most sensitive endpoint from McKee et al. 1989 was used for calculation of the SMCV.

Table 17. Salmonid sublethal response data for longer duration exposures (>7 days).

Organism	Life Stage	Test Duration	Endpoint	pH	Total Hardness (mg/L)	DOC (mg/L)	Reported Conc.[a] (µg/L)	Test CCC[b] (µg/L)	Norm. Conc.[c] (µg/L)	Reference
O. mykiss	Juvenile 1-3g	16 d	Growth Rate NOEC (8% red)	7.98	19.9	0.5 [d]	830	570	355	Gundersen et al. 1994
O. mykiss	Juvenile 1-3g	16 d	Growth Rate NOEC (7%)	7.98	103.5	0.5 [d]	740	**910**	311	Gundersen et al. 1994
O. mykiss	Juvenile 1-3g	16 d	Growth Rate NOEC (28%)	8.05	20.6	0.5 [d]	1,490	580	573	Gundersen et al. 1994
O. mykiss	Juvenile 1-3g	16 d	Growth Rate NOEC (50%)	8.02	102.1	0.5 [d]	1,520	910	611	Gundersen et al. 1994
O. mykiss	Juvenile 1-3g	16 d	Growth Rate (93%*)	8.14	20.3	0.5 [d]	3,200	590	1,073	Gundersen et al. 1994
O. mykiss	Juvenile 1-3g	16 d	Growth (83%*)	8.1	103.4	0.5 [d]	2,750	880	1,008	Gundersen et al. 1994
O. mykiss	Juvenile 1-3g	16 d	Growth (94*)	7.33	15	0.5 [d]	890	340	1,077	Gundersen et al. 1994
O. mykiss	Juvenile 1-3g)	16 d	Growth (50%*)	7.33	84.8	0.5 [d]	940	460	860	Gundersen et al. 1994
O. mykiss	Juvenile 1-3g	16 d	Growth (100%*)	7.35	16.9	0.5 [d]	2,110	350	2,427	Gundersen et al. 1994
O. mykiss	Juvenile 1-3g	16 d	Growth (100%*)	7.33	85.6	0.5 [d]	1,880	460	1,717	Gundersen et al. 1994
O. mykiss	Juvenile 1-3g	16 d	Growth (100%*)	7.34	16.3	0.5 [d]	4,490	350	5,276	Gundersen et al. 1994
O. mykiss	Juvenile 1-3g	16 d	Growth (100%*)	7.32	84.1	0.5 [d]	4,560	450	4,226	Gundersen et al. 1994
O. clarkii	Alevin	7d	Growth LOEC	6.0	42.5	2.0	43	**160**	109	Woodward et al. 1989
O. clarkii	S.U.F.	7d	Growth LOEC	6.0	42.5	2.0	43	**160**	109	Woodward et al. 1989

Organism	Life Stage	Test Duration	Endpoint	pH	Total Hardness (mg/L)	DOC (mg/L)	Reported Conc.[a] (µg/L)	Test CCC [b] (µg/L)	Norm. Conc.[c] (µg/L)	Reference
O. clarkii	Alevin	7d	Behavior	6.0	42.5	2.0	43	**160**	109	Woodward et al. 1989
O. clarkii	Alevin / larvae	7 d	Growth NOEC-LOEC	5	42.5	2	50- >50	15	340.8	Woodward et al. 1989
S. salar	ELS	60 d	Biomass EC_{20}	5.7	12.7	1.8 [d]	61.56	35	434.4	McKee et al. 1989; EPA 2018
S. salar	ELS *Newly hatched*	15 d	Growth NOEC	5.7	12.7	1.8 [d]	264	35	1,863	McKee et al. 1989
S. salar	ELS *Alevin*	30 d	Growth NOEC	5.7	12.7	1.8 [d]	264	35	1,863	McKee et al. 1989
S. salar	ELS *suf*	60 d	Growth LOEC	5.7	12.7	1.8 [d]	124	35	875	McKee et al. 1989
S. salar	ELS	60 d	Biomass EC_{20}	5.7	12.7	1.8 [d]	61.6	35	434.4	McKee et al. 1989 EPA 2020
S. fontinalis	ELS 15 dph		Growth NOEC	7.24	13.4	1.1 [d]	242	**440**	155	Cleveland et al. 1986
S. fontinalis	ELS 30 dph		Growth NOEC	7.24	13.4	1.1 [d]	242	**440**	155	Cleveland et al. 1986
S, fontinalis	Juvenile, 37-d	15 d	Growth NOEC	7.35	14.3	1.1 [d]	242	**520**	142	Cleveland et al. 1986
S. fontinalis	Juvenile, 37-d	30 d	Growth LOEC	7.35	14.3	1.1 [d]	242	**520**	142	Cleveland et al. 1986
S. fontinalis	Juvenile, 37-d	30 d	Growth LOEC	7.35	14.3	1.1 [d]	242	**520**	142	Cleveland et al. 1986
S. fontinalis	ELS Alevin	15 dph	Growth NOEC	6.6	12.3	1.8	350	190	776	Cleveland et al. 1989
S. fontinalis	ELS Fry	30 dph	Growth NOEC	6.5	12.3	1.8	57	**170**	144	Cleveland et al. 1989
S. fontinalis	ELS Fry	30 dph	Growth LOEC	6.5	12.3	1.8	88	**170**	222	Cleveland et al. 1989
S. fontinalis	ELS Fry	30 dph	Growth EC_{20}	6.6	12.3	1.8	69.4	**190**	154	Cleveland et al. 1989 (EPA 2020)

Organism	Life Stage	Test Duration	Endpoint	pH	Total Hardness (mg/L)	DOC (mg/L)	Reported Conc.[a] (µg/L)	Test CCC[b] (µg/L)	Norm. Conc.[c] (µg/L)	Reference
S. fontinalis	ELS Fry	45 dph	Growth NOEC	6.5	12.3	1.8	88	**170**	222	Cleveland et al. 1989
S. fontinalis	ELS Fry	45 dph	Growth LOEC	6.6	12.3	1.8	169	**190**	375	Cleveland et al. 1989
S. fontinalis	ELS Fry	45 dph	Growth EC20	6.6	12.3	1.8	371.7	190	824	Cleveland et al. 1989 (EPA 2020)
S. fontinalis	ELS Fry	60 dph	Growth NOEC	6.5	12.3	1.8	88	**170**	222	Cleveland et al. 1989
S. fontinalis	ELS Fry	60 dph	Growth LOEC	6.6	12.3	1.8	169	**190**	375	Cleveland et al. 1989
S. fontinalis	ELS Fry	60 dph	Growth EC_{20}	6.6	12.3	1.8	134.4	**190**	298	Cleveland et al. 1989 (EPA 2020)
S. fontinalis	ELS		Biomass EC_{20}	6.5	12.3	1.8	164.4	**170**	379	Cleveland et al. 1989; EPA 2018
S. fontinalis	ELS		Biomass EC_{20}	5.6	12.8	1.9	143.5	28	1,076	Cleveland et al. 1989; EPA 2018
S. fontinalis	ELS Fry	30 dph	Strike frequency LOEC	6.5	12.3	1.8	57	**170**	144	Cleveland et al. 1989
S. fontinalis	ELS Fry	30 dph	Locomotion / Position NOEC	6.6	12.3	1.8	169	**190**	375	Cleveland et al. 1989
S. fontinalis	ELS Fry	30 dph	Locomotion / Position LOEC	6.6	12.3	1.8	350	170	776	Cleveland et al. 1989
S. fontinalis	ELS Fry	30 dph	Swim performance NOEC	6.5	12.3	1.8	88	**170**	222	Cleveland et al. 1989
S. fontinalis	ELS Fry	30 dph	Swim performance LOEC	6.6	12.3	1.8	169	**170**	375	Cleveland et al. 1989

Abbreviations: ELS = Early life stage, where exposures began with embryo and continued through specified period of time (test subjects moved from eyed embryo to alevin to swimup fry); S.U.F. = swim up fry; dph = days post-hatch; rec = recovery; d = days; h = hours; * = statistically significantly different from controls

[a]Concentrations reported in the study for the specific pH, total hardness, and DOC concentrations reported in the previous three columns.

[b]The chronic IWQC for the test conditions. Bolded values indicate a test CCC that is greater than the reported concentration.

[c]Concentrations are normalized to the following chemistry: total hardness of 100 mg/L ($CaCO_3$), pH of 7, and DOC of 1 mg/L. These may be compared to the normalized CCC, which is 380 μg/L.

[d]Following EPA's approach, when definitive DOC values were not reported by the authors: a DOC value of 0.5 mg/L was used when dilution water was reconstituted, 1.1 mg/L when dilution water was Lake Superior, MN water, 2.8 mg/L when dilution water was Liberty Lake, WA water, 1.6 mg/L when dilution water was tap or well water, or half the detection limit when the reported value was less than the detection limit, based on recommendations in the 2007 Freshwater Copper AWQC (U.S. EPA 2007b).

In a separate test, the authors exposed six month old trout exposed to total aluminum at concentrations roughly half the test-specific CCC. Test organisms exhibited reduced feeding after six days of exposure and by day 30, these fish fed little (Freeman and Everhart 1971). After ten days of exposure, their coloration was darker and there was a noticeable decrease in their fright response to humans. Gill hyperplasia was evident in about half of the test organisms by the 21st day of exposure. By the last day of exposure, nearly all of the fish showed gill hyperplasia and the fish weighed about 58 percent of the controls, on average. After the 45-day exposure period, fish were allowed to recover in clean water for 18 days. During this recovery period, all symptoms of the exposure, except gill hyperplasia, disappeared within two days. Fish exhibited extremely rapid weight gain during the 16-day recovery period, although they didn't catch up to the controls. The authors also exposed fish to 51.6 µg/L total aluminum, which was about 20 times less than the test-specific chronic CCC. No changes in mortality, growth, or behavior was observed at this concentration (Freeman and Everhart 1971).

For their near-neutral pH tests, Freeman and Everhart (1971) exposed 11-week old trout to total aluminum concentrations (i.e., 514 µg/L) that were roughly twice the test-specific CCC. By the seventh day of exposure, fish coloration darkened and feeding activity was reduced. The first mortality occurred on day 23 and occurred regularly thereafter, but the exposure duration was not long enough for 50 percent of the organisms to die. At the conclusion of the test, fish fed erratically and average weight was 33% of the control group. After 74 days of recovery, average weight was 43% of the control group.

Holtze (1983) examined aluminum toxicity to different early life stages (i.e., cleavage embryo, eyed embryo, yolk sac fry, and swim up fry) of rainbow trout at different pH levels. Each life stage was exposed to elevated aluminum concentrations for 8 days, followed by a 12-day recovery period in which latent mortality was recorded. Survival and development of the cleavage egg, yolk sac fry, and swim up fry were unaffected by aluminum concentrations as high as 750 µg/L in test waters with a pH of 6.5 and 7.2. Eyed eggs were moderately sensitive in these near-neutral pH conditions, with a reported initial mortality of 4 to 22 percent and delayed mortality of 15 to 56 percent. Test concentrations were anywhere from 1.5 to 10 times greater than their corresponding chronic IWQC.

Cleveland et al. (1986) conducted an ELS test, where eyed eggs and their resulting larvae were exposed to aluminum concentrations of 242 µg/L for 30 days post hatch. In addition, the authors exposed 37-day old brook trout to the same aluminum concentrations for 30 days. In these tests, the exposure concentration was about 2.5 times greater than test-specific CCC. Mortality, growth, behavior, and biochemical responses were monitored. Tests were conducted in acidic (pH ~ 4.5 and 5.5) and near neutral (pH ~ 7.2) conditions. Only the near-neutral results are summarized here. Mortality of both ELS and 37-day old brook trout did not differ from that of the controls. Growth of the ELS organisms did not different from controls; however, growth of the 37-day old fish was significantly lower (five percent reduction in mean length) after 30 days of exposure to total aluminum. The strike frequency of ELS and 37-day old fish exposed to aluminum was significantly less (about 34 and 63 percent, respectively) than the controls.

Cleveland et al. (1989) reported that fish in treatment groups exposed to greater than or equal to 88 µg/L aluminum for 30 or more days were significantly smaller than the controls (Table 18).

On the contrary, fish exposed for 15 days were no different from the controls, and in some cases, were larger than the controls. The authors also examined behavioral responses such as feeding (quantified as the number of strikes at prey within a 5-minute period), locomotion (number of times a fish changed position during a 2-minute interval), swimming capacity (measured in a stamina tunnel), and buoyancy (position in the water column) at 30 and 60 days of exposure. The authors found that feeding behavior was the most sensitive endpoint, with test organisms showing significantly fewer strikes at prey after 30 days of exposure to aluminum concentrations of 57 μg/L. This concentration is roughly one-third of the test-specific chronic IWQC.

Gundersen et al. (1994) exposed juvenile rainbow trout to various concentrations of aluminum in neutral (pH ~ 7) or alkaline (pH ~ 8.5) test waters. The authors found that mortality was not concentration-dependent and was not observed in near-neutral pH until day 11; however, mortality was concentration-dependent and observed much sooner in the alkaline pH test. Using a generic dose-response curve with an assumed probit regression slope of 4.5, we calculated low-effects thresholds (LC_1) for each of the two reported LC_{50} values. One of the LC_1 values was equivalent to the CCC, and the other LC_1 value was approximately three times greater than the LC_1 estimate. These results suggest chronic criteria are protective against mortality (16 day exposures). In contrast, growth and food consumption was a more sensitive endpoint and were most affected by aluminum concentrations in the near-neutral test waters. Growth rates were lower in fish exposed to aluminum in near-neutral pH for 16 days relative to fish exposed to similar aluminum concentrations in weakly alkaline pH.

Summary. In summary, none of the test-specific chronic IWQC exceeded concentrations where mortality was observed to be statistically different from controls. Although not statistically significant, Freeman and Everhart (1971) reported low levels of mortality resulting from 45-day exposures to aluminum concentrations less than the test-specific CCC. The preponderance of data suggests that mortality, although it could occur, is not likely to be prevalent and is not likely to be very high. Sublethal effects, on the other hand, are more likely to result from exposure to the CCC. While some authors reported no reductions in growth at aluminum concentrations equal to or greater than the chronic IWQS, some authors did report reductions in growth at concentrations near, or less than, the chronic IWQS. Behavioral endpoints such as strike frequency and swimming performance were also reduced at concentrations less than the chronic IWQC. Overall, the criteria appear to be sufficiently protective under acidic conditions, when other organisms that are more sensitive than fish species are driving the SSD and resultant CCC. There are very few studies examining toxicity to rainbow trout in near-neutral and alkaline conditions. Oftentimes, the sublethal endpoints marked substantial changes; however, they were reported as NOECs.

2.5.4.3 Toxicity to Prey Items

An important consequence of toxic substances to ESA-listed species is the potential reduction of their prey base. For many substances, invertebrates tend to be among the most sensitive taxonomic groups and because juvenile salmonids depend on aquatic invertebrates during freshwater rearing, the potential loss of invertebrates due to contaminant exposure is an important indirect effect that must be assessed. General considerations and assumptions applicable to the evaluation of toxic impacts on forage base are described below.

In instances of a pulse of chemical disturbance such as insecticide spraying of forests or crops, effects to aquatic invertebrate communities ranging from increased drift to catastrophic reductions in abundance can result (Ide 1957; Gibson and Chapman 1972; Wallace and Hynes 1975; Wallace *et al*. 1986). In such cases, even if the fish are not directly harmed by the chemical, the temporary reduction in food from the reduction in invertebrate prey can lead to reduced growth, and reduced growth in juvenile salmonids can in turn be extrapolated to reduced survival and increased risk of population extinction (Kingsbury and Kreutzweiser 1987; Davies and Cooke 1993; Baldwin *et al*. 2009; Mebane and Arthaud 2010). However, such severe effects would not be expected in waters with chemical concentrations similar to the maximum allowed by aquatic life criteria. The criteria are intended to only allow adverse effects to a small minority of the species in aquatic communities.

This begs the question of whether the loss of a minority of invertebrate prey species could lead to a reduction in forage for juvenile salmonids that in turn could affect growth and survival. To address that question, NMFS reviewed a large number of studies on food habits of salmonids in streams, lakes, and reservoirs. The body of evidence indicates that juvenile salmonids are opportunistic predators on invertebrates, and so long as suitable, invertebrate prey items are abundant and diverse, the loss of a few "menu items" probably would not result in obvious, adverse effects. Suitable invertebrate prey items for juvenile salmonids are those that are small enough to be readily captured and swallowed, and vulnerable to capture (i.e., not taxa that are burrowers or are armored (Keeley and Grant 2001; Suttle *et al*. 2004; Quinn 2005). Some otherwise apparently suitable taxa such as water mites (Hydracarina) appear to taste bad to salmonids and others, like copepods, are too small to provide much energy for the effort it takes to eat them (Keeley and Grant 1997).

Freshwater aquatic invertebrates have such great diversity, that they have some ecological overlap and redundancy, so that the loss of a few species would be unlikely to disrupt the stream or lake ecology greatly (Covich *et al*. 1999). However, this apparent ecological redundancy is compromised in streams that have already lost substantial diversity to pollution. For instance, in copper-polluted Panther Creek, Idaho, during springtime in the early 1990s, the total count of invertebrates was just as abundant as in reference sites, although the abundance was composed of fewer species. Yet in October, the abundance in the polluted reaches was less than 10% of reference (Mebane 1994). With reduced diversity, after a single species hatches and leaves the streams, a large drop in remaining abundance can occur. Because all species do not hatch at the same time, with greater diversity, the swings in abundance would be less severe. Further, in copper-polluted tributaries to Panther Creek, the usually abundant mayflies were scarce and had been replaced by unpalatable mites and low-calorie copepods (Todd 2008).

One consistent theme in the literature on the feeding of salmonids in streams is the persistent importance of mayflies and chironomid midges (Chapman and Quistorff 1938; Chapman and Bjornn 1969; Sagar and Glova 1987, 1988; Mullan et al. 1992; Clements and Rees 1997; Rader 1997; White and Harvey 2007; Iwasaki et al. 2009; Syrjänen et al. 2011). In lakes zooplankton are disproportionally important, and as stream size increases and gradients drop, amphipods become popular food items with migrating and rearing juvenile salmon and steelhead (Tippets and Moyle 1978; Rondorf et al. 1990; Muir and Coley 1996; Budy et al. 1998; Karchesky and Bennett 1999; Steinhart and Wurtsbaugh 2003; Teuscher 2004). However, salmonids are opportunistic and will shift their feeding to whatever is abundant, accessible, and palatable, and have sometimes have been reported with their stomachs full of unexpected prey such as snails or hornets (Jenkins et al. 1970; NCASI 1989; Mullan et al. 1992).

In general, the body of the evidence suggests that there is some ecological redundancy among aquatic stream and lake invertebrates, and if a small minority of invertebrate taxa were eliminated by chemicals at criteria concentrations, but overall remain diverse and abundant, then aquatic invertebrate overall community structure and functions, and forage value of critical habitats would likely persist. However, case-by-case consideration of the data is required because the previous assumption is tempered by the fact that aquatic insects are typically underrepresented in criteria datasets and toxicity testing in general (Mebane 2010; Brix et al. 2011).

To evaluate the potential for indirect effects to ESA-listed species, EPA (2020) considered potential for reductions in species' forage base due to aluminum exposures at the chronic criterion. For each ESA-listed species (excluding SRKW, which are discussed in Section 2.5.9), a chronic aluminum SSD was created that only contained prey items, or reasonable surrogate prey items, for the species of concern.

Overall Prey Toxicity Assessment Methodology. To create a prey-based SSD, EPA removed non-prey items from the 2018 aluminum chronic toxicity sensitivity distribution and calculated a 5^{th} percentile hazard concentration (HC_5) value that was specific to: (1) A particular set of prey items specific to an ESA-listed species; and (2) the water chemistry for each sample where pH, DOC, and hardness were concurrently collected. The HC_5 value represents a 20% chronic effect to the 5^{th} centile of sensitive genera under long-term exposure scenarios. Therefore, the GMCVs for prey items of each species were all renormalized for each water chemistry scenario and ranked based on relative genera sensitivity to the specific water chemistry. This created a unique species-prey-item-specific SSD for all 19,274 water chemistries representative of Oregon waters. An HC_5 was then calculated for each unique listed-species-prey-prey-item-specific SSD following US EPA (1985).

Each individual HC_5 value was then compared to corresponding chronic IWQC to determine the number and percentage of time prey-item-specific HC_5 values were less than criteria. For many of the ESA-listed species assessed, prey-item-specific HC_5 values were less than corresponding criterion concentrations in a subset of Oregon water chemistries. Although the underlying data used for this prey vulnerability assessment was the same used for criteria derivation, the smaller number of species used to calculate the HC_5 reduced the HC_5 value relative to the CCC which is based on a greater number of species. Results of the prey-item-specific HC_5 comparisons to corresponding criterion magnitudes were influenced primarily by the number of species in the SSD, rather than prey item sensitivity itself. Therefore, the most sensitive GMCV from each prey-item SSD was also compared to the corresponding chronic IWQC. Comparing the most sensitive prey-item GMCV to each corresponding criterion was intended to provide an alternative measure of prey-item sensitivity that was not affected by the reduction in the number of species included.

EPA focused their assessment on the chronic criterion because most prey items for ESA-listed species were r-selected species with populations that are capable of recovering from short-term acute exposures. Assessing chronic exposures provided environmentally relevant conclusions, which better reflect exposures that could have long-term impacts to prey item availability.

Aluminum toxicity data are limited and the available data do not represent all potential prey items. At best, the species-specific prey-item SSD is adequately representative of the toxicity of aluminum to a species' set of prey items. Although NMFS recognizes the limitations associated with a small dataset, we believe that EPA's assessment strategy for salmonids, eulachon, and green sturgeon appropriately evaluates the potential for indirect impacts to these species due to prey reductions. This is because ESA-listed fish are opportunistic feeders and because the loss of a minority of taxa might not be a severe indirect effect if other prey were still diverse and abundant.

Anadromous Salmonids. Table 18 identifies the lists of prey species that EPA selected from when developing a species-specific, prey-item SSD. Foraging preferences of salmonids are similar across the species, and were broadly described above.

As described above, EPA compared site- and species-specific prey HC_5 values to the chronic IWQC. Comparisons were made for those samples within Level III ecoregions that overlapped the range of coho (n = 15,411), Chinook (n = 16,865), chum (n = 14,735), sockeye (n = 16,865), and steelhead (n = 16,865). The HC_5 values were generally (at least 78% of the time) below the chronic IWQC in samples collected within their range. In every instance, the most sensitive prey-item's GMCV was greater than the CCC.

Considering the most sensitive prey item GMCV was always greater than the criterion, and considering that salmonids are opportunistic feeders, prey resources are not expected to be diminished to a point which will negatively affect foraging behaviors or foraging success of salmonids.

Table 18. Salmonid prey items (or their surrogates) and GMCVs in rank order for reference water chemistry. Not all prey items are included in the SSD for each salmonid ESU/DPS.

Rank	Species	GMCV (µg/L)[a]
12	Oligochaete, *Aeolosoma sp.*	20,514
11	Midge, *Chironomus riparius*	5,099
10	Rotifer, *Brachionus calyciflorus*	3,539
9	Great pond snail, *Lymnaea stagnalis*	3,119
8	Fathead minnow, *Pimephales promelas*	2,407
7	Amphipod, *Hyalella azteca*	1,387
6	Zebrafish, *Danio rerio*	1,342
5	Cladoceran, *Ceriodaphnia dubia*	1,181
4	Fatmucket, *Lampsilis siliquoidea*	1,026
3	Cladoceran, *Daphnia magna*	985.3

Rank	Species	GMCV (µg/L)[a]
2	Brook trout, *Salvelinus fontinalis*	638.2
1	Atlantic salmon, *Salmo salar*	434.4

[a]The GMCVs are calculated for reference water conditions (pH = 7, total hardness = 100 mg/L, and DOC = 1 mg/L).

sDPS Green sturgeon. Similar to salmonids, green sturgeon are opportunistic predators and will consume a variety of available prey types. Burrowing shrimp species (e.g., *Neotrypaea* spp.) are an important dietary component for subadult and adult green sturgeon, but green sturgeon also eat fish (e.g., lingcod), crab (e.g., *Cancer* spp.), amphipods (e.g., *Anisogammarus* spp.), clams (e.g., *Cryptomya californica*), shrimp, and polychaetes (Dumbauld, Holden, and Langness 2008b; NMFS 2018). Similarly, juvenile green sturgeon feed upon shrimp, amphipods, isopods, clams, annelid worms, and an assortment of crabs and fish in the San Francisco Bay Delta Estuary (Ganssle 1966; NMFS 2018; Radtke 1966).

Table 19 is the EPA (2018) aluminum chronic criterion dataset modified to reflect species or surrogates that represent possible larval and juvenile green sturgeon prey items in freshwater where the criteria apply. The GMCVs (based on EC_{20} values) are at the reference water conditions (pH 7, total hardness=100 mg/L as $CaCO_3$ and DOC=1 mg/L).

As described above, EPA compared site-specific green sturgeon prey HC_5 values to the chronic IWQC. For green sturgeon, this analysis was limited to data collected within level-III ecoregions overlapping the green sturgeon sDPS (n = 4,768). The chronic aluminum CCC exceeded the green sturgeon prey-item HC_5 in 10.91% of water chemistry scenarios. The most sensitive prey-item's GMCV was greater than the CCC in 100% of water chemistries evaluated. Considering the most sensitive prey item GMCV was always greater than the criterion, green sturgeon prey resources are not expected to be diminished to a point which will negatively affect foraging behaviors or foraging success of green sturgeon.

Table 19. Green sturgeon prey items (or surrogates) and their GMCVs, in rank order for reference water chemistry.

Rank	Species	GMCV (µg/L)[a]
9	Oligochaete, *Aeolosoma sp.*	20,514
8	Midge, *Chironomus riparius*	5,099
7	Great pond snail, *Lymnaea stagnalis*	3,119
6	Fathead minnow, *Pimephales promelas*	2,407
5	Amphipod, *Hyalella azteca*	1,387
4	Zebrafish, *Danio rerio*	1,342
3	Cladoceran, *Ceriodaphnia dubia*	1,181
2	Fatmucket, *Lampsilis siliquoidea*	1,026
1	Cladoceran, *Daphnia magna*	985.3

[a]The GMCVs are calculated for reference water conditions (pH = 7, total hardness = 100 mg/L, and DOC = 1 mg/L).

Eulachon. Eulachon dietary information is limited, particularly for juveniles. River currents purportedly carry newly hatched young to the sea where they feed mainly on copepod larvae and other plankton (Willson et al. 2006). Adults are primarily plankton-feeders. However, during the fall season, studies have shown that their stomachs are not very full suggesting they do not actively feed during that time. Larval stages of Pacific eulachon eat phytoplankton, copepods, copepod eggs, mysids, barnacle larvae, worm larvae, and eulachon larvae (WDFW and ODFW 2001, as reported in Willson et al. 2006).

Table 20 is the EPA (2018) aluminum chronic criterion dataset modified to reflect species or surrogates that represent possible eulachon prey items in freshwater where the criteria apply. The GMCVs (based on EC_{20} values) are at the reference water conditions (pH 7, total hardness=100 mg/L as $CaCO_3$ and DOC=1 mg/L).

As described above, EPA compared site-specific eulachon prey HC_5 values to the chronic IWQC. This comparison was limited to data collected within level-III ecoregions overlapping the eulachon range (n = 13,973). The chronic aluminum CCC exceeded the eulachon prey-item HC_5 in 77% of water chemistry scenarios. The most sensitive prey-item's GMCV was greater than the CCC in 100% of water chemistries evaluated. Considering the most sensitive prey item GMCV was always greater than the criterion, eulachon prey resources are not expected to be diminished to a point which will negatively affect foraging behaviors or foraging success of eulachon.

Table 20. Eulachon prey items (or surrogates) and their GMCVs, in rank order for reference water chemistry.

Rank	Species	GMCV (µg/L)[a]
6	Oligochaete, *Aeolosoma sp.*	20,514
5	Midge, *Chironomus riparius*	5,099
4	Rotifer, *Brachionus calyciflorus*	3,539
3	Amphipod, *Hyalella azteca*	1,387
2	Cladoceran, *Ceriodaphnia dubia*	1,181
1	Cladoceran, *Daphnia magna*	985.3

[a]The GMCVs are calculated for reference water conditions (pH = 7, total hardness = 100 mg/L, and DOC = 1 mg/L).

2.5.4.4 Bioaccumulation

Aquatic organisms can accumulate metals from both aqueous and dietary exposure routes. The relative importance of each, however, is dependent upon the chemical. Aluminum adsorbs rapidly to gill surface from the surrounding water, but cellular uptake from the water is slow, with gradual accumulation by the internal organs over time (Dussault et al. 2001). Biomagnification of aluminum through trophic levels in aquatic food chains is not usually observed (Suedel et al. 1994; U.S. EPA 2007a; Herrmann and Frick 1995; Otto and Svensson 1983; Wren and Stephenson 1991). King et al. (1992) suggested the opposite phenomena (i.e., trophic dilution up the food chain) based on the lowest aluminum accumulation exhibited by fish predators (perch) and highest by the phytoplankton that their zooplankton prey were consuming.

Total uptake generally depends on the environmental aluminum concentration, exposure route and the duration of exposure (McGeer et al. 2003). Desouky et al. (2002) reported that the bioavailability of aluminum to a grazing invertebrate is influenced by both oligomeric silica and humic acid, and that aluminum bound to humic acid may still be bioavailable via grazing. Overall, bioaccumulation and toxicity via the diet are considered unlikely relative to direct waterborne aluminum toxicity (Handy 1993; Poston 1991).

Cleveland et al. (1989) found that as brook trout advanced from larvae to juveniles they either decreased their aluminum uptake or were able to excrete it quicker. This was evidenced in a 60-day exposure study where an initial increase in aluminum concentrations occurred during the first 15-30 days of exposure, followed by a decline in whole-body concentrations during the final 30 days of exposure. Cleveland et al. (1991a) exposed 30-day old brook trout to 200 µg/L of aluminum in test waters at three pH levels (5.3, 6.1, and 7.2) for 56 days. After 56 days, trout were transferred to water of the same pH with no aluminum amendments and held for 28 days. Fish were sampled for whole body aluminum on days 3, 7, 14, 28 and 56 of the exposure; and on day 3, 7, 14 and 28 of the depuration period. Bioconcentration factors (BCF) were inversely related to pH: 142 at pH 5.3, 104 at pH 6.1, and 14.2 at pH 7.2. Mortality was also highest at pH 5.3 and lowest at pH 7.2. In a separate study, Buckler et al. (1995) continuously exposed Atlantic salmon beginning as eyed eggs to four aluminum treatment levels (33, 71, 124, 264 µg/L) at pH 5.5 for 60 days after the median hatch date. Fish were sampled for whole body aluminum after 15, 30, 45, and 60 days post median hatch date. After 60 days, average mortality was 15% in the 124 µg/L treatment and 63% in the 264 µg/L treatment. The mortality NOEC and LOEC were 71 and 124 µg/L, respectively. BCFs were directly related to exposure concentration, and were 76, 154, and 190 at treatment levels 33, 71, and 124 µg/L, respectively. A BCF could not be calculated for the 264 µg/L treatment level because there were insufficient surviving fish to analyze. These BCFs are quite low (compared to BCFs for mercury or organic pollutants that are in the thousands to tens of thousands) suggesting bioaccumulation is of little concern.

Bioaccumulation and toxicity via the diet are considered highly unlikely based on the literature, 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. The low aluminum BCFs reported in the literature are supported by the slow waterborne uptake and the lack of dietary accumulation.

2.5.4.5 Mixture Toxicity

The toxicity analysis conducted by EPA (2020) and by NMFS in this opinion, evaluates aluminum as if it is the only chemical present in the aquatic environment. Despite this simplification, chemicals in water never occur in isolation, but rather always occur as mixtures, a concept recognized by the EPA (2020). 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 are beyond the present state of the ecotoxicology practice. Here, despite the complexities and many exceptions, we make a general assumption that, at their criteria concentrations, the effects of chemicals in mixtures would likely be more severe than would be the same concentration of the mixture components singly.

The EPA (2020) identified ODEQ's practice of requiring whole effluent toxicity (WET) testing in certain circumstances. WET testing accounts for the mixture toxicity of the effluent in a way that is relevant to the discharge and location. EPA (2020) included information on two facilities with NPDES permits that included aluminum and WET testing requirements (EPA 2020, Appendix E). Addressing mixture toxicity through the use of WET testing and instream bioassessment are practical and reasonable approaches for addressing the expected increased toxicity of a given concentration of a chemical in the presence of other chemicals. However, the assessment triggers on WET tests may not be sensitive enough to protect ESA-listed salmonids with reasonable certainty, and biomonitoring has not always been well defined (NMFS 2014b).

2.5.4.6 Summary of Toxicity to ESA-listed Species

The preponderance of data suggests that mortality of individual fish resulting from exposure to aluminum at concentrations equivalent to the acute criteria is possible. Mortality resulting from exposure to the chronic criteria is not likely to be prevalent and is not likely to be very high. Sublethal effects are likely to occur if individuals are exposed to the chronic criterion for sufficient periods of time. Sublethal effects included reduced growth, biomass, feeding, and swimming capabilities. Based on the information above, effects to ESA-listed fish species from biaccumulation or reduced forage is not expected to lead to any adverse effects.

Overall, because there is some risk of mortality and some risk of sublethal effects from exposures to the proposed criteria, we consider there to be a high risk of toxicity to individual fish. Scaling this up to the population- and species-level requires some qualitative judgements about the data. While the level of mortality that could occur across any given population is expected to be low (and population modeling suggests no changes in lambda), reduced growth is a potential sublethal effect that could subsequently lead to lower rates of survival for individual fish that could manifest into reductions in population abundance and productivity. However, we are not able to integrate these potential effects into the population model. In light of this uncertainty, the integrated risk of mortality that could cause population-level changes that could subsequently lead to lower species viability is considered to be ***moderate*** for most species considered in this Opinion. The only exception is that for SR sockeye salmon. Because these fish are believed to inhabit the Columbia River and LCRE estuary for relatively short periods of time, this species is assessed to have a ***low*** risk of population-level effects stemming from aluminum exposures during their migration.

2.5.5 Risk of Exposure to Aluminum

While the toxicity assessment assumes that individual fish have the potential to be exposed to aluminum at criteria concentrations, assuming this exposure occurs 100% of the time throughout a species range is extremely conservative in the case of aluminum. Although aluminum is ubiquitous in the environment, concentrations are typically far below the acute and chronic IWQC. As such, we characterized the risk of species to experience exposures to aluminum at concentrations at or near the IWQC. Exposures to elevated levels of aluminum could occur in: (1) Areas mineralized with aluminum ores; (2) areas that are urbanized where aluminum may be present in stormwater runoff; (3) areas of agricultural use where aluminum in soils can be mobilized to nearby streams; and (4) areas receiving discharge from a point source with aluminum in its effluent. To examine the risk of exposure, we considered land cover, point source discharges, life stage, and total aluminum concentrations within each species' range in Oregon. For purposes of this Opinion and based on findings of Freeman and Everhart (1971), we have assumed that fish spending portions of their life upstream of the action area have recovered from any aluminum exposure experienced in areas outside of the action area. These attributes are summarized in Table 21 and described in the following sections.

Table 21. Attributes considered in the risk of exposure evaluation.

Attribute	Description	Risk Categories		
		High	Medium	Low
Land Cover	Estimate of the percent of developed and agricultural land within the ESU/DPS boundary.	>25%	10%-25%	<10%
Point Source Discharges	Estimate of the number of currently authorized point source discharges with the potential for aluminum to be in the effluent.	>20	10-20	<10
Life Cycle Stage	The potential for experiencing toxic effects depends, in part, upon the life stage exposed and the duration of exposure. We considered information about whether species had the potential to spawn, rear, and/or migrate in Oregon waters as well as where exposures may occur and for how long those exposures could last.	See discussion	See discussion	Migration only
Total Aluminum Concentrations	Calculated the percent of samples where total aluminum concentrations were less than the IWQC and percent of samples that were ≤½ of the IWQC.	<25%	25%-75%	>75%

2.5.5.1 Land Cover

Land cover data was obtained from the National Land Cover Database (USGS 2019). More specifically, we used the 2016 land cover data. We calculated the area of land classified as developed (this includes the four developed categories of open space, low intensity, medium intensity, and high intensity) and agriculture (this includes the land cover classified as pasture/hay and cultivated crops) using ArcGIS. In some instances, the ESU/DPS boundary layer did not include the entire migration corridor along the Columbia or Willamette (where applicable) Rivers. In these instances, NMFS calculated the developed and agricultural land cover in the 10th-level HUCs bordering the migratory corridor and included those calculations for each applicable ESU/DPS. Only the ESU/DPS area within Oregon was considered. NMFS assigned a risk of high, medium, and low when the percent of developed/agricultural land cover was >25, 10 to 25, and <10, respectively. Table 22 summarizes the total percent of land cover classified as developed or agriculture within the ESU/DPS boundary with the assigned risk rating.

Table 22. Percent of the combined developed and agricultural land covers within each ESU/DPS boundary and its associated exposure risk.

Species	Developed & Agricultural Land Cover (%)	Land Use Risk
LCR Chinook	17	Moderate
LCR Steelhead	19	Moderate
LCR Coho	17	Moderate
ORC Coho	8	Low
SONCC Coho	9	Low
CR Chum	24	Moderate
UWR Chinook	32	High
UWR Steelhead	43	High
MCR Steelhead	16	Low
UCR Chinook	32	High
UCR Steelhead	32	High
SRS Chinook	16	Moderate
SRF Chinook	27	High
SRB Steelhead	32	High
SR sockeye	14	Moderate
Green sturgeon	7	Low
Eulachon	18	Moderate

2.5.5.2 Point Source Discharges

Section 2.4.3 described the potential point source discharges of aluminum. EPA provided NMFS with an ArcGIS layer of these 73 potential point source discharges, which are shown on Figure 3. Because the anodizing facilities do not discharge to surface water, they were not included in the summation of potential point source discharges of aluminum. Using ArcMap, we were able to identify the number and types of potential point source discharges within the range of each ESU/DPS, including migration corridors (as previously described). NMFS assigned a risk of high, medium, and low when the number of potential point sources discharges containing aluminum was >20, 10 to 20, and <10, respectively. Table 23 summarizes the number of NPDES permits with the potential to discharge aluminum along with the risk rating assigned. It is reasonable to assume that additional point sources, or increased discharges from existing point sources, will occur in the future. However, it is not possible to predict the number and location of new facilities that will be permitted. For purposes of this Opinion, we recognize that increases may occur; however, we assume that new or expanded discharges will not occur such that the NPDES risk summarized in Table 3 below will substantially change.

Table 23. Number of NPDES facilities with potential for aluminum in their discharge, and the associated NPDES exposure risk.

Species	# NPDES Facilities	NPDES Risk
LCR Chinook	6	Low
LCR Steelhead	4	Low
LCR Coho	6	Low
OC Coho	23	High
SONCC Coho	8	Low
CR Chum	7	Low
UWR Chinook	20	Moderate
UWR Steelhead	18	Moderate
MCR Steelhead	6	Low
UCR Chinook	5	Low
UCR Steelhead	5	Low
SRS Chinook	5	Low
SRF Chinook	5	Low
SRB Steelhead	5	Low
SR sockeye	5	Low
Green sturgeon	3	Low
Eulachon	2	Low

Point source discharges are allowed to discharge aluminum, and in some cases are allowed to discharge aluminum in concentrations greater than the criteria. This occurs when the ODEQ authorizes a mixing zone specific to aluminum. A mixing zone allows for a discharge to undergo initial dilution and mixing in the receiving stream; thus, it is possible aluminum concentrations may exceed the acute or chronic IWQC in small, localized areas near point source discharges. Oregon's regulations require ODEQ to provide specific information about the size, shape, location, and toxicity characteristics of a mixing zone in an NPDES permit. Mixing zones, by regulation, cannot impair the integrity of the water body as a whole; cannot be lethal to organisms passing through the mixing zone; and mixing zones cannot pose a health risk via any likely pathway to exposure. According to EPA (2020) based on a review of NPDES permits in early 2006, approximately 85 percent of individual permits in Oregon have authorized mixing zones (note, this analysis is not specific to mixing zones authorized for aluminum discharges).

EPA (2020) identified two NPDES facilities in Oregon with effluent limits for aluminum. The Fujimi Corporation facility discharges aluminum into an unnamed drainage ditch that flows into Coffee Lake Creek, which is in turn a tributary to the Willamette River. The Northwest Aluminum Specialties - Northwest Aluminum Company discharges aluminum to the Columbia River. Based on an analysis of the available data for these two facilities, EPA concluded that neither facility had reasonable potential to exceed the proposed criteria. Regardless, EPA established hypothetical scenarios for these facilities (i.e., assigned a hypothetical effluent limit). The calculation of these hypothetical NPDES permit limits was conducted to highlight, as an example, how future NPDES permit limits for facilities that are found to have reasonable potential to cause or contribute to an exceedance of the proposed aluminum aquatic life criteria could be implemented in NPDES permits. The protectiveness of these two hypothetical permits were examined by considering time-variable factors in a Monte-Carlo analysis to simulate receiving stream Al concentrations relative to the chronic low effect threshold values of listed species that were identified to be sensitive to chronic Al exposures (based on screening level chronic effects assessment). For brevity, we only summarize EPA's findings here; the analysis methodology employed by EPA is described in Appendix F to the BE (EPA 2020).

For the Fujimi facility, the results of the discharge simulation indicates aluminum concentrations at the edge of the chronic mixing zone (i.e., about 30 feet downstream of the point of discharge) may exceed listed species chronic low effect threshold values (i.e., EC_5) under very limited circumstances, less than 10% of the time. For the Northwest Aluminum facility, the results of the discharge simulation indicates aluminum concentrations at the edge of the mixing zone (i.e., 300 feet downstream of the point of discharge) would be less than half of the estimated low effect threshold value.

In summary, point source discharges of aluminum create additional risk for ESA-listed species to be exposed to harmful concentrations of aluminum, especially in cases where mixing zones are authorized. However, this risk can be minimized by ensuring mixing zones are as small as feasible and ensure new point sources discharges are not located in sensitive habitats.

2.5.5.3 Life History Stage and Use of Oregon Waters

Section 2.4.1 describes how each species utilizes the action area. The risk of population-level exposure to aluminum at criteria concentrations in Oregon was qualitatively assessed based on our understanding of how each species utilizes freshwater in Oregon. Species that are known to spawn and rear for extended periods of time in Oregon waters were considered to have a high risk associated with their overall life history stage. Species that do not spawn in Oregon waters, but that may spend extended periods of time rearing in Oregon waters such as the Columbia River or LCRE were considered to have a moderate risk associated with their overall life history stage. Finally, species do not spawn nor spend extended periods of time rearing in or migrating through Oregon waters were considered to have a low risk associated with their overall life history stage. The life history stage risk rating for each species if provided in Table 24.

Table 24. Overall life history stage risk of population-level exposure to aluminum at criteria concentrations for each ESU/DPS.

Species	Overall Life History Stage Risk
LCR Chinook	High
LCR Steelhead	High
LCR Coho	High
ORC Coho	High
SONCC Coho	High
CR Chum	High
UWR Chinook	High
UWR Steelhead	High
MCR Steelhead	High
UCR Chinook	Moderate
UCR Steelhead	Moderate
SRS Chinook	High
SRF Chinook	High
SRB Steelhead	High
SR sockeye	Low
Green sturgeon	Moderate
Eulachon	Moderate

2.5.5.4 Total Aluminum Concentrations

As described in Section 2.4.5, there were 417 samples with paired water quality (pH, DOC, and total hardness) and total aluminum measurements available within the ESU/DPS boundaries. This made it possible to calculate acute and chronic IWQC (EPA 2020) for comparison with the measured total aluminum concentrations (Table 25) (Figures 9, 10). The comparison was made by calculating the ratio of the total aluminum concentration to the IWQC. Results equal to or greater than one indicate an excursion above the criteria.

This comparison offers insight into how pervasive elevated aluminum is in the environment. The vast majority (i.e., >80% of samples) are less than one-half the acute IWQC. Very few of the samples were equivalent to, or greater than, the acute IWQC (Figure 9). As would be expected (because the chronic criterion is less than the acute criteria), more samples were greater than the chronic IWQC (Figure 10). Observed ratios greater than one most often occurred during February, March, April, May, October, and November. These are typically months of higher precipitation. As such, it is possible that these excursions above the criterion could be characterized by higher proportions of inorganically bound aluminum that is not bioavailable (refer to Section 2.4.5.4).

In assigning risk for population-level exposure to elevated aluminum concentrations, NMFS considered the percent of samples that were less than one-half the IWQC. We assigned a risk rating of high, medium, and low for <25, 25-75, and >75 percent of the samples that were less than one-half the IWQC.

Table 25. Comparison of the CMC and CCC to total aluminum concentrations, and the overall risk of population exposure to aluminum at criteria concentrations.

Species	# Sample Locations	N	# (%) > CMC	% < ½ CMC	# (%) ≥ CCC	% < ½ CCC	Risk of Population Exposure
LCR Chinook	24	48	3 (6%)	85	7 (15%)	71	Moderate
LCR Steelhead	16	33	3 (9%)	82	6 (18%)	70	Moderate
LCR Coho	24	48	3 (6%)	85	7 (15%)	71	Moderate
ORC Coho	42	103	7 (7%)	79	22 (21%)	65	Moderate
SONC Coho	14	44	1 (2%)	93	2 (5%)	80	Low
CR Chum	23	46	3 (7%)	85	7 (15%)	70	Moderate
UWR Chinook	56	137	7 (5%)	79	28 (20%)	68	Moderate
UWR Steelhead	46	108	2 (2%)	80	22 (20%)	67	Moderate
MCR Steelhead	23	69	0 (0%)	97	2 (3%)	86	Low
UCR Chinook	3	3	0 (0%)	100	0 (0%)	100	Low
UCR Steelhead	3	3	0 (0%)	100	0 (0%)	100	Low
SRS Chinook	8	19	0 (0%)	79	4 (21%)	58	Moderate
SRF Chinook	3	3	0 (0%)	100	0 (0%)	100	Low
SRB Steelhead	8	19	0 (0%)	79	4 (21%)	58	Moderate
SR Sockeye	3	3	0 (0%)	100	0 (0%)	100	Low
Green Sturgeon	15	32	2 (6)	84	6 (18)	63	Low
Eulachon	17	33	1 (3)	88	4 (12)	61	Low

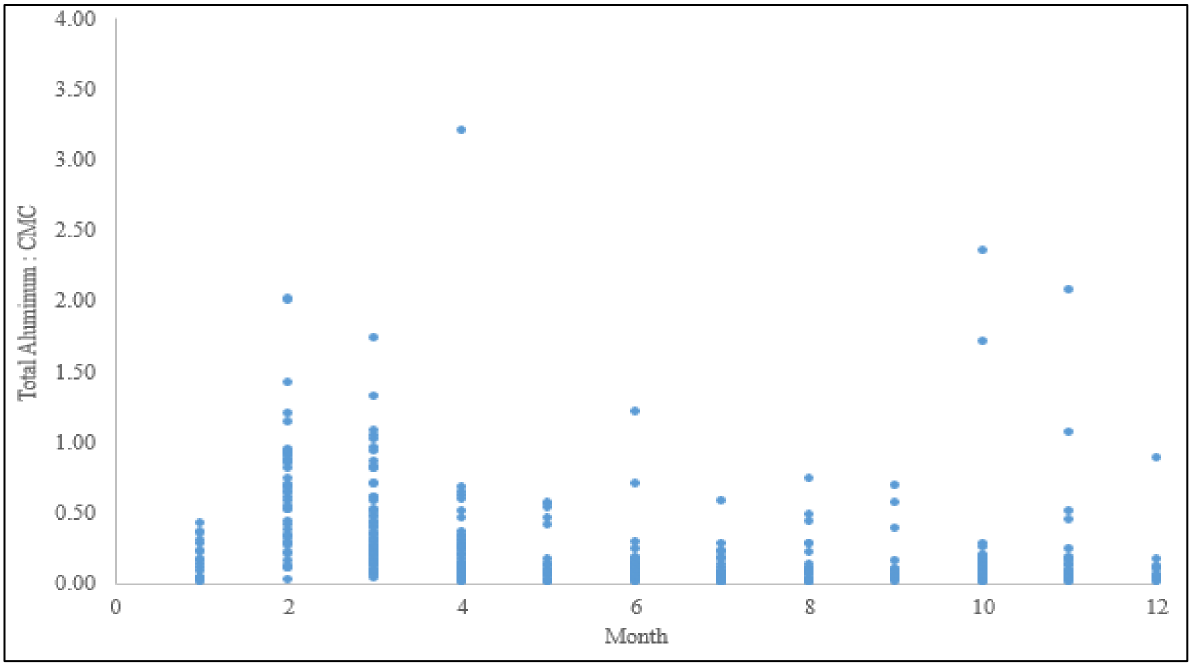

Figure 9. Ratios of total aluminum concentrations to the CMC for the 417 samples available within the action area. Data are grouped by month to identify potential seasonal variability.

Note: For scaling purposes, ratios greater than 4 are not shown, omitting two data points (both collected in December).

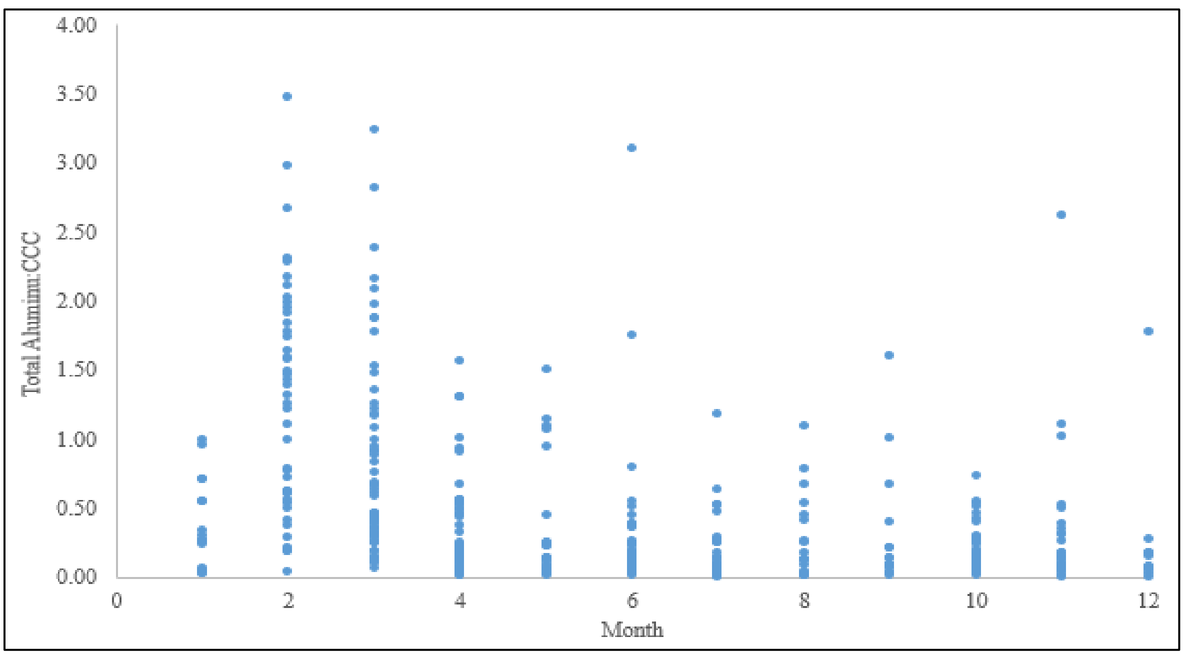

Figure 10. Ratios of total aluminum concentrations to the CCC for the 417 samples available within the action area. Data are grouped by month to identify potential seasonal variability.

Note: For scaling purposes, ratios greater than 4 are not shown, omitting eight data points (2 for February, October, and December; and 1 for April and November).

2.5.5.5 Integrating Exposure Risk Categories

We developed an overall risk of exposure to aluminum at criteria concentration by integrating the land use, NPDES, life history stage, and aluminum exposure risk categories for each fish ESU/DPS. Generally speaking, when three or more categories were rated as high or low, the overall risk was considered to be high or low, respectively. All other combinations were considered to present a moderate risk. Our risk decisions for the fish species considered in this Opinion are presented in Table 26.

Table 26. Exposure risk categories and summary of overall risk of exposure to aluminum at criterion concentrations. Risk ratings are summarized at the ESU/DPS scale.

Species	Land Use Risk	NPDES Risk	Life History Stage Risk	Risk of Exposure to IWQC	Overall Exposure Risk
LCR Chinook	Moderate	Low	High	Moderate	**Moderate**
LCR Steelhead	Moderate	Low	High	Moderate	**Moderate**
LCR Coho	Moderate	Low	High	Moderate	**Moderate**
ORC Coho	Low	High	High	Moderate	**Moderate**
SONC Coho	Low	Low	High	Low	**Low**
CR Chum	Moderate	Low	High	Moderate	**Moderate**
UWR Chinook	High	Moderate	High	Moderate	**Moderate**
UWR Steelhead	High	Moderate	High	Moderate	**Moderate**
MCR Steelhead	Low	Low	High	Low	**Low**
UCR Chinook	High	Low	Moderate	Low	**Low**
UCR Steelhead	High	Low	Moderate	Low	**Low**
SRS Chinook	Moderate	Low	High	Moderate	**Moderate**
SRF Chinook	High	Low	High	Low	**Moderate**
SRB Steelhead	High	Low	High	Moderate	**Moderate**
SR Sockeye	Moderate	Low	Low	Low	**Low**
Green Sturgeon	Low	Low	Moderate	Low	**Low**
Eulachon	Moderate	Low	Moderate	Low	**Low**

2.5.6 Effects to ESA-Listed Fish Species - Integration of Toxicity and Exposure Risks

In this step, the risk of exposure and risk of toxicity assessments are integrated to assign an overall risk of the proposed action to each fish ESU/DPS as high, medium, or low. This is illustrated in Figure 11. A "high" risk determination for a species is concluded when, there was a high exposure risk combined with a high toxicity risk ("high/high") and/or for species with a high/moderate" combinations (red squares in Figure 11). In similar fashion, a medium risk determination resulted when there was a moderate risk of species-level toxicity coupled with either a low or moderate risk of exposure (yellow squares in Figure 11). A low risk determination was made wherever there was a low risk of species-level toxicity risk, regardless of the exposure risk (green squares in Figure 11). A summary of the toxicity and exposure risk results and the overall species risk determinations are provided in Table 27.

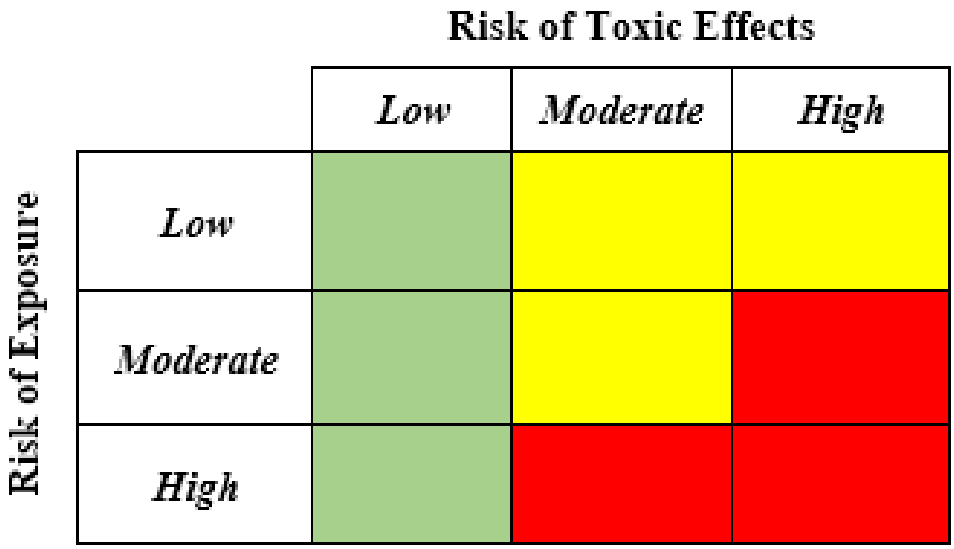

Figure 11. Overall species-level risk characterization based on the species-level risks of exposure and toxicity. Red squares indicate a high risk; yellow squares indicate medium risk; and green squares indicate low risk.

Table 27. Integration of the population-level toxicity and exposure risks into an overall species risk. .

Species	Risk of Toxicity	Risk of Exposure	Integrated Species Risk
LCR Chinook	Moderate	Moderate	**Medium**
LCR Steelhead	Moderate	Moderate	**Medium**
LCR Coho	Moderate	Moderate	**Medium**
ORC Coho	Moderate	Moderate	**Medium**
SONC Coho	Moderate	Low	**Medium**
CR Chum	Moderate	Moderate	**Medium**
UWR Chinook	Moderate	Moderate	**Medium**
UWR Steelhead	Moderate	Moderate	**Medium**
MCR Steelhead	Moderate	Low	**Medium**
UCR Chinook	Moderate	Low	**Medium**
UCR Steelhead	Moderate	Low	**Medium**
SRS Chinook	Moderate	Moderate	**Medium**
SRF Chinook	Moderate	Moderate	**Medium**
SRB Steelhead	Moderate	Moderate	**Medium**
SR Sockeye	Low	Low	**Low**
Green Sturgeon	Moderate	Low	**Medium**
Eulachon	Moderate	Low	**Medium**

Finally, we next qualitatively assessed whether the overall integrated species risk rose to a level that reasonably would be expected to appreciably reduce 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. A high integrated species risk would lead us to conclude that the proposed action would reasonably be expected to reduce the survival and recovery of a species. A low or medium integrated risk leads us to conclude that the proposed action would not be reasonably expected to reduce the likelihood of survival and recovery.

Our analysis suggests that the vast majority of species have a medium integrated species risk. We recognize there is risk of individual mortality as a result of exposure to aluminum at criteria concentrations, and depending on the species, the risk of toxicity manifesting itself at the population- or species-level ranges from low to moderate. Furthermore, we recognize that there is a low or moderate risk of exposure for the species considered in this Opinion. However, when considering the exposure and toxicity risk together, mortality events are expected to be episodic, localized, and will not impact multiple generations. Because such few individuals are expected to be killed, we do not believe there will be any reduction in survival and recovery of ESA-listed fish species.

2.5.7 Effects to SRKW

The proposed action will not directly affect SRKWs, but rather has the potential to indirectly affect the quality and quantity of SRKW prey items. As described in Section 2.5.4.4, aluminum is not anticipated to bioaccumulate through the food chain; therefore, the proposed action is not expected to affect the quality SRKW prey items. However, the proposed action has the potential to reduce the quantity of prey items.

The analysis of chronic criteria in Section 2.5.4.2 above showed potential for reduced growth of early life stages, which was also noted in the BE (EPA 2020). Reduced growth in juvenile salmonids can lead to reduced survival (Baldwin et al. 2009; Mebane and Arthaud 2010). In some studies, mortality occurred after exposure to aluminum over lengthy periods (Gunderson et al 1994). Similarly low levels of mortality from acute criteria are possible, as population models showed given the conservative assumptions of constant exposure. Together, both criteria could lead to reduced adult salmonid numbers from either juvenile mortality, or from reduced juvenile growth leading to lower survival to adult stages.

The first step in analyzing the effects on SRKWs from diminished abundance of adult Chinook or other salmonids is to consider the species found in whale diet studies. While SRKWs consume a variety of fish species (22 species) and one species of squid (Ford et al. 1998; Ford et al. 2000; Ford and Ellis 2006; Hanson et al. 2010; Ford et al. 2016), salmon are identified as their primary prey. Ongoing research from inland waters of Washington State and British Columbia, Canada during summer months, included direct observation, scale and tissue sampling of prey remains, and fecal sampling. The diet data suggest that SRKWs are consuming mostly larger (i.e., generally age 3 and up) Chinook salmon (Ford and Ellis 2006). While there are fewer Chinook than other salmonids in some areas and during certain time periods, they are the primary prey consumed (Ford and Ellis 2006). The species' large size, high fat and energy content, and year-round occurrence in the SRKWs' geographic range explain this in part. Chinook salmon have the highest value of total energy content compared to other salmonids because of their larger body size and higher energy density (kilocalorie/kilogram (kcal/kg))

(O'Neill et al. 2014). Research suggests that SRKWs are capable of detecting, localizing, and recognizing Chinook salmon through their ability to distinguish Chinook echo structure as different from other salmon (Au et al. 2010). The degree to which SRKWs are able to or willing to switch to non-preferred prey sources (i.e., prey other than Chinook salmon) is largely unknown, and likely varies depending on the time and location. The conservative approach to assessing impacts from prey reductions focuses on Chinook salmon, although previous genetics work has suggested that SRKWs switch from Chinook to other salmon in fall months (particularly coho and chum salmon) (Ford et al. 2016). Given Chinook salmon are consumed throughout the whales' range and prey samples indicate they are consumed the majority of the time, and because not much is known about SRKW prey preferences during the periods when abundance Chinook salmon are not available, we will focus on the Chinook stocks affected by the acute and chronic IWQC in the proposed action.

In an effort to characterize the coastal distribution of SRKWs, several satellite tags and acoustic recorders have been deployed primarily in Washington waters, but also off Oregon and California. Data from these deployments suggest differences in habitat use between the J pod and the K and L pods. The J pod appears to remain much more within the Salish Sea, while the K and L pods spend more time in coastal waters (Hanson et al. 2018). Considering these differences, they are likely to have differential responses to changes in the abundance of particular Chinook salmon stocks. An earlier analysis examined all three pods together, noting that statistical power is lost when analyzing one pod at a time due to lower sample sizes (NMFS 2020).

Chinook salmon abundances are tracked in fisheries in areas north and south of Cape Falcon Oregon. North of Cape Falcon, abundance estimates ranged from 819,183 to 2,446,093 Chinook salmon in 1992 – 2016, prior to fishing. Whales are observed in this area in all seasons, and certain species of Chinook leaving Columbia River waters will turn north and traverse this reach of the coast. However, Chinook salmon from rivers outside of Oregon also contribute to these ocean fisheries. In Oregon's coastal waters from Cape Falcon, OR south to Horse Mountain, CA, pre-fisheries abundance estimates ranged from 760,853 to 2,492,455 Chinook salmon during 1992 – 2016 with an average of approximately 1.5 million in the last 10 years. Again, these estimates are for the total number of Chinook salmon, which includes fish from outside of the action area. The reduction in abundance from salmon fisheries managed by the Pacific Fisheries Management Commission (PFMC) declined during the last decade from 13.5% to 7.0% on average (NMFS 2020).

To examine how the overlap between SRKWs and salmon runs could be affected by changes in abundance, prey and fecal samples were collected in winter and spring months. Although fewer observed predation events and collected fecal samples were from coastal waters than inland waters, recent data indicate Chinook salmon provide an important dietary component when the SRKWs overlap with the Chinook runs. Satellite tags were used to locate and follow the whales to obtain predation and fecal samples, for a total of 55 samples from northern California to northern Washington during 2013-2016. Chinook were the primary species detected in diet samples on the outer coast, although steelhead, chum, lingcod, and halibut were also detected in samples. Columbia River spring runs of Chinook salmon are part of the SRKW diets and were linked to the occurrence of K and L pods off the Columbia River in March. Genetic stock identification from samples collected in winter and spring in coastal waters from California through Washington included 12 U.S. west coast stocks, with over half the Chinook salmon consumed having originated in the Columbia River (Hanson et al. 2013; and Hanson et al. in prep cited in NMFS 2020). During these winter-spring periods, most of the Chinook prey samples originating from the Columbia River basin included LCR spring-run, MCR tule, and UCR summer/fall-runs. However, the Chinook stocks included fish from as far north as the Taku River (Alaska and British Columbia stocks) and as far south as the Central Valley California.

Scientists and managers from the U.S. and Canada created a priority stock report using observations of Chinook salmon stocks found in scat and prey scale/tissue samples, and by estimating the spatial and temporal overlap with Chinook salmon stocks ranging from Southeast Alaska to California (NOAA and WDFW 2018). They created a scoring system, based on diet contribution, reduced body condition or diverse diet for SRKWs, and spatio-temporal overlap. Rivers with species in the action area are shown in Table 28. In addition to the ESA-listed species analyzed for the effects of the proposed criteria, other species which are not listed as threatened or endangered are also shown as they could be affected by the proposed action, and in turn may affect the SRKW prey availability.

Table 28. Priority Chinook stocks modeled in 2018 SRKW Report (NOAA and WDFW 2018). A higher total score denotes an increased importance in the diet of SRKW.

ESU / Stock Group	Run Type	Rivers or Stocks in Group	Diet Contribution Score (0,1)[a]	SRKW Reduced Body Condition or Diverse Diet Score (0,1)[b]	Spatio-Temporal Overlap Score (0 - 3)[c]	Total Score (sum of factors)
LCR	Fall	Fall Tules and Fall Brights (Cowlitz, Kalama, Clackamas, Lewis, others)	1	1	2.63	4.63
UCR & SR	Fall	Upriver Brights	1	1	2.25	4.25
LCR	Spring	Lewis, Cowlitz, Kalama, Big White Salmon	1	1	2.25	4.25
MCR	Fall	Fall Brights	1	1	2.06	4.06
SR	Spring-Summer	Snake, Salmon, Clearwater	1	1	1.88	3.88
MCR & UCR	Spring	Columbia, Yakima, Wenatchee, Methow, Okanagan	1	1	1.31	3.31
MCR & UCR	Summer		1	1	1.31	3.31
Klamath River	Fall	Upper Klamath and Trinity	1	1	0.75	2.75
Klamath River	Spring	Upper Klamath and Trinity	1	1	0.75	2.75
UWR	Spring	Willamette	0	0	2.25	2.25
North & Central Oregon Coast	Fall	Northern (Siuslaw, Nehalem, Siletz) and Central (Coos, Elk, Coquille, Umpqua)	0	0	1.41	1.41
SONCC	Fall	Rogue, Chetco, Smith, lower Klamath	0	0	0.75	0.75
SONCC	Spring	Rogue	0	0	0.75	0.75

[a]If the Chinook stock was observed >=5% of the whales diet in summer or fall/winter/spring, the stock receives 1 point. If it was not observed in the diet, the stock receives 0 points.
[b]Reduced body condition and a diversified diet are seen to occur in non-summer months. If a stock is consumed during October through May, it receives 1 point. If it is consumed during June through September, the stock receives 0 points.
[c]For each space/time area described above, if more than 25% of the Chinook stock is distributed in that area, the area receives a sub-score of 2. For areas that contain between 5% and 25% of the Chinook stock, the area receives a sub-score of 1. If an area contains less than 5% of the Chinook stock, it receives a sub-score of 0. The sub-scores for each area are multiplied by an importance weight for each area. The final score for the Chinook stock/population is the sum of the products of the scores and weight for each area normalized such that the highest possible score of a given stock is equal to 3.

The abundance of Chinook salmon species rearing or migrating through Oregon rivers to coastal areas may be reduced through direct mortality of juveniles or through indirect mortality resulting from sublethal effects associated with aluminum exposures. In turn, SRKWs are likely to be harmed by reduced prey abundance. In the EPA analyses, chronic IWQC were greater than the EC_5 and EC_{10} concentrations over 90 and 58 percent of the time, respectively. None of the chronic IWQC exceeded EC_{15} values as noted above. In their analysis, the EPA (2020) assumed the biomass endpoint equated to mortality. Thus, a five percent reduction in biomass was interpreted to be representative of five percent mortality. The EPA methodology was limited to a subset of the available studies that have examined chronic toxicity. Section 2.5.4.2 of this Opinion summarizes other studies that examined aluminum toxicity under extended exposure durations. The preponderance of data suggests that mortality, although it could occur if fish are exposed to aluminum at concentrations less than the CCC for extended periods of time, is not likely to be prevalent and only a few individuals are expected to be killed. Sublethal effects (e.g., reduced growth, strike frequency, and swimming performance), on the other hand, are more likely to result from exposure to the CCC and can ultimately lead to death of individual fish. Most researchers observed measurable sublethal effects, but they were not statistically different from a control (e.g., NOEC). In some cases, the test-specific chronic IWQC calculated were higher than the concentration reported in the studies, suggesting effects could be experienced at concentrations below the criterion. In these studies, fish were exposed to the reported concentrations for longer than the chronic criteria allows (7- 60 days vs. 4 days).

Conservatively, we consider the EPA's estimate of five percent loss in biomass during juvenile life stages to translate into a potential loss in adults present during SRKW coastal feeding, noting that approximately one percent of Columbia River outmigrating juveniles tracked with PIT tags are found to return to the Columbia as adults (Smith et al. 2018). Hence, using adult Chinook counts in the Columbia River as a surrogate for those that would be present in the coastal area nearby for SRKW predation, we can calculate a rough loss in prey abundance as the product of the percent biomass loss (5 percent) and the percent smolt to adult return estimate (1 percent). This results in an estimated loss of 0.05 percent of the adults available to SRKW.

This 0.05 percent loss is likely an overestimate of the adults that could be missing from the ocean feeding areas on an annual basis when the SRKWs are present. This is because fish from areas outside of the Columbia River also contribute to the prey base for SRKW. Lower prey availability resulting from fewer adults is likely to harm SRKWs, with variable effects by year due to fluctuating Chinook abundance.

A recent analysis of harvest reductions in prey noted that when prey is scarce or in low density, SRKWs likely spend more time foraging. Greater energy expenditure and prey limitation can cause poor body condition and nutritional stress (NOAA 2020). Nutritional stress, the condition of being unable to acquire adequate energy and nutrients from prey resources, can lead to reduced body size of individuals and to lowered reproductive or survival rates in a population (Trites and Donnelly 2003). Body condition in whales can be influenced by a number of factors, including prey availability, increased energy demands, disease, physiological or life history status, and will vary over seasons or across years. Body condition data shows declines in some pods scattered across demographic and social groups (Fearnbach et al. 2018). Annual aerial surveys of the population from 2013-2017 (with exception of 2014) have detected declines in condition before the death of seven SRKWs (L52 and J8 as reported in Fearnbach et al. (2018); J14, J2, J28, J54, and J52 as reported in Durban et al. (2017)), including five of the six most recent mortalities (Trites and Rosen 2018). These data have provided evidence of a general decline in SRKW body condition since 2008.

The effects of energetic stress on adult females and juveniles caused by incremental increases in energy expenditures or incremental reductions in available energy have been studied extensively in adult females (Gamel et al. 2005; Schaefer 1996; Daan et al. 1996), and in juveniles (Trites and Donnelly 2003). Incremental increases in energy demands or incremental reductions in available energy from reductions in prey should effect an animal's energy budget similarly. Malnutrition and persistent or chronic stress can induce changes in immune function in mammals and may be associated with increased bacterial and viral infections, and lymphoid depletion (Mongillo et al. 2016; Neale et al. 2005; Maggini et al. 2018).

2.5.8 Effects to Designated Critical Habitat

The proposed action has the potential to affect the water quality and forage PBFs of critical habitat for ESA-listed fish species and the prey quantity PBF of SRKW designated and proposed critical habitat. To evaluate these effects we consider whether the proposed action will result in aluminum concentrations sufficient to reduce the conservation value of critical habitat via degradation of: (1) Water quality in spawning, rearing, and migratory habitats; or (2) prey in rearing and migratory habitats. Degradation of water quality means concentrations of aluminum are sufficient to elicit adverse effects to ESA-listed species. Degradation of prey means that either the quality (either through bioaccumulation of aluminum or by reduced species richness) or quantity of the forage base is reduced

The potential impacts of the proposed action on water quality are described in sections 2.5.4 through 2.5.6. Our analysis of effects to ESA-listed species captures the potential for adverse effects to the water quality PBF. Our conclusions for the potential effect to the designated critical habitat for each species is captured in Table 27. Overall, the proposed action may degrade the conservation value of water quality in small, discrete areas of critical habitat throughout each species range (e.g., near point source discharges or near areas heavily developed or used for agriculture), but that is not expected to degrade the conservation of critical habitat at larger watershed or subwatershed scales.

Section 2.5.4.3 discusses the potential of the proposed action to reduce the quantity of prey available for consumption, and Section 2.5.4.4 discusses the potential for aluminum bioaccumulation. As discussed in Section 2.5.4.3, EPA found that the most sensitive prey item GMCV was always greater than the chronic criteria, suggesting there is a low risk of toxicity to prey items. Considering the analysis was based on small subset of individual prey species, it is conceivable that other untested prey items of salmonids, eulachon, and green sturgeon may be negatively impacted if exposed to aluminum at criteria concentrations. However, juvenile salmonids are opportunistic predators, and the loss of a minority of taxa might not be a severe affect to the forage base PBF if other prey were still diverse and abundant. Regarding the quality of the prey, aluminum is not known to bioaccumulate or biomagnify up the food chain. Based on the risk of toxicity to prey, the risk of exposure, and the lack of bioaccumulation of aluminum, we do not believe that the overall quantity of forage available to ESA-listed species will be reduced to a degree that will diminish the conservation value of the forage PBF.

Section 2.5.7 discussed the potential for the proposed action to cause a reduction in the prey quantity PBF for SRKW. We estimated that there could be a 0.05 percent loss in prey availability for SRKW. However, we recognize this is likely an overestimate of the adults that could be missing from the ocean feeding areas on an annual basis when the SRKWs are present. This is because fish from areas outside of the Columbia River also contribute to the prey base for SRKW. While the proposed action may result in a slight loss prey availability, we do not believe the loss will appreciably reduce the conservation value of the prey quantity and availability PBF.

2.5.9 Implications of Climate Change

Climate change will continue to influence the viability of ESA-listed species. Of particular concern relative to this consultation is the potential for changes in flow regimes and increasing water temperatures. Lower baseflows lends less water for dilution of contaminants during late summer and fall. Because NPDES permits are to be reissued on a regular basis (i.e., at least every 5 years), there is opportunity to capture changes in low flows in the evaluation of the needed for, and where necessary establishment of, permit limits for contaminants. As more precipitation falls as rain, there is a greater potential for increased stormwater contributions of aluminum from point and nonpoint source pollution. While much of this aluminum is expected to be bound to inorganic material and not bioavailable, it is possible that some aluminum may become bioavailable with fluxes in water chemistry (e.g., changes in pH).

Thermal stress will continue into the future and is predicted to be exacerbated with climate change, depending on local site conditions. Elevated stream temperatures can reduce the ability of individual fish to tolerate elevated contaminant concentrations. As described in Section 2.5.2.1., toxicity testing is generally performed near optimal thermal conditions for fish, which fish are oftentimes most resistant to toxicity. As such, contaminant toxicity may be underestimated.

2.6 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 and 402.17(a)). 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 contribution of non-Federal activities to the current condition of ESA-listed species and designated critical habitats within the program-level action area was described in the Status of the Species and Critical Habitats and Environmental Baseline sections, above. Among those activities were agriculture, forest management, mining, road construction, urbanization, water development, implementation of CWA programs, and river restoration. Those actions were driven by a combination of economic conditions that characterized traditional natural resource-based industries, general resource demands associated with settlement of local and regional population centers, and the efforts of social groups dedicated to river restoration and use of natural amenities, such as cultural inspiration and recreational experiences.

Resource-based industries caused many long-lasting environmental changes that harmed ESA-listed species and their critical habitats, such as state-wide loss or degradation of stream channel morphology, spawning substrates, instream roughness and cover, estuarine rearing habitats, wetlands, riparian areas, water quality (e.g., temperature, sediment, dissolved oxygen, contaminants), fish passage, and habitat refugia. Those changes reduced the ability of populations of ESA-listed species to sustain themselves in the natural environment by altering or interfering with their behavior in ways that reduce their survival throughout their life cycle. The environmental changes also reduced the quality and function of critical habitat PBFs that are necessary for successful spawning, production of offspring, and migratory access necessary for adult fish to swim upstream to reach spawning areas and for juvenile fish to proceed downstream and reach the ocean. Without those features, the species cannot successfully spawn and produce offspring. However, the declining level of resource-based industrial activity and rapidly rising industry standards for resource protection are likely to reduce the intensity and severity of those impacts in the future.

The economic and environmental significance of the natural resource-based economy is currently declining in absolute terms and relative to a newer economy based on mixed manufacturing and marketing with an emphasis on high technology (Brown 2011). Nonetheless, resource-based industries are likely to continue to have an influence on environmental conditions within the action area for the indefinite future. Over time industries have adopted management practices that avoid or reduce many of their most harmful impacts, and this is anticipated to continue into the future.

While natural resource extraction within the Pacific Northwest may be declining, general resource demands are increasing with growth in the size and standard of living of the local and regional human population (Metro 2010; Metro 2011). Population growth is a good proxy for multiple, dispersed activities and provides the best estimate of general resource demands because as local human populations grow, so does the overall consumption of local and regional natural resources. According to Portland State University (2019), Oregon's population has grown by more than 400,000 people since 2010. That represents a population growth rate of approximately 11 percent. Most of the population centers in Oregon occur west of the Cascade Mountains. NMFS assumes that future private, state, and federal actions will continue within the action areas, increasing as population rises.

The ODEQ will continue to implement CWA programs in the state. Water quality assessment reports are prepared on a regular basis to identify where water quality is supporting its beneficial uses and where water quality is not attaining criteria necessary to support beneficial uses. The ODEQ will continue to prepare TMDLs that upon implementation, should help improve water quality conditions. Whether in response to TMDL implementation or as part of ongoing advances in land management, continued and improved implementation of best management practices to reduce the quantity of stormwater runoff and/or improve the quality of stormwater runoff will be needed to minimize nonpoint source contributions of aluminum to the aquatic environment. Where necessary, total aluminum concentrations in point source discharges will be characterized and effluent limits developed when required. Assuming the ODEQ is successful in characterizing the most toxic conditions in the receiving water, regulatory controls on point source discharges of aluminum should help minimize the potential for adverse effects.

The adverse effects of non-Federal actions stimulated by general resource demands are likely to continue in the future driven by changes in human population density and standards of living. These effects are likely to continue to a similar or reduced extent in the rural areas in the action area. Areas of growing population in the action area are likely to experience greater resource demands, and therefore more adverse environmental effects. Land use laws and progressive policies related to long-range planning will help to limit those impacts by ensuring that concern for a healthy economy that generates jobs and business opportunities is balanced by concern for protection of farms, forests, rivers, streams and natural areas (Metro 2000; Metro 2008; Metro 2011). In addition to careful land use planning to minimize adverse environmental impacts, larger population centers may also partly offset the adverse effects of their growing resource demands with more river restoration projects designed to provide ecosystem-based cultural amenities, although the geographic distribution of those actions, and therefore any benefits to ESA-listed species or critical habitats, may occur far from the centers of human populations.

Similarly, demand for cultural and aesthetic amenities continues to grow with human population, and is reflected in decades of concentrated effort by Tribes, states, and local communities to restore an environment that supports flourishing wildlife populations, including populations of species that are now ESA-listed (CRITFC 1995; OWEB 2017). Reduced economic dependence on traditional resource-based industries has been associated with growing public appreciation for the economic benefits of river restoration, and growing demand for the cultural amenities that river restoration provides. Thus, many non-Federal actions have become responsive to the recovery needs of ESA-listed species. Those actions included efforts to ensure that resource-based industries adopt improved practices to avoid, minimize, or offset their adverse impacts. Similarly, many actions are focused on completion of river restoration projects specifically designed to broadly reverse the major factors now limiting the survival of ESA-listed species at all stages of their life cycle. Those actions have improved the availability and quality of estuarine and nearshore habitats, floodplain connectivity, channel structure and complexity, riparian areas and large wood recruitment, stream substrates, stream flow, water quality, and fish passage. In this way, the goal of ESA-listed species recovery has become institutionalized as a common and accepted part of the economic and environmental culture. We expect this trend to continue into the future as awareness of environmental and at-risk species issues increases among the general public.

It is not possible to predict the future intensity of specific non-Federal actions related to resource-based industries at this program scale due to uncertainties about the economy, funding levels for restoration actions, and individual investment decisions. However, the adverse effects of resource-based industries in the action area are likely to continue in the future, although their net adverse effect is likely to decline slowly as beneficial effects spread from the adoption of industry-wide standards for more protective management practices. These effects, both negative and positive, will be expressed most strongly in rural areas where these industries occur, and therefore somewhat in contrast to human population density. The future effects of river restoration are also unpredictable for the same reasons, but their net beneficial effects may grow with the increased sophistication and size of projects completed and the additive effects of completing multiple projects in some watersheds.

Some continuing non-Federal activities are reasonably certain to contribute to climate effects within the action area. However, it is difficult if not impossible to distinguish between the action area's future environmental conditions caused by global climate change that are properly part of the environmental baseline *vs.* cumulative effects. Therefore, all relevant future climate-related environmental conditions in the action area are described in the rangewide status of the species and critical habitat (Section 2.2).

In summary, resource-based activities such as timber harvest, agriculture, mining, shipping, and energy development are likely to continue to exert an influence on the quality of freshwater and estuarine habitat in the action area. The intensity of this influence is difficult to predict and is dependent on many social and economic factors. However, the adoption of industry-wide standards to reduce environmental impacts and the shift away from resource extraction to a mixed manufacturing and technology based economy should result in a gradual decrease in influence over time. In contrast, the population of Oregon is expected to increase in the next several decades with a corresponding increase in natural resource consumption. Additional residential and commercial development and a general increase in human activities are expected to cause localized degradation of freshwater and estuarine habitat. Interest in restoration activities is also increasing as is environmental awareness among the public. This will lead to localized improvements to freshwater and estuarine habitat. When these influences are considered collectively, we expect trends in habitat quality to remain flat or improve gradually over time, although climate change is likely to present challenges. If habitat trends remain flat or gradually improve, this will, at best, have positive influence on population abundance and productivity for the species affected by this consultation. In a worst-case scenario, we expect cumulative effects will have a relatively neutral effect on population abundance trends. Similarly, we expect the quality and function of critical habitat PBFs to express a slightly positive to neutral trend over time as a result of the cumulative effects.

2.7 Integration and Synthesis

The Integration and Synthesis section is the final step in our assessment of the risk posed to species and critical habitat as a result of implementing the proposed action. In this section, we add the effects of the action (Section 2.5) to the environmental baseline (Section 2.4) and the cumulative effects (Section 2.6), taking into account the status of the species and critical habitat (Section 2.2), to formulate the agency's Opinion as to whether the proposed action is likely to: (1) Reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing its numbers, reproduction, or distribution; or (2) appreciably diminish the value of designated or proposed critical habitat as a whole for the conservation of the species.

2.7.1 Species

As described in Section 2.2, individuals of many ESA-listed salmon and steelhead species, and eulachon use the action area to fully complete the migration, spawning and rearing parts of their life cycle. Some salmon, steelhead, and green sturgeon species migrate and rear in the action area; and some species only migrate through the action area, once as out-migrating juveniles and then again as adult fish on upstream spawning migration. The SRKWs forage on salmon and steelhead that originate from within or migrate through the action area. This may occur along the entire coastal range of the SRKWs from California to Vancouver, British Columbia.

The status of each salmon and steelhead species addressed by this consultation varies considerably from very high risk (SR sockeye salmon) to moderate risk (e.g., OC coho salmon, MCR steelhead). Similarly, the hundreds of individual populations affected by the proposed program vary considerably in their biological status. The species addressed in this Opinion have declined due to numerous factors. The one factor for decline that all these species share is degradation of freshwater and estuarine habitat. Human development of the Pacific Northwest has caused significant negative changes to stream and estuary habitat across the range of these species. Poor water quality in some reaches contributes to the degraded habitat.

Eulachon use the estuaries and the first few miles of river mainstems for spawning, incubation, growth, maturation, and migration. Eulachon population abundance has declined significantly since the early 1990s. Although NMFS considers variation in ocean productivity to be the most important natural phenomenon affecting the productivity of this species, NMFS identified many other factors associated with the freshwater phase of their life cycle that are also limiting the recovery of these species. These factors include, but are not limited to, elevated water temperatures; excessive sediment; reduced access to spawning and rearing areas; reductions in habitat complexity, instream wood, and channel stability; degraded floodplain structure and function, and reduced flow.

The sDPS green sturgeon generally migrate in coastal waters of Oregon within the action area and prefer marine waters of less than a depth of 110 meters. The only known sDPS green sturgeon spawning population utilizes the Sacramento River (well outside the action area), and the current estimate of spawning adults is between 24 and 1,872. Limiting factors of green sturgeon within the action area include the lack of water quantity, poor water quality and poaching.

The SRKW DPS is one of the most at-risk species because of their endangered status, and declining population trend. The population has relatively high mortality and low reproduction unlike other resident killer whale populations that have generally been increasing since the 1970s (Carretta et al. 2019). The limiting factors described in the final recovery plan include reduced prey availability and quality, toxic chemicals that accumulate in top predators, and disturbances from vessels and sound.

The environmental baseline varies across the action area. There are relatively few point source discharges of aluminum; however, development and agricultural land use is prevalent in the western part of the state and in lowland coastal habitats. Some total aluminum data were available for characterizing environmental baseline conditions, and this relatively small dataset is assumed to be representative of water quality conditions experienced by ESA-listed species. Overall, aluminum concentrations are typically well below the proposed criteria, although there are some sample locations that warrant further investigation because of point source discharges or land uses nearby. Climate change is likely to exacerbate several of the ongoing habitat issues, in particular, increased summer temperatures, decreased summer flows in the freshwater environment, ocean acidification, and sea level rise in the marine environment. Elevated stream temperatures can reduce the ability of individual fish to tolerate elevated contaminant concentrations. The prey quantity for SRKWs is reduced with some wild salmon species throughout the whales' geographic range at just a fraction of historic levels.

There is limited aluminum toxicity information available for species considered in this Opinion. NMFS assumed that toxicity data for other surrogate salmonid species (e.g., rainbow trout, brook trout, Atlantic salmon, etc.) represent the potential toxicity of aluminum to coho, chum, Chinook, and sockeye salmon; steelhead; eulachon; and green sturgeon. The preponderance of data suggests that mortality resulting from exposure to the acute criteria is possible; however, the level of expected mortality is not expected to reduce the viability of any salmonid populations nor is it expected to reduce the viability of eulachon or green sturgeon populations. Mortality resulting from exposure to the chronic criteria is not likely to be prevalent and few, if any individuals would be expected to die. Sublethal effects are likely to occur if individuals are exposed to the chronic criterion for sufficient periods of time. Sublethal effects range from reduced growth to reduced feeding and swimming capabilities. These types of sublethal effects have the potential to reduce the abundance and productivity of populations if exposure is prolonged or widespread; based on available data, we do not believe the subject species will be exposed to concentrations of aluminum sufficient to elicit a sublethal response for a prolonged time period or throughout a significant part of their range. Contaminants oftentimes do not occur in isolation, but rather as part of complex mixtures. While we are not able to quantitatively assess mixture toxicity, we have recognized that mixtures may have additive toxicity. Aluminum does not appear to bioaccumulate or biomagnify in the food chain. Although there is potential for individual prey organisms to be negatively affected by aluminum concentrations near the proposed criterion, the species considered in this Opinion are opportunistic feeders and prey resources, taken as a whole, are not expected to be diminished to a point which will negatively affect foraging behaviors or foraging success of ESA-listed species. Localized impacts are expected to occur as a result of criteria implementation (e.g., mixing zone authorizations). We have assumed that ODEQ's implementation of CWA programs (water body assessments, TMDL development, and NPDES permit development) will be performed in a manner that will minimize adverse effects resulting from exposure to aluminum.

Overall, because there is some risk of mortality and some risk of sublethal effects from exposures to the proposed criteria, we consider there to be a high risk of toxicity to individual fish. Scaling this up to the population- and species-level requires some qualitative judgements about the information. While the level of mortality that could occur across any given population is expected to be low (and population modeling suggests no changes in lambda), reduced growth is a potential sublethal effect that could subsequently lead to lower rates of survival for individual fish that could manifest into reductions in population abundance and productivity which could subsequently lower the viability of a species. In light of the population modeling results and in light of the uncertainty about the degree to which sublethal effects experienced by individuals could have impacts at the species-level, we conservatively assigned a toxicity risk of moderate to the fish species considered in this Opinion. The SR sockeye salmon is the only exception to this. Because available information suggests little potential for SR sockeye salmon to rear in Oregon waters, we have concluded the risk of toxicity is low for this species. In addition to the risk of species-level toxicity, we considered the risk of species-level exposure to aluminum at criteria levels. We considered four categories of exposure sources: land use; point source dischargers, life history stage, and baseline aluminum concentrations. Upon integrating these four categories of exposure risk, we concluded ten species were at a moderate exposure risk and seven species had a low risk of exposure (Table 26).

The final step in our assessment of effects of the proposed action on ESA-listed species was to integrate the toxicity and exposure risks. This integration is described in Section 2.5.6. The overall risk of the proposed action was deemed to be medium for the following species: LCR Chinook, LCR steelhead, LCR coho, ORC coho, SONCC coho, CR chum, UWR Chinook, UWR steelhead, MCR steelhead, UCR Chinook, UCR steelhead, SRS Chinook, SRF Chinook, SRB steelhead, sDPS green sturgeon, and sDPS eulachon. The overall risk of the proposed action for SR sockeye salmon is considered to be low. As described in Section 2.5.6, the medium and low overall risk ratings indicate that the proposed action is not expected to reduce the abundance or productivity viability parameters for any of the ESA-listed fish species considered in this Opinion. Similarly, as described in Section 2.5.7, we estimated a potential 0.05 percent loss of prey available to SRKW. This is likely an overestimate of the actual number of Chinook lost as a result of the proposed action. While the proposed action may slightly reduce the number of prey available to SRKW, we do not believe such a small reduction will reduce the viability of the SRKW.

Cumulative effects, including commercial and residential development associated with population growth and resource-based activities such as timber harvest, agriculture, mining, shipping, and energy development are likely to continue to exert an influence on the viability of ESA-listed species. The ODEQ will continue to implement CWA programs, issuing NPDES permits and preparing TMDLs for waters not meeting criteria. Interest in restoration activities and environmental awareness among the public is expected to continue into the future, which will bring localized improvements to freshwater and estuarine habitats that in turn will help improve survival of ESA-listed species. When these influences are considered collectively, their impact on species viability is expected to remain flat or improve gradually over time.

In summary, after considering the status of the species, environmental baseline, effects of the action, and cumulative effects, NMFS concludes that EPA's promulgation of aquatic life criteria for aluminum in Oregon will not appreciably reduce the likelihood of survival and recovery of any of the 18 ESA-listed species address in this Opinion.

2.7.2 Critical Habitat

The action area includes designated critical habitat for all of the species considered in this Opinion. CHART teams determined that most designated critical habitat has a high conservation value, largely based on its potential for restoration.

The environmental baseline varies from excellent in wilderness to poor in highly urbanized areas. There are relatively few point source discharges of aluminum; however development and agricultural land use is prevalent in the western part of the state. Some total aluminum data were available for characterizing environmental baseline conditions, and this relatively small dataset is assumed to be representative of water quality conditions experienced by ESA-listed species. Overall, aluminum concentrations are typically well below the proposed criteria, although there are some sample locations that warrant further investigation. Climate change is likely to exacerbate several of the ongoing habitat issues, in particular, increased summer temperatures, decreased summer flows in the freshwater environment, ocean acidification, and sea level rise in the marine environment. Elevated stream temperatures can reduce the ability of individual fish to tolerate elevated contaminant, including aluminum, concentrations.

The proposed action has the potential to affect the water quality and forage PBFs. Our analysis of effects to ESA-listed species (Sections 2.5.4 through 2.5.6) captures the potential for adverse effects to the water quality PBF. Aluminum, if present in sufficient quantities, can be toxic to aquatic invertebrates and other prey organisms. The most sensitive prey item GMCV was always greater than the chronic criteria, suggesting there is a low risk of toxicity to prey items. Furthermore, juvenile salmonids are opportunistic predators, and the loss of a minority of taxa might not be a severe affect to the forage base PBF if other prey were still diverse and abundant. Aluminum is not a bioaccumulative contaminant, so the quality of the prey items is not expected to be negatively affected by the proposed action. Based on the risk of toxicity to prey, the risk of exposure of prey to aluminum at criteria concentrations, and the lack of bioaccumulation of aluminum, we do not believe that the overall quantity or quality of forage available to ESA-listed species will be reduced such that the foraging success or foraging behaviors will be altered at the scale of the population. In summary, while the proposed action may lead to degradation of the water quality and forage PBFs in localized areas, the conservation value of designated critical habitat at the larger watershed scale or at the designation scale is not expected to change. As previously stated, the proposed action may slightly reduce the number of prey available to SRKW. However, because the abundance and productivity viability parameters are not expected to be reduced, the conservation value of the prey PBF of SRKW designated critical habitat is not expected to change. This analysis includes both currently designated critical habitat and proposed critical habitat.

Cumulative effects, including resource-based activities such as timber harvest, agriculture, mining, shipping, and energy development are likely to continue to exert an influence on the quality of freshwater and estuarine habitat in the action area. In addition, population growth and its associated commercial and residential development will cause localized degradation of freshwater and estuarine habitat. The ODEQ will continue to implement CWA programs, issuing NPDES permits and preparing TMDLs for waters not meeting criteria. Interest in restoration activities and environmental awareness among the public is expected to continue into the future, which will bring localized improvements to freshwater and estuarine habitat. When these influences are considered collectively, we expect trends in habitat quality to remain flat or improve gradually over time.

In summary, after considering the status of the species, environmental baseline, effects of the action, and cumulative effects, NMFS concludes that EPA’s promulgation of aquatic life criteria for aluminum in Oregon will not appreciably diminish the value of designated or proposed critical habitat as a whole for the conservation of the species.

2.8 Conclusion

After reviewing the current status of the listed species, the environmental baseline within the action area, the effects of the proposed action, and cumulative effects, it is NMFS's biological opinion that the proposed action is not likely to jeopardize the continued existence of LCR Chinook salmon, UWR Chinook salmon, UCR Chinook salmon, SRS Chinook salmon, SRF Chinook salmon, CR chum salmon, LCR coho salmon, OC coho salmon, SONCC coho salmon, SR sockeye salmon, LCR steelhead, UWR steelhead, MCR steelhead, UCR steelhead, SRB steelhead, green sturgeon, eulachon, and SRKW, or result in the destruction or adverse modification of critical habitat that has been designated or proposed for these species.

2.9 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 listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. For this consultation, we interpret "harass" is to create the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering. Section 7(b)(4) and section 7(o)(2) provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA if that action is performed in compliance with the terms and conditions of this incidental take statement.

2.9.1 Amount or Extent of Take

The proposed criteria for aluminum apply to all freshwater habitats in Oregon, and ESA-listed anadromous species inhabit a subset of Oregon's freshwater. As described below, the proposed action is reasonably certain to cause incidental take of one or more of those fish species. Both adult and juvenile life stages have the potential to be exposed to aluminum at concentrations equivalent to the proposed criteria. Juvenile life stages are expected to be more sensitive to aluminum exposures, and juvenile fish exposed to aluminum at criteria concentrations for sufficient periods of time may experience death or sublethal effects such as reduced growth, reduced predator avoidance, and reduced swimming and foraging capacity that, in turn, could ultimately lead to death.

The potential reduction of juvenile Chinook salmon reaching the ocean is likely to result in some level of harm to SRKW. Reduced prey availability may cause SRKWs to forage for longer periods, travel to alternate locations, or abandon foraging efforts. All individuals of the SRKW DPS have the potential to be adversely affected in the action area. However, the K and L pods are known to use coastal waters off Washington, Oregon, and California where greater prey reduction may occur than in inland waters of the Salish Sea where the J pod primarily occurs.

NMFS is unable to quantify the amount of take that is associated with implementation of the aluminum criteria for the reasons listed below.

1. It is not possible to predict with any certainty the number or location of future point source discharges of aluminum, nor is it possible to predict the amount of aluminum that could be discharged into the environment from both point and nonpoint sources.

2. It is not possible to predict the number of individuals of a species exposed to aluminum at criteria concentrations. Furthermore, it is not possible to count the number of fish that may be adversely affected by such exposures as the majority of effects are anticipated to be sublethal or behavioral in nature.

3. The actual exposure of ESA-listed fish to harmful concentrations of aluminum and mixtures with other contaminants, and the duration of such exposures, is unpredictable. Furthermore, there is a large degree of variability in the effects that could occur as a result of these exposures.

4. There are no data available to help NMFS quantify impacts to foraging behavior or any changes to health of individual killer whales in the population from a specific amount of removal of potential prey resulting from the proposed criteria.

Because it is not practicable to quantify an amount of take, we will use a surrogate for take that is directly related to the potential for exposure to aluminum at criteria concentrations. Elevated aluminum concentrations are likely to be realized in areas where there are anthropogenic disturbances. As such, the number of point source discharges of aluminum to waters supporting ESA-listed species and/or critical habitat will serve as a surrogate. Currently, there are 73 potential dischargers of aluminum (either direct dischargers to surface water or indirect dischargers through sewage treatment plants). There are also a number of municipalities that are authorized to discharge stormwater to streams supporting ESA-listed species and/or designated critical habitat. For the sake of this ITS, we will assume that an additional 27 new point source discharges may be permitted in the future. As such our estimate of the extent of take for EPA's promulgation of aquatic life criteria for aluminum is 100 permitted point source discharges of aluminum, spread throughout the action area.

2.9.2 Effect of the Take

In the Opinion, NMFS determined that the amount or extent of anticipated take, coupled with other effects of the proposed action, is not likely to result in jeopardy to the species or destruction or adverse modification of critical habitat.

2.9.3 Reasonable and Prudent Measures

"Reasonable and prudent measures" are nondiscretionary measures that are necessary or appropriate to minimize the impact of the amount or extent of incidental take (50 CFR 402.02). NMFS believes the RPMs described below are necessary and appropriate to minimize the likelihood of incidental take of ESA-listed species due to implementation of the proposed action.

The EPA shall:

1. Minimize the potential for adverse effects associated with exposures to the promulgated aluminum criteria; and

2. Ensure completion of a monitoring and reporting program to confirm that the terms and conditions in the ITS are effective in avoiding and minimizing incidental take and ensuring the extent of incidental take is not exceeded.

2.9.4 Terms and Conditions

The terms and conditions described below are non-discretionary, and the EPA or any applicant must comply with them in order to implement the RPMs (50 CFR 402.14). The EPA or any applicant has a continuing duty to monitor the impacts of incidental take and must report the progress of the action and its impact on the species as specified in this ITS (50 CFR 402.14). If the EPA not comply with the following terms and conditions, protective coverage for the proposed action would likely lapse.

1. The following terms and conditions implement RPM 1:

    a. EPA shall review five draft state-issued NPDES permits for facilities that are known to discharge aluminum. While particular emphasis shall be placed on facilities with the greatest potential to discharge aluminum in harmful concentrations (i.e., industrial discharges or discharges from aluminum anodizing facilities), EPA shall work with ODEQ such that a variety of permit types (e.g., drinking water treatment facilities, POTWs, and stormwater facilities) are included in the review. Particular attention shall be placed on reviewing whether the most toxic conditions for aluminum are characterized appropriately; mixing zone authorizations are adequately minimized; and requirements for effluent and instream monitoring along with WET testing are adequate.

2. The following terms and conditions implement RPM 2:

    a. EPA will notify NMFS by email that a review is being initiated once EPA selects a draft permit to review in accordance with term and condition 1.a above. EPA will provide a written summary of the review to NMFS within six months of completing each review.[7] NMFS and EPA may agree to an extension of this timeframe if multiple permits are being reviewed within a given year.

    b. EPA will coordinate with NMFS to develop a list of facilities discharging aluminum to waters supporting ESA-listed species and/or designated critical habitats. This list shall include the following information: facility name, permit number, discharge location (latitude and longitude), receiving stream name, , aluminum effluent limits and number of effluent limit exceedances, and size of

[7] [7] If EPA provides written comments (that relate to aluminum) to ODEQ on a draft permit, sharing a courtesy copy with NMFS would satisfy the written summary requirement.

the authorized mixing zone. In the event that information is not available, the documentation will reflect this finding. This list will be submitted to NMFS after every Permit Quality Review cycle for Oregon. EPA and NMFS will revisit this reporting requirement after two Permit Quality Review cycles.

c. The reporting requirements described in term and conditions 2.a and 2.b above shall be submitted electronically to NMFS at nmfswcr.srbo@noaa.gov.

2.10 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. Specifically, conservation recommendations are suggestions 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 recommendations should be carried out by the EPA to achieve these purposes:

1. Given the paucity of toxicity testing on rainbow trout or other more closely related species in alkaline waters, EPA should fund or conduct additional aluminum toxicity testing to examine the potential for lethal and sublethal effects from chronic exposure to aluminum in alkaline waters. Particularly, EPA should examine whether freshwater aluminum exposures manifests in reduced survival of anadromous salmonids in marine environments. These tests should be sufficiently designed for assessing the protectiveness of the chronic criterion.

2. To improve the potential for recovery of ESA-listed species in the State of Oregon, the EPA should carry out management actions to reverse threats to survival as identified in the recovery plans for each species.

3. The EPA should carry out management actions (e.g., update technology-based treatment requirements as appropriate) that ensure point source discharges are employing the most effective treatment technologies available.

4. The EPA should collaborate with Federal and state agencies to ensure the most effective best management practices to reduce and treat stormwater runoff from nonpoint sources of pollution (e.g., agriculture, forestry, and development) are implemented.

5. The EPA should revise its 1985 *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and The Uses* to reflect more recent scientific advancements in the fields of ecotoxicology and salmon biology. As part of this effort, the EPA should collaborate with NMFS scientists to ensure the most sensitive and relevant toxicological endpoints (e.g., behavioral effects, olfaction, etc.), assessment methodologies, and effects thresholds are incorporated into the criteria development procedures. As part of this process, EPA should consider whether and how to incorporate temperature effects on species susceptibility to contaminant toxicity in the criteria derivation procedures.

6. The EPA should work with the State of Oregon to develop a monitoring protocol for toxic pollutants that establishes a consistent monitoring program across the state, and is designed to measure, in real-time, whether or not a particular point-source discharger is in compliance with the aquatic life criteria.

7. The EPA should work with the State of Oregon to minimize effects from chemical mixtures and decrease mixing zone dimensions such that no mixing zones overlap in space and time, or impact more than 5 percent of the cross-sectional area of the affected waterbody, and are calculated using the "one-day, once in ten year low flow" (1Q10) statistic or its equivalent.

8. The EPA should continue to explore and, where appropriate, modify EPA-approved testing methodologies that more aptly characterize the fraction of total aluminum that is bioavailable and that can exert toxic effects.

2.11 Reinitiation of Consultation

This concludes formal consultation for EPA's proposed promulgation of freshwater aquatic life criteria for aluminum in Oregon.

As 50 CFR 402.16 states, reinitiation of consultation is required and shall be requested by the federal agency or by the NMFS where discretionary federal agency involvement or control over the action has been retained or is authorized by law and if: (1) The amount or extent of incidental taking specified in the ITS is exceeded; (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion; (3) the identified action is subsequently modified in a manner that causes an effect to the listed species or critical habitat that was not considered in the biological opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action.

3. MAGNUSON-STEVENS FISHERY CONSERVATION AND MANAGEMENT ACT ESSENTIAL FISH HABITAT RESPONSE

Section 305(b) of the MSA directs federal agencies to consult with NMFS on all actions or proposed actions that may adversely affect EFH. The MSA (Section 3) defines EFH as "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity." Adverse effect means any impact that reduces quality or quantity of EFH, and may include direct or indirect physical, chemical, or biological alteration of the waters or substrate and loss of (or injury to) benthic organisms, prey species and their habitat, and other ecosystem components, if such modifications reduce the quality or quantity of EFH. Adverse effects on EFH may result from actions occurring within EFH or outside of it and may include site-specific or EFH-wide impacts, including individual, cumulative, or synergistic consequences of actions (50 CFR 600.810). Section 305(b) also requires NMFS to recommend measures that can be taken by the action agency to conserve EFH.

This analysis is based, in part, on the EFH assessment provided by the EPA and descriptions of EFH Pacific Coast groundfish (PFMC 2005), coastal pelagic species (PFMC 1998), and Pacific Coast salmon (PFMC 2014) contained in the fishery management plans developed by the PFMC and approved by the Secretary of Commerce.

3.1 Essential Fish Habitat Affected by the Project

The proposed action and action area for this consultation are described in Sections 1.3 and 2.3, of this document, respectively. The action area includes areas designated as EFH for various life-history stages of Chinook and coho salmon, groundfish, and coastal pelagic species. Because the action affects all freshwater habitats, including estuaries in Oregon, the action area includes all of the following habitat areas of particular concern (HAPCs) for salmon: complex channel and floodplain habitat, spawning habitat, thermal refugia, estuaries, and submerged aquatic vegetation. The action area also contains the estuarine and seagrass HAPCs for groundfish and coastal pelagic species.

3.2 Adverse Effects on Essential Fish Habitat

Based on information provided by the action agency and the analysis of effects presented in the ESA portion of this document, NMFS concludes that the proposed action will have adverse effects on EFH designated for Pacific Coast salmon in freshwater. Pacific salmon, groundfish and coastal pelagic species will also be adversely affected in estuaries, including estuarine areas designated as HAPCs in the LCR and at other river mouths and estuaries.

1. <u>Water Quality (spawning, rearing, and migration).</u> EPA's proposed approval of freshwater aquatic life criteria for aluminum establishes allowable concentrations of aluminum, depending upon site chemistry (e.g., pH, total hardness, and DOC). As described in Section 2.5 of this document, the proposed allowable concentrations could lead to some toxicity to aquatic organisms, and thus interfere with the habitats ability to fully support spawning, rearing, and migration. However, it is unlikely that water quality will be degraded by high concentrations of aluminum throughout EFH all of the time. Rather, these effects are anticipated to occur in localized areas and will most likely be influenced by human activities (point source discharges of aluminum or aluminum contributions associated with stormwater runoff from agricultural or urban areas). Based on available information, implementation of CWA programs by the ODEQ will be done in a manner that minimizes the potential for these potential adverse effects.

2. <u>Forage (rearing and migration).</u> Prey organisms may experience adverse effects when exposed to aluminum at proposed criteria concentrations. Thus forage may be impacted in small, localized areas, as described in Section 2.5.4.3.

3.3 Essential Fish Habitat Conservation Recommendations

Because the properties of EFH that are necessary for the spawning, breeding, feeding, or growth to maturity of managed species in the action area are the same or similar to the biological requirements of ESA-listed species as analyzed above, NMFS has provided three conservation recommendations. The following conservation recommendations are necessary to avoid, mitigate, or offset the impact of the proposed action on EFH:

1. EPA should review at least five state-issued NPDES permits for facilities that discharge aluminum. While particular emphasis should be placed on facilities with the greatest potential to discharge aluminum in harmful concentrations (i.e., industrial discharges or discharges from aluminum anodizing facilities), EPA should ensure that a variety of permit types (e.g., drinking water treatment facilities, POTWs, and municipal separate storm sewer systems) are included in the review. Particular attention should be placed on ensuring the most toxic conditions for aluminum are characterized appropriately; mixing zone authorizations are adequately minimized; and requirements for effluent and instream monitoring along with WET toxicity are adequate.

2. The EPA should carry out management actions (e.g., update technology-based treatment requirements as appropriate) that ensure point source discharges are employing the most effective treatment technologies available.

3. The EPA should collaborate with Federal and state agencies to ensure the most effective best management practices to reduce and treat stormwater runoff from nonpoint sources of pollution (e.g., agriculture, forestry, and development) are implemented.

Fully implementing these EFH conservation recommendations would protect, by avoiding or minimizing the adverse effects described in Section 3.2, above, all of the designated EFH for Pacific Coast salmon, Pacific Coast groundfish, and coastal pelagic species in Oregon.

3.4 Statutory Response Requirement

As required by section 305(b)(4)(B) of the MSA, the EPA must provide a detailed response in writing to NMFS within 30 days after receiving an EFH Conservation Recommendation. Such a response must be provided at least 10 days prior to final approval of the action if the response is inconsistent with any of NMFS' EFH Conservation Recommendations unless NMFS and the federal agency have agreed to use alternative timeframes for the federal agency response. The response must include a description of measures proposed by the agency for avoiding, minimizing, mitigating, or otherwise offsetting the impact of the activity on EFH. In the case of a response that is inconsistent with the Conservation Recommendations, the federal agency must explain its reasons for not following the recommendations, including the scientific justification for any disagreements with NMFS over the anticipated effects of the action and the measures needed to avoid, minimize, mitigate, or offset such effects (50 CFR 600.920(k)(1)).

In response to increased oversight of overall EFH program effectiveness by the Office of Management and Budget, NMFS established a quarterly reporting requirement to determine how many conservation recommendations are provided as part of each EFH consultation and how many are adopted by the action agency. Therefore, we ask that in your statutory reply to the EFH portion of this consultation, you clearly identify the number of conservation recommendations accepted.

3.5 Supplemental Consultation

The EPA must reinitiate EFH consultation with NMFS if the proposed action is substantially revised in a way that may adversely affect EFH, or if new information becomes available that affects the basis for NMFS' EFH Conservation Recommendations (50 CFR 600.920(l)).

4. DATA QUALITY ACT DOCUMENTATION AND PRE-DISSEMINATION REVIEW

The DQA specifies three components contributing to the quality of a document. They are utility, integrity, and objectivity. This section of the opinion addresses these DQA components, documents compliance with the DQA, and certifies that this Opinion has undergone pre-dissemination review.

4.1 Utility

Utility principally refers to ensuring that the information contained in this consultation is helpful, serviceable, and beneficial to the intended users. The intended user of this Opinion is the EPA. Other interested users could include the ODEQ or point source dischargers of aluminum. Individual copies of this Opinion were provided to the EPA. The document will be available within 2 weeks at the NOAA Library Institutional Repository [https://repository.library.noaa.gov/welcome]. The format and naming adheres to conventional standards for style.

4.2 Integrity

This consultation was completed on a computer system managed by NMFS in accordance with relevant information technology security policies and standards set out in Appendix III, 'Security of Automated Information Resources,' Office of Management and Budget Circular A-130; the Computer Security Act; and the Government Information Security Reform Act.

4.3 Objectivity

Information Product Category: Natural Resource Plan.

Standards: This consultation and supporting documents are clear, concise, complete, and unbiased; and were developed using commonly accepted scientific research methods. They adhere to published standards including the NMFS ESA Consultation Handbook, ESA regulations, 50 CFR 402.01 et seq., and the MSA implementing regulations regarding EFH, 50 CFR 600.

Best Available Information: This consultation and supporting documents use the best available information, as referenced in the References section. The analyses in this Opinion and EFH consultation contain more background on information sources and quality.

Referencing: All supporting materials, information, data and analyses are properly referenced, consistent with standard scientific referencing style.

Review Process: This consultation was drafted by NMFS staff with training in ESA and MSA implementation, and reviewed in accordance with West Coast Region ESA quality control and assurance processes.

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