Biological Evaluation of Freshwater Aluminum Water Quality Criteria for Oregon

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1. Background/History

1.1 Introduction

This Biological Evaluation (BE) prepared by the Region 10 U.S. Environmental Protection Agency (USEPA), addresses the proposed action in compliance with Section 7(c) of the Endangered Species Act (ESA) of 1973, as amended. Section 7 of the ESA assures that, through consultation (or conferencing for proposed species) with the National Marine Fisheries Service (NMFS) and/or the U.S. Fish and Wildlife Service (USFWS) (together, the "Services"), federal actions do not jeopardize the continued existence of any threatened, endangered or proposed species, or result in the destruction or adverse modification of critical habitat.

The federal action that is the subject of this biological evaluation is the USEPA's proposed rule to establish aluminum water quality criteria to protect aquatic life in freshwaters under the jurisdiction of the State of Oregon. The purpose of this BE is to assess the potential for effects of the proposed action on species listed as endangered or threatened under the ESA, and their designated critical habitat. This action has the potential to impact the ESA-listed species that occur in the action area, as provided in Table 1-1 and detailed more fully in Tables 2-x and 2-x.

Table 1-1. Listed Species residing in the geographic region in which the proposed action would occur.

Common Name	Species Name	Managing Service
Green sturgeon	*Acipenser medirostris*	NMFS
Eulachon	*Thaleichthys pacificus*	NMFS
Coho salmon	*Oncorhynchus kisutch*	NMFS
Chum salmon	*O. keta*	NMFS
Chinook salmon	*O. tshawytscha*	NMFS
Sockeye salmon	*O. nerka*	NMFS
Steelhead	*O. mykiss*	NMFS
Southern Resident Killer Whale	*Orcinus orca*	NMFS
Oregon spotted frog	*Rana pretiosa*	USFWS
Warner sucker	*Catostomus brevirostris*	USFWS
Shortnose sucker	*Chasmistes brevirostris*	USFWS
Lost River sucker	*Deltistes luxatus*	USFWS
Hutton tui chub	*Gila bicolor ssp.*	USFWS
Borax Lake chub	*Gila boraxobius*	USFWS
Lahontan cutthroat trout	*Oncorhynchus clarki henshawi*	USFWS
Foskett speckled dace	*Rhinichthys osculus ssp.*	USFWS
Bull trout	*Salvelinus confluentus*	USFWS
Vernal pool fairy shrimp	*Branchinecta lynchi*	USFWS

1.2 Organization of the Biological Evaluation

This biological evaluation follows the suggested format issued by the Services. The exception is the addition of a section (Section 8) to address Essential Fish Habitat:

Chapter 1.0 Background/History

Chapter 2.0 Proposed Action. Description of the Action and the Action Area. This section describes the federal rule for aluminum that USEPA is proposing (Proposed Action). The ESA-listed species within the action area for the BE are identified and those that could be affected by the proposed action (species of interest) are listed.

Chapter 3.0 Status of the Species. Listed Species and Critical Habitat in the Action Area. This section describes the species life-history, critical habitat, ESA listing history, current known range, and status for each of the ESA-listed fish species being considered.

Chapter 4.0 Environmental Baseline. This section describes effects of past and ongoing human and natural factors leading to the current status of the ESA-listed fish species in the State of Oregon, focusing on impacts related to land use, habitat, harvesting, hydropower, and hatcheries as well as current water quality.

Chapter 5.0 Effects of the Action. This section includes an analysis of the potential direct and indirect effects of the proposed action on the species and/or critical habitat and its interrelated and interdependent activities

Chapter 6.0 Effects Determinations, Cumulative Effects, Interdependent/Interrelated Actions. Cumulative Effects describes all "non-Federal" actions reasonably certain to occur in the foreseeable future. Includes state, local, private, and tribal actions (e.g. residential developments, watershed enhancement, etc.). Ongoing Environmental Effects includes ongoing environmental conditions and environmental cycles that could result in an additional stressor along with the Agency's Action. Uncertainty includes quantifiable and unquantifiable unknowns that may or may not be accountable for in the Agency's analysis of effects to the species.

Chapter 7.0 Conclusions. This section describes the USEPA's effects determination for each of the species addressed as well as critical habitat. The three possible effects determinations for each species are: 1) No Effect (NE); 2) May Affect, but Not Likely to Adversely Affect (NLAA); and 3) May Affect, and Likely to Adversely Affect (LAA).

Chapter 8.0 Essential Fish Habitat Analysis. In this section, Essential Fish Habitat (EFH) is assessed for potential adverse impacts resulting from the proposed action.

References

1.3 Consultation History

Table 1-2. Significant early coordination and pre-consultation activities with NMFS and USFWS

Date	Mode of communication	Participants	Topic
February 24, 2017	Letter	R10, NMFS	EPA notified NMFS of the upcoming AI rule and need for ESA consultation
August 9, 2017	Letter	R10, NMFS	NMFS notified EPA that staff had been assigned to the AI rule consultation
June 16, 2017	Teleconference	FWS, R10	EPA provided FWS with an update on the upcoming action
September 22, 2017	Teleconference	FWS, NMFS, R10, HQ	EPA introduced NMFS and FWS to the AI criterion, the AI rule; and, held a discussion of process and timing for the consultation
November 1, 2017	In-person meeting	FWS, NMFS, R10, HQ	Discussion among all participants of the proposed action in detail including EPA's effects analysis as it stood on Nov 1.
November 9, 2017	Teleconference	FWS, NMFS, R10, HQ	Follow-up discussion among all participants to address Services questions after having time to consider the content described on Nov 1st.
April 5, 2019	Teleconference	FWS, NMFS, R10, HQ	EPA update on AI criteria to Services, revised AI rule schedule and consultation timing.
May 22, 2019	Teleconference	FWS, NMFS, R10, HQ	EPA update Services on consultation schedule and analytical approach to the BE effects analysis
July 12, 2019	Teleconference	NOAA, R10	EPA update Services on BE effects analysis status
July 19, 2019	Teleconference	FWS, R10	EPA update Services on BE effects analysis status
August 12, 2019	Email	FWS, NMFS, R10	EPA transmitted Draft BE to Services
August 19, 2019	In-person meeting	FWS, NMFS, R10, HQ	Discussion among all participants of Draft BE and questions from the Services
August 19 – September 18, 2019	Series of emails	FWS, R10	Discussion among all participants of questions and comments from FWS regarding Draft BE
September 20, 2019	Letter and enclosure	FWS, NMFS, R10	EPA cover letter and BE shared with the Services
October 7, 2019	Letter of request	FWS	FWS request for supplementary data and analysis from EPA
October 15, 2019	Letter of request	NMFS	NMFS request for supplementary data and analysis from EPA

October 19, 2019	Letter and insufficiency determination	NMFS	BE insufficiency determination from NMFS
October 23, 2019	Letter and sufficiency determination	FWS	Email from FWS to EPA that the BE was sufficient to commence formal consultation
October 31, 2019	Teleconference	NMFS, R10, HQ	Discussion among all participants of NMFS insufficiency determination from 10/19/19
November 1, 2019	Teleconference	NMFS, R10, HQ	Discussion among all participants of NMFS insufficiency determination from 10/19/19
November 13, 2019	Teleconference	NMFS, R10, HQ	Discussion among all participants of NMFS insufficiency determination from 10/19/19
November 15, 2019	Teleconference	NMFS, R10, HQ	Discussion among all participants of NMFS insufficiency determination from 10/19/19
December 17, 2019	Email and phone call	NMFS, R10	NMFS email to EPA regarding a lack of complete information in the BE on designated critical habitat for Southern Oregon/Northern California Coho Salmon.
January 2, 2020	Letter and revised BE	EPA	Submission by EPA of revised BE to NMFS (cc to FWS)

On May 13, 2019 the USEPA transmitted a letter to NMFS and USFWS with a list of potentially affected listed species relevant to the Agency's Action. The EPA received responses back on May 22, 2019 and May 30, 2019 from the USFWS and NMFS, respectively, concurring with the USEPA's list of potentially affected listed species.

An interim draft Biological Evaluation was shared with the Services on August 12, 2019. Several email exchanges took place after the interim draft was reviewed by the Services. On October 19, 2019, NMFS replied to the EPA's Biological Evaluation with a determination of insufficiency, while on October 23, 2019 USFWS determined that the Biological Evaluation was sufficient.[1] On January 2, 2020, EPA submitted a revised BE to NMFS, with a courtesy copy to FWS.

[1] Personal communication from Jeffrey Dillon, USFWS, October 23, 2019.

2. Proposed Action

2.1 Overview of Water Quality Standards

A water quality standard (WQS) defines the water quality goals for a waterbody by designating the use or uses to be made of the water, by setting criteria necessary to protect the uses, and by preventing or limiting degradation of water quality through antidegradation provisions. The Clean Water Act (CWA) provides the statutory basis for the WQS program and defines broad water quality goals. For example, Section 101(a) states, in part, that wherever attainable, waters must achieve a level of quality that provides for the protection and propagation of fish, shellfish, and wildlife, and for recreation in and on the water (e.g., CWA "fishable/swimmable" goals).

CWA section 303(c) (33 U.S.C. 1313(c)) directs states to adopt WQS for their waters subject to the CWA. CWA section 303(c)(2)(A)[2] provides that WQS shall consist of designated uses of the waters and water quality criteria based on those uses. USEPA's regulations at 40 CFR 131.11(a)(1) provide that "[s]uch criteria must be based on sound scientific rationale and must contain sufficient parameters or constituents to protect the designated use [and] [f]or waters with multiple use designations, the criteria shall support the most sensitive use." In addition, 40 CFR 131.10(b) provides that "[i]n designating uses of a water body and the appropriate criteria for those uses, the [s]tate shall take into consideration the water quality standards of downstream waters and shall ensure that its water quality standards provide for the attainment and maintenance of the water quality standards of downstream waters."

States are required to review applicable WQS at least once every three years and, if appropriate, revise or adopt new WQS (CWA section 303(c)(1)[3] and 40 CFR 131.20). Any new or revised WQS must be submitted to USEPA for review and approval or disapproval (CWA section 303(c)(2)(A) and (c)(3)[4] and 40 CFR 131.20 and 131.21). USEPA reviews the standards to determine whether the analyses performed to support the revisions are scientifically defensible, and the criteria are protective of applicable designated uses. USEPA evaluates whether the standards meet the requirements of the CWA and USEPA's water quality standards regulations.

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[2] CWA section 303(c)(2)(A): Whenever the State revises or adopts a new standard, such revised or new standard shall be submitted to the Administrator. Such revised or new water quality standard shall consist of the designated uses of the navigable waters involved and the water quality criteria for such waters based upon such uses. Such standards shall be such as to protect the public health or welfare, enhance the quality of water and serve the purposes of this chapter. Such standards shall be established taking into consideration their use and value for public water supplies, propagation of fish and wildlife, recreational purposes, and agricultural, industrial, and other purposes, and also taking into consideration their use and value for navigation.

[3] CWA section 303(c)(1): The Governor of a State or the state water pollution control agency of such State shall from time to time (but at least once each three-year period beginning with October 18, 1972) hold public hearings for the purpose of reviewing applicable water quality standards and, as appropriate, modifying and adopting standards. Results of such review shall be made available to the Administrator.

[4] CWA section 303(c)(3): If the Administrator, within sixty days after the date of submission of the revised or new standard, determines that such standard meets the requirements of this chapter, such standard shall thereafter be the water quality standard for the applicable waters of that State. If the Administrator determines that any such revised or new standard is not consistent with the applicable requirements of this chapter, he shall not later than the ninetieth day after the date of submission of such standard notify the State and specify the changes to meet such requirements. If such changes are not adopted by the State within ninety days after the date of notification, the Administrator shall promulgate such standard pursuant to paragraph (4) of this subsection.

USEPA then formally notifies the state of these results. If USEPA finds that any such revised or new water quality standard is not consistent with the applicable requirements of the CWA and disapproves a state's new or revised WQS, USEPA is required to specify the disapproved portions and the changes needed to meet the CWA requirements. The CWA provides the state 90 days to adopt a revised WQS that meets CWA requirements, and if it fails to do so, the Agency shall promptly propose and then within 90 days promulgate such WQS unless the Agency approves a state replacement WQS first (CWA section 303(c)(3) and (c)(4)).[5]

Under CWA section 304(a), USEPA periodically publishes criteria recommendations for states to consider when adopting water quality criteria for particular pollutants to meet the CWA section 101(a)(2) goals. Where USEPA has published recommended criteria, states should establish numeric water quality criteria based on the Agency's CWA section 304(a) recommended criteria, CWA section 304(a) recommended criteria modified to reflect site-specific conditions, or other scientifically defensible methods (40 CFR 131.11(b)(1)). In all cases criteria must be sufficient to protect the designated use and be based on sound scientific rationale (40 CFR 131.11(a)(1)).

2.2 Background on Aluminum Aquatic Life Criteria in Oregon and Consultation History

On July 8, 2004, Oregon submitted 89 revised aquatic life criteria for 25 pollutants to USEPA for review under CWA section 303(c) including acute and chronic criteria for aluminum. Many of Oregon's revised criteria were the same as USEPA's national recommended CWA section 304(a) aquatic life criteria at the time. Oregon subsequently submitted revised WQS to the EPA for CWA section 303(c) review on April 23, 2007. USEPA did not take CWA section 303(c) action to approve or disapprove within the statutorily mandated timeline (CWA 303(c)(3)). On May 29, 2008, the U.S. District Court for the District of Oregon entered a consent decree setting deadlines for the EPA to take action under section 303(c) of the CWA on Oregon's July 8, 2004, submission of aquatic life criteria (Northwest Environmental Advocates v. U.S. EPA, No. 06-479-HA (D. Or. 2006)). On November 27, 2012, the District Court issued an extension of the applicable deadlines for the EPA's CWA section 303(c) action and amended the decree to require the Agency to act by January 31, 2013, on Oregon's July 8, 2004 submission of aquatic life criteria, as amended by subsequent submissions by Oregon dated April 23, 2007 and July 21, 2011.

USEPA initially considered approving Oregon's aluminum criteria. Prior to taking a final action on the aquatic life criteria, however, USEPA requested formal consultation with the National Marine Fisheries Service (NMFS) and the U.S. Fish and Wildlife Service (USFWS) on its proposed approval of the State's criteria, consistent with section 7(a)(2) of the ESA. USEPA initiated this consultation on January 14, 2008, by submitting a biological evaluation to NMFS and USFWS, which contained an analysis of the potential effects of the Agency's proposed

[5] CWA section 303(c)(4): The Administrator shall promptly prepare and publish proposed regulations setting forth a revised or new water quality standard for the navigable waters involved – (A) if a revised or new water quality standard submitted by such State under paragraph (3) of this subsection for such waters is determined by the Administrator not to be consistent with the applicable requirements of this Act…The Administrator shall promulgate any revised or new standard. . . not later than ninety days after he publishes such proposed standards, unless prior to such promulgation, such State has adopted a revised or new water quality standard which the Administrator determines to be in accordance with this chapter."

approval of Oregon's criteria, including criteria for aluminum, on threatened and endangered species in Oregon.

Before receiving a biological opinion from NMFS or USFWS, USEPA realized that the Agency's initial understanding that Oregon's criteria were entirely equivalent to the Agency's 1988 CWA section 304(a) recommended criteria was incorrect. While the EPA's 1988 CWA section 304(a) recommended aluminum criteria "apply at pH values of 6.5–9.0," the Agency later (before concluding consultation) identified a footnote to Oregon's revised aluminum criteria table specifying that Oregon's aluminum criteria applied "to waters with pH values less than 6.6 and hardness values less than 12 mg/L (as CaCO3)." The State had not supplied a scientific rationale to justify the application of the criteria to pH values less than 6.6 and hardness values less than 12 mg/L. As a result, USEPA prepared to disapprove the aluminum criteria. USEPA sent a letter to NMFS and USFWS identifying this change. USFWS had already completed and transmitted its biological opinion to USEPA by that point and the Agency was therefore unable to withdraw the consultation request for aluminum. USFWS's biological opinion (provided to USEPA on July 31, 2012) found that the Agency's proposed approval of Oregon's aquatic life criteria (which at the time of the consultation, was based on the application of the aluminum criteria to waters with pH 6.5 – 9.0) would not jeopardize the continued existence of listed species or result in the destruction or adverse modification of designated critical habitat under USFWS jurisdiction.

NMFS had not yet transmitted its analysis to USEPA at that time, so the Agency sent a letter to NMFS withdrawing its request for consultation on Oregon's acute and chronic aluminum criteria. NMFS acknowledged USEPA's request to withdraw the aluminum criteria from consultation in the biological opinion; however, NMFS did not modify the document to exclude the acute and chronic aluminum criteria. On August 14, 2012, NMFS concluded in its biological opinion that seven of Oregon's revised freshwater criteria would jeopardize the continued existence of endangered species in Oregon for which NMFS was responsible, including acute and chronic aluminum (applied to waters with pH 6.5 - 9.0).[6] NMFS acknowledged USEPA's request to withdraw the aluminum criteria from consultation and indicated that it would await a further request from USEPA regarding USEPA's future actions on Oregon's aluminum criteria.

On January 31, 2013, USEPA disapproved several of the State's revised aquatic life criteria under CWA section 303(c). USEPA disapproved the State's aluminum criteria because the State had not supplied a scientific rationale for the conditions under which the criteria would apply. On April 20, 2015, USEPA 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 USEPA to address the disapproved aquatic life criteria by either approving replacement criteria submitted by Oregon or by proposing and promulgating federal criteria. The State and USEPA have addressed the disapprovals for five of the criteria subject to the consent

[6] In addition to acute and chronic aluminum, the other criteria were the freshwater criteria Oregon adopted to protect aquatic life from adverse acute and chronic effects from ammonia and copper, as well as the criterion to prevent adverse acute effects from cadmium.

decree,[7] but the State has not yet addressed USEPA's 2013 disapproval of its freshwater criteria for acute and chronic aluminum (the sixth and seventh of the disapproved criteria). For the freshwater aluminum criteria, the consent decree originally established deadlines for USEPA to propose federal criteria by December 15, 2017, and to take final action on the proposal by September 28, 2018. On December 5, 2017, the District Court granted an extension of the applicable deadlines for USEPA's proposal and final action. At that time, the consent decree required USEPA to propose federal criteria for the State 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 USEPA'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 USEPA will sign a notice of federal rulemaking proposing aluminum criteria for Oregon. The consent decree includes a force majeure clause relating to "circumstances outside the reasonable control of EPA [that] could delay compliance with the deadlines specified in this Consent Decree. Such circumstances include…a government shutdown." Due to the 35-day government shutdown that occurred between December 22, 2018, and January 25, 2019, the deadline for signing a rule proposal was adjusted to April 19, 2019. As a result, USEPA proposed freshwater acute and chronic criteria for aluminum in Oregon in this rule in accordance with CWA section 303(c)(3) and (c)(4) requirements on April 18, 2019, consistent with the schedule established in the consent decree. Public comment for the proposed rule closed on June 17, 2019.The consent decree also requires that by March 27, 2020, USEPA will either approve aluminum criteria submitted by Oregon or sign a notice of final rulemaking.

2.3 Description of USEPA's Proposed Federal Rulemaking to Promulgate Aluminum Aquatic Life Criteria for Oregon's Freshwaters

USEPA proposes to establish federal CWA aquatic life criteria for fresh waters under the state of Oregon's jurisdiction, to protect aquatic life from the effects of exposure to harmful levels of aluminum. The federal action that is the subject of this biological evaluation is USEPA's Federal Rule "Aquatic Life Criteria for Aluminum in Oregon" (84 FR 18454, Docket ID No. EPA-HQ-OW-2016-0694). 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 (Final Aquatic Life Ambient Water Quality Criteria for Aluminum 2018, EPA 822-R-18-001, as cited in 83 FR 65663), which were developed consistent with the EPA's Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses (referred to as the "Aquatic Life Guidelines").[8] These criteria apply to fresh waters and account for water chemistry characteristics that affect aluminum bioavailability and toxicity.

The final 2018 recommended national criteria are based upon Multiple Linear Regression (MLR) models for fish and invertebrate species that use pH, DOC, and total hardness to quantify the

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[7] For more information on how the State and USEPA proceeded with regard to the other parameters, the proposed rule for copper and cadmium and final rule for cadmium are included in the docket for this rule.

[8] USEPA. 1985. *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses*. U.S. Environmental Protection Agency, Office of Research and Development, Duluth, MN, Narragansett, RI, Corvallis, OR. PB85-227049. https://www.epa.gov/sites/production/files/2016-02/documents/guidelines-water-quality-criteria.pdf

effects of these water chemistry parameters on the bioavailability and resultant toxicity of aluminum to aquatic organisms. The MLR models are used to normalize the available toxicity data to accurately reflect the effects of the water chemistry (pH, DOC, total hardness) on the toxicity of aluminum to tested species. These normalized toxicity test data are then used in a criteria calculator to generate criteria for specific water chemistry conditions, the water-chemistry-condition-specific criterion maximum concentration (CMC) and criterion continuous concentration (CCC) outputs. The numeric outputs of the final 2018 recommended national criteria models for a given set of conditions will depend on the specific pH, DOC, and total hardness entered into the models. The model outputs (CMC and CCC) for a given set of input conditions are numeric values that would be protective for that set of input conditions and protective of applicable aquatic life designated uses.

USEPA's rule provides that the criteria calculator, which incorporates pH, DOC, and total hardness as input parameters, be used to calculate protective acute and chronic 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. 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 will be promulgated by EPA.

Table 2-1 lists the aquatic life criteria for aluminum that are the subject of this evaluation. The aquatic life criteria in Table 2-1 apply to all fresh waters in Oregon to protect the fish and aquatic life designated uses.

Table 2-1. Proposed Federal Aluminum Aquatic Life Criteria for Oregon Fresh Waters

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 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.

In the preamble to the Oregon proposed rule, the EPA identified several possible approaches to reconciling multiple outputs of the criteria calculator, excerpted below.

The EPA's proposed rule provides that the criteria calculator, which incorporates pH, DOC, and total hardness as input parameters, be used to calculate protective acute and chronic 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. 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. The EPA strongly recommends that the State develop implementation materials to outline its approach.

The EPA suggests three methods that the State could use to reconcile model outputs and calculate criteria values that will result in protection of aquatic life at a site. Alternatively, the State may use its own alternate methods to reconcile outputs to generate protective criteria values. The appropriate method for each circumstance will depend primarily on data availability.

With method one, 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.

When using method two, 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.

In method three, 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.

Implementation of the standards is key to changing the current condition; however, the action under consideration at this time is USEPA's promulgation of the aluminum criteria. As the state of Oregon completes total maximum daily loads (TMDLs) designed to meet the federally-promulgated criteria, issues or reissues National Pollutant Discharge Elimination System (NPDES) permits in conjunction with those TMDLs, the condition of impaired waters, and thus the environmental baseline, will improve.

2.4 Species in the Action Area

USEPA accessed the online website of the USFWS (IPaC), to identify federally listed (or proposed) threatened or endangered species found within freshwater ecosystems in the state of Oregon. USEPA identified, and USFWS confirmed, the aquatic and aquatic-dependent species and associated critical habitat under the purview of the USFWS that would be subject to evaluation during the ESA consultation on USEPA's proposed action to promulgate the aluminum water quality criteria applicable to Oregon surface freshwaters. USEPA does not propose to evaluate any fully terrestrial listed species during this consultation on water quality standards for the protection of aquatic life.

Table 2-2a. USFWS species that May be Affected by the proposed action

Species Name	Status (T=threatened; E=endangered)	Designated Critical Habitat in Action Area?
Oregon spotted frog (*Rana pretiosa*)	T	Y
Warner sucker (*Catostomus warnerensis*)	T	Y
Shortnose sucker (*Chasmistes brevirostris*)	E	Y
Lost River sucker (*Deltistes luxatus*)	E	Y
Hutton tui chub (*Gila bicolor ssp.*)	T	N
Borax Lake chub (*Gila boraxobius*)	E, Proposed for Delisting	Y
Lahontan cutthroat trout (*Oncorhynchus clarki henshawi*)	T	N
Foskett speckled dace (*Rhinichthys osculus ssp*.)	T, Proposed for Delisting	N
Bull trout (*Salvelinus confluentus*)	T	Y
Vernal pool fairy shrimp (*Branchinecta lynchi*)	T	Y

USEPA accessed the online website of the NMFS, West Coast Region, to identify federally listed (or proposed) threatened or endangered species found within freshwater ecosystems in the state of Oregon. USEPA identified, and NMFS confirmed, the aquatic and aquatic-dependent species and associated critical habitat under the purview of the NMFS that would be subject to evaluation during any ESA consultation on the USEPA's action on the promulgation of aluminum water quality criteria applicable to Oregon surface freshwaters. USEPA does not propose to evaluate any fully terrestrial listed species during this consultation on water quality standards for the protection of aquatic life.

Table 2-2b. NMFS species that May be Affected by the proposed action.

Species Name	DPS/ESU	Status (T=threatened; E=endangered)	Designated Critical Habitat in Action Area?
Green Sturgeon (*Acipenser medirostris*)	Southern DPS	T	Y
Eulachon (*Thaleichthys pacificus*)	Southern DPS	T	Y
Southern Resident Killer Whales (*Orcinus Orca*)	Puget Sound	E	N (Y)
Coho Salmon (*Oncorhynchus kisutch*)	Lower Columbia River	T	Y
	Oregon Coast	T	Y
	Southern Oregon/Northern California Coast	T	Y
Chum Salmon (*Oncorhynchus keta*)	Columbia River	T	Y
Chinook Salmon (*Oncorhynchus tshawytscha*)	Upper Columbia River, spring-run	E	Y
	Snake River spring/summer-run	T	Y
	Snake River Fall-run	T	Y
	Upper Willamette River	T	Y
	Lower Columbia	T	Y
Sockeye Salmon (*Oncorhynchus nerka*)	Snake River	E	Y
Steelhead (*Oncorhynchus mykiss*)	Upper Columbia River	T	Y
	Snake River Basin	T	Y
	Middle Columbia River	T	Y
	Upper Willamette River	T	Y
	Lower Columbia River	T	Y

2.4.1 Species with "No Effect" Determinations

The listed and proposed species that will not be the focus of this consultation, based on previous scoping meetings with the USFWS and the NMFS, are marine turtles, terrestrial mammals, amphibians, and plants. It was determined that these species would not be directly impacted by USEPA's federal promulgation of aquatic life criteria for aluminum and thus USEPA's action would not affect these species. In other words, USEPA made 'No Effect' determinations for the following list of species:

Table 2-3. No Effect determinations.

Agency	Species – common name	Species – scientific name	Reason for No Effect Determination
NOAA	Leatherback sea turtle	*Dermochelys coriacea*	These are marine organisms and thus will not be affected by the freshwater Al criterion.
	Green sea turtle	*Chelonia mydas*	
	Loggerhead sea turtle	*Caretta caretta*	
FWS	Columbian white-tailed deer	*Odocoileus virginianus leucurus*	Minimal contact with waters affected by the Al criterion
	Grey wolf	*Canis lupis*	
	Macdonald's rockcress	*Arabis macdonaldiana*	These are terrestrial plants and therefore not affected by the freshwater Al criterion
	Applegate's milk vetch	*Astragalus applegatei*	
	Golden Indian paintbrush	*Castilleja levisecta*	
	MacFarlane's four o'clock	*Mirabilis macfarlanei*	
	Western lily	*Lilium occidentale*	
	Nelson's checker-mallow	*Sidalcea nelsoniana*	
	Willamette daisy	*Erigeron decumbens var decumbens*	
	Rough popcorn flower	*Plagiobothrys hirtus*	
	Howell's spectacular thelypody	*Thelypodium howellii ssp. spectabilis*	
	Water Howellia	*Howellia aquatilis*	These are wetland species, a habitat that does not receive toxic discharges, and therefore will not be affected by the freshwater Al criterion
	Bradshaw's lomatium	*Lomatium bradshawii*	

2.5 Description of Action Area

The action area of this consultation consists of all fresh waters of the state of Oregon to protect the fish and aquatic life designated uses. As the criteria were established by way of toxicity tests conducted in freshwaters only and because data were insufficient to establish saltwater criteria, the criteria will be applied only in freshwaters. However, because the Oregon Department of Environmental Quality (ODEQ) has no saltwater Al criteria and it is ODEQ's policy to apply the most protective (or only) criteria in estuarine locations, ODEQ may choose to apply the freshwater criteria to estuarine waters if pH, DOC and hardness values are within the bounds of the criteria model (ODEQ 2019[9]). Saline waters in Oregon include estuarine and marine waters, and estuarine are shown in Figure 2-1, with the estuarine extent defined using the federal Coastal and Marine Ecological Classification Standard (CMECS).[10] Based on analyses conducted for the

[9] ODEQ 2019: Methodology for Oregon's 2018 Water Quality Report and List of Water Quality Limited Waters September 2019 https://www.oregon.gov/deq/FilterDocs/ir2018assessMethod.pdf : "DEQ follows EPA's recommendation to use saltwater criteria for marine waters where the salinity is equal to or greater than 10 parts per thousand (approximately equivalent to conductivity 20,000 uS/cm) and use the more stringent of freshwater or saltwater aquatic life criteria in estuarine waters where salinity is between 1 and 10 parts per thousand."

[10] Available at https://www.fgdc.gov/standards/projects/cmecs-folder/CMECS_Version_06-2012_FINAL.pdf

2018 nationally recommended aluminum criteria (USEPA, 2018), the freshwater aluminum criteria are expected to be protective in estuarine waters. The Al criteria are discussed in detail in Section 5.0 of this BE. USEPA's federal promulgation of aluminum criteria does not apply to, and thus the action area does not include, any waters within Indian Country.

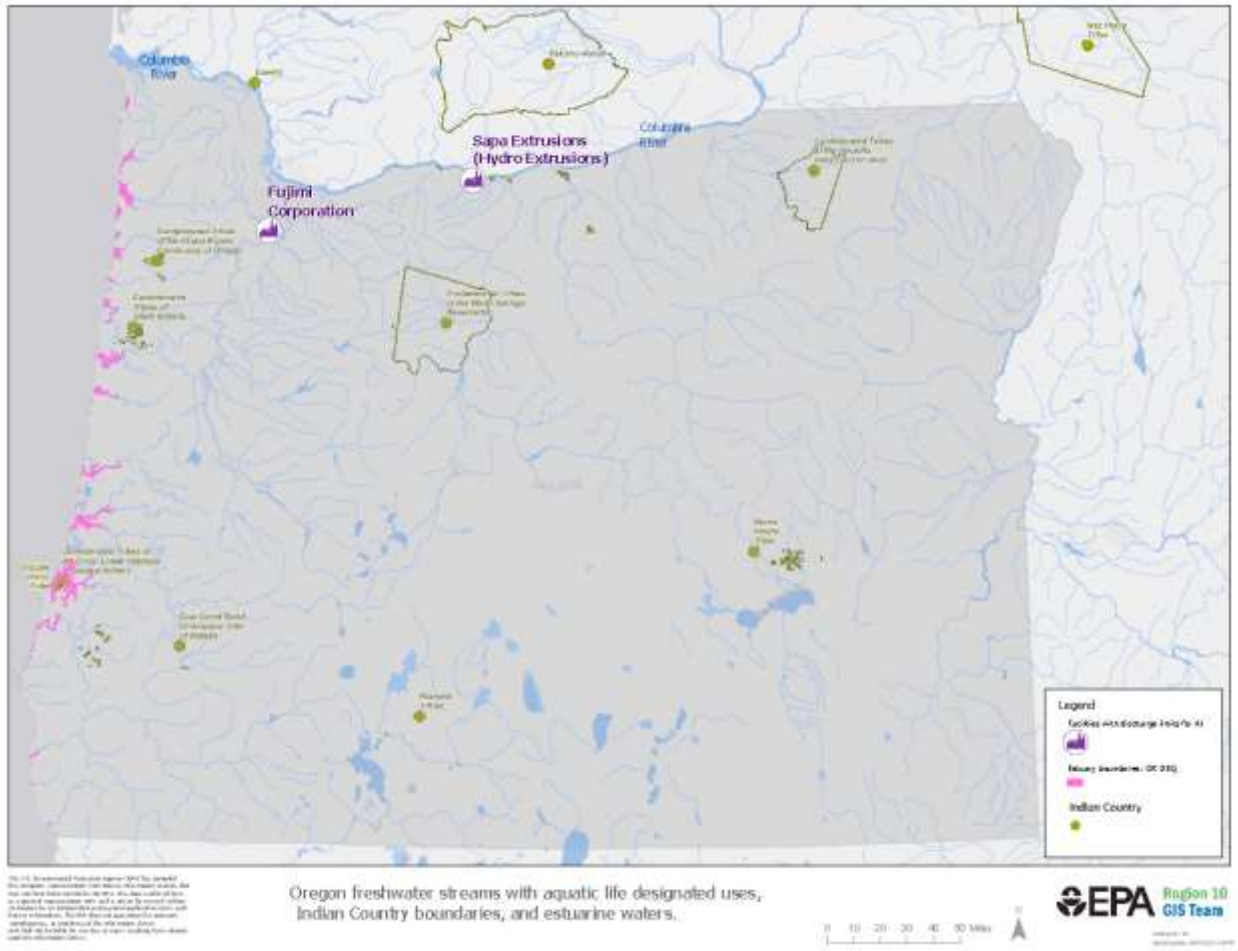


Figure 2-1. Map of action area (grey) showing Indian Country, freshwaters with aquatic life designated uses, and estuarine waters as well as two facilities with Al permit limits.

3. Status of the species

3.1 Green Sturgeon (*Acipenser medirostris*), Southern DPS

The southern DPS of green sturgeon was listed as threatened on April 6, 2005 (NMFS, 2006c) because the majority of spawning adults are concentrated in one spawning river (i.e., the Sacramento River) and at risk for extirpation due to catastrophic events. The ESA section 4(d) rule published by NMFS includes measures necessary to conserve the southern DPS of green sturgeon (NMFS, 2010b). The northern DPS of green sturgeon, which is not listed, spatially overlaps with the southern DPS, but is genetically distinct. Approximately 70% to 90% of the green sturgeon present in the Columbia River estuary and Willapa Bay are from the southern DPS, and 40% of green sturgeon in Grays Harbor are from the southern DPS (NMFS, 2015e).

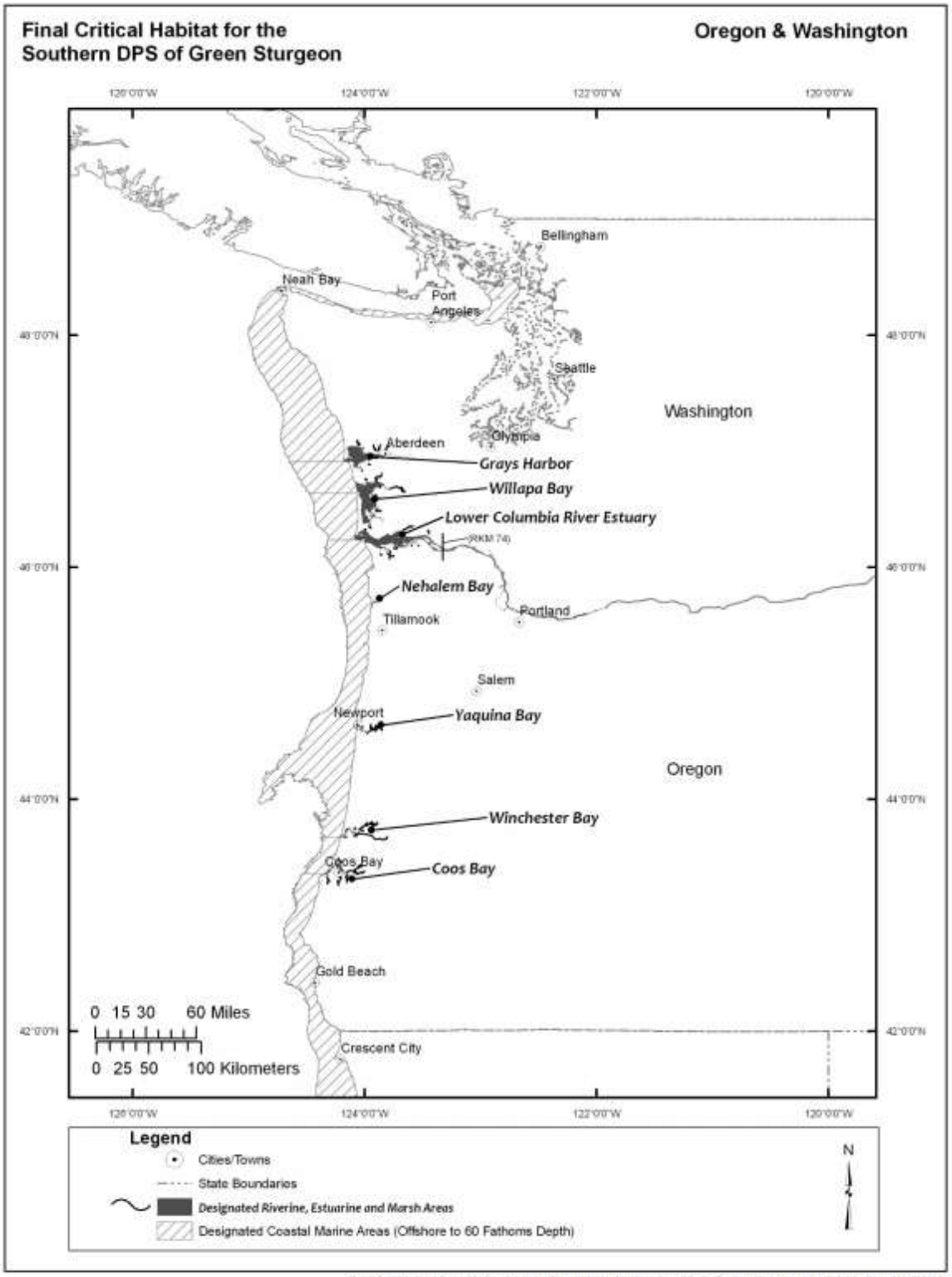
3.1.1 Distribution

Green sturgeon are distributed throughout the West Coast of North America (Colway & Stevenson, 2007; NMFS, 2015e; Rosales-Casian & Almeda-Jauregui, 2009), primarily north of Point Conception in California with seasonal (spring and winter) aggregation off Vancouver Island, British Columbia (NOAA Fisheries, 2016l). The only known spawning river for the entire DPS occurs in the Sacramento River in California (Poytress, Gruber, & Van Eenennaam, 2012), outside of the proposed action area.

Major areas of non-spawning aggregations in the proposed action area include Willapa Bay, Grays Harbor, and the Columbia River estuary (summer and fall), though these groups tend to be predominately composed of subadult sturgeon (WDFW & ODFW, 2002), as cited in (Lindley et al., 2011; Lindley et al., 2008; Moser & Lindley, 2007; NMFS, 2015e).

3.1.2 Critical Habitat

Critical habitat was designated for the southern DPS green sturgeon on October 9, 2009. Freshwater habitat was designated in the mainstem of the Sacramento River downstream of the Keswick Dam, in the Feather River below Oroville Dam, in the Yuba River below Dagueere Point Dam, and in the Sacramento-San Joaquin Delta. Marine critical habitat was designated in areas shallower than 110 m (361 feet) between Monterey Bay in California to the U.S.-Canada Border in Washington, including the following bays and estuaries: San Francisco, Humboldt, Coos, Winchester, Yaquina, and Newhalem Bays; Willapa and Grays Harbors; and the Lower Columbia River Estuary (up to river kilometer 74 [RM 46]). These critical habitat areas, where they overlap with the NW area, are shown in Figure 3-1.



Source: NOAA Fisheries (2009)

Figure 3-1. Critical Habitat for Southern DPS Green Sturgeon in Oregon and Washington

The following PBFs were identified for freshwater and estuarine green sturgeon habitats:

- Abundant prey for all life stages;
- Suitable substrates for egg deposition and development (e.g., bedrock sills, shelves, cobble or gravel, or hard clean sand; free of excessive siltation; availability of in-sediment voids for evading predators;
- Suitable flow regime to maintain normal behavior, growth, and survival of all life stages;
- Suitable water quality (i.e., temperature, salinity, dissolved oxygen, and "other chemical characteristics") for normal behavior, growth, and viability of all life stages;
- Migratory corridors that allow for safe and timely passage;
- Adequately deep holding pools (≥ 5 m (16.4 feet); and
- Suitable sediment quality for normal behavior, growth, and viability of all life stages.

The following PBFs were identified for coastal nearshore marine habitats:

- Migratory corridors that allow for safe and timely passage;
- Suitable water quality; and
- Abundant prey items (e.g., benthic invertebrates and fishes).

3.1.3 Life History

Green sturgeon are one of two West Coast sturgeon species. They are distinguished from white sturgeon by their greenish color, sharper but fewer scutes, a relatively elongated head, and a conspicuous stripe of color down their ventral side. They have an iteroparous, anadromous life history, typically spawning every three to four years in deep freshwater pools with coarse substrates (NMFS, 2015e). Sturgeon reach sexual maturity at approximately 15 years, and they can live for up to 70 years. Upon hatching, larval green sturgeon live on the bottom of rivers in coarse substrate, where they can avoid predators, absorb nutrients from their yolk sac, and grow into juveniles (NOAA Fisheries, 2016m). Larvae then disperse downstream, spending one to four years in their natal stream before migrating into estuaries and marine waters. Green sturgeon spend the majority of their lives in the ocean, migrating long distances (NOAA Fisheries, 2016m). However, sturgeon show strong stream fidelity when selecting a spawning river. Adults return in large numbers to Washington and Oregon estuaries during the summer and fall, and spawning migrations typically occur between April and June (NOAA Fisheries, 2016m). During the winter and spring, green sturgeon tend to migrate further north, where they form aggregations off Vancouver Island in British Columbia (NOAA Fisheries, 2016m).

Green sturgeon feed using an elongated mouth appendage that sucks food and sediment from the sediment surface (NOAA Fisheries, 2016l). Burrowing shrimp species (e.g., *Neotrypaea* spp.) are an important dietary component for subadult and adult green sturgeons, but they also eat fish (e.g., lingcod), crab (e.g., *Cancer* spp.), amphipods (e.g., *Anisogammarus* spp.), clams (e.g., *Cryptomya californica*), and polychaetes (Dumbauld, Holden, & Langness, 2008a). Predators of green sturgeon are not clear, but they may include pinnipeds (e.g., Steller sea lion [*Eumetopias jubatus*]), sharks (NMFS, 2015e), and humans (through poaching or bycatch) (Israel & May, 2007).

3.1.4 Current threats

The significant decline in green sturgeon that has occurred in the last century is primarily due to harvest pressure and the destruction of spawning habitat or migration corridors (NOAA Fisheries, 2016n). Since listing, the loss of freshwater habitat has remained a threat to the recovery of green sturgeon. Alterations to natural hydrology resulting from dams, channelization, sedimentation, and water withdrawal (for irrigation) are key culprits for degraded green sturgeon habitat (NOAA Fisheries, 2016n). Flow is an important factor for green sturgeon larval survival and to cue adult spawning migrations (NMFS, 2015e). Also, given the large number of spawning green sturgeon returning to the Sacramento River, genetic diversity may be relatively low. The impacts of invasive species, climate change, pesticide applications (i.e., carbaryl and imidacloprid), and future development are concerns (NMFS, 2015e), but more data on their effects on green sturgeon are needed.

3.2 Pacific Eulachon (*Ophiodon elongates*) Southern DPS

On March 16, 2010, the southern DPS of Pacific eulachon was listed as a threatened species (NMFS, 2010a).

3.2.1 Distribution

The southern eulachon DPS encompasses all populations within the states of Washington, Oregon, and California and extends from the Skeena River in British Columbia south to the Mad River in Northern California (inclusive).

Eulachon are endemic to the northeastern Pacific Ocean, ranging from northern California to southwest and south-central Alaska and into the southeastern Bering Sea. Puget Sound lies between two of the largest eulachon spawning rivers (the Columbia and Fraser Rivers) but lacks a regular eulachon run of its own (Gustafson, Ford, Teel, & Drake, 2010). Within the coterminous U.S., most eulachon production originates in the Columbia River Basin, and the largest and most consistent spawning runs return to the Columbia River mainstem and Cowlitz River. Adult eulachon have been found at several Washington and Oregon coastal locations, and they were previously common in Oregon’s Umpqua River and the Klamath River in northern California. Runs occasionally occur in many other rivers and streams, though often erratically, appearing in some years but not others and only rarely in some river systems (Gustafson et al., 2010; Hay & McCarter, 2000; Willson, Armstrong, Hermans, & Koski, 2006) Gustafson et al. 2010). Since 2005, eulachon in spawning condition have been observed nearly every year in the Elwha River by Lower Elwha Tribe fishery biologists. The Elwha is the only river within the U.S. portion of Puget Sound and the Strait of Juan de Fuca that supports a consistent run of eulachon.

3.2.2 Critical Habitat

Critical habitat was designated for the southern DPS of eulachon on October 20, 2011 (NMFS, 2011d). Sixteen specific areas were designated as critical habitat within the States of California, Oregon, and Washington. The designated areas are a combination of freshwater creeks and rivers and their associated estuaries, comprising approximately 539 km (335 miles) of habitat. No

marine areas were designated as critical habitat. Areas designated for critical habitat in Washington State include the Columbia (from the mouth to Bonneville Dam), Grays, Elochoman, Cowlitz, Toutle, Kalama, Lewis, Quinault, and Elwha Rivers and Skamokawa Creek. Lands of the Lower Elwha and Quinault Tribes are also excluded from critical habitat designation.

The PBFs essential to the conservation of the Pacific eulachon southern DPS were analyzed as three major categories reflecting key life history phases of eulachon. PBFs for freshwater spawning and incubation sites include water flow, quality, and temperature conditions; spawning and incubation substrates; and migratory access. PBFs for freshwater and estuarine migration corridors include waters free of obstruction; specific water flow, quality, and temperature conditions (for supporting larval and adult mobility); and abundant prey items (for supporting larval feeding after the yolk sac is depleted). The PBFs for marine nearshore and open water foraging habitat include suitable water quality and availability of prey.

3.2.3 Life History

Eulachon generally spawn in rivers fed by either glaciers or snowpack that experience spring freshets. Since these freshets rapidly move eulachon eggs and larvae to estuaries, it is believed that eulachon imprint and home to an estuary into which several rivers drain rather than individual spawning rivers (Hay & McCarter, 2000). Eulachon typically enter the Columbia River system from December to May, with peak entry and spawning during February and March (Gustafson et al., 2010). They spawn in the LCR mainstem and multiple tributaries of the LCR.

Eulachon eggs are commonly found attached to sand or pea-sized gravel, though eggs have been found on a variety of substrates, including silt, gravel-to-cobble sized rock, and organic detritus (Langer, Shepherd, & Vroom, 1977; Lewis, McGurk, & Galesloot, 2002; Smith & Saalfeld, 1955). Upon hatching, stream currents rapidly carry newly hatched larvae to the sea. Eulachon return to spawning rivers at ages ranging from two to five years as a single age class. Prior to entering their spawning rivers, eulachon hold in brackish waters while their bodies undergo physiological changes in preparation for freshwater and to synchronize their runs. Eulachon then enter rivers, move upstream, spawn, and die to complete their semelparous life cycle (COSEWIC, 2011a).

Eulachon are a short-lived, high-fecundity, high-mortality forage fish, and such species typically have extremely large population sizes. Fecundity estimates range from 7,000 to 60,000 eggs per female with egg-to-larva survival likely less than 1% (Gustafson et al., 2010). This may lead to recruitment events where only a small minority of spawning individuals contribute to subsequent generations (Hedgecock, 1994). Unlike other important forage fish species (e.g., Pacific herring), Columbia and Fraser River spawning stocks of Pacific eulachon appear to be limited to a single age class, which makes them vulnerable to environmental perturbations and catastrophic events (Gustafson et al., 2010).

Eulachon are an important link in the food chain between zooplankton and larger organisms. Small salmon, lingcod (*Ophiodon elongates*), white sturgeon (*Acipenser transmontanus*), and other fish feed on small larvae near river mouths. As eulachon mature, they are consumed by a wide variety of predators (Gustafson et al., 2010) (e.g., humpback whale).

3.2.4 Current Stressors and Threats

Climate change impacts on ocean habitat are the most serious threat to persistence of the southern DPS of Pacific eulachon (Gustafson et al., 2010). Physical changes associated with warming include increases in ocean temperature, increased stratification of the water column, and changes in the intensity and timing of coastal upwelling. These changes will alter primary and secondary productivity and the structure of marine communities (ISAB, 2007). In the marine environment, eulachon rely on cool ocean regions and the pelagic invertebrate communities therein (Willson et al., 2006). Warming ocean temperatures will likely alter these communities, making it more difficult for eulachon and their larvae to locate or capture prey (Roemmich & McGowan, 1995; Zamon & Welch, 2005). Warmer waters could also allow for the northward expansion of eulachon predator and competitor ranges, increasing the already high predation pressure on the species (McFarlane, King, & Beamish, 2000; Phillips et al., 2007; Rexstad & Pikitch, 1986). Decreased snowpack, increased peak flows, decreased base flow, changes in the timing and intensity of stream flows, and increased water temperatures may impact freshwater eulachon habitat (Morrison, Quick, & Foreman, 2002). In most rivers, eulachon typically spawn well before the spring freshet, near the seasonal flow minimum. Alterations to stream flow timing may cause eulachon to spawn earlier or be flushed out of spawning rivers at an earlier date. Early emigration may result in asynchrony between eulachon entering the marine environment and seasonal upwelling (Gustafson et al., 2010).

Historically, bycatch of eulachon in the pink shrimp fishery along the U.S. and Canadian coasts has been very high (composing up to 28% of the total catch by weight; (DFO, 2008; Hay & McCarter, 2000). Prior to the mandated use of bycatch-reduction devices in the pink shrimp fishery, 32% to 61% of the total catch in the pink shrimp fishery consisted of non-shrimp biomass, made up mostly of Pacific hake (*Merluccius productus*), various species of smelt, including Pacific eulachon, yellowtail rockfish (*Sebastes flavidus*), sablefish (*Anoplopoma fimbria*), and lingcod (Hannah & Jones, 2007). Bycatch of eulachon in these fisheries is still significant. The total estimated bycatch of eulachon in the Oregon and California pink shrimp fisheries ranged from 217,841 fish in 2004 to 1,008,260 fish in 2010 (Al-Humaidhi, Bellman, Jannot, & Majewski, 2010).

Hydroelectric dams block access to historical eulachon spawning grounds and affect the quality of spawning substrates through flow management, altered delivery of coarse sediments, and siltation. Dredging activities during the eulachon spawning run may entrain and kill adult and larval fish and eggs. Eulachon accumulate pollutants (USEPA, 2002), and although it has not been demonstrated that high contaminant loads in eulachon have increased mortality or reduced reproductive success, such effects have been shown in other fish species (e.g., (Kime, 1995)).

There are numerous activities that may affect the PBFs of Pacific eulachon critical habitat. Activities include dams and water diversions (e.g., Bonneville Dam); dredging and disposal of dredged material (i.e. on the Cowlitz and Columbia Rivers); in-water construction or alterations; contamination and runoff resulting in degraded habitat quality; tidal, wind, or wave energy projects; port and shipping terminals; and salmon habitat restoration projects, which benefit salmon to the detriment of species like Pacific eulachon. The activities may impact PBFs by altering stream hydrology; water level, flow, temperature and dissolved oxygen levels; erosion and sediment input/transport; physical habitat structure; vegetation; soils; nutrients and chemicals; fish passage; and estuarine/marine prey resources.

3.3 Coho Salmon

Coho salmon adults migrate to and spawn in small streams that flow directly into the ocean, or tributaries and headwater creeks of larger rivers (Peter B Moyle, 2002; P B Moyle, van Dyck, & Tomelleri, 2002; Sandercock, 1991). Adults migrate upstream to spawning grounds from September through late December, peaking in October and November. Spawning occurs mainly November through December, with fry emerging from the gravel in the spring, approximately three to four months after spawning. Juvenile rearing usually occurs in tributary streams with a gradient of 3% or less, although they may move up to streams of 4% or 5% gradient. Juveniles have been found in streams as small as 1 to 2 m (3.3 to 6.6 feet) wide. They may spend one to two years rearing in freshwater (Bell & Duffy, 2007), or emigrate to an estuary shortly after emerging from spawning gravels (Tschaplinski, 1988). With the onset of fall rains, coho salmon juveniles are also known to redistribute into non-natal rearing streams, lakes, or ponds, where they overwinter (N. P. Peterson, 1982). At a length of 38 to 45 millimeters (mm), fry may migrate upstream a considerable distance to reach lakes or other rearing areas (Nickelson, Rogers, Johnson, & Solazzi, 1992; Sandercock, 1991). Emigration from streams to the estuary and ocean generally takes place from March through June. The marine distribution of coho salmon extends from Alaska to California and west to Russia and Japan (NMFS, 2016e).

3.3.1 Lower Columbia River Coho Salmon

The LCR coho salmon ESU was listed as threatened pursuant to (NMFS, 2011c), and this determination was reaffirmed in 2011.

3.3.1.1 Distribution

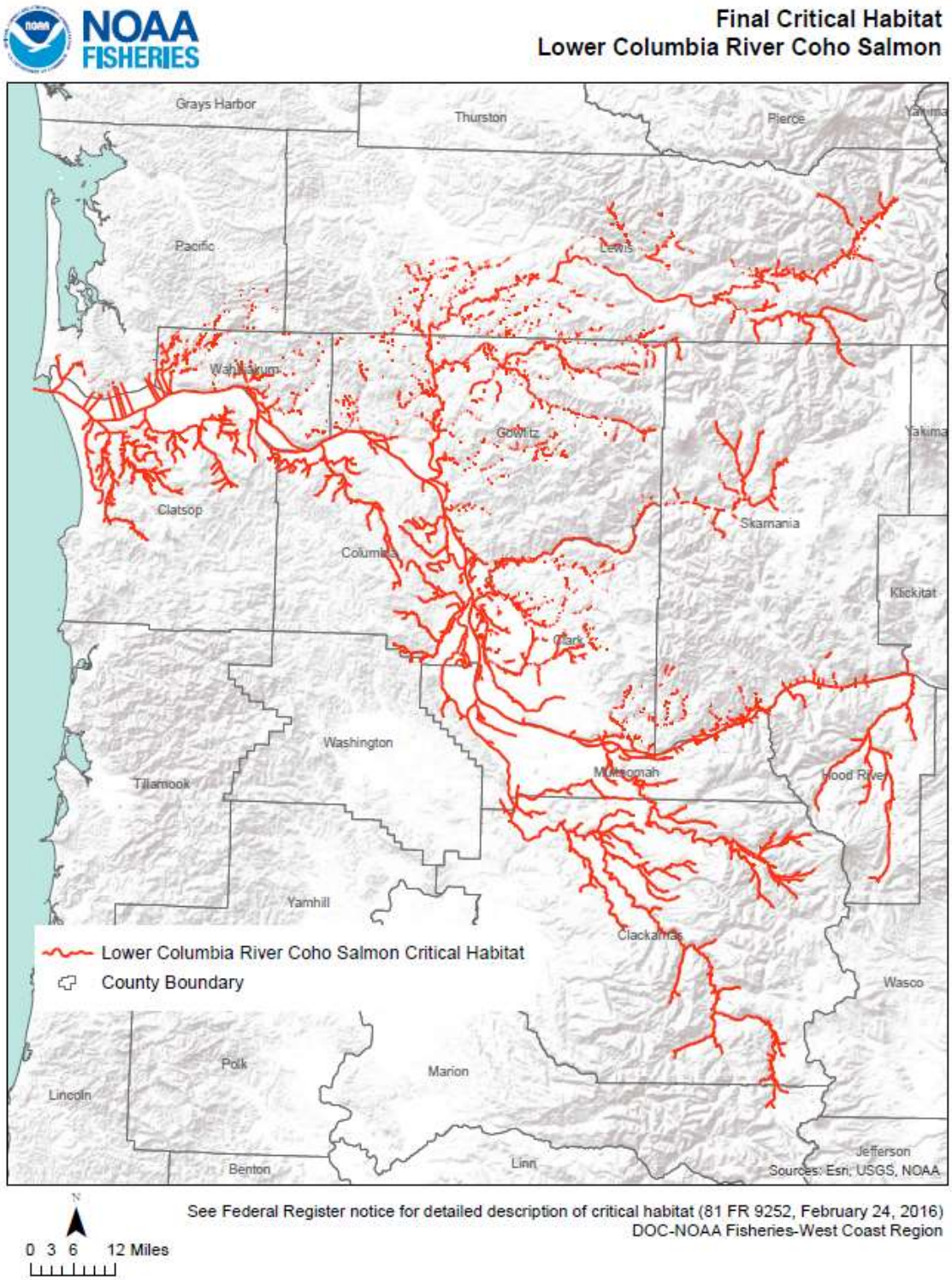
The LCR coho salmon ESU includes all naturally spawned coho salmon originating from the Columbia River and its tributaries downstream from the Big White Salmon and Hood Rivers (inclusive) and any such fish originating from the Willamette River and its tributaries below Willamette Falls. Also included in the ESU are coho salmon from 21 artificial propagation programs (NOAA Fisheries, 2014).

3.3.1.2 Critical Habitat

Critical habitat for LCR Coho salmon was designated for Oregon and Washington in 2016 and includes areas shown in Figure 3-2 (NMFS, 2016e). The PBFs for LCR coho salmon are the same as those for Chinook and chum salmon, and they are presented in (Table 3-1).

3.3.1.3 Life History

Adults typically spend approximately 18 months in freshwater streams and 18 months in marine waters before returning to natal streams to spawn as three-year olds (NMFS, 2015c). Two spawning groups have been identified, “type S” and “type N”; type S fish tend to enter rivers to spawn from mid-August to September and spawn in mid-October to early November, whereas type N fish enter rivers in late September to December and spawn between November and January (NMFS, 2015c). Type S fish tend to spawn higher in tributaries of the Columbia River, whereas type N fish spawn in lower tributaries.



Source: (NMFS, 2016a)

Figure 3-2. Critical Habitat for Lower Columbia River Coho Salmon

Table 3-1. Salmon and steelhead physical and biological features (PBFs) of critical habitats and corresponding species life history events.

PBFs for chum, coho, Chinook salmon, and sockeye salmon		
Site	**Site Attribute**	**Species Life History Event**
Spawning and juvenile rearing areas	Access (sockeye) Cover/shelter Food (juvenile rearing) Riparian vegetation Space (Chinook) Spawning gravel Water quality Water temperature (sockeye) Water quantity	Adult spawning Embryo incubation Alevin development Fry emergence Fry/parr growth and development Fry/parr smoltification Smolt growth and development
Juvenile migration corridors	Cover/shelter Food Riparian vegetation Safe passage Space Substrate Water quality Water quantity Water temperature Water velocity	Fry/parr seaward migration Smolt growth and development Smolt seaward migration
Adult migration corridors	Cover/shelter Riparian vegetation Safe passage Space) Substrate Water quality Water quantity Water temperature Water velocity	Adult sexual maturation Adult “reverse smoltification” Adult upstream migration Kelt (steelhead) seaward migration
PBFs for Steelhead		
Freshwater spawning	Spawning gravel /substrate Water quality Water quantity	Adult spawning Embryo incubation Alevin development
Freshwater rearing	Flood plain connectivity Forage Natural cover Water quality Water quantity	Fry emergence Fry/parr growth and development
Freshwater migration	Free of artificial obstructions Natural cover Water quality Water quantity	Adult sexual maturation Adult upstream migration Kelt (steelhead) seaward migration Fry/parr seaward migration

3.3.1.4 Current Stressors and Threats

Limiting factors for LCR coho salmon include migratory impediment caused by dams (i.e., Sediment Retention Structure on the North Fork Toutle River) (Fullerton et al., 2011); (NMFS, 2013); land development and habitat degradation; and potential effects from climate change and

coastal ocean conditions (e.g., reduced survival of emigrating smolts and corresponding drop in spawner abundance) (NOAA Fisheries, 2015b). Reduced complexity, connectivity, quantity, and quality of habitat used for spawning, rearing, foraging, and migrating are perhaps the most important limitations to LCR coho. Degradation or loss of habitat due to conversion to agricultural or urbanized uses (e.g., diking and draining of wetlands and floodplain), reduced complexity, and persistent inputs of wastewater, stormwater, and non-point-source runoff are key concerns (D L Bottom et al., 2005; LCFRB, 2010; Morace, 2012; Nilsen & Morace, 2014; NMFS, 2013; ODFW, 2010). Data collected by Ecology, the Oregon Department of Environmental Quality, and the Columbia River Contaminants and Habitat Characterization Project indicate that contaminants are present above levels of concern (Alvarez, Perkins, Nilsen, & Morace, 2014; Counihan, Hardiman, & Waste, 2013); (Nilsen & Morace, 2014); (Nilsen et al., 2014) and that Total Maximum Daily Loads are warranted for the LCR.

Per NOAA's 2016 status review, although populations in this ESU have generally improved, recent poor ocean conditions suggest that population declines might occur in the upcoming return years. The ESU is still considered to be at moderate risk (NOAA Fisheries, 2015b).

3.3.2 Oregon Coast Coho Salmon

The Oregon Coast coho salmon ESU was listed as threatened in 1998, but the listing was overturned by court decision in 2001. The ESU was re-listed as threatened in February 2008 (NMFS, 2008a), and that status was confirmed on June 20, 2011.

3.3.2.1 Distribution

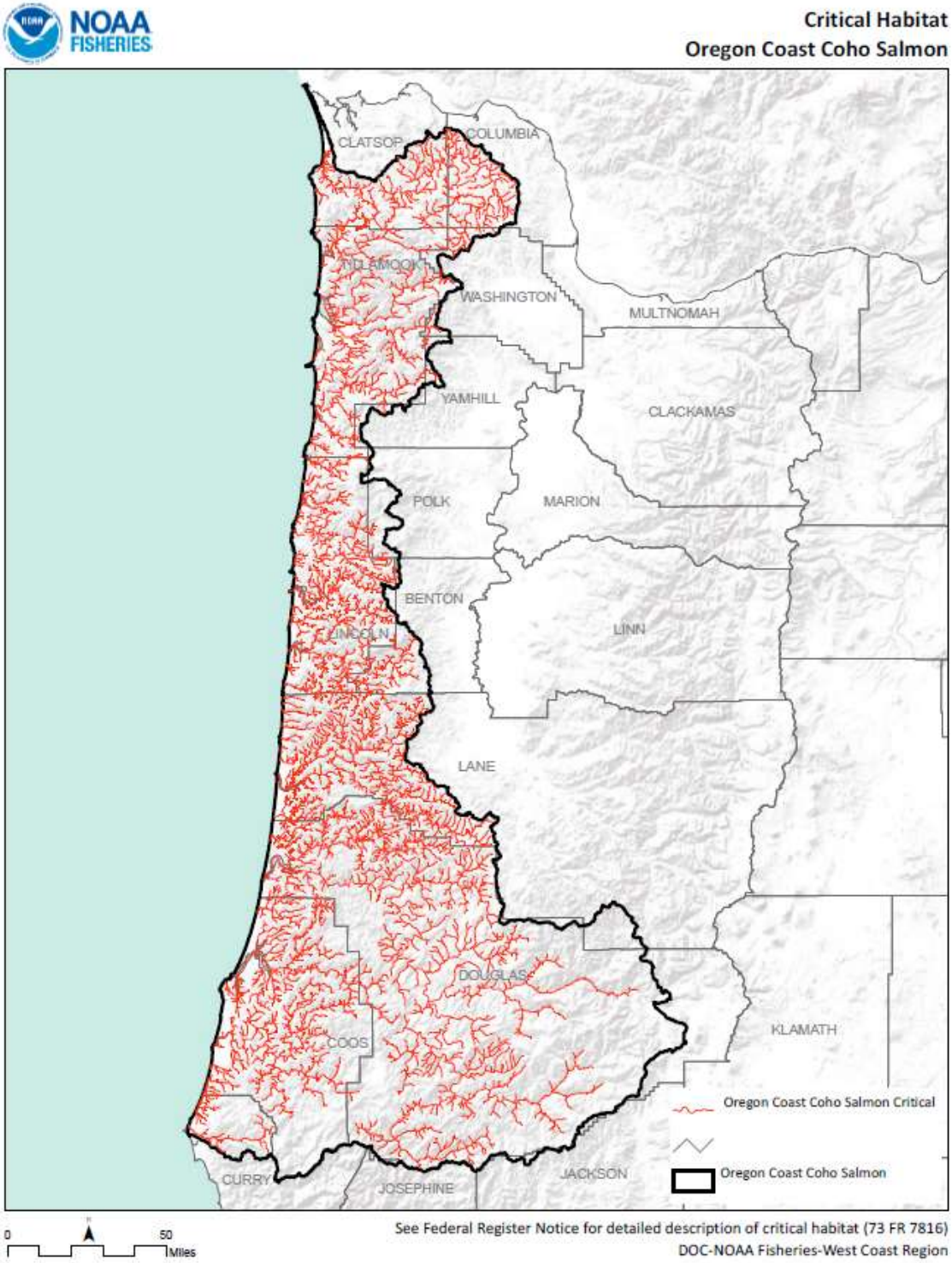
The Oregon Coast coho salmon ESU includes populations of coho salmon in Oregon coastal streams south of the Columbia River and north of Cape Blanco, Oregon. The Cow Creek stock (South Umpqua population) is included as part of the ESU because the original brood stock was founded from the local, non-hatchery population. Non-hatchery coho salmon from the South Umpqua population have been incorporated into the Cow Creek brood stock on a regular basis.

3.3.2.2 Critical Habitat

Critical habitat for OC coho salmon was designated for Oregon in 2008 and includes areas shown in Figure 3-3 (NMFS, 2008a). The PBFs for Oregon Coast coho salmon are listed in Table 3-1.

3.3.2.3 Life History

Coho salmon have a fixed, approximately three-year lifespan that includes approximately 15 months in freshwater rearing streams and 21 months in marine waters (as subadults and adults) (NMFS, 2015c). A small portion of "jacks" return to spawn after only six months at sea (i.e., two years of age) (NMFS, 2015c). Foraging occurs in the ocean in coastal and offshore North Pacific Ocean waters (NMFS, 2015c). Spawning migration starts in late summer and fall and ends by mid-winter (NMFS, 2015c).



Source: (NOAA Fisheries, 2016f)

Figure 3-3. Critical Habitat for Oregon Coast Coho Salmon

3.3.2.4 Current Stressors and Threats

The primary limiting factors in freshwater habitats for Oregon Coast coho salmon include reduced floodplain connectivity and function, reduced channel structure and complexity, degradation or loss of riparian habitat, reduced supply of large woody debris, altered stream substrate (e.g., increased embeddedness with fine sediment), altered stream flow, degraded water quality, installation or maintenance of barriers to migration (that limit access to spawning and rearing habitats) (NOAA Fisheries, 2008). In the marine environment, key limiting factors include adverse effects of climate, altered marine and estuarine productivity, and degradation of ocean ecosystem conditions (e.g., acidification) (NOAA, 2011; Stout et al., 2012). Per NOAA's 2016 status review (NOAA Fisheries, 2016f), no reclassifications for OC coho salmon ESU are appropriate, and the OC coho salmon ESU remains listed as threatened.

3.3.3 Southern Oregon/Northern California Coho Salmon

The Southern Oregon/Northern California (SONC) coho salmon ESU was federally listed as threatened in 1999 (NMFS, 1999d), and this listing was reaffirmed in 2005 (NMFS, 2005b).

3.3.3.1 Distribution

The SONC coho salmon ESU includes all naturally spawned populations of coho salmon in coastal streams from the Elk River in Oregon, through the Mattole River in California. It also includes three artificial propagation programs: the Cole Rivers Hatchery in the Rogue River Basin, and the Trinity and Iron Gate Hatcheries in the Klamath-Trinity River Basin. Recovery of tagged SONC coho salmon suggests that they generally travel from California to Oregon but do not migrate much farther north (NMFS, 2014).

3.3.3.2 Critical Habitat

Critical habitat for SONC coho salmon was designated for California and Oregon in 1999 and includes accessible reaches, estuarine areas, and tributaries between the Mattole and Elk Rivers (NMFS, 1999b).

Critical habitat was designated for SONC coho salmon on May 5, 1999 (NMFS, 1999b) and includes all river reaches accessible to listed coho salmon in coastal streams south of Cape Blanco, Oregon, and north of Punta Gorda, California. Critical habitat consists of the water, substrate, and adjacent riparian zone of estuarine and riverine reaches (including off-channel habitats) in the following counties: Klamath, Jackson, Douglas, Josephine, and Curry Counties in Oregon, and Humboldt, Mendocino, Trinity, Glenn, and Del Norte Counties in California. Major rivers, estuaries, and bays known to support coho salmon include the Rogue, Smith, Klamath, Mad, Eel, and Mattole Rivers and Humboldt Bay. Many smaller coastal rivers and streams also provide essential estuarine habitat for coho salmon, but access is often constrained by seasonal fluctuations in hydrologic conditions. Within these areas, PBF of coho salmon critical habitat include adequate substrate, water quality, water quantity, water temperature, water velocity, cover/shelter, food, riparian vegetation, space, and safe passage conditions.

The PBFs for SONC coho salmon are listed in Table 3-1.

3.3.3.3 Life History

SONC coho salmon tend to follow a stream-type life history, spending a year or two in their natal streams before emigrating to the ocean (NMFS, 2014). Downstream migration of SONC coho juveniles occurs between April and June, generally depending on the stream (NMFS, 2014). There are also ocean-type individuals that occur in the SONC coho ESU, which are found in streams with adequately large estuaries for rearing large numbers of juvenile fish. Emigration in ocean-type fish tends to coincide with heavy seasonal rains, though most fish use off-channel refuge habitat to avoid strong river flows (NMFS, 2014). Large woody debris is vital for this species because it helps to create refuges.

Returning SONC coho adults tend to be three years old, but some "jacks" return earlier (NMFS, 2014). Returning fish typically migrate upriver between October and March with peak migration occurring between November and January (NMFS, 2014).

3.3.3.4 Current Stressors and Threats

The primary threats to the SONC coho salmon ESU include timber harvest, road construction, damming and diversions, suction dredge mining, development in floodplains, agriculture, and hatchery inputs (C. D. Williams & Williams, 2002). Timber harvest, road construction, floodplain development, and agriculture all can increase sedimentation, water temperature (through sediment and clearing of riparian vegetation), pollutant loading, and other stressors as well as decrease the delivery of wood (and habitat complexity) and terrestrial nutrients (e.g., insects). Suction dredge mining has the effect of increasing scour in possible spawning habitats (C. D. Williams & Williams, 2002). Damming and diversion has the effect of impeding migration and altering hydrology, both of which can interrupt spawning or emigration and decrease water quality (e.g., temperature) and quantity. Hatchery inputs can reduce the overall genetic diversity of the population, making the ESU more prone to sudden collapse. As with other species, climate change has the potential to impact this species through increased temperatures and declining oceanic conditions (e.g., altered food webs).

Per NOAA's 2016 status review, no reclassifications for Southern Oregon/Northern California coho salmon ESU are appropriate, and the SO/NC coho salmon ESU remains listed as threatened.

3.4 Chum Salmon (*Oncorhynchus keta*)

Chum salmon are a species of anadromous salmonid that typically live for four years and grow to 6.8 kilograms (kg) (15 pounds) and can grow up 1.1 m (3.6 feet) long (NMFS, 2015a). They take on a characteristic greenish blue color that becomes striped with red slashes during spawning, and spawning adult males develop elongated "canine" teeth, which explains the colloquial name for this species, "dog salmon" (NMFS, 2015a). Chum spawn once before dying in freshwater streams.

Juvenile chum salmon quickly migrate into the marine environment after hatching, where, unlike other salmonids, they congregate in schools (NMFS, 2015a). The diet of chum salmon tends to shift from insects and other benthic invertebrates while in freshwater to crustaceans, fish, mollusks, squid, and tunicates while in the ocean (NMFS, 2015a).

The distribution of chum salmon in the marine environment is not well understood; however, it appears that chum salmon migrate as far north as Alaska, as far south as California, and as far west as Russia and Japan (Beamish & Bouillon, 1993).

3.4.1 Columbia River Chum Salmon

The Columbia River chum salmon ESU was listed as threatened pursuant to (NMFS, 2011c), and this determination was reaffirmed in 2011.

3.4.1.1 Distribution

The Columbia River chum salmon ESU includes naturally spawned chum salmon originating from the Columbia River and its tributaries in Washington and Oregon. Also, the ESU includes chum salmon from two artificial propagation programs, the Grays River Program and the Washougal River Hatchery/Duncan Creek Program (NOAA Fisheries, 2014). The Columbia River chum salmon ESU consists of 17 historical populations in three Major Population Groups (i.e., the Coastal, Cascade, and Gorge groups) (NMFS, 2013).

3.4.1.2 Critical Habitat

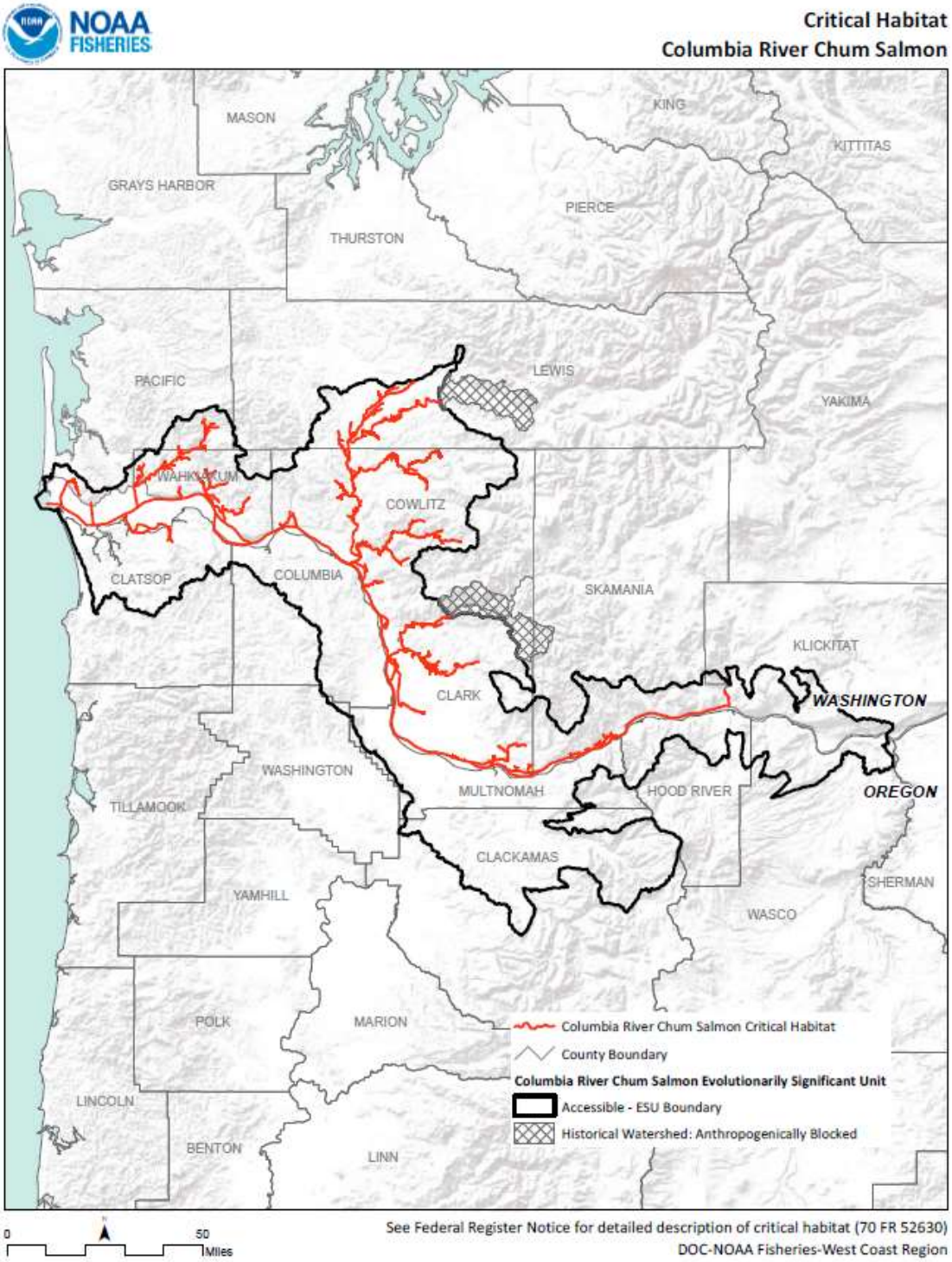
Critical habitat for Columbia River chum salmon was designated for Oregon and Washington in 2005 and includes areas shown in Figure 3-4 (NMFS, 2005c). The PBFs for Columbia River chum salmon are listed in Table 3-1.

3.4.1.3 Life History

Columbia River Chum salmon run in the fall, spawning in tributaries of the Columbia River below the Bonneville Dam (e.g., Grays River and Hardy and Hamilton Creeks) (NMFS, 2015a). Chum salmon fry emerge from spawning gravel and almost immediately drift downstream toward the ocean (NMFS, 2015a). Unlike other salmonids, chum salmon do not have a distinct smolt life stage. As subadults and adults, chum salmon feed in marine nearshore and open waters of the North Pacific Ocean. After three to five years, chum salmon return to their natal streams to spawn, typically between mid-October and early December (spawning until mid-January) in the mainstem Columbia River or in the lower portions of tributaries (NMFS, 2015a).

3.4.1.4 Current Stressors and Threats

Limiting factors for Columbia River chum salmon include poor ocean conditions in the near future (e.g., reduced or altered food web due to climate change and acidification) (limiting of juvenile survival) (NOAA Fisheries, 2015b), reduced freshwater habitat quality (limiting of spawning and early rearing success in some basins), and land development, especially in the low gradient reaches that chum salmon prefer. Based on projected increases in the population of the greater Vancouver-Portland area and the LCR overall (Metro, 2014), land development is expected to continue to limit the recovery of most chum salmon populations.



Source: (NOAA Fisheries, 2016b)

Figure 3-4. Critical Habitat for Columbia River Chum Salmon

Per NOAA's 2016 status review, the ESU remains at moderate to high risk (NOAA Fisheries, 2015b). For the CR chum salmon ESU, the majority of the populations in this ESU are at high to very high risk, with very low abundances (NOAA Fisheries, 2015b).

3.5 Chinook Salmon (*Oncorhynchus tshawytscha*)

Chinook salmon, also called king salmon, are the largest and least abundant species of Pacific salmon (NMFS, 2005a). Chinook salmon are anadromous, requiring both freshwater and saltwater to complete their life cycle. Juveniles generally spend three months to two years in freshwater before migrating to estuarine waters and eventually to sea, where they spend one to six years. Adults spend most of their lives in the ocean before migrating back to natal freshwater streams to spawn and subsequently die. Compared to other Pacific salmon species, Chinook prefer larger and deeper stream habitat (NMFS, 2005a). Juveniles feed on terrestrial and aquatic invertebrates, while subadults (i.e., post-smolt stage) and adults consume larger prey such as shrimp, squid, and small fish (e.g., herring [*Clupea* spp.] and sand lance [*Ammodytidae* spp.]) (Scott & Crossman, 1973). The distribution of Chinook salmon in the marine environment is not well characterized; however, they may be found as far north as Alaska, as far south as California, and as far west as Russia and Japan (NMFS, 2016c).

National Oceanic and Atmospheric Administration (NOAA) Fisheries recognizes six ESA-listed ESUs of Chinook salmon that spawn in Washington, Oregon, and Idaho: two Snake River ESUs were listed in April 1992 (NMFS, 1992a); the Upper Willamette River (UWR) ESU was listed in March 1999 (NMFS, 1999d); and the two Columbia River ESUs and single Puget Sound ESU were listed in August 1999 (NMFS, 1999e) . In 2005, NOAA published a scientific report entitled *Updated Status of Federally Listed ESUs of West Coast Salmon and Steelhead*, which includes an updated status of Chinook salmon (Good et al., 2005). The five-year status review completed in 2010 (NMFS, 2011c) concluded that all Chinook salmon ESUs should remain listed. Each ESU is treated as a separate species under the ESA (NMFS, 2011c). ESUs may include both naturally spawned and artificially propagated (hatchery stock) fish.

3.5.1 Snake River Fall-run Chinook salmon

The Snake River fall Chinook salmon ESU was listed as threatened on April 22, 1992 (NMFS, 1992a), and the threatened status was reaffirmed in 2005 (NMFS, 2005b).

3.5.1.1 Distribution

The Snake River Chinook salmon ESU occupies the Snake River basin, which drains portions of southeastern Washington, northeastern Oregon, and north/ central Idaho. The Snake River fall Chinook salmon ESU includes one extant population of fish spawning in the mainstem of the Snake River and the lower reaches of several 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 (NMFS, 2005b). Historically, this ESU also included a large population that spawned in the mainstem of the Snake River upstream of the Hells Canyon Dam complex, which is currently an impassable barrier to migration (NOAA Fisheries, 2015a).

3.5.1.2 Critical Habitat

Critical habitat for Snake River fall-run Chinook salmon was designated in 1993 and includes reaches of the Columbia, Snake, and Salmon Rivers and passable tributaries of the Snake and Salmon Rivers (NMFS, 1993). The geographic extent of critical habitat is the Snake River to Hells Canyon Dam; Palouse River from its confluence with the Snake River upstream to Palouse Falls; Clearwater River from its confluence with the Snake River upstream to Lolo Creek; North Fork Clearwater River from its confluence with the Clearwater River upstream to Dworshak Dam; and all other river reaches presently or historically accessible within the Lower Clearwater, Hells Canyon, Imnaha, Lower Grande Ronde, Lower Salmon, Lower Snake, Lower Snake–Asotin, Lower North Fork Clearwater, Palouse, and Lower Snake–Tucannon subbasins.

The PBFs for Snake River fall-run Chinook salmon are listed in Table 3-1.

3.5.1.3 Life History

Snake River fall Chinook salmon enter the Columbia River in July and August, and migrate past the lower Snake River mainstem dams from August through November. Spawning takes place from October through early December in the mainstem of the Snake River, primarily between Asotin Creek and Hells Canyon Dam, and in the lower reaches of several of the associated major tributaries including the Tucannon, Grande Ronde, Clearwater, Salmon, and Imnaha Rivers (Connor & Burge, 2003; M J Ford et al., 2011a). Spawning has occasionally been observed in the tailrace areas of the four mainstem dams (Dauble, Johnson, & Garcia, 1999; Dauble et al., 1994; Dauble, Johnson, Mueller, & Geist, 1995; Mueller, 2009). Juveniles emerge from the gravels in March and April of the following year.

Until relatively recently, Snake River fall Chinook were assumed to follow an “ocean-type” life history (Dauble & Geist, 2000; Good et al., 2005; Healey, 1991; NMFS, 1992a) where they migrate to the Pacific Ocean during their first year of life, normally within three months of emergence from spawning substrate (as young-of-year smolts), to spend their first winter in the ocean. Ocean-type Chinook salmon juveniles tend to display a “rear as they go” strategy in which they continually move downstream through shallow shoreline habitats during the first summer and fall until they reach the ocean by winter (Connor & Burge, 2003; Coutant & Whitney, 2006). However, a substantial number Snake River fall Chinook juvenile exhibit a “reservoir-type” life history by which they begin their seaward migration later than ocean-types, arrest their migration and overwinter in reservoirs on the Snake and Columbia Rivers, then resume migration, entering the ocean in early spring as age-1 smolts (Connor & Burge, 2003; Connor, Burge, & Waitt, 2002; Connor, Sneva, Tiffan, Steinhorst, & Ross, 2005; Hegg, Kennedy, Chittaro, & Zabel, 2013). Analysis of fish scales taken from non-hatchery, adult, fall-run Chinook salmon indicate that approximately half of the returns passing Lower Granite Dam are reservoir type Snake River fall Chinook and overwintered in freshwater (M J Ford et al., 2011b). (Tiffan & Connor, 2012) showed that young-of-year fish favor water less than 1.8 meters (m) (6 feet) deep.

3.5.1.4 Current Stressors and Threats

Stressors to Snake River Chinook salmon include commercial and recreational harvest, bycatch, and natural predation; reduced habitat and prey quality and quantity; and impeded migration pathways.

Spawning and rearing habitat quality in tributary streams in the Snake River varies from excellent in wilderness and roadless areas to poor in areas subject to intensive human land uses (NMFS, 2015c). Critical habitat throughout much of the Interior Columbia (which includes the Snake River and the Middle Columbia River; MCR) has been degraded by intensive agriculture, alteration of stream morphology (i.e., channel modifications and diking), riparian vegetation disturbance, wetland draining and conversion, livestock grazing, dredging, road construction and maintenance, logging, mining, and urbanization. Reduced summer streamflows, impaired water quality, and reduction of habitat complexity are common problems for critical habitat in non-wilderness areas. Human land use practices throughout the basin have caused streams to become straighter, wider, and shallower, thereby reducing rearing habitat and increasing water temperature fluctuations.

In many stream reaches designated as critical habitat in the Snake River basin, streamflows are substantially reduced by water diversions (NMFS, 2015c). Withdrawal of water, particularly during low-flow periods that commonly overlap with agricultural withdrawals, often increases summer stream temperatures, blocks fish migration, strands fish, and alters sediment transport (Spence, Lomnicky, Hughes, & Novitzki, 1996).

Many stream reaches designated as critical habitat in the Snake River basin are on the Clean Water Act 303(d) list for impaired water quality (e.g., due to elevated water temperature) (IDEQ, 2011). Many areas that were historically suitable rearing and spawning habitat are now unsuitable due to high summer stream temperatures, such as some stream reaches in the Upper Grande Ronde. Removal of riparian vegetation, alteration of natural stream morphology, and withdrawal of water for agricultural or municipal use all contribute to elevated stream temperatures. Water quality in spawning and rearing areas in the Snake River has also been impaired by high levels of sedimentation and by metal contamination potentially from mine waste (IDEQ, 2001; IDEQ & EPA, 2003).

Migration habitat quality for Snake River salmon has also been severely degraded, primarily by the development and operation of dams and reservoirs on the mainstem Columbia and Snake Rivers (NMFS, 2008c). Hydroelectric development has modified natural flow regimes in the migration corridor causing higher water temperatures and changes in fish community structure that have led to increased rates of piscivorous and avian predation on juvenile salmon, and delayed migration for both adult and juveniles. Physical features of dams, such as turbines, also kill migrating fish.

According to NOAA's 2015 Pacific Salmon and Steelhead Status Review Update (NOAA Fisheries, 2015b) and NOAA's 2016 5-Year Review of Snake River Salmon and Steelhead (NMFS, 2016b), "Overall, while new information indicates an improvement in ESU abundance, uncertainty about population productivity and diversity indicate that the biological risk category has not changed enough since the last status review to achieve the desired viability status of highly viable and support delisting (NMFS, 2015c, 2016b). The status report concluded that the Snake River fall-run Chinook salmon ESU should remain listed as threatened.

More recently, fall Chinook returns have declined overall (approximately 50% of the 10-year average 2007-2017 in 2017), and SR fall-run returns also reflect this downturn. The following Table 3-2 from (W. T. Peterson et al., 2018) shows the counts for returning fall Chinook at the Bonneville Dam over the past 20 years.

Table 3-2. Adult returns to Bonneville Dam (Source: (W. T. Peterson et al., 2018), Table ARD-02).

Adult returns by Year of Ocean Entry[1]

Year	OPIH Coho (adults:smolts)	Bonneville spring Chinook (n)	Bonneville fall Chinook (n)	Klamath River fall Chinook (n est.)
1998	0.0128	178,302	192,793	123,856
1999	0.0227	391,367	400,205	187,333
2000	0.0459	268,813	473,786	160,788
2001	0.0258	192,010	610,075	191,948
2002	0.0399	170,152	583,332	78,943
2003	0.0282	74,038	417,057	65,227
2004	0.0193	96,456	299,161	61,374
2005	0.0238	66,624	161,256	132,131
2006	0.0250	125,543	314,995	70,554
2007	0.0255	114,525	283,691	100,644
2008	0.0461	244,385	467,524	90,860
2009	0.0251	167,097	401,576	101,977
2010	0.0234	158,075	350,083	295,322
2011	0.0092	83,299	952,944	165,025
2012	0.0174	188,078	854,503	160,396
2013	0.0675	220,250	954,140	77,821
2014	0.0131	137,176	440,945	24,582
2015	0.0128	83,616	316,833	31,838[2]
2016	0.0156	87,890	186,862	—
2017	0.0171[2]	—	—	—

[1] Counts of spring and fall Chinook salmon are lagged by 2 years. Return ratios for coho salmon are lagged by 1 year.

[2] Estimate based on jack returns.

The following figure depicts interannual variability in total fall-Chinook returns to the Lower Granite Dam, indicating that a steep decline occurred in 2017 and 2018 (Figure 3-5). Further, the Snake River Fall-run natural origin Chinook have also steeply declined (Table 3-3).

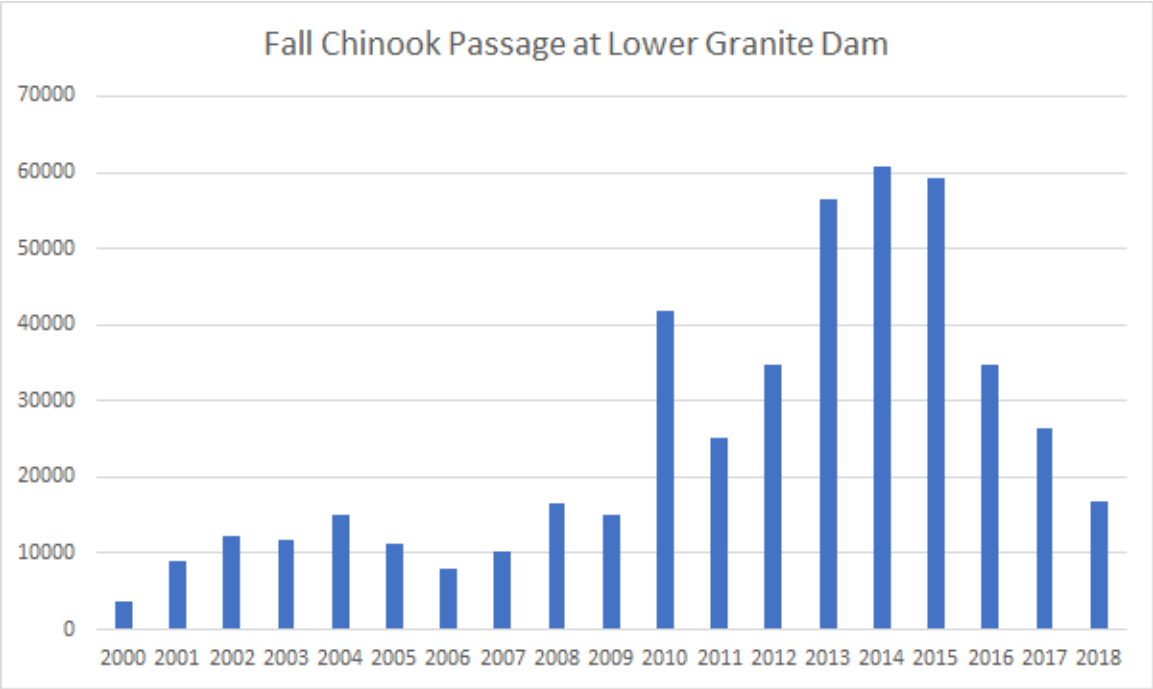


Figure 3-5. Interannual fall Chinook adult return data at Lower Granite Dam. (Columbia River DART).

Table 3-3. Estimated Columbia River return of Snake River natural origin fall Chinook adults 1986-2017 (Source: Table 5 in WDFW, ODFW joint status report, 2018 (WDFW & ODFW, 2018).

Year	Columbia River Return	Non-Treaty Zone 1-5 Harvest	Bonneville Dam Count	Treaty Zone 6 Harvest	Non-Treaty Above BON Harvest [1]	Lower Granite Run Size [2]
1986	2,830	652	2,178	723	12	449
1987	1,783	491	1,292	379	2	253
1988	3,558	944	2,614	965	7	368
1989	1,981	373	1,608	608	8	295
1990	508	71	437	169	2	78
1991	1,846	275	1,571	379	17	318
1992	1,289	112	1,178	202	6	549
1993	1,475	107	1,368	270	11	742
1994	958	0	958	173	1	406
1995	1,296	10	1,286	225	9	350
1996	1,729	95	1,634	350	3	639
1997	1,839	99	1,740	459	7	797
1998	730	21	709	165	4	306
1999	2,395	163	2,232	515	11	905
2000	2,612	179	2,432	520	9	1,148
2001	14,133	778	13,355	2,020	63	5,163
2002	3,665	250	3,416	709	11	2,116
2003	8,093	675	7,417	953	33	4,257
2004	8,174	706	7,467	877	21	7,055
2005	9,500	779	8,721	1,434	49	5,299
2006	12,202	928	11,274	2,136	34	4,713
2007	9,878	567	9,311	1,492	64	3,914
2008	8,738	622	8,115	1,615	30	3,937
2009	15,576	1,568	14,008	3,831	53	4,653
2010	12,855	971	11,884	2,141	34	7,302
2011	17,156	2,228	14,928	2,918	53	8,370
2012	19,360	2,641	16,719	3,433	61	12,797
2013	34,669	3,462	31,208	6,429	141	21,124
2014	20,752	2,484	18,268	4,096	32	14,172
2015	24,054	2,530	21,523	4,319	87	16,212
2016	14,493	2,023	12,568	2,907	96	9,772
2017	11,750	1,403	10,997	3,308	86	6,966

[1] *Recent year harvest data for non-treaty recreational fisheries upstream of Bonneville Dam considered preliminary until catch record card data is finalized.*

[2] *Includes release mortalities*

3.5.2 Snake River spring/summer-run Chinook salmon

The Snake River spring/summer Chinook salmon ESU was listed as threatened on April 22, 1992 (NMFS, 1992a) and the threatened status was reaffirmed in 2005 (NMFS, 2005b). The spring/summer run and fall run subpopulations are distinguished from one another by the seasons during which they return to freshwater streams.

3.5.2.1 Distribution

Snake River Chinook salmon occupy the Snake River basin in southeastern Washington, northeastern Oregon, and north/central Idaho. The Snake River ESU includes all naturally spawning populations of spring/summer Chinook in the mainstem Snake River (below Hells Canyon Dam) and in the Tucannon River, Grande Ronde River, Imnaha River, and Salmon River subbasins (NMFS, 1992b), as well as the progeny of 15 artificial propagation programs (NMFS, 2005b). The historical Snake River ESU likely also included populations in the Clearwater River drainage and extended above the Hells Canyon Dam complex; however, current runs returning to the Clearwater River drainages are not considered to be a part of the Snake River spring/summer run Chinook salmon ESU.

3.5.2.2 Critical habitat

Critical habitat for Snake River spring/summer-run Chinook salmon was designated in 1993 and 1999 and includes reaches of the Columbia, Snake, and Salmon Rivers and accessible tributaries of the Snake and Salmon Rivers (NMFS, 1993, 1999a). The geographic extent of critical habitat includes all Snake River reaches upstream to Hells Canyon Dam; all river reaches presently or historically accessible to Snake River spring/summer Chinook salmon within the Salmon River basin; and all river reaches presently or historically accessible to Snake River spring/summer Chinook salmon within the Hells Canyon, Imnaha, Lower Grande Ronde, Upper Grande Ronde, Lower Snake-Asotin, Lower Snake-Tucannon, and Wallowa subbasins.

The PBFs for Snake River Chinook salmon (spring/summer and fall runs) are listed in Table 3-1.

3.5.2.3 Life History

Snake River spring/summer Chinook salmon are characterized by their return times. Spring runs are counted at Bonneville Dam beginning in early March and ending the first week of June. Summer runs include Chinook adults that pass Bonneville Dam from June through August. Returning adults will hold migration in deep mainstem and tributary pools until late summer, when they move up into tributary areas to spawn. In general, spring-run Chinook salmon tend to spawn in higher-elevation reaches of major Snake River tributaries in mid- to late August, and summer-run Chinook salmon tend to spawn lower in Snake River tributaries in late August and September. The spawning areas of the two runs may overlap.

Spring/summer Chinook follow a “stream-type” life history characterized by rearing for a full year in spawning habitat before migrating to the sea (Healey, 1991). Eggs are deposited in late summer and early fall, incubate through the winter, and hatch between late winter and early spring. Juveniles rear through the summer, and most overwinter and migrate to the sea in the spring of their second year. Depending on the tributary and the specific habitat conditions,

juveniles may migrate extensively from natal reaches into alternative summer-rearing or overwintering areas. Snake River spring/summer Chinook salmon return from the ocean to spawn primarily as four- and five-year-old fish, after two to three years in the ocean. A small fraction of the fish return as three-year old “jacks” (precocious spawners), of which the majority are males(Good et al., 2005).

3.5.2.4 Current Stressors and Threats

Limiting factors for Snake River spring/summer run Chinook salmon are the same as those listed above for the Snake River fall-run Chinook salmon subpopulation.

Per NOAA’s 2016 (NOAA Fisheries, 2016o) status review, the Snake River spring/summer Chinook salmon ESU should remain listed as threatened.

3.5.3 Upper Columbia River spring-run Chinook Salmon

On March 24, 1999, the NMFS listed the Upper Columbia River (UCR) spring-run Chinook salmon as an endangered species (NMFS, 1999d). The status of this ESU was reaffirmed on June 28, 2005 (NMFS, 2005b) and again on August 15, 2011, after the five-year status review (NMFS, 2011c).

3.5.3.1 Distribution

The ESU includes all naturally spawned populations of Chinook salmon in all river reaches accessible to Chinook salmon in the Columbia River tributaries upstream of Rock Island Dam and downstream of Chief Joseph Dam (barrier to upstream movement), excluding the Okanogan River (NMFS, 1999d). Six artificial propagation programs are included in this ESU: The Twisp River, Chewuch River, Methow Composite, Winthrop National Fish Hatchery, Chiwawa River, and White River spring-run Chinook hatchery programs.

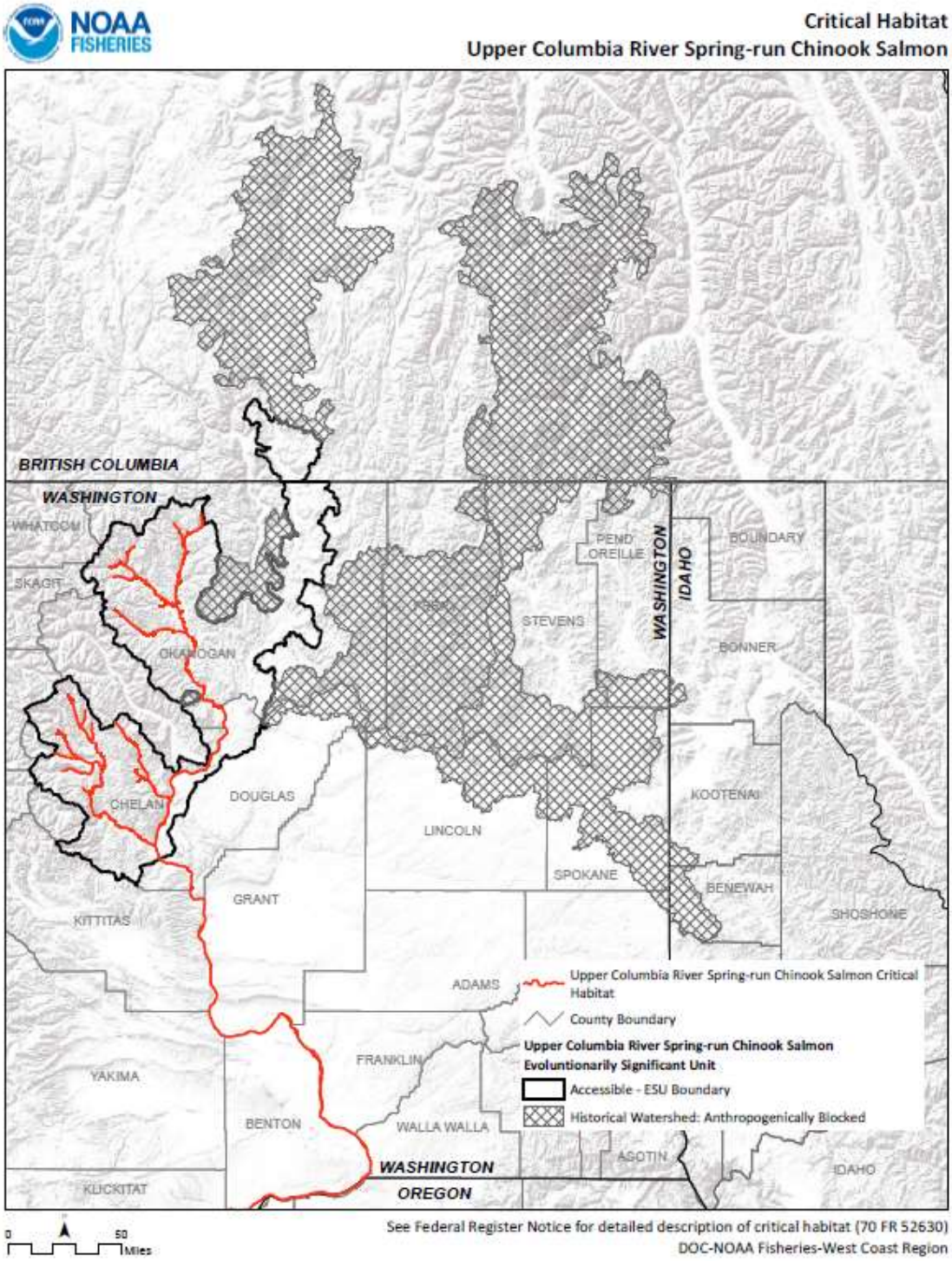
3.5.3.2 Critical Habitat

Critical habitat for UCR Chinook salmon was designated for Oregon and Washington in 2005 and includes freshwater areas shown in Figure 3-6 (NMFS, 2005c).

The PBFs for UCR Chinook salmon are listed in Table 3-1.

3.5.3.3 Life History

UCR spring-run Chinook salmon exhibit stream-type life history strategies. Adults begin returning from the ocean in the early spring, with the run into the Columbia River peaking in mid-May. They then enter UCR tributaries from April through July, where they hold until spawning occurs in the late summer, peaking in mid to late August. Juvenile spring-run Chinook salmon spend a year in freshwater before migrating to the ocean. Most UCR spring-run Chinook salmon return as adults after two or three years in the ocean.



Source: (NOAA Fisheries, 2016h)

Figure 3-6. Critical Habitat for Upper Columbia River Spring-run Chinook Salmon

3.5.3.4 Current Stressors and Threats

Limiting factors for UCR spring-run Chinook salmon include adverse effects from Columbia River hydropower (i.e., modified hydrograph and increase in lentic conditions/decrease in riverine conditions, passage barriers, temperature; dissolved oxygen problems, and invasive species), riparian degradation and reduced large wood recruitment, altered floodplain connectivity and function, altered channel structure and complexity, reduced streamflow, and hatchery-related adverse effects (e.g., reduced genetic diversity) (NMFS, 2011a).

Per NOAA's 2016 (NOAA Fisheries, 2016b) review, the UCR spring-run Chinook salmon ESU should remain listed as endangered.

3.5.4 Lower Columbia River Chinook Salmon

The LCR Chinook salmon ESU was listed as threatened pursuant to (NMFS, 2011c), and this determination was reaffirmed in 2011.

3.5.4.1 Distribution

The LCR Chinook salmon ESU includes all naturally spawned Chinook salmon originating from the Columbia River and its tributaries downstream of a transitional point east of the Hood and White Salmon Rivers and any such fish originating from the Willamette River and its tributaries below Willamette Falls. Also this ESU includes Chinook salmon from 15 artificial propagation programs (NOAA Fisheries, 2014). The following individuals are not included in the ESU:

- Spring-run Chinook salmon originating from the Clackamas River;
- Fall-run Chinook salmon originating from the UCR bright hatchery stocks that spawn in the mainstem Columbia River below Bonneville Dam or in other tributaries upstream from the Sandy River to the Hood and White Salmon Rivers;
- Spring-run Chinook salmon originating from the Round Butte Hatchery (Deschutes River, Oregon) and spawning in the Hood River;
- Spring-run Chinook salmon originating from the Carson National Fish Hatchery and spawning in the Wind River; and
- Naturally spawning Chinook salmon originating from the Rogue River Fall Chinook Program.

Dam removal projects have reopened historical habitat once blocked to migrating LCR Chinook salmon. The removal of Marmot Dam in the Sandy River eliminated migration delays and injuries associated with holding at the dam's fish ladder. Additionally, the removal of the diversion dam on the Little Sandy River restored access and flow to historical salmon habitat. The removal of Condit Dam on the White Salmon River provides an opportunity for the reestablishment of a spring-run population with renewed access to historical spawning grounds. Spring-run Chinook salmon in the Hood River are largely from the Deschutes River spring-run (managed under the MCR spring-run ESU) and are not considered to benefit the status of the LCR ESU. However, some LCR spring-run Chinook salmon have been detected in the Hood River (NOAA Fisheries, 2015b).

3.5.4.2 Critical Habitat

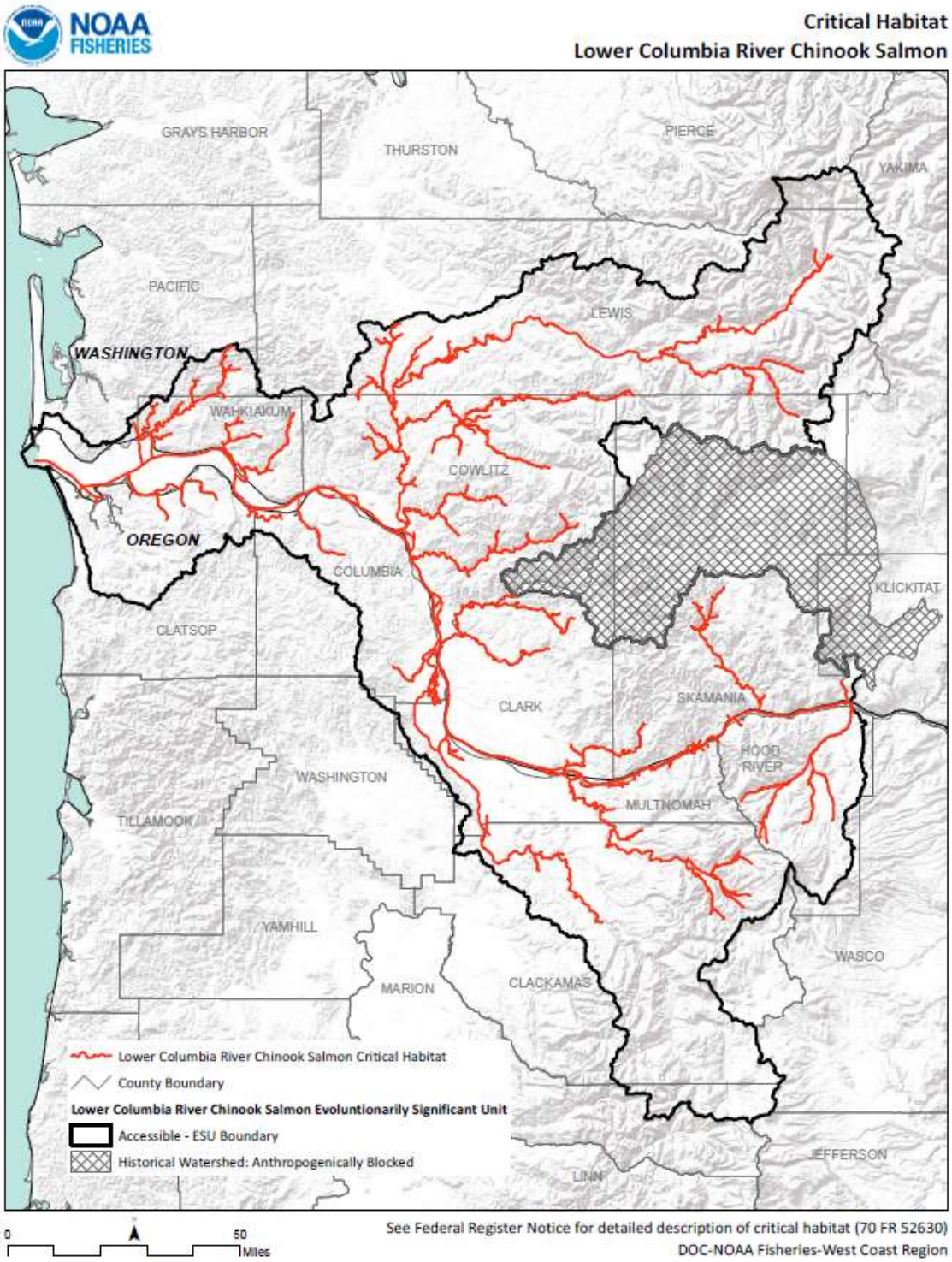
Critical habitat for LCR Chinook salmon was designated for Oregon and Washington in 2005 and includes watersheds shown in Figure 3-7 (NMFS, 2005c). The PBFs for LCR Chinook salmon are listed in Table 3-1.

3.5.4.3 Life History

LCR Chinook salmon generally follow an ocean-type (fall-run) life history cycle. These salmon migrate to the ocean within the first year, after one to four months of rearing in freshwater habitat in the spring (NOAA Fisheries, 2005) CHART. Chinook fry emerge in April and quickly find protection in off-stream refuge habitat. Some Chinook remain in their natal streams until the spring after hatching at which point they emigrate as yearlings. Ocean-type Chinook return to the Columbia River at approximately three to four years of age. Entry to the Columbia River occurs generally from August to September; spawning begins in late September through November and peaks in mid-October (NOAA, 2006; NOAA Fisheries, 2005). Spawning occurs in the lower reaches of Columbia River tributaries. (NOAA, 2006; NOAA Fisheries, 2005). There is also a subpopulation of fall-run LCR Chinook called “brights,” which are presumed to be genetically linked to Chinook introduced from the Rogue River and the UCR (NOAA, 2006). Brights enter the Columbia River slightly later than the rest of the population, from August to October and spawn from November to January. Peak bright spawning occurs in mid-November.

Spring-run or stream-type LCR Chinook spawn in August to September in Columbia River headwaters from the age of four to five years, a full year after returning to freshwater. Generally, the return of spring-run LCR Chinook occurs in March and April. Juvenile spring-run Chinook emerge from sediments between November and March with peak emergence occurring between December and January (NOAA Fisheries, 2005).

There are 32 populations in the LCR Chinook salmon ESU: nine spring-run, 21 fall-run, and two late-fall-run (named according to the seasonal return to streams).



Source: (NOAA Fisheries, 2016c)

Figure 3-7. Critical Habitat for Lower Columbia River Chinook Salmon

3.5.4.4 Current Stressors and Threats

Limiting factors for LCR Chinook salmon include relatively high harvest rates, especially for the spring-run, and low abundance of fall-run populations (NMFS, 2012a); migratory impediment caused by dams (i.e., Mossyrock Dam on the Cowlitz River); land development and habitat degradation; and potential effects from climate change and coastal ocean conditions (e.g., reduced survival of emigrating smolts and corresponding drop in spawner abundance) (NOAA Fisheries, 2015b). Reduced complexity, connectivity, quantity, and quality of habitat used for spawning, rearing, foraging, and migrating are perhaps the most important limitations to LCR Chinook population growth. Degradation or loss of habitat due to conversion to agricultural or urbanized uses (e.g., diking and draining of wetlands and floodplain) is also of particular concern (D L Bottom et al., 2005; NMFS, 2013). Reduced habitat complexity has resulted in a concomitant increase in water temperatures (LCFRB, 2010; NMFS, 2013; ODFW, 2010). Contamination of salmon habitat from wastewater treatment plant effluent, stormwater runoff, and nonpoint source pollution is a growing concern (Morace, 2012; Nilsen & Morace, 2014; NMFS, 2013). Data collected by Ecology, the Oregon Department of Environmental Quality and the Columbia River Contaminants and Habitat Characterization Project indicate that contaminants are present above levels of concern (Alvarez et al., 2014; Counihan et al., 2013; Nilsen & Morace, 2014; Nilsen et al., 2014) and that Total Maximum Daily Loads are warranted for the LCR.

According to NOAA's 2016 status update (NMFS, 2016a), for the LCR Chinook salmon ESU, only two of 32 populations (Lewis River late fall-run and Sandy River late fall-run) could be considered viable or nearly so.

3.5.5 Upper Willamette River Chinook Salmon

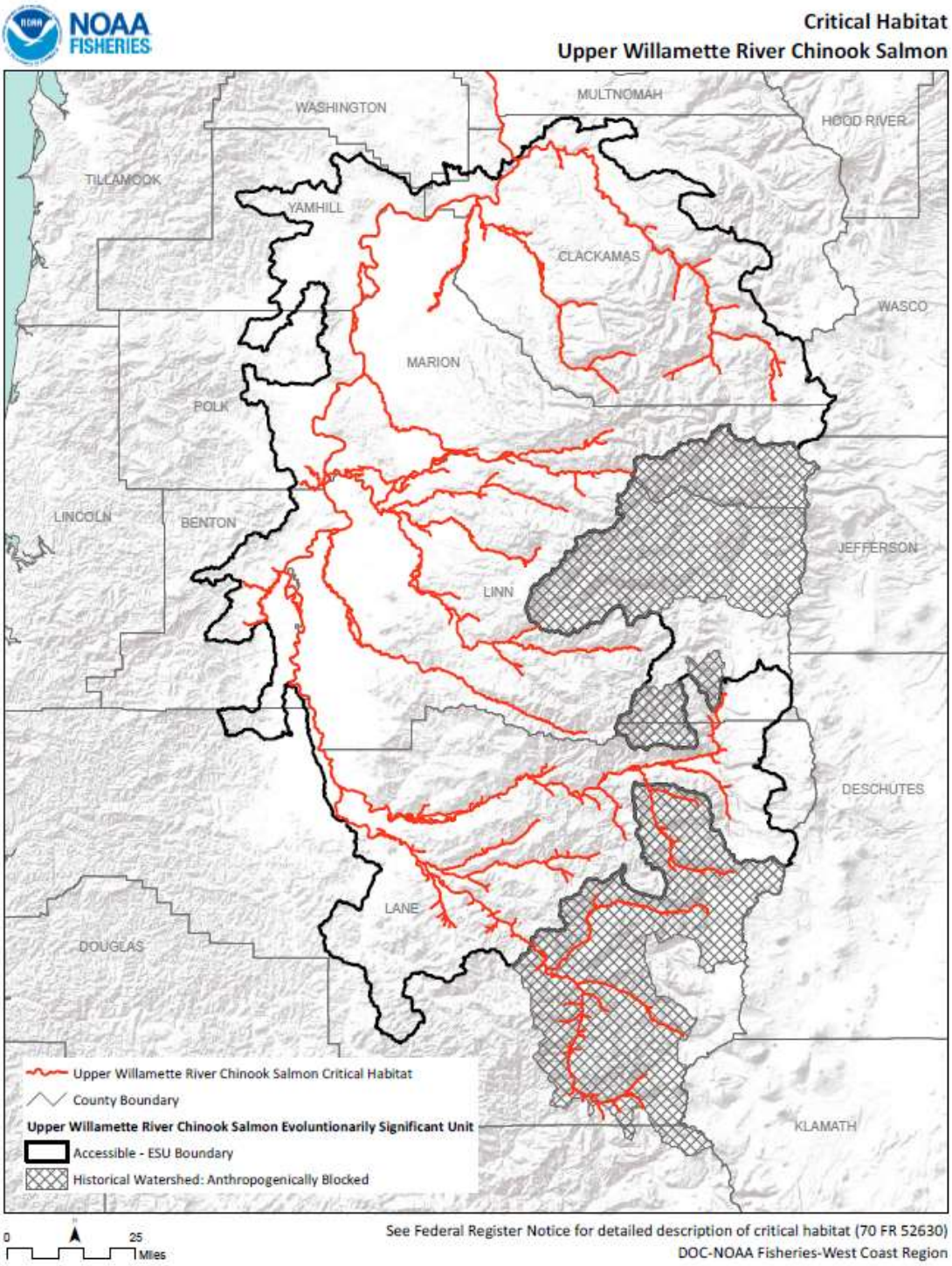
The UWR Chinook salmon ESU was listed as threatened pursuant to (NMFS, 2011c), and this determination was reaffirmed in 2011.

3.5.5.1 Distribution

The UWR Chinook salmon ESU includes all naturally spawned populations of spring-run Chinook salmon in the Clackamas River and in the Willamette River and its tributaries above Willamette Falls, Oregon, and progeny of six artificial propagation programs (USDI, 2016). All seven historical populations of UWR Chinook salmon are contained within a single ecological subregion, the western Cascade Range (NMFS, 2016c).

3.5.5.2 Critical Habitat

Critical habitat for UWR Chinook salmon ESU was designated for Oregon in 2005 and includes freshwater areas shown in Figure 3-8 (NMFS, 2005c). The PBFs for UWR Chinook salmon are listed in Table 3-1.



Source: (NOAA Fisheries, 2016j)

Figure 3-8. Critical Habitat for Upper Willamette River Chinook Salmon

3.5.5.3 Life History

As with other ESUs, UWR Chinook salmon can be either “ocean type” or “stream type,” which is determined by their propensity to migrate to the ocean at less than one year old or to remain in streams until they are approximately one year old (NMFS, 2015b). Spawning runs in this ESU occur in spring. The abundance of UWR Chinook salmon is likely fewer than 10,000 fish, and the majority of those individuals are from the McKenzie and Clackamas River populations (ODFW & NMFS, 2011) .

3.5.5.4 Current Stressors and Threats

Limiting factors for UWR Chinook salmon ESU (and their critical habitat) include widespread loss and degradation of peripheral and transitional habitats (e.g., through diking and draining of wetlands and floodplains) (M J Ford et al., 2011a; Good et al., 2005) and degraded freshwater habitat (e.g., reduced floodplain connectivity and function, channel structure and complexity, riparian habitat, large wood recruitment, and water quality). Also, recovery of this ESU is limited by increased disease incidence, altered stream flows, reduced access to spawning and rearing habitats (because of tributary dams), altered food web (due to reduced inputs of detritus), and predation by and competition with native and non-native species (including hatchery fish). Harvest and bycatch are also key limitations (ODFW & NMFS, 2011).

Per NOAA’s 2016 status update (NOAA Fisheries, 2016j), the Upper Willamette River Chinoook DPS should remain listed as threatened.

3.6 Sockeye Salmon (*Oncorhynchus nerka*)

Sockeye salmon are the second most abundant of the seven Pacific salmon species (Quinn, 2005). They display more life history diversity than all other members of the *Oncorhynchus* genus (Burgner, 1991). Sockeye salmon are generally anadromous, but distinct populations of non-anadromous *O. nerka* also exist; these fish are commonly referred to as kokanee (*O. nerka kennerlyi*) or silver trout (Wydoski & Whitney, 2003). The vast majority of sockeye populations spawn in or near lakes. Spawning can take place in lake tributaries, lake outlets, rivers between lakes, and on lake shorelines or beaches where suitable upwelling or intra-gravel flow is present. Spawn timing is often determined by water temperature. In spawning habitats with cooler water temperatures, sockeye typically spawn earlier (August) than in warmer habitats (November) (Burgner, 1991). Sockeye fry that are spawned in lake tributaries typically exhibit a behavior of rapid downstream migration to the nursery lake after emergence, whereas lake/beach spawned sockeye rapidly migrate to open limnetic waters after emergence. Lake-rearing juveniles typically spend one to three years in their nursery lake before emigrating to the marine environment (Gustafson et al., 1997). Other life history variants include ocean-type and river-type sockeye. Ocean-type populations typically use large rivers and side channels or spring-fed tributary systems for spawning and emigrate to sea soon after emergence. River-type sockeye rear in rivers for one year before emigrating to sea. (Quinn, 2005) describes the differences between ocean-type and river-type sockeye as a continuum of rearing patterns rather than as two discrete types.

Upon smoltification, sockeye emigrate to the ocean. Peak emigration occurs in mid-April to early May in southern sockeye populations (generally south of 52ºN latitude) and as late as early July in northern populations (62ºN latitude and north) (Burgner, 1991). Typically, river-type sockeye populations make little use of estuaries during their emigration to the marine environment (Quinn, 2005). Estuarine habitats may be more extensively used by ocean-type sockeye (Quinn, 2005). Upon entering marine waters, sockeye may reside in the nearshore or coastal environment for several months but are typically distributed offshore by fall (Burgner, 1991).

In the marine environment, North American sockeye stocks are limited to the zone north of 46ºN latitude. Within these zones, sockeye salmon have a wide distribution. In North America, their range is south to the Sacramento River in California (historically) and as far north as Kotzebue Sound in Alaska. In the Western Pacific, sockeye can be found from the Kuril Islands of Japan to Cape Chaplina in Russia.

3.6.1 Snake River Sockeye Salmon

The Snake River sockeye salmon ESU was first listed as endangered under the ESA in 1991, and the listing was reaffirmed in 2005 (NMFS, 2005b). On May 26, 2016, in the most recent five-year review for Pacific salmon and steelhead, the NMFS concluded that the species should remain listed as endangered (NMFS, 2016d).

3.6.1.1 Distribution

This ESU includes all anadromous and resident sockeye salmon from the Snake River basin in Idaho, as well as artificially propagated sockeye salmon from the Redfish Lake captive propagation program.

3.6.1.2 Critical Habitat

Critical habitat for Snake River sockeye salmon was designated in 1993 and includes the Snake and Salmon Rivers, Alturas Lake Creek, Valley Creek, Stanley Lake, Redfish Lake, Yellowbelly Lake, Pettit Lake, Alturas Lake, and all inlet/outlet creeks to the aforementioned lakes (NMFS, 1993). PBFs for Snake River sockeye salmon critical habitat are described in Table 3-4.

Table 3-4. Physical and Biological Features of Snake River Sockeye Critical Habitat

Site	Physical and Biological Feature
Adult spawning and juvenile rearing	Suitable spawning gravel, water quality and quantity, water temperature, and access to spawning areas
Migration (adult and juvenile)	Suitable substrate, water quality and quantity, water temperature, water velocity, cover/shelter, food[a], riparian vegetation, space, and safe passage

Note:
a Food applies to juvenile migration only.

3.6.1.3 Life History

Snake River sockeye salmon adults enter the Columbia River primarily during June and July and arrive in the Sawtooth Valley, peaking in August. The Sawtooth Valley supports the only remaining run of Snake River sockeye salmon. The adults spawn in lakeshore gravels, primarily in October (Bjornn, Craddock, & Corley, 1968). Eggs hatch in the spring between 80 and 140 days after spawning. Fry remain in gravels for three to five weeks, emerge from April through May and move immediately into lakes. Once there, juveniles feed on plankton for one to three years before they migrate to the ocean, leaving their natal lake in the spring from late April through May (Bjornn et al., 1968). Snake River sockeye salmon usually spend two to three years in the Pacific Ocean and return to Idaho in their fourth or fifth year of life.

3.6.1.4 Current Stressors and Threats

Spawning and rearing habitat quality in tributaries of the Snake River varies from excellent in wilderness areas to poor in areas of intensive human land uses (NMFS, 2015c). Critical habitat throughout much of the Interior Columbia (which includes the Snake River and the MCR) has been degraded by intensive agriculture, alteration of stream morphology (i.e., channel modifications and diking), riparian vegetation disturbance, wetland draining and conversion, livestock grazing, dredging, road construction and maintenance, logging, mining, and urbanization. Reduced summer streamflows, impaired water quality, and reduction of habitat complexity are common problems for critical habitat in non-wilderness areas. Human land use practices throughout the basin have caused streams to become straighter, wider, and shallower, thereby reducing rearing habitat and increasing water temperature fluctuations. In many stream reaches designated as critical habitat in the Snake River basin, streamflows are substantially reduced by water diversions (NMFS, 2015c). Withdrawal of water, particularly during low-flow periods that commonly overlap with agricultural withdrawals, often increases summer stream temperatures, blocks fish migration, strands fish, and alters sediment transport (Spence et al., 1996).

Many stream reaches designated as critical habitat in the Snake River basin are on the Clean Water Act 303(d) list for impaired water quality (IDEQ, 2011). In addition to elevated temperatures, water quality in spawning and rearing areas in the Snake River has been impaired by high levels of sedimentation and by metal contamination potentially from mine waste (e.g., IDEQ (2001); IDEQ and EPA (2003)).

Migration habitat quality for Snake River salmon has also been severely degraded, primarily by the development and operation of dams and reservoirs on the mainstem Columbia and Snake Rivers (NMFS, 2008c). Hydroelectric development has modified natural flow regimes in the migration corridor causing higher water temperatures and changes in fish community structure that have led to increased rates of piscivorous and avian predation on juvenile salmon, and delayed migration for both adult and juveniles. Physical features of dams such as turbines also kill migrating fish.

Per NOAA’s 2016 status review (NMFS, 2016b), the Snake River Sockeye Salmon ESU should remain listed as endangered.

3.7 Steelhead Trout (*Oncorhynchus mykiss*)

Steelhead trout is an anadromous salmonid fish that can live up to 11 years and grow up to 25 kg (55 pounds) and 120 cm (47 inches) long, though most fish tend to be much smaller than (NMFS, 2016f). They are distinguishable from other salmonids by their dark olive color, speckled body, and pinkish red stripe along their sides, though they tend to remain more silver while in the marine environment (than the non-migratory rainbow trout [*O. mykiss*]) (NMFS, 2016f).

Steelhead in the NW mature in one of two distinct modes, either stream-maturing or ocean-maturing (NMFS, 2016f). Stream-maturing individuals (also called summer-run steelhead) return to freshwater streams prior to becoming fully mature, typically between May and October; spawning occurs several months later. Ocean-maturing individuals (also called winter-run steelhead) mature while at sea and reenter freshwater streams during November and April. Coastal streams tend to be dominated by ocean-maturing groups, whereas inland streams tend to be dominated by stream-maturing groups (NMFS, 2016f).

Spawning occurs over coarse substrates (gravel) in cold, fast-flowing streams with highly oxygenated waters, and spawning may occur more than once (NMFS, 2016f). After hatching (three to four weeks after spawning), steelhead may reside in freshwater streams for up to seven years before migrating into estuaries to smoltify, and they may reside in marine environments for three years (NMFS, 2016f). A small number of steelhead actually return to freshwater after their first year only to migrate back out without spawning; this behavior is irregular among salmonid species.

Steelhead typically feed on zooplankton as juveniles and shift to larger insects, mollusks, crustaceans, and fish as adults (NMFS, 2016f).

3.7.1 Snake River Basin Steelhead Trout

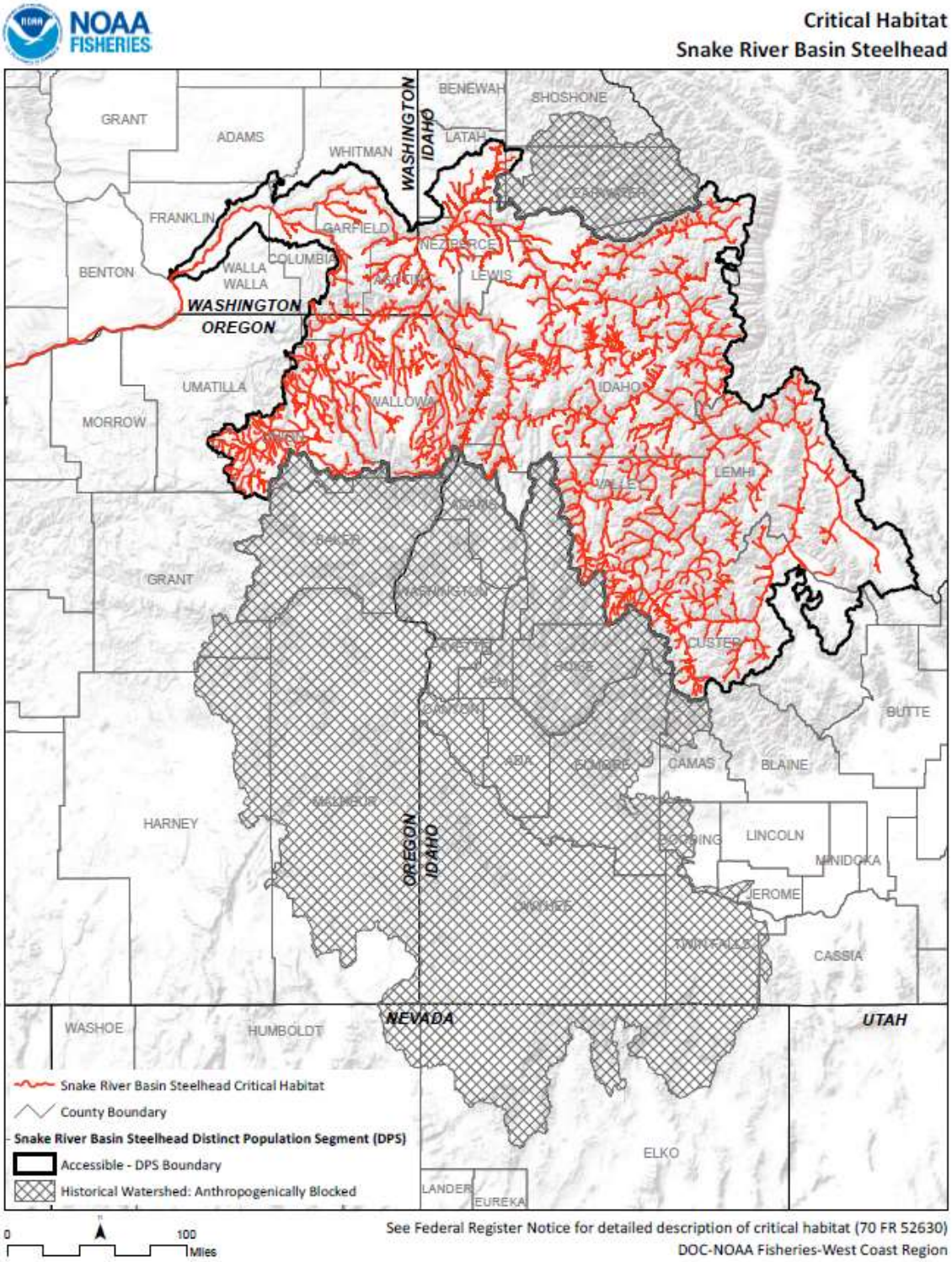
The Snake River Basin (SRB) steelhead trout was listed as a threatened ESU on August 18, 1997 (NMFS, 1997), with a revised listing as a DPS on January 5, 2006 (NMFS, 2006a).

3.7.1.1 Distribution

The SRB steelhead DPS occupies the SRB, which drains portions of southeastern Washington, northeastern Oregon, and north/central Idaho. This species includes all naturally spawning steelhead populations below natural and manmade impassable barriers in streams in the SRB, as well as the progeny of six artificial propagation programs (NMFS, 2006a). The SRB steelhead listing does not include resident *O. mykiss* (rainbow trout) that co-occur with (migratory) steelhead.

3.7.1.2 Critical Habitat

Critical habitat for the SRB steelhead trout DPS was designated for Idaho, Oregon, and Washington in 2005 and includes areas shown in Figure 3-9 (NMFS, 2005c). Specific stream reaches are designated within the Lower Snake, Salmon, and Clearwater River Basins. Habitat areas within the DPS's geographical range that are excluded from critical habitat designation are defined in (NMFS, 2005c). Table 3-5 describes the PBFs for steelhead critical habitat for multiple DPS/ESUs, including the SRB steelhead DPS.



Source: (NOAA Fisheries, 2016g)

Figure 3-9. Critical Habitat for Snake River Basin Steelhead Trout

Table 3-5. Physical and Biological Features of Designated Steelhead Critical Habitats

Habitat Type	Physical and Biological Feature(s)	Applicable Life Stage
Freshwater spawning	Water quality, water quantity, and substrate	Spawning, incubation, and larval development
Freshwater rearing	Water quantity and floodplain connectivity to form and maintain physical habitat conditions	Juvenile growth and mobility
	Water quality and forage[a]	Juvenile development
	Natural cover[b]	Juvenile mobility and survival
Freshwater migration	Free of artificial obstructions, water quality and quantity, and natural cover[b]	Juvenile and adult mobility and survival
Estuarine	Water quality, water quantity, and salinity	Juvenile and adult physiological transitions
	Natural cover and forage[a]	Juvenile growth and maturation and adult conservation
Nearshore marine	Water quality, water quantity, forage[a], natural cover[b]	Juvenile growth and maturation
Offshore marine	Water quality and forage[a]	Juvenile growth and maturation

Notes:
[a] Forage includes aquatic invertebrate and fish species that support growth and maturation.
[b] Natural cover includes shade, large wood, log jams, beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks.
Key:
MCR Middle Columbia River
SRB Snake River Basin
UCR Upper Columbia River

3.7.1.3 Life History

Adult SRB steelhead enter the Columbia River from late June to October to begin their migration inland. After holding over the winter in larger rivers in the SRB, steelhead disperse into smaller tributaries to spawn from March through May. Earlier dispersal occurs at lower elevations, and later dispersal occurs at higher elevations. Juveniles emerge from the gravels four to eight weeks after hatching, and move into shallow, low-velocity areas in side channels and along channel margins, where they are able to escape high velocities and predators (Everest & Chapman, 1972). Juvenile steelhead then progressively move toward deeper water as they grow in size (Bjornn & Reiser, 1991). Juveniles typically reside in freshwater for one to three years, although this species displays a wide diversity of life histories. Smolts migrate downstream during spring runoff, which occurs from March to mid-June depending on elevation, and typically spend one to two years in the ocean.

SRB steelhead exhibit a diversity of life-history strategies, including variations in freshwater and marine residence times. Traditionally, fisheries managers have classified SRB steelhead into two groups, A-run and B-run, based on age at return to freshwater, adult size at return, and migration timing. A-run steelhead tend to be smaller than B-run steelhead, and they predominantly spend one year in the ocean. Conversely, B-run steelhead are larger, and most individuals return after two

years in the ocean. Most Snake River populations support a mixture of the two run types. The highest percentage of B-run fish are in the upper Clearwater River and the South Fork Salmon River; moderate percentages of B-run fish are in the Middle Fork Salmon River; and a very low percentages of B-run fish are in the Upper Salmon River, Grande Ronde River, and Lower Snake River (NOAA Fisheries, 2015b). A-run fish make up the remainder of those populations.

3.7.1.4 Current Stressors and Threats

Limiting factors for Snake River basin steelhead trout include substantial modification of the seaward migration corridor by hydroelectric power development on the mainstem Snake and Columbia Rivers, widespread habitat degradation and reduced streamflows throughout the Snake River basin (Good et al., 2005), and reduced genetic integrity caused by a high proportion of hatchery fish (M J Ford et al., 2011b; Good et al., 2005).

Spawning and rearing habitat quality in tributary streams in the Snake River varies from excellent in wilderness and roadless areas to poor in areas subject to intensive human land uses (NMFS, 2015d). Critical habitat throughout much of the Interior Columbia (which includes the Snake River and the MCR) has been degraded by intensive agriculture, alteration of stream morphology (i.e., channel modifications and diking), riparian vegetation disturbance, wetland draining and conversion, livestock grazing, dredging, road construction and maintenance, logging, mining, and urbanization. Reduced summer streamflows, impaired water quality, and reduction of habitat complexity are common problems for critical habitat in non-wilderness areas. Human land use practices throughout the basin have caused streams to become straighter, wider, and shallower, thereby reducing rearing habitat availability and impairing water temperature.

In many stream reaches designated as critical habitat in the SRB, streamflows are substantially reduced by water diversions (NMFS, 2015d). Withdrawal of water, particularly during low-flow periods that commonly overlap with agricultural withdrawals, often increases summer stream temperatures, blocks fish migration, strands fish, and alters sediment transport (Spence et al., 1996). Reduced tributary streamflow has been identified as a major limiting factor for SRB steelhead in particular (NMFS, 2015d).

Many stream reaches designated as critical habitat for these species are on the Clean Water Act 303(d) list for impaired water quality (e.g., due to elevated water temperature) (IDEQ, 2011). Many areas (e.g., Upper Grande Ronde) that were historically suitable rearing and spawning habitat are now unsuitable due to high summer stream temperatures. Removal of riparian vegetation, alteration of natural stream morphology, and withdrawal of water for agricultural or municipal use all contribute to elevated stream temperatures. Water quality in spawning and rearing areas in the Snake River has also been impaired by high levels of sedimentation and by heavy metal contamination from mine waste (e.g., (IDEQ, 2011; IDEQ & EPA, 2003)).

Migration habitat quality for SRB steelhead has also been severely degraded, primarily by the development and operation of dams and reservoirs on the mainstem Columbia and Snake Rivers (NMFS, 2008c). Hydroelectric development has modified natural flow regimes in the migration corridor, causing higher water temperatures and changes in fish community structure. This has led to increased rates of piscivorous and avian predation on juvenile steelhead, and delayed migration for both adult and juveniles. Physical features of dams such as turbines also kill migrating fish.

Per NOAA's 2016 status review (NMFS, 2016f), the Snake River basin steelhead DPS should remain listed as threatened.

3.7.2 Upper Columbia River Steelhead Trout

The UCR steelhead DPS was listed as endangered on August 18, 1997 (NMFS, 1997), and its status was upgraded to threatened on January 5, 2006 (NMFS, 2006a). The threatened status was affirmed on August 15, 2011 after the 5-year status review (NMFS, 2011c).

3.7.2.1 Distribution

The UCR steelhead DPS includes all naturally spawned populations of steelhead in streams in the Columbia River Basin upstream from the Yakima River in Washington to the US-Canada border (NMFS, 1997). There are four populations of UCR steelhead included in the UCR steelhead DPS: the Wenatchee, Entiat, Methow, and Okanogan populations. Six artificial propagation programs are also considered part of the DPS.

3.7.2.2 Critical Habitat

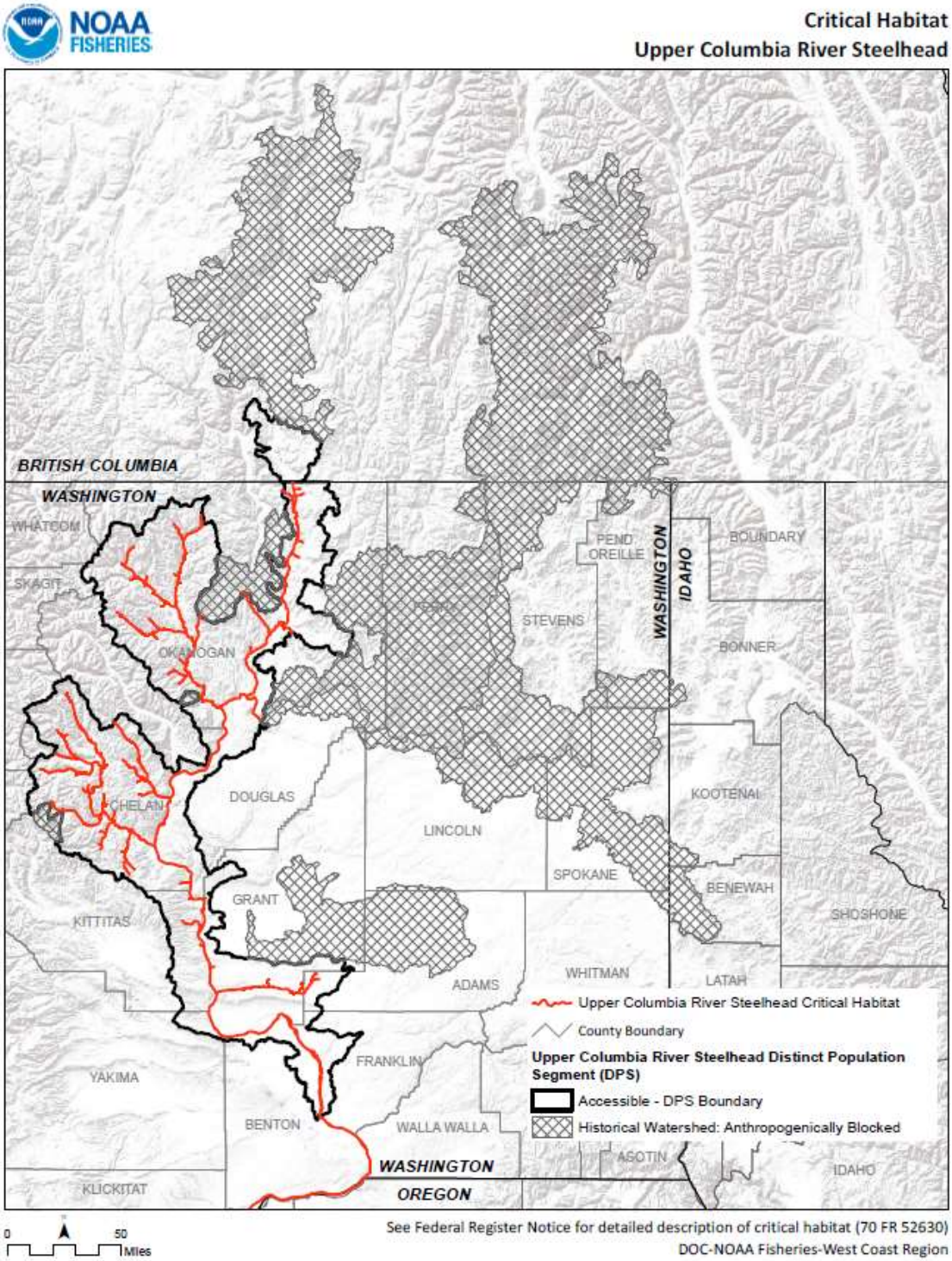
Critical habitat for UCR steelhead trout was designated for Oregon and Washington in 2005 and includes freshwater areas shown in Figure 3-10 (NMFS, 2005c). The PBFs of freshwater spawning sites include water flow, water quality, temperature conditions, and suitable substrate for spawning and incubation. These features are essential to conservation because without them the species cannot successfully spawn and produce offspring. However, there are only a few locations where spawning occurs in the Columbia River for UCR steelhead. The PBFs for West Coast steelhead critical habitat are provided in Table 3-1.

3.7.2.3 Life History

The life-history pattern of UCR steelhead is complex (Peven, Whitney, & Williams, 1994). Adults return to the Columbia River in the late summer and early fall. Unlike spring-run Chinook salmon, most steelhead do not move quickly up to spawning areas (i.e., tributaries). A portion of the returning run overwinters in mainstem reservoirs, passing over the UCR dams (up to Chief Joseph Dam, which is impassible) in April and May of the following year. Spawning occurs in the late spring. Juvenile steelhead generally spend one to three years (up to seven years) rearing in freshwater before migrating to the ocean. Most adult steelhead return to the UCR after one or two years at sea.

3.7.2.4 Current Stressors and Threats

Limiting factors for UCR steelhead trout include adverse impacts from hydropower operations (i.e., modified hydrograph, increase in lentic conditions/ decrease in riverine conditions, passage barriers, altered temperatures and dissolved oxygen, and invasive species), riparian habitat degradation, decreased large wood recruitment, altered floodplain connectivity and function, altered channel structure and complexity, reduced streamflows, and hatchery-related adverse effects (i.e., reduced genetic diversity) (NMFS, 2011c).



Source1981: (NOAA Fisheries, 2016i)

Figure 3-10. Critical Habitat for Upper Columbia River Steelhead Trout

Habitat quality in tributary streams in the UCR range from excellent in wilderness and roadless areas to poor in areas subject to relatively heavy agricultural and urban development (NMFS, 2009b; Wissmar et al., 1994). Critical habitat throughout much of the UCR has been degraded by intense agriculture, alteration of stream morphology (i.e., channel modifications and diking), riparian vegetation removal, wetland draining and conversion, livestock grazing, dredging, road construction and maintenance, logging, mining, and urbanization. Reduced summer stream flows, impaired water quality, and reduction of habitat complexity are common problems for critical habitat in developed areas.

Many stream reaches designated as critical habitat in the UCR are over-allocated under state water law, resulting in greater extraction of water than existing streamflow conditions can support. Withdrawal of water, particularly during low-flow periods that commonly overlap with agricultural withdrawals, often results in increased summer stream temperatures. Withdrawal can also block fish migration, strand fish, and alter sediment transport (Spence et al., 1996). Reduced tributary stream flow has been identified as a major limiting factor for all listed salmon and steelhead species in this area (NMFS, 2007, 2011b).

Per NOAA's 2016 status review (NOAA Fisheries, 2016k), the UCR steelhead DPS should remain listed as threatened.

3.7.3 Middle Columbia River Steelhead Trout

The MCR steelhead DPS was listed as threatened on March 25, 1999 (NMFS, 1999c). The threatened status was reaffirmed on June 28, 2005 (NMFS, 2005b) and updated on April 14, 2014 (NOAA Fisheries, 2014).

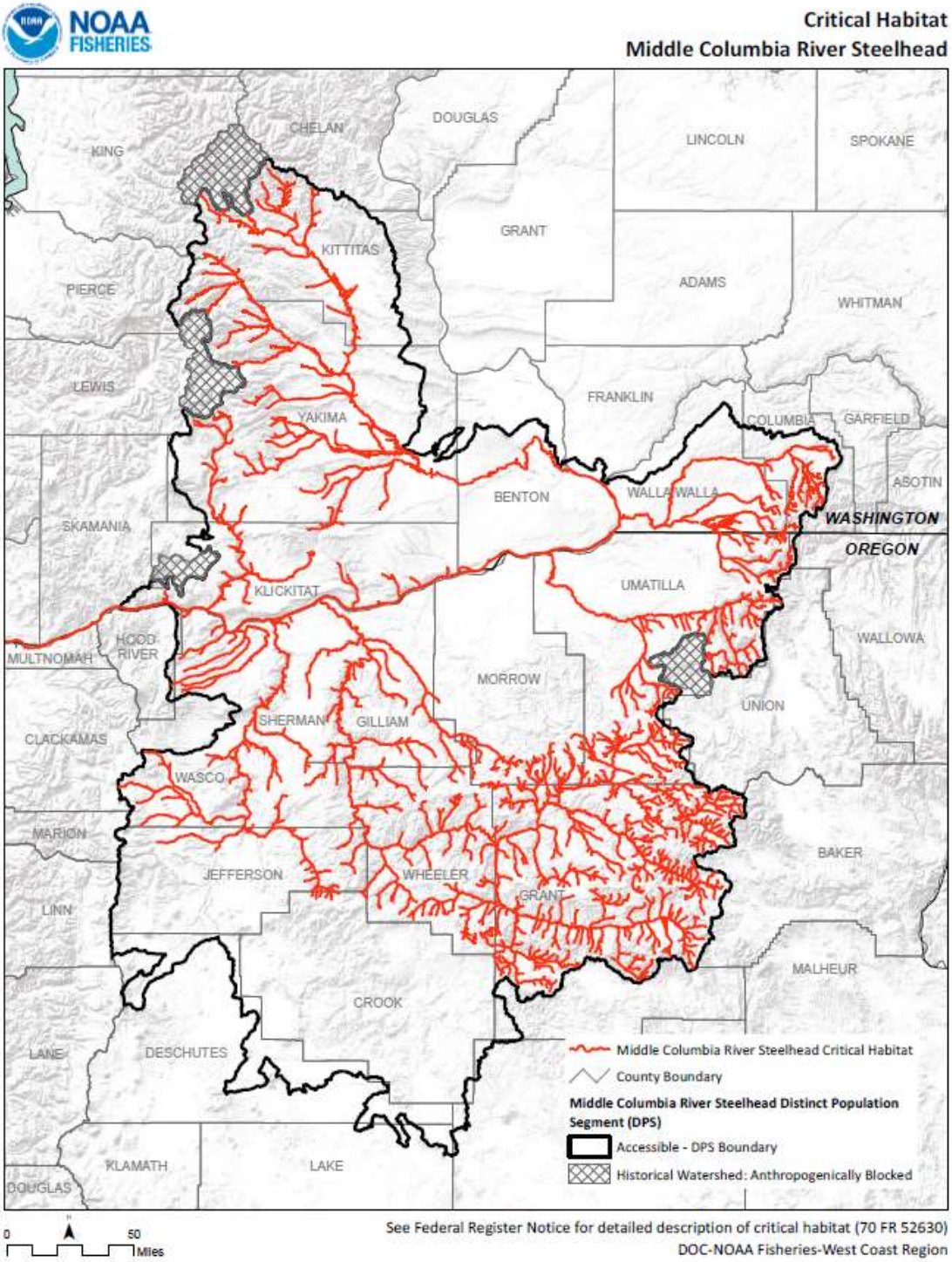
3.7.3.1 Distribution

The MCR steelhead DPS includes all naturally spawned anadromous *O. mykiss* populations below impassable barriers in streams from above the Wind River in Washington and the Hood River in Oregon, upstream to, and including, the Yakima River in Washington but excluding *O. mykiss* from the Snake River Basin. Seven artificial propagation programs are also included in the DPS.

There are 17 extant populations (and three historically extirpated populations) in the MCR steelhead DPS (ICTRT 2005; 2003). The populations are further classified into four Major Population Groups: John Day River (five extant populations), Umatilla/Walla Walla Rivers (three extant and one extirpated populations), Yakima River (four extant populations), and the Eastern Cascades group (five extant and two extirpated populations).

3.7.3.2 Critical Habitat

Critical habitat for the MCR steelhead DPS was designated for Oregon and Washington in 2005 and includes freshwater areas shown in Figure 3-11 (NMFS, 2005c). The PBFs for West Coast steelhead critical habitat are provided in Table SRBS.



Source: (NOAA Fisheries, 2016e)

Figure 3-11. Critical Habitat for Middle Columbia River Steelhead Trout

3.7.3.3 Life History

MCR steelhead trout follow a summer-run pattern (consistent with other inland steelhead), and they mature in streams for up to one year before spawning (DOI, 2011). Spawning migration starts in mid-May, and fish pass over Bonneville Dam in July and August (DOI, 2011). Fry emerge from gravel between May and June, and juvenile MCR steelhead tend to smolt after two years in freshwater streams, after which they spend one to three years in the ocean before returning to freshwater (DOI, 2011). MCR steelhead co-occur with non-anadromous rainbow trout, and they may not be reproductively isolated (Carmichael, 2006).

3.7.3.4 Current Stressors and Threats

Stressors and threats to the MCR steelhead DPS and critical habitat are similar to those for the UCR steelhead DPS.

Per NOAA's 2016 status review (NOAA Fisheries, 2016i), the Middle Columbia River Steelhead DPS should remain listed as threatened.

3.7.4 Lower Columbia River Steelhead Trout

Lower Columbia River steelhead were federally listed as threatened in 2006 (NMFS, 2006a).

3.7.4.1 Distribution

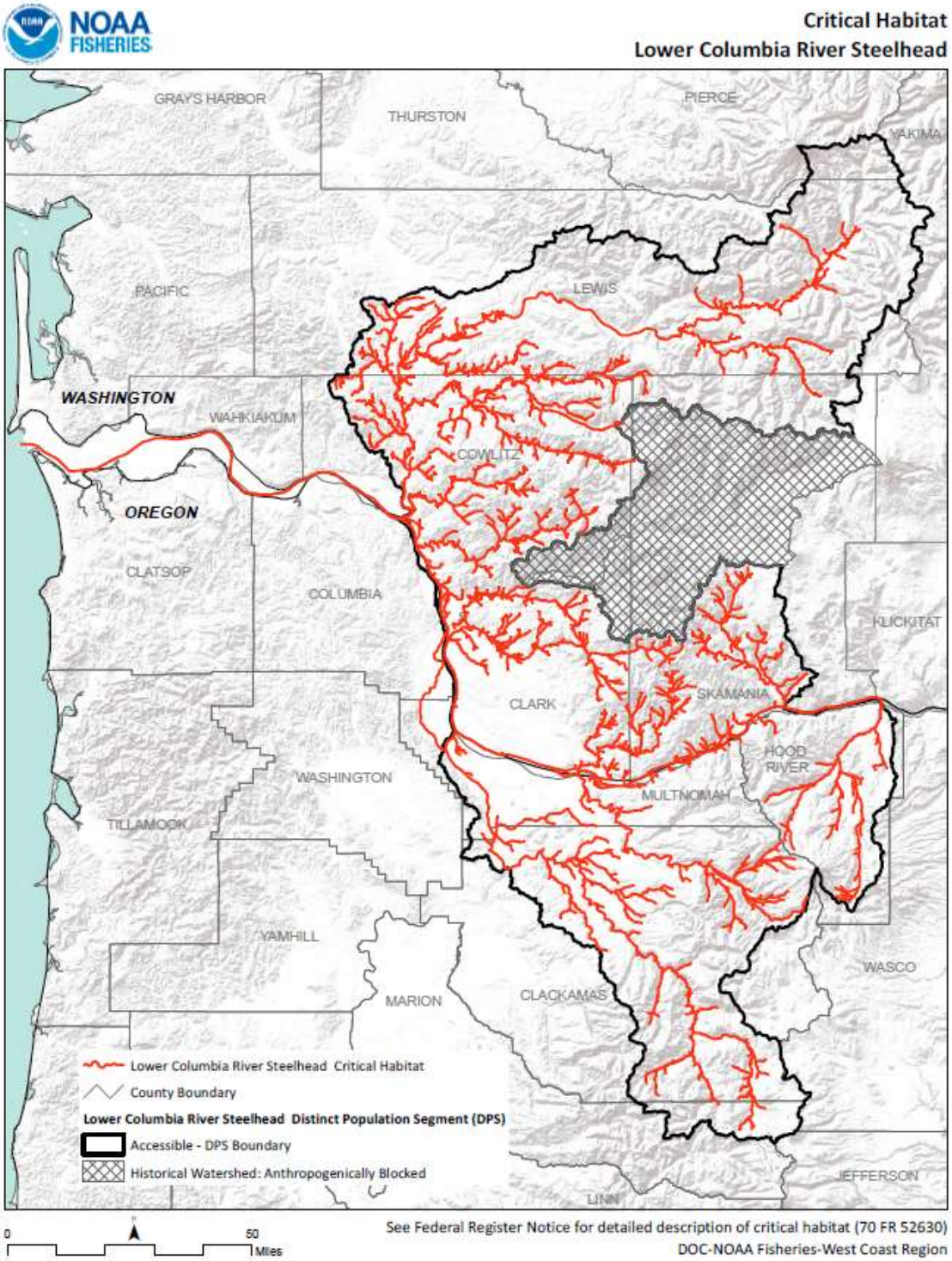
The LCR steelhead DPS includes all naturally spawned anadromous *O. mykiss* originating from below impassable barriers from rivers between the Cowlitz and Wind Rivers (inclusive) and the Willamette and Hood Rivers (inclusive). The DPS excludes fish originating from the UWR Basin above Willamette Falls. This DPS also includes steelhead from seven artificial propagation programs (NOAA Fisheries, 2014).

3.7.4.2 Critical Habitat

Critical habitat for LCR steelhead trout was designated for Oregon and Washington in 2005 and includes areas shown in Figure 3-12 (NMFS, 2005c). The PBFs for LCR steelhead critical habitat are described in Table SRBS.

3.7.4.3 Life History

Steelhead in the LCR smolt after two years spent in freshwater, then spend an additional two years in marine waters before returning to freshwater to spawn (NOAA Fisheries, 2015b). Steelhead may linger in freshwater streams for up to a year before spawning (NOAA Fisheries, 2015b). Unlike other Pacific salmonids, steelhead are iteroparous (can spawn multiple times), though multiple spawning events are rare and mostly restricted to females (Nickelson et al., 1992), as cited by NOAA Fisheries (2015b).



Source: (NOAA Fisheries, 2016d)

Figure 3-12. Critical Habitat for Lower Columbia River Steelhead Trout

3.7.4.4 Current Stressors and Threats

Limitations for the LCR steelhead DPS include interactions with hatchery fish (resulting in reduced genetic diversity), limited fish passage at dams, and habitat degradation. The conversion of floodplain habitat to agricultural or urbanized land uses has reduced steelhead habitat availability throughout the LCR region. Channelization and hydrological changes have reduced habitat complexity in the lower tributary/mainstem Columbia River interface, and the concomitant change in water temperatures is a likely stressor on the LCR steelhead DPS (LCFRB, 2010; NMFS, 2013; ODFW, 2010). Contamination is a growing concern for aquatic life and designated uses of the LCR (Alvarez et al., 2014; Counihan et al., 2013; Morace, 2012; Nilsen & Morace, 2014; Nilsen et al., 2014) and Total Maximum Daily Loads are needed for many pollutants.

In NOAA's 2016 status report, they determined that the DPS continues to be at moderate risk (NOAA Fisheries, 2015b).

3.7.5 Upper Willamette River Steelhead Trout

A recovery plan was published for the UWR steelhead trout DPS in 2011 (ODFW & NMFS, 2011). Recovery goals for this DPS include the maintenance of all populations, including two populations with high probabilities of persistence.

3.7.5.1 Distribution

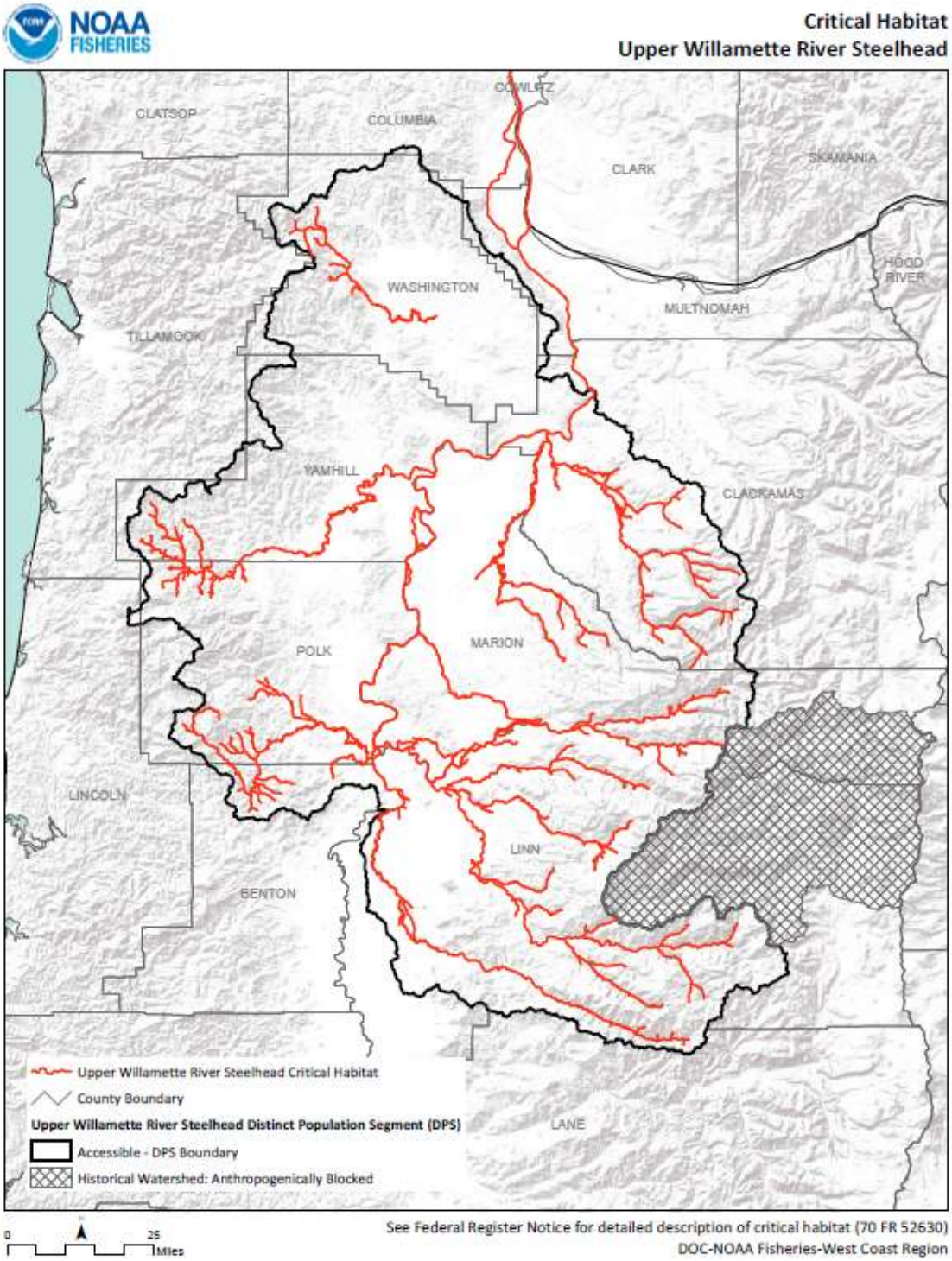
This species includes all naturally spawned steelhead populations below impassable barriers in the Willamette River, Oregon, and its tributaries upstream from Willamette Falls to and including the Calapooia River (USDI, 2016). One stratum and four extant populations of UWR steelhead occur within the DPS. Hatchery summer-run steelhead that are released in the subbasins are from an out-of-basin stock, so they are not included in the DPS. Additionally, historically hatchery-raised summer-run steelhead in the McKenzie River (now an established population) are not included in the DPS (ODFW & NMFS, 2011).

3.7.5.2 Critical Habitat

Critical habitat for the UWR steelhead DPS was designated for Oregon in 2005 and includes freshwater areas shown in Figure 3-13 (NMFS, 2005c). The PBFs for UWR steelhead critical habitat are described in Table SRBS.

3.7.5.3 Life History

UWR steelhead migrate in winter, entering freshwater between January and April (ODFW & NMFS, 2011). Like other steelhead DPSs, juveniles hatch and rear for two years before smoltifying, after which they migrate into the ocean for another two years. Spawning occurs at four or five years of age (NOAA Fisheries, 2015b).



Source: (NOAA Fisheries, 2016k)

Figure 3-13. Critical Habitat for Upper Willamette River Steelhead Trout

3.7.5.4 Current Stressors and Threats

Limiting factors for UWR steelhead DPS include degradation of freshwater habitat (e.g., reductions in floodplain connectivity and function, channel structure and complexity, riparian habitat availability, large wood recruitment, and stream flow); degraded water quality (e.g., increased water temperature); reduced access to spawning and rearing habitats; reduced genetic diversity resulting from summer-run steelhead hatchery releases; and predation and competition related to non-native species and out-of-basin salmonid populations (NMFS, 2011b; ODFW & NMFS, 2011).

Per NOAA's 2016 status update (NOAA Fisheries, 2016k), the Upper Willamette River Steelhead Trout DPS should remain listed as threatened.

3.8 Bull Trout (*Salvelinus confluentus*)

The coterminous US population of bull trout was listed as threatened on November 1, 1999 (USFWS, 1999).

3.8.1 Distribution

Bull trout generally occur in the Klamath River Basin of south-central Oregon; the Jarbidge River in Nevada; the Willamette River Basin in Oregon; Pacific Coast drainages of Washington, including Puget Sound; major rivers within the Columbia River Basin in Idaho, Oregon, Washington, and Montana; and the St. Mary-Belly River, east of the Continental Divide in northwestern Montana (Bond, 1992; Brewin & Brewin, 1997; Cavender, 1978; Leary & Allendorf, 1997).

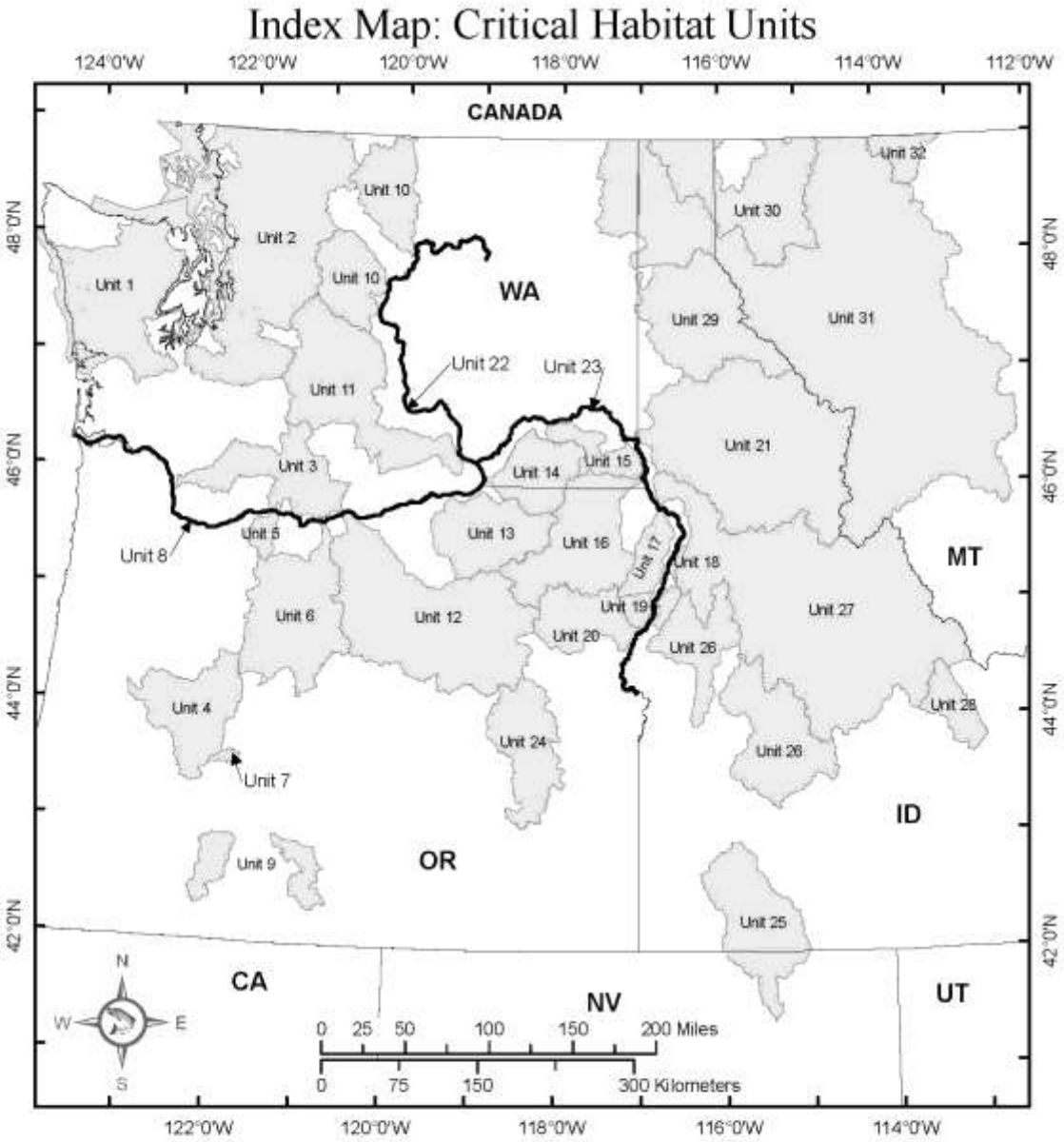
3.8.2 Critical Habitat

A final ruling on critical habitat for bull trout in the coterminous U.S. was made on October 18, 2010 (effective November 17, 2010) (USFWS, 2010). Critical habitat for bull trout includes approximately 32,187 km (20,000 miles) of riverine habitat, 1,207 km (750 miles) of marine shoreline, and 197,487 ha (488,000 acres) of lacustrine habitat. Critical habitat spans Washington, Oregon, Idaho, Nevada, and Montana (Figure 3-14). The PBFs of bull trout critical habitat include space for individual and population growth and for normal behavior; food, water, air, light, minerals, or other nutritional or physiological requirements; cover or shelter; sites for breeding, reproduction, or rearing (or development) of offspring; and habitats that are protected from disturbance or are representative of the historical, geographical, and ecological distributions of a species (USFWS, 2010).

The physical or biological features (PBFs) determined to be essential to the conservation of bull trout are:

- Springs, seeps, groundwater sources, and subsurface water connectivity (hyporheic flows) to contribute to water quality and quantity and provide thermal refugia;
- Migration habitats with minimal physical, biological, or water quality impediments between spawning, rearing, overwintering, and freshwater and marine foraging habitats, including but not limited to permanent, partial, intermittent, or seasonal barriers;

- An abundance of food, including terrestrial organisms of riparian origin, aquatic macroinvertebrates, and forage fish;
- Complex shorelines with features such as large wood, side channels, pools, undercut banks, and unembedded substrates, to provide a variety of depths, gradients, velocities, and structure;
- Water temperatures ranging from 2 to 15°C (36 to 59°F), with adequate thermal refugia available for temperatures that exceed the upper end of this range;
- Sufficient and appropriate substrate in spawning and rearing areas;
- Water flows approximating natural timing (historic and seasonal ranges) for peak, high, low, and base flow;
- Sufficient water quality and quantity to sustain normal reproduction, growth, and survival; and
- Low occurrence of nonnative predatory (e.g., lake trout, walleye, northern pike, smallmouth bass), interbreeding (e.g., brook trout), or competing (e.g., brown trout) species.



1	Olympic Peninsula	11	Yakima River	22	Mainstem Upper Columbia River
2	Puget Sound	12	John Day River	23	Mainstem Snake River
3	Lower Columbia River Basins	13	Umatilla River	24	Malheur River Basin
4	Upper Willamette River	14	Walla Walla River Basin	25	Jarbidge River
5	Hood River	15	Lower Snake River Basins	26	Southwest Idaho River Basins
6	Lower Deschutes River	16	Grande Ronde River	27	Salmon River Basin
7	Odell Lake	17	Imnaha River	28	Little Lost River
8	Mainstem Lower Columbia River	18	Sheep / Granite Creeks	29	Coeur d'Alene River Basin
9	Klamath River Basin	19	Hells Canyon Complex	30	Kootenai River Basin
10	Upper Columbia River Basins	20	Powder River Basin	31	Clark Fork River Basin
		21	Clearwater River	32	Saint Mary River Basin

Source: (USFWS, 2010)

Figure 3-14. Critical Habitat Units for Bull Trout of the Coterminous U.S.

3.8.3 Life History

Bull trout exhibit both resident and migratory life history strategies. Both resident and migratory forms may be found together, and either form may produce offspring exhibiting either resident or migratory behavior (B E Rieman & McIntyre, 1993). Resident bull trout complete their entire life cycle in or near tributary streams where they spawn and rear. Migratory bull trout spawn in tributary streams where juvenile fish rear for one to four years before migrating to a lake, river (Fraley & Shepard, 1989; Goetz, 1989), or saltwater (Cavender, 1978; McPhail & Baxter, 1996; WDFW, 1997). Bull trout reach sexual maturity in four to seven years and may live longer than 12 years. They are iteroparous, meaning that they may spawn more than once in a lifetime. Bull trout typically spawn from August through November during periods of increasing flows and decreasing water temperatures. Spawning areas are often associated with cold-water springs, groundwater infiltration, and the coldest streams in a given watershed (Baxter, Taylor, Devlin, Hagen, & McPhail, 1997; Pratt, 1992; Bruce E Rieman, Lee, & Thurow, 1997; B E Rieman & McIntyre, 1993). Fry normally emerge from early April through May, depending on water temperatures and increasing stream flows (Howell & Buchanan, 1992; Pratt, 1992; Ratliff & Howell, 1992). Bull trout are primarily found in colder streams (below 15°C; 59°F) (Fraley & Shepard, 1989; Pratt, 1992; B E Rieman & McIntyre, 1993), though they may be found in warmer waters that have access to colder refuges.

Bull trout have more specific habitat requirements than most other salmonids (B E Rieman & McIntyre, 1993). Habitat components that influence bull trout distribution and abundance include water temperature (as described above), availability of cover, channel form and stability, valley form, spawning and rearing substrate, and migratory corridors (Bond, 1992; Fraley & Shepard, 1989; Goetz, 1989; Hoelscher & Bjornn, 1989; Howell & Buchanan, 1992; Pratt, 1992; Rich, 1996; B E Rieman & McIntyre, 1993; Bruce E Rieman & McIntyre, 1995; Sedell & Everest, 1991; G. Watson & Hillman, 1997). All life history stages of bull trout are associated with complex forms of cover, including large woody debris, undercut banks, boulders, and pools (Fraley & Shepard, 1989; Goetz, 1989; Hoelscher & Bjornn, 1989; Pratt, 1992; Rich, 1996; Sedell & Everest, 1991; Sexauer & James, 1997; Thomas, 1992; G. Watson & Hillman, 1997). Early life stages of bull trout, specifically the developing embryo, require the highest inter-gravel dissolved oxygen levels, and are the most sensitive life stage to reduced oxygen levels. The oxygen demand of embryos depends on temperature and stage of development, with the greatest dissolved oxygen required just prior to hatching.

Bull trout are opportunistic feeders, with food habits primarily a function of size and life-history strategy. Resident and juvenile migratory bull trout prey on terrestrial and aquatic insects, macrozooplankton, and small fish (Boag, 1987; Donald & Alger, 1993; Goetz, 1989). Bull trout may also feed heavily on fish eggs of other salmon (Lowery & Beauchamp, 2015). Subadult and adult migratory bull trout feed on various fish species (L. G. Brown, 1994; Donald & Alger, 1993; Fraley & Shepard, 1989). In marine nearshore areas of western Washington, bull trout feed on Pacific herring, Pacific sand lance, and surf smelt (Goetz, Jeanes, & Beamer, 2004; WDFW, 1997). Bull trout of sizes greater than fry have been found to eat fish up to half their length (Beauchamp & Van Tassell, 2001).

3.8.4 Current Stressors and Threats

Throughout their range, bull trout are threatened by the combined effects of habitat degradation, fragmentation, and alterations associated with dewatering, road construction and maintenance, mining, grazing, the blockage of migratory corridors by dams or other diversion structures, entrainment in diversion channels, and introduced non-native species (USFWS, 1999). Although all salmonids are likely to be affected by climate change, bull trout are especially vulnerable given that spawning and rearing are constrained by their location in upper watersheds and the requirement for cold water temperatures (Battin et al., 2007; B. Rieman et al., 2007). Additional threats to bull trout include industrial development and urbanization, timber harvest, and poaching or bycatch.

The iteroparous reproductive strategy of bull trout has important repercussions for the management of this species. Bull trout require passage both upstream and downstream for both spawning and foraging, and passage must be allowed for multiple spawning migrations. However, most fish ladders were designed specifically for anadromous, semelparous salmonids (spawning once before death). Therefore, fish passage facilities (e.g., fish ladders) at barriers to migration may be a factor in isolating bull trout populations because they do not provide downstream passage for adults and subadults. Additionally, in some core areas, bull trout that migrate to marine waters must pass both upstream and downstream through areas with net fisheries at river mouths. This can increase the likelihood of mortality during spawning and foraging migrations.

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. Average regional temperatures are likely to increase by 3°F to 10°F over the next century (USGCRP, 2009). Overall, about one-third of the current cold-water habitat in the Pacific Northwest is likely to exceed water temperature thresholds for bull trout by the end of this century (USGCRP, 2009). Significant reductions in both total snow pack and low-elevation snow pack in the Pacific Northwest are predicted over the next 50 years (Mote & Salathé, 2010), which will shrink the extent of the snowmelt-dominated habitat available to salmonids and cause warmer temperatures after snowmelt has run off (ISAB, 2007; USGCRP, 2009). As the snow pack diminishes and seasonal hydrology shifts to more frequent and severe early large storms, stream flow timing and increased peak river flows may limit salmonid survival (Mantua, Tohver, & Hamlet, 2010). Similarly, marine conditions adverse to salmonids may be more likely under a warming climate (Zabel, Scheuerell, McClure, & Williams, 2006).

3.9 Oregon Spotted Frog (*Rana pretiosa*)

The Oregon spotted frog was listed as threatened on August 29, 2014 (USFWS, 2014). This species is named for the black spots that cover the head, back, sides, and legs. The dark spots are characterized by ragged edges and light centers that grow and darken with age (USFWS, 2019b). Body color also varies with age. Juveniles are usually brown or, occasionally, olive green on the back and white, cream, or flesh-colored with reddish pigments on the underlegs and abdomen, developing with age (McAllister & Leonard, 1997). Adults range from brown to reddish brown but tend to become redder with age. The spotted frog is medium-sized, ranging from 4.3 to 10.1 cm (1.7 to 4 inches) in body length. Females are typically larger than males and can reach up to S10 cm (4 inches) or more (USFWS, 2013a).

3.9.1 Species Distribution

Historically, the Oregon spotted frog ranged from British Columbia to the Pit River basin in northeastern California (McAllister & Leonard, 1997). Oregon spotted frogs have been documented at 61 historical localities in 48 watersheds (three in British Columbia, 13 in Washington, 29 in Oregon, and three in California) in 31 sub-basins (COSEWIC, 2011b; McAllister & Leonard, 1997; USFWS, 2014).

Currently, the Oregon spotted frog is found within 15 sub-basins, ranging from extreme southwestern British Columbia south through the Puget Trough, and the Cascades Range from south-central Washington to at least the Klamath Basin in southern Oregon. Oregon spotted frogs occur in lower elevations in British Columbia and Washington and are restricted to high elevations in Oregon (C. Pearl, Clayton, & Turner, 2010). In addition, Oregon spotted frogs currently have a very limited distribution west of the Cascade crest in Oregon and are considered to be extirpated from the Willamette Valley in Oregon (Cushman & Pearl, 2007).

In Washington, Oregon spotted frogs are known to occur only within six sub-basins/watersheds: the Sumas River, a tributary to the Lower Chilliwack River watershed and Fraser River sub-basin; the Black Slough in the lower South Fork Nooksack River, a tributary of the Nooksack River; the Samish River; the Black River, a tributary of the Chehalis River; Outlet Creek (Conboy Lake), a tributary to the Middle Klickitat River; and Trout Lake Creek, a tributary of the White Salmon River. The Klickitat and White Salmon Rivers are tributaries to the Columbia River. The Oregon spotted frogs in each of these sub-basins/watersheds are isolated from frogs in other sub-basins (USFWS, 2014).

In Oregon, Oregon spotted frogs are known to occur only within eight sub-basins: Lower Deschutes River, Upper Deschutes River, Little Deschutes River, McKenzie River, Middle Fork Willamette, Upper Klamath, Upper Klamath Lake, and the Williamson River. The Oregon spotted frogs in most of these sub-basins are isolated from frogs in other sub-basins, although Oregon spotted frogs in the lower Little Deschutes River are aquatically connected with those below Wickiup Reservoir in the Upper Deschutes River sub-basin. Oregon spotted frog distribution west of the Cascade Mountains in Oregon is restricted to a few lakes in the upper watersheds of the McKenzie River and Middle Fork Willamette River sub-basins, which represent the remaining 2 out of 12 historically occupied sub-basins west of the Cascades in Oregon (USFWS, 2014).

3.9.2 Critical Habitat

The USFWS designated critical habitat for Oregon spotted frog on 26,319 ha (65,038 acres) and 32.7 stream km (20.3 stream miles) in Washington and Oregon on May 11, 2016 (USFWS, 2016). Critical habitat for the Oregon spotted frog is proposed within 14 units, delineated by river sub-basins where spotted frogs are extant: (1) Lower Chilliwack River; (2) South Fork Nooksack River; (3) Samish River; (4) Black River; (5) White Salmon River; (6) Middle Klickitat River; (7) Lower Deschutes River; (8) Upper Deschutes River; (9) Little Deschutes River; (10) McKenzie River; (11) Middle Fork Willamette River; (12) Williamson River; (13) Upper Klamath Lake; and (14) Upper Klamath. Descriptions of ownership, acreages, and threats for each unit are stated in the critical habitat designation (USFWS, 2016).

Three Primary Constituent Elements exist for this species. These are:

- Primary constituent element 1.— Nonbreeding (N), Breeding (B), Rearing mstockstill on DSK3G9T082PROD with RULES2 VerDate Sep2014 17:56 May 10, 2016 Jkt 238001 PO 00000 Frm 00041 Fmt 4701 Sfmt 4700 E:\FR\FM\11MYR2.SGM 11MYR2 29376 Federal Register / Vol. 81, No. 91 / Wednesday, May 11, 2016 / Rules and Regulations (R), and Overwintering (O) Habitat. Ephemeral or permanent bodies of fresh water, including, but not limited to, natural or manmade ponds, springs, lakes, slow-moving streams, or pools within or oxbows adjacent to streams, canals, and ditches, that have one or more of the following characteristics: (A) Inundated for a minimum of 4 months per year (B, R) (timing varies by elevation but may begin as early as February and last as long as September); (B) Inundated from October through March (O); (C) If ephemeral, areas are hydrologically connected by surface water flow to a permanent water body (e.g., pools, springs, ponds, lakes, streams, canals, or ditches) (B, R); (D) Shallow-water areas (less than or equal to 12 inches (30 centimeters), or water of this depth over vegetation in deeper water (B, R); (E) Total surface area with less than 50 percent vegetative cover (N); (F) Gradual topographic gradient (less than 3 percent slope) from shallow water toward deeper, permanent water (B, R); (G) Herbaceous wetland vegetation (i.e., emergent, submergent, and floating-leaved aquatic plants), or vegetation that can structurally mimic emergent wetland vegetation through manipulation (B, R); (H) Shallow-water areas with high solar exposure or low (short) canopy cover (B, R); and (I) An absence or low density of nonnative predators (B, R, N).
- Primary constituent element 2.— Aquatic movement corridors. Ephemeral or permanent bodies of fresh water that have one or more of the following characteristics: (A) Less than or equal to 3.1 miles (5 kilometers) linear distance from breeding areas; and (B) Impediment free (including, but not limited to, hard barriers such as dams, impassable culverts, lack of water, or biological barriers such as abundant predators, or lack of refugia from predators).
- Primary constituent element 3.— Refugia habitat. Nonbreeding, breeding, rearing, or overwintering habitat or aquatic movement corridors with habitat characteristics (e.g., dense vegetation and/or an abundance of woody debris) that provide refugia from predators (e.g., nonnative fish or bullfrogs). (3) Critical habitat does not include manmade structures (such as buildings, aqueducts, runways, roads, and other paved areas) and the land on which they are located existing within the legal boundaries on June 10, 2016.

3.9.3 Life History

The Oregon spotted frog is highly aquatic; it is almost always found in or near a perennial body of water that includes zones of shallow water and abundant emergent or floating aquatic plants, which it uses for basking and cover. Conditions required for completion of the species' life cycle are shallow water areas for egg and tadpole survival; perennially deep, moderately vegetated pools for adult and juvenile survival in the dry season; and perennial water for protecting all age classes during cold, wet weather (J. W. Watson, McAllister, & Pierce, 2003).

Oregon spotted frogs breed in shallow pools near flowing water or in shallow pools that may be connected to larger bodies of water during seasonally high water or at flood stage. These locations are most often defined by shallow, often temporary, pools of water; gradually receding shorelines; location on benches of seasonal lakes and marshes; or location in wet meadows. These sites are usually associated with the previous year's emergent vegetation and are generally no more than 14 inches deep (C. A. Pearl & Hayes, 2004).

Oregon spotted frogs concentrate breeding efforts in relatively few locations (Hayes et al., 2001; McAllister & White, 2001). The availability of the unique characteristics of egg-laying sites is limited, and adults may have limited flexibility to switch sites. This inflexibility may make the Oregon spotted frog particularly vulnerable to modification of egg-laying sites (USFWS, 2014).

After breeding, during the dry season, Oregon spotted frogs move to deeper, permanent pools or creeks, where they are often observed near the water surface basking and feeding in beds of floating and submerged vegetation (G. Watson & Hillman, 1997). Larger sites are more likely to provide the seasonal microhabitats required by Oregon spotted frogs, have a more reliable prey base, and include overwintering habitat. It is thought that a minimum wetland size of 3.6 ha (9 acres) may be necessary to reach suitably warm temperatures and support a large enough population to persist despite high predation rates (Hayes, 1994). However, Oregon spotted frogs also occupy smaller sites and are known to occur at sites as small as 1 ha (2.5 acres) and as large as 1,989 ha (4,915 acres) (C. A. Pearl & Hayes, 2004). Smaller sites generally have a small number of frogs and, as described above, are more vulnerable to extirpation. (C. A. Pearl & Hayes, 2004) believe that these smaller sites were historically subpopulations within a larger breeding complex and that Oregon spotted frogs may only be persisting in these small sites because the sites exchange migrants or because seasonal habitat needs are provided nearby.

Known overwintering sites for the Oregon spotted frog are associated with flowing systems, such as springs and creeks, that provide water with high oxygen content (Hayes et al., 2001; Tattersall & Ultsch, 2008) and sheltering locations protected from predators and freezing (J. W. Watson et al., 2003). Oregon spotted frogs burrow in mud, silty substrate, clumps of emergent vegetation, woody accumulations within the creek, and holes in creek banks when inactive during periods of prolonged or severe cold (McAllister & Leonard, 1997; J. W. Watson et al., 2003). They are intolerant of anoxic conditions and are unlikely to burrow into the mud for more than a day or two because survival under anoxic conditions is only a matter of four to seven days (Tattersall & Ultsch, 2008). This species remains active during the winter and selects microhabitats that can support aerobic metabolism and minimize exposure to predators (Hayes et al., 2001; Tattersall & Ultsch, 2008).

Oregon spotted frog tadpoles are grazers, having rough tooth rows for scraping plant surfaces and ingesting plant tissue and bacteria. They also consume algae, detritus, and probably carrion. Post-metamorphic spotted frogs feed on live animals, primarily insects (USFWS, 2019b).

3.9.4 Current Stressors and Threats

Large historical losses of wetland habitat have occurred across the range of the Oregon spotted frog. Wetland losses are estimated at 30% to 85% across the species' range, with the greatest percentage lost having occurred in British Columbia. These wetland losses have directly influenced the current fragmentation and isolation of remaining Oregon spotted frog populations

(USFWS, 2014). The historical loss of Oregon spotted frog habitat and lasting anthropogenic changes in natural disturbance processes are exacerbated by the introduction of reed canarygrass (*Phalaris arundinacea*), nonnative predators, and, potentially, climate change. In addition, current regulatory mechanisms and voluntary incentive programs designed to benefit fish species have inadvertently led to the continuing decline in quality of Oregon spotted frog habitat in some locations (USFWS, 2014).

In the Final Rule to list the frog as threatened, the USFWS determined that the Oregon spotted frog is impacted by one or more of the following factors to the extent that the species meets the definition of a threatened species under the ESA:

- Habitat necessary to support all life stages continuing to be impacted and/or destroyed by human activities that result in the loss of wetlands to land conversions;
- Hydrologic changes resulting from operation of existing water diversions/ manipulation structures, new and existing residential and road developments, drought, and removal of beavers (*Castor canadensis*);
- Changes in water temperature and vegetation structure resulting from reed canarygrass invasions, plant succession, and restoration plantings;
- Increased sedimentation, increased water temperatures, reduced water quality, and vegetation changes resulting from the timing and intensity of livestock grazing (or, in some instances, removal of livestock grazing at locations where it maintains early seral stage habitat essential for breeding);
- Predation by nonnative species, including nonnative trout and bullfrogs;
- Inadequate existing regulatory mechanisms that result in significant negative impacts such as habitat loss and modification; and
- Other natural or manmade factors, including small and isolated breeding locations, low connectivity, low genetic diversity within occupied sub-basins, and genetic differentiation between sub-basins.

The Oregon spotted frog faces cumulative effects from threats as well. All occupied sub-basins are subjected to multiple threats, which cumulatively pose a risk to individual populations. Many of these threats are intermingled, and the magnitude of the combined threats to the species is greater than the individual threats (USFWS, 2014).

3.10 Lost River Sucker (*Deltistes luxatus*)

Lost River sucker was listed as endangered on July 18, 1988 (USFWS, 1988), along with the shortnose sucker (*Chasmistes brevirostris*), which shares much of the same geographic range. A final recovery plan for the Lost River and shortnose suckers was published in 1993 and was revised in 2012 (USFWS, 2012c).

3.10.1 Distribution

Lost River sucker was historically present in Upper Klamath, Tule, Lower Klamath, and Clear Lakes and their tributaries, though the species is now limited to the Upper Klamath River and its tributaries and outlet as well as the Iron Gate Reservoir, J.C. Boyle Reservoir, and Sheepy and Lower Klamath Lakes (USFWS, 2012c). Of those water bodies, only the Upper Klamath Lake

and J.C. Boyle Reservoirs are in the proposed action area (Oregon). Strong spawning populations were historically found in the Lost River system and in Big Springs near Bonanza, Oregon (USFWS, 2016). Current spawning areas are concentrated in Upper Klamath Lake, the Williamson River (from RM 6 to the confluence with the Sprague River), and in sections of the Sprague River. The Chiloquin Dam, which was removed in 2008, provided a significant hindrance to sucker migration in the Sprague River for nearly a century (Ellsworth, Tyler, VanderKooi, & Markle, 2009). The removal of the dam should allow for a large number of spawning Lost River sucker to migrate up the Sprague River and into its tributaries, though the effect of the dam removal is not yet known.

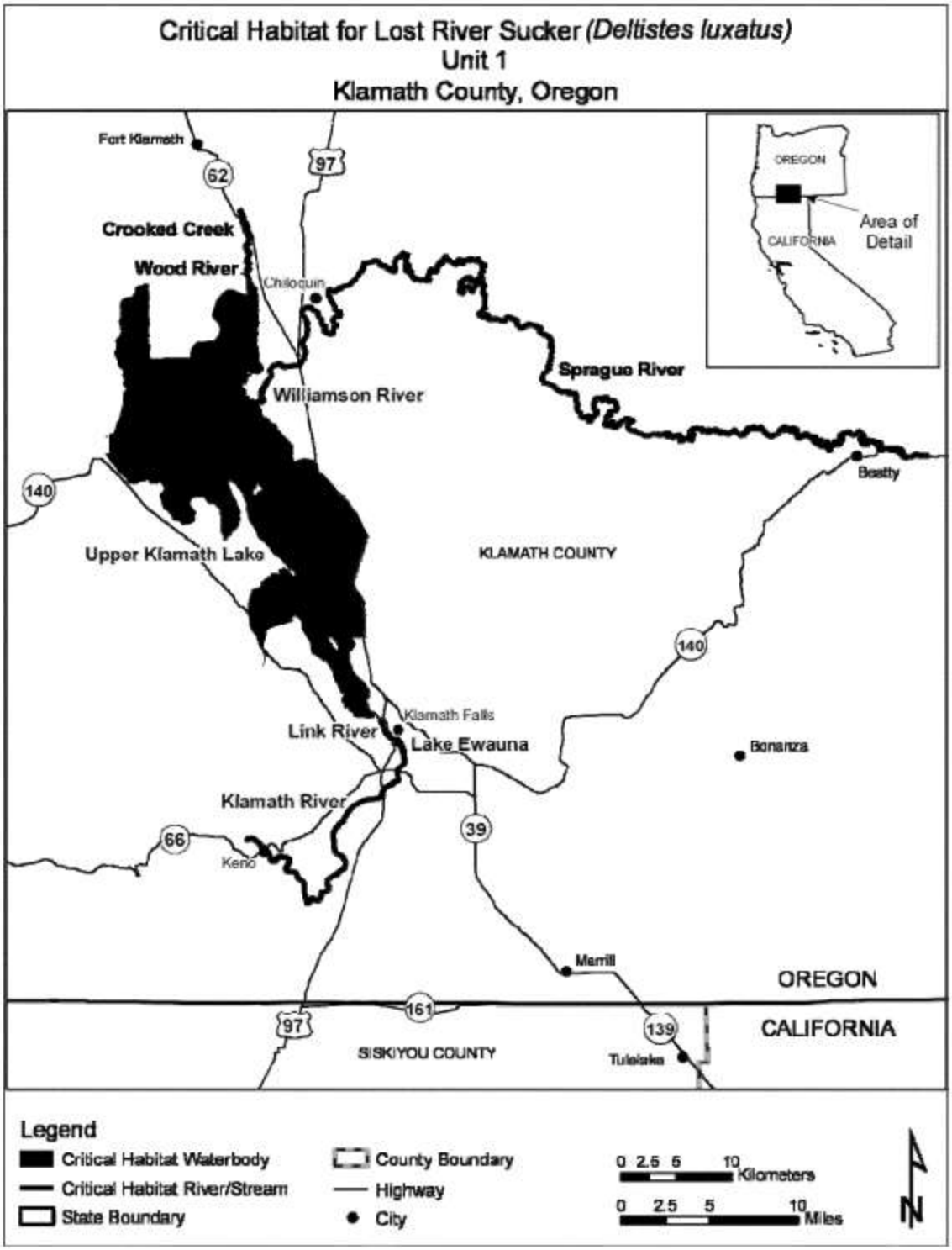
3.10.2 Critical Habitat

Critical Habitat for the Lost River sucker (and shortnose sucker) was listed on December 11, 2012 (USFWS, 2012b), effective January 10, 2013. A total of 235 km (146 miles) of streams and 47,691 ha (117,848 acres) of lake and reservoir habitat were protected in this way for Lost River sucker. The critical habitat is distributed in Klamath and Lake Counties, Oregon, and Modoc County, California. Figure 3-15 presents the location of Lost River sucker critical habitat in Oregon. Water bodies included in critical habitat include Upper Klamath Lake; the Williamson, Sprague, and Wood Rivers and Crooked Creek upstream of Upper Klamath Lake; and the Link and Klamath Rivers and Lake Ewauna downstream of Upper Klamath Lake.

The PBFs for Lost River and shortnose suckers include sufficient water quantity, complexity, depth, quality, and connectivity for survival including multiple water depths for various life stages, water temperatures <28°C (82.4°F), water pH <9.75 [11], dissolved oxygen concentrations > 4 milligrams per liter, low levels of microcystin[12], and unionized ammonia <0.5 milligrams per liter. Natural (or like-natural) flow regimes are another PBF for sucker critical habitat. Spawning and rearing habitats must have suitable substrates (e.g., gravel and cobble) in <1.3 m (4.3 feet) of water with an adequate stream velocity and emergent vegetation. Critical habitats must also provide an adequate supply of food items (e.g., insects and crustaceans).

[11] This upper limit is specified in (USFWS, 2012b), but a lower limit is not. Assumedly very low pH would also be detrimental to suckers.

[12] Microcystins are a group of toxic chemicals produced naturally by cyanobacteria.



Source: (USFWS, 2012b)

Figure 3-15. Critical Habitat for the Lost River Sucker in Oregon

3.10.3 Life History

Lost River sucker spawn from February to May in streams and rivers with gravel substrates (Buettner & Scoppettone, 1990) as cited in (ODFW, 2016) and in shallow, stream-fed areas along the east side of Upper Klamath Lake (Buettner & Scoppettone, 1990), as cited by (Leary & Allendorf, 1997; ODFW, 2016) as cited by (ODFW, 2016). Individual female Lost River suckers produce between 44,000 and 200,000 eggs per year, with older females producing far more eggs than younger, smaller females. A small fraction of fertilized embryos survive to hatch. Lost River sucker reach maturity at between four and nine years old and may live up to 55 years, spawning many times in their lifetime. Larval suckers emerge from gravels soon after hatching (by mid-July) and drift into lakes or rivers to rear (M. Cooperman & Markle, 2003); (USFWS, 2016).

Lost River sucker use several habitat types throughout their life including stream, river, lake, marsh, and shoreline spring habitats of varying depths (USFWS, 2012c). Adults tend to use open waters of lakes (between 1.5 and 3 m deep [5 and 10 feet deep]) when not spawning, though some individuals may stay in rivers, particularly in the Lost River. Upon entering lakes, larvae tend to seek out cover provided by emergent vegetation along lake shorelines and in shallow water (between 0.15 and 0.91 m deep [0.5 and 3 feet deep]) (M. S. Cooperman & Markle, 2004). Juveniles, as they grow, shift from shallower to deeper lake waters away from emergent vegetation and over substrates ranging from mud to cobble(Burdick, Hendrixson, & VanderKooi, 2008). Habitat use may be driven by patterns in dissolved oxygen, which has a pronounced effect on juvenile sucker growth and survival (Rasmussen, 2011); (Perkins, Kann, & Scoppettone, 2000); (Saiki, Monda, & Bellerud, 1999).

The size structure of the Lost River sucker population shifted from larger to smaller individuals between the mid-1980s and mid-1990s, resulting from significant recruitment of young individuals born in the early 1990s and relatively poor recruitment before then (Janney, Shively, Hayes, & Barry, 2008); (Terwilliger, Reece, & Markle, 2010). Recruitment between the mid-1990s and mid-2000s appeared to be lower, as the size structure trended again toward larger individuals (Janney et al., 2008).

3.10.4 Current threats

The primary threats to the Lost River sucker are habitat loss or alteration, the introduction of predatory fishes (e.g., fathead minnow), poor water quality and quantity, adverse biological factors like toxic algae or pathogens, and entrainment in agricultural irrigation systems (Buettner, 2005); (Burdick et al., 2015); (Martin & Saiki, 1999); (USFWS, 2013b); (Markle & Dunsmoor, 2007); (Perkins et al., 2000)). Global climate change may also impact this species (USFWS, 2012c); (USFWS, 2013b).

3.11 Shortnose Sucker

As noted above, shortnose sucker was listed as endangered on July 18, 1988 (USFWS, 1988), along with the Lost River sucker, which shares much of the same geographic range. A final recovery plan for the shortnose and Lost River suckers was published in 1993 and was revised in 2012 (USFWS, 2012c).

3.11.1 Distribution

Shortnose sucker was historically present in Upper Klamath, Tule, Lower Klamath, and Clear Lakes and their tributaries, though the species is now limited to the Upper Klamath River and its tributaries and outlet as well as the Keno, Copco, Iron Gate, and J.C. Boyle Reservoirs (USFWS, 2012c). Of those water bodies, only the Upper Klamath Lake and J.C. Boyle Reservoirs are in the proposed action area (Oregon). Strong spawning populations were historically found in the Lost River system and in Big Springs near Bonanza, Oregon (USFWS, 2016). Current spawning areas are concentrated in Upper Klamath Lake, the Williamson River (from RM 6 to the confluence with the Sprague River), and in sections of the Sprague River.

Adult suckers in Upper Klamath Lake tend to congregate in Pelican Bay at the north end of the lake in summer, where water quality (e.g., dissolved oxygen) tends to be more suitable (Hendrixson, Janney, & Shively, 2003) as cited in (ODFW, 2016); (Peck, 2000) as cited by (ODFW, 2016); (Banish et al., 2009). They then move into other parts of the lake starting in September (Banish et al., 2009).

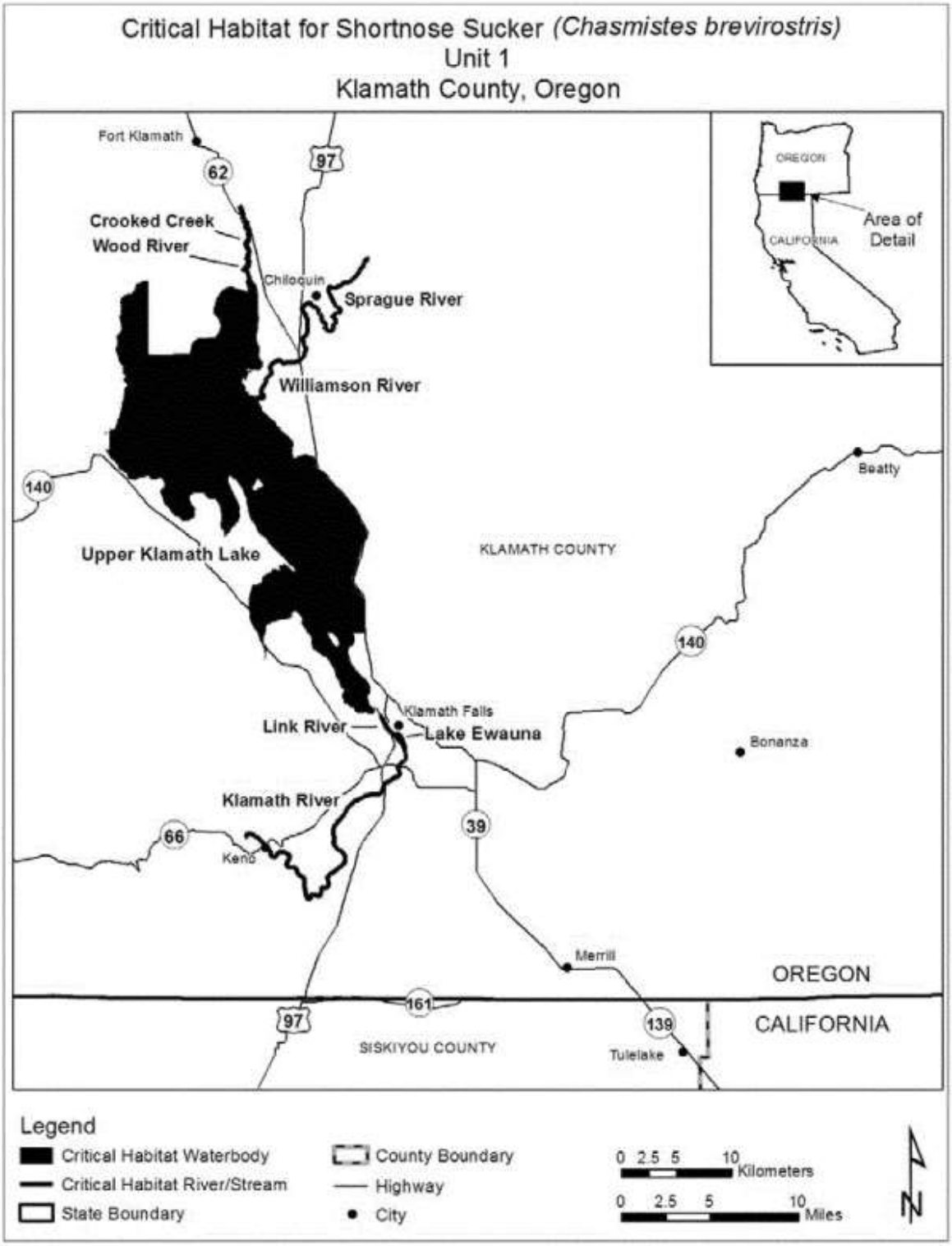
3.11.2 Critical Habitat

Critical habitat for the shortnose sucker (and Lost River sucker) was listed on December 11, 2012 (USFWS, 2012b), effective January 10, 2013. A total of 219 km (136 miles) of streams and 50,015 ha (123,590 acres) of lake and reservoir habitat were protected for shortnose sucker. The critical habitat is distributed in Klamath and Lake Counties, OR, and Modoc County, California. Figures 3-16 and 3-17 present the location of shortnose sucker critical habitat. Like Lost River sucker, shortnose sucker critical habitat is present in the Upper Klamath Lake and rivers both upstream and downstream of the lake (Figure 3-16); however, unlike the Lost River sucker, shortnose sucker critical habitat also includes several reservoirs in Oregon to the east of Upper Klamath Lake and associated streams and tributaries (e.g., Dry Prairie and Gerber Reservoirs and Ben Hall, Long Branch, Barnes Valley, Lapham, and Pitch Log Creeks).

The PBFs for shortnose sucker critical habitat are the same as those for the Lost River sucker, described above.

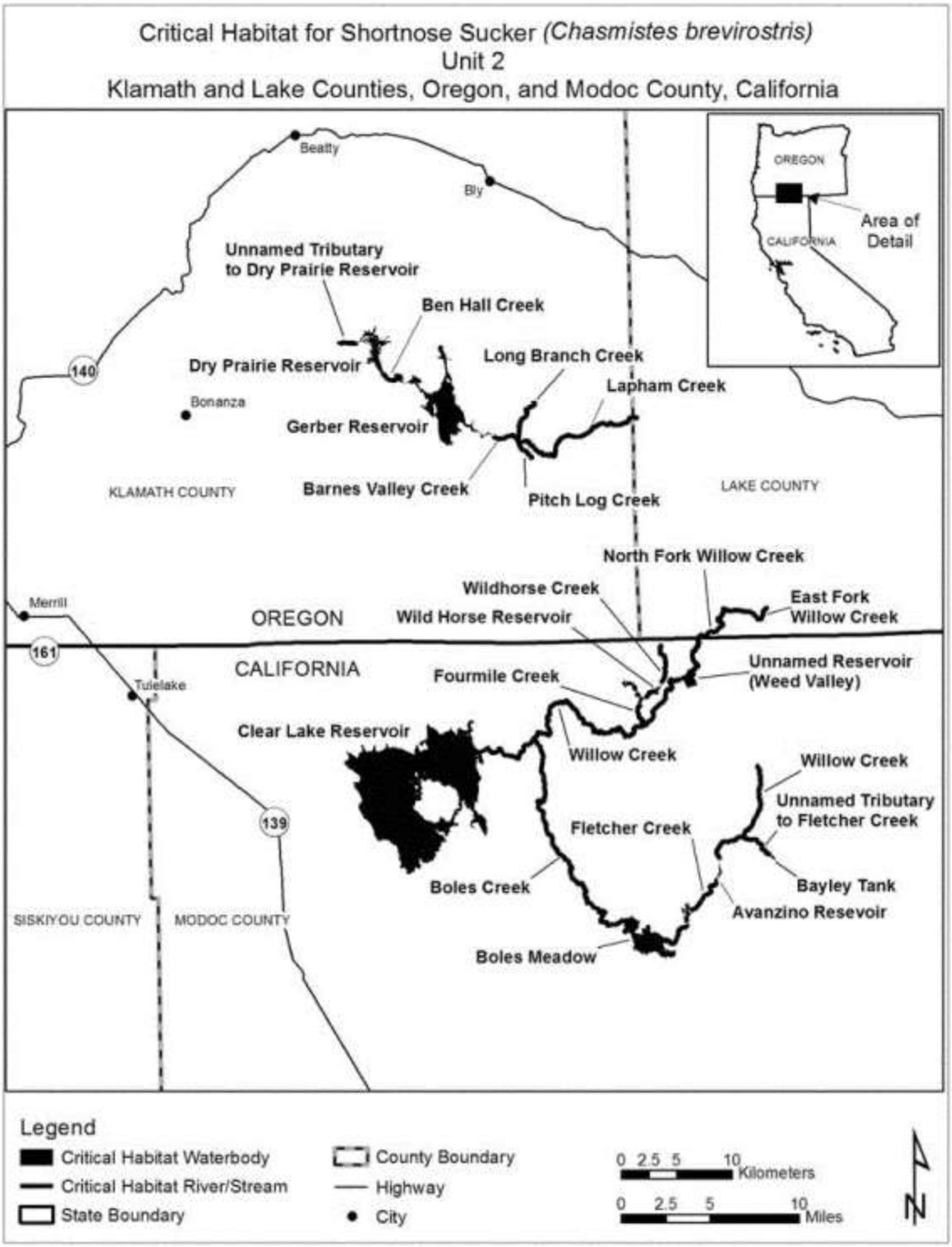
3.11.3 Life History

Shortnose sucker spawn from February to May in streams and rivers with gravel substrates and along the eastern shoreline of Upper Klamath Lake (Buettner, 2005), as cited by (ODFW, 2016). Individual female shortnose suckers produce between 18,000 and 70,000 eggs per year, with older females producing far more eggs than younger, smaller females. A small fraction of fertilized embryos survive to hatch. Shortnose sucker reach maturity at between four and six years old and may live up to 33 years, spawning many times in their lifetime. Larval suckers emerge from gravels soon after hatching (by July) and drift into lakes or rivers to feed and rear (M. Cooperman & Markle, 2003); (USFWS, 2016); (USFWS, 2012c).



Source: (USFWS, 2012b)

Figure 3-16. Critical Habitat for Shortnose Sucker, Western Areas



Source: (USFWS, 2012b)

Figure 3-17. Critical habitat for Shortnose Sucker, Eastern Areas

Shortnose sucker use several habitat types throughout their life including stream, river, lake, marsh, and shoreline spring habitats of varying depths (USFWS, 2012c). Adults tend to use open waters of lakes (between 1.5 and 3 m deep [5 and 10 feet deep]) when not spawning, though some individuals may stay in rivers, particularly in the Lost River. Larvae, upon entering lakes, tend to seek out cover provided by emergent vegetation along lake shorelines and in shallow water (between 0.15 and 0.91 m deep [0.5 and 3 feet deep]) (M. S. Cooperman & Markle, 2004). Juveniles, as they grow, shift from shallower to deeper lake waters away from emergent vegetation and over substrates ranging from mud to cobble (Burdick et al., 2008). Habitat use may be driven by patterns in dissolved oxygen, which has a pronounced effect on juvenile sucker growth and survival (Perkins et al., 2000; Rasmussen, 2011; Saiki et al., 1999).

In Upper Klamath Lake, there was an apparent shift in population size from large to smaller individuals between the mid-1980s to mid-1990s followed by a shift back to larger individuals up to the mid-2000s (Janney et al., 2008); (Terwilliger et al., 2010). This suggests that recruitment in the early-to-mid-1990s was high but that recruitment decreased during late 1990s and early-to-mid 2000s.

3.11.4 Current threats

Threats to shortnose sucker and their critical habitat are the same as those for Lost River sucker, which are described above. In addition, genetic monitoring of shortnose sucker in the Klamath River Basin indicates that there is significant hybridization between the shortnose sucker and the Klamath largescale sucker (*Catostomus snyderi*) (Tranah & May, 2006), which reduces the number of viable offspring produced by shortnose sucker.

3.12 Warner Sucker (*Catostomus warnerensis*) - Threatened

The Warner sucker was first listed on September 27, 1985. It is currently designated as threatened (USFWS, 1985b) in the entire range. Within the area covered by this listing, this species is known to occur only in Lake County Oregon.

3.12.1 Geographic Distribution

The Warner sucker is endemic to the streams and lakes of the Warner Basin in south-central Oregon. The species is presently known to occur in portions of Crump and Hart Lakes, the spillway canal north of Hart Lake, and portions of Snyder, Honey, Twentymile, and Twelvemile Creeks (USFWS, 1985b).

3.12.2 Critical Habitat

Critical habitat is designated (USFWS, 1985b) and includes: sections of Twelvemile and Twentymile Creeks; Spillway Canal north of Hart Lake; Snyder and Honey Creeks. Constituent elements of all areas proposed as critical habitat include streams 15 feet to 80 feet wide with gravel-bottom shoal and riffle areas with intervening pools. Streams should have clean, unpolluted flowing water and a stable riparian zone. The streams should support a variety of aquatic insects, crustaceans, and other small invertebrates for food.

3.12.3 Life History

The Warner sucker is a slender-bodied species that attains a maximum fork length of 456 mm. Pigmentation of sexually mature adults can be striking. The dorsal two-thirds of the head and body are blanketed with dark pigment, which borders creamy white lower sides and belly. During spawning season, males have a brilliant red lateral band along the midline of the body, females are lighter (USFWS, 2019a).

The Warner basin provides two generally continuous aquatic habitat types; a temporally more stable stream environment, and a temporally less stable lake environment. A common phenomenon among fishes is phenotypic plasticity induced by changes in environmental factors. Life history for the Warner sucker is evidently plastic. The lake and stream morphs of the Warner sucker probably evolved with frequent migration and gene exchange between them. The larger, presumably longer-lived, lake morphs are capable of surviving through several continuous years of isolation from stream spawning habitats due to drought or other factors. Stream morphs probably serve as sources for recolonization of lake habitats in wet years following droughts, such as the refilling of Warner Lakes in 1993 following their desiccation in 1992. Lake morph Warner suckers occupy the lakes and, possibly, deep areas in the low elevation creeks, reservoirs, sloughs and canals. The loss of either lake or stream morphs to drought, winter kill, excessive flows and a flushing of the fish in a stream, in conjunction with the lack of safe migration routes and the presence of predaceous game fishes (such as crappie), may strain the ability of the species to rebound. Irrigation diversions have also reduced available habitat and blocked migration. Lake morph suckers are generally much larger than steam morphs, however, growth rates in either habitat have not been studied. Sexual maturity is believed to usually occur at an age of 3-4 years (USFWS, 2019a).

Spawning usually occurs in April and May. Temperature and flow cues appear to trigger spawning, with most spawning taking place at 14to 20°C when stream flows are relatively high. The feeding habits of the Warner sucker depend to a large degree on habitat and life history stage, with adult suckers becoming less specialized than juveniles and YOY. Larvae have terminal mouths and short digestive tracts, enabling them to feed selectively in mid-water or on the surface. Invertebrates, particularly planktonic crustaceans, make up most of their diet. As the suckers grow, they gradually become generalized benthic feeders. Adult stream morph suckers forage nocturnally over a wide variety of substrates. Adult lake morphs are thought to have a similar diet, though food is taken over predominantly muddy substrates (USFWS, 2019a)

Warner suckers spawn in sand, or gravel beds in pools. Possible important spawning habitats and a source of recruitment for lake recolonization are in the upper Honey Creek drainage and the tributary Snyder Creek where the warm, constant temperatures of Source Springs are located. In years when access to stream spawning areas is limited by low flow or by physical in-stream blockages (such as beaver dams), suckers may attempt to spawn on gravel beds along the lake shorelines (USFWS, 2019a).

Larvae generally occupy shallow backwater pools or stream margins with abundant macrophytes, where there is little or no current. Larvae venture near higher flows during the daytime to feed on planktonic organisms but avoid the mid-channel water current at night. Spawning habitat may also be used for rearing during the first few months of life because when young eventually become immersed in high stream flows they do not appear to drift large distances downstream; i.e., the YOY remain in spawning habitat areas. YOY are often found

over deep, still water from mid-water to the surface, but also move into faster flowing areas near the heads of pools. For both runs and pools, YOY usually occupy quiet water close to shore (USFWS, 2019a).

Both juveniles and adults prefer areas of the streams which are protected from the main flow, seeking out deep pools. Beaver ponds may offer important refugia. Preferred pools tend to have: undercut banks; large beds of aquatic macrophytes; root wads or boulders; a surface to bottom temperature differential of at least 2 degrees C. (at low flows); maximum depth greater than 1.5 meters; and overhanging vegetation (often *Salix* ssp). Although suckers may be found almost anywhere in calmer sections of streams, the fish will not be far from larger pools (approximately 1/4 mile up- or down- steam).

When submersed and floating vascular macrophytes are present, they often form a major component of sucker-inhabited pools, providing cover and harboring planktonic crustaceans which make up most of the YOY sucker diet. Rock substrates are important in providing surfaces for epilitihic organisms upon which adult stream morph suckers feed, and finer gravel or sand is used for spawning. Embeddedness (e.g., from silt) has been negatively correlated with total sucker density.

Habitat use by lake morph suckers appears similar to that of stream morph suckers in that adult suckers are generally found in the deepest available water where food and cover are plentiful. Deep water also provides refuge from aerial predators.

By day, juveniles and adult suckers take shelter in the deepest available water and/or undercut banks. Deep pools also allow suckers to mitigate temperature extremes by moving vertically in the water column. With the absence of aquatic macrophytes, suckers can be seen schooling near the bottoms of these deep pools during the day. At night they disperse thorough various habitat types and water depths to forage for food.

3.12.4 Population Trends and Risks

The probable historic range of the Warner sucker includes the main Warner lakes (Hart, Crump and Pelican), ephemeral lakes, sloughs, and lower-gradient streams. Historically abundant and widely-distributed in the basin, the Warner sucker still maintains sizable numbers in a few habitats. It is still known to occur in most lakes, sloughs, and potholes, except during drought years. Stream resident populations are found in Honey and Twentymile creeks and in Deep Creek below Deep Creek falls. In most habitats the Warner sucker is rare, although aggregations of spawning adults or young-of-the-year may be encountered (USFWS, 2019a).

Drought in the late 1980's and early 1990's dried most lake and slough habitats and basin-wide surveys conducted from 1993-1997, after the lakes had refilled, documented the recolonization of these habitats by native and non-native fish. Prior to the drought, the lake population of suckers was comprised of only large older individuals indicating a lack of successful reproduction or recruitment to lake habitats. During the same time, non-native piscivorous fishes [crappie (Pomoxis sp.) and brown bullhead (*Ictalurus nebulosus*)], comprised approximately 87 percent of the fish fauna in the Warner lakes. Following the drought, recolonization by native fishes, including the Warner sucker, was found to occur at a much faster rate than for non-native fishes. Surveys in 1997 indicated that native fish [Warner suckers, tui chubs (*Gila bicolor* ssp) and redband trout (*Oncorhynchus mykiss* ssp.)] comprised approximately 80 percent of the total

catch. Information collected from 1993-1997 suggests that the drought may have had a significantly greater impact on non-native fishes as compared to the native species that evolved under fluctuating environmental conditions. However, over time it is anticipated that the number of crappie and brown bullhead will increase significantly to levels observed prior to the drying of lake habitat in 1992 (USFWS, 2019a).

Historically abundant and widely-distributed in the Warner Valley basin, the Warner sucker still maintains sizable numbers in a few habitats. The largest remaining population occurs in Hart Lake, Oregon, where successful reproduction has been documented, but there is no evidence of recruitment to the adult population (J. E. Williams, Stern, Munhall, & Anderson, 1990).

3.13 Hutton Tui Chub (*Gila bicolor ssp.*) - Threatened

The Hutton tui chub was first listed on March 28, 1985. It is currently designated as threatened (USFWS, 1985a) in the Hutton subspecies. This species is known only to occur in Lake County in southern Oregon.

3.13.1 Geographic Boundaries and Spatial Distribution

The Hutton tui chub occurs in Hutton Spring, Lake County, Oregon. The size of the springhole of Hutton Spring varies with excavations made by the owner. It has ranged from 20 feet to nearly 40 feet in diameter, and is about 15 feet deep in the center. Hutton Spring is occupied in part by tules (*Scirpus americanus*). Other vegetation present includes sedge (*Carex sp*.), saltgrass (*Distichlus sp*.), and squirreltail (*Sitanion hystrix*). The fish use the vegetation and whatever debris is present for cover. Some of the larger individuals use the deep spring hole as cover. The recorded water temperature is 64°F. (17.7°C), during the summer (May - October). The outflow from the spring forms a small area of wetland adjacent to the sources. This is occupied by grasses, water parsley, and sedges. The spring is in a grassy area bordered to the north and west by shrubby rangeland and to the east and south by the lake bed of pluvial (rain-influenced) Alkali Lake. A low dry ridge with sagebrush is immediately south of the spring area. Elevation at the site is 4500 feet (Scheerer & Jacobs, 2005).

3.13.2 Critical Habitat

No critical habitat has been designated for the Hutton tui chub.

3.13.3 Life History

The Hutton tui chub is robust, with the greatest depth of body immediately behind the head. The size of the head ranges from 0.30 to 0.31 m with a depth 0.20 m (SL). The eyes appear large and are usually slightly more than 0.06 m (SL). This subspecies is distinguished from other tui chubs in adjacent basins by morphology of the head. It also has the heaviest, strongest pharyngeal teeth of any *Gila bicolor* known, and its mouth lining and tongue are purplish (Bills, 1978; Bond, 1974). The mouth does not extend to the eye; dorsum dusky olive to dark green, sides brassy brown (often mottled in adult), venter silver-white to yellow; fins clear to dusky- olive; dusky stripe alongside in young; large individual may have orange lower sides and yellow to copper fins with red, pink, or orange base.

There is very little information regarding the ecology of the Hutton tui chub (Bills, 1978) examined gut content and found the Hutton tui chub to be omnivorous with a majority of food eaten being filamentous algae. It appears that a dense aquatic alga is needed for spawning and rearing of young (USFWS, 1998). No information is available on growth rates, age of reproduction or behavioral patterns.

The Hutton tui chub lives in clear springs and outflow channels. The size of the Hutton Spring holes range from 20-40 feet in width, and about 15 ft in depth at its center. The smaller spring hole is about 10 feet across and 2 feet deep. The spring holes are surrounded by water parsley, sedge, saltgrass, and squirreltail, which the chub uses for cover (BLM, 2005).

3.13.4 Population Trends and Risks

There is very little information regarding the population numbers of this species; the last estimation of 450 individuals was made in 1977 (Bond, 1974). In 1977, the distribution of the Hutton tui chub included only two springs in the Alkali subbasin, Hutton Spring and an unnamed spring. Attempts to find this unnamed spring in 1996 were unsuccessful and this population may have been lost. Threats to habitat include ground water pumping for irrigation, excessive trampling of the habitats by livestock, channeling of the springs for agricultural purposes, other mechanical manipulation of the spring habitats, and the presence of chemical waste disposal site near Hutton Spring. The habitat is currently fenced to exclude cattle and is in stable condition (USFWS & OFWO, 2017). The Oregon Department of Environmental Quality (DEQ) is currently monitoring groundwater contamination from the chemical disposal site to the south of Hutton Spring. The DEQ has determined the contaminated plume is spreading from east to west away from Hutton Spring and thus does not currently constitute a threat to the water quality in Hutton Spring (USFWS & OFWO, 2017).

3.14 Borax Lake Chub (*Gila boraxobius*) - Endangered

The Borax Lake chub was first listed on May 28, 1980. It is currently designated as endangered in the entire range (USFWS, 1982). This species is currently known to occur only in the Alvord Basin in Harney County in southern Oregon**.**

3.14.1 Geographic Boundaries and Spatial Distribution

This species is endemic to Borax Lake, Harney County, Oregon. Borax Lake is a 10.2 acre, clear, shallow hot spring, with more than 50 smaller hot springs within about one-half mile to the north of the lake. Borax Lake is a natural lake fed from the waters of several thermal springs and is perched atop large sodium-borate deposits in the Alvord Desert. Spring inputs near the bottom of a deep vent (100 feet below the surface) range from 40-148 °C. Surface water temperatures typically range from 16-38 °C but fluctuations occur and temperatures occasionally exceed 38 °C causing fish kills as water exceeds the chub's critical thermal maximum (Scoppettone, Shea, & Buettner, 1995). At temperatures above 30 °C, fishes form loose schools around algae-coated carbonate nodules.

Water flows from Borax Lake into surrounding marshes, small pools and Lower Borax Lake. Individuals may also reside in the outflow of the lake, and a pond (Lower Borax Lake) fed by the outflow. Chubs are fairly evenly distributed throughout the lake, though there is a tendency toward avoidance of shallow water (< 40 mm in depth) and areas of sparse vegetation; there are times when virtually the entire population occurs within or under the flocculent-like algal layer covering the bottom of much of the lake (Scoppettone & Vinyard, 1991). Young are common in shallow coves around lake margins in spring.

3.14.2 Critical Habitat

The critical habitat for the Borax Lake chub encompasses Borax Lake and the aquatic environments associated with its outflow located in T37S; R33E; SW1/4 Sec. 11. W1/2 Sec. 14, and E3/4 of the SE1/4 Sec. 15 and SE1/4 of the NE1/4 Sec. 15, Harney County, Oregon. Borax Lake is situated within a 160-acre parcel of privately-owned land purchased by the Nature Conservancy in 1993. The total area of the Critical Habitat is 640 acres. Principal constituent elements of this habitat for the Borax Lake chub are considered to be the constant temperature and flow of water into Borax Lake and the natural water flow out of Borax Lake into associated aquatic environs and the aquatic and terrestrial food organisms of this ecosystem.

3.14.3 Historical Information

The vast majority of chubs reside in Borax Lake. Seven population estimates of the chub in Borax Lake ranged from 3,934 to 13,319 during 1986-88. Population estimates ranged from 10,625 to 34,634 during 1991-97. Lower Borax Lake, which contained several thousand chubs during wet years, was dry from 1989-91 and currently has no fish (BLM, 2005).

3.14.4 Life History

The Borax Lake chub is a dwarf species of the genus Gila (a wide-spread desert minnow). Adults are typically 33-50 mm (max. 93 mm) in size. This species has a large, concave head, and large eyes. It is olive green on the upper part of the head and body, and is speckled with small melanophores that extend nearly to the ventral surface. A dark line also extends along the length of the dorsal midline. Males have longer fins, nuptial tubercles are only found in males longer than 28mm and are restricted to the body and paired fins (J. E. Williams & Bond, 1980).

Males, and some females, reach reproductive maturity within 1 year. Spawning can occur year-round, but primarily takes place in the fall and spring (Scoppettone & Vinyard, 1991; J. E. Williams & Bond, 1983). Spawning occurs primarily in gravel, rock outcrop, and sand habitats, which comprise only 16% of the substrate of Borax Lake (Perkins, Mace, Scoppettone, & Rissler, 1996). Longevity usually is about 1 year; few of the chubs live beyond 4 years (Scoppettone & Vinyard, 1991; Sigler & Sigler, 1987).

The Borax lake chub is an opportunistic omnivore (J. E. Williams & Williams, 1980). Juveniles and adults eat essentially the same things including aquatic insects; terrestrial insects; algae; mollusks and mollusk eggs; aquatic worms; fish scales; spiders; and seeds.

Borax Lake is situated approximately 30 ft. above the desert floor on a bed of deposited salts, making the bottom of the lake inhospitable to rooted plants. However, some plants grow in spots along the lakes perimeter. The lake depends upon several subterranean springs for its water supply. Inflowing water typically is 35-40 °C. However, the Borax lake chub prefers water of 29-30 °C (J. E. Williams & Bond, 1983), and may lose equilibrium in water above 34 °C. The Borax lake chub lives at the lakes perimeter, where the water is within the chubs preferred temperature range. These shallow areas make up only a few acres and thus decreasing the water level significantly affects the chub's habitat. For cover, this species uses boulder-like concretions of deposited mineral; filamentous algae and encrusting diatoms; aquatic plants; and when disturbed the chub may retreat to deeper water.

3.14.5 Population Trends and Risks

The vast majority of chubs reside in Borax Lake. Seven population estimates of the chub in Borax Lake ranged from 3,934 to 13,319 during 1986-88, and from 10,625 to 34,634 during 1991-97 (BLM, 2005). Lower Borax Lake, which may contain several thousand chubs during wet years, was dry from 1989-91 and currently has no fish. A 1980 modification of the lake diverted water and lowered the water level by approximately 1 foot decreasing the habitat and increasing water temperatures. Changes in salinity and increases in temperature can cause the chub to lose equilibrium resulting in death. An increase in salinity also decreases the lakes capacity to support aquatic vegetation, thus reducing a food source.

Development of geothermal resources may adversely affect the Borax Lake chub by modifying or destroying its aquatic habitat. The Steens Mountain Cooperative Management and Protection Act of 2000 has withdrawn geothermal and mineral development rights from the Borax Lake area. Other threats include irrigation and overgrazing by livestock. The fragile salt-crust shoreline of the lake also is easily damaged by off-road vehicle use and by frequent human visitation; water chemistry analysis have shown that lake conditions no longer support deposition of rim material through chemical precipitation (Schneider & McFarland, 1995). The area is currently fenced to exclude livestock. Conservation actions needed include: 1) closure of the Critical Habitat area to vehicle entry, 2) restoration of Lower Borax Lake and adjacent marshes, and 3) monitoring of fish, invertebrates and habitat characteristics.

3.15 Lahontan Cutthroat Trout (*Oncorhynchus clarki henshawi*) - Endangered

The Lahontan cutthroat trout was first listed as endangered on October 13, 1970 (USFWS, 1970), but was downlisted to "threatened" on July 16, 1975 (USFWS, 1975). Within the area covered by this listing, this species is known to occur in: California, Nevada, Oregon and Utah. In Oregon, the species is present in Harney and Malheur counties (Southeast Oregon).

The Lahontan cutthroat trout is the largest of all cutthroat races. Although coloration is variable, this species is generally heavily marked with large, rounded black spots, more or less evenly distributed over the sides, head, and abdomen. Spawning fish generally develop bright red coloration on the underside of the mandible and on the opercle. In spawning males, coloration is generally more intense than in females.

3.15.1 Geographic Distribution

The range of Lahontan cutthroat trout is primarily in streams of the Lahontan and Coyote Lake Basins in southeastern Oregon and occurs in the following streams: Willow Creek, Whitehorse Creek, Little Whitehorse Creek, Doolittle Creek, Fifteen Mile Creek (from the Coyote Lake Basin), and Indian, Sage, and Line Canyon Creeks, tributaries of McDermitt Creek in the Quinn River Basin (Nevada). This subspecies has been reintroduced into several stream systems throughout the Lahontan Basin and Pyramid and Walker Lakes (USFWS, 2003b).

The Coyote Lake Basin has the only native population of Lahontan cutthroat trout in Oregon that is without threat of hybridization and is broadly distributed throughout a waterbody. In October 1994, the number of Lahontan cutthroat in the basin was estimated at 39,500 fish, and fish were limited to 56 km of stream habitat available (approximately 25,000 in the Whitehorse Creek drainage and about 15,000 cutthroat occupied the Willow Creek drainage). Distribution was limited by dry channels and thermal and physical barriers to movement, which created two disconnected populations in the Willow Creek and Whitehorse Creek drainages and influenced population density, structure, and life history (USFWS, 2003b).

3.15.2 Critical Habitat

Critical habitat has not been designated for this species.

3.15.3 Life History

Lahontan cutthroat trout are obligate but opportunistic stream spawners. Typically, they spawn from April through July, depending on water temperature and flow characteristics. Autumn spawning runs have been reported from some populations. The fish may reproduce more than once, though post-spawning mortality is high (60-90%). Lake residents migrate into streams to spawn, typically in riffles on well washed gravels. The behavior of this subspecies is typical of stream spawning trout; adults court, pair, and deposit and fertilize eggs in a redd dug by the female.

Like other cutthroat races, the Lahontan cutthroat is an opportunistic feeder, with the diet of small individuals dominated by invertebrates, and the diet of larger individuals composed primarily of fish.

The Lahontan cutthroat trout is one subspecies of the wide-ranging cutthroat trout species (*Oncorhynchus clarki*), which includes at least 14 recognized forms in the western United States. Cutthroat trout have the most extensive range of any inland trout species in western North America, and occur in anadromous, non-anadromous, fluvial, and lacustrine populations (USFWS, 2003b). Many of the basins in which cutthroat trout occur contain remnants of much more extensive bodies of water which were present during the wetter period of the late Pleistocene epoch (USFWS, 2003b).

These fish are usually tolerant of both high temperatures (>27 °C) and large daily fluctuations (up to 20 °C). They are also quite tolerant of high alkalinity (>3,000 mg/L) and dissolved solids (>1,000 mg/L). They are apparently intolerant of competition or predation by nonnative salmonids, and rarely coexist with them (USFWS, 2003b).

3.15.4 Population Trends and Risks

Lahontan cutthroat trout historically occurred in most cold waters of the Lahontan Basin of Nevada and California, including the Humboldt, Truckee, Carson, Walker, and Summit Lake/Quinn River drainages. Large alkaline lakes, small mountain streams and lakes, small tributary streams, and major rivers were inhabited, resulting in the present highly variable subspecies. Only remnant populations remain in a few streams in the Truckee, Carson, and Walker basins out of an estimated 1,020 miles of historic habitat (Gerstung, 1986). Although mechanisms of stream colonization outside of the Lahontan basin by this subspecies are uncertain, transport by humans is suspected. Subsequently, resident stream populations were used to stock Oregon streams during the 1970's and 1980's.

Surveys conducted by ODFW indicated that Lahontan cutthroat trout populations were reduced from 1985 to 1989 by 62 percent on Willow Creek, 69 percent on Whitehorse Creek, 93 percent on Little Whitehorse Creek, and 42 percent on Doolittle Creek. No Lahontan cutthroat trout were found in either the 1985 or 1989 ODFW surveys on Fifteen Mile Creek (USFWS, 2003b). These declining numbers prompted ODFW to close area streams to fishing (by special order) in 1989. The closure remains in effect. Fish surveys of area streams were conducted again in October of 1994. Although methods vary among the conducted surveys (1985, 1989, and 1994), fish numbers have increased in general from approximately 8,000 fish in the mid-1980s to approximately 40,000 fish in 1994; however, in many areas, stream conditions remain less than favorable for the cutthroat.

The overall status of Lahontan cutthroat trout is unknown, although the population has experienced a severe decline in range and numbers. Riparian and upland habitats have been degraded by intensive grazing by cattle and sheep during the past 130 years. Drought and cold periods during the past decade have further affected the quantity and quality of the aquatic habitat. The ability of local populations to interact is important to the long-term viability of a metapopulation. The population of Lahontan cutthroat in the Whitehorse Creek subbasin has been fragmented by numerous barriers into four discrete local populations. The Willow Creek subbasin is largely free of migration barriers. Seasonally, all streams in the drainages have disjunct populations because of high summer temperatures (>26 °C) or dry channels.

3.16 Invertebrates

3.16.1 Vernal Pool Fairy Shrimp (*Branchinecta lynchi*) - Threatened

The Vernal pool fairy shrimp (VPFS) was first listed on September 19, 1994. It is currently designated as threatened (USFWS, 1994b), September 19, 1994) in the entire range. Within the area covered by this listing, this species is known to occur in California and Oregon. In the state of Oregon, it occurs only in Jackson county in southern Oregon.

3.16.1.1 Geographic Boundaries and Spatial Distribution

The populations of the vernal pool fairy shrimp are restricted to southern Oregon and northern, central, and portions of southern California. Vernal pool fairy shrimp are known from a total of 32 populations located in an area extending from Stillwater Plain in Shasta County through most

of the length of the Central Valley to Pixley in Tulare County, and along the central coast range from northern Solano County to Pinnacles in San Benito County (Eng, Belk, & Eriksen, 1990; Sugnet, 1993). Five additional, disjunctive populations exist near Soda Lake in San Luis Obispo County, in the mountain grasslands of northern Santa Barbara County, on the Santa Rosa Plateau in Riverside County, near Rancho California in Riverside County and near the city of Medford, Jackson County, Oregon. Populations of this species are known from Beale, Mather, and Castle Air Force Bases (Eng et al., 1990). Several refuges have been established on Nature Conservancy preserves in the Vina Plains, the Haystack Mountain area, and the Santa Rosa Plateau. The areas surrounding the refuges are cultivated fields (Ridgely et al., 2003). The TNC owns two preserves occupied by VPFS and owns a conservation easement on a third property with VPFS. The Preserves are the Agate Desert Preserve (White City, OR), Whetstone Savannah Preserve (Central Point, OR), and Rogue Plains Preserve (Eagle Point, OR). The Bureau of Land Management owns 432 acres of VPFS habitat on the Upper and Lower Table Rocks north of White City. Oregon Dept of Transportation owns and manages 196 acres of VPFS habitat as a mitigation/conservation bank in Central Point, OR. And the Bureau of Reclamation owns 154 acres of VPFS habitat around the Agate Reservoir, near White City.

3.16.1.2 Critical Habitat

Critical habitat for the vernal pool fairy shrimp was designated on August 6, 2003 in both California and Oregon and updated on February 10, 2006. Total area of critical habitat in these two states is 597,821 acres. Four units of habitat were designated within Jackson county, Oregon; totaling 7,574 acres. Of this area, 432 acres are Federally owned land, and the remaining 7,142 acres are located on privately owned lands.

Primary constituent elements for this include: (i) Topographic features characterized by mounds and swales and depressions within a matrix of surrounding uplands that result in complexes of continuously, or intermittently, flowing surface water in the swales connecting the pools, providing for dispersal and promoting hydroperiods of adequate length in the pools; (ii) Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water for a minimum of 18 days, in all but the driest years; thereby providing adequate water for incubation, maturation, and reproduction. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands; (iii) Sources of food, expected to be detritus occurring in the pools, contributed by overland flow from the pools' watershed, or the results of biological processes within the pools themselves, such as single-celled bacteria, algae, and dead organic matter, to provide for feeding; and (iv) Structure within the pools described above (ii), consisting of organic and inorganic materials, such as living and dead plants from plant species adapted to seasonally inundated environments, rocks, and other inorganic debris that may be washed, blown, or otherwise transported into the pools, that provide shelter (USFWS, 2006).

3.16.1.3 Historical Information

At the time of its listing, the vernal pool fairy shrimp was known to occur only in California, extending from Tulare County in the south to Shasta County in the north. In 1998, these fairy shrimp were discovered in vernal pools in Jackson County, Oregon, in an area north of Medford

known as the Agate Desert. Prior to this discovery, the most northerly known location for the species was south of Mount Shasta, California, some 80 miles south of the Agate Desert (USFWS, 2005b).

Like the other species of vernal pool branchiopods, the numbers of *B. lynchi* populations have declined primarily because of destruction or degradation of vernal pools through development of urban, suburban, and agricultural projects. In addition to direct habitat loss, vernal pool fairy shrimp populations have declined from of a variety of activities that degrade existing vernal pools by altering pool hydrology (water regime). Vernal pool hydrology can be altered by a variety of activities, including the construction of roads, trails, ditches, or canals that can block the flow of water into or drain water away from the vernal pool complex (USFWS, 2005b).

3.16.1.4 Life History

The vernal pool fairy shrimp ranges in size from 10.9 to 25.0 mm (Eng et al., 1990). Vernal pool fairy shrimp are almost translucent, but can be whitish or have some orange body parts. Fairy shrimp have delicate elongate bodies, large stalked compound eyes, no carapace, and 11 pairs of swimming legs. They swim or glide gracefully upside down by means of complex beating movements of the legs that pass in a wave-like anterior to posterior direction.

The second pair of antennae in the adult females are cylindrical and elongate, but in the males' antennae are greatly enlarged and specialized for clasping the females during copulation. The females carry the fertilized eggs in an oval or elongate ventral brood sac. The eggs are either dropped to the bottom or remain attached until the female dies and sinks (Pennak, 1989). A key adaptation of the fairy shrimp is the production of drought-resistant eggs. When the vernal pools dry, the eggs remain on the surface of the pool or embedded within the top few centimeters of soil. There they survive the hot, dry summers and cold, wet winters that follow until the vernal pools and swales fill with rainwater and conditions are right for hatching (Geer & Foulk, 2000). When the pools refill in the same or subsequent seasons some, but not all, of the eggs may hatch. The egg bank in the soil may be comprised of the eggs from several years of breeding (Donald, 1983). Although the animal can mature quickly, allowing populations to persist in short-lived shallow pools, it also persists later into the spring where pools are longer lasting (Simovich & Fugate, 1992).

The vernal pool fairy shrimp consumes algae and plankton by scraping and straining them from surfaces within the vernal pool. In addition, the vernal pool fairy shrimp also feeds on a diet of bacteria, protozoa, rotifers, and small amounts of detritus (Pennak, 1989). Fairy shrimp play an important role in the community ecology of ephemeral water bodies (Loring, MacKay, & Whitford, 1988). They are fed upon by migratory waterfowl (Krapu, 1974; Swanson, Krapu, Bartonek, Serie, & Johnson, 1974) and other vertebrates, such as spadefoot toad tadpoles.

Vernal pool fairy shrimp populations occur in ephemeral freshwater habitats, such as vernal pools and swales. None are known to occur in running or marine waters or other permanent bodies of water. Vernal pools are unique seasonal wetlands that support a wide variety of wildlife, from waterfowl to amphibians– all of which rely on the protein-rich food sources found in these ecosystems (Geer & Foulk, 2000).

The distribution of vernal pools is highly discontinuous and some of the aquatic invertebrates that are found in this habitat occur only in specific geographic areas. Due to local topography and geology, the pools are usually clustered into pool complexes (Holland & Jain, 1988). Pools within a complex typically are separated by distances on the order of meters and may form dense, interconnected mosaics of small pools or a sparser scattering of larger pools. “This species has a sporadic distribution within vernal pool complexes (Jones and Stokes, 1992, 1993; County of Sacramento 1990; Patton 1984; Stromberg 1993; Sugnet and Associates 1993b) wherein the majority of pools in a given complex typically are not inhabited by the species” (USFWS, 1994a).

Although the vernal pool fairy shrimp has a relatively wide range, the majority of known populations inhabit vernal pools with clear to tea-colored water, most commonly in grass or mud bottomed swales, or basalt flow depression pools in unplowed grasslands, but one population occurs in sandstone rock outcrops and another population in alkaline vernal pools (Collie & Lathrop, 1976). In California, within monitored vernal pool ecosystems a tremendous spatial and temporal variability in redox processes and in reducible substrates was shown (Hobson & Dahlgren, 1996). They are ecologically dependent on seasonal fluctuations in their habitat, such as absence or presence of water during specific times of the year, duration of inundation, and other environmental factors that include specific salinity, conductivity, dissolved solids, and pH levels. Water chemistry is one of the most important factors in determining the distribution of fairy shrimp (Belk, 1977). The water in pools inhabited by this species has low total dissolved solids (TDS), conductivity, alkalinity, and chloride (Collie & Lathrop, 1976). The vernal pools the animal inhabits vary in size from over 10 ha to only 20 square meters. The vernal pool fairy shrimp occurs at temperatures between 6-20 degrees C in soft and poorly buffered waters (Eng et al., 1990).

3.16.1.5 Population Trends and Risks

The populations of the vernal pool fairy shrimp are restricted to southern Oregon and northern, central, and portions of southern California. (Sugnet, 1993) listed 178 records for the species out of 3092 “discrete locations” containing potential habitat. Current population estimates number in the 1000s of individuals (Friedman, 2019). Simovich and Fugate (1992) reported that the vernal pool fairy shrimp typically is found at low population densities. Only rarely does the vernal pool fairy shrimp co-occur with other fairy shrimp species, but where it does, the vernal pool fairy shrimp is never the numerically dominant one (Eng et al., 1990). The vernal pool fairy shrimp has been collected from early December to early May (Collie & Lathrop, 1976). Several of these populations are comprised of a single inhabited pool.

The greatest threat for vernal pool fairy shrimp survival is altered hydrology and the elimination of vernal pools. Pollution is also a suspected risk factor. More specifically, risks include disruption of habitat by human developments, which impacts hydrology such as through the construction of cuts or rips through hardpan soils that could drain vernal pools. Addition of toxicants/pollutants such as aluminum might reduce fairy shrimp prey or directly kill fairy shrimp. Pools are not connected to waterways and ditches, but if a roadside was constructed to bisect a vernal pool, storm events might lead to water overflow from ditches and expose vernal pools to road runoff and potential contaminants. Their small population size makes additional risk factors potentially more significant compared to species with a larger population size. And

as previously stated in section 3.16, vernal pool hydrology can be altered by a variety of activities, including the construction of roads, trails, ditches, or canals that can block the flow of water into, drain water away from, or drain water into the vernal pool complex (OFWO, 2005b).

3.16.2 Foskett Speckled Dace (*Rhinichthys osculus*) - Threatened

The Foskett speckled dace was first listed on March 28, 1985. It is currently designated as threatened (USFWS, 1985a) in the Foskett subspecies. Within the area covered by this listing, this species is known to occur only in Lake County in southern Oregon. This form of speckled dace is related to the speckled dace of Twentymile Creek in the Warner Basin, but is distinguished by a shorter lateral line and larger eye. Distinguishing characteristics are as follows: lateral line much reduced with about 15 scales with pores; eyes large; dorsal fin set well behind pelvic insertion (BLM, 2003).

3.16.2.1 Geographic Boundaries and Spatial Distribution

There is currently only one known population of Foskett speckled dace which is found in Foskett Spring in the Coleman subbasin; Lake County, southern Oregon.

3.16.2.2 Critical Habitat

Critical habitat has not been designated for this species.

3.16.2.3 Historical Information

Foskett speckled dace were probably distributed throughout prehistoric Coleman Lake of the Warner Basin during times that it held substantial amounts of water. The Warner Basin includes portions of southeast Oregon, northern Nevada and northern California. As the lake dried, the salt content of the lake water increased. Suitable habitat would have been reduced from a large lake to any spring systems that provided enough habitat for survival. There is currently only one known population of Foskett speckled dace which is found in Foskett Spring in the Coleman subbasin. Dace Spring, a short distance away, may have been occupied by Foskett speckled dace in the past, but there were none found in a 1970 survey. In 1979 and 1980, 100 dace from Foskett spring were transplanted into an excavated area at Dace Spring. Over time the artificial habitat filled in and fish in Dace Spring were last seen in the spring outflow in 1997 (BLM, 2003).

Foskett Spring has the only known native population of Foskett speckled dace. The spring originates in a pool about 5 meters (16.6 feet) across, then flows toward Coleman Lake in a narrow, shallow channel (approximately 5 centimeters (2 inches) deep and 5 centimeters (2 inches) wide). The source pool has a loose sandy bottom and is choked with macrophytes (large plants that are visible to the naked eye). The spring brook (outflow channel) eventually turns into a marsh and finally dries up before reaching the bed of Coleman Lake. Bond (USFWS, 1985a) estimated the population of Foskett speckled dace in Foskett Spring to be 1,500 individuals. Dambacher estimated there to be about 204 Foskett speckled dace in the source pool, 702 in the spring brook, and 26,881 in the shallow pool/marsh (BLM, 2003).

3.16.2.4 Life History

Little is known about the biology/ecology of the Foskett speckled dace. The Foskett speckled dace grows to about 4 inches (10 cm) in length. Their coloring on the back is dusky to dark olive; sides grayish green with dark lateral stripe which is often obscured by dark speckles or blotches. Spawning usually occurs in June or July (Matthews & Moseley, 1990). Breeding males are usually reddish on their lips and fin bases.

The speckled dace is one of the most widespread minnow in western waters with several subspecies characterized in this highly variable species. The speckled dace is an important forage fish for some trout streams. These fish feed on insects, detritus and eggs of other fish (Matthews & Moseley, 1990). They prefer cool to warm creeks, rivers and lakes over rocks or gravel. Pant species found around the springs which they inhabit include rushes, sedges, *Mimulus*, Kentucky bluegrass *(Poa pretensis)*, thistle, and saltgrass *(Distichlis. spicata*). Foskett Spring is a coolwater spring with temperatures recorded at a constant 18 degrees Celsius over a 2-year period.

3.16.2.5 Population Trends and Risks

There is currently only one known population of Foskett speckled dace which is found in Foskett Spring in the Coleman subbasin. Dace Spring, a short distance away, may have been occupied by Foskett speckled dace in the past, but there were none found in a 1970 survey. In 1979 and 1980, 100 dace from Foskett spring were transplanted into an excavated area at Dace Spring. Over time the artificial habitat filled in and fish in Dace Spring were last seen in the spring outflow in 1997 (BLM, 2003).

3.17 Southern Resident killer whale (*Orcinus orca*) - Endangered

The distinct population segment (DPS) of Southern Resident killer whales (Orcinus orca) was listed as endangered under the ESA on November 18, 2005 (USFWS, 2005a). Southern Residents are designated as “depleted” and “strategic” under the Marine Mammal Protection Act (MMPA) (USFWS, 2003a). NMFS issued the final recovery plan for Southern Residents in January 2008 (NMFS, 2008b).

Killer whales (often referred to as “orcas”) are the largest odontocete (toothed) dolphin species; adults tend to be 6.1 to 7.3 m (20 to 24 ft) in length, but killer whales may grow as large as 9.8 m (32 ft) (NOAA Fisheries, 2017a). Southern Resident killer whales (SRKWs) live in inland and coastal marine waters, generally 49 to 55 m (160 to 180 ft) deep (Noren & Hauser, 2016). Southern Residents are highly mobile and can travel up to 86 miles in a single day (Baird, 2000; Erickson, 1978).

Killer whales of related matrilines group together to form pods. Three pods – J, K, and L – make up the Southern Resident community. Clans are composed of pods with similar vocal dialects and all three pods of the Southern Residents are part of J clan. The following description of the SRKW summarizes information taken largely from the recovery plan and recent 5-year status review (NMFS, 2011e), as well as more recent information.

3.17.1.1 Range and Distribution

SRKWs occur throughout the coastal waters of Washington, Oregon, and Vancouver Island and are known to travel as far south as central California and as far north as southeast Alaska. They are present in the Salish Sea (Puget Sound, Strait of Juan de Fuca, and the Strait of Georgia) from spring to fall each year (NOAA Fisheries, 2017a). In winter, some SRKWs remain in the Salish Sea, while others travel along the Washington, Oregon, and California coasts (as far south as central California) (NOAA Fisheries, 2015b). SRKWs may also travel north along the British Columbia border as far as the Queen Charlotte Islands and southeast Alaska. Between late spring and early autumn, SRKWs spend a significant portion of time in the Georgia Basin (Canada) and around the San Juan Islands of Washington following incoming salmon runs (NOAA Fisheries, 2017a). Satellite- tagged animals and tracking has identified an important winter through spring foraging area along the west coast of Washington down to the mouth of the Columbia River (M. B. Hanson, Emmons, Ward, Nystuen, & Lammers, 2013). Although SRKWs can occur along the outer coast of Washington and Oregon at any time of the year, occurrence along the outer coast is more likely from late autumn to early spring.

SRKWs co-exist in areas with West Coast transient killer whales, but resident and transient groups generally do not have significant interactions (e.g., socializing or attacking one another) (Barrett-Lennard & Heise, 2007).

3.17.1.2 Life History

3.17.1.2.1 Reproduction

Southern Resident killer whales are a long-lived species with late onset of sexual maturity (review in NMFS (2008b)). Females produce a low number of surviving calves over the course of their reproductive life span (Bain, 1990; Olesiuk, Bigg, & Ellis, 1990). The average interbirth interval for reproductive Southern Resident females is 6.1 years (Olesiuk et al., 1990). Mothers and offspring maintain highly stable social bonds throughout their lives, which is the basis for the matrilineal social structure in the Southern Resident population (Baird, 2000; Bigg, Olesiuk, Ellis, Ford, & Balcomb, 1990; J. K. B. Ford, Ellis, & Balcomb, 2000).

3.17.1.2.2 Social groups

The familial pods include 20 to 40 individuals led by a dominant matriarch (NOAA Fisheries, 2017b). Stable social groups tend to include 2 to 15 individuals at a time, but large, temporary aggregations of the entire population occur, particularly in the summer (NOAA Fisheries, 2017a). Aggregation and separation of groups tends to follow seasonal trends in prey availability and courtship and mating activities. Temporary associations of the pods, called “superpods,” of 50 or more individuals may form for a matter of days during late summer, consistent with when whales are mating (Barrett-Lennard & Heise, 2007).

Other social types are transient killer whales and offshore killer whales. Transient killer whales generally travel in small groups and will hunt marine mammals. Offshore killer whales are uncommon, although groups of over 100 have been observed.

3.17.1.2.3 Behavior

Observations of SRKW behavior indicates that their active time is primarily budgeted to travel (70.4%), followed by foraging (21%), rest (6.8%), and socialization (1.8%) (Noren & Hauser, 2016). Others have suggested that foraging accounts for a greater amount of activity, 40 to 67% (J. Ford & M. Ellis, 2006). Diving tends to be concentrated within the upper 30 m (98 ft) of the water column, with deeper dives of 100 to 200 m (328 to 656 ft) (or more) being occasional (Baird, Hanson, & Dill, 2005). Diving activity is greatest during the day, and dive depths and frequencies are greater for males than females (in adults) but are not greater for adults than juveniles (on average) (Baird et al., 2005). Killer whales are relatively recognizable due to their distinctive coloring and high level of surface activity (e.g., breaching and tail slapping), though SRKWs cannot easily be differentiated from transient individuals. Killer whales communicate, navigate, and hunt using several types of calls, whistles, and clicks (NOAA Fisheries, 2017b).

3.17.1.2.4 Diet

Salmon are identified as their primary prey of SRKW with a high percentage consumed during spring, summer and fall (J. K. B. Ford, 2006; M. Hanson et al., 2010). Feeding records for Southern and Northern Residents show a predominant consumption of Chinook salmon during late spring to fall (J. K. B. Ford et al., 1998). Chum salmon are also taken in significant amounts, especially in fall. Other salmon eaten include coho, pink, steelhead, and sockeye. They also consume non-salmonid fishes included Pacific herring, sablefish, Pacific halibut, quillback and yelloweye rockfish (*Sebastes maliger*), lingcod (*Ophiodon elongates*), and Dover sole (*Microstomus pacificus*) (J. K. B. Ford et al., 1998; M. Hanson et al., 2010) and squid.

SRKWs preferentially consume Chinook salmon. Chinook salmon were the primary prey despite the much lower abundance of Chinook salmon in the study area in comparison to other salmonid fishes (primarily sockeye salmon). Though mechanisms are not well known, factors of potential importance include the species' large size, high fat and energy content, and year-round occurrence in the area. Killer whales also captured older (i.e., larger) than average Chinook salmon (J. K. B. Ford, 2006). Recent research suggests that killer whales are capable of detecting, localizing and recognizing Chinook salmon through their ability to distinguish Chinook salmon echo structure as different from other salmon (Au, Horne, & Jones, 2010).

Scale and tissue sampling in inland waters from May to September indicate that the Southern Residents' diet consists of a high percentage of Chinook salmon, with an overall average of 88% Chinook across the timeframe and monthly proportions as high as >90% Chinook salmon (i.e., July: 98% and August: 92%, (M. Hanson et al., 2010). The significance of the dominance of Chinook in the diet of SRKW is discussed at the end of this section.

3.17.1.2.5 Movement

Based on acoustic activity of whales, it is inferred that whale movements and presence are driven by local availability and abundance of salmon (M. B. Hanson et al., 2013), suggesting that the prey base is the most important habitat element for SRKWs. Recent evidence shows that K and L pods are spending significantly more time off of the Columbia River in March than was previously recognized, suggesting the importance of Columbia River spring Chinook salmon in their diet (M. B. Hanson et al., 2013).

Late summer and early fall movements of Southern Residents in the Georgia Basin are consistent, with strong site fidelity shown to the region generally and high occurrence in the San Juan Island area (M. B. Hanson & Emmons, 2010; Hauser, Logsdon, Holmes, VanBlaricom, & Osborne, 2007). There is inter-annual variability in arrival time and days present in inland waters from spring through fall, with late arrivals and fewer days present during spring in recent years potentially related to weak returns of spring and early summer Chinook salmon to the Fraser River (M. B. Hanson & Emmons, 2010). Similarly, recent high occurrence in late summer may relate to greater than average Chinook salmon returns to South Thompson tributary of the Fraser River (M. B. Hanson & Emmons, 2010). During fall and early winter, Southern Resident pods, (J pod in particular) expand their routine movements into Puget Sound, likely to take advantage of chum and Chinook salmon runs (B. Hanson, Emmons, Sears, & Ayres, 2010; Osborne, 1999).

3.17.1.3 Stressors and Threats

Because of this population's small abundance, it is susceptible to demographic stochasticity — randomness in the pattern of births and deaths among individuals in a population. This can contribute to variance in a population's growth and extinction risk. Small populations are also vulnerable to environmental fluctuations that drive fluctuations in birth and death rates. Finally, small populations can have more vulnerability to variation in birth or death rates of individuals because of differences in their individual fitness.

Several factors identified in the final recovery plan for Southern Residents may be limiting recovery. It is likely that multiple threats are acting in concert to impact the whales. Although it is not clear which threat or threats are most significant to the survival and recovery of Southern Residents, all of the threats identified are potential limiting factors in their population dynamics (NMFS, 2008b).

Factors of concern include the follow:

Water quality: Water quality in areas inhabited by SRKW. Elevated concentrations of pollutants/contaminants in the Salish Sea and elsewhere have been linked to elevated concentrations in salmon and in killer whales (Hickie, Ross, MacDonald, & Ford, 2007; Krahn et al., 2007; Krahn et al., 2009; Lachmuth et al., 2010). Once in the environment, many contaminants accumulate in biological tissues, and some biomagnify up the food chain, reaching high levels in long-lived apex predators like SRKWs. Maternal transfer of persistent and bioaccumulative contaminants from mother to offspring increases killer whale body burdens in subsequent generations (by increasing the baseline burden at birth) (Krahn et al., 2009). Elevated concentrations of pollutants may result in reduced immune function and/or reproductive capability and mortality (Krahn et al., 2007; Krahn et al., 2009).

Reduced quality and quantity of prey: Human influences have had profound impacts on the abundance of many prey species in the northeastern Pacific during the past 150 years, including salmon. As presented in the sections on threats to salmon, the health and abundance of wild salmon stocks have been negatively affected by altered or degraded freshwater and estuarine habitat, including numerous land use activities, from hydropower systems to urbanization, forestry, agriculture and development, harmful artificial propagation practices, and overfishing. Finally, the consequences of climate change contribute to habitat alterations (both in fresh and marine environments) that may negatively impact salmon and reduce productivity. The

availability of adult salmon may be reduced in years following unfavorable conditions to the early life-stage growth and survival of salmon.

Nutritional stress: Another consequence of a reduced pry base is energy expenditure. When prey is scarce, whales likely spend more time foraging than when it is plentiful. Increased energy expenditure and prey limitation can cause nutritional stress. Nutritional stress is the condition of being unable to acquire adequate energy and nutrients from prey resources and as a chronic condition can lead to reduced body size and condition of individuals and lower reproductive and survival rates of a population (e.g., Trites and Donnelly (2003)).

Other human activities: fishing, and disturbance caused by vessels and noise pollution (e.g., caused by military activities) and risk from oil spills (NMFS, 2008b) all pose possible risks to SRKWs.

3.17.1.4 Population Trend and Risk

The historical minimum abundance of SRKWs is estimated to 140 individuals (Krahn et al., 2004; Olesiuk et al., 1990). Data are insufficient to estimate an upper bound but Several lines of evidence (i.e., known kills and removals Olesiuk, 1990 #382), salmon declines, and genetics (Michael J Ford et al., 2011; Krahn et al., 2004) all indicate that the population used to be much larger than it is now. A reasonable assumption is 400 as an upper bound (Krahn et al., 2004).

As of July 1, 2015, there were 81 SRKW (27 whales in J pod, 19 whales in K pod, and 35 whales in L pod). As of December 31, 2016, there were a total of 78 whales (CWR, 2016). Of the three pods, the L pod is the largest at 35 members followed by J, which has 24 members, and then K, which only has 19 members (CWR, 2016). The most recent count as of mid-2019 is 73 whales. The estimated effective size of the population (based on the number of breeders under ideal genetic conditions) is very small at approximately 26 whales, or roughly 1/3 of the current population size (Michael J Ford et al., 2011).

There are several demographic factors of the Southern Resident population that are cause for concern, namely the small number of breeding males (particularly in J and K pods), reduced fecundity, sub-adult survivorship in L pod, and the total number of individuals in the population (review in NMFS (2008b)).

At present, the Southern Resident population has declined to essentially the same size that was estimated during the early 1960s, when it was considered likely to be depleted (Olesiuk et al., 1990) The population suffered an almost 20% decline from 1996-2001 (from 97 whales in 1996 to 81 whales in 2001), largely driven by lower survival rates in L pod. Since then, the overall population has fluctuated but still remained fairly consistent from 2002 to present (from 83 whales in 2002 to 81 whales on July 1, 2015). Over a recent 32-year period (1983-2014), population growth has been variable, with an average annual population growth rate of 0.1% and standard deviation of ± 3.2%.

Extinction Risk: In conjunction with the 2004 status review, NMFS conducted a population viability analysis (PVA) for Southern Resident killer whales (Krahn et al., 2004). Demographic information from the 1970s to fairly recently (1974-2003, 1990-2003, and 1994-2003) were considered to estimate extinction and quasi-extinction risk. We defined “quasi-extinction” as the stage at which 10 or fewer males or females remained, a threshold from which the population

was not expected to recover. The analysis indicated that the Southern Resident killer whales have a range of extinction risk from 0.1 to 18.7% in 100 years and 1.9 to 94.2% in 300 years, and a range of quasi-extinction risk from 1 to 66.5% in 100 years and 3.6 to 98.3% in 300 years.

3.17.1.5 Critical Habitat

Approximately 6,630 sq km (2,560 square miles) of critical habitat were designated for the SRKW at the end of 2006 (NMFS, 2006b). This includes all US waters within the Salish Sea, excluding 18 areas designated for military use (291 sq km; 112 square miles), any waters less than 6.1 m (20 ft) deep (at extreme high tide), and Hood Canal. Military installations were excluded from critical habitat as a matter of national security. The critical habitat was subdivided into three areas that provide necessary habitat elements: a core summer area (Haro Strait and San Juan Islands), Puget Sound, and the Strait of Juan de Fuca. These subareas correspond with seasonal prey (e.g., salmon) concentrations.

The NMFS announced a 12-month finding on a petition from the Center for Biological Diversity to revise the critical habitat designation for the Southern Resident killer whale (Orcinus orca) Distinct Population Segment (DPS) (February 2014, NMFS (2015d)). In February 2015, NMFS announced their intention to proceed with revisions to critical habitat. Although projected to have this complete by 2017, no new rules have been announced (Fed Reg. checked May, 2019).

PBFs for this critical habitat are stated in (NMFS, 2006b) as: water quality to support growth and development; prey species of sufficient quantity, quality, and availability to support individual growth, reproduction and development, as well as overall population growth; and passage conditions to allow for migrating, resting, and foraging.

3.17.1.6 Chinook as Prey for SRKW

In the BE we include a review of the quantity and quality of prey in the action area, because this is the relevance of the action area for SRKW—<u>the quantity and quality of the prey that could be affected by the proposed action</u>. As demonstrated in the above sections, Chinook salmon are important in the diet of SRKW. Two Snake River Chinook stocks, Snake River fall Chinook and Snake River spring-summer Chinook salmon, appear on the list developed by NOAA Fisheries and the Washington Department of Fish and Wildlife regarding West Coast Chinook salmon stocks most important to SRKWs (NOAA Fisheries, 2017a) and (WDFW & ODFW, 2018)

As stated in the life history section, Killer whales, including both the Southern Residents and other populations in Canada and Alaska, are large consumers of West Coast Chinook salmon in terms of biomass and numbers of adult Chinook salmon. The ~75 Southern Resident killer whales, a small subset of all killer whales on the West Coast, consume an estimated 190,000 to 260,000 adult Chinook salmon each year. The Southern Residents depend on a diversity of salmon stocks that together provide the food they need throughout the year. The more diverse and healthy stocks available to the whales, the better they can withstand variable ocean conditions, climate change, and other factors that may affect the availability of salmon. (NMFS, 2018c).

Given their more coastal distribution in fall, winter, and spring, K and L Pods are most likely to prey on Columbia/Snake River stocks directly. They would encounter Snake River fall Chinook along the outer coast, as fall Chinook typically migrate closer to the coast than spring-summer Chinook, which spread out much more widely over the Gulf of Alaska. J Pod might encounter Snake River fall Chinook on the outer coast during forays to the west side of Vancouver Island, but would be less likely to encounter Snake River spring-summer Chinook. (NMFS, 2018a)

Genetic analysis of the Hanson et al. (2010c); 1076 samples indicate that when Southern Resident killer whales are in inland waters from May to September, they consume Chinook stocks that originate from the Fraser River (including Upper Fraser, Mid Fraser, Lower Fraser, N. Thompson, S. Thompson and Lower Thompson), Puget Sound (N. and S. Puget Sound), the Central British Columbia Coast and West and East Vancouver Island. M. Hanson et al. (2010) found that the whales are likely consuming Chinook salmon stocks at least roughly proportional to their local abundance, as inferred by Chinook run-timing pattern and the stocks represented in killer whale prey for a specific area of inland waters, the San Juan Islands. Ongoing studies also confirm a shift to chum salmon in fall (J. Ford, Wright, Ellis, & Candy, 2010; M. Hanson et al., 2010). Although less is known about the diet of Southern Residents off the Pacific coast, the available information indicates that salmon, particularly Chinook, are also important when the whales occur in coastal waters.

Krahn et al. (2002) examined the ratios of DDT (and its metabolites) to various PCB compounds in the whales, concluded that the whales feed primarily on salmon throughout the year rather than other fish species. The predominance of Chinook salmon in their diet in inland waters, even when other species are more abundant, combined with information to date about prey in coastal waters (above), makes it reasonable to expect that Chinook salmon is equally predominant in the whales' diet when available in coastal waters. It is also reasonable to expect that the diet of Southern Residents is predominantly larger Chinook when available in coastal waters. The diet of Southern Residents in coastal waters is a subject of ongoing research.

4. Environmental Baseline

The purpose of this section is to identify "the past and present effects of all Federal, State, or private activities in the action area, the anticipated effects of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the effect of State or private actions which are contemporaneous with the consultation in process" ("Endangered Species Act of 1973, As Amended,"), definition of "effects of the action". These factors affect the species' environment or critical habitat in the action area. The factors are described in relation to species' biological requirements in the action area.

The action area is defined as "all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action" ("Endangered Species Act of 1973, As Amended,"). USEPA's action, the promulgation of new freshwater aluminum water quality criteria for Oregon affects all waters within the state boundaries that are used by ESA-listed species. Seventeen aquatic species were identified for evaluation in this biological evaluation.

Due to the ubiquitous nature of aluminum in the environment, coupled with the paucity of data from both point source dischargers and nonpoint sources, the following analysis is considered somewhat speculative regarding existing point source and nonpoint source contributions of aluminum.

4.1 Current Status of the Environment

The *Oregon State of the Environment Report 2000* published by the Oregon Progress Board in September 2000 (Risser, 2000) provides a comprehensive review of Oregon's environmental baseline. The report was developed using a combination of analyses of existing data and best professional judgment. Aquatic ecosystems, marine ecosystems, estuarine ecosystems, freshwater wetland ecosystems, and riparian ecosystems were among the resources considered. A set of indicators of ecosystem health was proposed for each resource system and as benchmarks for the state's use in evaluating past decisions and for planning future policies to improve Oregon's environment and economy. The report also included findings regarding the environmental health of Oregon's nine ecoregions and conclusions about future resource management needs. Information from this report and other sources has been used to describe the current environment in terms of factors affecting ESA-listed species within the action area.

To supplement the *Oregon State of the Environment Report 2000*, the Oregon Conservation Strategy (ODFW, 2016) was reviewed. This report provides more current information on each of Oregon's nine Level III (Omernik, 1987) ecoregions providing ecoregional characteristics, conservation issues and priorities by ecoregion, ecoregional limiting factors, recommended conservation approaches, as well as strategy species and habitats. ODFW (2016) define strategy species and strategy habitats as species or native habitats of conservation concern including wildlife, fish, invertebrates, plants and algae. ODFW (2016) uses the EPA Level III Ecoregions but combines the Snake River Plain with the Northern Basin and Range. The Nearshore ecoregion is bounded by Oregon's Territorial Sea at three nautical miles, to the supratidal zone, then into portions of estuaries with saltwater-dependent species.

4.2 Habitat Effects

Past and present actions within the state of Oregon have adversely impacted the habitat of ESA-listed species, bringing about changes in the amount and timing of water flows, reductions in water quality, introduction of nonnative species, and the alteration, elimination, fragmentation, and degradation of riparian and aquatic habitat.

The quality and quantity of freshwater habitat in much of Oregon has been adversely impacted by water management activities. Oregon's currently available water supplies are fully or often over-allocated during low flow months of summer and fall. In the Columbia Plateau ecoregion, less than 20 percent of instream water rights can expect to receive their full allocation 9 months of the year. In the Willamette Valley and Cascades ecoregions, more than 80 percent of the instream water rights can expect to receive their full allocation in the winter, but only about 25 percent in the early fall. Increased demand for water is linked to the projected 34 percent increase in human population in the state over the next 25 years. Expected increases in certain land uses will further increase demand on already strained water resources. For example, projected increases in national output of metals is expected to increase 4.3% annually along with increases in irrigated agricultural output (NMFS, 2012b). Water quantity is a limiting factor for fish species where seasonally low flows can limit habitat suitability and, in turn, overall reproductive success. Depletion and storage of natural flows have altered natural hydrological cycles in basins occupied by listed ESUs. This may cause juvenile salmon mortality through migration delay resulting from insufficient flows or habitat blockages, loss of sufficient habitat due to dewatering and blockage, stranding of fish resulting from rapid flow fluctuations, entrainment of juveniles into poorly screened or unscreened diversions, and increased juvenile mortality resulting from increased water temperatures (Spence et al., 1996). Reduced flows have also negatively affected fish habitats because they have resulted in increased deposition of fine sediments in spawning gravels, decreased recruitment of new spawning gravels, and encroachment of riparian and exotic vegetation into spawning and rearing areas. Consumptive use of freshwater in the upper watersheds has reduced freshwater inflow to estuaries by as much as 60-80 percent, thus reducing the natural dilution and flushing of pollutants. Further, some climate models predict 10-25 percent reductions in late spring, summer, and early fall runoff amounts in the coming decades. As the human demand for freshwater increases, groundwater supplies decrease, further reducing the amount of water required as source water for inland aquatic habitats.

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 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. Temperatures above 59 degrees Fahrenheit are known to impede bull trout foraging and migration. The presence of toxicants may elicit effects from death or injury, to reduced reproduction potential (NMFS, 2012b). The very specific habitat requirements leave this species especially vulnerable when habitat characteristics are less than ideal. Widespread channel widening and reduced base flows further exacerbate seasonal water temperature extremes.

Generally, water quality in Oregon is poor for salmon except in mountainous areas. Areas with excellent or good water quality occur most often in forested uplands. Poor or very poor water quality occurs most often in non-forested lowlands where land has been converted to agricultural and urban uses. Most ecoregions include some rivers and streams with excellent water quality and others with very poor water quality. Only the Cascades ecoregion has close to excellent water quality overall as shown by annual OWQI measurements (Table 4-1) (ODEQ, 2018b). The Willamette Valley, Columbia Plateau, Northern Basin and Range, and the southern end of the Eastern Cascade Slope ecoregions have poor water quality indices (Table 4-1).

Toxic pollutants, which are not considered by the OWQI, can also degrade water quality. The effects of pesticides and fertilizers, especially nitrates, on water supplies and aquatic habitats are a significant concern. Future actions causing diffuse and concentrated urban growth are expected to increase the overall contaminant loading from wastewater treatment plants as well as water/sediment runoff which can carry elevated loads of oil, heavy metals, PAHs and other chemical pollutants to surface waters (NMFS, 2012b). Sedimentation from logging, mining, urban development, and agriculture is a primary cause of salmon habitat degradation. In general, the effects of sedimentation on salmon are well documented and include (1) clogging and abrasion of gills and other respiratory surfaces; (2) adhering to the chorion of eggs; (3) providing conditions conducive to entry and persistence of disease-related organisms; (4) inducing behavioral modifications; (5) entombing early life stages; (6) altering water chemistry by the absorption of chemicals; (7) affecting usable habitat by scouring and filling pools and riffles and changing bedload composition; (8) reducing photosynthetic growth and primary production; and (9) reducing intergravel permeability and dissolved oxygen (Spence et al., 1996).

Nonnative species now represent a significant portion of Oregon's estuarine flora and fauna. Some, such as the European green crab, pose serious threats to the native estuarine communities necessary to support healthy salmon populations. More than 32 species of freshwater fish have been introduced into Oregon, and are now self-sustaining, making up approximately one-third of Oregon's freshwater fish fauna. Introduced species are frequently predators on native species, compete for food resources, and alter freshwater habitats. In 1998, introduced species were found to represent 5 percent of the number of species found in the upper Willamette River, but accounted for 60 percent of the observed species in the lower river near Portland. As another example, the invasive common carp (*Cyprinus carpio*) has greatly impacted waterfowl production by altering the ecological dynamics and water quality of Lake Malheur.

Table 4-1. OWQI for each of Oregon's ecoregions for 10 year spans from 2001 – 2010 through 2009 – 2018 (ODEQ, 2018a).

Ecoregion	2001-10 Score	2002-11 Score	2003-12 Score	2004-13 Score	2005-14 Score	2006-15 Score	2007-16 Score	2008-17 Score	2009-18 Score
Blue Mountains	81	81	82	82	82	82	82	82	82
Cascades	88	89	90	87	85	85	87	89	90
Coast Range	78	79	79	80	80	81	82	82	82
Columbia	80	81	76	76	76	76	76	77	77
Eastern Cascades	81	81	84	82	80	77	77	78	79
Klamath	64	66	69	69	69	70	70	71	71
Northern Basin and Range	-	-	63	78	70	74	73	75	75
Snake River	30	30	30	31	30	31	32	32	33
Willamette	79	80	79	79	79	79	79	79	78

[1] – OWQI Range 90 – 100 = excellent; 85 – 89 = good; 80 – 84 = fair; 60 – 79 = poor; <60 = very poor

Depending on the species, salmon spend from a few days to 1 or 2 years in an estuary before migrating to the ocean. Natural variability and extremes in temperature, salinity, tides, and river flow make estuarine ecosystems and organisms relatively resilient to disturbance; however, alterations such as filling, dredging, the introduction of nonnative plant and animal species, and excessive waste disposal have changed Oregon's estuaries, reducing their natural resiliency and functional capacity and also potentially threatening the economic productivity of resource waters. The most significant historical changes in Oregon's estuaries are the diking, draining, and filling of wetlands; and the stabilization, dredging, and maintenance of navigation channels. Between 1870 and 1970, approximately 50,000 acres, or 68 percent, of the original tidal wetland areas in Oregon estuaries were lost. Despite these significant historical wetland conversions and continuing degradation by pollutants, nuisance species, and navigational improvement, much of the original habitat that existed in the mid-1800s is still relatively intact. Hundreds of acres of former estuarine marshes are now being restored.

Oregon contains approximately 114,500 miles of rivers and streams. No statewide measurements exist of the area of riparian vegetation, although some estimates have been made for more localized regions. Using the conservative estimate of a 100-yard riparian corridor on each side of the stream, the total area of riparian habitats for flowing water in Oregon may be 22,900 square miles. That is equal to approximately 15 percent of the total area of the state.

With the exception of fall chinook, which generally spawn and rear in the mainstem, most salmon and steelhead spawning, and rearing habitat is found in tributaries where riparian areas are a major habitat component. Healthy riparian areas retain the structure and function of natural landscapes as they were before the intensive land use and land conversion that have occurred over the past 150 to 200 years. However, land use activities have reduced the numbers of large trees, the amount of closed-canopy forests, and the proportion of older forests in riparian areas. In western Oregon, riparian plant communities have been altered along almost all streams and rivers.

In the western Cascades, Willamette Valley, Coast Range, and Klamath Mountains, riparian areas on privately owned land are dominated by younger forests because of timber harvest, whereas riparian areas on public lands have more mature conifers. Old coniferous forests now account for approximately 20 % of the riparian forests in the Cascades, but only 3 % in the Coast Range. Older forests historically occurred along most of the McKenzie River, but now account for less than 15 % of its riparian forests. Along the mainstem of the upper Willamette River, channel complexity has been reduced by 80 % and the total area of riparian forest has been reduced by more than 80 % since the 1850s. Downstream portions of the Willamette River have experienced little channel change, but more than 80 % of the historical riparian forest has been lost.

Oregon’s Nearshore ecoregion can be negatively impacted both directly or indirectly by various human activities. Habitats within the Nearshore ecoregion may be impacted by alterations in sediment transport, coastal development, shoreline armoring, beach maintenance, climate change, non-native species introductions, and bottom fishing methods, among other factors (ODFW, 2016).

Effects caused from direct human use of the landscape can have deleterious effects on the nearshore environment. Land use conversion and urbanization contributes to habitat loss and fragmentation which can isolate populations of wildlife. Residential development is sometimes

sought after in the little remaining ecologically sensitive areas of the state. An increasing amount of recreation can also have negative impacts on the nearshore ecoregion when the activity occurs on or near sensitive habitats (ODFW, 2016).

4.2.1 Hydrosystem Effects

Beginning in the early 1800s, riparian areas in eastern and southern Oregon were extensively changed by such activities as beaver trapping, logging, mining, livestock grazing, agriculture, and associated water diversion projects. Very little of the once-extensive riparian vegetation remains to maintain water quality and provide habitats for threatened salmon. The Willamette River, located in Willamette Valley Ecoregion, has been disconnected from its floodplain where braided channels historically helped to disperse flood waters, deposit fertile soil, moderate temperature and flow, and allow for slow-water habitats. Dams affected flow, sedimentation, and gravel patterns, which in turn have diminished regeneration and natural succession of riparian vegetation along downstream rivers. Introduced plant species pose a risk to some riparian habitat by dominating local habitats and reducing the diversity of native species. Improper grazing in riparian areas is another significant threat.

In the Columbia River Basin, anadromous salmonids, especially those above Bonneville Dam, have been dramatically affected by the development and operation of the Federal Columbia River Power System (FCRPS). Storage dams have eliminated spawning and rearing habitat and have altered the natural hydrograph of the Snake and Columbia Rivers, decreasing spring and summer flows and increasing fall and winter flows. Power operations cause flow levels and river elevations to fluctuate, affecting fish movement through reservoirs and riparian ecology, and stranding fish in shallow areas. The eight dams in the migration corridor of the Snake and Columbia Rivers alter smolt and adult migrations. Smolts experience a high level of mortality as they pass through the dams. The dams also have converted the once-swift river into a series of slow-moving reservoirs, slowing the smolts' journey to the ocean and creating habitat for predators. Water velocities throughout the migration corridor now depend far more on volume runoff than before development of the mainstem reservoirs.

Even before mainstem dams were built, habitat was lost or severely damaged in small tributaries as a result of the construction and operation of irrigation dams and diversions; inundation of spawning areas by impoundments; and siltation and pollution from sewage, farming, logging, and mining (Fulton, 1968). Recently, the construction of hydroelectric and water storage dams without adequate provision for adult and juvenile passage in the Upper Snake River has kept fish from all spawning areas upstream of Hells Canyon Dam.

The lower reaches of the Columbia River are highly modified by urbanization and dredging for navigation. The upland areas covered by this ESU are extensively logged, affecting water quality in the smaller streams used primarily by summer runs. In addition, all major tributaries used by Lower Columbia River (LCR) steelhead have some form of hydraulic barrier that impedes fish passage. Barriers range from impassible structures in the Sandy River Basin that block access to extensive, historically occupied, steelhead habitat, to passable but disruptive projects on the Cowlitz and Lewis Rivers. The Biological Review Team (BRT, 1997) viewed the overall effect of hydrosystem activities on this ESU as an important determinant of extinction risk.

Human activities have had vast effects on the salmonid populations in the Willamette River Basin. First, the Willamette River, once a highly braided river system, has been dramatically simplified through channelization, dredging, and other activities that have reduced rearing habitat (i.e., stream shoreline) by as much as 75 %. In addition, the construction of 37 dams in the basin has blocked access to over 377.7 miles of stream and river spawning habitat. The dams also alter the temperature regime of the Willamette and its tributaries, affecting the timing of development of naturally spawned eggs and fry. Water quality is also affected by development and other economic activities. Agricultural and urban land uses on the valley floor, as well as timber harvesting in the Cascade and Coast ranges, contribute to increased erosion and sediment load in Willamette River Basin streams and rivers. Finally, since at least the 1920s, the Lower Willamette River has suffered municipal and industrial pollution.

Detroit and Big Cliff Dams cut off 291.4 miles of spawning and rearing habitat in the North Santiam River. In general, habitat in this ESU has become substantially simplified since the 1800s by removal of large woody debris to increase the river's navigability.

Spawning and rearing habitat in the Columbia River and its tributaries upstream of the Yakima River include dry areas where conditions are less conducive to steelhead survival than in many other parts of the Columbia River Basin (Mullan, Rockhold, & Chrisman, 1992). Salmon in the Upper Columbia River (UCR) ESU must pass up to nine federal and private dams, and Chief Joseph Dam prevents access to historical spawning grounds farther upstream. Degradation of remaining spawning and rearing habitat continues to be a major concern associated with urbanization, irrigation projects, and livestock grazing along riparian corridors. Overall harvest rates are low for this ESU, currently less than 10 % (ODFW, 1998).

Construction of the Chief Joseph and Grand Coulee Dam caused blockages of substantial habitat, as did the construction of smaller dams on tributary rivers. Habitat issues for the UCR ESU relate mostly to irrigation diversions and hydroelectric dams, as well as to degraded riparian and instream habitat from urbanization and livestock grazing.

Hydrosystem projects create substantial habitat blockages in the Snake River (SR) ESU; the major ones are the Hells Canyon Dam complex (mainstem Snake River) and Dworshak Dam (North Fork Clearwater River). Minor blockages are common throughout the region. Steelhead spawning areas have been degraded by overgrazing, as well as by historical gold dredging and sedimentation due to poor land management. Habitat in the Snake River Basin is warmer, drier, and often more eroded than elsewhere in the Columbia River Basin or in coastal areas.

The only substantial habitat blockage now present in the Middle Columbia River (MCR) ESU is at Pelton Dam on the Deschutes River, but minor blockages occur throughout the region. Water withdrawals and overgrazing have seriously reduced summer flows in the principal summer steelhead spawning and rearing tributaries of the Deschutes River. This is significant because high summer and low winter temperatures are limiting factors for salmonids in many streams in this region (D L Bottom, Howell, & Rodgers, 1985).

The water quality of the Nearshore ecoregion is often affected by coastal as well as inland development that results in contaminated runoff. Elevated water temperatures resulting from altered hydrology and depth could result in such negative effects as significant blooms of harmful algae and subsequent high levels of marine biotoxins.

4.2.2 Hatchery Effects

For more than 100 years, hatcheries in the Pacific Northwest have been used to replace natural production lost as a result of the FCRPS and other development, not to protect and rebuild natural populations. Consequently, most salmon populations in this region are primarily hatchery fish. In 1987, for example, 95 % of the Coho, 70 % of the spring Chinook, 80 % of the summer Chinook, 50 % of the fall Chinook, and 70 % of the steelhead returning to the Columbia basin originated in hatcheries (CBFWA, 1990).

While hatcheries certainly have contributed greatly to the overall numbers of salmon, only recently has the effect of hatcheries on native wild populations been demonstrated. In many cases, these effects have been substantial. For example, production of hatchery fish, among other factors, has contributed to the 90 % reduction in wild Coho salmon runs in the Lower Columbia River over the past 30 years (Weitkamp et al., 1995). Hatcheries have traditionally focused on providing fish for harvest, with less attention given to identifying and resolving factors causing declines of native runs.

NMFS has identified four primary categories of risk that hatcheries can pose on wild-run salmon and steelhead (1) ecological effects, (2) genetic effects, (3) overharvest effects, and (4) masking effects (NMFS, 2000). Ecologically, hatchery fish can increase predation on, displace, and/or compete with, wild fish. These effects are likely to occur when fish are released in poor condition and do not migrate to marine waters, but rather remain in the streams for extended rearing periods during which they may prey on or compete with wild fish. Hatchery fish also may transmit hatchery-borne diseases, and hatcheries themselves may release diseases into streams via water effluents.

4.2.3 Harvest Effects

The Biological Opinion prepared by the NMFS for ESA consultation on the operation of the FCRPS (NMFS, 2012b) describes the effects of commercial harvests on Columbia River Basin salmonids. Overharvesting of spring and summer chinook beginning in the 1800s is cited as one of the factors contributing to the species' decline (Ricker, 1959). Following the sharp declines in spring and summer chinook in the late 1800s, fall chinook salmon became a more important component of the catch. Fall chinook have provided the greatest contribution to Columbia River salmon catches in most years since 1890. Through the first part of the 20th century, the commercial catch was usually canned for marketing. The peak year of commercial sales was 1911, when 49.5 million pounds of fall chinook were landed. Columbia River chinook salmon catches were generally stable from the beginning of commercial exploitation until the late 1940s, when landings declined by about two-thirds to a level that remained stable from the 1950s through the mid-1980s (ODFW, 1998). Since 1938, total salmonid landings (all species) have ranged from a high of about 2,112,500 fish in 1941 to a low of about 68,000 fish in 1995 (Figure A.1 in ODFW (1998)).

Over harvest of Southern Oregon Northern California (SONC) Coho for commercial recreational, scientific, or educational purposes has been suggested as one of the factors contributing to the decline of this ESU. Harvest management practiced by Native American tribes is conservative and has resulted in limited impact on the Coho stock in the Klamath and Trinity Rivers.

Overfishing in non-tribal fisheries is believed to have been a significant factor in the decline of Coho salmon. Marked hatchery Coho may be harvested in the Rogue River. All other recreational Coho salmon fisheries in the Oregon portion of this ESU are closed. Collection for scientific research and educational programs has had little or no impact on Coho populations in the ESU.

The Oregon Conservation Strategy (ODFW, 2016) indicates that nearshore species may be impacted by commercial/recreational overharvest, as well as through bycatch and discard of non-target species. It also indicates that estimates of abundance and complete life history of nearshore species is largely incomplete. Habitat and species monitoring are very important to evaluate status and trends, especially in terms of predator-prey population dynamics, the introduction of non-native species, algal blooms, climate change, and ocean acidification, among other issues. An understanding of these topics is needed to understand the nearshore ecosystem and effects due to human interaction.

4.3 Sources of Aluminum in Oregon

According to the Agency for Toxic Substances and Disease Registry (ATSDR, 2008), aluminum is the most abundant metal in the Earth's crust and it occurs ubiquitously in natural waters as a result of the weathering of aluminum-containing rocks and minerals. The presence of aluminum ions in streams may result from industrial wastes but is more likely to come from the wash water of drinking water treatment plants. Many aluminum salts are readily soluble; however, there are some that are very insoluble. Those that are insoluble will not exist long in surface water but will precipitate and settle. In addition to point source discharges, aluminum concentrations in surface waters can also be increased by several nonpoint sources, such as urban stormwater runoff, agriculture, forestry, acid mine drainage, and atmospheric deposition (Figure 4-1).

Aluminum may enter the environment through many pathways. The most direct routes are through acid mine drainage from active and abandoned mines and point source discharges from industrial activities such as plating, textile, tanning, and steel industries. Municipal waste water treatment plants may be a source of Al to the environment while urban runoff could be a significant source of aluminum to the environment. Particulate metals from combustion and dust can be transported through the air.

Aluminum can enter the aquatic environment in a dissolved form or be attached to organic and inorganic particulate 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. Aluminum can flux between different states and forms in an aquatic environment due to changes in pH and temperature. 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. Dredging and disposal operations can result in substantial suspension and re-suspension of particulates in the water column, including those contaminated with aluminum. Section 5 discusses in more detail the flux between dissolved and particulate aluminum and how the form of Al is related to its toxicity to aquatic organisms.

Table 4-2. Potential number of facilities that utilize aluminum in their operations in Oregon based on EPA-HQ-OW-0694-0125, *Economic Analysis for the Proposed Rule: Aquatic Life Criteria for Aluminum in Oregon* (USEPA, 2019c).

Facility Type[1]	Potential Number of Individual Facilities
Aluminum anodizing facility (Discharge to POTWs)	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

Notes:

1, For purposes of the Economic Analysis associated with the proposal and because of the lack of data for aluminum in point source discharges in the State, the EPA additionally identified potential costs for point source dischargers that utilize aluminum in their operations and, therefore, could potentially be affected by the proposed rule by presuming a need for additional controls or product substitution, without facility-specific data. For this supplemental point source analysis, the EPA evaluated potential costs to three types of facilities (aluminum anodizing facilities, drinking water treatment plants, and wastewater treatment facilities) that could incur costs under the proposed rule (if the facilities were found to have RP). This analysis required the EPA to make a number of assumptions, and this analysis is only as accurate as those assumptions.

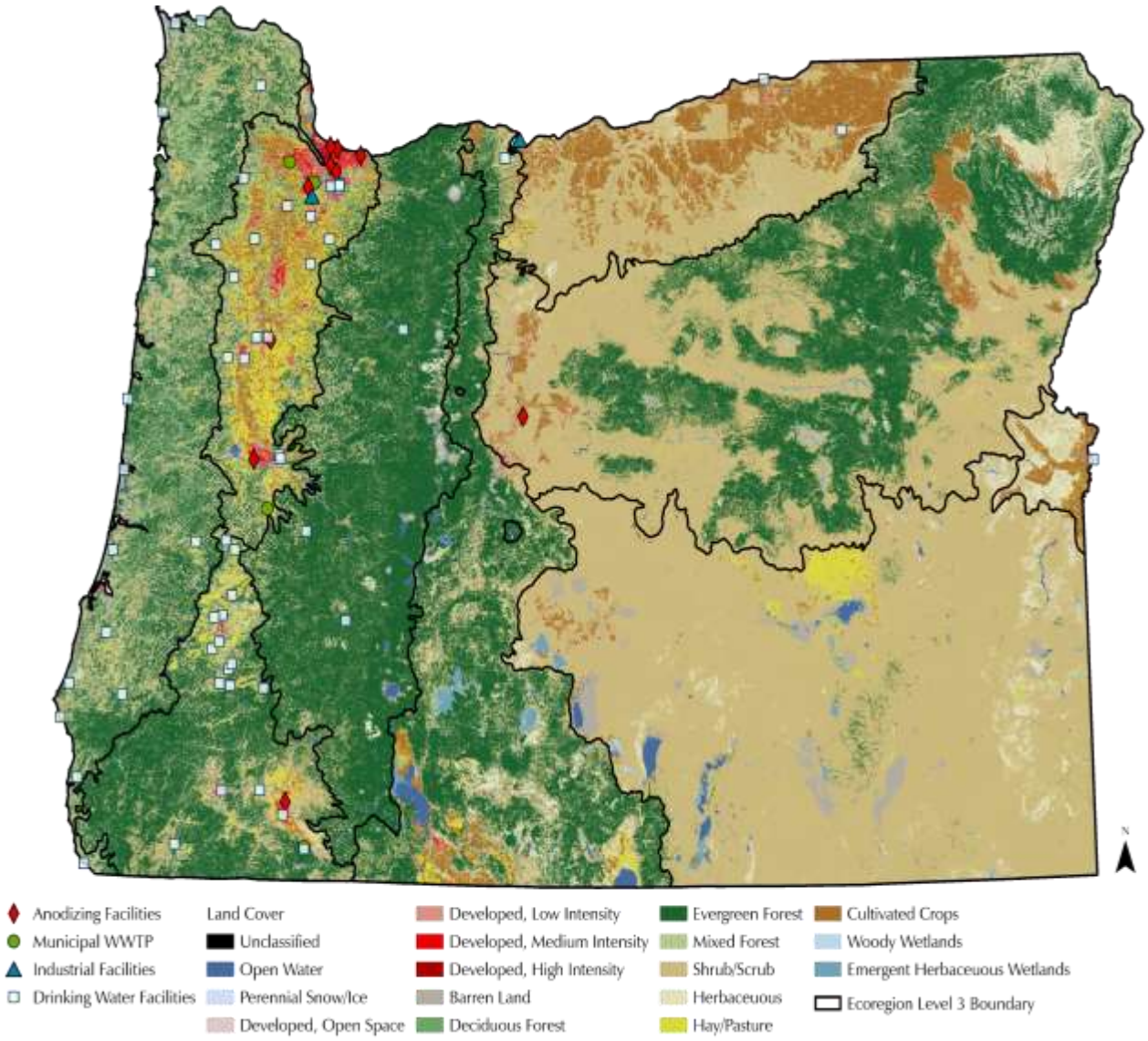


Figure 4-1. Known and potential point sources of aluminum in Oregon which include wastewater treatment plants, indirect anodizing facilities, industrial facilities and drinking water facilities.

4.3.1 Municipal and Industrial Dischargers

The EPA's Integrated Compliance Information System NPDES (ICIS-NPDES) database identifies individual NPDES permits for 330 industrial dischargers and POTWs in Oregon, with 69 of those facilities being major facilities (USEPA, 2017). For POTWs, major facilities are those that have a design flow of one million gallons per day (mgd) or greater, serve a population of 10,000 people or more, are required to have a pretreatment program, or have the potential to cause significant water quality impacts. Industrial dischargers are classified as a major facility when the point value reaches a specific level which is based on several criteria, including toxic pollutant potential, flow volume, and water quality factors such as the impairment of the receiving water. One major facility and one minor facility (both industrial dischargers) in Oregon currently have effluent limitations for aluminum in their individual NPDES permits. These limits

are based on the minimum contaminant level of 50 μg/L (major) or a chronic aquatic life criterion of 87 μg/L, which was adopted into ODEQ's standards in 2004.

ODEQ issues NPDES general permits for a wide variety of discharger categories, including cooling water/heat pumps, filter backwash, fish hatcheries, log ponds, boiler blowdown, seafood processing, suction dredges, stormwater from gravel mining, stormwater construction activities, industrial stormwater, oily stormwater runoff, tank cleanup and treatment of groundwater, wash water, noncontact geothermal water, and small municipal separate storm sewer systems (MS4). Currently there are no specific effluent limits or monitoring requirements for aluminum in these general permits. EPA identified existing facilities that are not currently regulated for aluminum in their discharge (albeit speculative, given minimal data) that could be sources of aluminum. Currently, there are 57 drinking water facilities with NPDES permits in Oregon (USEPA, 2017). Aluminum sulfate (alum) and aluminum chlorohydrate/polyaluminum chloride are often used as coagulants in the treatment of drinking water. Coagulation/flocculation is a process used to remove turbidity, color, and some bacteria from water. In a 2011 national study of drinking water treatment plant residuals, including approximately half a dozen plants in Oregon, the EPA estimated that long-term average aluminum concentrations discharged from drinking water plants ranged from 0.177 to 2.16 mg/L (177.0 to 2,160 μg/L) based on discharge monitoring report data (USEPA, 2011).

In addition, metal salts (alum, sodium aluminate, or polyaluminum chloride) are commonly utilized in chemical treatment to remove phosphorus from wastewater. Chemical treatment is the most common method used for phosphorus removal to meet effluent concentrations below 1.0 mg/L total phosphorous. The required chemical dose is related to the liquid phosphorus concentration. Four individually permitted wastewater treatment facilities in Oregon currently have effluent limitations for phosphorus and use metal salts that contain aluminum in the treatment process (Table 4-2). Aluminum anodizing facilities may generate wastewater containing aluminum. In Oregon, there are no direct dischargers and 12 aluminum anodizing facilities that are indirect dischargers to POTWs (Figure 4-1).

4.3.2 Urban Stormwater

Stormwater discharges are generated by precipitation runoff from land, pavement, building rooftops, and other surfaces. Urban stormwater from any municipal and industrial areas may contribute metals, including aluminum, to surface waters. ODEQ regulates stormwater discharges from MS4s through NPDES permits. The MS4 permits require the discharger to develop and implement a stormwater management program (SWMP), with the goal of controlling pollutant discharges to the maximum extent practicable (MEP). In the State's Phase II general permits, ODEQ interprets the MEP requirement as being consistent with requiring all controls that are reasonable and available per ORS 468.020(2)(b). The management programs specify the best management practices (BMPs) that will be used in the following areas: public education and outreach; public participation and involvement; illicit discharge detection and elimination; construction runoff control; post-construction runoff control; and pollution prevention and good housekeeping. Permittees implement BMPs through an adaptive management or iterative approach.

ODEQ has issued four area-wide "Phase I" MS4 permits covering Clackamas County Group; Portland Group; Multnomah County, Gresham Group, City of Salem, City of Eugene; and the Oregon Department of Transportation (ODEQ, 2017a). Phase I MS4s are those systems serving populations equal to or greater than 100,000. "Phase II" (or "small") MS4s are those systems serving municipalities with populations less than 100,000 located within Census Bureau-defined urbanized areas. There are 19 jurisdictions in Oregon covered by 15 Phase II MS4 permits (with some jurisdictions covered as co-permittees). These Phase II MS4s were previously covered by individual NPDES permits; however, ODEQ is in the process of developing a general permit to cover all current and future Phase II permittees (ODEQ, 2017b). Federal regulations also provide States the discretion to require other MS4s outside of urbanized areas to apply for a permit.

Industrial dischargers, including those engaged in manufacturing, transportation, mining, and steam electric power generation, as well as scrap yards, landfills, certain sewage treatment plants, and hazardous waste management facilities may have stormwater requirements in their NPDES permits. ODEQ requires industrial stormwater dischargers to develop a Stormwater Pollution Control Plan (SWPCP) that includes a monitoring plan and discussion of the site controls that the discharger will implement to prevent stormwater pollution (ODEQ, 2017c).

4.3.3 Agriculture and Forestry

Aluminum is the most plentiful metal in the Earth's crust (ATSDR, 2008). As such, activities that increase soil erosion may transport aluminum to surface waters (see discussion in CCME (2003)). Because agriculture and forestry involve disturbances to substantial land areas, these sectors represent potentially significant sources of aluminum through erosion and runoff to surface waters. However, most of this Al is likely to be particulate bound and thus not-bioavailable to organisms.

Aluminum may also be present in pesticides used in agriculture. Aluminum phosphide is a fumigant that is primarily used to control insects and rodents (USEPA, 1998). While data from the United States Geological Survey (USGS) (USGS, 2015) show that aluminum phosphide was used widely on Oregon agricultural lands as late as 1996, there was little to no use of the pesticide in the State from 1997 through 2014.

The Oregon Department of Agriculture (ODA)'s Agricultural Water Quality Management Program (AWQMP) is charged with assisting the agriculture industry in reducing pollution from agricultural sources. The legal foundation of the AWQMP is the 1993 Agricultural Water Quality Management Act, also known as Senate Bill 1010 (SB1010). SB1010 provides the department with enforcement authority to address situations where corrective action is needed but is not voluntarily being taken by an operator. In those cases where a farmer or rancher refuses to take action, the law authorizes the department to require corrective measures or use civil penalties to address the issue.

ODA has identified 38 watershed-based agricultural water quality management areas. Each management area has a local advisory committee consisting of water quality specialists, local farmers, ranchers, and community leaders. All 38 local advisory committees have approved Agricultural Water Quality Management Area Plans along with enforceable administrative rules to ensure all landowners do their part to avoid and resolve water quality problems. These plans

must describe a program to achieve WQS and ensure that landowners will prevent and control water pollution from agricultural activities and soil erosion (ODA, 2016).

Oregon's Forest Practices Act (FPA) regulates operations in the commercial forestry, and a large portion of the act aims to protect surface waters. Landowners are required to implement BMPs such as maintenance of riparian buffers, restricted use of forest roads during wet weather, and construction of roads using placement and methods to minimize runoff (ODOF, 2017; OFRI, 2011).

4.3.4 Mining

Active and abandoned or inactive mine sites, whether they are for the extraction of aluminum or other materials, can contribute to elevated aluminum concentrations in surface waters (USGS (2011); ESRI (2019)) (Figures 4-2 and 4-3). Mining activities require large amounts of water to process ore, which results in discharged effluent, seepage from tailings storage facilities, and potential accidental releases. In addition to this direct contamination, exploratory activities such as road building and erosion of mineralized overburden during mine construction can also contribute to the contamination of surface water by heavy metals (Jennings, D.R. Neuman, & Blicker., 2008). The leeching of heavy metals from excavated rock is especially problematic when mine wastes with high acid-generating potential are oxidized through exposure to surface conditions.

Abandoned and inactive mines exist throughout Oregon, including mining districts in eastern Oregon, southwest Oregon, and the Willamette Basin (Figure 4-2 and 4-3). Over the last century, minerals and metals such as gold, silver, copper, zinc, and mercury have been extracted from these mines. 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 (ODEQ, 2012a). 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 the Middle Creek (USEPA, 2008).

4.3.5 Atmospheric Deposition

Atmospheric deposition is a potential nonpoint source of aluminum to surface waters (ATSDR, 2008) and can contribute to surface water loads through either direct or indirect deposition. Direct deposition occurs when pollutants are deposited directly on surface waters from the atmosphere. Indirect deposition reflects the process by which metals deposited on the land surface are washed off during storm events and enter surface water through stormwater runoff.

Atmospheric deposition is not directly addressed through any existing regulation and if it were a significant source could be indirectly addressed through TMDLs by inclusion in the load allocation portion of the TMDLs.

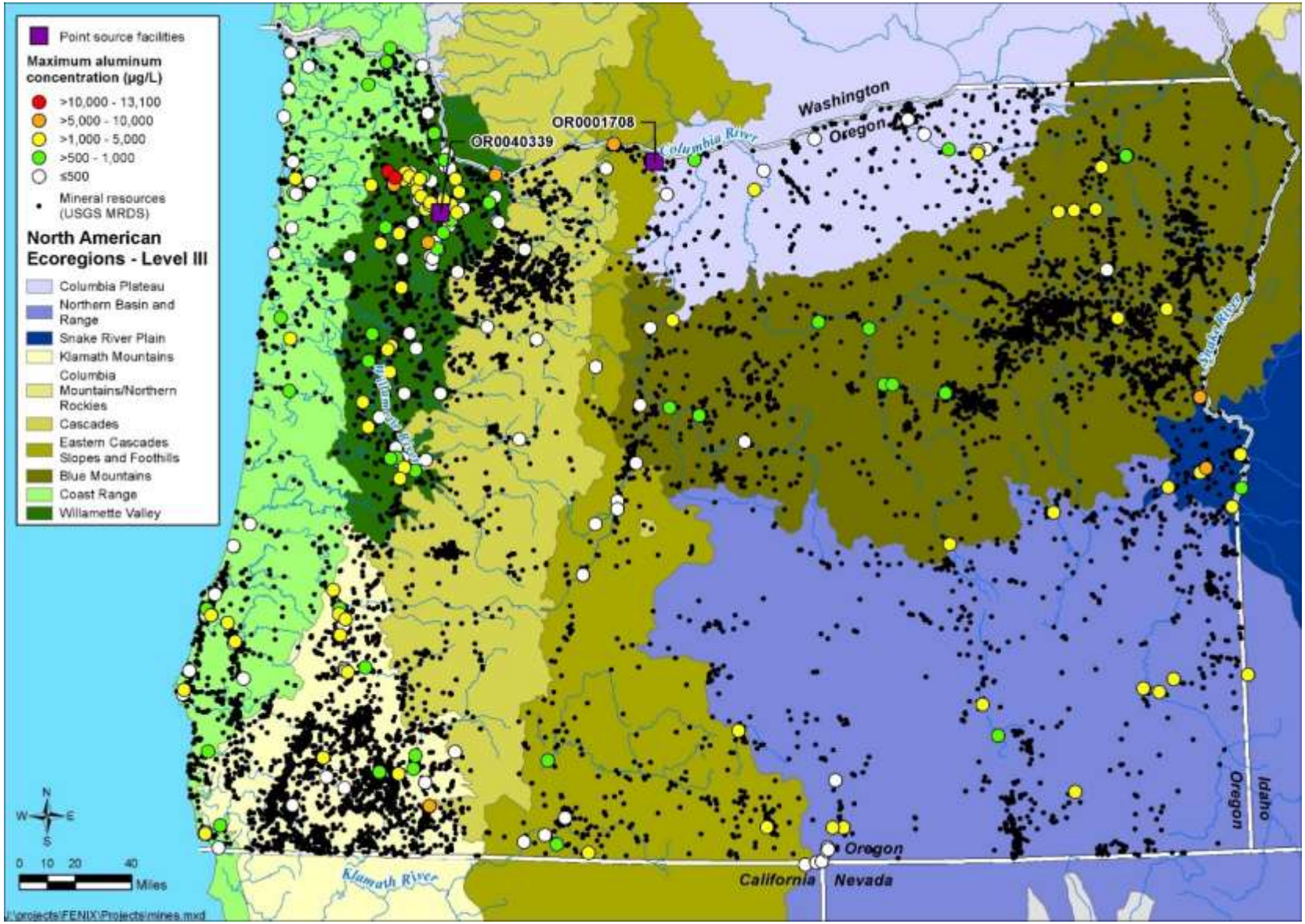


Figure 4-2. Mine sites, prospects, and known deposits of hard-rock minerals in Oregon, grouped by ecoregion (ESRI, 2019; USGS, 2011). Total recoverable Al shown.

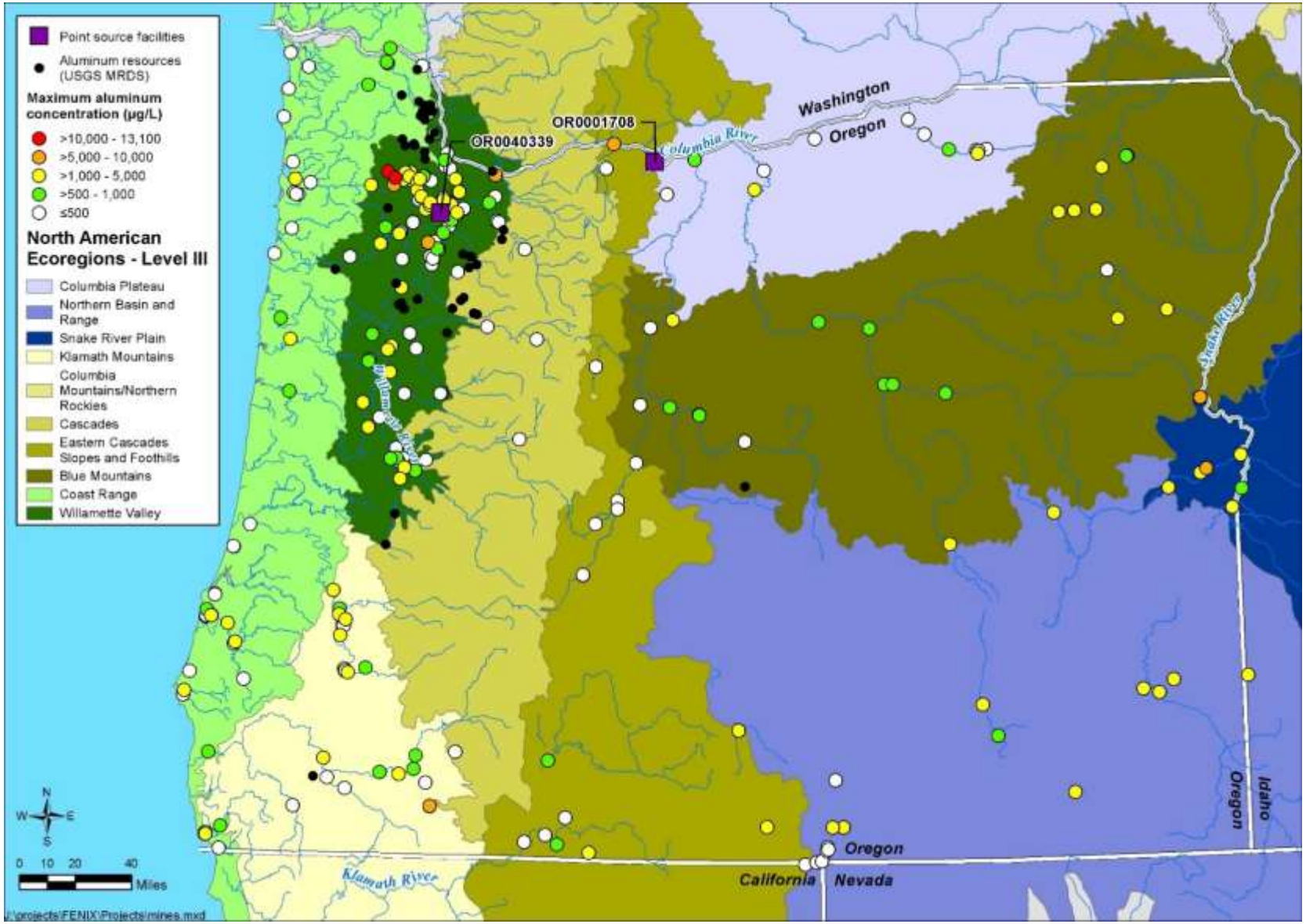


Figure 4-3. Mine sites, prospects and known deposits where aluminum has been identified in Oregon, grouped by ecoregion (ESRI, 2019; USGS, 2011). Total recoverable Al shown.

4.4 Oregon Water Quality

Under section 303(d) of the CWA, states and tribes are required to provide EPA a biennial list of water body segments that do not meet water quality standards. On its 2012 303(d) list Oregon identified more than 13,300 stream miles listed for at least one pollutant. Pollutants identified on the 303(d) list fall into several major groups including sediment, nutrients, metals, bacteria, oxygen demand, and toxic organics. Exceedances of temperature and bacteria are the most prevalent, followed by dissolved oxygen. Other commonly listed pollutants on the Oregon 303(d) list are sediment, metals, and nutrients. Most water bodies in Oregon on the 303(d) list did not meet water quality standards for temperature. Many factors can cause high stream temperatures, but they are typically related to land-use practices and non-point pollution rather than point-source discharges. Some common actions that result in high stream temperatures have been the removal of trees or shrubs that directly shade streams, excessive water withdrawals for irrigation or other purposes, and warm irrigation return flows. Loss of wetlands and increases in groundwater withdrawals have contributed to lower base-stream flows, which in turn contributed to temperature increases. Channel-widening and other land uses that reduced stream depth also cause temperature increases.

As of 2012, Oregon did not have any waters identified as impaired for aluminum, nor any aluminum TMDLs. Three sections of the Lower Columbia River were placed into Category 2[13] for aluminum (pursuant to the narrative criteria for toxic substances) based on an assessment conducted in 1998. No other aluminum classifications were itemized in ODEQ's 2012 Integrated Report database (ODEQ, 2014).

4.4.1 Effect of Surface Water Chemistry on Aluminum Toxicity

Aluminum is one of the most abundant elements and metals in the earth's crust and occurs in many rocks and ores, but never as a pure metal. The presence of aluminum ions in streams may result from industrial wastes but is more likely to come from the wash water of drinking water treatment plants. Many aluminum salts are readily soluble; however, there are some that are very insoluble. Those that are insoluble will not exist long in surface water but will precipitate and settle and are not typically thought to be toxic to aquatic life. Waters containing high concentrations of aluminum can become toxic to aquatic life if the pH is lowered (e.g., from acid rain or in acid mine drainage).

Aluminum, like other metals, generally acts as a surface active toxicant, exerting its damage by binding to anionic sites on respiratory surfaces of aquatic animals, such as a fish gill (Wood & McDonald, 1987). The physiological manifestation of these deleterious surface effects at the gill include both ionoregulatory and respiratory effects. Ionoregulatory effects of aluminum predominate at low pH (e.g., less than pH 5.0) and include a mechanism similar to hydrogen ion toxicity alone, i.e., sodium uptake blockade (Playle & Wood, 1989). In moderately acidic water, it is generally the respiratory effects of aluminum that predominate. Respiratory effects are likely the result of the physical coating of the gills which occurs when aluminum-rich water passes into the more basic gill microenvironment (Robert W. Gensemer & Playle, 1999). Overall, the acute and chronic aluminum toxicity to fish species is substantially greater at low pH, particularly for

[13] Category 2: Available data and information indicate that some designated uses are supported, and the water quality standard is attained.

salmonids. 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, Cleveland, Little, and Brumbaugh (1995); Delonay et al. (1993)). Aluminum at extreme low pH (pH <5) can protect against the direct toxic effects of hydrogen ion. Several factors ameliorate aluminum toxicity at low pH, including, but probably not limited to: calcium ion (D. J. A. Brown, 1983; Ingersoll, Mount, Gulley, Point, & Bergman, 1990), silicic acid (Birchall, Exley, Chappell, & Phillips, 1989), fluoride (Wilkinson, Campbell, & Couture, 2011), and dissolved and natural organic matter (Parkhurst et al., 2011; Roy & Campbell, 1997). Aquatic invertebrates are generally less sensitive to aluminum compared with fish.

Factors such as pH, temperature, and the presence of complexing ions influence the fate and transport of aluminum in the environment. The toxicity of aluminum in ambient water is largely dependent on these various water quality characteristics. 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's and the effect of pH on the derivation of water quality criteria. Additionally, section 5 of this BE assesses the influence of these parameters on the aluminum criteria in Oregon.

4.4.2 Data Sources for Oregon Water Quality

Ambient water quality data were acquired and processed into numerous samples, as described below.

Water Chemistry data were obtained from the following four sources:

a) ODEQ's Ambient Water Quality Monitoring System (AWQMS) web portal (https://orwater.deq.state.or.us/Login.aspx);

b) ODEQ's Laboratory Analytical Storage and Retrieval (LASAR) database;

c) Washington Department of Ecology's (WDoE) Environmental Information Management System (EIM, https://fortress.wa.gov/ecy/eimreporting/Default.aspx); and

d) US Geological Survey's (USGS) National Water Information System (NWIS, https://waterdata.usgs.gov/nwis/qwdata).

4.4.2.1 pH

Ambient water quality monitoring data provided by US Geological Survey and Oregon DEQ for the period of 2000–2017 (USEPA, 2019a, 2019b) indicated an average pH for all stations (19,274 samples from 1554 statewide stations) in all ecoregions of 7.7. The highest mean pH values occurred at Coast Range Ecoregion sites (8.4) and Snake River Ecoregion sites (8.4). These two ecoregions also produced the lowest individual pH value (5.6 for both ecoregions). The highest individual pH values occurred within the Eastern Cascade Slope Ecoregion (10.3) and the Columbia Plateau (10.1). Table 4-3 and Figure 4-4 provide a summary of the ranges of all pH data acquired for this BE.

Table 4-3. pH data summary for Oregon Level III Ecoregions – ODEQ.

Ecoregion	N	# of Stations	Mean	Minimum	Maximum
Coast Range (1)	4768	326	8.4	5.6	9.0
Willamette Valley (3)	6325	357	7.4	6.2	9.4
Cascades (4)	1050	211	7.7	6.5	9.4
Eastern Cascade Slope (9)	1830	207	7.9	6.5	10.3
Columbia Plateau (10)	762	45	8.3	7.1	10.1
Blue Mountains (11)	2130	223	8.2	6.9	9.9
Snake River (12)	528	21	8.4	5.6	9.0
Klamath Mountains (78)	1438	107	8.0	5.9	9.7
Northern Basin and Range (80)	443	57	8.2	7.0	9.2
Oregon	19274	1554	7.7	5.6	10.3

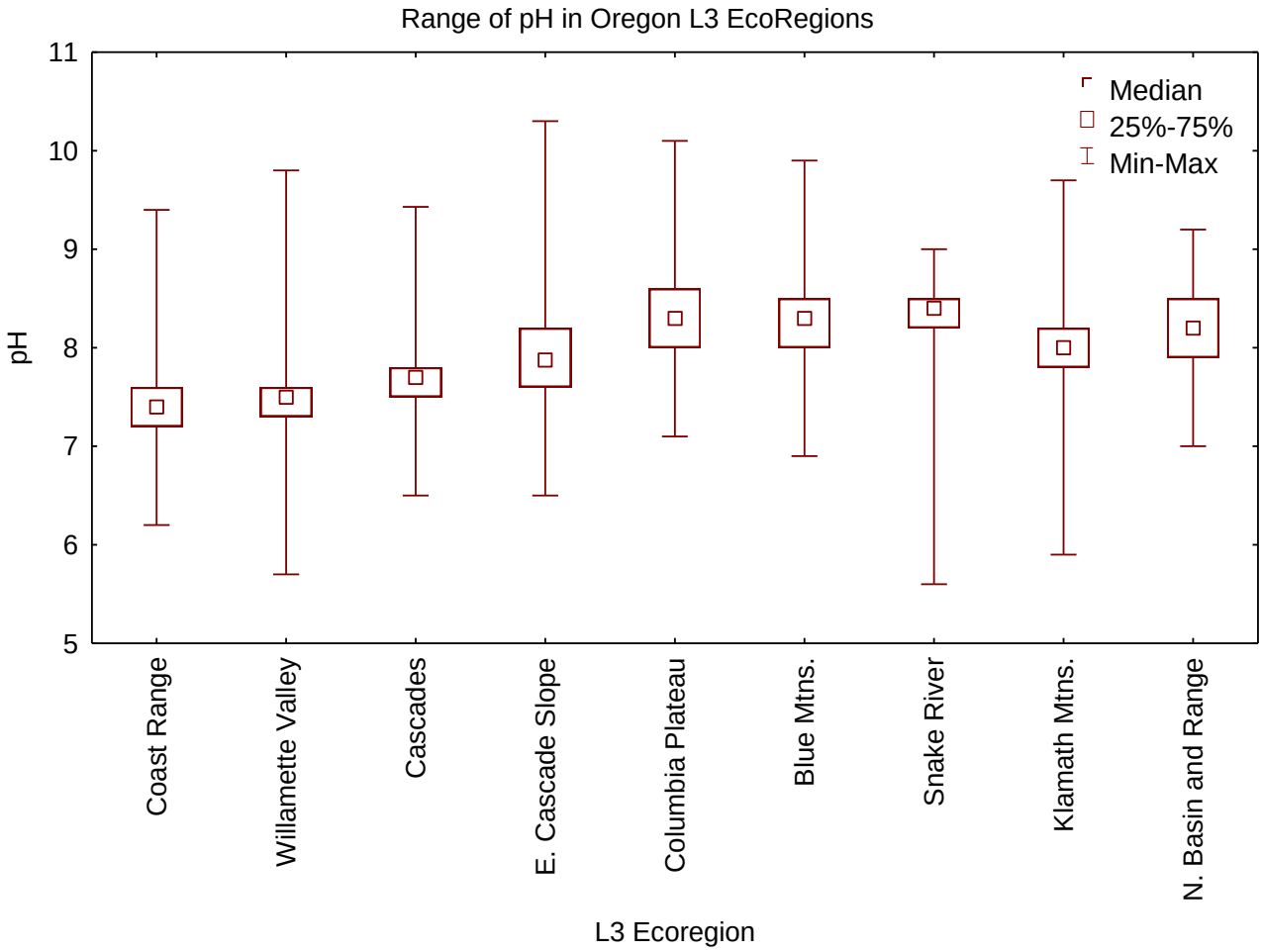


Figure 4-4. Summary of pH values measured in Oregon Level III Ecoregions by ODEQ from 2000 – 2017 (n=19,274).

4.4.2.2 DOC

Ambient water quality monitoring data provided by U.S. Geological Survey and Oregon DEQ for the period of 2000 – 2017 (USEPA, 2019a, 2019b) indicated an average DOC concentration for all stations in all ecoregions of 2.8 mg/L. The highest mean DOC concentrations were found at Snake River sites (4.4 mg/L) and Eastern Cascade Slope sites (4.1 mg/L). Minimum DOC values for all samples in all ecoregions were within the range of 0.1 – 0.8 mg/L. Maximum DOC values for all samples in all were ecoregions were in the range of 14.9 (Cascades) – 132.0 mg/L (Willamette Valley). Table 4-4 and Figure 4-5 provide a summary of the ranges of all DOC data used in this analysis.

Table 4-4. DOC (mg/L) data summary for Oregon Level III Ecoregions.

Ecoregion	N	# of Stations	Mean	Minimum	Maximum
Coast Range (1)	4768	326	1.8	0.2	28.2
Willamette Valley (3)	6325	357	2.5	0.2	132.0
Cascades (4)	1050	211	1.1	0.1	14.9
Eastern Cascade Slope (9)	1830	207	4.1	0.3	50.0
Columbia Plateau (10)	762	45	2.7	0.8	28.2
Blue Mountains (11)	2130	223	2.8	0.3	36.5
Snake River (12)	528	21	4.4	0.8	46.5
Klamath Mountains (78)	1438	107	2.2	0.6	18.3
Northern Basin and Range (80)	443	57	3.3	0.5	28.2
Oregon	19274	1554	2.5	0.1	132.0

4.4.2.3 Hardness

Ambient water quality monitoring data provided by USGS and Oregon DEQ for the period of 2000 – 2017 (USEPA, 2019a, 2019b) indicated hardness values for all stations in all ecoregions in the range of 19.0 (Cascades) to 210.2 (Snake River). The highest individual hardness value was found in the Klamath Mountain Ecoregion (589.0 mg/L). The lowest value was found in the Willamette Valley Ecoregion (0.05 mg/L). Table 4-5 and Figure 4-6 provide a summary of the ranges of all hardness data used in this analysis.

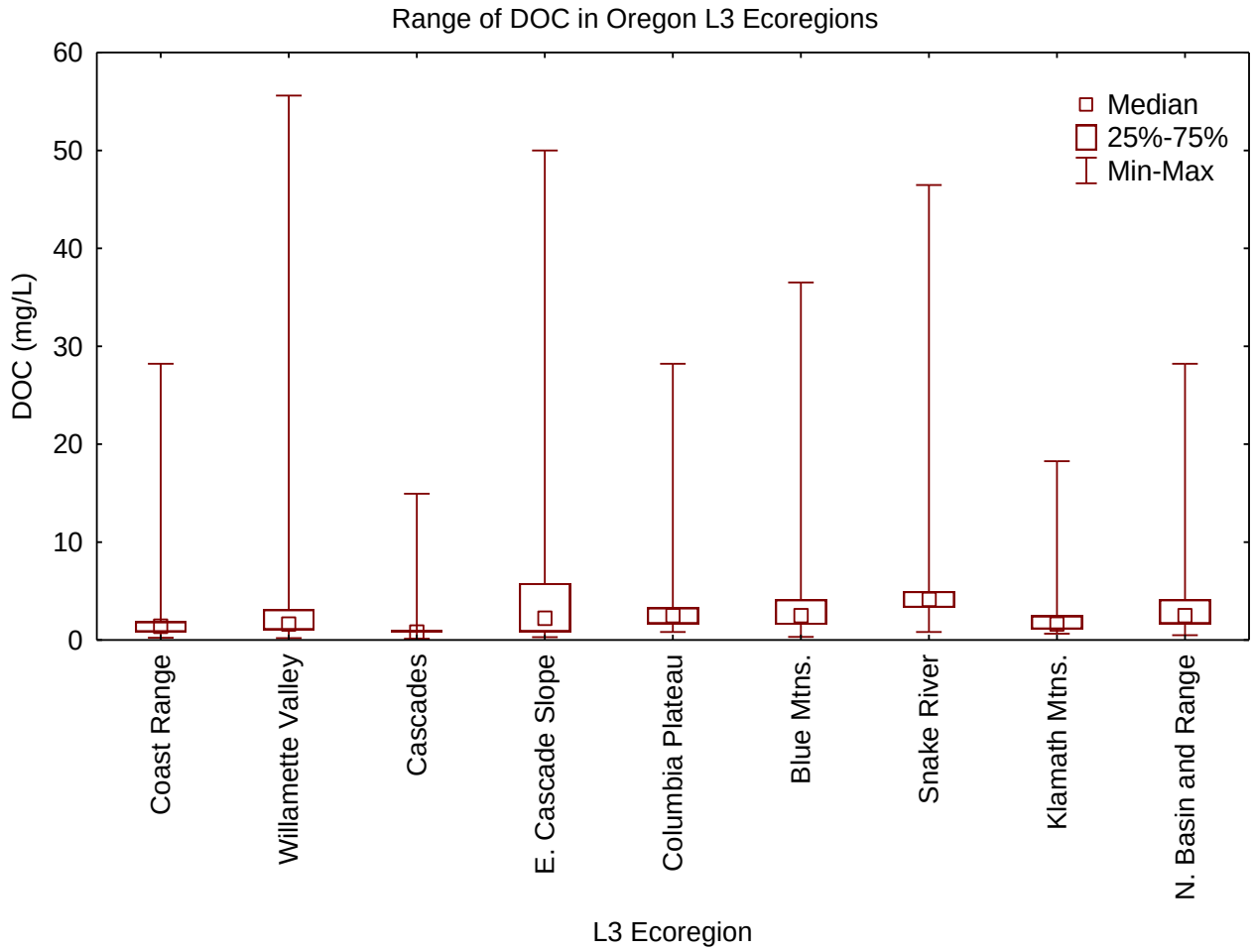


Figure 4-5. Summary of DOC concentrations measured in Oregon Level III Ecoregions by ODEQ from 2000 – 2017. Two additional DOC concentrations (79.4 and 132 mg/L) reported for Willamette Valley were removed from the analysis for display.

Table 4-5. Hardness data (n=19274) summary for Oregon Level III Ecoregions.

Ecoregion	N	# of Stations	Mean	Minimum	Maximum
Coast Range (1)	4768	326	31.8	2.4	413.7
Willamette Valley (3)	6325	357	39.3	0.05	235.0
Cascades (4)	1050	211	19.0	6.0	216.0
Eastern Cascade Slope (9)	1830	207	44.3	10.1	461.0
Columbia Plateau (10)	762	45	61.6	15.8	265.0
Blue Mountains (11)	2130	223	70.7	8.7	343.0
Snake River (12)	528	21	210.2	46.9	385.4
Klamath Mountains (78)	1438	107	44.6	15.4	589.0
Northern Basin and Range (80)	443	57	67.9	22.2	200.5
Oregon	19274	1554	46.9	0.05	589.0

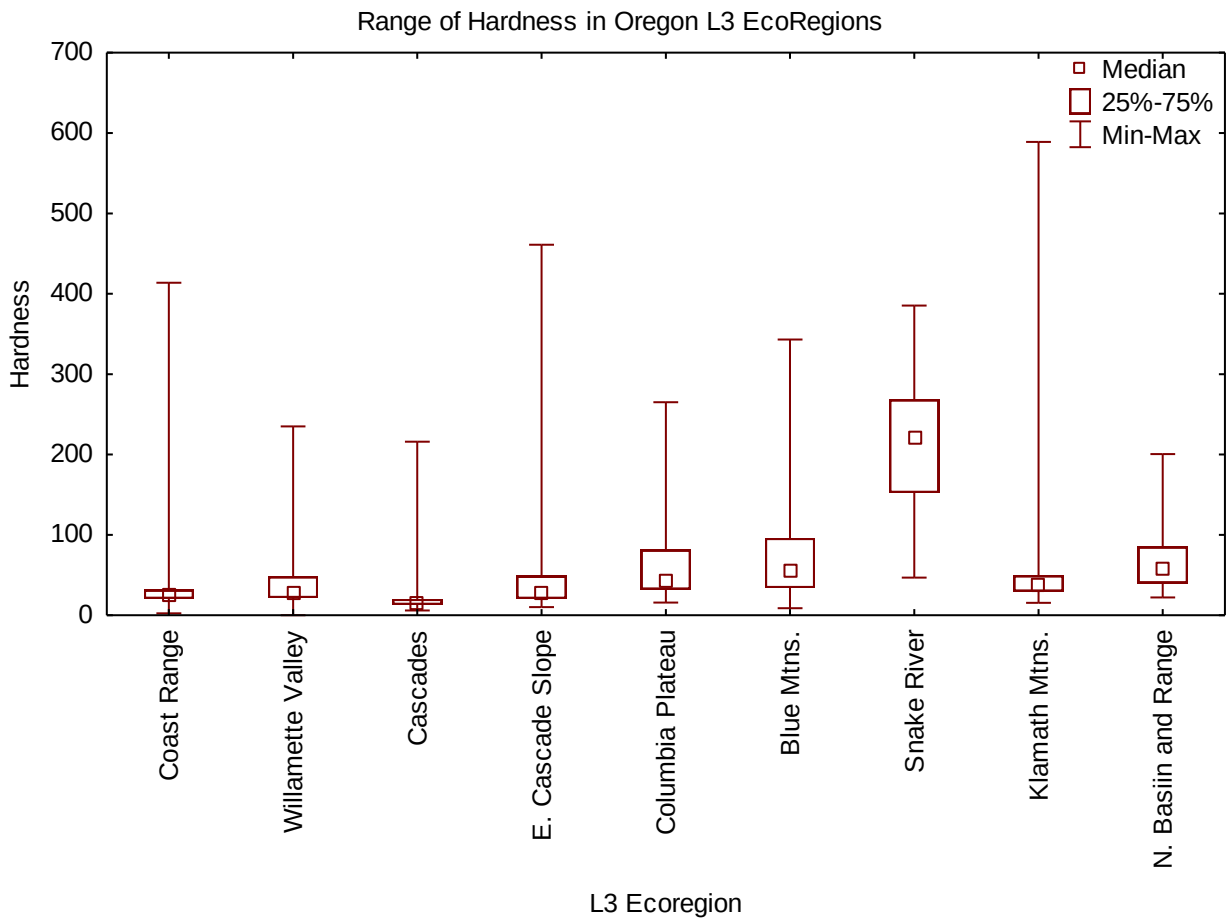


Figure 4-6. Summary of hardness concentrations measured in Oregon Level III Ecoregions by ODEQ from 2000 – 2017.

4.4.2.4 Aluminum

Monitoring data provided by USGS and ODEQ (2006–2017) measuring total recoverable and dissolved aluminum from sites within all Level III Ecoregions is provided in Table 4-6 and Figures 4-7 and 4-8. Results indicated a wide range of concentrations both within and among Ecoregions. The highest mean concentrations of both total recoverable and dissolved aluminum occurred in the Snake River Ecoregion (6,741 and 167.0 µg/L, respectively). The lowest mean concentrations of both total recoverable and dissolved aluminum occurred in the Cascades Ecoregion (58.8 and 17.0 µg/L, respectively).

The highest individual total recoverable Al concentrations came from the Willamette Valley (10,600 µg/L) and Snake River Ecoregions (67,000 µg/L).

Table 4-6. Summary of aluminum concentration (ug/L) data for Oregon Level III Ecoregions.

Ecoregion	# of Stations with Al Data	Fraction	N	Mean	Minimum	Maximum	Std. Dev.
Cascades (4)	6	Total Recoverable	26	58.8	8.2	379.0	79.9
		Dissolved	23	17.0	<0.05	61.6	15.0
Coast Range (1)	49	Total Recoverable	135	289.2	<0.01	3520.0	478.8
		Dissolved	335	9.07	<0.01	110.0	17.5
Blue Mountains (11)	25	Total Recoverable	121	552.1	14.5	9,340.0	1212.6
		Dissolved	235	21.1	<0.05	366	41.0
Klamath Mountains (78)	23	Total Recoverable	95	392.7	<0.05	2,110.0	495.3
		Dissolved	101	24.5	<0.05	119.0	22.7
Columbia Plateau (10)	12	Total Recoverable	54	489.9	<0.05	2340.0	656.4
		Dissolved	86	25.5	<0.05	238.0	40.1
Willamette Valley (3)	104	Total Recoverable	231	610.9	<0.05	10,600.0	1194.4
		Dissolved	282	34.9	<0.05	557.0	73.4
Eastern Cascade Slope (9)	18	Total Recoverable	86	439.0	<0.05	5,930.0	798.1
		Dissolved	97	30.3	<0.05	220	36.9
Northern Basin and Range (80)	17	Total Recoverable	59	693.2	28.1	2,480.0	611.8
		Dissolved	81	165.3	<0.05	2390.0	417.7
Snake River (12)	6	Total Recoverable	35	6,743.1	55.4	67,000.0	13,002.3
		Dissolved	67	167.0	<0.05	1250.0	279.7
Oregon	260	Total Recoverable	842	744.6	<0.01	67,000.0	3033.1
		Dissolved	1307	38.6	<0.01	2390.0	136.3

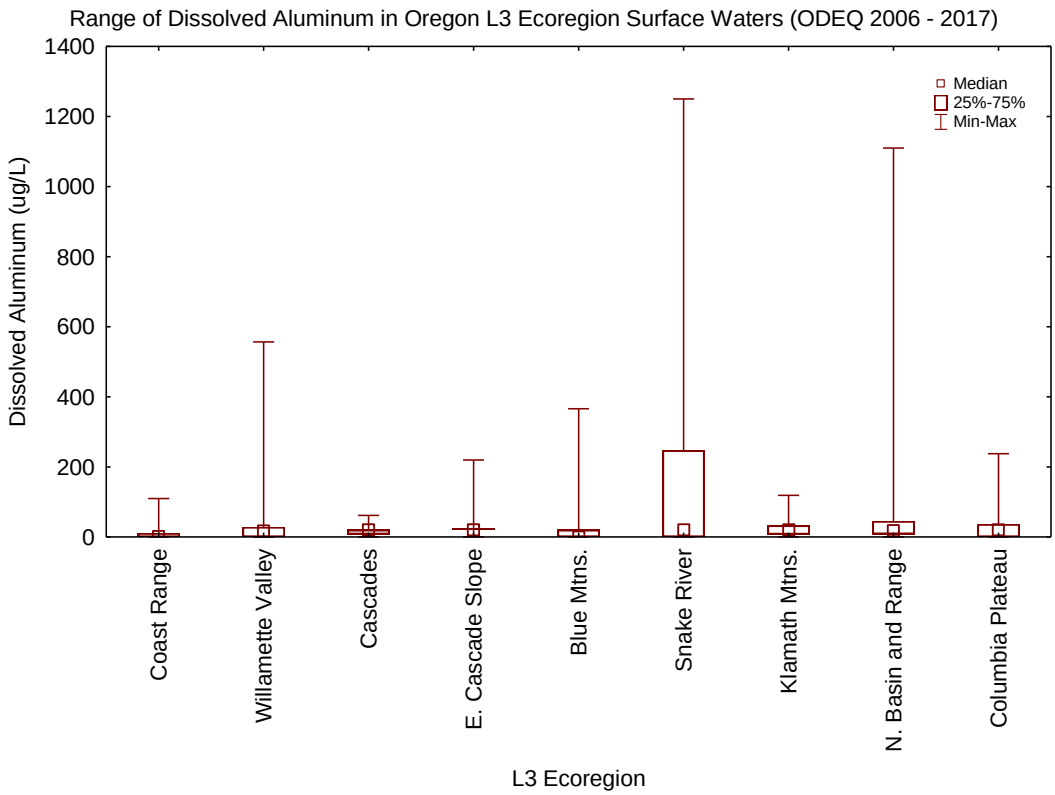


Figure 4-7. Summary of dissolved aluminum measured in Oregon Level III Ecoregions from 2006 – 2017. Three additional dissolved aluminum concentrations (1640, 1720, and 2390 µg/L) reported for Ecoregion 80 were removed from the analysis for display.

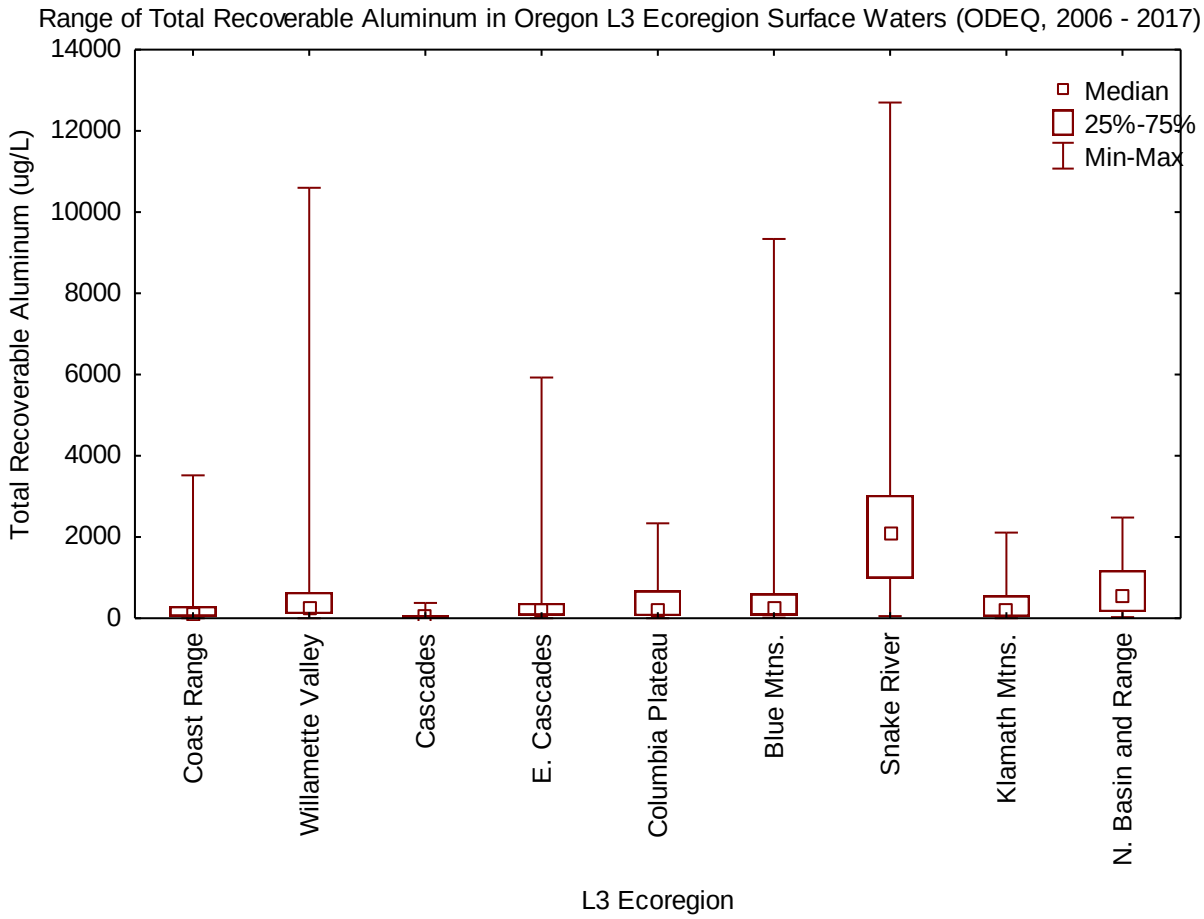


Figure 4-8. Summary of total recoverable aluminum measured in Oregon Level III Ecoregions from 2006 – 2017. Four additional total recoverable aluminum concentrations (67,000; 34,000; 28,200; and 19,600 µg/L) reported for Ecoregion 12 were removed from the analysis

4.5 Land Uses Affecting Aluminum Concentrations

As noted in the previous sections, the concentration of aluminum in surface waterbodies can be impacted by non-point sources correlated with land-use types including agriculture, forest, and developed urban areas. The land covers in each EPA Level 3 Ecoregion are summarized in Table 4-7 and reviewed in the following subsections with respect to their potential as a source of aluminum relevant to ESA-Listed species (Table 4-8).

Table 4-7. Summary of land uses by Level III Ecoregion in Oregon (USEPA, 2013).

Ecoregion Level III	Ecoregion Level III Name	Total Stream/Coastline Miles	% of state waters	Open Water	Perennial Snow/Ice	Developed Open Space	Developed Low Intensity	Developed Medium Intensity	Developed High Intensity	Barren Land	Deciduous Forest	Evergreen Forest	Mixed Forest	Shrub/Scrub	Herbaceous	Hay/Pasture	Cultivated Crops	Woody Wetland	Emergent Herbaceous Wetland
1	Coast Range	10583.1	12.7%	0.5%	0.0%	5.1%	0.6%	0.2%	0.1%	1.2%	1.7%	43.4%	18.5%	18.8%	5.6%	1.4%	0.2%	1.5%	1.2%
3	Willamette Valley	6378	7.6%	1.3%	0.0%	4.5%	6.5%	3.9%	1.5%	0.4%	1.1%	14.7%	6.1%	8.5%	4.4%	26.3%	17.2%	2.9%	0.8%
4	Cascades	11166.4	13.3%	0.9%	0.1%	0.5%	0.1%	0.0%	0.0%	1.6%	0.1%	78.6%	0.6%	13.5%	3.4%	0.2%	0.2%	0.0%	0.1%
9	Eastern Cascades Slopes and Foothills	8448.9	10.1%	2.0%	0.0%	1.0%	0.4%	0.1%	0.0%	0.9%	0.1%	53.0%	0.1%	24.9%	9.1%	2.2%	3.7%	0.1%	2.4%
10	Columbia Plateau	7703.7	9.2%	0.3%	0.0%	2.4%	0.5%	0.2%	0.0%	0.2%	0.0%	1.2%	0.0%	56.5%	3.9%	1.1%	33.2%	0.1%	0.3%
11	Blue Mountains	7703.7	9.2%	0.2%	0.0%	0.8%	0.3%	0.1%	0.0%	0.4%	0.0%	37.6%	0.0%	50.2%	6.0%	0.4%	3.3%	0.1%	0.6%
12	Snake River Plain	1164.8	1.4%	0.6%	0.0%	3.2%	0.9%	0.3%	0.0%	0.0%	0.0%	0.0%	0.0%	23.9%	38.4%	3.9%	28.4%	0.0%	0.2%
78	Klamath Mountains	6131.3	7.3%	0.3%	0.0%	3.2%	1.1%	0.5%	0.2%	0.4%	0.3%	56.3%	3.4%	20.6%	6.0%	6.1%	0.9%	0.6%	0.2%
80	Northern Basin and Range	24375.8	29.1%	1.1%	0.0%	0.4%	0.1%	0.0%	0.0%	3.1%	0.0%	0.8%	0.0%	86.4%	3.9%	2.0%	0.9%	0.2%	1.0%
Oregon		83655.7	100%	100%	0.0%	1.9%	0.8%	0.4%	0.1%	1.5%	0.4%	30.4%	3.2%	44.2%	5.5%	3.7%	5.8%	0.6%	0.9%

Table 4-8. Summary of the occurrence of threatened and endangered species in Oregon by Level III Ecoregion and Oregon Euregion.

Species		Status	Georegion									
			Coastal		Willamette Valley	Cascades	Eastern					Columbia River
			Ecoregion									
			1	78	3	4	9	10	11	12	80	NA
			Coast Range	Klamath Mountains	Willamette Valley	Cascades	Eastern Cascades	Columbia Plateau	Blue Mountains	Snake River Plain	Northern Basin & Range	NA
Green Sturgeon	Southern DPS	T	x									
Eulachon	Southern DPS	T	x		x	x	x					x
Coho salmon	Lower Columbia River	T	x		x	x	x					x
	Oregon Coast	T	x	x								
	Southern Oregon/ Northern California Coast*	T	x	x		x						
Chum salmon	Columbia River	T	x		x	x	x	x				x
Chinook salmon	Upper Columbia River spring	E	x		x	x	x	x				x
	Snake River spring/summer	T	x		x	x	x	x	x			x
	Snake River fall	T	x		x	x	x	x	x			x
	Upper Willamette River	T	x		x	x						x
	Lower Columbia	T	x		x	x	x					x
Sockeye salmon	Snake River	E	x		x	x	x	x	x			x
Steelhead	Upper Columbia	T	x		x	x	x	x				x
	Snake River Basin	T	x		x	x	x	x	x			x
	Middle Columbia	T	x		x	x	x	x	x			x
	Upper Willamette River	T	x		x	x						x
	Lower Columbia	T	x		x	x	x					x

Species	Status	Georegion									
		Coastal		Willamette Valley	Cascades	Eastern					Columbia River
		Ecoregion									
		1	78	3	4	9	10	11	12	80	NA
		Coast Range	Klamath Mountains	Willamette Valley	Cascades	Eastern Cascades	Columbia Plateau	Blue Mountains	Snake River Plain	Northern Basin & Range	NA
Oregon spotted frog	T				x	x					
Warner sucker	T									x	
Shortnose sucker	E					x					
Lost River sucker	E					x					
Hutton tui chub	T					x				x	
Borax Lake chub	E									x	
Lahontan cutthroat trout	T									x	
Foskett speckled dace	T									x	
Bull trout	T	x		x	x	x	x	x	x	x	x
Vernal pool fairy shrimp	T		x								

Table 4-9 shows an amalgamation of similar land covers (e.g., "Developed Lands" included Developed Open Spaces, Developed Low Intensity, Developed Medium Intensity, Developed High Intensity; "Forest" included Deciduous, Evergreen, Mixed, Shrub/Scrub, and Herbaceous) to determine if certain land covers may be linked to higher Al in surface water across ecoregions. However, it appears Al is ubiquitous throughout the landscape such that no clear pattern can be discerned. Areas like the Coast Range and Northern Basin have a high percentage of forested land cover, 88 and 91.1%, respectively, but mean dissolved Al is low (9.07 mg/L) for the Coast Range and high (165.3 mg/L) for the Northern Basin. A similar trend for agriculture land cover can be observed within ecoregions as demonstrated by the Columbia Plateau and Snake River each with similar agriculture land cover (34.4% and 32.3%), yet the Al concentrations are significantly different, 25.5 and 167 mg/L, respectively.

Table 4-9. Summary of ecoregion land cover and surface water dissolved and total aluminum concentrations.

Ecoregion Level III	Ecoregion Level III Name	Developed Lands[1]	Wetlands and Open Water[2]	Agriculture[3]	Perennial Snow/Ice & Barren Land[4]	Forest[5]	Mean Dissolved Aluminum (mg/L)	Total Recoverable Aluminum (mg/L)	Total : Dissolved Ratio
1	Coast Range	6.0%	3.1%	1.6%	1.2%	88.0%	9.07	289.2	31.9
3	Willamette Valley	16.5%	4.9%	43.4%	0.4%	34.8%	34.9	610.9	17.5
4	Cascades	0.6%	1.0%	0.4%	1.7%	96.2%	17	58.8	3.5
9	Eastern Cascades Slopes and Foothills	1.5%	4.5%	5.9%	0.9%	87.2%	30.3	439	14.5
10	Columbia Plateau	3.1%	0.7%	34.4%	0.2%	61.6%	25.5	489.9	19.2
11	Blue Mountains	1.2%	0.9%	3.7%	0.4%	93.8%	21.1	552.1	26.2
12	Snake River Plain	4.5%	0.8%	32.3%	0.0%	62.4%	167	6,743.1	40.4
78	Klamath Mountains	5.0%	1.1%	7.0%	0.4%	86.6%	24.5	392.7	16.0
80	Northern Basin and Range	0.6%	2.4%	2.9%	3.1%	91.1%	165.3	693.2	4.2
Oregon		3.2%	2.3%	9.5%	1.5%	83.5%	38.6	744.6	19.3

1 – includes Developed Open Spaces, Developed Low Intensity, Developed Medium Intensity, and Developed High Intensity.
2 – Includes Open Water, Woody Wetland, and Emergent Herbaceous Wetland
3 – Includes Hay/Pasture and Cultivated Crops
4 – Includes Perennial Snow/Ice and Barren Land
5 – Includes Deciduous Forest, Evergreen Forest, Mixed Forest, Shrub/Scrub, and Herbaceous

Except for the Northern Basin and Range and the Eastern Cascades, higher mean dissolved aluminum may be correlated with higher percentage of agricultural land use including hay/pasture and cultivated crops. These land uses have been correlated with Al concentrations and therefore may present a higher potential for exposing organisms to Al in the future.

4.5.1 Land Use Surrounding Each Surface Water Use Location

Due to the lack of distinct correlations between the five land cover categories and the mean total recoverable or dissolved aluminum concentration across the EPA Level III Ecoregions, individual site-specific land covers were quantified. Land covers within a ¼ mile buffer to the sites, where surface water aluminum concentrations were collected, were calculated to determine if there was a correlation between land cover and surface water aluminum concentration. Non-detects for aluminum were removed from the analysis due to variability in the reporting limit (50 – 2500 µg/L).

A correlation analysis for total recoverable and dissolved aluminum and the percentage of each of the five land cover categories was conducted. The Pearson r coefficient, also called linear or product-moment correlation, assumes that the two variables are measured on at least interval scales and determines the extent to which values of the two variables are "proportional" to each other. Proportional means linearly related; that is, the correlation is high if it can be "summarized" by a straight line (sloped upwards or downwards).

For dissolved aluminum across the entire state, only forested land cover was significantly positively correlated ($p < 0.05$) with surface water dissolved aluminum: as forested land cover percentage increased so did the measured surface water dissolved aluminum (Table 4-10 and Figure 4-9). Based on individual ecoregions, both the Coast Range and Northern Basin Al concentrations, were significantly positively correlated ($p < 0.05$) to percentage of forested land cover (Table 4-10, Figures 4-10 and 4-11). For all other ecoregions the % forested land cover was not significantly correlated ($p > 0.05$) to measured dissolved aluminum, but a trend of increasing aluminum with increased forested land cover occurred in 7 of 9 ecoregions and for the state overall. Across Oregon, the measured surface water concentration of aluminum was significantly negatively correlated ($p < 0.05$) with the percentage of developed lands and wetlands/open water (Table 4-10). Therefore, as the percentage of these land covers increase the measured dissolved aluminum concentration decreases. Individual ecoregions, including the Blue Mountains, Cascades, and Klamath Mountains, also noted this trend for developed land cover (Table 4-10). Although not significantly correlated ($p > 0.05$), the decreasing trend was noted in four of the other six ecoregions. Agriculture was significantly negatively correlated ($p < 0.05$) to dissolved aluminum in the Coast Range (Table 4-10) but only indicated a decreasing trend in two of the other eight ecoregions. Barren land also indicated a decreasing trend in aluminum concentration in four of six ecoregions (three ecoregions including the Columbia Plateau, Northern Basin, and Snake River did not have barren land in any of the site-specific land uses) (Table 4-10).

Table 4-10. Summary of correlations (Pearson r correlation coefficient) between land covers and surface water dissolved aluminum concentrations. Green shade indicates a statistically significant (p < 0.05) negative correlation and red shading indicates a statistically significant (p < 0.05) positive correlation between land cover % and dissolved Al concentration.

Variable	All Groups Correlations (Spreadsheet280) Marked correlations are significant at p < .05000 N=554 (Casewise deletion of missing data) Include condition: v2='Aluminum, dissolved' Exclude condition: v4='U'('U' quality assurance code)									
	Whole State	**Blue Mtns.**	**Cascades**	**Coast Range**	**Columbia Plateau**	**Eastern Cascades**	**Klamath Mtns.**	**Northern Basin**	**Snake River**	**Willamette Valley**
N	554	66	11	95	31	36	53	55	30	177
Developed Lands	-0.110722	-0.251297	-0.927902	-0.064850	0.301439	-0.097148	-0.333317	-0.146730	0.335479	-0.047886
Wetlands and Open Water	-0.132588	-0.009400	-0.370824	0.033877	-0.307746	-0.135289	-0.140062	-0.231228	-0.082446	-0.031930
Agriculture	0.030556	0.133056	0.211417	-0.243811	-0.208408	-0.074755	0.142176	-0.242972	0.011453	0.066007
Barren Lands	-0.045681	-0.111971	-0.369762	-0.078889		0.285638	0.081438			-0.119226
Forest	0.142462	0.057976	0.596891	0.221602	-0.145519	0.070968	0.200200	0.358691	-0.143250	0.020578

Table 4-11. Summary of correlations (Pearson r correlation coefficient) between land covers and surface water total recoverable aluminum concentrations. Green shade indicates a statistically significant (p < 0.05) negative correlation and red shading indicates a statistically significant (p < 0.05) positive correlation between land cover % and total recoverable Al concentration.

Variable	All Groups Correlations (Spreadsheet280) Marked correlations are significant at p < .05000 N=811 (Casewise deletion of missing data) Include condition: v2='Aluminum, total recoverable' Exclude condition: v4='U' ('U' quality assurance code)									
	Whole State	Blue Mtns.	Cascades	Coast Range	Columbia Plateau	Eastern Cascades	Klamath Mtns.	Northern Basin	Snake River	Willamette Valley
N	811	121	25	128	52	79	88	55	35	228
Developed Lands	-0.041913	0.079663	0.110520	0.021348	0.377296	-0.055065	0.142192	0.004240	-0.219655	-0.037770
Wetlands and Open Water	-0.061208	-0.140300	0.088569	0.046806	-0.004968	-0.100660	-0.049660	-0.216824	-0.204321	-0.087547
Agriculture	0.200094	0.022177	0.301012	0.050013	-0.370427	0.011304	0.159779	0.119976	0.536257	0.135815
Barren Lands	-0.022903	-0.091978	0.121653	-0.044518		0.272527	-0.157476			-0.086157
Forest	-0.099571	0.010516	-0.202059	-0.066891	-0.215572	-0.007434	-0.206830	0.047949	-0.203127	-0.033426

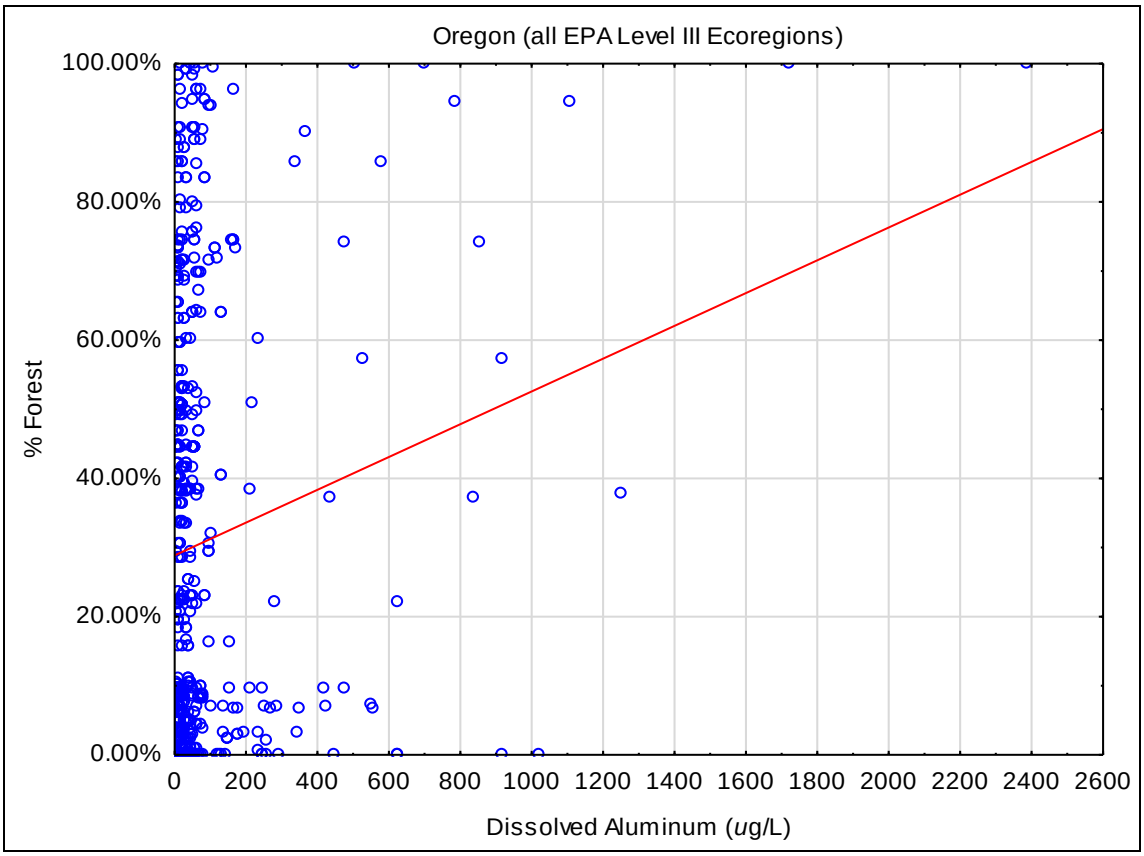


Figure 4-9. Percentage of forested land cover correlated (r = 0.142; p < 0.05) to dissolved aluminum surface water concentrations in the whole state of Oregon.

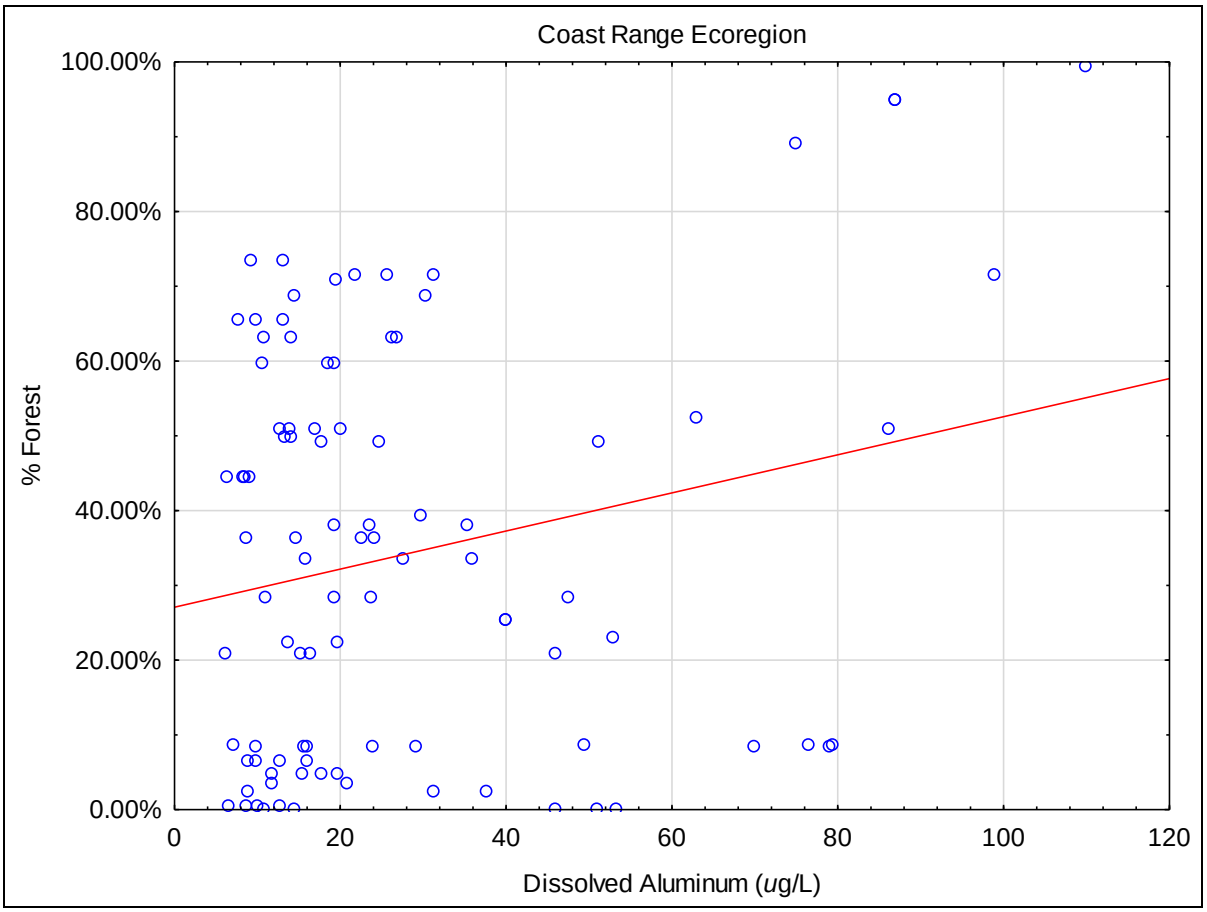


Figure 4-10. Percentage of forested land cover correlated (r = 0.222; p < 0.05) to dissolved aluminum surface water concentrations in the Coast Range ecoregion.

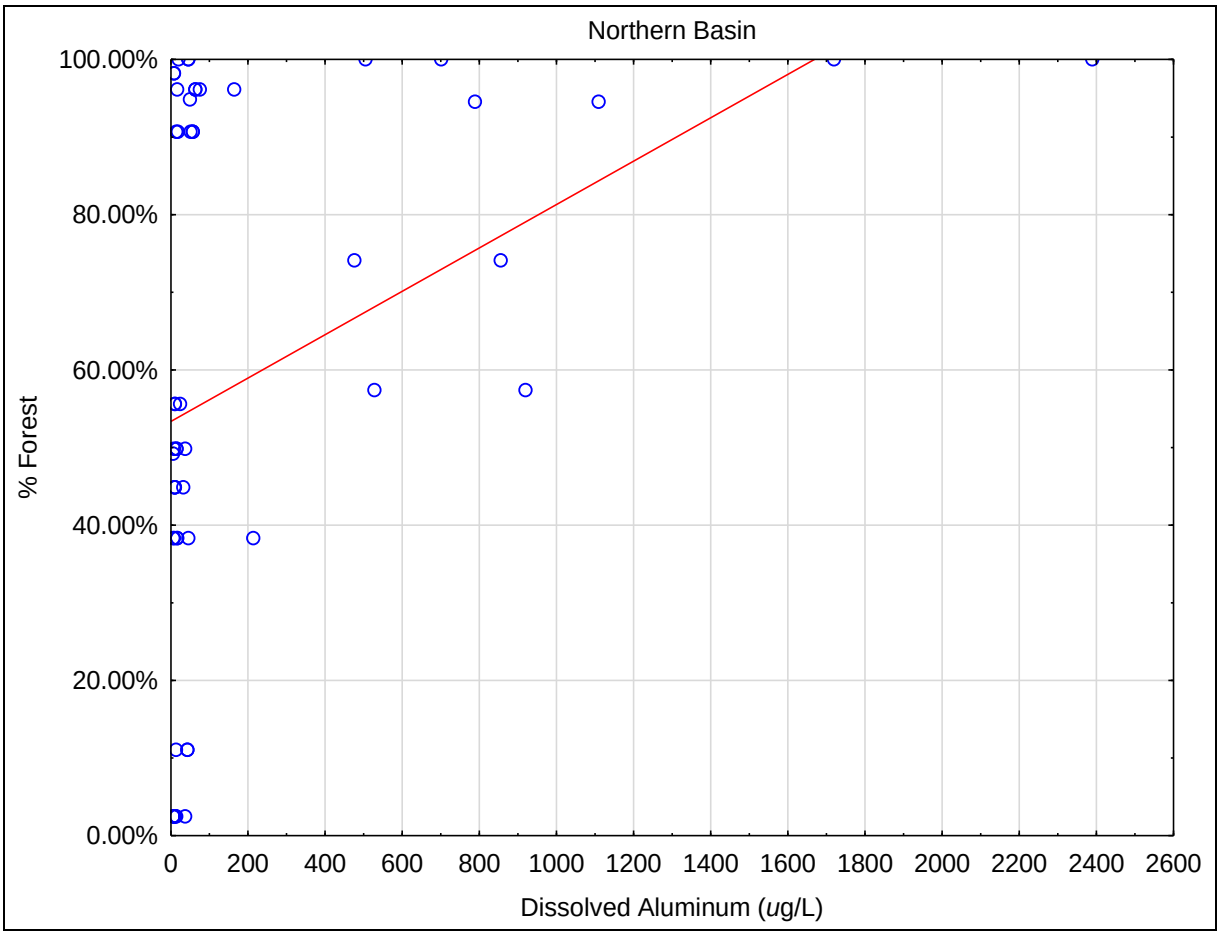


Figure 4-11. Percentage of forested land cover correlated (r = 0.359; p < 0.05) to dissolved aluminum surface water concentrations in the Northern Basin ecoregion.

For total recoverable aluminum across the entire state, agriculture was significantly positively correlated ($p < 0.05$), and forested lands were significantly negatively correlated ($p < 0.05$) (Table 4-11 and Figure 4-12). Based on individual ecoregions, there were no ecoregions with statistically significant positive correlations ($p < 0.05$) between total recoverable Al and the percentage of forested land cover (Table 4-11). Based on individual ecoregions, both the Snake River and Willamette Valley total recoverable Al concentrations were significantly positively correlated ($p < 0.05$) to percentage of agriculture land cover, while the Columbia Plateau was significantly negatively correlated (Table 4-11 and Figures 4-13 and 4-14). For all other ecoregions, the trend of increasing total recoverable aluminum with increased agriculture cover occurred (Table 4-11), but the r values were not statistically significant. For the entire state and three individual ecoregions (i.e., Eastern Cascades, Snake River, and Willamette Valley) there was a trend of increased developed lands with a decrease in total recoverable Al (Table 4-11). However, the other six ecoregions, as developed land percentage increased so too did the total recoverable Al concentrations, including the Columbia Plateau (which was a statistically significant correlation ($p < 0.05$) (Table 4-11 and Figure 4-15). The Eastern Cascades were the only other ecoregion that had a land cover category, barren land, that was positively correlated ($p < 0.05$) with total recoverable aluminum concentrations (Figure 4-16). The trend across the other ecoregions with barren land is that as barren land increases the total recoverable Al concentration decreases (Table 4-11).

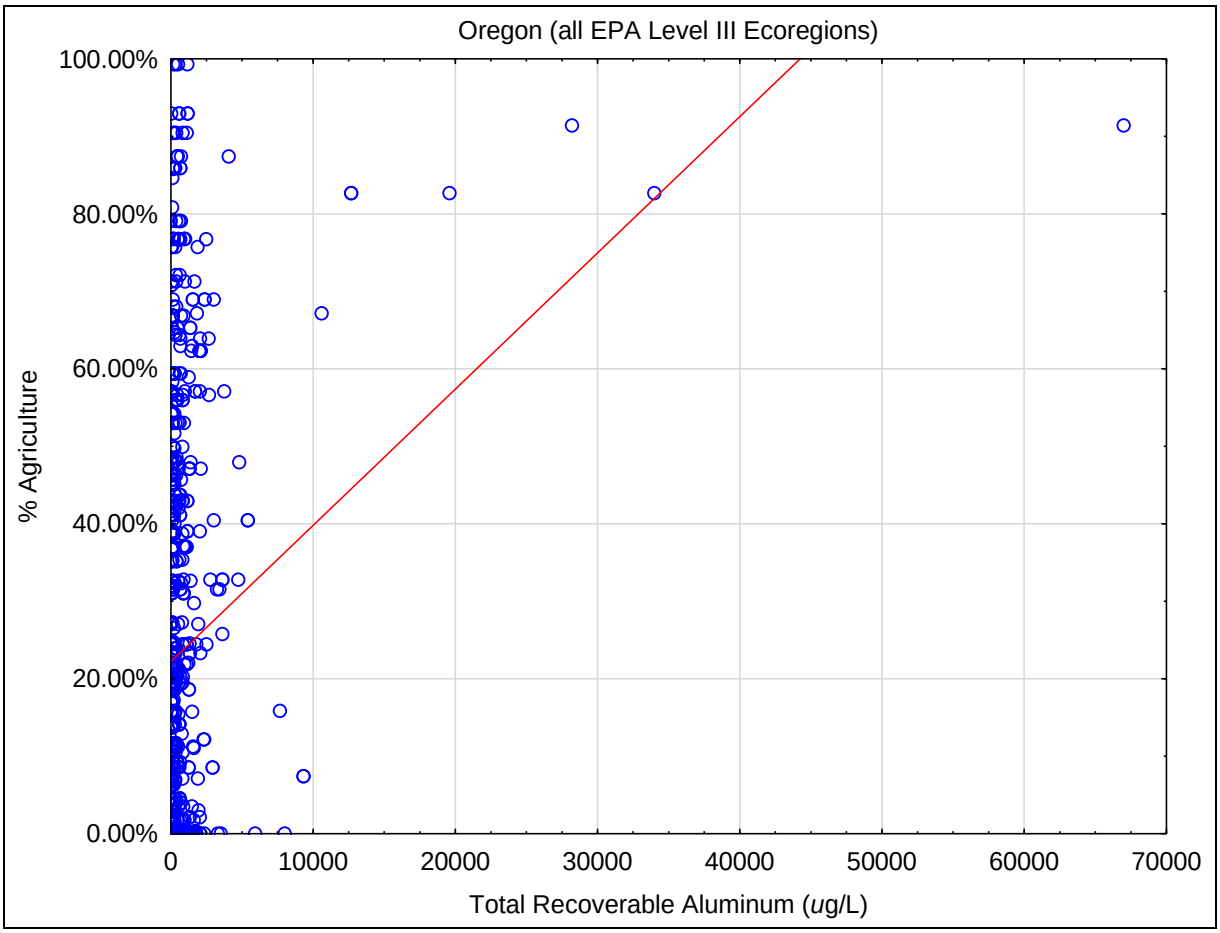


Figure 4-12. Percentage of agriculture land cover correlated (r = 0.200; p < 0.05) to total recoverable aluminum surface water concentrations in the whole state of Oregon.

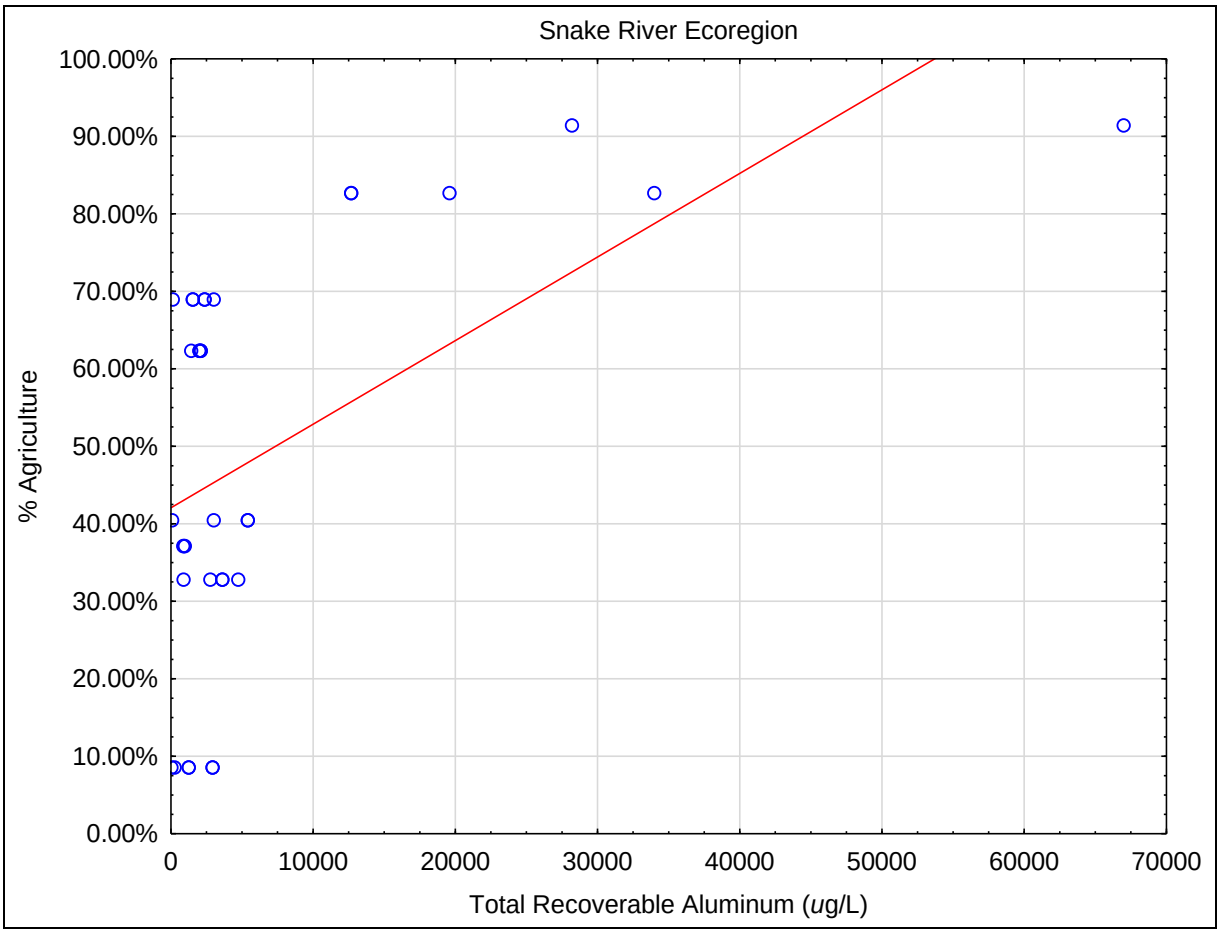


Figure 4-13. Percentage of agriculture land cover correlated ($r = 0.536$; $p < 0.05$) to total recoverable aluminum surface water concentrations in the Snake River ecoregion.

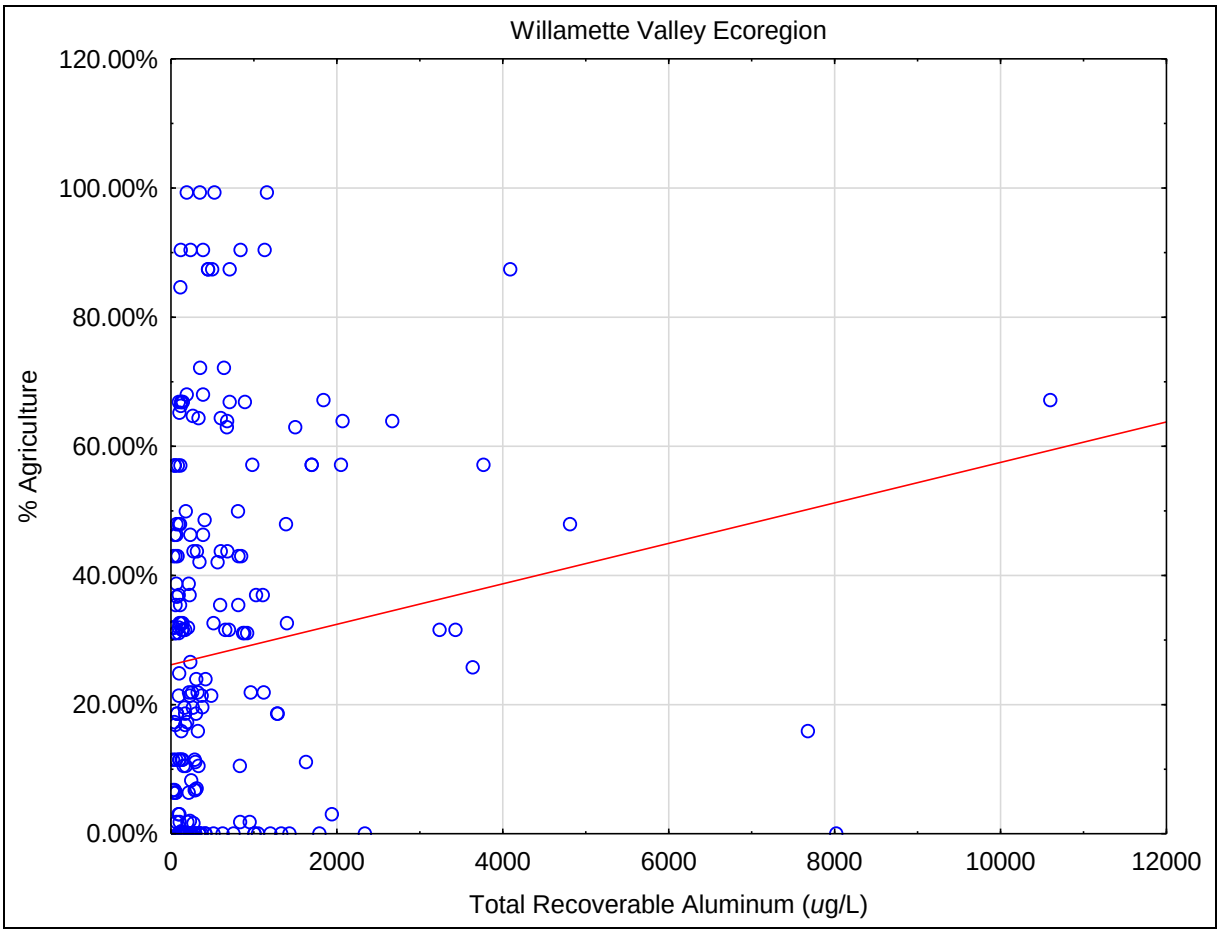


Figure 4-14. Percentage of agriculture land cover correlated (r = 0.136; p < 0.05) to total recoverable aluminum surface water concentrations in the Willamette Valley ecoregion.

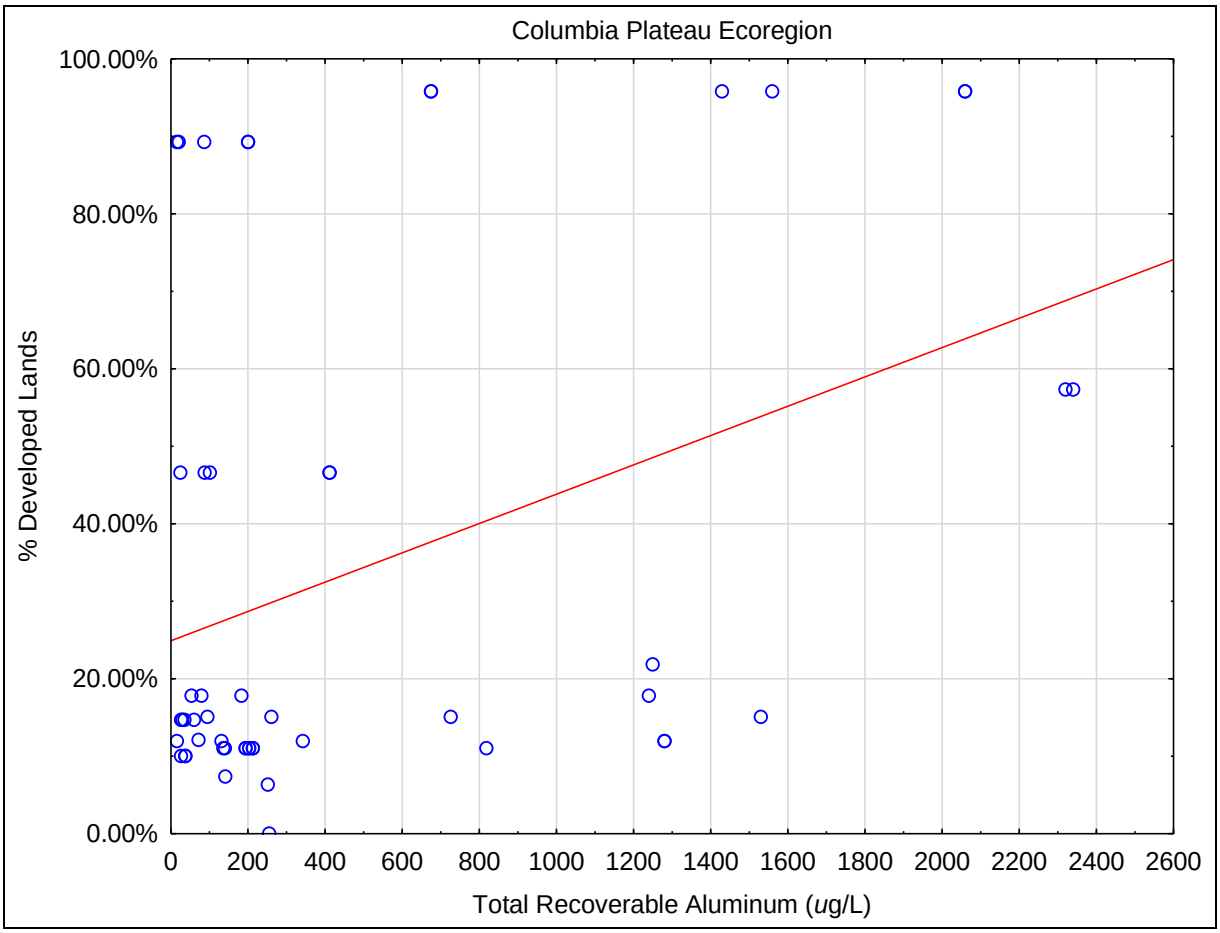


Figure 4-15. Percentage of developed land cover correlated (r = -0.370; p < 0.05) to total recoverable aluminum surface water concentrations in the Columbia Plateau ecoregion.

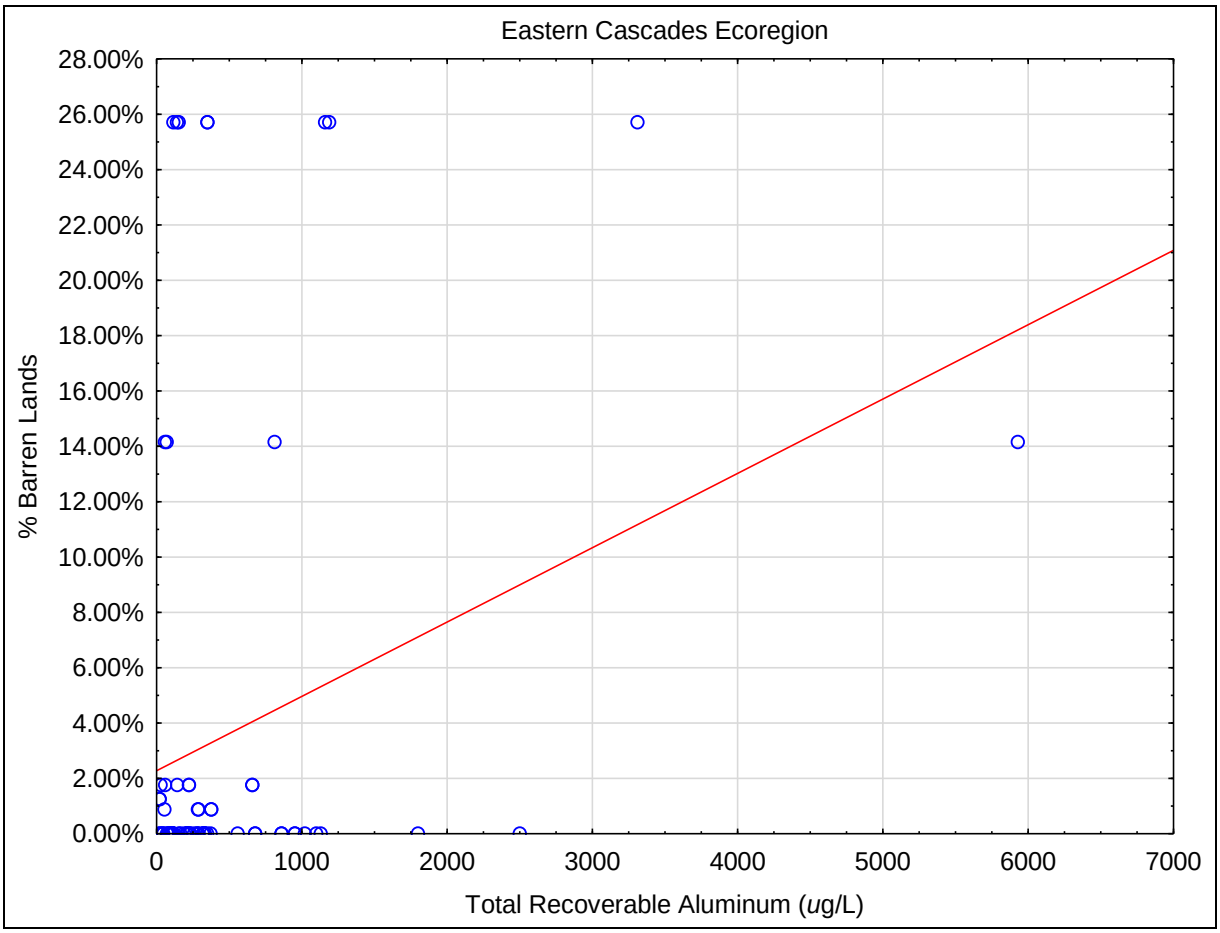


Figure 4-16. Percentage of barren land cover correlated (r = 0.273; p < 0.05) to total recoverable aluminum surface water concentrations in the Eastern Cascades ecoregion.

Overall, for both dissolved and total recoverable Al concentrations, the percentage of agriculture land use was the most consistently positively correlated, indicating as agriculture land use increases so does the concentration of Al in surface waters (Tables 4-10 and 4-11). However, correlations were only statistically significantly positive for total recoverable Al, perhaps indicating higher levels of Al bound to total suspended solids that was not liberated into dissolved Al due to non-acidic conditions.

4.6 Potential Exposure of Aluminum to Threatened and Endangered Species

In the next sections, each of the nine Level III ecoregions are examined in more detail with respect to the potential past exposure of threatened and endangered species to aluminum in surface waters. The threatened and endangered species known to occur in each ecoregion are noted. Potential point sources of aluminum, including municipal WWTPs, industrial facilities, anodizing facilities and drinking water facilities are enumerated and evaluated, as well as the known mineral deposits and those mineral deposits associated with aluminum. The potential non-point sources including the different land covers are summarized with respect to how each was associated with previously measured Al concentrations. The measured water chemistry that is known to affect the toxicity of aluminum in each ecoregion is also summarized.

Figures 4-17 through 4-26 depict StreamNet[14] dervived fish habitat/distribution and critical habitat for threatened and endangered species. The StreamNet Fish Distribution ("Essential Habitat") dataset documents fish distribution and activity for all fish species available from data submitted to the regional StreamNet project by partner agencies. This dataset was compiled from separate spatial datasets contributed by StreamNet partners as of January 31, 2019. Distribution is based upon survey data and, in some cases, the best professional judgement of local fish biologists, in the Pacific Northwest Region. The datasets used here also contain information for Federally Designated Critical Habitat for threatened and endangered species derived by the United States Fish and Wildlife Service (USFWS) and obtained by EPA in October 2016 and March 2017.

4.6.1 Coast Range

There are 8 threatened and endangered species present in the Coast Range ecoregion (see Table 4-8) with some species having multiple ESUs present, including:

- Green Sturgeon – Southern DPS
- Eulachon – Southern DPS
- Coho Salmon – Lower Columbia River, Oregon Coast, Southern OR/Northern CA Coast
- Chum Salmon – Columbia River
- Chinook Salmon – Upper Columbia River Spring Run, Snake River Spring/Summer Run, Snake River Fall Run, Upper Willamette River, Lower Columbia River
- Sockeye Salmon – Snake River
- Steelhead – Upper Columbia River, Snake River, Middle Columbia River, Upper Willamette River, Lower Columbia River
- Bull trout

Figure 4-17 illustrates habitats identified by StreamNet across Oregon where T&E species have occurred including those within the Coast Range ecoregion, as well as the location of potential point sources of aluminum including municipal WWTPs, industrial facilities, anodizing facilities, and drinking water facilities. As illustrated on this map, there are 15 facilities that may be potential point sources of aluminum to surface waters within this ecoregion, all of them drinking water facilities. Many of the downstream areas are used by various salmon and trout species. Figure 4-18 further illustrates the known fish habitat of the T&E species within the Coast Range ecoregion. For many of the T&E species that occupy the Coast Range it is for migration from the Pacific Ocean to further inland spawning locations (Figure 4-18).

[14] https://www.streamnet.org/; accessed June - July 2019

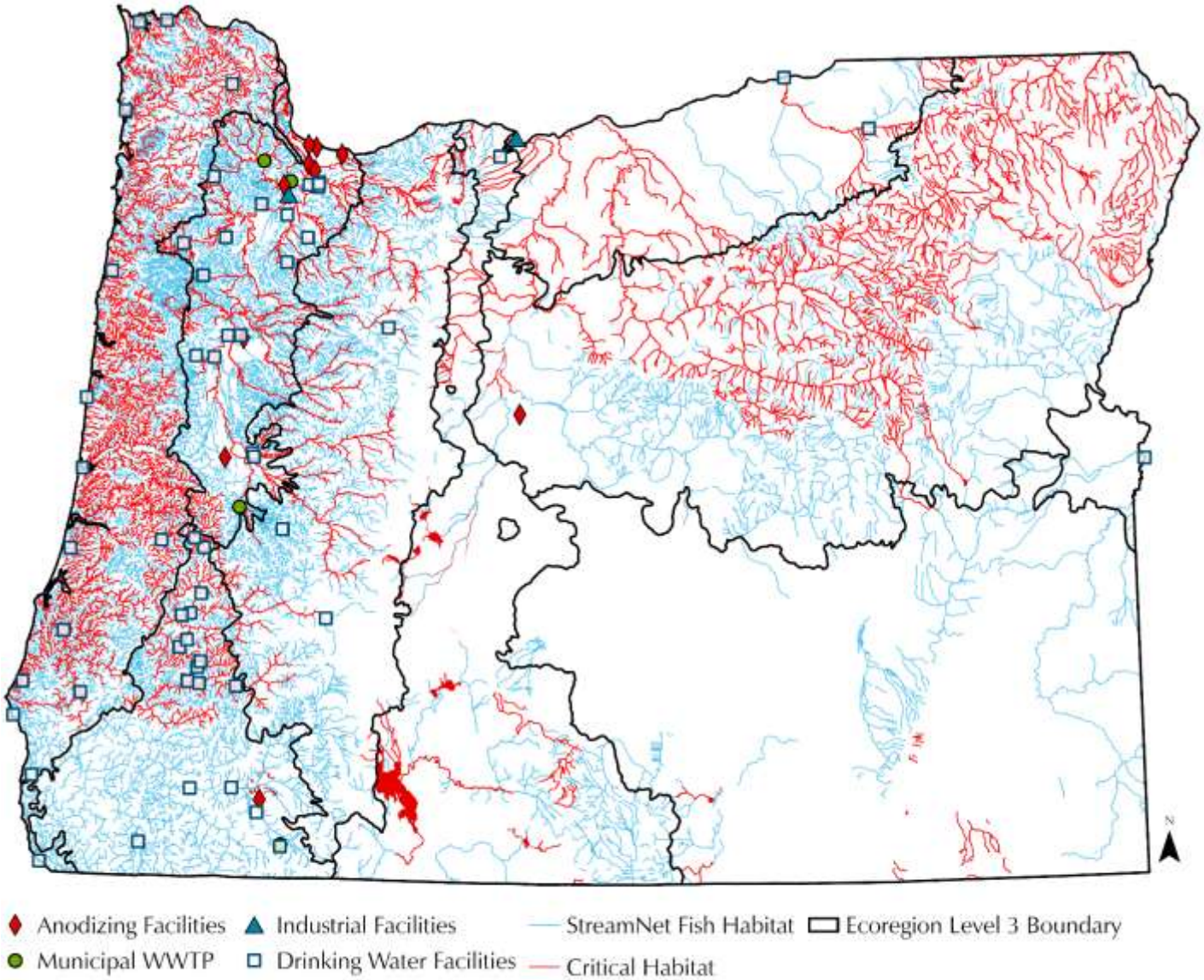


Figure 4-17. Summary of essential and critical habitat across Oregon's ecoregions and the location of point source discharges.

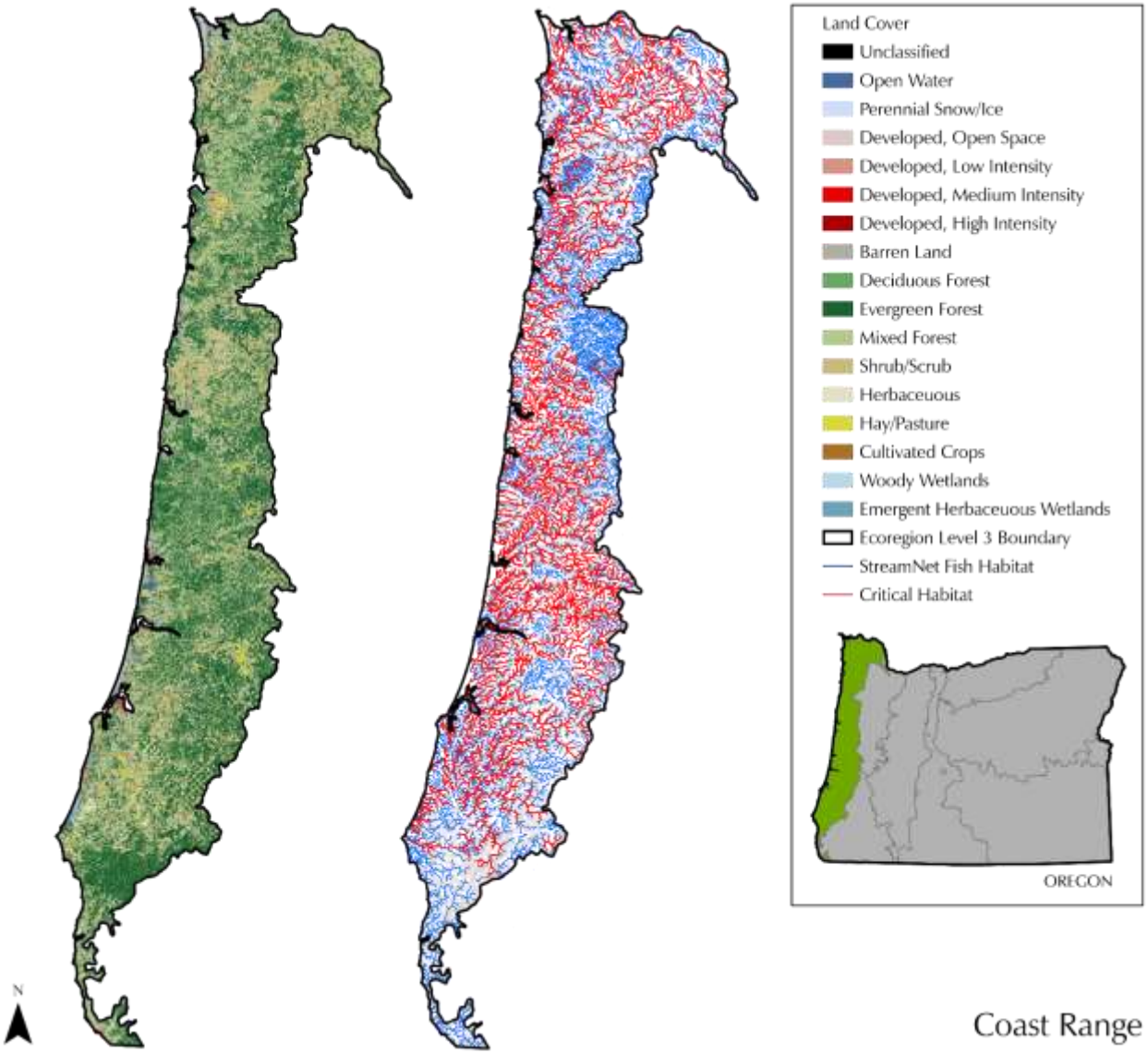


Figure 4-18. Land cover in the Oregon Coast Range Level III Ecoregion. Shading in species habitat map is indicative of elevation, higher elevations are darker than lower elevations.[15]

[15] StreamNet dervived fish habitat/distribution and critical habitat for threatened and endangered species. The StreamNet Fish Distribution (Essential Habitat) dataset documents fish distribution and activity for all fish species available from data submitted to the regional StreamNet project by partner agencies. This dataset was compiled from separate spatial datasets contributed by StreamNet partners as of January 31, 2019. Distribution is based upon survey data and, in some cases, the best professional judgement of local fish biologists, in the Pacific Northwest Region. This dataset also contains information for Critical Habitat for threatened and endangered species derived by the United States Fish and Wildlife Service (USFWS).

The Coast Range had the highest mean pH (8.4 su) and the lowest minimum pH measured (5.6 su) (Table 4-3). As noted in Section 4.3.1, acute and chronic aluminum toxicity is substantially greater at lower pH. DOC in the Coast Range ranged between 0.2 and 28.2 mg/L with a mean concentration of 1.8 mg/L (Table 4-4), while hardness ranged from 2.4 to 413.7 mg/L with a mean of 31.8 (Table 4-5). The concentrations of DOC and hardness were not at the extremes measured in Oregon but could have a substantial impact on the dissociation of aluminum into a dissolved form which is more toxic. Total recoverable surface water aluminum in the Coast Range ecoregion ranged from <0.01 to 3520.0 µg/L with a mean of 289.2 µg/L, while dissolved aluminum surface water concentrations ranged from<0.01 to 110 µg/L with a mean of 9.07 µg/L (Table 4-6). As noted in Figures 4-2 and 4-3, the Coast Range ecoregion has known mineral deposits that may be associated with aluminum in the ecoregion. Of the 1,637 mineral deposits in the Coast Range (Figure 4-2), 12 are known to be associated with aluminum deposits (Figure 4-3) (USGS, 2011). These known mineral deposits along with the range of pH, DOC, and hardness may lead to potential concentrations of aluminum in the surface waters of the Coast Range.

Non-point sources of aluminum to the Coast Range ecoregion include potential inputs from various land covers. Evergreen Forest was the highest land cover in the Coast Range ecoregion at 43% (Table 4-9). Followed by Mixed Forest and Shrub/Scrub both at 19%. Coast Range also had 10,583 stream miles (Table 4-9), which is 12.7% of the total stream miles (83,655 miles) in Oregon. The forested land cover within the Coast Range was positively significantly correlated to dissolved surface water aluminum (Table 4-10, Figure 4-11). As described in Section 4.4.1, since aluminum is one of the most abundant elements in the earth's crust, any disruption in forest soils could potentially represent substantial sources of aluminum in this watershed. Since there is a higher percentage of water overall in this ecoregion as measured by the number of stream miles, the chances for aluminum inputs through runoff are higher as well. Although a small percentage of this ecoregion contained agriculture, it was found to be negatively significantly correlated to dissolved aluminum in this region, (Table 4-11) indicating as the percentage of agricultural land cover increased the dissolved aluminum concentration decreased.

4.6.2 Willamette Valley

There are 7 threatened and endangered species present in the Willamette Valley ecoregion (see Table 4-8) with some species having multiple ESUs present, including:

- Eulachon – Southern DPS
- Coho Salmon – Lower Columbia River
- Chum Salmon – Columbia River
- Chinook Salmon – Upper Columbia River Spring Run, Snake River Spring/Summer Run, Snake River Fall Run, Upper Willamette River, Lower Columbia River
- Sockeye Salmon – Snake River
- Steelhead – Upper Columbia River, Snake River, Middle Columbia River, Upper Willamette River, Lower Columbia River
- Bull trout

Figure 4-17 illustrates habitats identified by StreamNet across Oregon where T&E species have occurred including those within the Willamette Valley ecoregion, as well as the location of potential point sources of aluminum including municipal WWTPs, industrial facilities, anodizing

facilities, and drinking water facilities. As illustrated on this map, there are 32 facilities that may be potential point sources of aluminum to surface waters within this ecoregion including 3 municipal WWTPs, 1 industrial facility, 10 anodizing facilities, and 18 drinking water facilities. Many of the downstream areas are used by various salmon and trout species. Figure 4-19 further illustrates the known fish habitat of the T&E species within the Willamette Valley ecoregion. Many of the T&E species that occupied the Willamette Valley ecoregion streams used them as migration corridors as well as spawning and rearing locations (Figure 4-19). Therefore, not only were adults present but other life stages including eggs, fry and juveniles may have been exposed to surface water aluminum in the Willamette Valley ecoregion.

The Willamette Valley ecoregion had a mean pH of 7.4 su and a minimum pH of 6.2 su (Table 4-3). As noted in Section 4.3.1, acute and chronic aluminum toxicity was substantially greater at lower pH. DOC in the Willamette Valley ranged between 0.2 and 132 mg/L with a mean concentration of 2.5 mg/L (Table 4-4), while hardness ranged from 0.05 to 235 mg/L with a mean of 39.3 (Table 4-5). The concentrations of DOC and hardness were not at the extremes measured in Oregon but could have a substantial impact on the dissociation of aluminum into a dissolved form which is more toxic. Total recoverable surface water aluminum in the Willamette Valley ecoregion ranged from <0.05 to 10,600 µg/L with a mean of 610.9 µg/L, while dissolved aluminum surface water concentrations ranged from <0.05 to 557 µg/L with a mean of 34.9 µg/L (Table 4-6). Of the 1,533 mineral deposits in the Willamette Valley (Figure 4-2), 30 are known to be associated with aluminum deposits (Figure 4-3) (USGS, 2011). These known mineral deposits along with the range of pH, DOC, and hardness may lead to potential concentrations of aluminum in the surface waters of the Willamette Valley.

The Willamette Valley ecoregion is one of the most developed in the State of Oregon, and therefore, has a variety of land covers (Figure 4-19). The most common is Hay/Pasture (26%, Table 4-9), followed by Cultivated Crops (17%), Evergreen Forest (15%), Shrub/Scrub (9%), Developed Low Density (7%), and Mixed Forest (6%). The Willamette Valley's percentage of agriculture land cover was positively significantly correlated to levels of total recoverable aluminum (Table 4-11, Figure 4-14).

4.6.3 Cascades

There are 8 threatened and endangered species present in the Cascades ecoregion (see Table 4-8) with some species having multiple ESUs present, including:

- Eulachon – Southern DPS
- Coho Salmon – Lower Columbia River, Southern OR/Northern CA Coast
- Chum Salmon – Columbia River
- Chinook Salmon – Upper Columbia River Spring Run, Snake River Spring/Summer Run, Snake River Fall Run, Upper Willamette River, Lower Columbia River
- Sockeye Salmon – Snake River
- Steelhead – Upper Columbia River, Snake River, Middle Columbia River, Upper Willamette River, Lower Columbia River
- Oregon spotted frog
- Bull trout

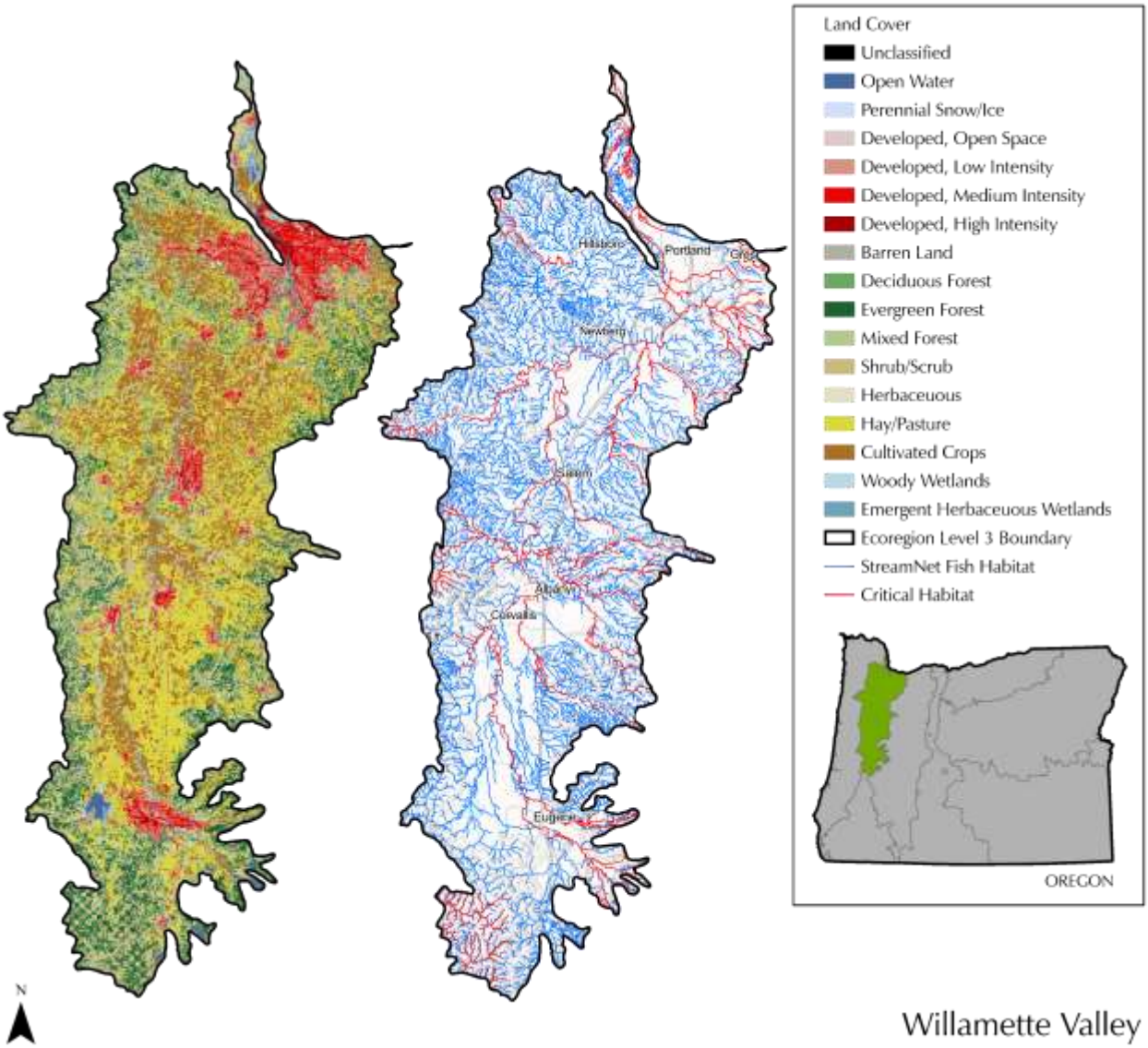


Figure 4-19. Land cover in the Oregon Willamette Valley Level III Ecoregion. Shading in species habitat map is indicative of elevation, higher elevations are darker than lower elevations.[16]

[16] StreamNet derived fish habitat/distribution and critical habitat for threatened and endangered species. The StreamNet Fish Distribution (Essential Habitat) dataset documents fish distribution and activity for all fish species available from data submitted to the regional StreamNet project by partner agencies. This dataset was compiled from separate spatial datasets contributed by StreamNet partners as of January 31, 2019. Distribution is based upon survey data and, in some cases, the best professional judgement of local fish biologists, in the Pacific Northwest Region. This dataset also contains information for Critical Habitat for threatened and endangered species derived by the United States Fish and Wildlife Service (USFWS).

Figure 4-17 illustrates habitats identified by StreamNet across Oregon where T&E species have occurred including those within the Cascades ecoregion, as well as the location of potential point sources of aluminum including municipal WWTPs, industrial facilities, anodizing facilities, and drinking water facilities. As illustrated on this map, there are 3 facilities that may be potential point sources of aluminum to surface waters within the Cascades ecoregion that are all drinking water facilities. Many of the upstream headwaters areas are used by various salmon and trout species for spawning as well as juvenile rearing. Figure 4-20 further illustrate the known fish habitat of the T&E species within the Cascades ecoregion. For many of the T&E species that occupy the Cascades it is for spawning and rearing locations (Figure 4-20).

The Cascades ecoregion had a mean pH of 7.7 su and a minimum pH of 6.5 su (Table 4-3). As noted in Section 4.3.1, acute and chronic aluminum toxicity is substantially greater at lower pH. DOC in the Cascades ecoregion ranged between 0.1 and 14.9 mg/L with a mean concentration of 1.1 mg/L (Table 4-4), while hardness ranged from 6 to 216 mg/L with a mean of 19.0 (Table 4-5). The concentrations of DOC and hardness were not at the extremes measured in Oregon but could have a substantial impact on the dissociation of aluminum into a dissolved form which is more toxic. Total recoverable surface water aluminum in the Cascades ecoregion ranged from 8.2 to 379 µg/L with a mean of 58.8 µg/L, while dissolved aluminum surface water concentrations ranged from <0.05 to 61.6 µg/L with a mean of 17 µg/L (Table 4-6). Of the 1,799 mineral deposits in the Cascades (Figure 4-2), 5 are known to be associated with aluminum deposits (Figure 4-3) (USGS, 2011). These known mineral deposits along with the range of pH, DOC, and hardness may lead to potential concentrations of aluminum in the surface waters of the Cascades.

This ecoregion is very similar to the Cascades, with most of the land use being Evergreen Forest (79%), followed by Shrub/Scrub (14%) (Figure 4-20). The measured surface water concentration of dissolved aluminum in this region was negatively significantly correlated to developed lands (Table 4-10). As development in the ½ mile area surrounding the surface water sampling location increased in the Cascades, the dissolved aluminum concentration decreased.

4.6.4 Eastern Cascades Slopes and Foothills

There are 11 threatened and endangered species present in the Eastern Cascades Slopes and Foothills ecoregion (see Table 4-8) with some species having multiple ESUs present, including:

- Eulachon – Southern DPS
- Coho Salmon – Lower Columbia River
- Chum Salmon – Columbia River
- Chinook Salmon – Upper Columbia River Spring Run, Snake River Spring/Summer Run, Snake River Fall Run, Lower Columbia River
- Sockeye Salmon – Snake River
- Steelhead – Upper Columbia River, Snake River, Middle Columbia River, Lower Columbia River
- Oregon spotted frog
- Short nose sucker
- Lost River sucker
- Hutton Tui chub
- Bull trout

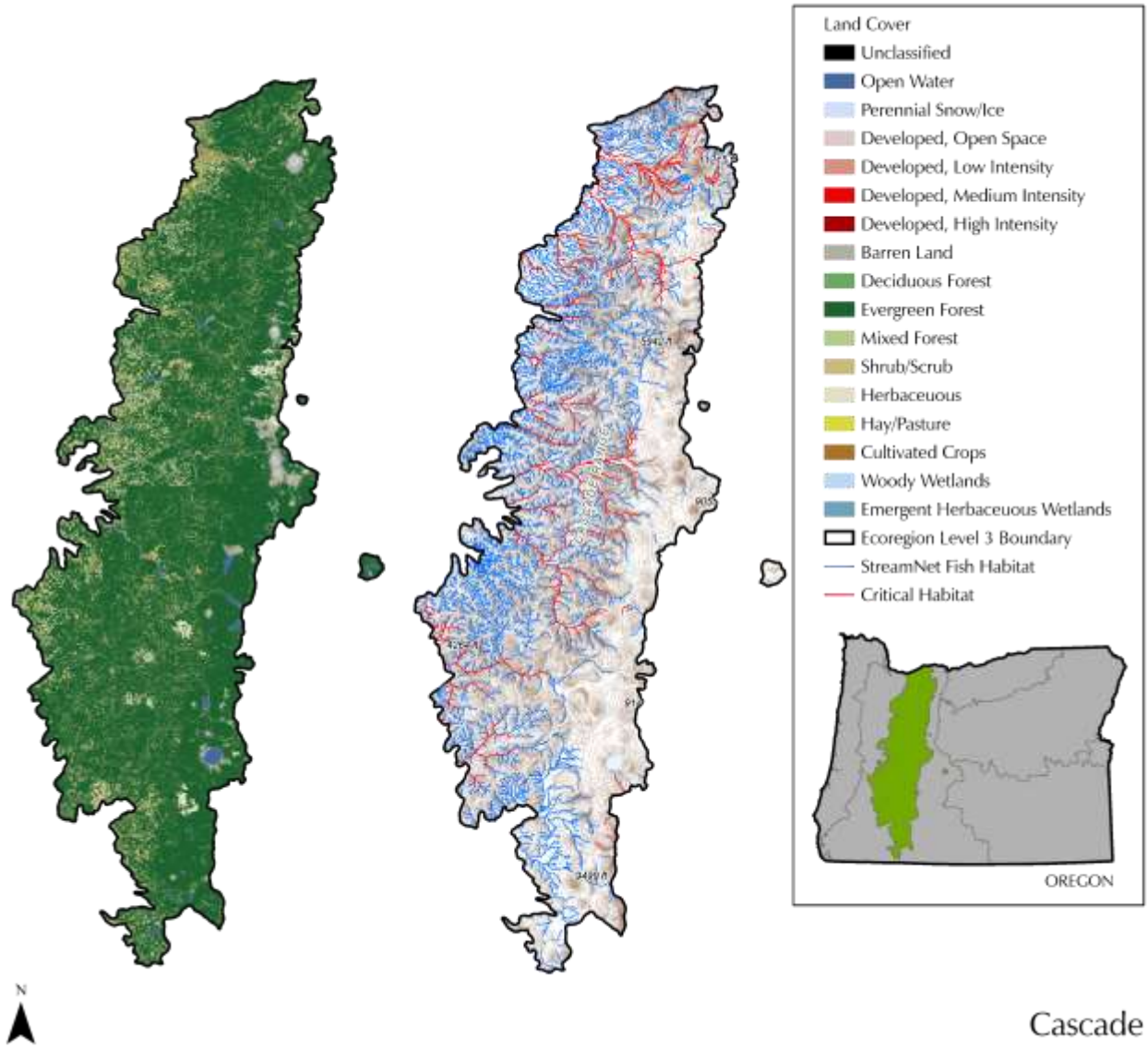


Figure 4-20. Land use in the Oregon Cascades Level III Ecoregion. Shading in species habitat map is indicative of elevation, higher elevations are darker than lower elevations.[17]

[17] StreamNet dervived fish habitat/distribution and critical habitat for threatened and endangered species. The StreamNet Fish Distribution (Essential Habitat) dataset documents fish distribution and activity for all fish species available from data submitted to the regional StreamNet project by partner agencies. This dataset was compiled from separate spatial datasets contributed by StreamNet partners as of January 31, 2019. Distribution is based upon survey data and, in some cases, the best professional judgement of local fish biologists, in the Pacific Northwest Region. This dataset also contains information for Critical Habitat for threatened and endangered species derived by the United States Fish and Wildlife Service (USFWS).

Figure 4-17 illustrates habitats identified by StreamNet across Oregon where T&E species have occurred including those within the Eastern Cascades Slopes and Foothills ecoregion, as well as the location of potential point sources of aluminum including municipal WWTPs, industrial facilities, anodizing facilities, and drinking water facilities. As illustrated on this map, there are one facility that may be potential point sources of aluminum to surface waters within the Eastern Cascades Slopes and Foothills ecoregion, a drinking water facility. Many of the upstream headwaters areas are used by various salmon and trout species for spawning as well as juvenile rearing. Figure 4-21 further illustrates the known habitat of T&E species within the Eastern Cascades Slopes and Foothills ecoregion. For many of the anadromous fish T&E species that occupy the Eastern Cascades Slopes and Foothills use it for spawning and rearing locations (Figure 4-21), while other species including the Oregon spotted frog, Short nose sucker, Lost River sucker, and Hutton Tui chub, the entire life span may occur in this ecoregion.

This ecoregion is also very forested with most of the land use is Evergreen Forest (53%), Shrub/Scrub (25%), and Herbaceous (9%) (Figure 4-21). Table 4-7 shows that 10.1% (8,449 miles) of Oregon's streams (83,655 miles) are found in this ecoregion. Although it was a small percentage overall (1%, Figure 4-21), the Barren land use was the only land use significantly correlated with aluminum and it was positively significantly correlated with total recoverable aluminum (Figure 4-16).

The Eastern Cascades Slopes and Foothills ecoregion had a mean pH of 7.9 su and a minimum pH of 6.5 su (Table 4-3). As noted in Section 4.3.1, acute and chronic aluminum toxicity is substantially greater at lower pH. DOC in the Eastern Cascades Slopes and Foothills ecoregion ranged between 0.3 and 50.0 mg/L with a mean concentration of 4.1 mg/L (Table 4-4), while hardness ranged from 10.1 to 461 mg/L with a mean of 44.3 (Table 4-5). The concentrations of DOC and hardness were not at the extremes measured in Oregon but could have a substantial impact on the dissociation of aluminum into a dissolved form which is more toxic. Total recoverable surface water aluminum in the Eastern Cascades Slopes and Foothills ecoregion ranged from <0.05 to 5930 µg/L with a mean of 439 µg/L, while dissolved aluminum surface water concentrations ranged from <0.05 to 220 µg/L with a mean of 30.3 µg/L (Table 4-6). Of the 748 mineral deposits in the Eastern Cascades Slopes and Foothills (Figure 4-2), there are none that are known to be associated with aluminum deposits (Figure 4-3) (USGS, 2011). Thus, known mineral deposits may not be leading to elevated aluminum in the Eastern Cascades Slopes and Foothills.

4.6.5 Columbia Plateau

There are 5 threatened and endangered species present in the Columbia Plateau ecoregion (see Table 4-8) with some species having multiple ESUs present, including:

- Chum Salmon – Columbia River
- Chinook Salmon – Upper Columbia River Spring Run, Snake River Spring/Summer Run, Snake River Fall Run
- Sockeye Salmon – Snake River
- Steelhead – Upper Columbia River, Snake River, Middle Columbia River
- Bull trout

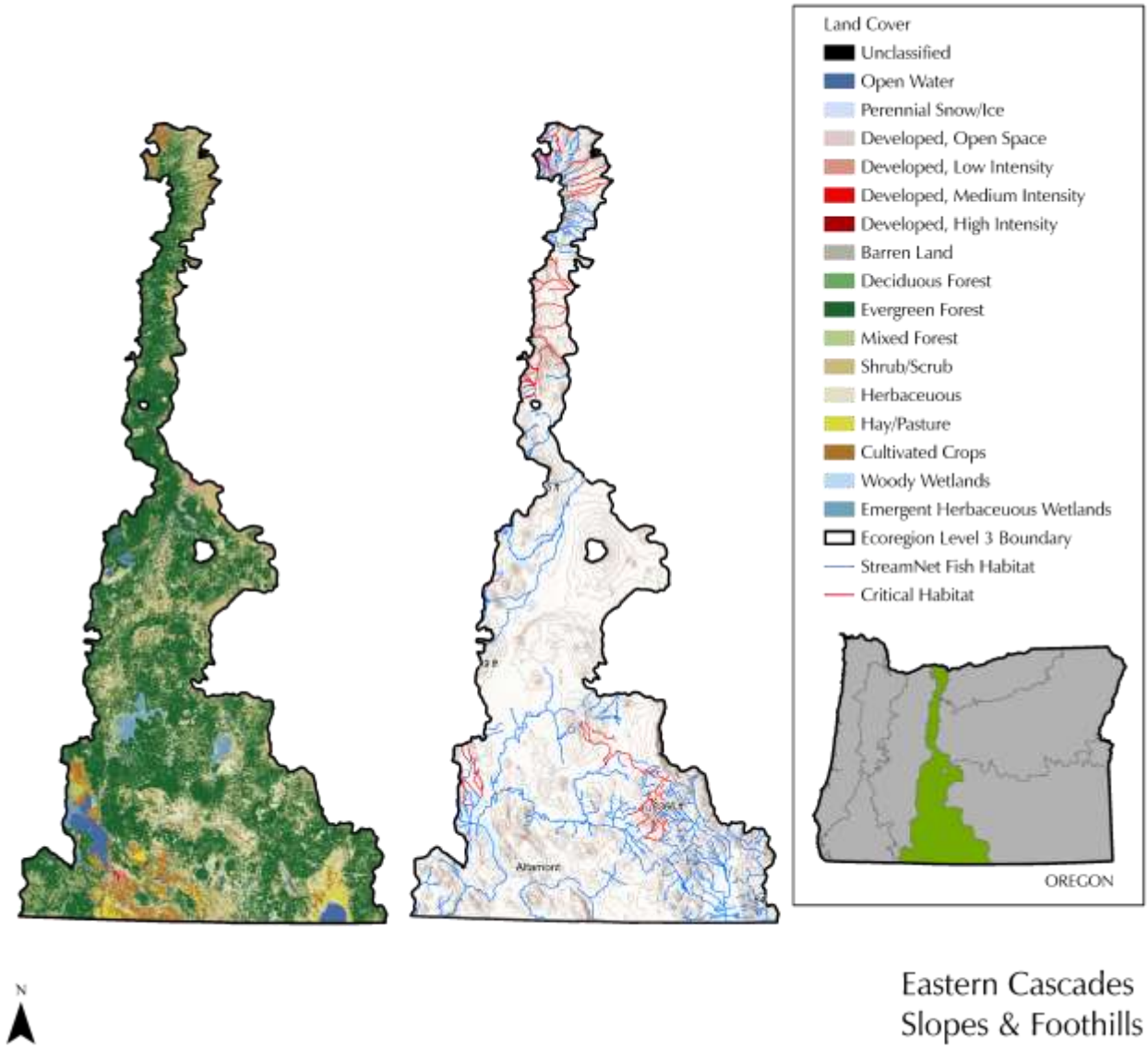


Figure 4-21. Land use in the Oregon Eastern Cascades Slopes and Foothills Level III Ecoregion. Shading in species habitat map is indicative of elevation, higher elevations are darker than lower elevations.[18]

[18] StreamNet dervived fish habitat/distribution and critical habitat for threatened and endangered species. The StreamNet Fish Distribution (Essential Habitat) dataset documents fish distribution and activity for all fish species available from data submitted to the regional StreamNet project by partner agencies. This dataset was compiled from separate spatial datasets contributed by StreamNet partners as of January 31, 2019. Distribution is based upon survey data and, in some cases, the best professional judgement of local fish biologists, in the Pacific Northwest Region. This dataset also contains information for Critical Habitat for threatened and endangered species derived by the United States Fish and Wildlife Service (USFWS).

Figure 4-17 illustrates habitats identified by StreamNet across Oregon where T&E species have occurred including those within the Columbia Plateau ecoregion, as well as the location of potential point sources of aluminum including municipal WWTPs, industrial facilities, anodizing facilities, and drinking water facilities. As illustrated on this map, there are 3 facilities that may be potential point sources of aluminum to surface waters within the Columbia Plateau ecoregion including 1 industrial facility, and 2 drinking water facilities. Many of the upstream headwaters areas are used by various salmon and trout species for spawning as well as juvenile rearing. Figure 4-22 further illustrates the known habitat of T&E species within the Columbia Plateau ecoregion. For many of the anadromous fish T&E species that occur in the Columbia Plateau ecoregion it is for spawning and rearing purposes, thus not only do adults have the potential to be exposed to surface water aluminum. eggs, fry, and juveniles may also be potentially exposed (Figure 4-22).

The Columbia Plateau is a mix of forest and agricultural land. It is predominantly Shrub/Scrub (57%), but there is also 33% of the area used as Cultivated Crops (Figure 4-22). Although this ecoregion only makes up 9.2% (7,704 miles) of the total Oregon stream miles (83.655 miles) (Table 4-7), the land cover potentially could allow for inputs into surface water. Agriculture and developed lands in this ecoregion were the only land uses significantly correlated to surface water aluminum. Developed land was negatively significantly correlated to total recoverable aluminum (Table 11; Figure 4-15).

The Columbia Plateau ecoregion had a mean pH of 8.3 su and a minimum pH of 7.1 su (Table 4-3). As noted in Section 4.3.1, acute and chronic aluminum toxicity is substantially greater at lower pH. DOC in the Columbia Plateau ecoregion ranged between 0.8 and 28.2 mg/L with a mean concentration of 2.7 mg/L (Table 4-4), while hardness ranged from 15.8 to 265 mg/L with a mean of 61.6 (Table 4-5). The concentrations of DOC and hardness were not at the extremes measured in Oregon but could have a substantial impact on the dissociation of aluminum into a dissolved form which is more toxic. Total recoverable surface water aluminum in the Columbia Plateau ecoregion ranged from <0.05 to 2340 µg/L with a mean of 489.9 µg/L, while dissolved aluminum surface water concentrations ranged from <0.05 to 238 µg/L with a mean of 25.5 µg/L (Table 4-6). Of the 701 mineral deposits in the Columbia Plateau (Figure 4-2), only 1 is known to be associated with aluminum deposits (Figure 4-3) (USGS, 2011). These known mineral deposits along with the range of pH, DOC, and hardness may lead to potential concentrations of aluminum in the surface waters of the Columbia Plateau.

4.6.6 Blue Mountains

There are only 4 threatened and endangered species present in the Blue Mountains ecoregion (see Table 4-8) with some species having multiple ESUs present, including:

- Chinook Salmon –Snake River Spring/Summer Run, Snake River Fall Run
- Sockeye Salmon – Snake River
- Steelhead – Snake River, Middle Columbia River
- Bull trout

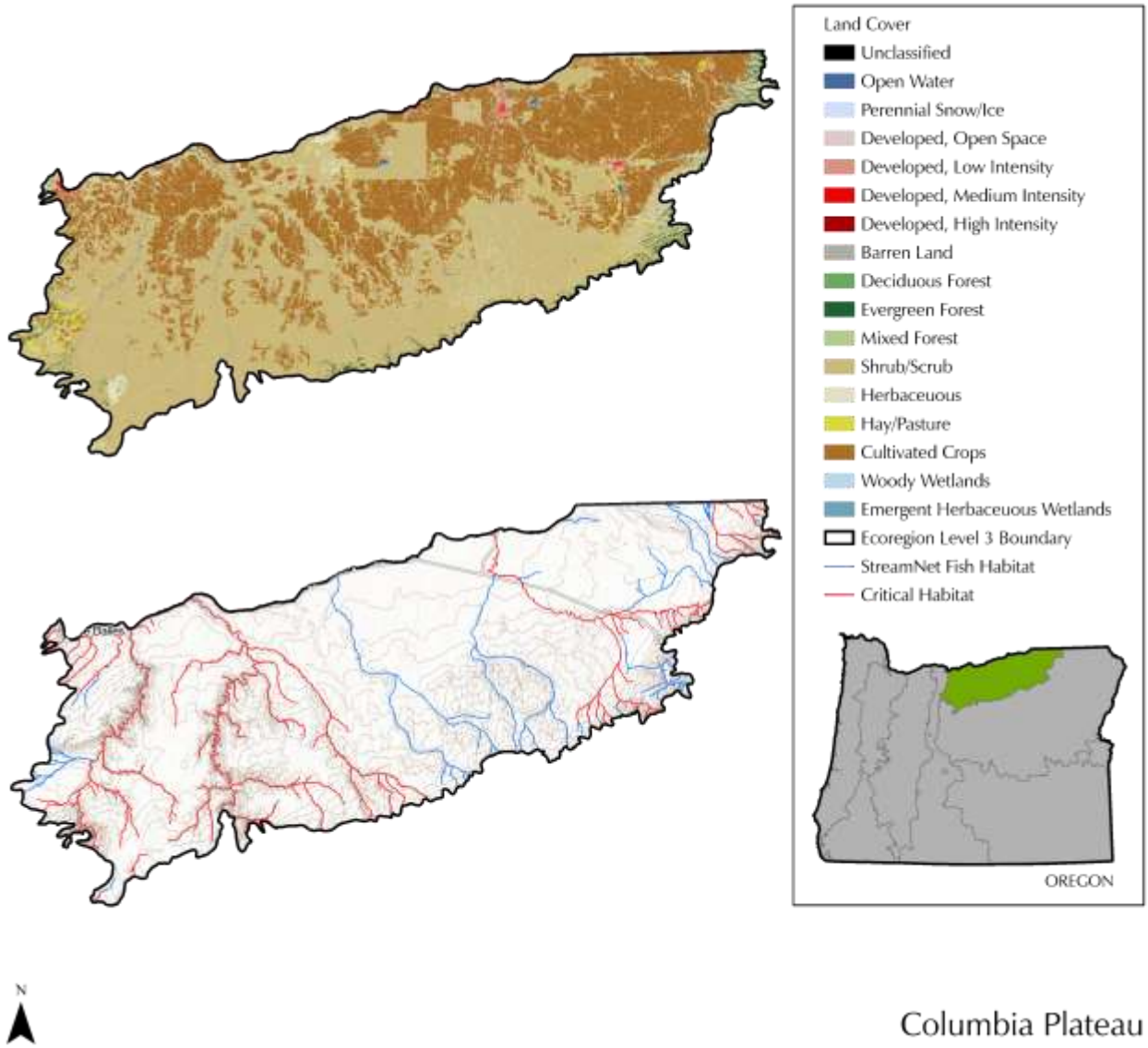


Figure 4-22. Land use in the Oregon Columbia Plateau Level III Ecoregion. Shading in species habitat map is indicative of elevation, higher elevations are darker than lower elevations.[19]

[19] StreamNet dervived fish habitat/distribution and critical habitat for threatened and endangered species. The StreamNet Fish Distribution (Essential Habitat) dataset documents fish distribution and activity for all fish species available from data submitted to the regional StreamNet project by partner agencies. This dataset was compiled from separate spatial datasets contributed by StreamNet partners as of January 31, 2019. Distribution is based upon survey data and, in some cases, the best professional judgement of local fish biologists, in the Pacific Northwest Region. This dataset also contains information for Critical Habitat for threatened and endangered species derived by the United States Fish and Wildlife Service (USFWS).

Figure 4-17 illustrates habitats identified by StreamNet across Oregon where T&E species have occurred including those within the Blue Mountains ecoregion, as well as the location of potential point sources of aluminum including municipal WWTPs, industrial facilities, anodizing facilities, and drinking water facilities. As illustrated on this map, there is one facility that may be a potential point sources of aluminum to surface waters within the Blue Mountains ecoregion, an anodizing facility. Many of the upstream headwaters areas are used by various salmon and trout species for spawning as well as juvenile rearing. Figure 4-23 further illustrates the known habitat of the T&E species within Blue Mountain ecoregion. For many of the anadromous fish T&E species that occur in the Blue Mountain ecoregion it is for spawning and rearing purposes, thus not only do adults have the potential to be exposed to surface water aluminum. eggs, fry, and juveniles may also be exposed (Figure 4-23).

The Blue Mountain ecoregion is almost entirely made up of natural areas (Shrub/Scrub, 50% and Evergreen Forest, 38%) (Figure 4-23). The Blue Mountains also contribute 9.2% (7,704 miles) of Oregon's stream miles (83,655 miles) (Table 4-9). As such, any forestry or silviculture work could contribute to higher levels of aluminum in the watershed. In the Blue Mountains, surface water concentration of dissolved aluminum was the only significantly correlated land use and it was significantly negatively correlated with the percentage of developed lands (Table 4-10). Thus, as the % of developed land increased in the ½ mile surrounding the surface water sampling location, the concentration of dissolved aluminum decreased.

The Blue Mountains ecoregion had a mean pH of 8.2 su and a minimum pH of 6.9 su (Table 4-3). As noted in Section 4.3.1, acute and chronic aluminum toxicity is substantially greater at lower pH. DOC in the Blue Mountains ecoregion ranged between 0.3 and 36.5 mg/L with a mean concentration of 2.8 mg/L (Table 4-4), while hardness ranged from 8.7 to 343 mg/L with a mean of 70.7 (Table 4-5). The concentrations of DOC and hardness were not at the extremes measured in Oregon but could have a substantial impact on the dissociation of aluminum into a dissolved form which is more toxic. Total recoverable surface water aluminum in the Blue Mountains ecoregion ranged from 14.5 to 9,340 µg/L with a mean of 552.1 µg/L, while dissolved aluminum surface water concentrations ranged from <0.05 to 366 µg/L with a mean of 21.1 µg/L (Table 4-6). Of the 4,427 mineral deposits in the Blue Mountains (Figure 4-2), only 1 is known to be associated with aluminum deposits (Figure 4-3) (USGS, 2011). These known mineral deposits along with the range of pH, DOC, and hardness may lead to potential concentrations of aluminum in the surface waters of the Blue Mountains.

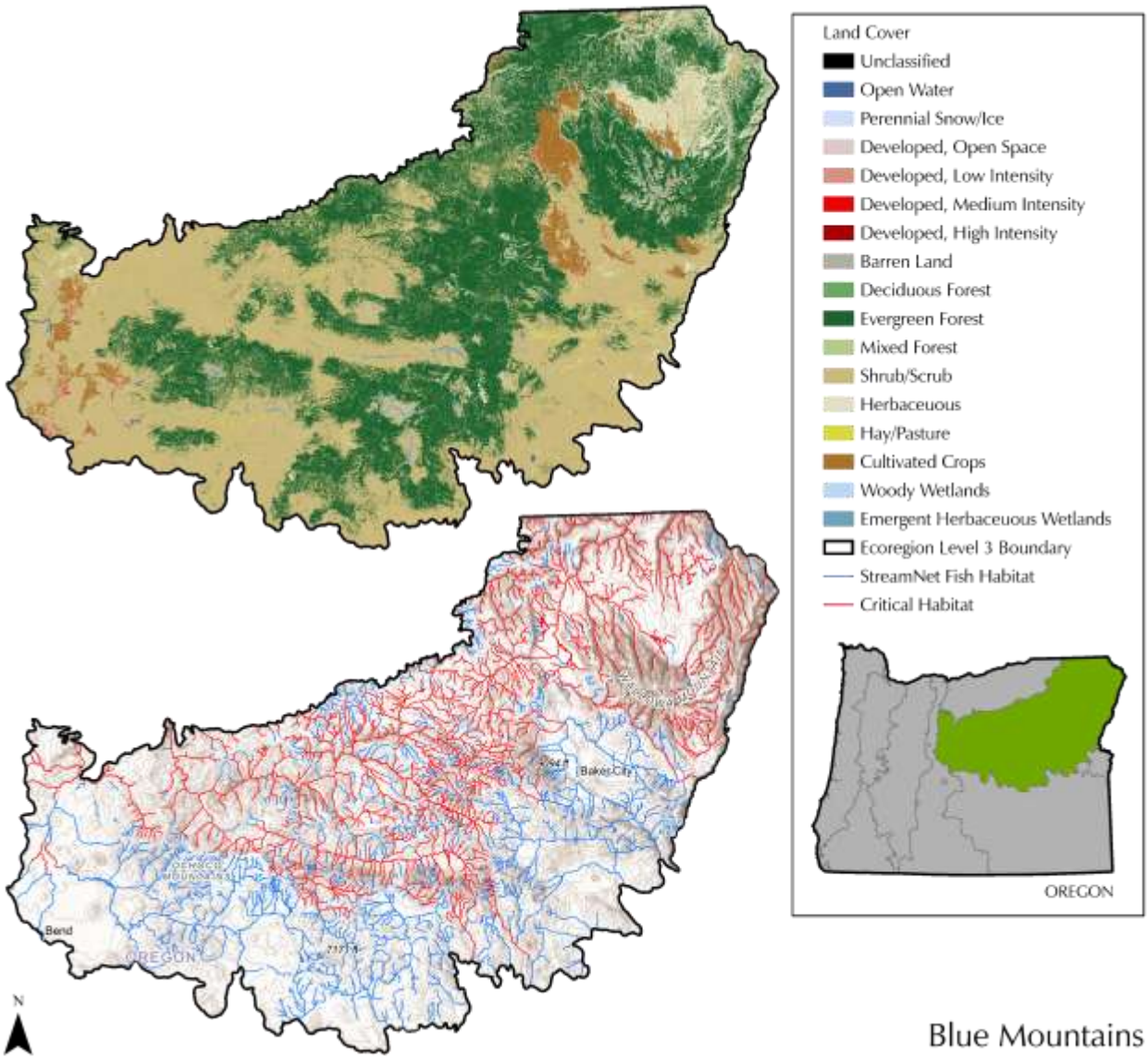


Figure 4-23. Land use in the Oregon Blue Mountains Level III Ecoregion. Shading in species habitat map is indicative of elevation, higher elevations are darker than lower elevations.[20]

[20] StreamNet dervived fish habitat/distribution and critical habitat for threatened and endangered species. The StreamNet Fish Distribution (Essential Habitat) dataset documents fish distribution and activity for all fish species available from data submitted to the regional StreamNet project by partner agencies. This dataset was compiled from separate spatial datasets contributed by StreamNet partners as of January 31, 2019. Distribution is based upon survey data and, in some cases, the best professional judgement of local fish biologists, in the Pacific Northwest Region. This dataset also contains information for Critical Habitat for threatened and endangered species derived by the United States Fish and Wildlife Service (USFWS).

4.6.7 Snake River Plain

There is only one threatened and endangered species present in the Snake River Plain ecoregion (see Table 4-8), the bull trout.

Figure 4-17 illustrates habitats identified by Streamnet across Oregon where T&E species have occurred including those within the Snake River Plain ecoregion, as well as the location of potential point sources of aluminum including municipal WWTPs, industrial facilities, anodizing facilities, and drinking water facilities. As illustrated on this map, there is only 1 facility that may be a potential point source of aluminum to surface waters within the Snake River Plain ecoregion, a drinking water facility.

This ecoregion is a mixture of agricultural (Cultivated Crops, 28%) and natural areas (Herbaceous, 38% and Shrub/Scrub, 24%) (Figure 4-24). Agriculture was the only land use significantly correlated to surface water aluminum concentration and it was significantly positively correlated to total recoverable aluminum in this ecoregion (Table 4-11, Figure 4-13) However, only 1.4% (1164.8) of the state's stream miles (83,655 miles) runs through the Snake River Plain (Table 4-7).

The Snake River Plain ecoregion had a mean pH of 8.4 su and a minimum pH of 5.6 su (Table 4-3). As noted in Section 4.3.1, acute and chronic aluminum toxicity is substantially greater at lower pH. DOC in the Snake River Plain ecoregion ranged between 0.8 and 46.5 mg/L with a mean concentration of 4.4 mg/L (Table 4-4), while hardness ranged from 46.9 to 589 mg/L with a mean of 210.2 (Table 4-5). The Snake River Plain ecoregion had the highest mean DOC and hardness measured of all the Oregon ecoregions. Total recoverable surface water aluminum in the Snake River Plain ecoregion ranged from 55.4 to 67,000 µg/L with a mean of 6743.1 µg/L, while dissolved aluminum surface water concentrations ranged from <0.05 to 1250 µg/L with a mean of 167 µg/L (Table 4-6). Of the 201 mineral deposits in the Snake River Plain (Figure 4-2), none are associated with aluminum deposits (Figure 4-3) (USGS, 2011). Thus, known mineral deposits may not be leading to elevated aluminum in the Snake River Plain.

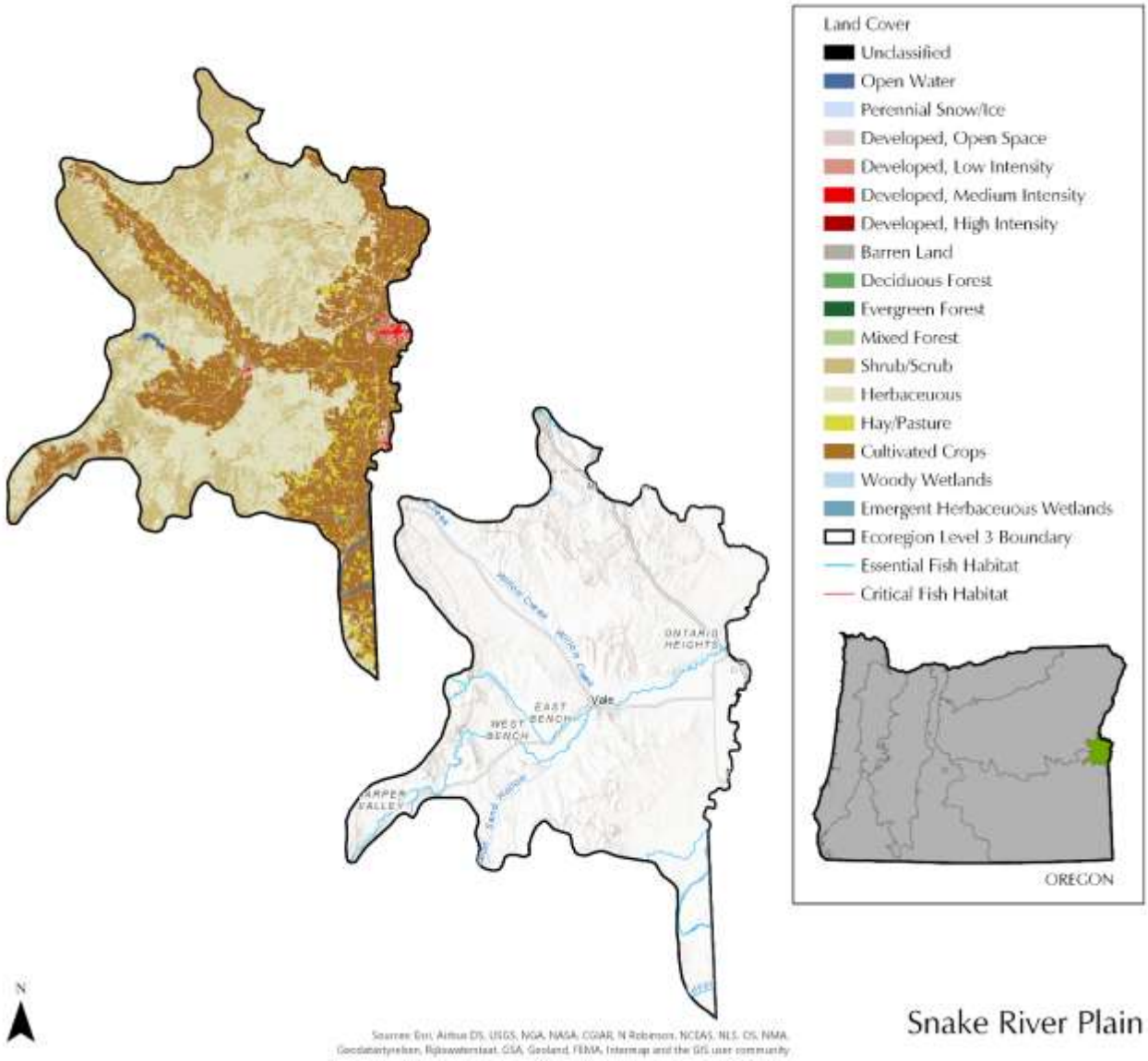


Figure 4-24. Land use in the Oregon Snake River Plain Level III Ecoregion. Shading in species habitat map is indicative of elevation, higher elevations are darker than lower elevations.[21]

[21] StreamNet dervived fish habitat/distribution and critical habitat for threatened and endangered species. The StreamNet Fish Distribution (Essential Habitat) dataset documents fish distribution and activity for all fish species available from data submitted to the regional StreamNet project by partner agencies. This dataset was compiled from separate spatial datasets contributed by StreamNet partners as of January 31, 2019. Distribution is based upon survey data and, in some cases, the best professional judgement of local fish biologists, in the Pacific Northwest Region. This dataset also contains information for Critical Habitat for threatened and endangered species derived by the United States Fish and Wildlife Service (USFWS).

4.6.8 Klamath Mountains

There are only 2 threatened and endangered species present in the Klamath Mountains ecoregion (see Table 4-8) with some species having multiple ESUs present, including:

- Coho Salmon – Oregon Coast, Southern OR/Northern CA Coast
- Vernal Pool Shrimp

Figure 4-17 illustrates habitats identified by StreamNet across Oregon where T&E species have occurred including those within the Klamath Mountains ecoregion, as well as the location of potential point sources of aluminum including municipal WWTPs, industrial facilities, anodizing facilities, and drinking water facilities. As illustrated on this map, there are 19 facilities that may be potential point sources of aluminum to surface waters within the Klamath Mountains ecoregion including 1 municipal WWTP, 1 anodizing facility, and 17 drinking water facilities. Many of the upstream headwaters areas are used by various salmon and trout species for spawning as well as juvenile rearing. Figure 4-25 further illustrates the known habitat of the T&E species within Klamath Mountains ecoregion. For many of the anadromous fish T&E species that occur in the Klamath Mountains ecoregion it is for spawning and rearing purposes, thus not only do adults have the potential to be exposed to surface water aluminum, eggs, fry, and juveniles may also be potentially exposed (Figure 4-25).

Evergreen forest (56%) is the prevailing land use in this ecoregion, followed by Shrub/Scrub (21%) (Figure 4-25). There are 6,131 stream miles in this ecoregion (Table 4-7), 7.3% of Oregon's total (83,655 miles). Only one land cover type was significantly correlated to surface water aluminum. Correlation analysis indicated that the surface water concentration of dissolved aluminum in the Klamath Mountain ecoregion was significantly negatively correlated with the percentage of developed lands (Table 4-9). Thus, as the percentage of developed land increased in the ½ mile surrounding the surface water sampling location, the concentration of dissolved aluminum decreased.

The Klamath Mountains ecoregion had a mean pH of 8.0 su and a minimum pH of 5.9 su (Table 4-3). As noted in Section 4.3.1, acute and chronic aluminum toxicity is substantially greater at lower pH. DOC in the Klamath Mountains ecoregion ranged between 0.6 and 18.3 mg/L with a mean concentration of 2.2 mg/L (Table 4-4), while hardness ranged from 15.4 to 589 mg/L with a mean of 44.6 (Table 4-5). The concentrations of DOC and hardness were not at the extremes measured in Oregon but could have a substantial impact on the dissociation of aluminum into a dissolved form which is more toxic. Total recoverable surface water aluminum in the Klamath Mountains ecoregion ranged from <0.05 to 2,110 µg/L with a mean of 392.7 µg/L, while dissolved aluminum surface water concentrations ranged from <0.05 to 119 µg/L with a mean of 24.5 µg/L (Table 4-5). Of the 4,136 mineral deposits in the Klamath Mountains (Figure 4-2), only 2 are known to be associated with aluminum deposits (Figure 4-3) (USGS, 2011). These known mineral deposits along with the range of pH, DOC, and hardness may lead to potential concentrations of aluminum in the surface waters of the Klamath Mountains.

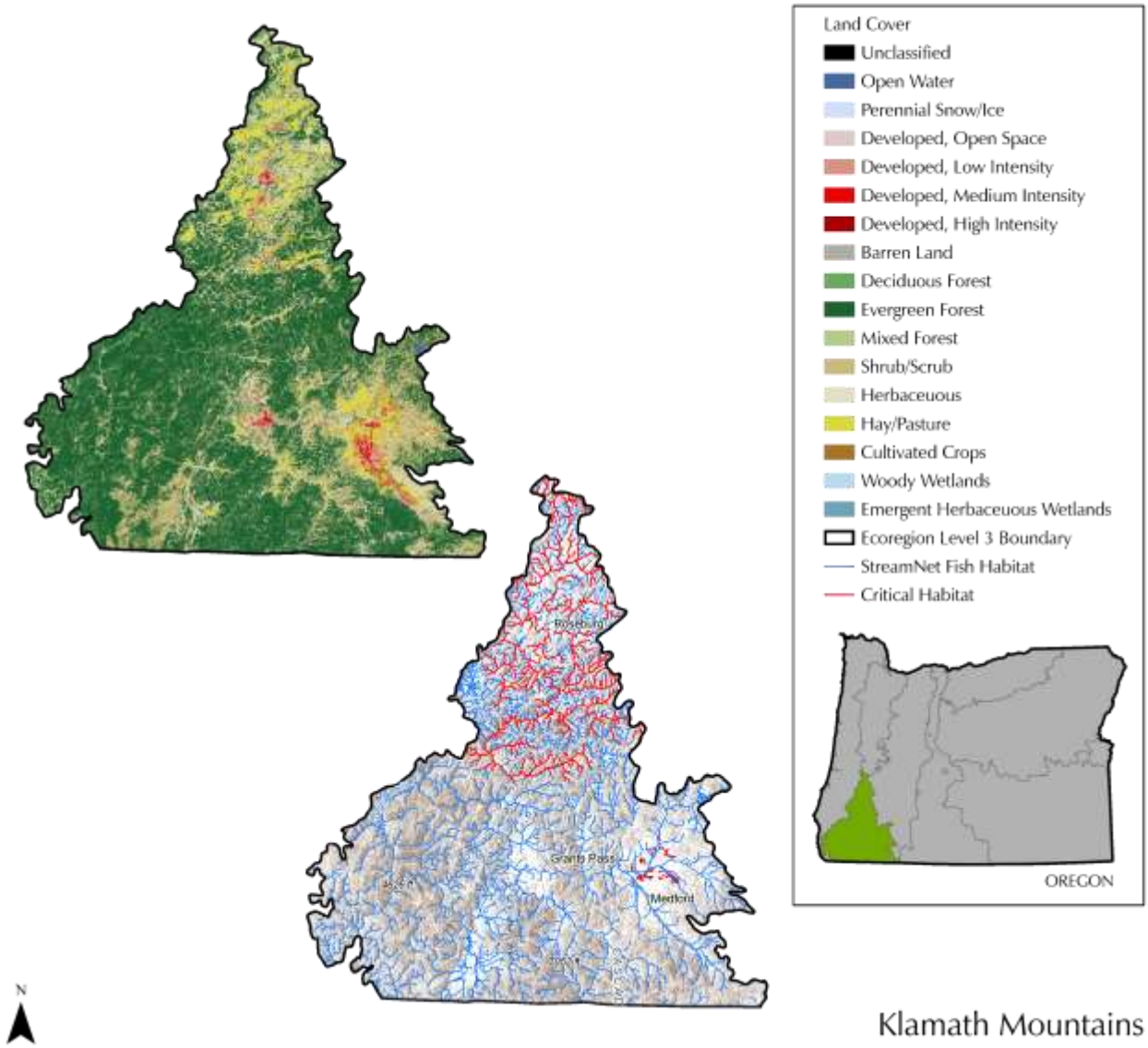


Figure 4-25. Land use in the Oregon Klamath Mountains Level III Ecoregion. Shading in species habitat map is indicative of elevation, higher elevations are darker than lower elevations.[22]

[22] StreamNet dervived fish habitat/distribution and critical habitat for threatened and endangered species. The StreamNet Fish Distribution (Essential Habitat) dataset documents fish distribution and activity for all fish species available from data submitted to the regional StreamNet project by partner agencies. This dataset was compiled from separate spatial datasets contributed by StreamNet partners as of January 31, 2019. Distribution is based upon survey data and, in some cases, the best professional judgement of local fish biologists, in the Pacific Northwest Region. This dataset also contains information for Critical Habitat for threatened and endangered species derived by the United States Fish and Wildlife Service (USFWS).

4.6.9 Northern Basin and Range

There are 6 threatened and endangered species present in the Northern Basin and Range ecoregion (see Table 4-7), including:

- Warner sucker
- Hutton Tui chub
- Borax Lake chub
- Lahontan Cutthroat trout
- Foskett Speckled dace
- Bull trout

There are no municipal WWTPs, industrial facilities, anodizing facilities or drinking water facilities in this ecoregion.

Figure 4-17 illustrates habitats identified by StreamNet across Oregon where T&E species have occurred including those within the Northern Basin and Range ecoregion, as well as the location of potential point sources of aluminum including municipal WWTPs, industrial facilities, anodizing facilities, and drinking water facilities. As illustrated on this map, there are no facilities that may be potential point sources of aluminum to surface waters within the Northern Basin and Range ecoregion. Many of the upstream headwaters areas are used by various salmon and trout species for spawning as well as juvenile rearing. Figure 4-26 further illustrates the known habitat of the T&E species within Northern Basin and Range ecoregion. For many of the T&E species that occur in the Northern Basin and Range ecoregion it is for the entire life span (Figure 4-26).

This is a predominantly forested ecoregion (Evergreen forest, 86%) (Figure 4-26). Twenty-nine % (24,376 miles) of Oregon's total stream miles (83,655) are found here (Table 4-7). Similar to the Coast Range, the forested land use in the Northern Basin was significantly positively correlated to surface water dissolved aluminum (Table 4-9, Figure 4-11).

The Northern Basin and Range ecoregion had a mean pH of 8.2 su and a minimum pH of 7.0 su (Table 4-3). As noted in Section 4.3.1, acute and chronic aluminum toxicity is substantially greater at lower pH. DOC in the Northern Basin and Range ecoregion ranged between 0.5 and 28.2 mg/L with a mean concentration of 3.3 mg/L (Table 4-4), while hardness ranged from 22.2 to 200.5 mg/L with a mean of 67.9 (Table 4-5). The concentrations of DOC and hardness were not at the extremes measured in Oregon but could have a substantial impact on the dissociation of aluminum into a dissolved form which is more toxic. Total recoverable surface water aluminum in the Northern Basin and Range ecoregion ranged from 28.1 to 2,480 µg/L with a mean of 693.2 µg/L, while dissolved aluminum surface water concentrations ranged from <0.05 to 2,390 µg/L with a mean of 165.3 µg/L (Table 4-5). Of the 1,195 mineral deposits in the Northern Basin and Range (Figure 4-2), none are known to be associated with aluminum deposits (Figure 4-3) (USGS, 2011). Thus, known mineral deposits may not be leading to elevated aluminum in the Northern Basin and Range.

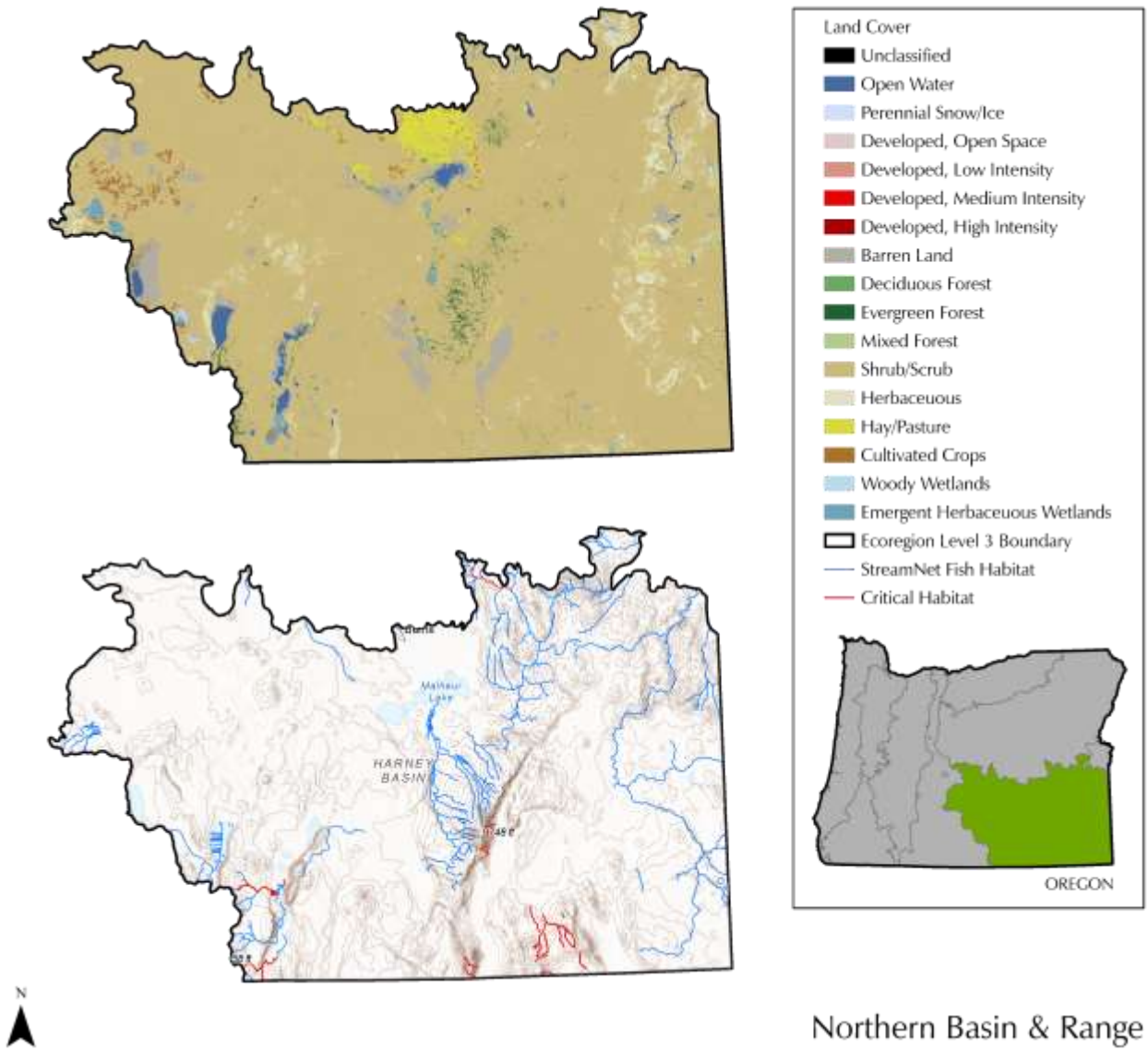


Figure 4-26. Land use in the Oregon Northern Basin and Range Level III Ecoregion. Shading in species habitat map is indicative of elevation, higher elevations are darker than lower elevations.[23]

[23] StreamNet dervived fish habitat/distribution and critical habitat for threatened and endangered species. The StreamNet Fish Distribution (Essential Habitat) dataset documents fish distribution and activity for all fish species available from data submitted to the regional StreamNet project by partner agencies. This dataset was compiled from separate spatial datasets contributed by StreamNet partners as of January 31, 2019. Distribution is based upon survey data and, in some cases, the best professional judgement of local fish biologists, in the Pacific Northwest Region. This dataset also contains information for Critical Habitat for threatened and endangered species derived by the United States Fish and Wildlife Service (USFWS).

4.7 Baseline Evaluation Regarding Mixing Zone

Mixing zones are components of National Pollutant Discharge Elimination System (NPDES) permits administered under the federal Clean Water Act. 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. A mixing zone allows for a discharge to undergo initial dilution and mixing in the receiving stream. Water quality criteria are suspended or lessened within that zone provided several of the conditions listed below are met. 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.

Oregon's administrative rules identify the rules that Oregon implements in establishing mixing zones in NPDES permits.[24] The Oregon administrative rules for mixing zones are excerpted below:

Department of Environmental Quality Chapter 340 Division 41
WATER QUALITY STANDARDS: BENEFICIAL USES, POLICIES, AND CRITERIA FOR OREGON
340-041-0053
Mixing Zones

(1) The Department may allow a designated portion of a receiving water to serve as a zone of dilution for wastewaters and receiving waters to mix thoroughly and this zone will be defined as a mixing zone;

(2) The Department may suspend all or part of the water quality standards, or set less restrictive standards in the defined mixing zone, provided that the following conditions are met:

(a) A point source for which the mixing zone is established may not cause or significantly contribute to any of the following:

(A) Materials in concentrations that will cause acute toxicity to aquatic life as measured by a Department approved bioassay method. Acute toxicity is lethal to aquatic life as measured by a significant difference in lethal concentration between the control and 100 percent effluent in an acute bioassay test. Lethality in 100 percent effluent may be allowed due to ammonia and chlorine only when it is demonstrated on a case-by-case basis that immediate dilution of the effluent within the mixing zone reduces toxicity below lethal concentrations. The Department may on a case-by-case basis establish a zone of immediate dilution if appropriate for other parameters;

(B) Materials that will settle to form objectionable deposits;

(C) Floating debris, oil, scum, or other materials that cause nuisance conditions; and

(D) Substances in concentrations that produce deleterious amounts of fungal or bacterial growths.

[24] Accessible at: https://secure.sos.state.or.us/oard/viewSingleRule.action?ruleVrsnRsn=68770

(b) A point source for which the mixing zone is established may not cause or significantly contribute to any of the following conditions outside the boundary of the mixing zone:

(A) Materials in concentrations that will cause chronic (sublethal) toxicity. Chronic toxicity is measured as the concentration that causes long-term sublethal effects, such as significantly impaired growth or reproduction in aquatic organisms, during a testing period based on test species life cycle. Procedures and end points will be specified by the Department in wastewater discharge permits;

(B) Exceedances of any other water quality standards under normal annual low flow conditions.

(c) The limits of the mixing zone must be described in the wastewater discharge permit. In determining the location, surface area, and volume of a mixing zone area, the Department may use appropriate mixing zone guidelines to assess the biological, physical, and chemical character of receiving waters, effluent, and the most appropriate placement of the outfall, to protect instream water quality, public health, and other beneficial uses. Based on receiving water and effluent characteristics, the Department will define a mixing zone in the immediate area of a wastewater discharge to:

(A) Be as small as feasible;

(B) Avoid overlap with any other mixing zones to the extent possible and be less than the total stream width as necessary to allow passage of fish and other aquatic organisms;

(C) Minimize adverse effects on the indigenous biological community, especially when species are present that warrant special protection for their economic importance, tribal significance, ecological uniqueness, or other similar reasons determined by the Department and does not block the free passage of aquatic life;

(D) Not threaten public health;

(E) Minimize adverse effects on other designated beneficial uses outside the mixing zone.

(d) Temperature Thermal Plume Limitations. Temperature mixing zones and effluent limits authorized under 340-041-0028(12)(b) will be established to prevent or minimize the following adverse effects to salmonids inside the mixing zone:

(A) Impairment of an active salmonid spawning area where spawning redds are located or likely to be located. This adverse effect is prevented or minimized by limiting potential fish exposure to temperatures of 13 degrees Celsius (55.4 Fahrenheit) or more for salmon and steelhead, and 9 degrees Celsius (48 degrees Fahrenheit) or more for bull trout;

(B) Acute impairment or instantaneous lethality is prevented or minimized by limiting potential fish exposure to temperatures of 32.0 degrees Celsius (89.6 degrees Fahrenheit) or more to less than 2 seconds);

(C) Thermal shock caused by a sudden increase in water temperature is prevented or minimized by limiting potential fish exposure to temperatures of 25.0 degrees Celsius (77.0 degrees Fahrenheit) or more to less than 5 percent of the cross section of 100 percent of the 7Q10 low flow of the water body; the Department may develop additional exposure timing restrictions to prevent thermal shock; and

(D) Unless the ambient temperature is 21.0 degrees of greater, migration blockage is prevented or minimized by limiting potential fish exposure to temperatures of 21.0 degrees Celsius (69.8 degrees Fahrenheit) or more to less than 25 percent of the cross section of 100 percent of the 7Q10 low flow of the water body.

(e) The Department may request the applicant of a permitted discharge for which a mixing zone is required, to submit all information necessary to define a mixing zone, such as:

(A) Type of operation to be conducted;

(B) Characteristics of effluent flow rates and composition;

(C) Characteristics of low flows of receiving waters;

(D) Description of potential environmental effects;

(E) Proposed design for outfall structures.

(f) The Department may, as necessary, require mixing zone monitoring studies and/or bioassays to be conducted to evaluate water quality or biological status within and outside the mixing zone boundary;

(g) The Department may change mixing zone limits or require the relocation of an outfall, if it determines that the water quality within the mixing zone adversely affects any existing beneficial uses in the receiving waters.

In addition to the rule language, Oregon has published guidance on the application of mixing zones in NPDES permits.[25]

A review of NPDES permits in early 2006 indicates that 85% of individual permits issued in Oregon have a mixing zone (313 of 366). The permits and fact sheets for facilities (Figure 2-1) that have Al permit limits are available for NOAA to understand how mixing zones have been implemented by ODEQ.

[25] Available at: https://www.oregon.gov/deq/wq/wqpermits/Pages/NPDES-Mixing-Zones.aspx

5. Effects of the Action

5.1 Effects Assessment Methodologies

Biological effects to aquatic listed species were assessed through a hybrid tiered/weight of evidence approach. Please see Figure 5-1 below for a conceptual diagram outlining the decision-making process used to inform final effects determinations.

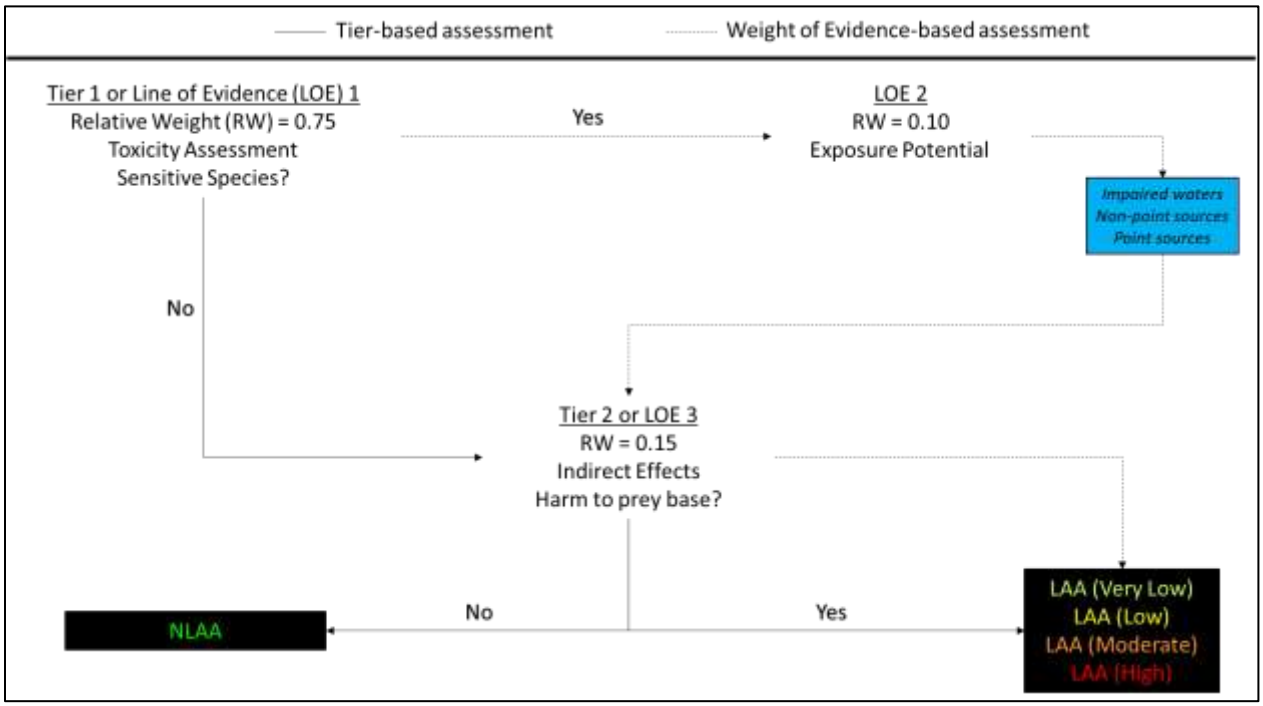


Figure 5-1. Effects assessment methodology to inform final effect determination based on direct and indirect biological effects.

5.1.1 Hybrid Tiering/Weight of Evidence Decision Making Process for the Analysis of Effects of the Action on Listed Species

EPA considered multiple factors regarding the protectiveness of the proposed aluminum aquatic life criteria for Oregon for specific species in order to determine if a species is or is not likely to be adversely affected (LAA or NLAA, respectively) by its action to promulgate criteria for Oregon. The analysis was with respect to the effect of the action on individual members of a species. The multiple factors were considered in a systematic way that either involved a series of binomial decision points (tiering) or a range of options (weight of evidence), depending upon the species sensitivity to Al, in order to develop a ranked based conclusion process for NLAA or LAA determinations in Section 6 (Figure 5-1). Effects determinations are classified as LAA or NLAA but are given a qualifier in this BE of Very Low, Low, Medium, or High depending on the level of information or evidence indicative of an adverse effect on an individual member of a species. The qualifying label is only intended to provide the Services with a rapid way to gauge the relative extent of the information supportive of a LAA determination. The qualifying terms

are not intended to and do not meet a regulatory threshold. The idea for this type of qualification can also be found in the Biological Evaluations produced by the EPA's Office of Pesticide Programs for diazinon, chlorpyrifos, and malathion, although those three BEs contained a more quantitative rendition of WOE than this BE. The driving impetus in using a weight of evidence approach was to transparently rank the relative strength evidence for conclusions EPA makes in Section 6. As a note of context, ecological risk assessment, and perhaps the field of statistics, is moving away from making decisions on thresholds to making decisions based on gradients or probabilities and a weighing of multiple lines of evidence[26]. As the information in this BE (and other similar BEs) involves a level of uncertainty in exposure and effects of chemical substances on T/E species, it is important to recognize this uncertainty and when possible and appropriate, formally and transparently integrate it into the assessment. Although the approach herein does not produce a formal quantitative assessment of risk, it does provide a means to relatively rank the level of evidence for LAA/NLAA for each species assessed. However, the hybrid of tiering and weight of evidence approach was used to demonstrate EPA's higher level of confidence in Tier 1 for the overall effect determinations made in Chapter 6 of this BE and as a way to defer to the protection of the species in the face of uncertainty.

The following tiers or lines of evidence provide information on both the toxic effect on organisms as well as the potential for exposure to aluminum in Oregon surface freshwaters.

- *Tier 1* or *Line of Evidence 1*: Protectiveness of Al criteria magnitude[27] for each listed entity (species or critical habitat) in a range of Oregon water chemistry conditions
- *Line of Evidence 2*: Potential for exposure to Al from point and non-point sources
- *Tier 2* or *Line of Evidence 3*: Indirect effects of the Al criteria on species

5.1.1.1 Tiered/Weight of Evidence Hybrid for the Analysis of Effects

As shown in Figure 5-1, to make an effects determination for each species, EPA proceeded in two distinct paths depending on the sensitivity[28] of a species. For insensitive species, EPA evaluated the indirect effects of the action on species, after which, EPA completed its analysis and effects determination. This Tiered Approach was thus only used for insensitive species. For sensitive species, EPA proceeded to evaluate the weight of the evidence (WoE) for LAA using

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[26] Suter, G. Weight of Evidence in Ecological Assessment. US EPA Office of Research and Development, Washington, DC, EPA100R16001, 2016; INSIGHT: Weight-of-Evidence Best Way to Manage Chemical Risks (https://news.bloombergenvironment.com/environment-and-energy/insight-weight-of-evidence-best-way-to-manage-chemical-risks); Nature Comment. Scientists rise up against statistical significance. 2019. https://www.nature.com/articles/d41586-019-00857-9

[27] The Al criteria employ three elements to protect aquatic life: criteria magnitude, duration, and frequency of exceedance. However, as existing Al toxicity data indicate a limited effect of exposure duration on the manifestation of toxicity and because limited data exist to assess the effect of frequency of exposure on toxicity, EPA focused its assessment on criteria magnitude. The goal of the assessment of criteria magnitude was to evaluate the protectiveness of the authorized criteria concentrations for listed species rather than to evaluate the effect of measured Al concentrations in ambient waters on listed species, as the latter is part of the Environmental Baseline (Section 4).

[28] A "sensitive" species is one in which the relevant toxicity value (chronic Effective Concentration at the xth centile (ECx) or Lethal Concentration at the xth centile (LCx)) is less than the respective criterion value (CMC or CCC) respectively) in more than 0% of Oregon waters. Insensitive species have toxicity values that are never less than the criteria value.

multiple lines of evidence (LOE). Therefore, the overall process is referred to as a hybrid approach and is described in more detail below.

5.1.1.2 Tiered Effects Assessment Approach

The tiered effects assessment begins with analyzing a given species sensitivity to Al and whether the criteria are protective for direct toxicity to a species. Tier 1: If a species chronic or acute toxicity value (EC_5 or LC_5, respectively) was > all corresponding criteria values (CCC or CMC, respectively) across water chemistries relevant to that species (See Section 5.1.2-3), the criteria were determined to be protective for direct toxicity, so EPA proceeded to Tier 2, indirect effects assessment in which toxicity to prey species was considered. If, however, a species toxicity value was < a corresponding criteria value at any water chemistry, the EPA proceeded to Line of Evidence 2 as described below (Section 5.1.1.3).

5.1.1.3 Weight of Evidence Approach

Each line of evidence (LOE) was weighted, on a relative scale from 0-1.0, according to the level of uncertainty associated with each LOE, as well as each LOE's likely contribution to the effects of the action. Reduced uncertainty coupled with a high contribution to the effects of the action tended to increase the relative weight of that LOE. Each LOE and the relative weighting factors are described in Table 5-1. The way in which EPA translated each LOE into an overall WOE assessment is described below.

On a species by species basis, each LOE received a **score** of 0, 1, 2, 3, or 4 depending on the level of information indicating that an adverse effect is likely to occur to an individual member of a Listed species. The score for each LOE was then multiplied by the **relative weight** to obtain a **LOE Score**. LOE Scores were added and divided by 4 (i.e., the maximum possible score indicating high certainty of adverse effects) and multiplied 100% to obtain a **WOE Score** that provides a relative measure of evidence for a LAA determination. The resultant species-specific WOE Score was used to provide information on the level of evidence for the potential that an adverse effect would occur as a result of the proposed action.

- *Line of Evidence 1*: Direct toxicity assessment
    - The percent of samples in which the criteria value (CCC or CMC) was > species toxicity value ($EC_{5,10,15}$ or $LC_{5,10,15}$)
        - This information was used to estimate percent of a population that might be affected (e.g., if 99% of samples have a CCC > EC5, then 4.95% (i.e., 0.99 x 0.05 = 0.0495) of the individuals exposed *at the CCC concentration* may be affected at the 5% effect level (by comparison, if the CCC > EC5 for 1% of samples, this would suggest a 0.05% chance of effect). Therefore, the scoring system is considered conservative for this reason and given that it relies on the assumption that water concentrations in the environment are at the criterion concentration long enough to exert toxicity. As exposure at the criteria concentration is not expected to occur in all circumstances, EPA proceeded to an exposure analysis for LOEs 2 and indirect effect analysis in LOE 3.

    - A LOE Score of 0-4 was determined as follows:
        - 0: 0% EC5 < CCC
        - 1: 1-25% EC5 < CCC
        - 2: 26-50% EC5 < CCC
        - 3: 51-75% EC5 < CCC
        - 4: 76-100% EC5 < CCC
- *Line of Evidence 2*: Exposure potential of a listed species to Al in locations where the Al criteria may be used (e.g., for effluent permits, assessments, total maximum daily load limitations, or site cleanup activities). Further details of LOE 2 calculation procedures are described in Section 5.3. Briefly, scoring was as follows:
    - Geographic overlap of a species range and a potentially impaired water?
        - LOE Score of Yes (1) or No (0)
    - Percent of land use associated with a statistically significant potential to be a source for Al in surface waters within an ecoregion in which a species occurs
        - 0% = 0
        - 1-50% = 1
        - 51-100% = 2
    - Existence of any facilities that could discharge Al into waters within an ecoregion in which a species occurs
        - LOE Score of Yes (1) or No (0)
    - A LOE Score of 0-4 was determined as follows:
        - 0: Values sum to 0
        - 1: Values sum to 1
        - 2: Values sum to 2
        - 3: Values sum to 3
        - 4: Values sum to 4
- *Tier2/Line of Evidence 3*: Indirect effects assessment scoring system.
  A **Tier 2/LOE 3 Score** was calculated to determine the level (i.e., strength) of evidence for LAA by comparing the CCC across Oregon water chemistries to the corresponding 5th centile hazard concentration (HC_5) for prey in the same water chemistry condition. The HC5 was determined from the species prey-based species sensitivity distribution (SSD) (see Section 5.5 for further information). See below for scoring and Table 5-2 for the interpretation of Tier 2/LOE 3 Scores. Tier 2 Scores were used if the WOE approach was not conducted (i.e., when Tier 1 suggested a species would not experience direct adverse effects).

  Tier 2/LOE 3 Scoring Process
    - Scores were 0 or 4 based on the percent of water chemistry conditions in which the HC_5 < CCC
        - 0: 0-99% of water chemistry conditions
        - 4: 100% of water chemistry conditions

The justification for the above scoring categories is based on two factors. (1) The calculation of HC5 values is an imprecise process in that the calculation is greatly affected by the number of species (the N) in a prey-based SSD. Because toxicity data are limited and do not represent all potential prey items, the SSD is at best adequately representative of the toxicity of Al to a species set of prey items. More likely, the SSD is a subset of prey items and is

artificially smaller (low N) than it should be. As the diet of each listed species differs, the available toxicity data differ, and when the N becomes smaller than 59 prey species toxicity values, the HC5 is conservatively reduced, as part of the statistical procedure to calculate the HC5. (2) The SSD to calculate the HC5 is built from EC20 values for each prey genus. EC20 concentrations are statistically robust low, chronic effects concentrations. So, strictly speaking, the HC5 is the Al concentration that would affect 5% of prey items at the 20% effects level. Thus, even if the HC5 is < CCC for 100% of the water chemistry conditions, the overall effect is a 1% reduction in the growth, survival, or reproduction (depending on the nature of the toxicity value selected for each species/genus) of prey items (calculated as 100% x (1* 0.05 * 0.20) = 1% probability for chronic effects to prey species). EPA interprets anything less than 1% to be discountable and insignificant. It is discountable because the minimal effects to the diet are quite likely surmountable through the varied diets of the species evaluated. It is insignificant because the effects can't be measured in the field, given the statistical imprecision at low effects concentrations (e.g., < EC20, HC_5), complexities of population level monitoring of survival, growth, or reproduction across numerous taxa, locations and times as well as the imprecision in field measurements themselves. As this analysis was focused on indirect rather than the direct toxic effects to listed species, the EPA considered this decision to be a reasonable level of caution. Given item #2 above, EPA took the additional step of comparing the CCC to the lowest genus mean chronic value in the prey-based SSD across Oregon water chemistries. This information was used to verify that prey species would not be adversely impacted, given the data but was not used in the scoring system (see Section 5.5 for further information).

- In what percent of Oregon water chemistry conditions is the prey-based HC_5 < CCC?
  - If 0-99%, Tier 2/LOE 3 Score set to 0.
  - If 100%, Tier 2/LOE 3 Score set to 4.

Table 5-1. Relative weights assigned to each line of evidence (LOE) in the weight of evidence calculations.

LOE	Relative Weight	Explanation for Relative Weight
1	0.75	High level of certainty compared to the other LOEs[1]; direct effect on species; action agency has authority to mitigate effects through adjustments to its recommended criteria
2	0.10	Low level of certainty in the analyses given limited data collected for other purposes; uncertain effect on species, given the use of total recoverable versus bioavailable Al concentrations; action agency has some oversight authority outside the scope of this action to mitigate effects through implementation programs (Permits, Assessments, TMDLs)
3	0.15	Medium level of certainty; indirect effect on species; action agency has authority to mitigate effects through adjustments to its recommended criteria

[1] Level of certainty based on relevance and reliability of data used for the assessment

Using the available information EPA calculated WOE Scores for each species as follows (also see Attachment 2):

$$\frac{\sum_{i=1}^{n} LOE\ Score\ \times Relative\ Weight}{4} \times 100 = \text{WOE Score}$$

Where, n = 3

The WOE Score can be interpreted as the Level of Evidence indicating the potential for an adverse effect to an individual member of a species to occur under the assumptions used in the analysis (interpreted as LAA) (Table 5-2). The calculations of scores, LOE Scores, and WOE Scores can be found in Attachments 1 (LOE 2) and 2 (WOE), respectively. The effects determinations based on these analyses can be found in Section 6.2.

Table 5-2. Interpretation of Tier 2 Score and WOE Scores into the level of evidence for LAA

Tier 2 Score	WOE Score	Level of Evidence for LAA	Interpretation
0	0	Insignificant	NLAA
4	1-25	Very Low	LAA
	26-50	Low	LAA
	51-75	Moderate	LAA
	76-100	High	LAA

5.1.2 Tier 1 (Screening Level)/Line of Evidence 1: Direct Biological Effects Assessment Methodology

5.1.2.1 Direct Acute Effects to Freshwater Animals

5.1.2.1.1 Deriving Acute Low Effect Threshold Values (LC_5) and Acute Effect Concentrations (LC_{10} and LC_{15}) in Reference Waters

The protectiveness of the freshwater acute aluminum criterion magnitude was assessed by identifying or estimating acute toxicity values (i.e., LC_{50}) for Oregon aquatic listed species that were then adjusted to represent protective low effect threshold concentrations as described below. Appendix A of EPA's 2018 Aluminum 304(a) aquatic life criteria document (USEPA, 2018) was used to identify high-quality acute toxicity values[29] for listed species in the same

[29] While the 2018 aluminum aquatic life criteria accounts for the effects of site water chemistry and aluminum toxicity, the initial data pull focused on acute toxicity values applicable to one set of water quality conditions. Specifically, all values presented within Appendix A of USEPA (2018) were normalized to the same set of water quality conditions: pH 7, total hardness of 100 mg/L as $CaCO_3$, and dissolved organic carbon (DOC) of 1 mg/L.

order (i.e., Acipenseriformes, Anostraca, Anura, Cypriniformes, Osmeriformes, Salmoniformes) as aquatic listed species occurring in Oregon.

Ideally, species-specific toxicity data for listed species of concern are available to support an acute effects assessment; however, data limitations required use of surrogate toxicity data. Although imperfect, considering surrogate toxicity data at the most phylogenetically-related taxonomic level possible (e.g., least phylogenetic distance) is intended to account for genetically-derived traits conserved across taxa that influence sensitivity to a pollutant (Buchwalter, Sweeney, & Funk, 2008; Garland, Bennett, & Rezende, 2005; LaLone et al., 2016). For example, Raimondo, 2010 #825 suggested interspecies correlations estimates (ICE) were 91% accurate (i.e., predicted values within 5-fold of actual value; see table 1 of (Raimondo, Jackson, & Barron, 2010) when predicting within the same family, with order and class-level predictions being only 5% and 13% less accurate than family-level estimates, respectively. Declines in cross validation success of ICE models became strongly apparent at the phylum level (Raimondo et al., 2010). Similarly, (Slooff, Van Oers, & De Zwart, 1986) observed higher correlation coefficients between effect concentrations of species within the same class than correlation coefficients between species of greater taxonomic distances. Suter, Barnthouse, Breck, Gardner, and O'Neill (1985)demonstrated correlation between paired LC_{50} values decreases at increasing taxonomic distances, with systematic differences in sensitivity becoming evident at the class-level. Based on available literature and collaboration with the U.S. Fish and Wildlife Service and National Oceanic and Atmospheric Administration Fisheries (NOAA Fisheries) at the national-level, EPA Office of Pesticide Programs (OPP) considered order-level surrogate toxicity data to be of "high confidence" in the pilot biological evaluations for diazinon, malathion, and chlorpyrifos (USEPA, 2016a, 2016b). Furthermore, the mode of action (MOA) by which Al causes toxicity is relatively broad, which further supports the use of taxonomic surrogacy to provide toxicity estimates for species lacking acceptable toxicity test data. For example, Raimondo et al. (2010) also constructed MOA-specific ICE models and observed MOA-specific models improved estimation for unrelated species but had no significant improvement for related species. Therefore, EPA considered acute toxicity data at the closest taxonomic level possible to calculate geometric mean acute toxicity values for each species assessed (i.e., LC_{50}).

Geometric mean acute toxicity values at the genus-, family-, and order-level were calculated as the geometric mean of lower taxonomic-level geometric mean values, since these mean values were meant to represent the sensitivity for a particular taxon. Species-specific and surrogate acute toxicity data obtained from EPA's 2018 Aluminum aquatic life criteria (USEPA, 2018) represent sensitivity expressed as a concentration that will acutely affect half of the species population. Acute toxicity data (expressed as an LC_{50}) representative of listed species were, therefore, transformed to an acute minimum effect threshold concentration (i.e., LC_5), which theoretically represents a concentration that could affect 5% of individuals of a listed species population under long-term and continuous exposure conditions. In practicality, however, a species (or surrogate)-specific 5% effect concentration constitutes an effect level that is generally statistically indistinguishable from control organisms (i.e., unexposed to aluminum) and LC_0 values (which are theoretically not quantifiable). For example, studies are often considered acceptable for water quality criteria derivation when control mortality is ≤ 10% because such low-level responses reflect natural mortality. As a result, 5% effect concentrations are considered within the statistical noise of control responses and, therefore, represent an insignificant effect so minor that it is not possible (with the data available in standard toxicity tests with limited

organisms and limited (e.g., 4 - 6) exposure concentrations to meaningfully measure, detect, or evaluate adverse effects at, or below, such levels. For example, LC_5 values and corresponding 95% confidence intervals (CI) from all acceptable acute salmonid toxicity studies that had data to support C-R modeling are described in Table 5-3. These specific curves and their acceptability are described in detail in section 5.2.1.1.2.

Table 5-3. Acute salmonid concentration-response curves from acceptable acute toxicity studies with model score and a description of the confidence intervals around the LC_5 relative to other LC_x values. For a detailed discussion of these models, see section 5.2.1.1.2 and Supplemental Information A.

Curve ID	Model Acceptability Score	Description of LC_5 confidence bounds
Al-Acute-53	3	95% Confidence bounds around the LC_5 and LC_{50} overlapped
Al-Acute-54	2	95% Confidence bounds around the LC_5 and LC_{50} overlapped
Al-Acute-55	2	95% Confidence bounds around the LC_5 and LC_{50} overlapped
Al-Acute-56	3	95% Confidence bounds around the LC_5 and LC_{50} overlapped. 95% Confidence bounds around the LC_5 went into negative numbers.
Al-Acute-66	2 – This model was used quantitatively because of data limitations	95% Confidence bounds around the LC_5 and LC_{15} overlapped. Because this model was used quantitatively, lower effect levels were retroactively calculated and compared to the LC_5. 95% confidence intervals around the LC_5 and LC_1 overlapped (see Figure 5-2, Panel A).
Al-Acute-67	2	95% Confidence bounds around the LC_5 and LC_{20} overlapped.
Al-Acute-68	2 – This model was used quantitatively because of data limitations	95% Confidence bounds around the LC_5 and LC_{20} overlapped. Because this model was used quantitatively, lower effect levels were retroactively calculated and compared to the LC_5. 95% Confidence bounds around the LC_5 and LC_1 overlapped (see Figure 5-2, Panel B).

Overall, there were seven acute salmonid C-R curves available. As an example of the uncertainty associated with a 5% effect level, 95% CIs for four of the seven C-R models overlapped with the 95% CIs from their corresponding LC_{50} values, with the 95% CI around the LC_5 from curve Al-Acute-56 extending into the negative number range. 95% CIs for two of the three remaining C-R models overlapped with the 95% CIs from their corresponding LC_{20} values, with the 95% CI around the LC_5 from curve Al-Acute-66 overlapping with the 95% CI from the corresponding LC_{15} value. Although these comparisons do not specifically address how the LC_5 compares to lower effect levels, they do demonstrate the wide range of uncertainty associated with quantifying low effect levels.

Only curves Al-Acute-68 and Al-Acute-66 were used quantitatively in the acute effect assessment. As a result, only these two curves were recalculated to retroactively determine LC_1 through LC_4 values, in addition to the LC_x values reported in **Supplemental Information A**. LC_1 through LC_5 values for curves Al-Acute-66 and Al-Acute-68 and corresponding 95% CIs are displayed in Figure 5-2 below. For both curves, 95% CIs between LC_5 and corresponding LC_1 values overlapped, with the LC_5 value from curve Al-Acute-68 falling directly within the CI of the corresponding LC_1 value. While LC_5 values are indistinguishable from control responses,

their application as acute low effect threshold values was also selected because use of a five percent toxicity value to represent an acute minimum effect threshold to an individual is consistent with the reasonable and prudent alternatives (RPAs) outlined in a recent biological opinion (NOAA Fisheries, 2012).

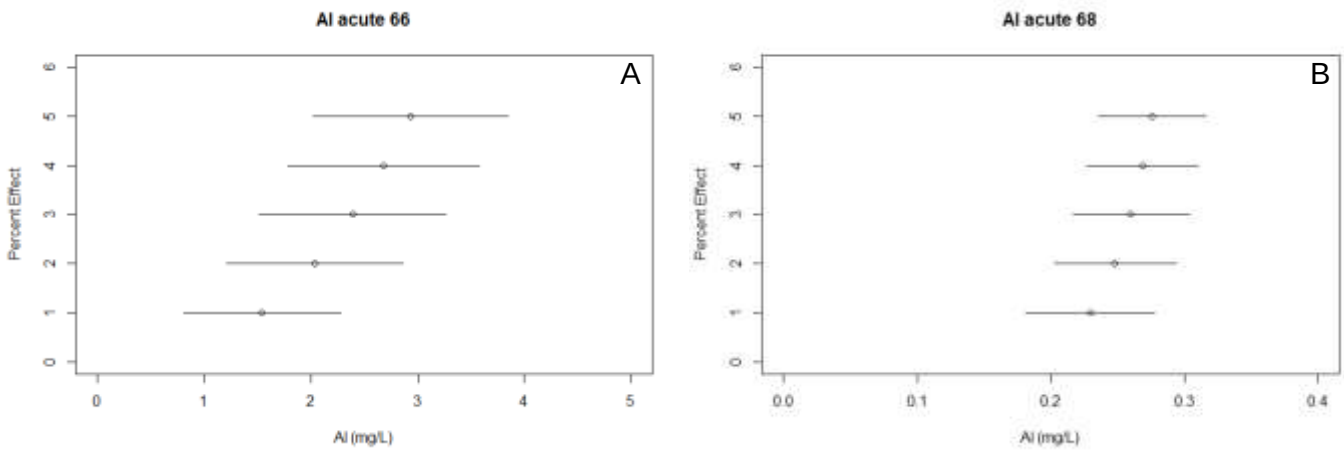

Figure 5-2. Panel A displays LC_1 through LC_5 values as percent effect (Y-axis) relative to aluminum concentration (X-axis) and corresponding 95% confidence interval (CI) for each LCx value as a horizontal line for curve Al-Acute-66. For comparative purposes, the LC_5 was 2.934 mg/L (CI lower = 2.015 mg/L; CI upper = 3.852 mg/L) and the LC_1 was 1.544 mg/L (CI lower = 0.804 mg/L; CI upper = 2.283 mg/L). Panel B displays LC_1 through LC_5 values as percent effect (Y-axis) relative to aluminum concentration (X-axis) and corresponding 95% confidence interval (CI) for each LCx value as a horizontal line for curve Al-Acute-68. For comparative purposes, the LC_5 was 0.275 mg/L (CI lower = 0.235 mg/L; CI upper = 0.316 mg/L) and the LC_1 was 0.2291 mg/L (CI lower = 0.181 mg/L; CI upper = 0.277 mg/L).

Raw empirical acute toxicity data may be used to calculate LC_5 values directly from the concentration-response (C-R) models of the listed species-specific toxicity tests, when available. However, not all published acute tests provide concentration-response data or the concentration-response data may not support robust curve fits at low-level effects. Therefore, species-specific, or surrogate LC_{50} values (which represent listed species 50% acute effect level), were transformed to an acute minimum effect threshold concentration by applying an acute taxonomic adjustment factor (TAF) or an acute mean adjustment factor (MAF)[30]. Acute TAFs were calculated by averaging (geometric mean) the ratios of LC_{50}:LC_5 from acute aluminum toxicity tests (raw data[31] were obtained from studies used to calculate the acute criterion, see Appendix A of (USEPA, 2018) with the most closely related species (e.g., same species, genus, family, or order) to the listed species that is being assessed. Therefore, acute TAFs could be calculated and paired to listed species at the species-, genus-, family-, or order-level. Similar to the process for identifying acute surrogate toxicity values (i.e., LC_{50}), genus-, family-, and order-level acute

[30] See Appendix B.1 for concentration-response (C-R) data obtained from USEPA (2018), C-R modeling methodology, and C-R model scoring to inform the use of quantitatively- and qualitatively-acceptable C-R models to derive acute LC_{50}, LC_{15}, LC_{10}, and LC_5 values.

[31] Raw data used to calculate LC_{50}:LC_5 ratios (as well as LC_{50}:LC_{15} and LC_{50}:LC_{10} ratios) were not normalized to reference water chemistry because the ratios based on the same C-R curve would not be affected by changing water chemistry.

TAFs were calculated as the geometric mean of lower taxonomic-level acute TAFs to ensure adequate representation of all lower-level taxa for a particular taxon (Table 5-4).

When data availability did not allow for the development of an acute TAF within the same order as the species being assessed, EPA considered applying an acute invertebrate or vertebrate TAF (depending on whether the listed species assessed was an invertebrate or vertebrate). The acute invertebrate TAF and the acute vertebrate TAF were calculated as the geometric mean of genus-level LC_{50}:LC_5 ratios of invertebrates and vertebrates, respectively. Vertebrate and invertebrate TAFs were calculated as the geometric mean of the respective genus-level LC_{50}:LC_5 ratios to limit the influence of LC_{50}:LC_5 ratios from species that were overly-represented in a dataset, similar to criteria derivations (Stephen et al., 1985). An acute MAF is typically calculated as the geometric mean of all genus-level TAFs together, irrespective of vertebrate and invertebrate species. Generally, an acute MAF is used to adjust species effect concentrations (i.e., LC_{50}) to low effect threshold concentrations (i.e., LC_5) when: 1) an acute TAF was not available within the same order as the listed species being assessed, and 2) when the acute invertebrate TAF and the acute vertebrate TAF were not significantly different according to a two-sample t-test assuming unequal variances ($\alpha = 0.05$).

In the case of this acute effect assessment for aluminum in freshwater, no quantitatively acceptable LC_{50}:LC_5 ratios for vertebrates were available (Table 5-4). Because quantitatively-acceptable C-R curves were limited and the acute MAF was based on invertebrate species only, EPA considered the application of qualitatively-acceptable curves for vertebrate species, where necessary:

- *Salmonids* - A family-level TAF (i.e., Salmonidae TAF) was calculated from two brook trout LC_{50}:LC_5 ratios that were originally determined to be qualitatively acceptable C-R curve fits (see footnote a of Table 5-4). This salmonid family-level TAF was calculated and used for salmonids instead of the acute MAF because 1) it ensured conclusions for salmonids were based on more taxonomically-related data and 2) the two brook trout LC_{50}:LC_5 ratios used to calculate the salmonid family-level TAF were relatively robust curve fits, only being determined to be qualitatively-acceptable because the Q-Q Plot used in curve diagnostics indicated certain data points within the curves may be over influential (note: all other curve diagnostics indicated curves were quantitatively-acceptable; see curves Cd-Acute-66 and Cd-Acute-68 in Supplemental Information A).
- *Non-salmonid Vertebrate Species* – The only quantitatively-acceptable C-R curves for vertebrate species were based on salmonids and would have provided class to phylum-level adjustment factors (depending on the species assessed) for non-salmonid vertebrate species. Because no quantitatively-acceptable curves were available for non-salmonid vertebrate species within the order level, the MAF (based on invertebrate species only) was retained as the initial approach to transform representative LC_{50} values into LC_5 values (Table 5-4). For non-salmonid vertebrates the MAF was used as the initial approach because it was based on quantitatively-acceptable curves. The application of the MAF to non-salmonid vertebrates was a conservative approach because the MAF (3.881) was greater than the best estimate of an adjustment factor based on vertebrate species (i.e., Salmonidae-TAF = 1.967). Nevertheless, applying invertebrate-based surrogacy to represent vertebrate species solely for the sake of conservatism may be inappropriate considering the physiological differences between vertebrates and invertebrates. Therefore, if the application of the MAF suggested a non-salmonid invertebrate species was acutely sensitive to Al exposures at the acute criterion magnitude (i.e., green sturgeon), then EPA also calculated LC_5 values

through the application of the Salmonidae-TAF to present results that were based less on conservative assumptions and more on phylogenetic relatedness.

Table 5-4. Acute LC_{50}:LC_5 ratios from quantitatively acceptable C-R models with freshwater aquatic organisms used to derive the aluminum acute mean adjustment factor (MAF).

Order	Family	Species	LC_{50} (µg/L)	LC_{05} (µg/L)	LC_{50}: LC_{05}	C-R Curve Label[a]	Citation	Species-level TAF (LC_{50}:LC_{05})	Genus-level TAF (LC_{50}:LC_{05})
Basommatophora	Physidae	*Physa sp.*	29,267	18,969	1.543	Al-Acute 5	(Call et al., 1984)	3.597	3.597
Basommatophora	Physidae	*Physa sp.*	61,966	7,391	8.385	Al-Acute 7	(Call et al., 1984)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	737.9	84.11	8.773	Al-Acute 8	(ENSR, 1992a)	4.189	4.189
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	1,898	290.2	6.541	Al-Acute 9	(ENSR, 1992a)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	2,822	218.9	12.90	Al-Acute 10	(ENSR, 1992a)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	72.07	17.30	4.165	Al-Acute 14	(European A1 Association, 2009)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	792.4	371.3	2.134	Al-Acute 16	(European A1 Association, 2009)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	1,005	423.0	2.378	Al-Acute 17	(European A1 Association, 2009)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	8,801	3,202	2.748	Al-Acute 20	(European A1 Association, 2009)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	10,636	2,071	5.136	Al-Acute 21	(European A1 Association, 2009)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	24,345	2,767	8.797	Al-Acute 27	(European A1 Association, 2009)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	10,766	8,927	1.206	Al-Acute 28	(European A1 Association, 2009)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	309.3	53.37	5.795	Al-Acute 45	(European A1 Association, 2010)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	120.6	31.24	3.859	Al-Acute 48	(European A1 Association, 2010)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	79.27	33.98	2.333	Al-Acute 52	(European A1 Association, 2010)		

[a] Raw empirical acute toxicity data and model output for all C-R models are provided in Appendix B.1. C-R models from two brook trout (*Salvelinus fontinalis*) acute tests (Al-Acute 66 and Al-Acute 68) that were used to derive an acute salmonid TAF are not shown in the table because they were scored to be qualitatively acceptable and were, therefore, excluded from MAF derivation. The Salmonidae family-level acute TAF is 1.967, calculated as the geometric mean of 2.788 (Al-Acute 66) and 1.388 (Al-Acute 68); see Appendix B.1.

Listed species-specific or surrogate LC_{50} values were then divided by an appropriate adjustment factor (i.e., acute TAF or acute MAF depending on data availability) to derive an acute minimum effect threshold concentration. Dividing LC_{50} values by an adjustment factor to identify a minimum-level effect concentration is an approach that is fundamentally similar to acute criteria derivation[32], but is more specific to the chemical and species assessed.

While acute minimum effect threshold estimates for listed species were based on LC_5 values, EPA also calculated LC_{10} and LC_{15} values for Oregon listed species. LC_{10} and LC_{15} values were calculated according to the same process used to determine acute low effect threshold values (i.e., LC_5), except the appropriate adjustment factor (i.e., acute TAF or acute MAF depending on data availability) used to transform acute LC_{50} values into LC_5 values were based on LC_{50}:LC_{10} ratios and LC_{50}:LC_{15} ratios, respectively (determined from the same C-R curves used to identify LC_{50}:LC_5 ratios). Listed species' LC_{10} and LC_{15} values were developed with the intent to illustrate the magnitude of effect for scenarios where the acute criterion magnitude may exceed a listed species acute low effect threshold (i.e., LC_5). Appendix C.1 and Appendix C.2 outline the same acute toxicity C-R curves used to derive acute TAFs and the acute MAF (Table 5-4) but have been refined to show corresponding LC_{15} and LC_{10} estimates as well as LC_{50}:LC_{15} ratios and LC_{50}:LC_{10} ratios, respectively.

5.1.2.1.2 Assessing Acute Effect Concentrations Across Oregon Water Chemistries

Typically, minimum effect threshold concentrations are compared to corresponding criteria magnitudes (i.e., criterion maximum concentration, CMC) under reference water chemistry (i.e., the water chemistry that all acute LC_{50} data were normalized to in Appendix A of (USEPA, 2018). For most aquatic life criteria, results of the comparison in reference waters are applicable to all water chemistries because the criteria magnitudes and species sensitivity vary proportionally across all water chemistries. However, because the acute aluminum criterion is based on Multiple Linear Regression (MLR) models using different vertebrate and invertebrate slopes to model aluminum toxicity across water chemistry, the criterion magnitude and species sensitivity do not always change proportionally across all water chemistries. Consequently, comparing aluminum criteria magnitudes to species sensitivity in reference water chemistry does not produce results that are broadly applicable to all water chemistries.

To account for the difference in criteria magnitudes and individual species sensitivities, EPA obtained an Oregon-specific database of paired hardness, pH, and DOC measurements (see Chapter 4.4.2) paired in space and time to calculate an acute criterion magnitude and renormalize species' acute low effect thresholds (i.e., LC_5), LC_{10} values, and LC_{15} values for each set of paired water chemistry data (n = 19,274). Depending on the species being assessed, the EPA used either the vertebrate or invertebrate MLR equation (USEPA, 2018) to renormalize acute

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[32] The Final Acute Value (FAV; fifth percentile of genus mean acute values) is divided by 2.0 to derive the **Criterion** Maximum Concentration (CMC). The FAV was divided by 2.0 to ensure the CMC is representative of a concentration that will not severely adversely affect populations. To support the development of the 1985 Guidelines, a Federal Register notice published in 1978 (Vol 43, pp. 21506-21518; (USEPA, 1978) outlined the derivation of a generic LC_{50} to LC_{low} (i.e., 0-10% effect) adjustment factor of 0.44 (or divide by 2.27). The adjustment factor of 2.27 was derived as the geometric mean of the quotients of the highest concentration that killed 0-10% of the organisms divided by the LC_{50} in 219 acute toxicity tests. The geometric mean adjustment factor (2.27) outlined in the 1978 Federal Register notice was subsequently rounded to 2.0 in the 1985 Guidelines (Stephen et al., 1985).

toxicity values (i.e., LC_5; LC_{10} and LC_{15}) and then paired each value to its corresponding acute criterion magnitude. Paired acute criterion magnitudes and corresponding listed species acute effect concentrations (i.e., LC_5, LC_{10}, LC_{15}) values could then be compared to determine the number of times, and percentage of samples, an acute criterion magnitude exceeded a species effect concentration. Furthermore, a scatter plot of acute effect concentrations (Y-axis) and corresponding criterion magnitudes (X-axis) was constructed to visually express the comparison. Each point on the regression represents an acute effect concentration and an acute criterion magnitude, both based on the same water chemistry data. Starting at the origin, a linear (1:1) line was overlaid on the scatter plot to indicate points that occur above or below the linear line. Any point below the linear line symbolizes a scenario where the acute criterion magnitude exceeded the corresponding acute low effect concentration. For each species, the comparisons between acute criterion magnitudes and corresponding acute low effect concentrations (i.e., LC_5, LC_{10}, and LC_{15}) were made only using water quality data collected from level-III ecoregions that contained the species (i.e., its native range) being assessed. Additionally, for each species with a Distinct Population Segment (DPS) or Evolutionarily Significant Unit (ESU), the comparison between acute criterion magnitudes and acute effect concentration (LC_5, LC_{10}, LC_{15}) were further parsed out by individual DPSs or ESUs. Tables and scatter plots comparing acute effect concentrations and criterion magnitudes for an individual species DPS or ESU only included water chemistry data collected from level-III ecoregions for the particular ESU or DPS that was being assessed.

A species was considered to not be affected through direct acute effects of aluminum exposure if the species did not occur in any ecoregions where the CMC was greater than that species corresponding LC_5. For such species, direct biological effects of chronic exposures were further assessed before determining whether or not a refined exposure assessment was necessary before assessing indirect biological effects (i.e., Tier 2/LOE 3 assessment) and making final effect determinations

A species was considered to be affected through direct acute effects of aluminum exposure if the species did occur in any ecoregions where the CMC was more than that species corresponding LC_5 in any of the water chemistry scenarios. For such species, the potential for chronic biological effects were then assessed before a refined exposure assessment, based on point and non-point source dischargers, was conducted to fully inform the potential for Al exposure and resultant direct biological effects on listed species. Indirect biological effects (i.e., Tier 2/LOE 3 assessment) were then assessed before making a final effect determination. That is, final effect determinations were based on the potential for direct biological effects (i.e., Tier 1/LOE 1 and LOE 2) and indirect biological effects (Tier 2/LOE 3) through a weight of evidence approach.

5.1.3 Tier 1 (Screening Level)/Line of Evidence 1: Direct Biological Effects Assessment Methodology

5.1.3.1 Direct Chronic Effects to Freshwater Animals

5.1.3.1.1 Deriving Chronic Low Effect Threshold Values (EC_5) and Chronic Effect Concentrations (EC_{10} and EC_{15}) in Reference Waters

The protectiveness of the freshwater chronic aluminum criterion magnitude was assessed by identifying or estimating chronic toxicity values (i.e., EC_{20}) for Oregon aquatic listed species that were then adjusted to represent protective low effect threshold concentrations as described below. Appendix C of EPA's 2018 Aluminum 304(a) aquatic life criteria (USEPA, 2018) was used to identify high-quality chronic toxicity values[33] for listed species in the same order (i.e., Acipenseriformes, Anostraca, Anura, Cypriniformes, Osmeriformes, Salmoniformes) as aquatic listed species occurring in Oregon.

Ideally, species-specific toxicity data for listed species of concern are available to support a chronic effects assessment; however, data limitations required use of surrogate toxicity data. EPA considered chronic toxicity data at the closest taxonomic level possible to calculate geometric mean chronic toxicity values for each species assessed (i.e., EC_{20}). Considering surrogate toxicity data at the most phylogenetically-related taxonomic level possible accounts for the genetically-derived traits conserved across taxa that directly influence sensitivity to a pollutant (see discussion of taxonomic surrogacy and references cited in Section 5.1.2.1.1). Geometric mean chronic toxicity values at the genus-, family-, and order-level were calculated as the geometric mean of lower taxonomic-level geometric mean values, since these mean values were meant to represent the sensitivity for a particular taxon. Species-specific and surrogate chronic toxicity data obtained from EPA's 2018 Aluminum aquatic life (USEPA, 2018) represent sensitivity expressed as a concentration that will affect 20% of the species population from chronic exposures.

Unlike acute criteria derivation, which typically uses a generic LC_{50} to LC_{low} adjustment factor (i.e., 2.0^7; (Stephen et al., 1985), chronic criteria are based directly on chronic effect concentrations (e.g., EC_{20}) and do not incorporate a generic EC_x to EC_{low} adjustment factor. However, a concentration that potentially results in chronic effects to 20% of a listed species population may not be considered acceptable for listed species under the Endangered Species Act. Therefore, a similar convention used for the acute assessment methodology was applied to the chronic effects assessment methodology to determine a chronic minimum effect threshold concentration (i.e., EC_5) from chronic toxicity values (i.e., EC_{20} values). Chronic toxicity data (expressed as EC_{20}) representative of listed species were, therefore, transformed to a chronic minimum effect threshold concentration (i.e., EC_5), which represents a concentration that theoretically affects 5% of individuals of a listed species population under long-term and continuous exposure conditions. In practicality, however, a species (or surrogate)-specific 5%

[33] While the 2018 aluminum aquatic life criteria accounts for the effects of site water chemistry and aluminum toxicity, the initial data pull focused on chronic toxicity values applicable to one set of water quality conditions. Specifically, all values presented within Appendix C of USEPA (2018) are normalized to the same set of water quality conditions: pH 7, total hardness of 100 mg/L as $CaCO_3$, and dissolved organic carbon (DOC) of 1 mg/L

effect concentration constitutes an effect level that is generally statistically indistinguishable from control organisms (i.e., unexposed to aluminum) and LC_0 values (which are theoretically not quantifiable) and constitutes an insignificant effect.

Following the same fundamental process described the acute effect assessment methodology, representative listed species EC_{20} values were transformed into EC_5 values. Briefly, EC_{20} values were transformed to represent lower effect concentrations through the application of a chronic taxonomic adjustment factor (TAF) or a chronic mean adjustment factor (MAF[34]; see Table 5-5)[35]. When data availability did not allow for the development of a chronic TAF within the same order as the species being assessed, EPA considered applying a chronic invertebrate or vertebrate TAF (depending on whether the listed species assessed was an invertebrate or vertebrate), in the same manner as to the acute effect assessment methodology described above. While the invertebrate TAF was based eight EC_{20}:EC_5 ratios, the vertebrate TAF was only based on a single quantitatively acceptable EC_{20}:EC_5 ratio (fathead minnow, *Pimephales promelas*, Order Cypriniformes), which did not allow for a two-sample t-test. Consequently, EPA chose to use the chronic MAF over an invertebrate/vertebrate TAF for species where data availability did not permit the use of a TAF at or below the order-level (Table 5-5). Similar to the acute effect assessment, quantitatively-acceptable vertebrate curves were relatively limited, and EPA considered the application of qualitatively-acceptable curves for vertebrate species, where necessary:

- *Salmonids* - No quantitatively acceptable C-R models were available for salmonids. For species within the Family Salmonidae, a family-level TAF was calculated from one brook trout EC_{20}:EC_5 ratio that was originally determined to be only qualitatively acceptable based on C-R curve fits (see footnote a of Table 5-5). This salmonid family-level TAF was calculated and used in favor of the chronic MAF because 1) it ensured conclusions for salmonids were based on more taxonomically-related data and 2) the brook trout EC_{20}:EC_5 ratio used to calculate the salmonid family-level chronic TAF was based on a relatively robust curve fits, only being determined to be qualitatively acceptable because the Q-Q Plot used in curve diagnostics indicated certain data points within the curve may be over influential (note: all other curve diagnostics indicated curves were quantitatively acceptable; see curve Al-Chronic-9 in Appendix B.2 and Supplemental Information A). Moreover, the chronic salmonid family-level TAF (1.696) is greater than the chronic vertebrate TAF (1.668) or chronic MAF (1.403), resulting in lower EC_5 estimates.
- *Non-salmonid Vertebrate Species* – Unlike the acute MAF, the chronic MAF contained a vertebrate-based C-R curve that was quantitively acceptable, therefore, EPA did not conduct further analyses for non-salmonid vertebrate species that required the application of the chronic MAF (i.e., Oregon spotted frog, green sturgeon).

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[34] See Appendix B.2 for concentration-response (C-R) data obtained from USEPA (2018), C-R modeling methodology, and C-R model scoring to inform the use of quantitatively and qualitatively acceptable C-R models to derive chronic EC_{20}, EC_{15}, EC_{10}, and EC_5 values.

[35] Raw data used to calculate EC_{20}:EC_5 ratios (as well as EC_{20}:EC_{15} and EC_{20}:EC_{10} ratios) were not normalized to reference water chemistry because the ratios based on the same C-R curve would not be affected by changing water chemistry.

Table 5-5. Chronic EC_{20}:EC_5 ratios from quantitatively acceptable C-R models with freshwater aquatic organisms used to derive the aluminum chronic mean adjustment factor (MAF).

Order	Family	Species	EC_{20} (µg/L)	EC_{05} (µg/L)	EC_{20}: EC_{05}	C-R Curve Label[a]	Citation	Species-level TAF (EC_{20}:EC_{05})	Genus-level TAF (EC_{20}:EC_{05})
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	263.6	166.8	1.580	Al-Chronic 1	(CECM, 2014; Robert W Gensemer et al., 2018)	1.685	1.685
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	931.3	563.1	1.654	Al-Chronic 4	(CECM, 2014; Robert W Gensemer et al., 2018)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	833.1	709.6	1.174	Al-Chronic 5	(CECM, 2014; Robert W Gensemer et al., 2018)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	1,022	614.9	1.661	Al-Chronic 6	(CECM, 2014; Robert W Gensemer et al., 2018)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	808.1	394.6	2.048	Al-Chronic 11	(ENSR, 1992b)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	596.8	449.6	1.327	Al-Chronic 12	(ENSR, 1992b)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	185.0	111.8	1.655	Al-Chronic 17	(European A1 Association, 2010; Robert W Gensemer et al., 2018)		
Diplostraca	Daphniidae	*Ceriodaphnia dubia*	438.8	154.8	2.835	Al-Chronic 21	(European A1 Association, 2010; Robert W Gensemer et al., 2018)		
Cypriniformes	Cyprinidae	*Pimephales promelas*	6,428	5,503	1.168	Al-Chronic 24	(Kimball, 1978)	1.168	1.168

[a] Raw empirical chronic toxicity data and model output for all C-R models are provided in Appendix B.2.The C-R model from a brook trout (*Salvelinus fontinalis*) chronic test (Al-Chronic-9) that was used to derive a chronic salmonid EC_{20} to EC_5 adjustment factor is not shown in the table because the model was scored to be qualitatively acceptable and was, therefore, excluded from chronic MAF derivation. The Salmonidae family-level EC_{20} to EC_5 chronic adjustment factor is 1.696, based on the brook trout EC_{20}:EC_5 ratio (Al-Chronic 9; see Appendix B.2 and Supplemental Information A).

Listed species-specific or surrogate EC_{20} values were then divided by an appropriate adjustment factor (i.e., chronic TAF or chronic MAF depending on data availability) to derive a chronic minimum effect threshold concentration. Chronic minimum effect threshold concentrations were then compared to the corresponding criterion magnitude (i.e., criterion continuous concentration, CCC) to assess potential adverse effects of aluminum exposures at the chronic criterion concentration.

Similar to the acute analyses, EPA also calculated EC_{10} and EC_{15} values for Oregon listed species. Listed species' EC_{10} and EC_{15} values were developed, in addition to EC_5 values, to illustrate the magnitude of effect for scenarios where the chronic criterion magnitude may exceed a listed species chronic low effect threshold (i.e., EC_5). Appendix C.3 and Appendix C.4 outline the same chronic toxicity C-R curves used to derive chronic TAFs and the chronic MAF (Table 5-10) but have been refined to show corresponding EC_{15} and EC_{10} estimates as well as EC_{20}:EC_{15} ratios and EC_{20}:EC_{10} ratios, respectively.

5.1.3.1.2 Assessing Chronic Effect Concentrations Across Oregon Water Chemistries

Typically, minimum effect threshold concentrations are compared to corresponding criteria magnitudes (i.e., criterion continuous concentration, CCC) under reference water chemistry (i.e., the water chemistry that all chronic EC_{20} data were normalized to in Appendix C of (USEPA, 2018). For most aquatic life criteria, results of the comparison in reference waters are applicable to all water chemistries because the criteria magnitudes and species sensitivity vary proportionally across all water chemistries. However, because the chronic aluminum criterion is based on Multiple Linear Regression (MLR) models using different vertebrate and invertebrate slopes to model aluminum toxicity across water chemistry, the criterion magnitude and species sensitivity do not always change equally across all water chemistries. Consequently, comparing aluminum criteria magnitudes to species sensitivity in reference water chemistry does not produce results that are broadly applicable to all water chemistries. Following the same methodology outlined in Section 5.1.1.2, listed species chronic effect thresholds (e.g., EC_5, EC_{10}, EC_{15} values) were renormalized and compared the corresponding chronic criterion magnitude in the same water chemistry scenarios.

A species was considered to not be affected through direct chronic effects of aluminum exposure if the species did not occur in any ecoregions where the CCC was less than that species corresponding EC_5. For such species (that were also tolerant to acute exposures), indirect biological effects (i.e., Tier 2/LOE 3 assessment) were further assessed before making a final effects determination.

A species was considered to be affected through direct chronic effects of aluminum exposure if the species did occur in any ecoregions where the CCC was more than that species corresponding EC_5 in any water chemistry scenarios. For such species, a refined exposure assessment, based on point and non-point source dischargers, was conducted to fully inform the potential for Al exposure and resultant direct biological effects on listed species. Indirect biological effects (i.e., Tier 2/LOE 3 assessment) were then assessed before making a final effect determination. Final effect determinations were based on based on direct biological effects (i.e., Tier 1/LOE 1 as well as LOE 2 and indirect biological effects (Tier 2/LOE 3) through a weight of evidence approach.

5.1.4 Tier 2/LOE 3: Indirect Effects Assessment

Following assessment of direct biological effects, EPA considered and assessed potential indirect biological effects of the water quality standard action on listed aquatic organisms in Oregon. To assess potential indirect effects, EPA considered potential effects to listed animal prey items as well as listed species diets and feeding behaviors.

5.2 Aluminum Tier 1/LOE 1 Screening-Level Direct Biological Effects Assessment

5.2.1 Chum Salmon (Oncorhynchus keta)

5.2.1.1 Chum Salmon Acute Aluminum Effects Assessment

5.2.1.1.1 Identifying Chum Salmon Acute Aluminum Data

Species-level acute toxicity data were not available for chum salmon. The only acute toxicity data available for members of the genus *Oncorhynchus* were based on rainbow trout (see Appendix A.1). The rainbow trout species mean acute value (SMAV; 3,312 µg/L total aluminum, normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) was therefore used to represent the sensitivity of chum salmon to acute aluminum exposures (Table 5-6). The *Oncorhynchus* genus mean acute value (GMAV) was based on eight LC_{50} values from flow-through, measured acute toxicity tests conducted on juvenile rainbow trout tested under various water chemistry conditions (see Gundersen, Bustaman, Seim, and Curtis (1994) in Appendix A.1). Four of the eight rainbow trout LC_{50} values used to determine the *Oncorhynchus* GMAV were non-definitive LC_{50} values ranging from >5,164 to >7,216 µg/L total aluminum (normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L). These non-definitive LC_{50} values are represented as greater than values because study treatment concentrations were not sufficiently high enough to generate 50% mortality. As a result, their inclusion in the *Oncorhynchus* GMAV results in a conservative *Oncorhynchus* GMAV estimate.

Table 5-6. Data used to calculate the *Oncorhynchus* genus mean acute value (GMAV), representative of chum salmon sensitivity to acute aluminum exposure.

Family	Species	SMAV (µg/L)[a]	GMAV (µg/L)[a]
Salmonidae	Chum salmon, *Oncorhynchus keta*	N/A	3,312
Salmonidae	Rainbow trout, *Oncorhynchus mykiss*	3,312	

[a] All values expressed as total aluminum, normalized to pH 7, DOC of 1 mg/L and 100 mg/L hardness as $CaCO_3$; normalized using MLR equations identified in (USEPA, 2018);.

N/A: not available.

EPA also evaluated other acute toxicity data from several tests with rainbow trout and Atlantic salmon to provide comparative information (Table 5-7 provides a summary of these tests). The normalized effect concentrations from these tests were not used for acute criteria derivation primarily due to insufficient exposure durations for salmonid species or low pH of test waters. Qualitatively acceptable toxicity data that were not used for criteria derivation, but provide supportive auxiliary information, are reported in Appendix H of USEPA (2018). Table 5-7 only includes a subset of all the salmonid data included in Appendix H of USEPA (2018). Data in Table 5-7 were specifically selected for discussion here because these studies adequately reported water chemistry data to populate the multiple linear regression model to normalize effect concentrations to the same reference water chemistry (pH = 7.0; hardness = 100 mg/L; DOC = 1 mg/L). These data, taken together, appear to support the *Oncorhynchus* GMAV (3,312 µg/L total aluminum at pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) that is representative and protective of all anadromous salmonids assessed in this evaluation.

Table 5-7. Summary of data from other tests representative of the chronic aluminum sensitivity of chum salmon (from Appendix H in EPA's 2018 Aluminum Criteria Document).

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	DOC (mg/L)	pH	Effect	Concentration (µg/L)	Reference	Normalized Concentration (µg/L)[a]
Rainbow trout (juvenile), *Oncorhynchus mykiss*	Aluminum sulfate	96 hr	25	1.6	8	40% dead	50,000	Hunter et al. 1980	18,009
Rainbow trout (juvenile), *Oncorhynchus mykiss*	Aluminum sulfate	42 hr	25	1.6	8.5	100% dead	50,000	Hunter et al. 1980	10,025
Rainbow trout (juvenile), *Oncorhynchus mykiss*	Aluminum sulfate	42 hr	25	1.6	9	100% dead	50,000	Hunter et al. 1980	5,581
Rainbow trout (3.5 g), *Oncorhynchus mykiss*	Aluminum sulfate	6 d	11.2	1.6	5.09-5.31	LC50	175	Orr et al. 1986	2,837
Rainbow trout (alevin, 23-26 dph), *Oncorhynchus mykiss*	Aluminum sulfate	6 d	10.3	2	5.8	LC50	>1,050	Hickie et al. 1993	6,738
Rainbow trout (alevin, 16-19 dph), *Oncorhynchus mykiss*	Aluminum sulfate	6 d	10.3	2	4.9	LC50	88	Hickie et al. 1993	1,851
Rainbow trout (alevin, 23-26 dph), *Oncorhynchus mykiss*	Aluminum sulfate	6 d	10.3	2	4.9	LC50	91	Hickie et al. 1993	1,914
Atlantic salmon (juvenile, 1.4 g), *Salmo salar*	Aluminum sulfate	5 d	10.6	0.5	4.47	LC50	259	Roy and Campbell 1995	39,797
Atlantic salmon (juvenile, 1.4 g), *Salmo salar*	Aluminum sulfate	5 d	10.6	0.5	4.42	LC50	283	Roy and Campbell 1995	47,329
Atlantic salmon (juvenile, 1.4 g), *Salmo salar*	Aluminum sulfate	5 d	10.6	0.5	4.83	LC50	121	Roy and Campbell 1995	10,210

Species	Chemical	Duration	Total Hardness (mg/L as $CaCO_3$)	DOC (mg/L)	pH	Effect	Concentration (µg/L)	Reference	Normalized Concentration (µg/L)[a]
Atlantic salmon (juvenile, 1.4 g), *Salmo salar*	Aluminum sulfate	5 d	10.6	0.5	5.26	LC50	54	Roy and Campbell 1995	2,209
Atlantic salmon (juvenile, 1.4 g), *Salmo salar*	Aluminum sulfate	5 d	10.6	0.5	5.24	LC50	51	Roy and Campbell 1995	2,170
Atlantic salmon (juvenile, 6.8 g), *Salmo salar*	Aluminum sulfate	96 hr	10.6	0.5	4.86	LC50	75.54	Roy and Campbell 1995	6,041
Atlantic salmon (juvenile, 1.8 g), *Salmo salar*	Aluminum sulfate	96 hr	10.6	0.5	4.99	LC50	79.60	Roy and Campbell 1997	5,121
Atlantic salmon (juvenile, 1.8 g), *Salmo salar*	Aluminum sulfate	96 hr	10.6	0.5	4.96	LC50	124.1	Roy and Campbell 1997	8,396
Geometric Mean of all Normalized Concentrations (µg/L)									6,519.51

[a] All values expressed as total aluminum, normalized to pH 7, DOC of 1 mg/L and 100 mg/L hardness as $CaCO_3$; normalized using MLR equations identified in USEPA (2018).

5.2.1.1.2 Deriving Chum Salmon Acute Adjustment Factors

Raw empirical acute toxicity data for rainbow trout were fit to C-R models using R statistical software (version 3.5.2; see Appendix E.1 for curve fitting methodology and C-R model scoring methodology) to calculate LC_{50} and corresponding LC_5, LC_{10}, and LC_{15} values for four of the eight toxicity tests conducted by Gundersen et al. (1994). Four tests reported by Gundersen et al. (1994) had non-definitive LC_{50} values and C-R models could, therefore, not be generated. Two of the remaining four models were scored as being unacceptable model fits (see Al-Acute-53 and Al-Acute-56 in Appendix B.1 for raw empirical toxicity data and model output). The final two models from Gundersen et al. (1994) were scored as being qualitatively acceptable (see Al-Acute-54 and Al-Acute-55 in Appendix B.1). Therefore, no quantitatively acceptable C-R models were available for species within the genus *Oncorhynchus* to derive an acute genus-level TAF. Moreover, no acute TAFs based on quantitatively acceptable C-R curves were available for any member of the Order Salmoniformes or any other vertebrate species (see Table 5-8).

Due to the lack of TAFs representative of vertebrates and salmonid species, EPA re-evaluated all salmonid C-R models that were considered to be qualitatively acceptable C-R models (Appendix B.1). C-R models from three brook trout (*Salvelinus fontinalis*) acute toxicity tests reported by Tandjung (1982); Al-Acute-66, Al-Acute-67, Al-Acute-68) were classified as qualitatively acceptable in addition to the two rainbow trout C-R models that were scored qualitatively acceptable (Al-Acute-54 and Al-Acute-55). Of these five qualitatively acceptable acute C-R models for salmonid species, curves Al-Acute-54 and Al-Acute-55 were not further considered because both curves had limited partial effects and overly influential data points. Curve Al-Acute-67 was also not further considered because of a lack of low-level effect data points (lowest treatment level resulted in 30% mortality) to inform responses between 5% and 15% mortality. Curves Al-Acute-66 and Al-Acute-68 (see Appendix B.1; both curves based on acute brook trout tests) were retained to derive a Salmonidae family-level TAF that was used to represent chum salmon. Curves Al-Acute-66 and Al-Acute-68 were retained because these fitted curves only had minor issues with curve statistics relative to the other acute salmonid C-R curves that were considered qualitatively acceptable. For example, curves Al-Acute-66 and Al-Acute-68 both have adequate partial effects, contain low-level response, but were determined to be qualitatively acceptable only because both models had several data points that may be overly influential (as indicated by QQ plots, see Supplemental Information A).

Table 5-8. Acute point estimates and corresponding LC_{50}:LC_{15}, LC_{50}:LC_{10} and LC_{50}:LC_5 ratios from two qualitatively acceptable acute concentration-response (C-R) curves that were retained for quantitative use due to limited salmonid acute C-R data. Raw data used to calculate LC_{50}:LC_5 ratios (as well as LC_{50}:LC_{15} and LC_{50}:LC_{10} ratios) were not normalized to reference water chemistry because the ratios based on the same C-R curve would not be affected by changing water chemistry.

Species	C-R Curve Label	LC_{50} (µg/L)	LC_{15} (µg/L)	LC_{10} (µg/L)	LC_5 (µg/L)	LC_{50}:LC_{15} Ratio	LC_{50}:LC_{10} Ratio	LC_{50}:LC_5 Ratio
Brook trout, Salvelinus fontinalis	Al-Acute-66	8,170	4,620	3,890	2,930	1.770	2.100	2.788
Brook trout, Salvelinus fontinalis	Al-Acute-68	383.1	315.7	299.8	275.9	1.213	1.277	1.388

Taken together, the geometric mean LC_{50}:LC_5 ratio from curves Al-Acute-66 and Al-Acute-68 (see Table 5-8) is 1.967, which serves as the Salmonidae family-level TAF (i.e., LC_{50}:LC_5 adjustment factor) representative of the chum salmon. Similarly, the geometric mean of the LC_{50}:LC_{10} ratios from curves Al-Acute-66 and Al-Acute-68 (see Table 5-8) is 1.638 and the geometric mean of the LC_{50}:LC_{15} ratios from curves Al-Acute-66 and Al-Acute-68 is 1.466.

5.2.1.1.3 Calculating Chum Salmon Acute Aluminum Effect Threshold Values

The Salmonidae family-level TAF (i.e., LC_{50}:LC_5 adjustment factor) as well as the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were applied, as a denominator, to the *Oncorhynchus* GMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value representative of the chum salmon:

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) = **LC_5** = 3,312 µg/L / 1.967 = **1,683 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) = **LC_{10}** = 3,312 µg/L / 1.638 = **2,022 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) = **LC_{15}** = 3,312 µg/L / 1.466 = **2,260 µg/L***

5.2.1.1.4 Chum Salmon: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude (which uses the lower of the vertebrate or invertebrate value for each set of water chemistry) relative to vertebrate species sensitivity across water chemistry, chum salmon acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate an acute criterion magnitude and corresponding chum salmon acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and chum salmon acute effect threshold data (i.e., LC_5, LC_{10}, and LC_{15}) were compared with the chum salmon Columbia River ESU. The comparison between acute effect threshold data and criterion magnitudes for the chum salmon Columbia River ESU only considered the specific water quality data that were collected from level-III ecoregions within the Columbia River ESU. To visually display the comparison, criterion magnitudes (X-axis) and corresponding acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1)

line for the chum salmon Columbia River ESU (see Figure 5-3[36]). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, LC_{10}, or LC_{15}) and are therefore protective at low effect levels, and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and chum salmon Columbia River ESU acute effect threshold data are reported in Table 5-9.

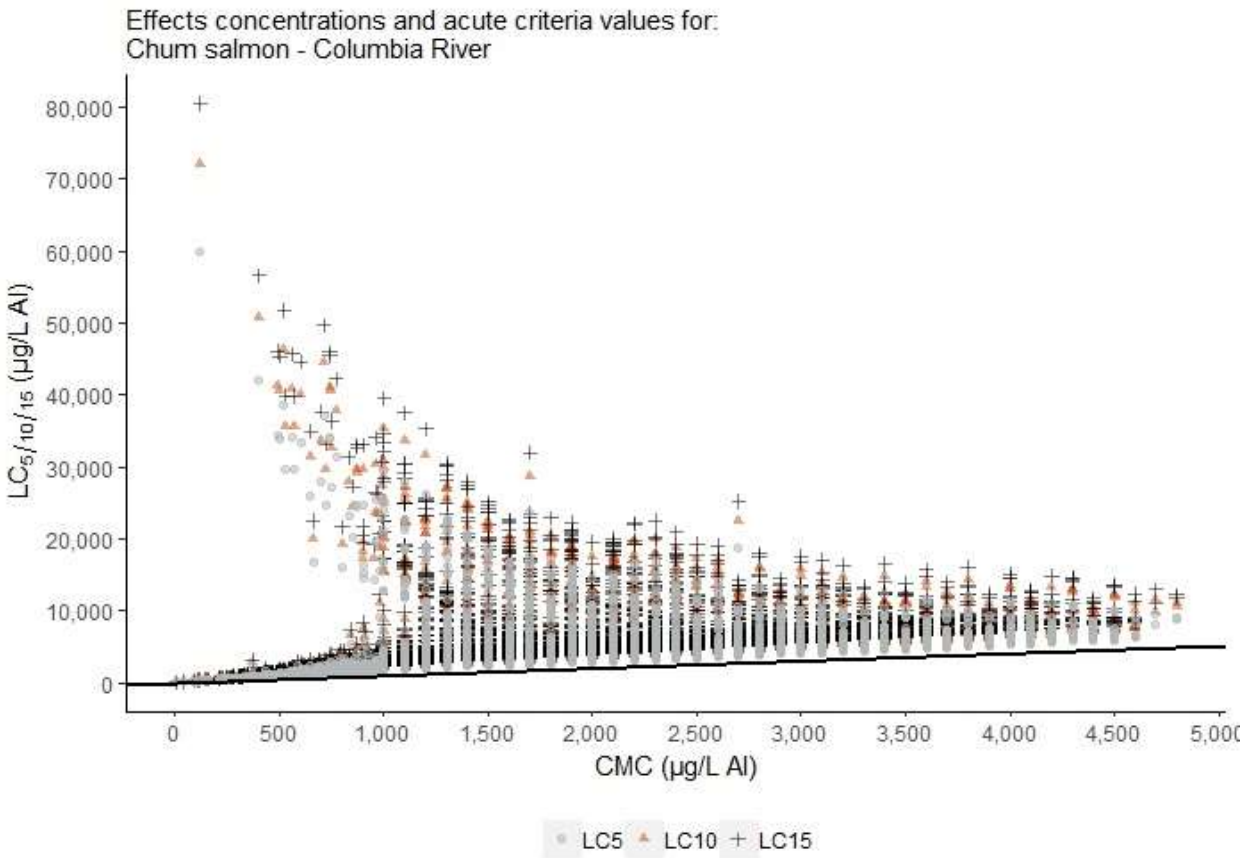

Figure 5-3. Paired chum salmon acute effect concentrations and acute criterion values in Oregon ecoregions containing the chum salmon Columbia River Evolutionarily Significant Unit (ESU).

[36] The structure and interpretation of Figure 5-3 is fundamentally similar all acute figures in this effect assessment. Individual data points in the scatter plot represent acute criterion magnitudes (X axis) and corresponding acute effect concentrations (Y axis) for the listed species calculated from the same water chemistry data (i.e., pH, hardness, DOC). Each acute criterion concentration is compared to three effect levels of 5%, 10%, and 15%, denoted by the light gray, orange, and dark gray colors, respectably. The linear (1:1) line beginning at the origin represents the point at which an individual point has either a greater X axis value (acute criterion concentration) or a greater Y axis value (LC_5, LC_{10}, or LC_{15}). Data points above the line indicate scenarios where water chemistry results in an acute criterion concentration that is less than the corresponding acute effect level, while data points below the line indicate scenarios where water chemistry results in an acute criterion concentration that is greater than the corresponding acute effect level.

Table 5-9. The number and percentage of occurrences where the acute criterion magnitude exceed chum salmon LC_5, LC_{10}, and LC_{15} values for each chum salmon Evolutionarily Significant Unit in Oregon (ESU). Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing the range of each ESU. Aggregate Effect Level (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (mortality) at each effects concentration.

Evolutionarily Significant Unit	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}			CMC > LC_{15}		
Name		n	%	[1]AEL	n	%	AEL	n	%	AEL
Columbia River	14,735	0	0	0%	0	0	0%	0	0	0%

1, AEL was estimated by multiplying the lethal concentration percentile (5, 10, 15th) by the percent of samples in which the criteria value was higher than the toxicity value (i.e., 100*(0.05 or 0.10 or 0.15 x 0.00) = 0.00% probability of adverse effects occurring in exposed fish.

5.2.1.1.5 Chum Salmon: Tier I Acute Assessment Conclusion

The acute aluminum CMC was less than the chum salmon acute low effect threshold (i.e., LC_5) under all water chemistry scenarios in Oregon ecoregions that overlap with chum salmon Columbia River ESU (Table 5-9). The LC_5 range was 36 to 59,906 µg/L and the CMC range was 4.8 to 4,800 µg/L. As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for chum salmon based on the acute criterion.

5.2.1.2 Chum Salmon Chronic Aluminum Effects Assessment

5.2.1.2.1 Identifying Chum Salmon Chronic Aluminum Data

Species-level chronic aluminum toxicity data were not available for chum salmon or any other members within the genus *Oncorhynchus*. Therefore, chronic toxicity data for members of the Family Salmonidae were used to estimate the Salmonidae family mean chronic value (FMCV) representative of the chum salmon (Table 5-10). The Salmonidae FMCV was based on three early-life stage (ELS) toxicity tests (one test with the Atlantic salmon, *Salmo salar*, and two tests with the brook trout, *Salvelinus fontinalis*; see Appendix A.2 of this document). The FMCV (526.5 µg/L total aluminum) was the geometric mean of 434.4 and 638.2 µg/L total aluminum (i.e., the SMCVs for Atlantic salmon and brook trout, respectively; normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L).

Table 5-10. Data used to calculate the Salmonidae family mean chronic value (FMCV), representative of the chronic aluminum sensitivity of chum salmon.

Family	Species	SMCV (µg/L)[a]	GMCV (µg/L)[a]	FMCV (µg/L)[a]
Salmonidae	Chum salmon, *Oncorhynchus keta*	N/A	N/A	526.5
Salmonidae	Atlantic salmon, *Salmo salar*	434.4	434.4	
Salmonidae	Brook trout, *Salvelinus fontinalis*	638.2	638.2	

[a] All values expressed as total aluminum, normalized to pH 7, DOC of 1 mg/L and 100 mg/L hardness as $CaCO_3$; normalized using MLR equations identified in (USEPA, 2018)

N/A: not available.

EPA also evaluated other chronic toxicity data from several tests with early-life stages of golden, cutthroat, and rainbow trout to provide comparative information (Table 5-11 provides a summary of these tests). The normalized effect concentrations from these tests were not used for chronic criteria derivation primarily due to insufficient exposure durations for salmonid species, or the use of inappropriate endpoints for the chronic test result to be used for criteria derivation (e.g., EC_{50} or LC_{50}). Qualitatively acceptable toxicity data that were not used for criteria derivation, but provide supportive auxiliary information, are reported in Appendix H of USEPA (2018). Table 5-10 only includes a subset of all the salmonid data included in Appendix H of USEPA (2018). Data in Table 5-10 were specifically selected for discussion here because these studies adequately reported water chemistry data to populate the multiple linear regression model to normalize effect concentrations to the same reference water chemistry (pH = 7.0; hardness = 100 mg/L; DOC = 1 mg/L). Furthermore, these tests were also selected for a comparative assessment because they represent early life stage tests, which is the most sensitive salmonid life stage and the same life stage used to calculate the Salmonidae FMCV. While these data, taken together, appear to support the Salmonidae FMCV (526.5 µg/L total aluminum at pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L), the range in normalized no effect and effect concentrations, as well as endpoint variability, also indicate the potential for epistemic uncertainty and natural variability that can confound estimates of chum salmon (and other *Oncorhynchus* sp.) sensitivity to chronic aluminum exposures.

Table 5-11. Summary of data from other tests representative of the chronic aluminum sensitivity of chum salmon (*Oncorhynchus spp.*) (from Appendix H in EPA's 2018 Aluminum Criteria Document (USEPA, 2018).

Species	Chemical	Duration	pH	Total Hardness (mg/L as $CaCO_3$)	DOC (mg/L)	Effect	Reported Conc. (µg/L)	Normalized Conc. (µg/L)[a]	Reference
Golden trout (alevin), *Oncorhynchus aguabonita*	Aluminum sulfate	7 d	5	4.89	0.5	NOEC-LOEC (survival)	97-293	18,359	(Delonay, 1991; Delonay et al., 1993)
Cutthroat trout (alevin, 2 d post hatch), *Oncorhynchus clarki*	-	7 d	5	42.5	2	NOEC-LOEC (survival)	50-100	482.0	(Woodward, Farag, Mueller, Little, & Vertucci, 1989)
Cutthroat trout (alevin/larvae), *Oncorhynchus clarki*	-	7 d	5	42.5	2	NOEC-LOEC (growth)	50->50	340.8	(Woodward et al., 1989)
Rainbow trout (embryo/larvae), *Oncorhynchus mykiss*	Aluminum chloride	28 d	7.4	104	0.5	EC50 (death and deformity)	560	457.4	(Birge, 1978; Birge, Hudson, Black, & Westerman, 1978)
Rainbow trout (embryo/larvae), *Oncorhynchus mykiss*	Aluminum sulfate	8 d	6.5	14.3	1.6	No effect	1,000	2,544	(Holtze, 1983)
Rainbow trout (embryo/larvae), *Oncorhynchus mykiss*	Aluminum sulfate	8 d	7.2	14.3	1.6	No effect	1,000	1,023	(Holtze, 1983)
Rainbow trout (embryo), *Oncorhynchus mykiss*	Aluminum chloride	7-12 d	7.4	100	0.5	LC50	560	460.0	(Birge, Westerman, & Spromberg, 2000)
Geometric Mean Chronic Value (µg/L)								1,074.29	NA

[a] All values expressed as total aluminum, normalized to pH 7, DOC of 1 mg/L and 100 mg/L hardness as $CaCO_3$; normalized using MLR equations identified in (USEPA, 2018);

5.2.1.2.2 Deriving Chum Salmon Chronic Adjustment Factors

Raw empirical chronic toxicity data for the three toxicity tests used to generate the Salmonidae FMCV (Table 5-10) were fit to C-R models (see Appendix E.1 for curve fitting methodology and C-R model scoring methodology). Of these, no C-R models were considered quantitatively acceptable models, one of two brook trout tests (Al-Chronic 9) produced a qualitatively acceptable C-R model, while remaining two C-R models (Al-Chronic-8 and Al-Chronic-26) for Salmonidae species were considered unacceptable (see Appendix B.2 and Supplemental Information A for raw empirical toxicity data and model output). No quantitatively acceptable C-R models were available for salmonids and the only quantitatively acceptable C-R model available for a vertebrate species was based on the fathead minnow. As a result, a family-level TAF was calculated for use based on one brook trout EC_{20}:EC_5 ratio that was originally determined to be only a qualitatively acceptable C-R curve fit.

The salmonid family-level TAF was calculated and used instead of the chronic MAF because: 1) it ensured conclusions for salmonids were based on more taxonomically-related data, and 2) the brook trout EC_{20}:EC_5 ratio used to calculate the salmonid family-level chronic TAF was based on a relatively robust curve fit, only being determined to be qualitatively acceptable because the Q-Q Plot used in curve diagnostics indicated certain data points within the curve may be over influential (note: all other curve diagnostics indicated the curve was quantitatively acceptable; see curve Al-Chronic-9 in Appendix B.2 and Supplemental Information A). Moreover, the chronic salmonid family-level TAF (1.696) was greater than the chronic vertebrate TAF (1.668, based on a fathead minnow) and the chronic MAF (1.403), resulting in lower EC_5 estimates. The EC_{20}:EC_{10} ratio and the EC_{20}:EC_{15} ratio from curve Al-Chronic-9 was 1.316 and 1.125, respectively (Table 5-12).

Table 5-12. Chronic point estimates and corresponding EC_{20}:EC_{15}, EC_{20}:EC_{10} and EC_{20}:EC_5 ratios from a qualitatively acceptable chronic concentration-response (C-R) curve that was retained for quantitative use due to limited salmonid chronic C-R data.

Species	C-R Curve Label	EC_{20} (µg/L)	EC_{15} (µg/L)	EC_{10} (µg/L)	EC_5 (µg/L)	EC_{20}:EC_{15} Ratio	EC_{20}:EC_{10} Ratio	EC_{20}:EC_5 Ratio
Brook trout, *Salvelinus fontinalis*	Al-Chronic-9	162.3	144.2	123.3	95.70	1.125	1.316	1.696

5.2.1.2.3 Calculating Chum Salmon Chronic Aluminum Minimum Effect Threshold Values

The Salmonidae family-level TAF (i.e., EC_{20}:EC_5 adjustment factor) as well as the EC_{20}:EC_{10} and EC_{20}:EC_{15} adjustment factors were applied, as a denominator, to the Salmonidae FMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the chum salmon:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) = **EC_5** = 526.5 µg/L / 1.696 = **310.4 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) = **EC_{10}** = 526.5 µg/L / 1.316 = **400.0 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) = **EC_{15}** = 526.5 µg/L / 1.125 = **467.9 µg/L***

5.2.1.2.4 Chum Salmon: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and vertebrate species sensitivity across water chemistry, chum salmon chronic EC_5, EC_{10}, and EC_{15} values were renormalized the Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding chum salmon chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and chum salmon chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were compared with the chum salmon Columbia River ESU. The comparison between chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) and criterion magnitudes for the chum salmon Columbia River ESU only considered the specific water quality data that were collected from level-III ecoregions within the Columbia River ESU. To visually display the comparison, criterion magnitudes (X-axis) and corresponding chronic effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line for the chum salmon Columbia River ESU (see Figure 5-4[37]). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and chum salmon Columbia River ESU chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) are reported in Table 5-13.

[37] The structure and interpretation of Figure 5-4 is fundamentally similar all chronic figures in this effect assessment. Individual data points in the scatter plot represent chronic criterion magnitudes (X axis) and corresponding chronic effect concentrations (Y axis) calculated from the same water chemistry data (i.e., pH, hardness, DOC). Each chronic criterion concentration is compared to three effect levels of 5%, 10%, and 15%, denoted by the light gray, orange, and dark gray colors, respectably. The linear (1:1) line beginning at the origin represents the point at which an individual point has either a greater X axis value (chronic criterion concentration) or a greater Y axis value (EC_5, EC_{10}, or EC_{15}). Data points above the line indicate scenarios where water chemistry results in a chronic criterion concentration that is less than the corresponding chronic effect level, while data points below the line indicate scenarios where water chemistry results in chronic criterion concentration that is greater than the corresponding chronic effect level.

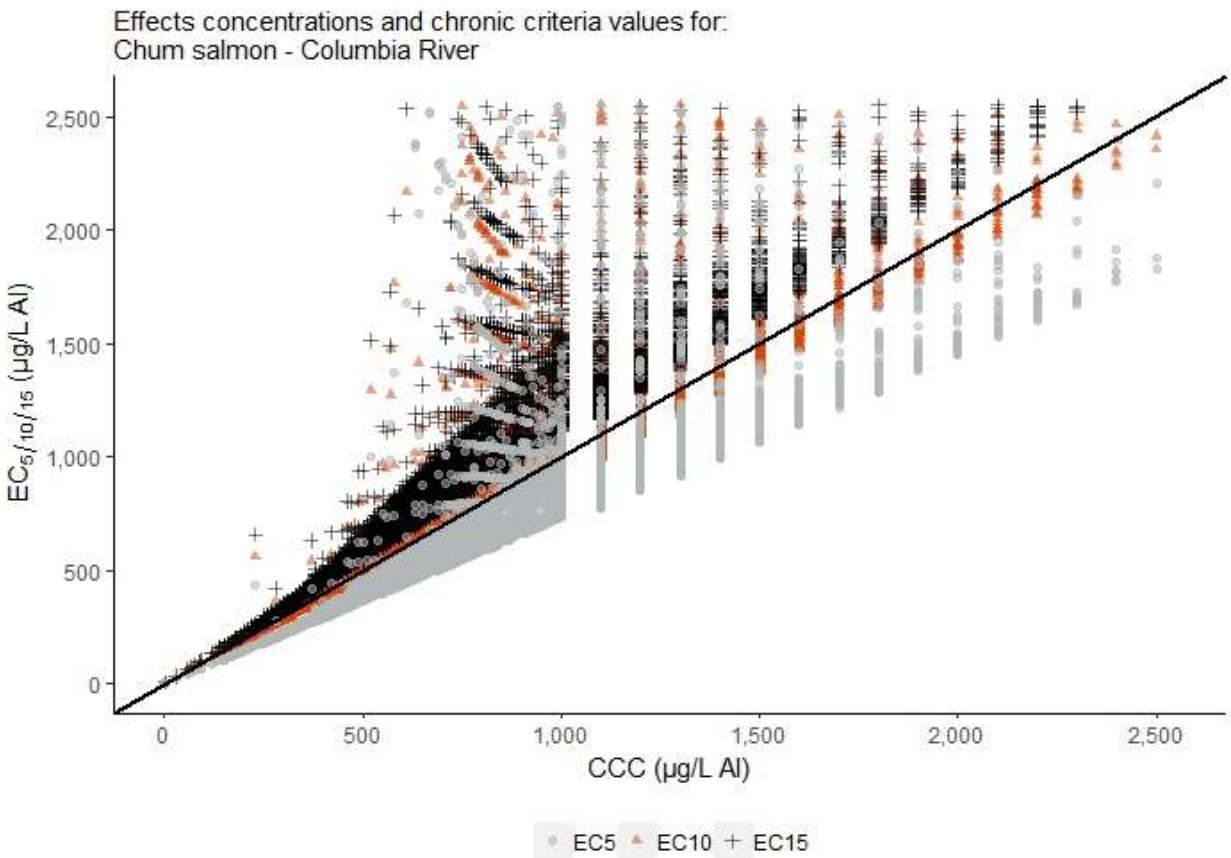

Figure 5-4. Paired chum salmon chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the chum salmon Columbia River Evolutionarily Significant Unit. Note, 1.97% ($n_{removed}$ = 869) of available data points were removed from Figure 2-2 as the Y axis was scaled down (Y axis maximum = 16,649) to aid in data visualization.

Table 5-13. The number and percentage of occurrences where the chronic criterion magnitude exceed chum salmon EC_5, EC_{10}, and EC_{15} values for each chum salmon Evolutionarily Significant Unit in Oregon (ESU). Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing the range of each ESU. Aggregate Effect Levels (AEL) are also shown, defined as the percent of exposed individuals that would experience adverse effects (reduced growth/biomass).

Evolutionarily Significant Unit	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}			CCC > EC_{15}		
Name		n	%	[1]AEL	n	%	AEL	n	%	AEL
Columbia River	14,735	13,771	93.46	4.7%	8,830	59.93	6.0%	0	0	0%

1, AEL was estimated by multiplying the lethal concentration percentile (5, 10, 15th) by the percent of samples in which the criteria value was higher than the toxicity value (i.e., 100*(0.05 x 0.9346) = 4.7% reduction in growth across the population.

5.2.1.2.5 Chum Salmon: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC, across the range of ecoregions that contain the chum salmon Columbia River ESU, was greater than the chum salmon chronic low effect threshold (i.e., EC_5) in 93.46% of water chemistry scenarios and exceeded the chum salmon EC_{10} in 59.93% of water chemistry scenarios. The chronic Al CCC never exceeded the chum salmon EC_{15} (Table 5-13). Because the CCC exceeded the chum salmon chronic low effect threshold, consideration of a refined exposure assessment through a weight of evidence approach was required for chum salmon based on the chronic criterion.

5.2.2 Coho Salmon (Oncorhynchus kisutch)

5.2.2.1 Coho Salmon Acute Aluminum Effects Assessment

5.2.2.1.1 Identifying Coho Salmon Acute Aluminum Data

Species-level acute toxicity data were not available for coho salmon. The only acute toxicity data available for members of the genus *Oncorhynchus* were based on rainbow trout (see Appendix A.1). The rainbow trout species mean acute value (SMAV; 3,312 µg/L total aluminum, normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) was therefore used to represent the sensitivity of coho salmon to acute aluminum exposures (see Section 5.2.1.1.1 and Table 5-6).

5.2.2.1.2 Deriving Coho Salmon Acute Adjustment Factors

No quantitatively acceptable acute TAFs were available for coho salmon or any other vertebrate. Qualitatively acceptable C-R models from two acute brook trout tests (see Al-Acute 66 and Al-Acute-68 in Appendix B.1) were, therefore, used to derive a family-level TAF (i.e., LC_{50}:LC_5 adjustment factor) of 1.967, representative of fish in the Family Salmonidae, including coho salmon (see Section 5.2.1.1.2 and Table 5-8). Al-Acute-66 and Al-Acute-68 C-R models (see Appendix B.1) were also used to calculate LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors of 1.638 and 1.466, respectively that are representative of coho salmon (see Section 5.2.1.1.2 and Table 5-8).

5.2.2.1.3 Calculating Coho Salmon Acute Aluminum Minimum Effect Threshold Values

The Salmonidae family-level TAF (i.e., LC_{50}:LC_5 adjustment factor) as well as the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were applied, as a denominator, to the *Oncorhynchus* GMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value representative of the coho salmon:

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) =* ***LC_5*** *= 3,312 µg/L / 1.967 =* ***1,683 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) =* ***LC_{10}*** *= 3,312 µg/L / 1.638 =* ***2,022 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) =* ***LC_{15}*** *= 3,312 µg/L / 1.466 =* ***2,260 µg/L***

5.2.2.1.4 Coho Salmon: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude and vertebrate species sensitivity across water chemistry, coho salmon acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate an acute criterion magnitude and corresponding coho salmon acute effect threshold values (i.e., LC_5, LC_{10}, LC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and coho salmon acute effect threshold data (i.e., LC_5, LC_{10}, and LC_{15}) were parsed out by coho salmon ESUs. The comparisons between acute effect threshold data and criterion magnitudes for each coho salmon ESU only considered the specific water quality data that were collected from level-III ecoregions within each respective ESU that was assessed. To visually display the comparisons, criterion magnitudes (X-axis) and corresponding acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line for each coho salmon ESU (see Figure 5-5 through Figure 5-7). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, LC_{10}, or LC_{15}) and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and coho salmon ESU acute effect threshold data (i.e., LC_5, LC_{10}, LC_{15}) are reported in Table 5-14.

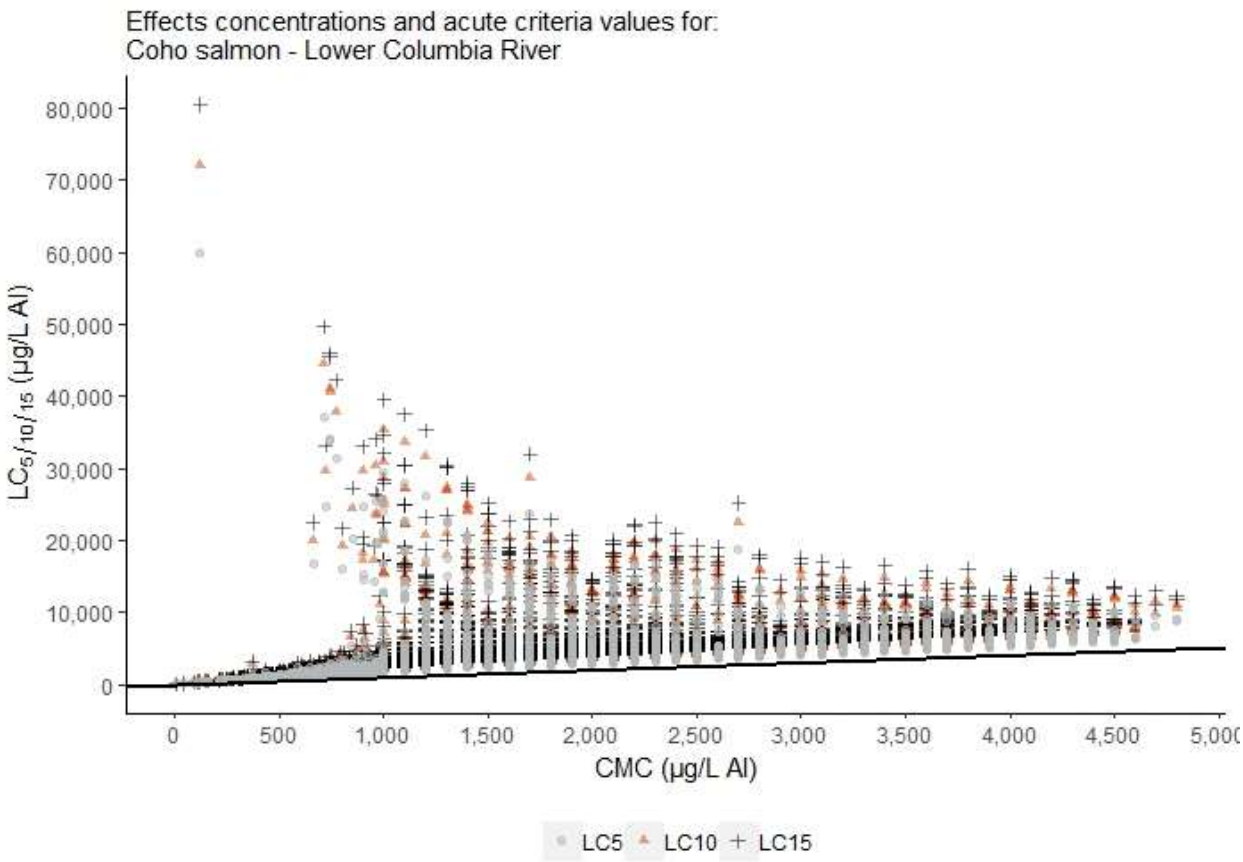

Figure 5-5. Paired coho salmon acute effect concentrations and acute criterion values in Oregon ecoregions containing the coho salmon Lower Columbia River Evolutionarily Significant Unit.

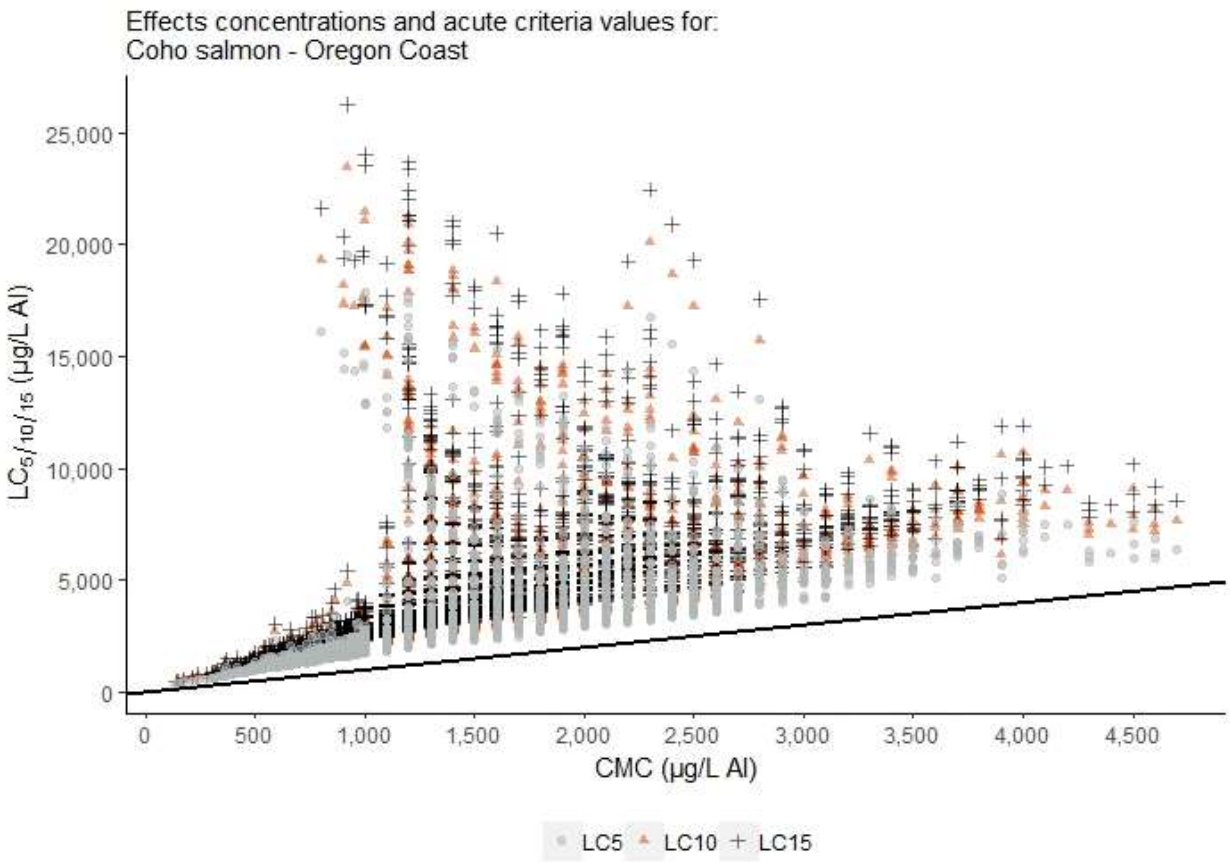

Figure 5-6. Paired coho salmon acute effect concentrations and acute criterion values in Oregon ecoregions containing the coho salmon Oregon Coast Evolutionarily Significant Unit.

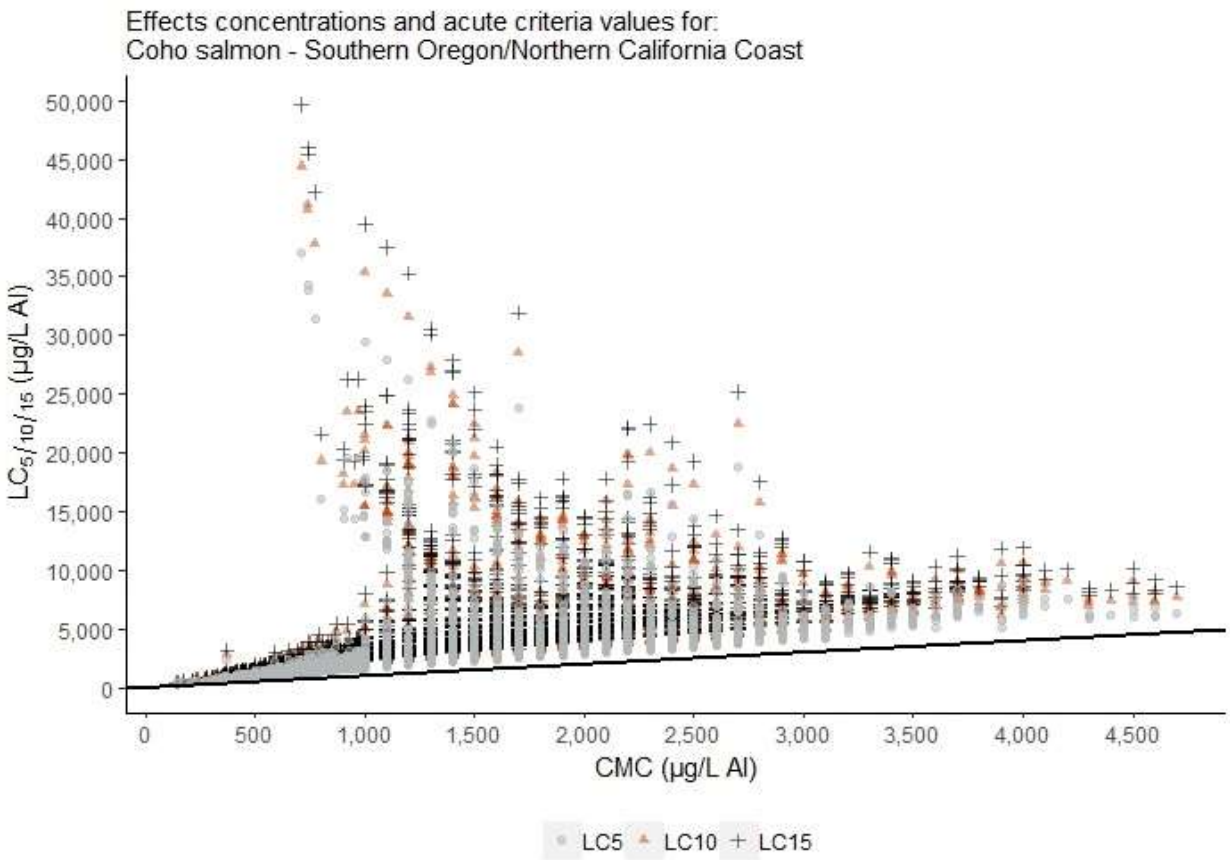

Figure 5-7. Paired coho salmon acute effect concentrations and acute criterion values in Oregon ecoregions containing the coho salmon Southern Oregon/Northern California Coast Evolutionarily Significant Unit.

Table 5-14. The number and percentage of occurrences where the acute criterion magnitude exceed coho salmon LC_5, LC_{10}, and LC_{15} values for each coho salmon Evolutionarily Significant Unit in Oregon (ESU). Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing the range of each ESU. Aggregate Effect Levels (AEL) are also shown, defined as the percent of exposed individuals that would experience adverse effects (mortality).

Evolutionarily Significant Unit	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}			CMC > LC_{15}		
Name		n	%	[1]AEL	n	%	AEL	n	%	AEL
Lower Columbia River	13,973	0	0	0%	0	0	0%	0	0	0%
Oregon Coast	6,206	0	0	0%	0	0	0%	0	0	0%
Southern Oregon/Northern California Coast	7,256	0	0	0%	0	0	0%	0	0	0%

1, AEL was estimated by multiplying the lethal concentration percentile (5, 10, 15th) by the percent of samples in which the criteria value was higher than the toxicity value (i.e., 100*(0.05 or 0.10 or 0.15 x 0.00) = 0.00% probability of adverse effects occurring in exposed fish.

5.2.2.1.5 Coho Salmon: Tier I Acute Assessment Conclusion

For all ESUs, the acute aluminum CMC was less than the coho salmon acute low effect threshold (i.e., LC_5) under all water chemistry scenarios in Oregon ecoregions within the coho salmon ESUs (Table 5-14). The LC_5 range was 36 to 59,906 µg/L and the CMC range was 4.8 to 4,800 µg/L for the Lower Columbia River ESU. The LC_5 range was 353 to 19,546 µg/L and the CMC range was 140 to 4,700 µg/L for the Oregon Coast and Southern Oregon/Northern California Coast ESUs. As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for coho salmon based on the acute criterion.

5.2.2.2 Coho Salmon Chronic Aluminum Effects Assessment

5.2.2.2.1 Identifying Coho Salmon Chronic Aluminum Data

Species-level chronic aluminum toxicity data were not available for coho salmon or any other members within the genus *Oncorhynchus*. Therefore, chronic toxicity data for members of the Family Salmonidae were used to estimate the Salmonidae FMCV of 526.5 µg/L (total aluminum pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) that was representative of coho salmon (see Section 5.2.1.2.1 and Table 5-10). EPA also evaluated qualitatively acceptable chronic toxicity data (Appendix H of USEPA (2018)) from several tests with early-life stages of golden, cutthroat, and rainbow trout to provide comparative information (Table 5-11 provides a summary of these tests). Overall, results of these qualitatively acceptable chronic toxicity tests appear to support the Salmonidae FMCV; however, the range in normalized no effect and effect concentrations, as well as endpoint variability, also indicate the potential for epistemic uncertainty and natural variability that can confound estimates of coho salmon (and other *Oncorhynchus* sp.) sensitivity to chronic aluminum exposures.

5.2.2.2.2 Deriving Coho Salmon Chronic Adjustment Factors

No quantitatively acceptable chronic TAFs were available for coho salmon or any other member of the Order Salmoniformes. The only quantitatively acceptable C-R model available for a vertebrate species was based on the fathead minnow. As a result, a family-level TAF was calculated from a brook trout EC_{20}:EC_5 ratio that was originally determined to be a qualitatively acceptable C-R curve fit (see Appendix B.2, curve Al-Chronic-9 and Section 5.2.1.2.2). The chronic salmonid family-level TAF (i.e., EC_{20}:EC_5) was 1.696 and the corresponding EC_{20}:EC_{10} ratio and the EC_{20}:EC_{15} ratio from curve Al-Chronic-9 was 1.316 and 1.125, respectively (see Section 5.2.1.2.2 and Table 5-12).

5.2.2.2.3 Calculating Coho Salmon Chronic Aluminum Minimum Effect Threshold Values

The Salmonidae Family-level TAF (i.e., EC_{20}:EC_5 adjustment factor) as well as the EC_{20}:EC_{10} and EC_{20}:EC_{15} adjustment factors were applied, as a denominator, to the Salmonidae FMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the coho salmon:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) =* ***EC_5*** *= 526.5 µg/L / 1.696 =* ***310.4 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) =* ***EC_{10}*** *= 526.5 µg/L / 1.316 =* ***400.0 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) =* ***EC_{15}*** *= 526.5 µg/L / 1.125 =* ***467.9 µg/L***

5.2.2.2.4 Coho Salmon: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and vertebrate species sensitivity across water chemistry, coho salmon chronic EC_5, EC_{10}, and EC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding coho salmon chronic effect threshold values (i.e., EC_5, EC_{10}, EC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and coho salmon chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were parsed out by coho salmon ESU. The comparisons between chronic effect threshold data and criterion magnitudes for each coho salmon ESU only considered the specific water quality data that were collected from level-III ecoregions within each respective ESU that was assessed. To visually display the comparisons, criterion magnitudes (X-axis) and corresponding chronic effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line for each coho salmon ESU (see Figure 5-8 through Figure 5-10). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and coho salmon ESU chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) are reported in Table 5-15.

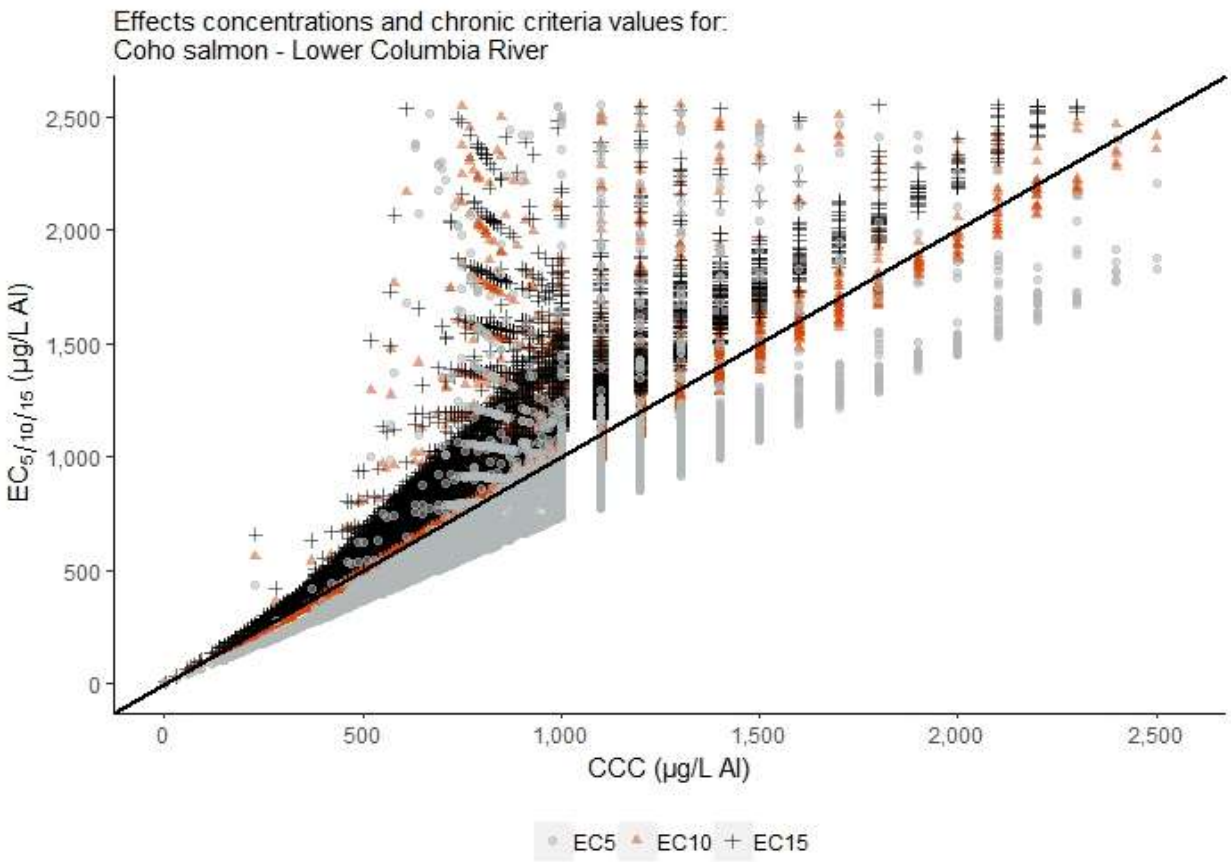

Figure 5-8. Paired coho salmon chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the coho salmon Lower Columbia River Evolutionarily Significant Unit. Note, 1.3% ($n_{removed}$ = 543) of available data points were removed from Figure 5-7 as the Y axis was scaled down (Y axis maximum = 16,649) to aid in data visualization.

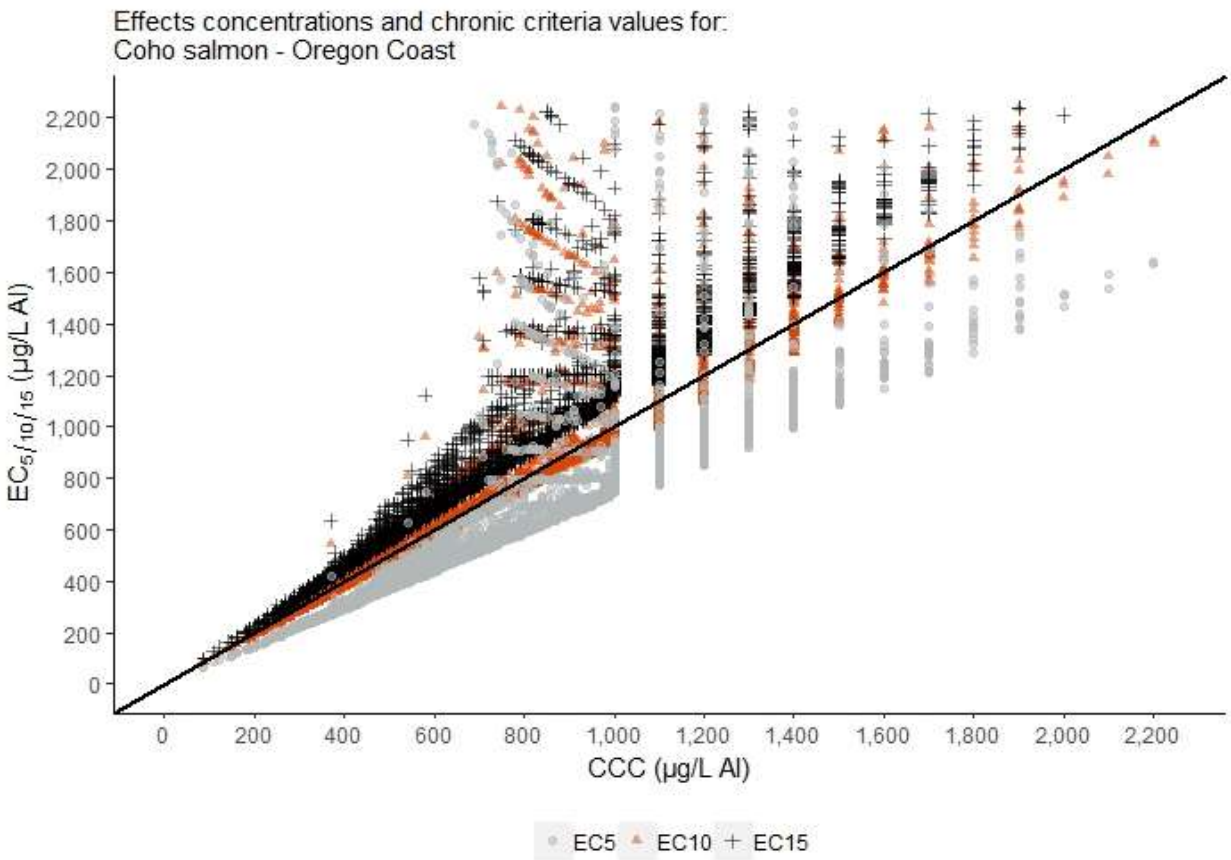

Figure 5-9. Paired coho salmon chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the coho salmon Oregon Coast Evolutionarily Significant Unit. Note, 1.70% ($n_{removed}$ = 317) of available data points were removed from Figure 5-8 as the Y axis was scaled down (Y axis maximum = 5,432) to aid in data visualization.

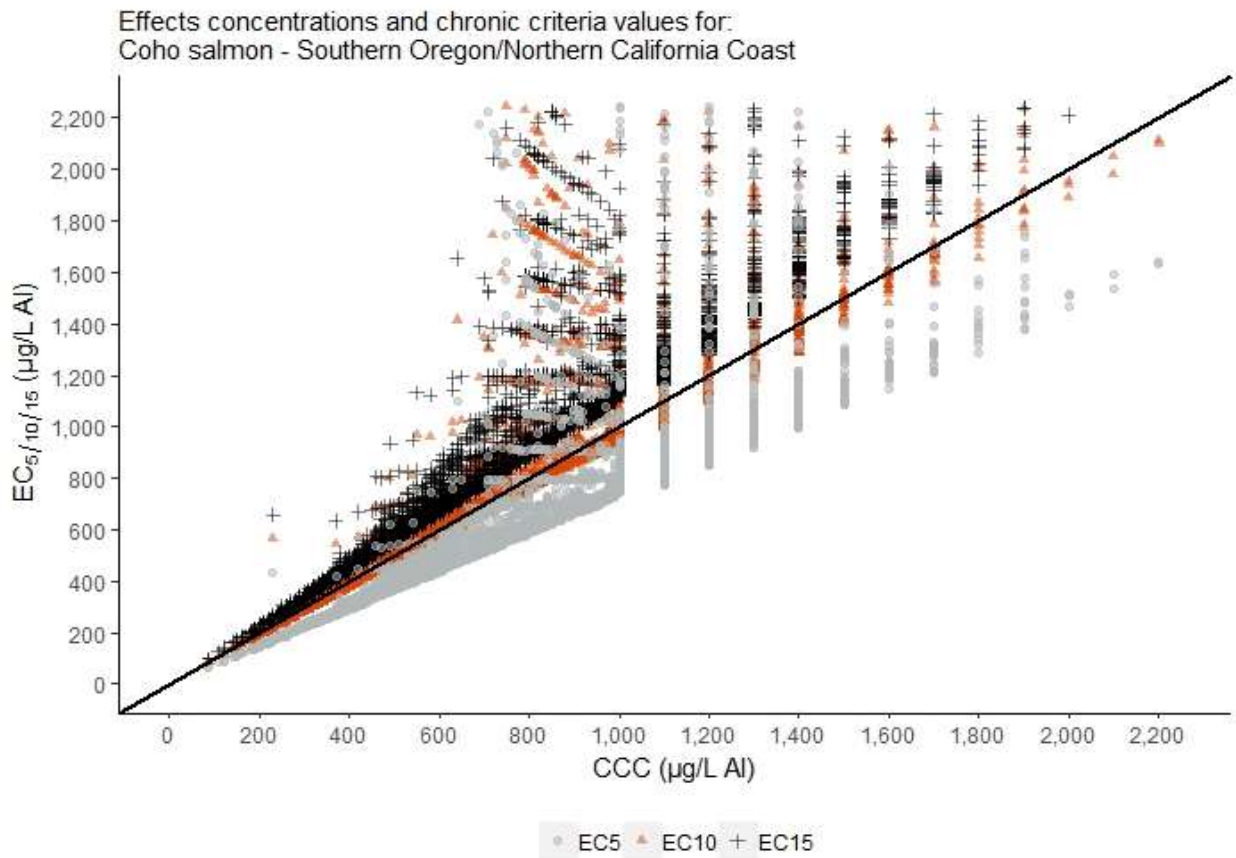

Figure 5-10. Paired coho salmon chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the coho salmon Southern Oregon/Northern California Coast Evolutionarily Significant Unit. Note, 2.04% ($n_{removed}$ = 445) of available data points were removed from Figure 5-9 as the Y axis was scaled down (Y axis maximum = 10,275) to aid in data visualization.

Table 5-15. The number and percentage of occurrences where the chronic criterion magnitude exceed coho salmon EC_5, EC_{10}, and EC_{15} values for each coho salmon Evolutionarily Significant Unit in Oregon (ESU). Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing the range of each ESU. Aggregate Effect Levels (AEL) are also shown, defined as the percent of exposed individuals that would experience adverse effects (reduced growth/biomass).

Evolutionarily Significant Unit	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}			CCC > EC_{15}		
Name		n	%	AEL	n	%	AEL	n	%	AEL
Lower Columbia River	13,973	13,232	94.70	4.7%	8,513	60.92	6.1%	0	0	0
Oregon Coast	6,206	5,864	94.49	4.7%	4,429	71.37	7.1%	0	0	0
Southern Oregon/Northern California Coast	7,256	6,801	93.73	4.7%	5,090	70.15	7.0%	0	0	0

5.2.2.2.5 Coho Salmon: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC exceeded the chum salmon EC_5 between 93.73% – 94.70% of exposure conditions, depending on the specific ESU assessed. The chronic aluminum CCC exceeded the coho salmon EC_{10} between 60.92% – 71.37% of exposure conditions, depending on the specific ESU assessed. The chronic Al CCC never exceeded the coho salmon EC_{15} (Table 5-15). Because the CCC exceeded the coho salmon chronic low effect threshold, consideration of a refined exposure assessment through a weight of evidence approach was required for coho salmon based on the chronic criterion.

5.2.3 Sockeye Salmon (Oncorhynchus nerka)

5.2.3.1 Sockeye Salmon Acute Aluminum Effects Assessment

5.2.3.1.1 Identifying Sockeye Salmon Acute Aluminum Data

Species-level acute toxicity data were not available for sockeye salmon. The only acute toxicity data available for members of the genus *Oncorhynchus* were based on rainbow trout (see Appendix A.1). The rainbow trout species mean acute value (SMAV; 3,312 µg/L total aluminum, normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) was therefore used to represent the sensitivity of sockeye salmon to acute aluminum exposures (see Section 5.2.1.1.1 and Table 5-6).

5.2.3.1.2 Deriving Sockeye Salmon Acute Adjustment Factors

No quantitatively acceptable acute TAFs were available for sockeye salmon or any other vertebrate. Qualitatively acceptable C-R models from two acute brook trout tests (see Al-Acute 66 and Al-Acute-68 in Appendix B.1) were, therefore, used to derive a family-level TAF (i.e., LC_{50}:LC_5 adjustment factor) of 1.967, representative of fish in the Family Salmonidae, including sockeye salmon (see Section 5.2.1.1.2 and Table 5-8). Al-Acute-66 and Al-Acute-68 C-R models were also used to calculate LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors of 1.638 and 1.466, respectively, that are representative of sockeye salmon (see Section 5.2.1.1.2 and Table 5-8).

5.2.3.1.3 Calculating Sockeye Salmon Acute Aluminum Minimum Effect Threshold Values

The Salmonidae family-level TAF (i.e., LC_{50}:LC_5 adjustment factor) as well as the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were applied, as a denominator, to the *Oncorhynchus* GMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value representative of the sockeye salmon:

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) = **LC_5** = 3,312 µg/L / 1.967 = **1,683 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) = **LC_{10}** = 3,312 µg/L / 1.638 = **2,022 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) = **LC_{15}** = 3,312 µg/L / 1.466 = **2,260 µg/L***

5.2.3.1.4 Sockeye Salmon: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude and vertebrate species sensitivity across water chemistry, sockeye salmon acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate an acute criterion magnitude and corresponding sockeye salmon acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and sockeye salmon acute effect threshold data (i.e., LC_5, LC_{10}, and LC_{15}) were compared to the sockeye salmon Snake River ESU. The comparison between acute effect threshold data (i.e., LC_5, LC_{10}, LC_{15}) and criterion magnitudes for the sockeye salmon Snake River ESU only considered the specific water quality data that were collected from level-III ecoregions within the Snake River ESU. To visually display the comparison, criterion magnitudes (X-axis) and corresponding acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line for the sockeye salmon Snake River ESU (see Figure 5-11). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, LC_{10}, or LC_{15}) and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and sockeye salmon Snake River ESU acute effect threshold data (i.e., LC_5, LC_{10}, LC_{15}) are reported in Table 5-16.

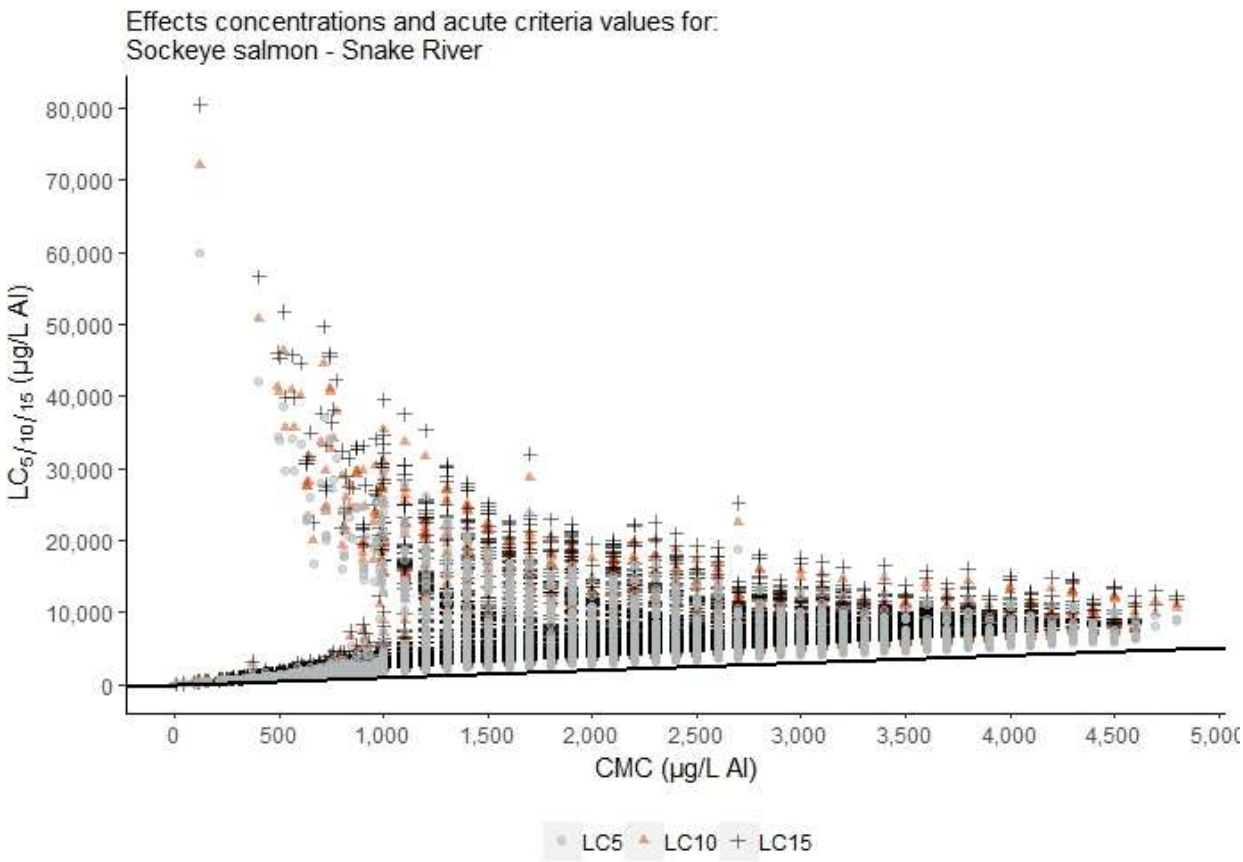

Figure 5-11. Paired sockeye salmon acute effect concentrations and acute criterion values in Oregon ecoregions containing the sockeye salmon Snake River Evolutionarily Significant Unit.

Table 5-16. The number and percentage of occurrences where the acute criterion magnitude exceed sockeye salmon LC_5, LC_{10}, and LC_{15} values for each sockeye salmon Evolutionarily Significant Unit in Oregon (ESU). Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing the range of each ESU. Aggregate Effect Levels (AEL) are also shown, defined as the percent of exposed individuals that would experience adverse effects (mortality).

Evolutionarily Significant Unit	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}			CMC > LC_{15}		
Name		n	%	AEL	n	%	AEL	n	%	AEL
Snake River	16,865	0	0	0%	0	0	0%	0	0	0%

5.2.3.1.5 Sockeye Salmon: Tier I Acute Assessment Conclusion

The acute aluminum CMC was less than the sockeye salmon Snake River ESU acute low effect threshold (i.e., LC_5) in all water chemistry scenarios in Oregon ecoregions that overlap with the sockeye salmon ESU (Table 5-16). The LC_5 range was 36 to 59,906 µg/L and the CMC range was 4.8 to 4,800 µg/L. As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for sockeye salmon based on the acute criterion.

5.2.3.2 Sockeye Salmon Chronic Aluminum Effects Assessment

5.2.3.2.1 Identifying Sockeye Salmon Chronic Aluminum Data

Species-level chronic aluminum toxicity data were not available for sockeye salmon or any other members within the genus *Oncorhynchus*. Therefore, chronic toxicity data for members of the Family Salmonidae were used to estimate the Salmonidae FMCV of 526.5 µg/L (total aluminum pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) that was representative of sockeye salmon (see Section 5.2.1.2.1 and Table 5-10). EPA also evaluated qualitatively acceptable chronic toxicity data (Appendix H of USEPA (2018)) from several tests with early-life stages of golden, cutthroat, and rainbow trout to provide comparative information (Table 5-9 provides a summary of these tests). Overall, results of these qualitatively acceptable chronic toxicity tests appear to support the Salmonidae FMCV; however, the range in normalized no effect and effect concentrations, as well as endpoint variability, also indicate the potential for epistemic uncertainty and natural variability that can confound estimates of sockeye salmon (and other *Oncorhynchus* sp.) sensitivity to chronic aluminum exposures.

5.2.3.2.2 Deriving Sockeye Salmon Chronic Adjustment Factors

No quantitatively acceptable chronic TAFs were available for sockeye salmon or any other member of the Order Salmoniformes. The only quantitatively acceptable C-R model available for a vertebrate species was based on the fathead minnow. As a result, a family-level TAF was calculated from a brook trout EC_{20}:EC_5 ratio that was originally determined to be a qualitatively acceptable C-R curve fit (see Appendix B.2, curve Al-Chronic-9 and Section 5.2.1.2.2). The chronic salmonid family-level TAF (i.e., EC_{20}:EC_5) was 1.696 and the corresponding EC_{20}:EC_{10} ratio and the EC_{20}:EC_{15} ratio from curve Al-Chronic-9 was 1.316 and 1.125, respectively (see Section 5.2.1.2.2 and Table 5-12).

5.2.3.2.3 Calculating Sockeye Salmon Chronic Aluminum Minimum Effect Threshold Values

The Salmonidae family-level TAF (i.e., EC_{20}:EC_5 adjustment factor) as well as the EC_{20}:EC_{10} and EC_{20}:EC_{15} adjustment factors were applied, as a denominator, to the Salmonidae FMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the sockeye salmon:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) = **EC_5** = 526.5 µg/L / 1.696 = **310.4 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) = **EC_{10}** = 526.5 µg/L / 1.316 = **400.0 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) = **EC_{15}** = 526.5 µg/L / 1.125 = **467.9 µg/L***

5.2.3.2.4 Sockeye Salmon: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and vertebrate species sensitivity across water chemistry, sockeye salmon chronic EC_5, EC_{10}, and EC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding sockeye salmon chronic low effect threshold values (i.e., EC_5, EC_{10}, EC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and sockeye salmon chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were compared to the sockeye salmon Snake River ESU. The comparison between chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) and criterion magnitudes for the sockeye salmon Snake River ESU only considered the specific water quality data that were collected from level-III ecoregions within the Snake River ESU. To visually display the comparison, criterion magnitudes (X-axis) and corresponding chronic effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line for the sockeye salmon Snake River ESU (see Figure 5-12). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and sockeye salmon Snake River ESU chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) are reported in Table 5-17.

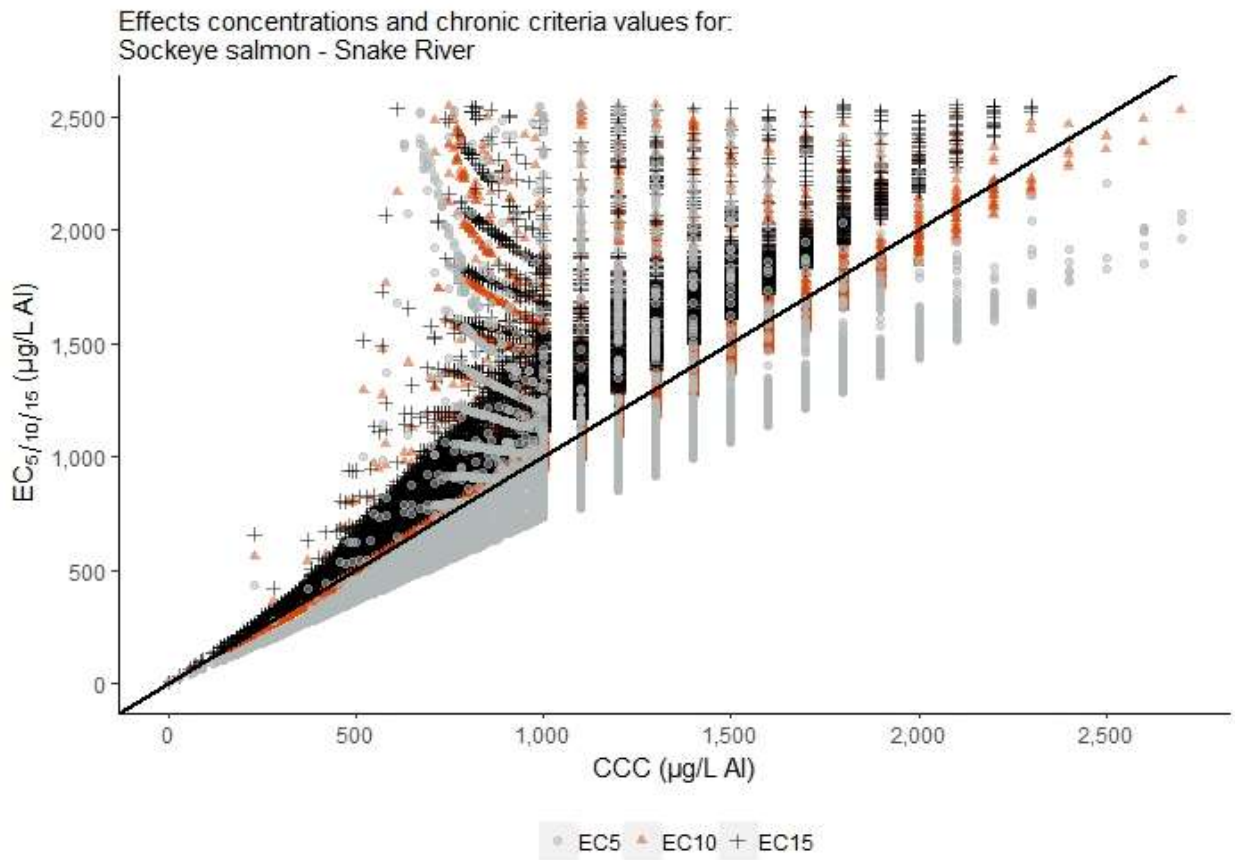

Figure 5-12. Paired sockeye salmon chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the sockeye salmon Snake River Evolutionarily Significant Unit. Note, 2.3% ($n_{removed}$ = 1,166) of available data points were removed from Figure 5-11 as the Y axis was scaled down (Y axis maximum = 16,649) to aid in data visualization.

Table 5-17. The number and percentage of occurrences where the chronic criterion magnitude exceed sockeye salmon EC_5, EC_{10}, and EC_{15} values for each sockeye salmon Evolutionarily Significant Unit in Oregon (ESU). Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing the range of each ESU. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (reduced growth/biomass) at each effects concentration.

Evolutionarily Significant Unit	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}			CCC > EC_{15}		
Name		n	%	AEL	n	%	AEL	n	%	AEL
Snake River	16,865	15,529	92.08	4.6%	9,834	58.31	5.8%	0	0	0

5.2.3.2.5 Sockeye Salmon: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC, across the range of ecoregions that contain the sockeye salmon Snake River ESU, was greater than the sockeye salmon chronic low effect threshold (i.e., EC_5) in 92.08% of water chemistry scenarios and exceeded the sockeye salmon EC_{10} in 58.31% of water chemistry scenarios. The chronic Al CCC never exceeded the sockeye salmon EC_{15} (Table 5-17). Because the CCC exceeded the sockeye salmon chronic low effect threshold, a refined exposure assessment through a weight of evidence approach was required for sockeye salmon based on the chronic criterion.

5.2.4 Chinook Salmon (Oncorhynchus tshawytscha)

5.2.4.1 Chinook Salmon Acute Aluminum Effects Assessment

5.2.4.1.1 Identifying Chinook Salmon Acute Aluminum Data

Species-level acute toxicity data were not available for Chinook salmon. The only acute toxicity data available for members of the genus *Oncorhynchus* were based on rainbow trout (see Appendix A.1). The rainbow trout species mean acute value (SMAV; 3,312 µg/L total aluminum, normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) was therefore used to represent the sensitivity of Chinook salmon to acute aluminum exposures (see Section 5.2.1.1.1 and Table 5-6).

5.2.4.1.2 Deriving Chinook Salmon Acute Adjustment Factors

No quantitatively acceptable acute TAFs were available for Chinook salmon or any other vertebrate. Qualitatively acceptable C-R models from two acute brook trout tests (see Al-Acute 66 and Al-Acute-68 in Appendix B.1) were, therefore, used to derive a family-level TAF (i.e., LC_{50}:LC_5 adjustment factor) of 1.967, representative of fish in the Family Salmonidae, including Chinook salmon (see Section 5.2.1.1.2 and Table 5-8). Al-Acute-66 and Al-Acute-68 C-R models (see Appendix B.1) were also used to calculate LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors of 1.638 and 1.466, respectively that are representative of Chinook salmon (see Section 5.2.1.1.2 and Table 5-8).

5.2.4.1.3 Calculating Chinook Salmon Acute Aluminum Minimum Effect Threshold Values

The Salmonidae family-level TAF (i.e., LC_{50}:LC_5 adjustment factor) as well as the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were applied, as a denominator, to the *Oncorhynchus* GMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value representative of the Chinook salmon:

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) =* ***LC_5*** *= 3,312 µg/L / 1.967 =* ***1,683 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) =* ***LC_{10}*** *= 3,312 µg/L / 1.638 =* ***2,022 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) =* ***LC_{15}*** *= 3,312 µg/L / 1.466 =* ***2,260 µg/L***

5.2.4.1.4 Chinook Salmon: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude and vertebrate species sensitivity across water chemistry, Chinook salmon acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate an acute criterion magnitude and corresponding Chinook salmon acute effect threshold values (i.e., LC_5, LC_{10}, LC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and Chinook salmon acute effect threshold data (i.e., LC_5, LC_{10}, and LC_{15}) were parsed out by each Chinook salmon ESU. The comparisons between acute effect threshold data and criterion magnitudes for each Chinook salmon ESU only considered the specific water quality data that were collected from level-III ecoregions within each respective ESU that was assessed. To visually display the comparisons, criterion magnitudes (X-axis) and corresponding acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line for each Chinook salmon ESU (see Figure 5-13 through Figure 5-17). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, LC_{10}, or LC_{15}) and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and Chinook salmon ESU acute effect threshold data (i.e., LC_5, LC_{10}, LC_{15}) are reported in Table 5-18.

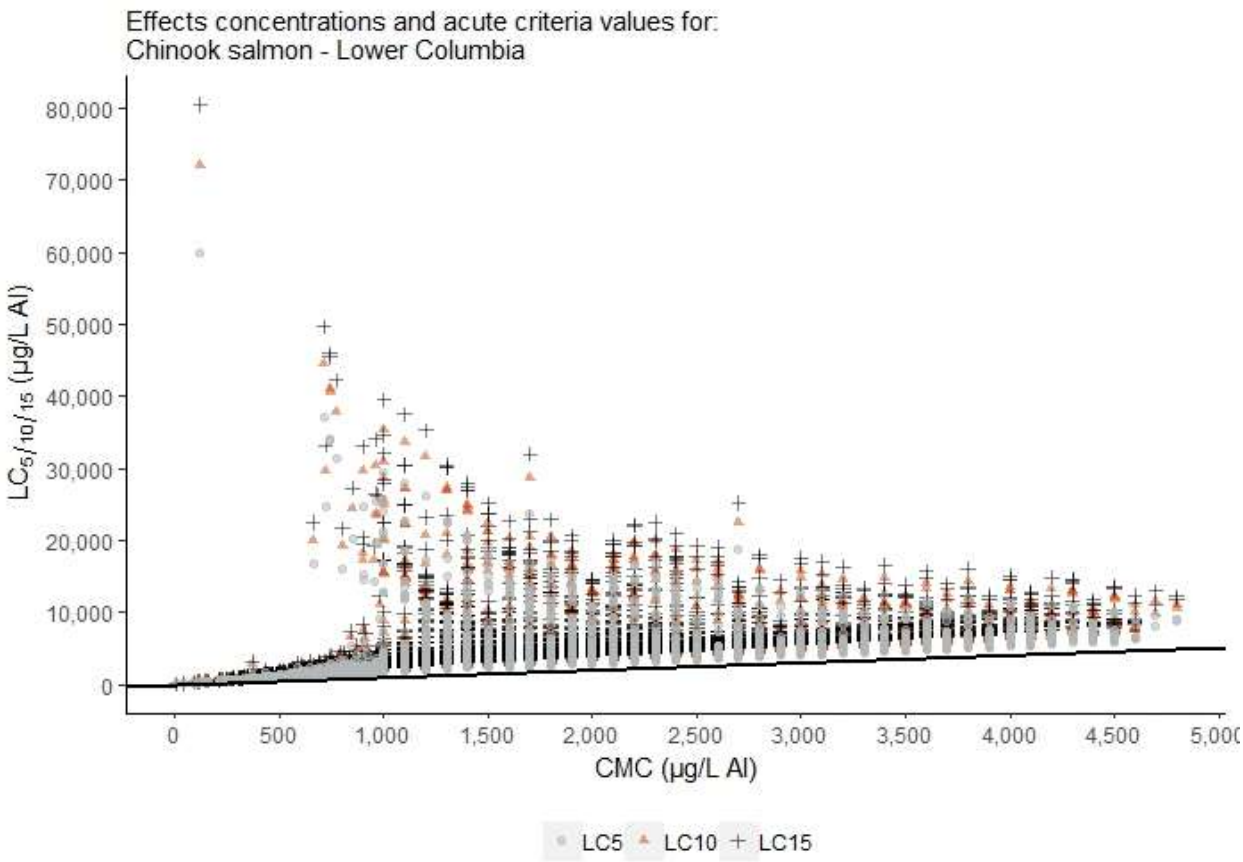

Figure 5-13. Paired Chinook salmon acute effect concentrations and acute criterion values in Oregon ecoregions containing the Chinook salmon Lower Columbia River Evolutionarily Significant Unit.

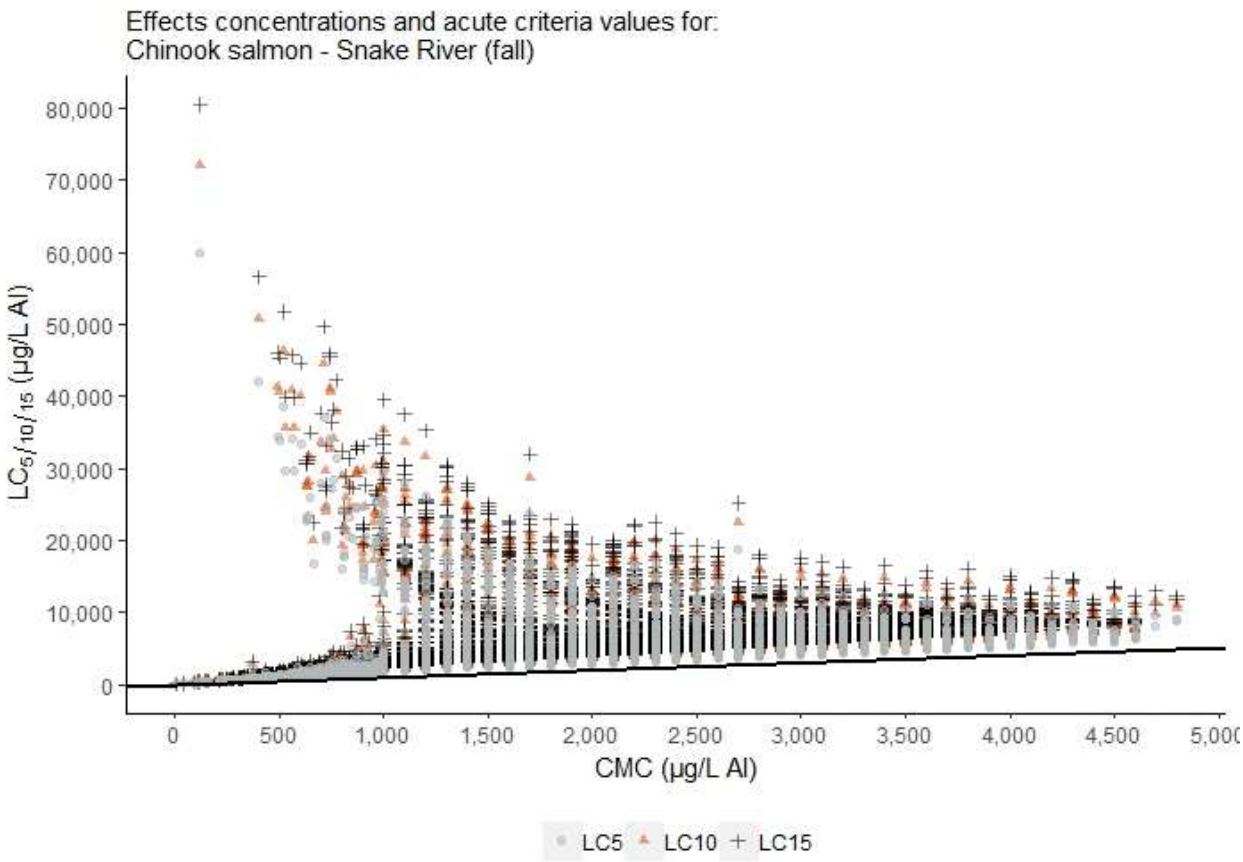

Figure 5-14. Paired Chinook salmon acute effect concentrations and acute criterion values in Oregon ecoregions containing the Chinook salmon Snake River (Fall) Evolutionarily Significant Unit.

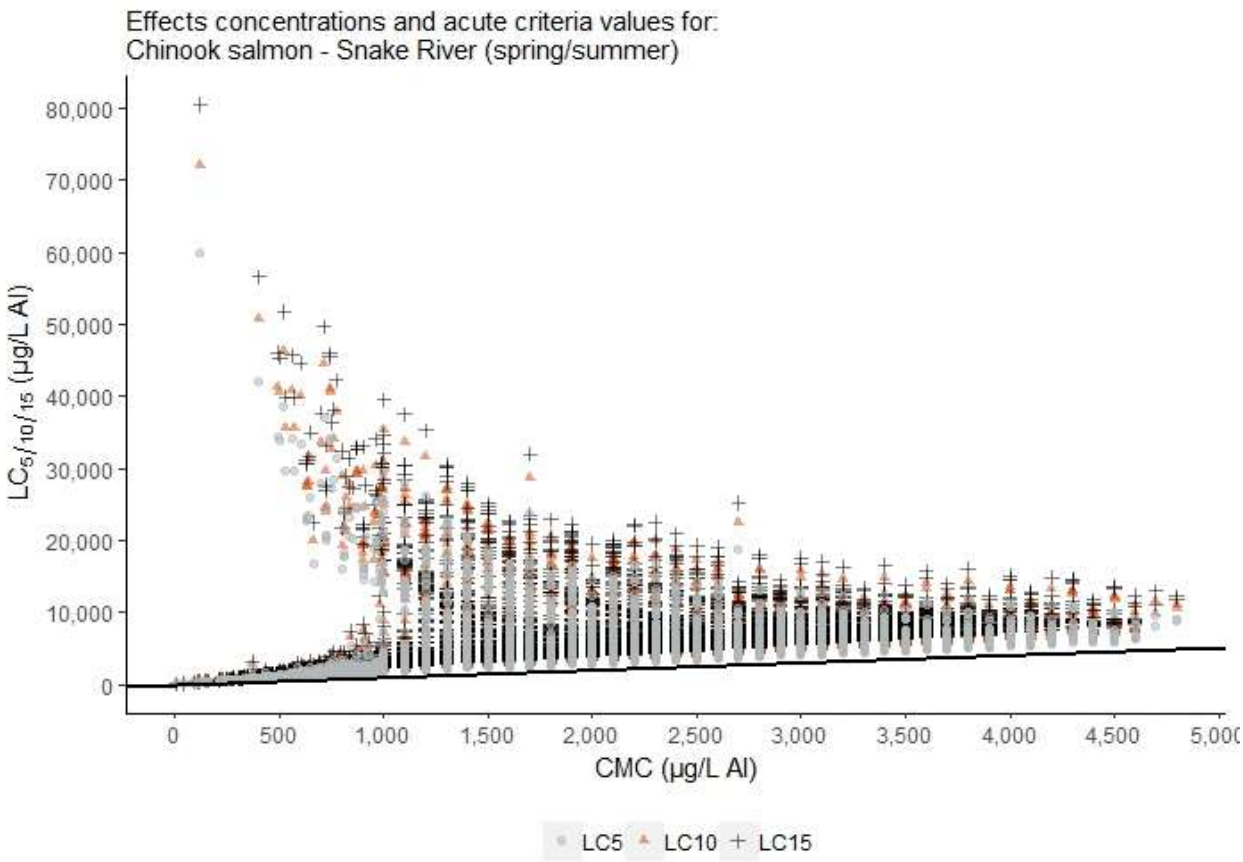

Figure 5-15. Paired Chinook salmon acute effect concentrations and acute criterion values in Oregon ecoregions containing the Chinook salmon Snake River (Spring/Summer) Evolutionarily Significant Unit.

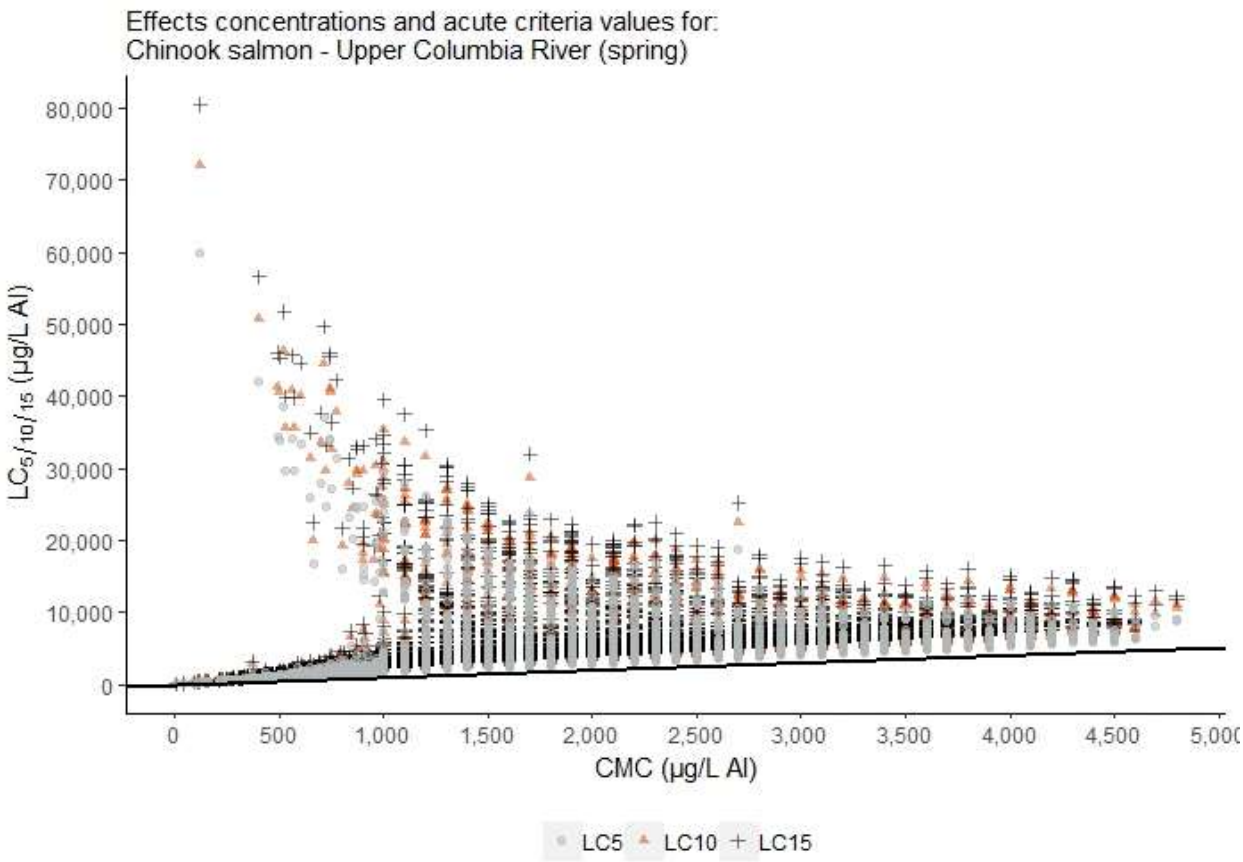

Figure 5-16. Paired Chinook salmon acute effect concentrations and acute criterion values in Oregon ecoregions containing the Chinook salmon Upper Columbia River (Spring) Evolutionarily Significant Unit.

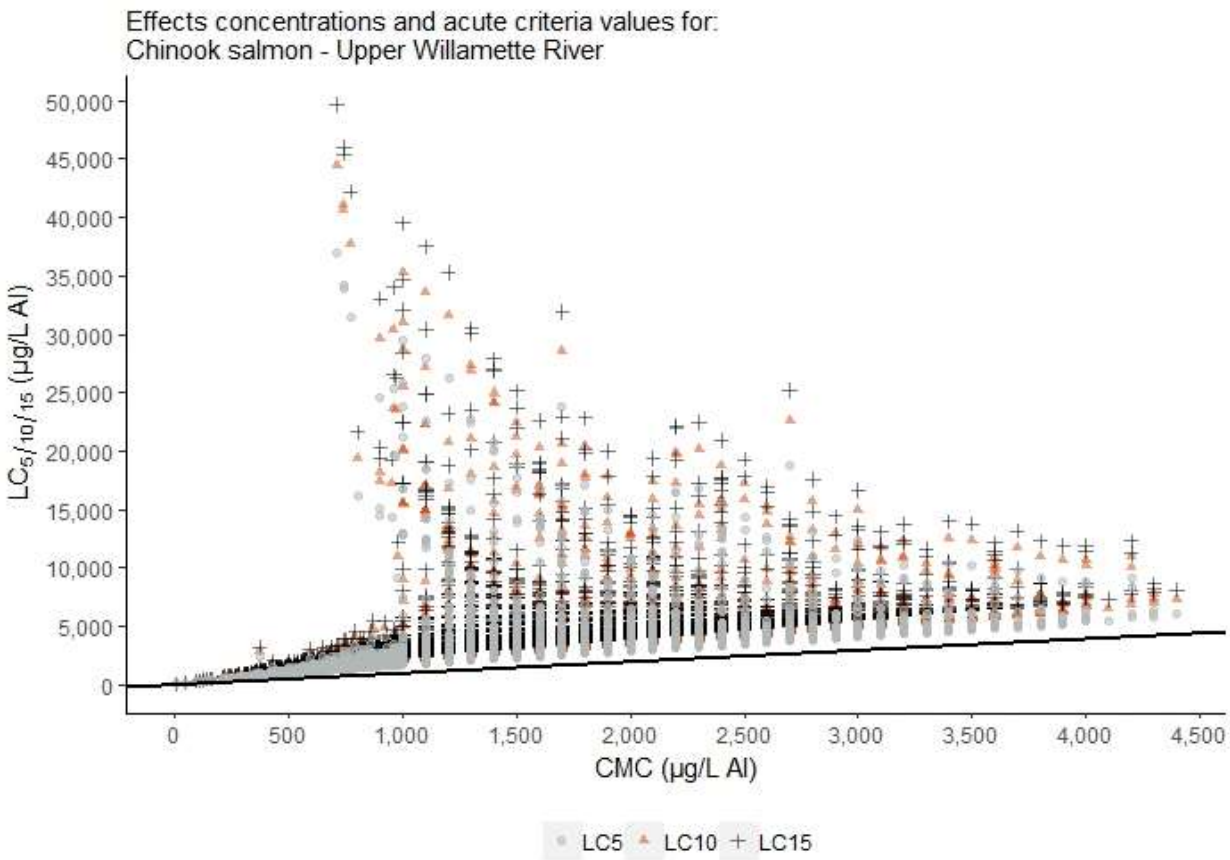

Figure 5-17. Paired Chinook salmon acute effect concentrations and acute criterion values in Oregon ecoregions containing the Chinook salmon Upper Willamette River Evolutionarily Significant Unit.

Table 5-18. The number and percentage of occurrences where the acute criterion magnitude exceed Chinook salmon LC_5, LC_{10}, and LC_{15} values for each Chinook salmon Evolutionarily Significant Unit in Oregon (ESU). Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing the range of each ESU. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (mortality) at each effects concentration.

Evolutionarily Significant Unit	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}			CMC > LC_{15}		
Name		n	%	AEL	n	%	AEL	n	%	AEL
Lower Columbia River	13,973	0	0	0%	0	0	0%	0	0	0%
Snake River (fall)	16,865	0	0	0%	0	0	0%	0	0	0%
Snake River (spring/summer)	16,865	0	0	0%	0	0	0%	0	0	0%
Upper Columbia River (spring)	14,735	0	0	0%	0	0	0%	0	0	0%
Upper Willamette River	12,143	0	0	0%	0	0	0%	0	0	0%

5.2.4.1.5 Chinook Salmon: Tier I Acute Assessment Conclusion

For all ESUs, the acute aluminum CMC was less than the Chinook salmon acute low effect threshold (i.e., LC_5) in all water chemistry scenarios in Oregon ecoregions within Chinook salmon ESUs (Table 5-18). The LC_5 range was 36 to 36,972 µg/L and the CMC range was 4.8 to 4,400 µg/L for the Upper Willamette River ESU. For all other ESUs, the LC_5 range was 36 to 59,906 µg/L and the CMC range was 4.8 to 4,800 µg/L. As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for Chinook salmon based on the acute criterion.

5.2.4.2 Chinook Salmon Chronic Aluminum Effects Assessment

5.2.4.2.1 Identifying Chinook Salmon Chronic Aluminum Data

Species-level chronic aluminum toxicity data were not available for Chinook salmon or any other members within the genus *Oncorhynchus*. Therefore, chronic toxicity data for members of the Family Salmonidae were used to estimate the Salmonidae FMCV of 526.5 µg/L (total aluminum pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) that was representative of Chinook salmon (see Section 5.2.1.2.1 and Table 5-10). EPA also evaluated qualitatively acceptable chronic toxicity data (Appendix H of USEPA (2018))from several tests with early-life stages of golden, cutthroat, and rainbow trout to provide comparative information (Table 5-11 provides a summary of these tests). Overall, results of these qualitatively-acceptable chronic toxicity tests appear to support the Salmonidae FMCV; however, the range in normalized no effect and effect concentrations, as well as endpoint variability, also indicate the potential for epistemic uncertainty and natural variability that can confound estimates of Chinook salmon (and other *Oncorhynchus* sp.) sensitivity to chronic aluminum exposures.

5.2.4.2.2 Deriving Chinook Salmon Chronic Adjustment Factors

No quantitatively acceptable chronic TAFs were available for Chinook salmon or any other member of the Order Salmoniformes. The only quantitatively acceptable C-R model available for a vertebrate species was based on the fathead minnow. As a result, a family-level TAF was calculated from a brook trout EC_{20}:EC_5 ratio that was originally determined to be a qualitatively acceptable C-R curve fit (see Appendix B.2, curve Al-Chronic-9 and Section 5.2.1.2.2). The chronic salmonid family-level TAF (i.e., EC_{20}:EC_5) was 1.696 and the corresponding EC_{20}:EC_{10} ratio and the EC_{20}:EC_{15} ratio from curve Al-Chronic-9 was 1.316 and 1.125, respectively (see Section 5.2.1.2.2 and Table 5-12).

5.2.4.2.3 Calculating Chinook Salmon Chronic Aluminum Minimum Effect Threshold Values

The Salmonidae family-level TAF (i.e., EC_{20}:EC_5 adjustment factor) as well as the EC_{20}:EC_{10} and EC_{20}:EC_{15} adjustment factors were applied, as a denominator, to the Salmonidae FMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the Chinook salmon:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) =* ***EC_5*** *= 526.5 µg/L / 1.696 =* ***310.4 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) =* ***EC_{10}*** *= 526.5 µg/L / 1.316 =* ***400.0 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) =* ***EC_{15}*** *= 526.5 µg/L / 1.125 =* ***467.9 µg/L***

5.2.4.2.4 Chinook Salmon: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and vertebrate species sensitivity across water chemistry, Chinook salmon chronic EC_5, EC_{10}, and EC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding Chinook salmon chronic effect threshold values (i.e., EC_5, EC_{10}, EC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and Chinook salmon chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were parsed out by each Chinook salmon ESU. The comparisons between chronic effect threshold data and criterion magnitudes for each Chinook salmon ESU only considered the specific water quality data that were collected from level-III ecoregions within each respective ESU that was assessed. To visually display the comparisons, criterion magnitudes (X-axis) and corresponding chronic effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line for each Chinook salmon ESU (see Figure 5-18 through Figure 5-22). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and Chinook salmon ESU chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) are reported in Table 5-19.

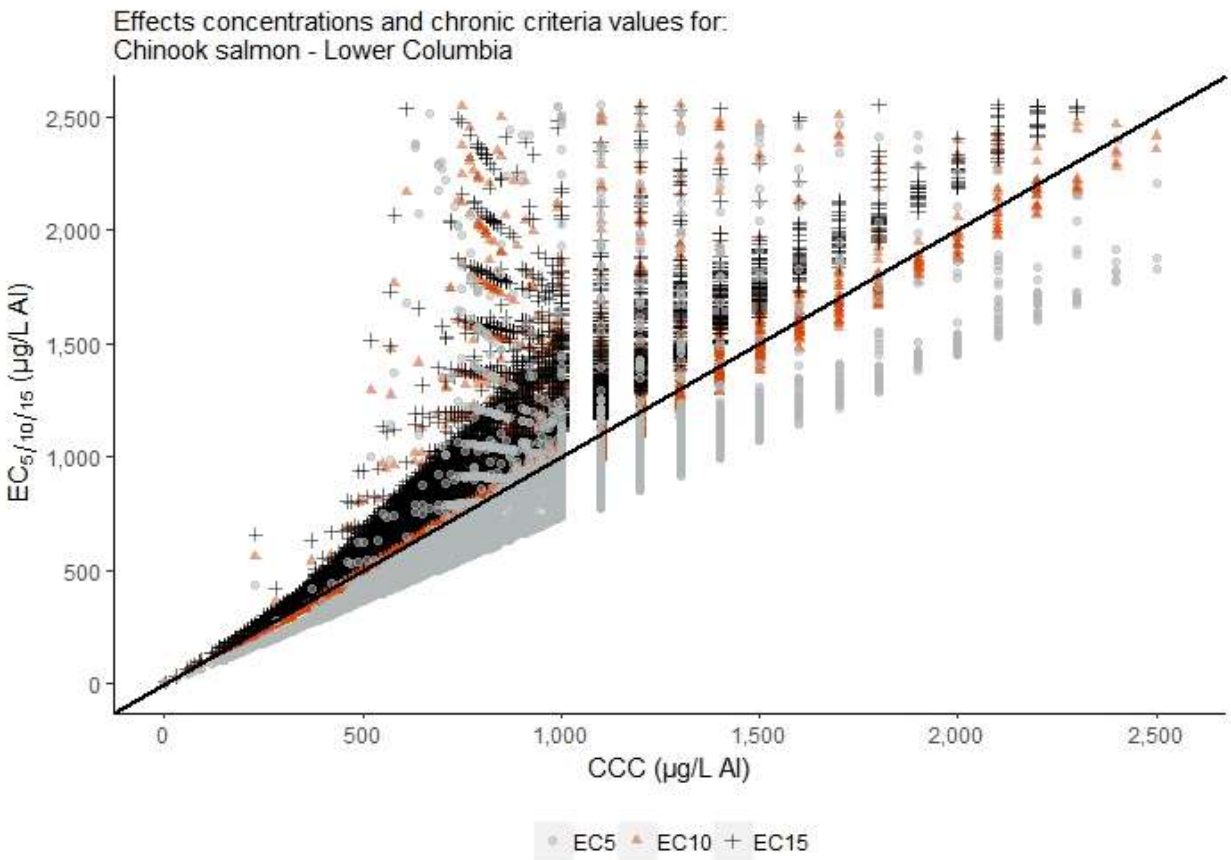

Figure 5-18. Paired Chinook salmon chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the Chinook salmon Lower Columbia River Evolutionarily Significant Unit. Note, 1.3% ($n_{removed}$ = 543) of available data points were removed from Figure 5-17 as the Y axis was scaled down (Y axis maximum = 16,649) to aid in data visualization.

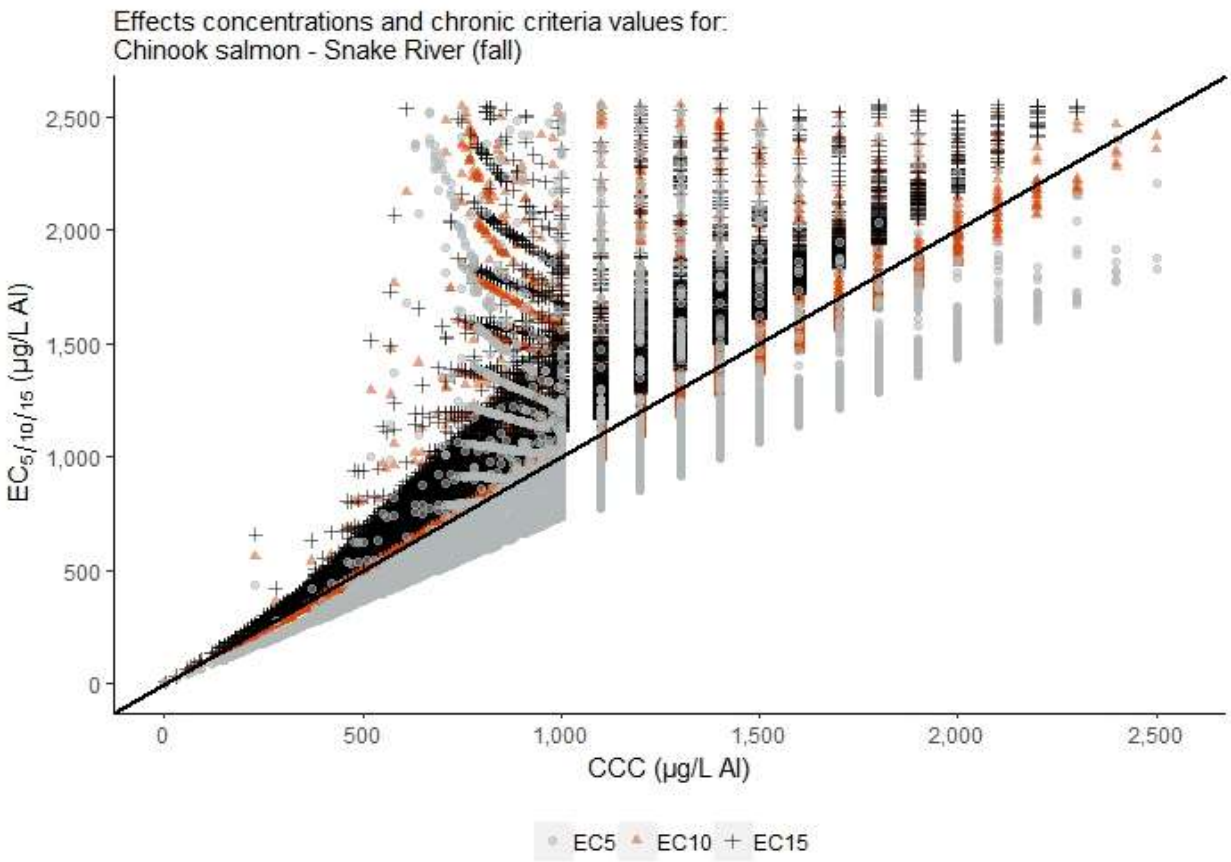

Figure 5-19. Paired Chinook salmon chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the Chinook salmon Snake River (Fall) Evolutionarily Significant Unit. Note, 2.33% ($n_{removed}$ = 1,178) of available data points were removed from Figure 5-18 as the Y axis was scaled down (Y axis maximum = 16,649) to aid in data visualization.

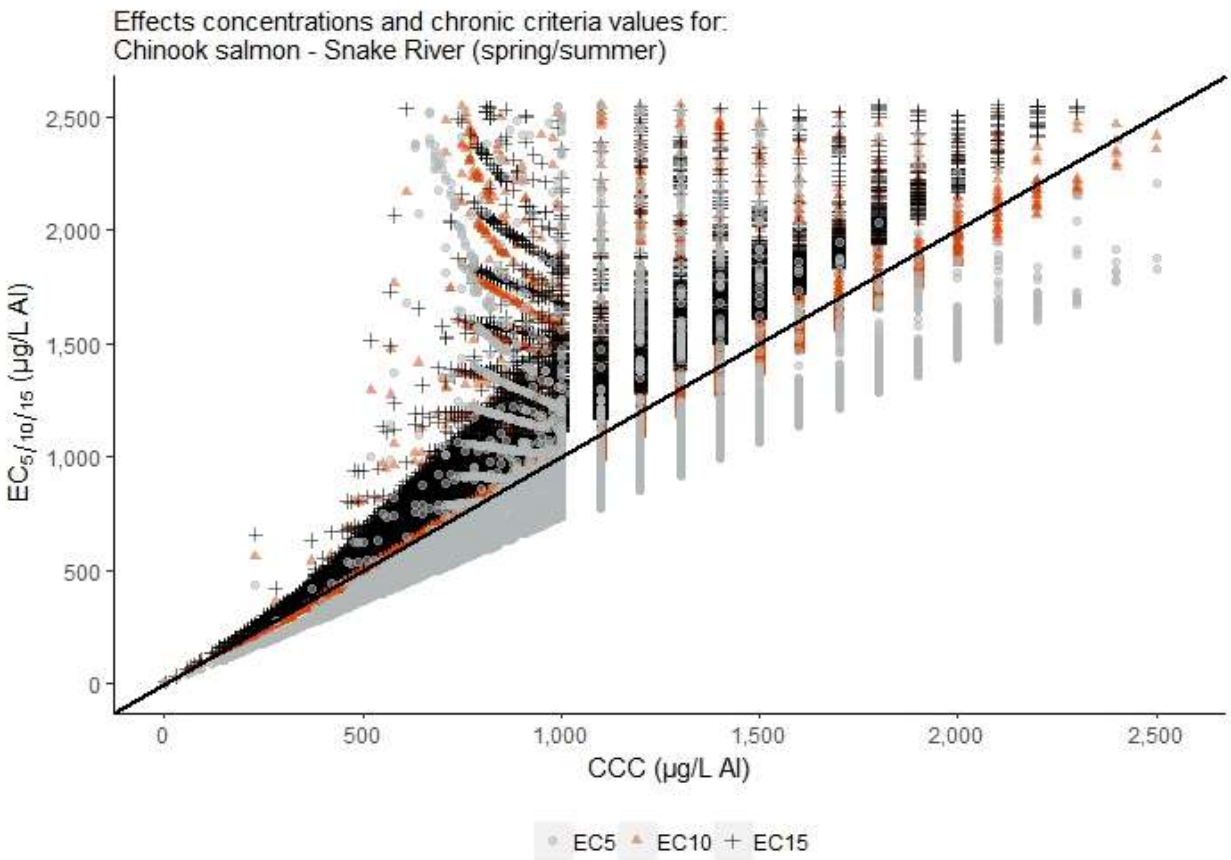

Figure 5-20. Paired Chinook salmon chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the Chinook salmon Snake River (Spring/Summer) Evolutionarily Significant Unit. Note, 2.33% ($n_{removed}$ = 1,178) of available data points were removed from Figure 5-19 as the Y axis was scaled down (Y axis maximum = 16,649) to aid in data visualization.

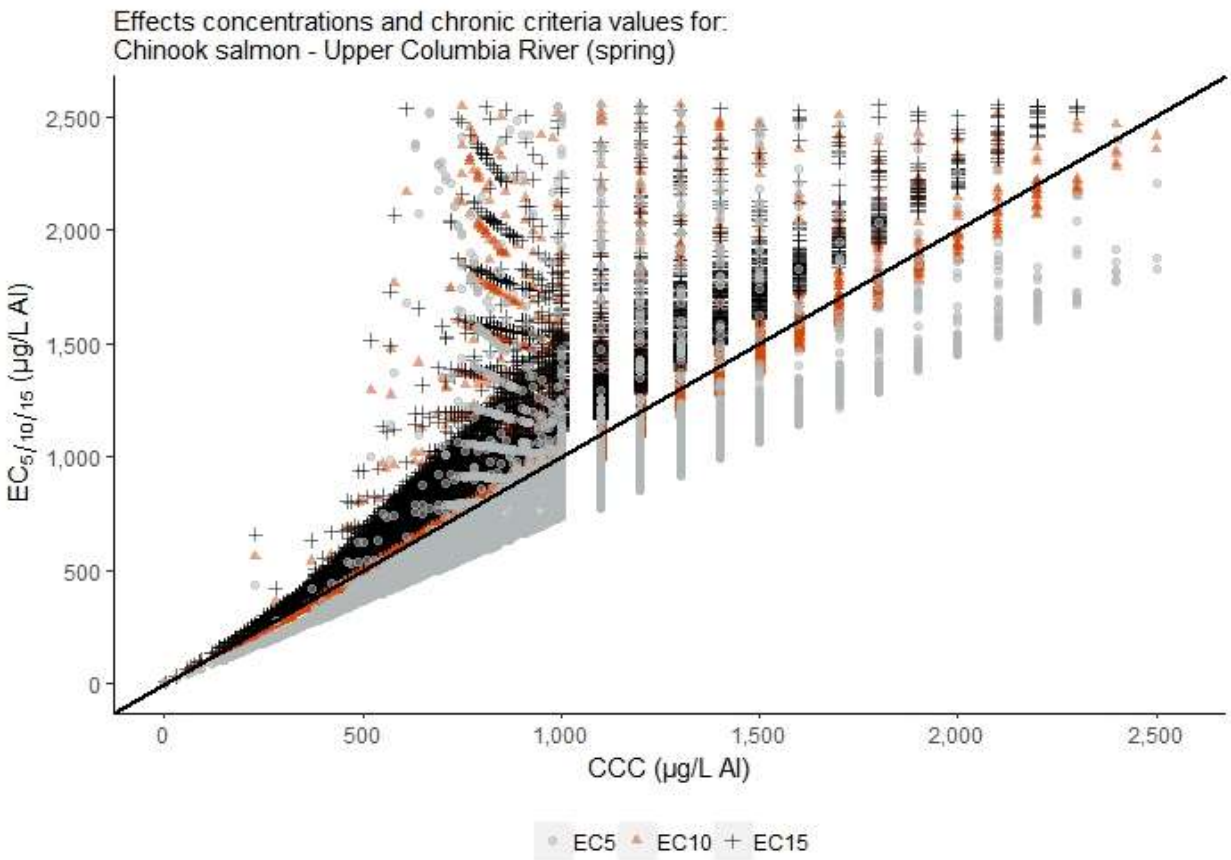

Figure 5-21. Paired Chinook salmon chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the Chinook salmon Upper Columbia River (Spring) Evolutionarily Significant Unit. Note, 1.97% ($n_{removed}$ = 869) of available data points were removed from Figure 5-20 as the Y axis was scaled down (Y axis maximum = 16,649) to aid in data visualization.

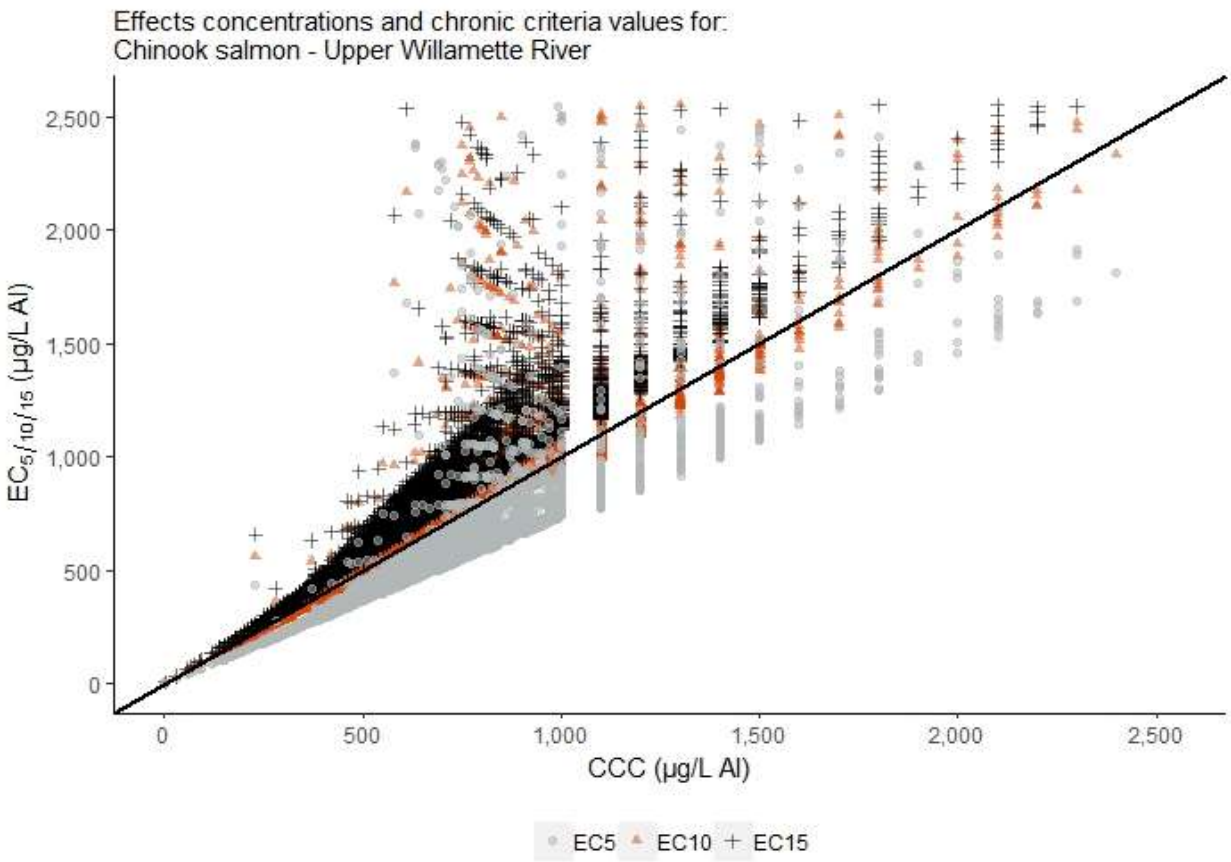

Figure 5-22. Paired Chinook salmon chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the Chinook salmon Upper Willamette River Evolutionarily Significant Unit. Note, 0.87% ($n_{removed}$ = 316) of available data points were removed from Figure 5-21 as the Y axis was scaled down (Y axis maximum = 10,275) to aid in data visualization.

Table 5-19. The number and percentage of occurrences where the chronic criterion magnitude exceed Chinook salmon EC_5, EC_{10}, and EC_{15} values for each Chinook salmon Evolutionarily Significant Unit in Oregon (ESU). Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing the range of each ESU. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (reduced growth/biomass) at each effects concentration.

Evolutionarily Significant Unit	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}			CCC > EC_{15}		
Name		n	%	AEL	n	%	AEL	n	%	AEL
Lower Columbia River	13,973	13,232	94.70	4.74	8,513	60.92	6.09	0	0	0
Snake River (fall)	16,865	15,529	92.08	4.60	9,834	58.31	5.83	0	0	0
Snake River (spring/summer)	16,865	15,529	92.08	4.60	9,834	58.31	5.83	0	0	0
Upper Columbia River (spring)	14,735	13,771	93.46	4.67	8,830	59.93	5.99	0	0	0
Upper Willamette River	12,143	11,714	96.47	4.82	7,735	63.70	6.37	0	0	0

5.2.4.2.5 Chinook Salmon: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC exceeded the Chinook salmon EC_5 between 92.08% – 96.47% of exposure conditions, depending on the specific ESU assessed. The chronic aluminum CCC exceeded the Chinook salmon EC_{10} between 58.31% – 63.70% of exposure conditions, depending on the specific ESU assessed. The chronic Al CCC never exceeded the Chinook salmon EC_{15} (Table 5-19). Because the CCC exceeded the Chinook salmon chronic low effect threshold, a refined exposure assessment through a weight of evidence approach was required for Chinook salmon based on the chronic criterion.

5.2.5 Steelhead (Oncorhynchus mykiss)

5.2.5.1 Steelhead Acute Aluminum Effects Assessment

5.2.5.1.1 Identifying Steelhead Acute Aluminum Data

Species-level acute toxicity data were available for the steelhead (rainbow trout; see Appendix A.1). The steelhead species mean acute value (SMAV) was 3,312 µg/L total aluminum, normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L (see Section 5.2.1.1.1 and Table 5-6).

5.2.5.1.2 Deriving Steelhead Acute Adjustment Factors

No quantitatively acceptable acute TAFs were available for steelhead or any other vertebrate. Qualitatively acceptable C-R models from two acute brook trout tests (see Al-Acute 66 and Al-Acute-68 in Appendix B.1) were, therefore, used to derive a family-level TAF (i.e., LC_{50}:LC_5 adjustment factor) of 1.967, representative of fish in the Family Salmonidae, including steelhead (see Section 5.2.1.1.2 and Table 5-8). Al-Acute-66 and Al-Acute-68 C-R models (see Appendix B.1) were also used to calculate LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors of 1.638 and 1.466, respectively that are representative of steelhead (see Section 5.2.1.1.2 and Table 5-8).

5.2.5.1.3 Calculating Steelhead Acute Aluminum Minimum Effect Threshold Values

The Salmonidae Family-level TAF (i.e., LC_{50}:LC_5 adjustment factor) as well as the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were applied, as a denominator, to the steelhead SMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for the steelhead:

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) = **LC_5** = 3,312 µg/L / 1.967 = **1,683 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) = **LC_{10}** = 3,312 µg/L / 1.638 = **2,022 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) = **LC_{15}** = 3,312 µg/L / 1.466 = **2,260 µg/L***

5.2.5.1.4 Steelhead: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude and vertebrate species sensitivity across water chemistry, steelhead acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate an acute criterion magnitude and corresponding steelhead acute effect threshold values (i.e., LC_5, LC_{10}, LC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and steelhead acute effect threshold data (i.e., LC_5, $LC_{10,}$ and LC_{15}) were parsed out by each steelhead ESU. The comparisons between acute effect threshold data and criterion magnitudes for each steelhead ESU only considered the specific water quality data that were collected from level-III ecoregions within each respective ESU that was assessed. To visually display the comparisons, criterion magnitudes (X-axis) and corresponding acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line for each steelhead ESU (see Figure 5-23 through Figure 5-27). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, $LC_{10,}$ or LC_{15}) and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and each steelhead ESU acute effect threshold data (i.e., LC_5, $LC_{10,}$ LC_{15}) are reported in Table 5-20.

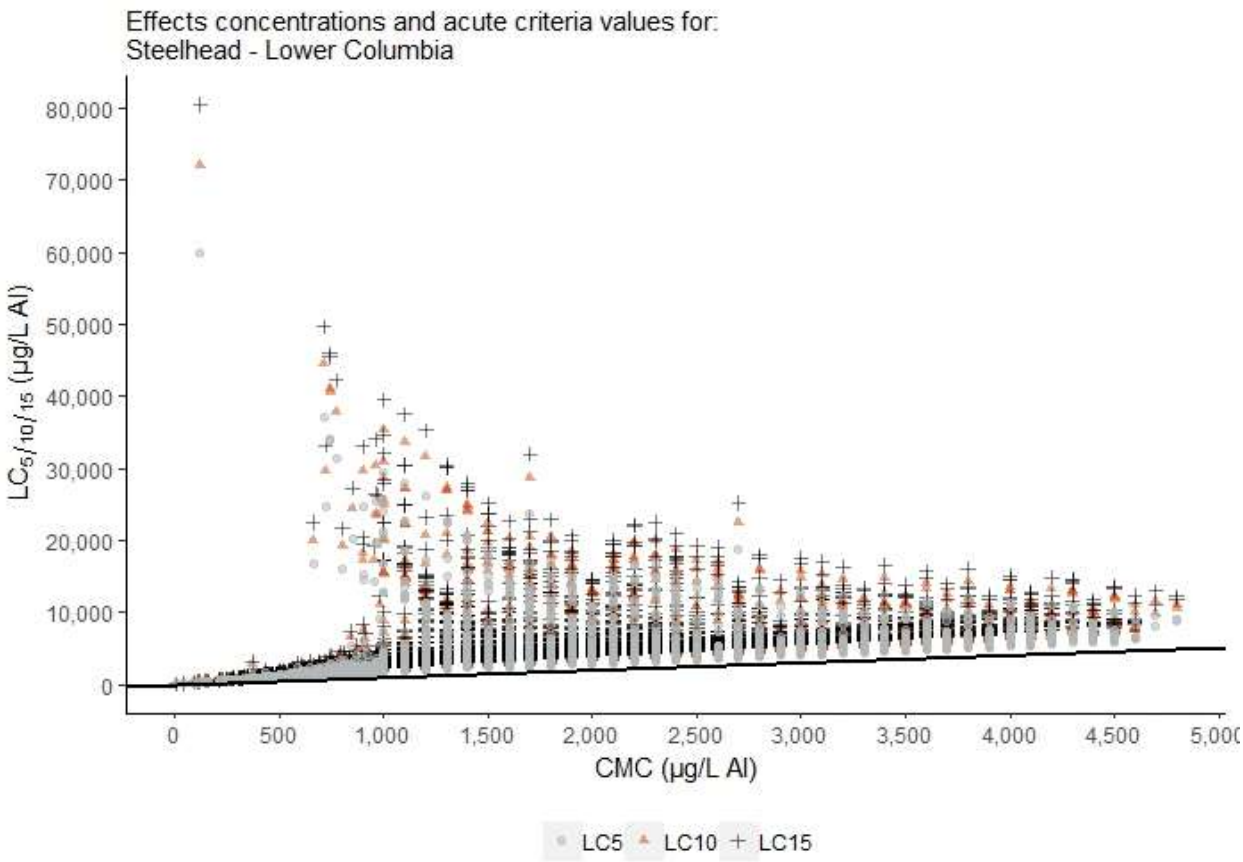

Figure 5-23. Paired steelhead acute effect concentrations and acute criterion values in Oregon ecoregions containing the steelhead Lower Columbia River Evolutionarily Significant Unit.

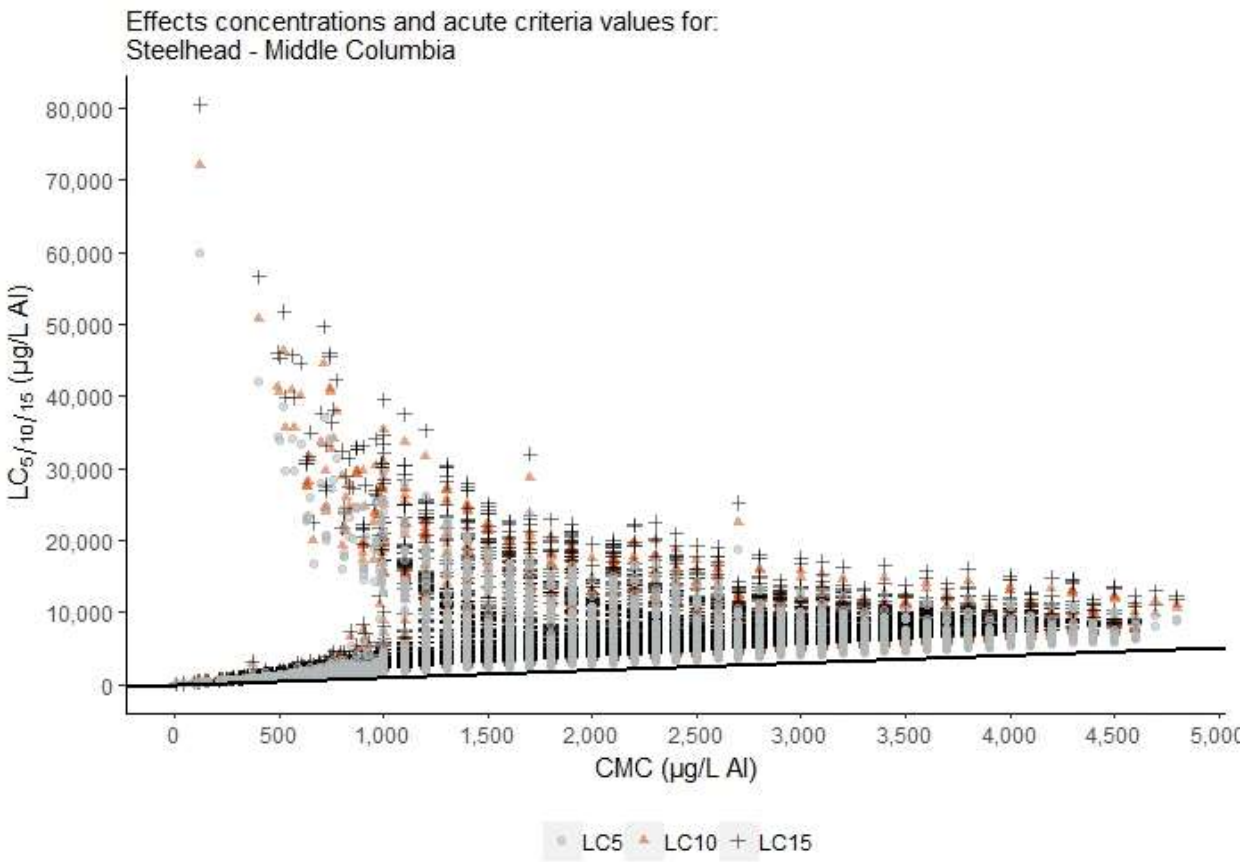

Figure 5-24. Paired steelhead acute effect concentrations and acute criterion values in Oregon ecoregions containing the steelhead Middle Columbia River Evolutionarily Significant Unit.

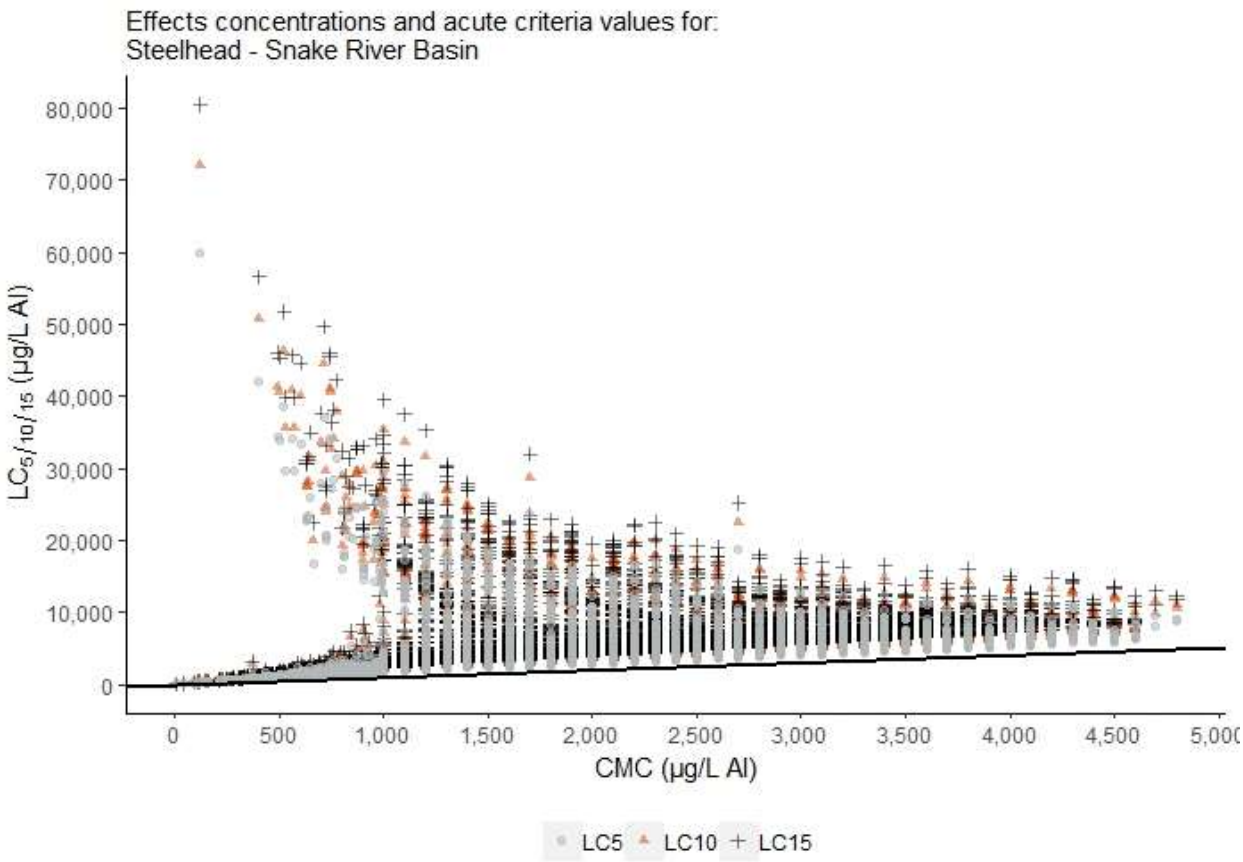

Figure 5-25. Paired steelhead acute effect concentrations and acute criterion values in Oregon ecoregions containing the steelhead Snake River Basin Evolutionarily Significant Unit.

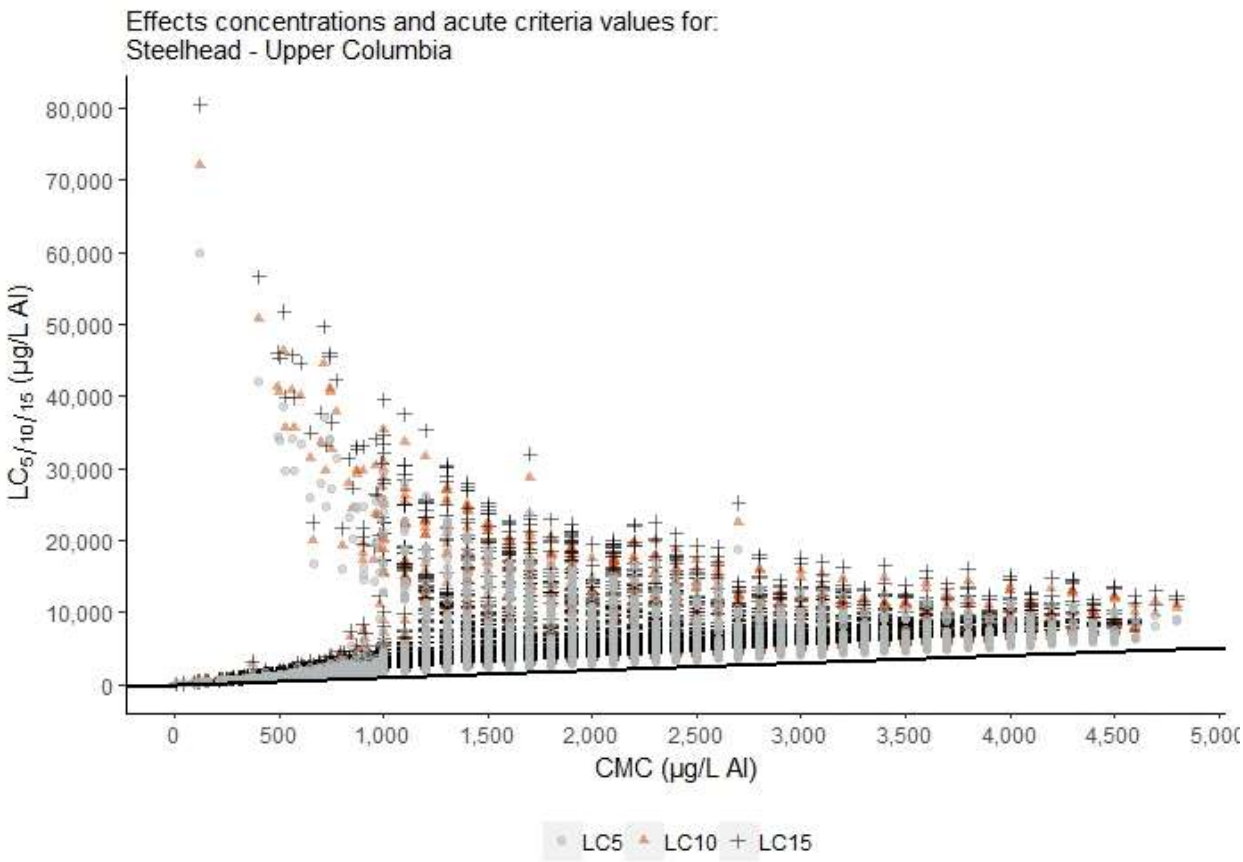

Figure 5-26. Paired steelhead acute effect concentrations and acute criterion values in Oregon ecoregions containing the steelhead Upper Columbia River Evolutionarily Significant Unit.

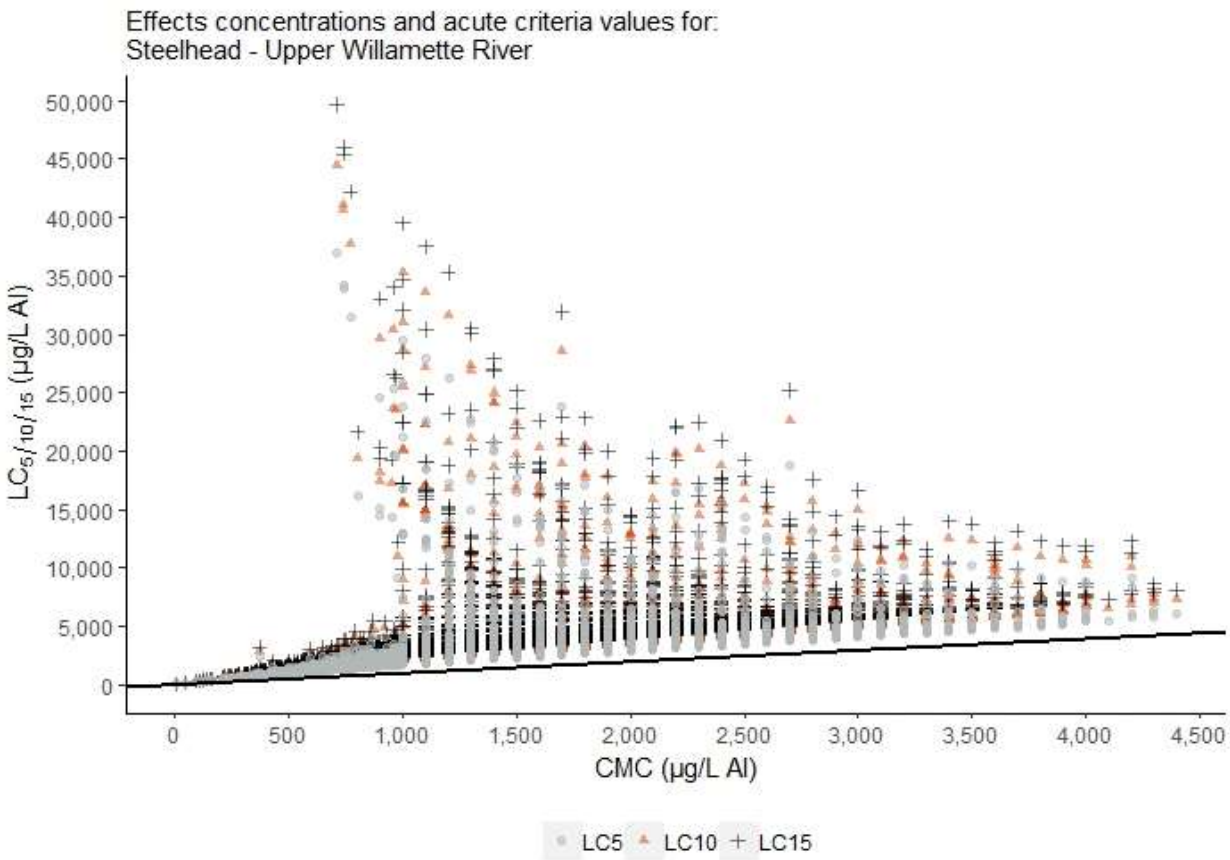

Figure 5-27. Paired steelhead acute effect concentrations and acute criterion values in Oregon ecoregions containing the steelhead Upper Willamette River Evolutionarily Significant Unit.

Table 5-20. The number and percentage of occurrences where the acute criterion magnitude exceed steelhead LC_5, LC_{10}, and LC_{15} values for each steelhead Evolutionarily Significant Units in Oregon (ESU). Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing the range of each ESU. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (mortality) at each effects concentration.

Evolutionarily Significant Unit	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}		CMC > LC_{15}	
Name		n	%	AEL[1]	n	%	n	%
Lower Columbia River	13,973	0	0	0	0	0	0	0
Middle Columbia	16,865	0	0	0	0	0	0	0
Snake River Basin	16,865	0	0	0	0	0	0	0
Upper Columbia River	14,735	0	0	0	0	0	0	0
Upper Willamette River	12,143	0	0	0	0	0	0	0

Notes: 1, AEL shown here applies to each LCx value.

5.2.5.1.5 Steelhead: Tier I Acute Assessment Conclusion

For all ESUs, the acute aluminum CMC was less than the steelhead acute low effect threshold (i.e., LC_5) in all water chemistry scenarios in Oregon ecoregions within steelhead ESUs (Table 5-20). The LC_5 range was 36 to 36,972 µg/L and the CMC range was 4.8 to 4,400 µg/L for the Upper Willamette River ESU. For all other ESUs, the LC_5 range was 36 to 59,906 µg/L and the CMC range was 4.8 to 4,800 µg/L. As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for steelhead based on the acute criterion.

5.2.5.2 Steelhead Chronic Aluminum Effects Assessment

5.2.5.2.1 Identifying Steelhead Chronic Aluminum Data

Quantitatively acceptable chronic aluminum toxicity data were not available for steelhead or any other members within the genus *Oncorhynchus*. Therefore, chronic toxicity data for members of the Family Salmonidae were used to estimate the Salmonidae FMCV of 526.5 µg/L (total aluminum pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) that was representative of steelhead (see Section 5.2.1.2.1 and Table 5-10). EPA also evaluated qualitatively acceptable chronic toxicity data (Appendix H of USEPA (2018)) from several tests with early-life stages of trout to provide comparative information (Table 5-11 provides a summary of these tests). Table 5-11 specifically contains four steelhead (rainbow trout) tests that document death and deformity of early life stages occurring at a concentration as low as 457.5 µg/L after 28-day exposures to no observable effects at concentration as high as 2,544 µg/L after 6.5-day exposures[38]. Overall, results of these qualitatively acceptable chronic toxicity tests appear to support the Salmonidae FMCV; however, the range in normalized no effect and effect concentrations, as well as endpoint variability, also indicate the potential for epistemic uncertainty and natural variability that can confound estimates of steelhead sensitivity to chronic aluminum exposures.

5.2.5.2.2 Deriving Steelhead Chronic Adjustment Factors

No quantitatively acceptable chronic TAFs were available for steelhead or any other member of the Order Salmoniformes. The only quantitatively acceptable C-R model available for a vertebrate species was based on the fathead minnow. As a result, a family-level TAF was calculated from a brook trout EC_{20}:EC_5 ratio that was originally determined to be qualitatively acceptable C-R curve fit (see Appendix B.2, curve Al-Chronic-9 and Section 5.2.1.2.2). The chronic salmonid family-level TAF (i.e., EC_{20}:EC_5) was 1.696 and the corresponding EC_{20}:EC_{10} ratio and the EC_{20}:EC_{15} ratio from curve Al-Chronic-9 was 1.316 and 1.125, respectively (see Section 5.2.1.2.2 and Table 5-12).

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[38] Rainbow trout tests in Table 2-5 were excluded from criteria derivation due to unacceptable test durations and a lack of exposure details (Stephen et al., 1985). Reported effect concentrations are represented as total aluminum in a water chemistry of: pH = 7, total hardness = 100 mg/L as $CaCO_3$, and DOC = 1 mg/L.

5.2.5.2.3 Calculating Steelhead Chronic Aluminum Minimum Effect Threshold Values

The Salmonidae family-level TAF (i.e., EC_{20}:EC_5 adjustment factor) as well as the EC_{20}:EC_{10} and EC_{20}:EC_{15} adjustment factors were applied, as a denominator, to the Salmonidae FMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the steelhead:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) = **EC_5** = 526.5 µg/L / 1.696 =* ***310.4 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) = **EC_{10}** = 526.5 µg/L / 1.316 =* ***400.0 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) = **EC_{15}** = 526.5 µg/L / 1.125 =* ***467.9 µg/L***

5.2.5.2.4 Steelhead: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and vertebrate species sensitivity across water chemistry, steelhead chronic EC_5, EC_{10}, and EC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding steelhead chronic effect threshold values (i.e., EC_5, EC_{10}, EC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and steelhead chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were parsed out by each steelhead ESU. The comparisons between chronic effect threshold data and criterion magnitudes for each steelhead ESU only considered the specific water quality data that were collected from level-III ecoregions within each respective ESU that was assessed. To visually display the comparisons, criterion magnitudes (X-axis) and corresponding chronic effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line for each steelhead ESU (see Figure 5-28 through Figure 5-32). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and steelhead ESU chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) are reported in Table 5-21.

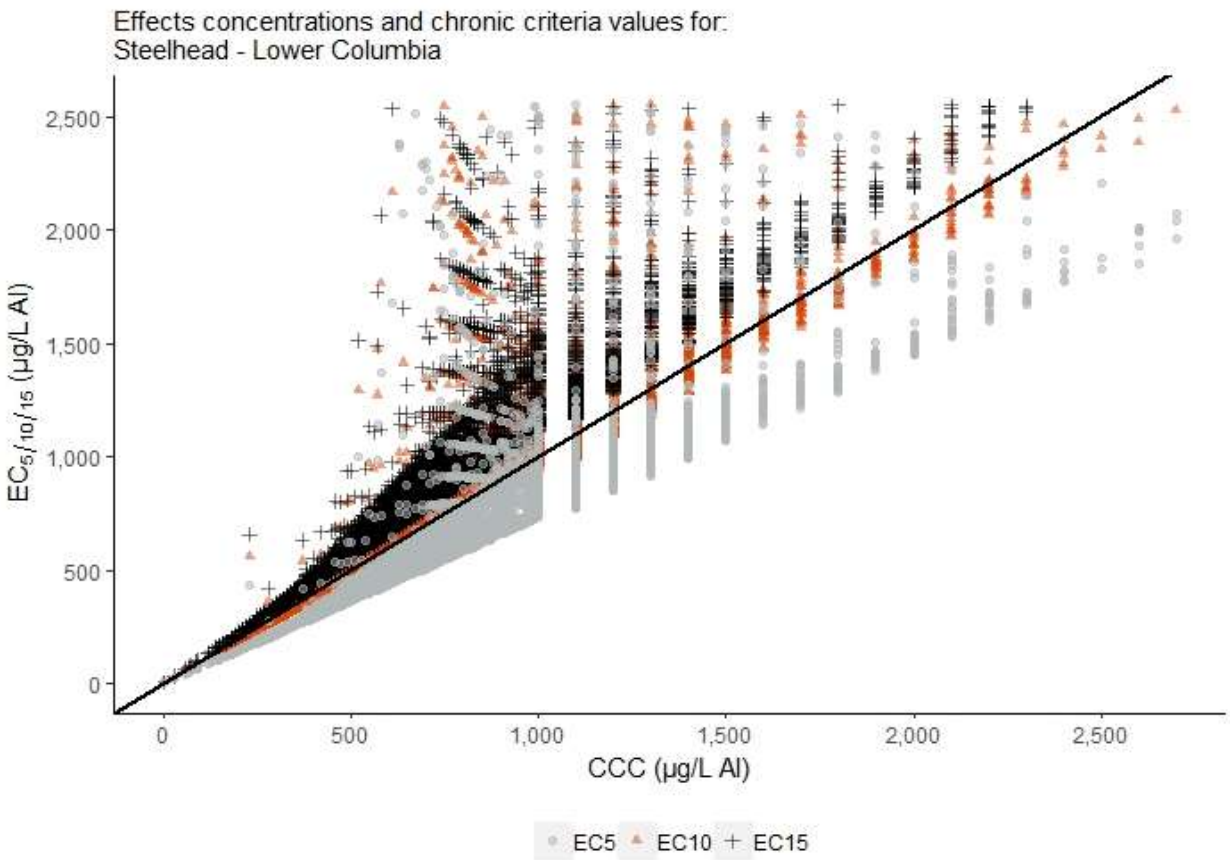

Figure 5-28. Paired steelhead chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the steelhead Lower Columbia River Evolutionarily Significant Unit. Note, 1.27% ($n_{removed}$ = 531) of available data points were removed from Figure 5-27 as the Y axis was scaled down (Y axis maximum = 16,649) to aid in data visualization.

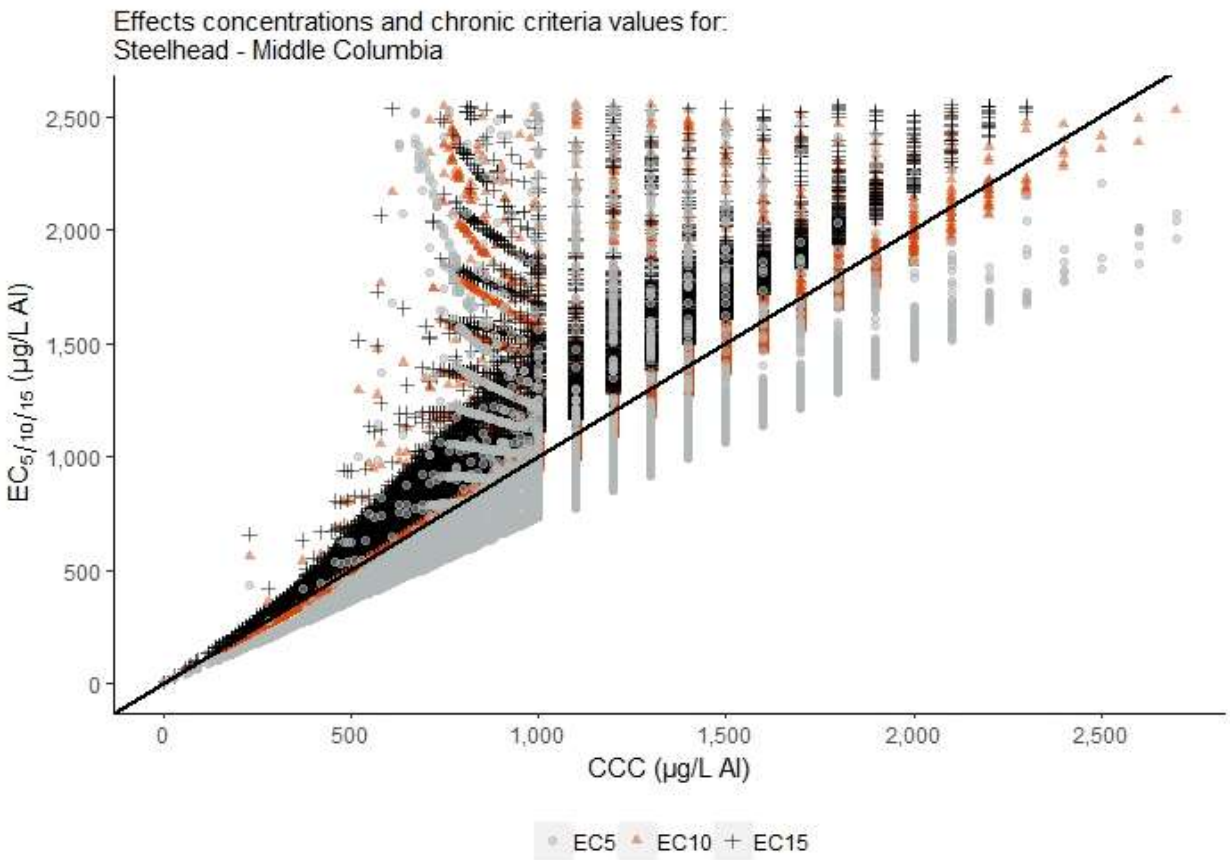

Figure 5-29. Paired steelhead chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the steelhead Middle Columbia River Evolutionarily Significant Unit. Note, 2.3% ($n_{removed}$ = 1,166) of available data points were removed from Figure 5-28 as the Y axis was scaled down (Y axis maximum = 16,649) to aid in data visualization.

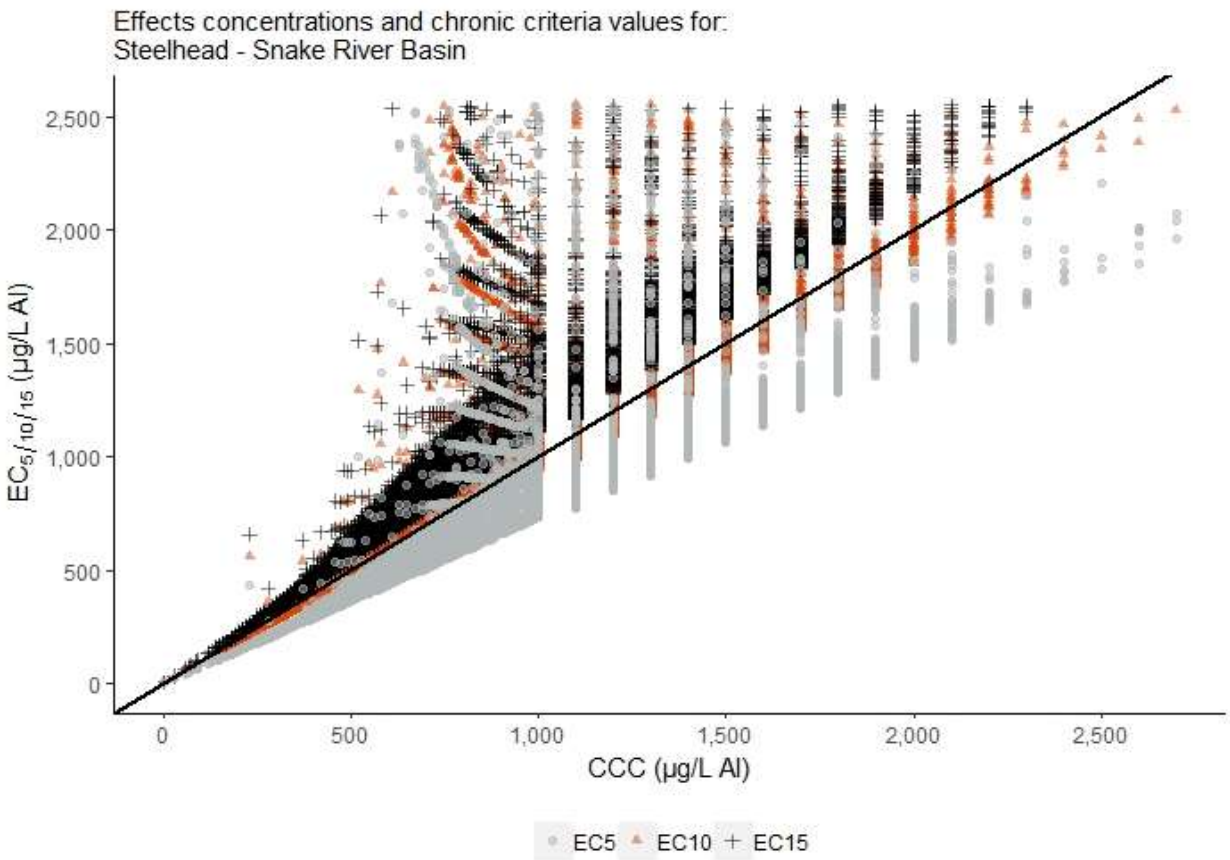

Figure 5-30. Paired steelhead chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the steelhead Snake River Basin Evolutionarily Significant Unit. Note, 2.3% ($n_{removed}$ = 1,166) of available data points were removed from Figure 5-29 as the Y axis was scaled down (Y axis maximum = 16,649) to aid in data visualization.

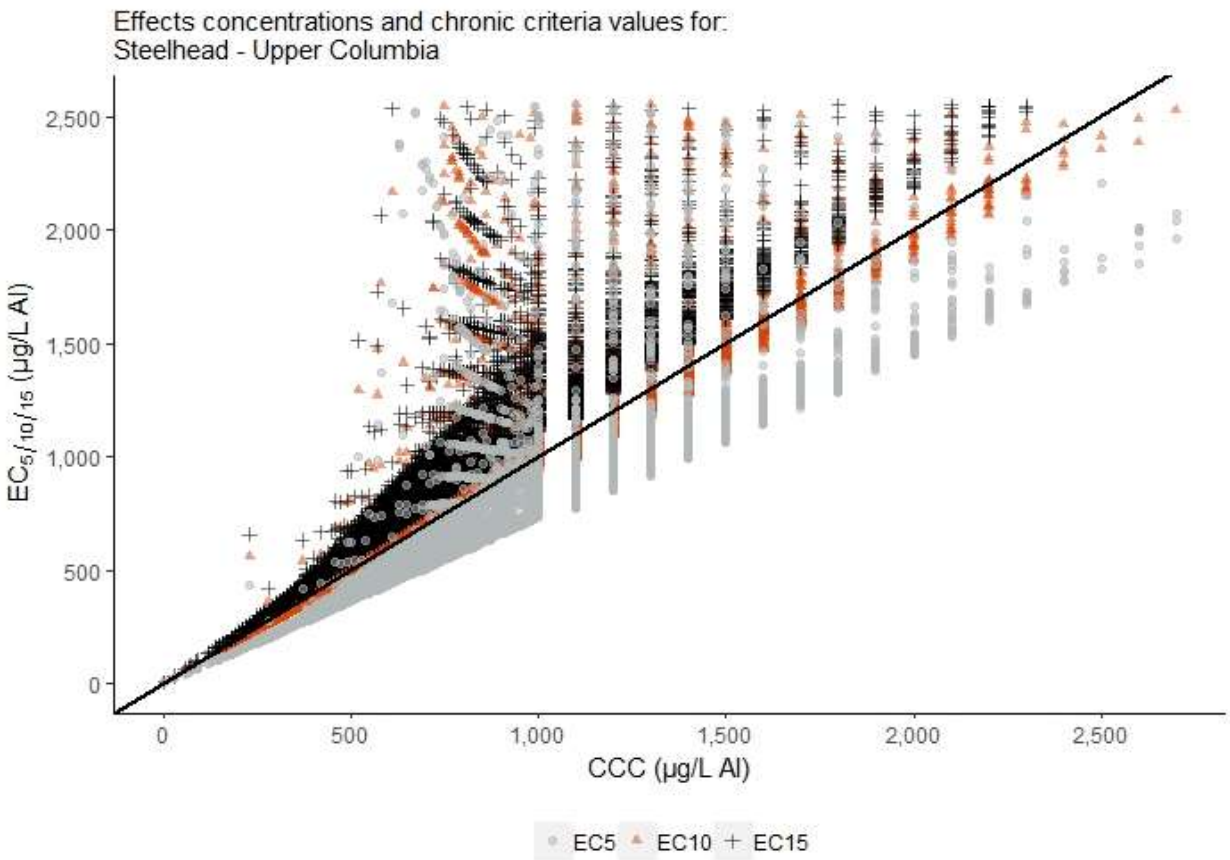

Figure 5-31. Paired steelhead chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the steelhead Upper Columbia River Evolutionarily Significant Unit. Note, 1.94% ($n_{removed}$ = 857) of available data points were removed from Figure 5-30 as the Y axis was scaled down (Y axis maximum = 16,649) to aid in data visualization.

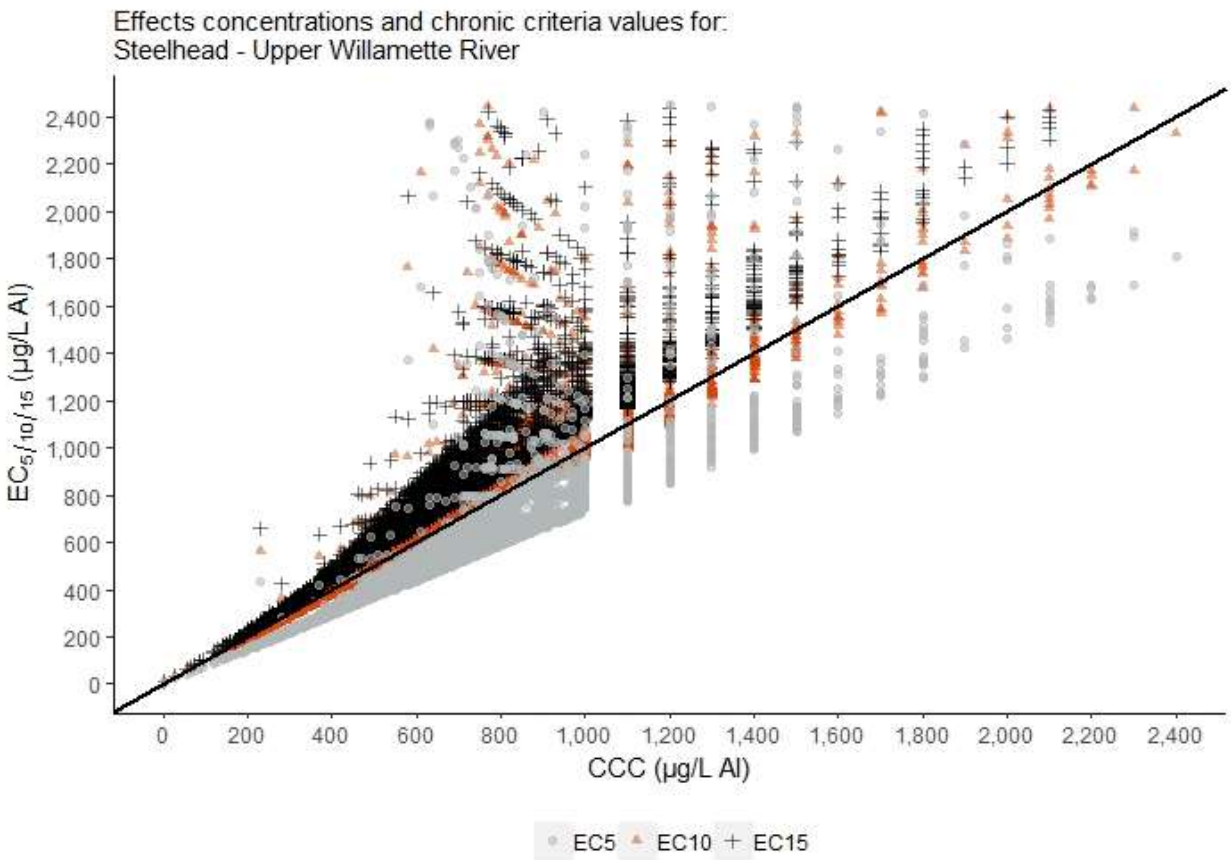

Figure 5-32. Paired steelhead chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the steelhead Upper Willamette River Evolutionarily Significant Unit. Note, 0.95% ($n_{removed}$ = 347) of available data points were removed from Figure 5-31 as the Y axis was scaled down (Y axis maximum = 10,275) to aid in data visualization.

Table 5-21. The number and percentage of occurrences where the chronic criterion magnitude exceed steelhead EC_5, EC_{10}, and EC_{15} values for each steelhead Evolutionarily Significant Unit in Oregon (ESU). Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing the range of each ESU. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (growth/biomass) at each effects concentration.

Evolutionarily Significant Unit	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}			CCC > EC_{15}		
Name		n	%	AEL	n	%	AEL	n	%	AEL
Lower Columbia River	13,973	13,232	94.70	4.74	8,513	60.92	6.09	0	0	0
Middle Columbia River	16,865	15,529	92.08	4.60	9,834	58.31	5.83	0	0	0
Snake River Basin	16,865	15,529	92.08	4.60	9,834	58.31	5.83	0	0	0
Upper Columbia River	14,735	13,771	93.46	4.67	8,830	59.93	5.99	0	0	0
Upper Willamette River	12,143	11,714	96.47	4.82	7,735	63.70	6.37	0	0	0

5.2.5.2.5 Steelhead: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC exceeded the steelhead EC_5 between 92.08% – 96.47% of exposure conditions, depending on the specific ESU assessed. The chronic aluminum CCC exceeded the steelhead EC_{10} between 58.31% – 63.70% of exposure conditions, depending on the specific ESU assessed. The chronic Al CCC never exceeded the steelhead EC_{15} (Table 5-21). Because the CCC exceeded the steelhead chronic low effect threshold, consideration of a refined exposure assessment through a weight of evidence approach was required for steelhead based on the chronic criterion.

5.2.6 Green Sturgeon (Acipenser medirostris)

5.2.6.1 Green Sturgeon Aluminum Acute Effects Assessment

5.2.6.1.1 Identifying Green Sturgeon Acute Aluminum Data

Acute aluminum toxicity data from the EPA's 2018 aluminum aquatic life criteria document were not available for any species within the Order Acipenseriformes (see Appendix A.1). Therefore, a representative acute toxicity value was estimated from the traditional module of EPA's web-based Interspecies Correlation Estimation (web-ICE) program[39]. Web-ICE did not contain a green sturgeon-specific model, but did contain an *Acipenser* model, capable of predicting acute Al toxicity to members of the genus *Acipenser* from as many as 15 surrogate species. Of the 15 surrogate species, only rainbow trout and fathead minnow had quantitatively acceptable SMAVs reported in Appendix A of USEPA (2018); to be used as input data (see Appendix A.1). The rainbow trout (surrogate)-to-*Acipenser* (predicted) ICE model was

[39] Web-ICE, operated using the standard Aquatic module; https://www3.epa.gov/webice/

determined to be more robust than fathead minnow (surrogate)-to-*Acipenser* (predicted) ICE model based on model selection guidance reported by Willming, Lilavois, Barron, and Raimondo (2016); see Appendix D.1 for ICE model parameters).

The rainbow trout SMAV (3,312 µg/L total aluminum, normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) was input into the rainbow trout (surrogate)-to-*Acipenser* (predicted) ICE model to determine an *Acipenser* GMAV of 3,593 µg/L that was representative of green sturgeon. Because the input toxicity value (i.e., rainbow trout SMAV) was normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L, the resultant output toxicity value was representative of *Acipenser* acute sensitivity to Al in that same water chemistry.

5.2.6.1.2 Deriving Green Sturgeon Acute Adjustment Factors

Because no raw empirical acute toxicity data were available for fish species in the same order as the green sturgeon (Acipenseriformes), no acute order-level TAF could be calculated. As a result, EPA obtained and analyzed C-R data for all acute tests in the USEPA (2018) acute aluminum dataset where such data were reported or could be obtained to derive an acute vertebrate TAF or acute MAF, if necessary (i.e., if the vertebrate and invertebrate acute TAFs do not differ from one another). As discussed in Section 5.1.2.1.1, no quantitatively acceptable LC_{50}:LC_5 ratios for vertebrates were available. Also, there were no qualitatively acceptable models (up to the order level) that can be used to estimate minimum effect concentrations (as in the case of salmonids). Thus, the acute MAF was used to estimate green sturgeon low acute effect concentrations (see Table 5-4). The application of the MAF to the green sturgeon (and other non-salmonid vertebrates) was a conservative approach because the MAF (3.881) was greater than the best estimate of an adjustment factor based on vertebrate species (i.e., Salmonidae-TAF = 1.967). Nevertheless, applying invertebrate-based surrogacy to represent vertebrate species solely for the sake of conservatism is advised against, considering the physiological differences between vertebrates and invertebrates. Therefore, if the application of the MAF suggested the green sturgeon (<u>or any other non-salmonid invertebrate species</u>) was acutely sensitive to Al exposures at the acute criterion magnitude, then EPA also calculated LC_5, LC_{10}, and LC_{15} values through the application of the Salmonidae-TAF to present results that were based less on conservative assumptions and more on phylogenetic relatedness.

5.2.6.1.3 Calculating Green Sturgeon Acute Aluminum Minimum Effect Threshold Values

The MAF (i.e., LC_{50}:LC_5 adjustment factor) as well as the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were applied, as a denominator, to the *Acipenser* GMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for the green sturgeon

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) = **LC_5** = 3,593 µg/L / 3.881 = **925.6 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) = **LC_{10}** = 3,593 µg/L / 2.787 = **1,289 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) = **LC_{15}** = 3,593 µg/L / 2.261 = **1,589 µg/L***

5.2.6.1.4 Green Sturgeon: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude and vertebrate species sensitivity across water chemistry, green sturgeon acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate an acute criterion magnitude and corresponding green sturgeon acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and green sturgeon acute effect threshold data (i.e., LC_5, LC_{10}, and LC_{15}) were compared to the green sturgeon Southern DPS. The comparison between acute effect threshold data and criterion magnitudes for the green sturgeon Southern DPS only considered the specific water quality data that were collected from level-III ecoregions within the green sturgeon Southern DPS. To visually display the comparison, criterion magnitudes (X-axis) and corresponding acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line for the green sturgeon southern DPS (see Figure 5-33). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, LC_{10}, or LC_{15}) and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and green sturgeon Southern DPS acute effect threshold data (i.e., LC_5, LC_{10}, LC_{15}) are reported in Table 5-22. The LC_5 range was 194 to 9,193 µg/L and the CMC range was 140 to 4,300 µg/L.

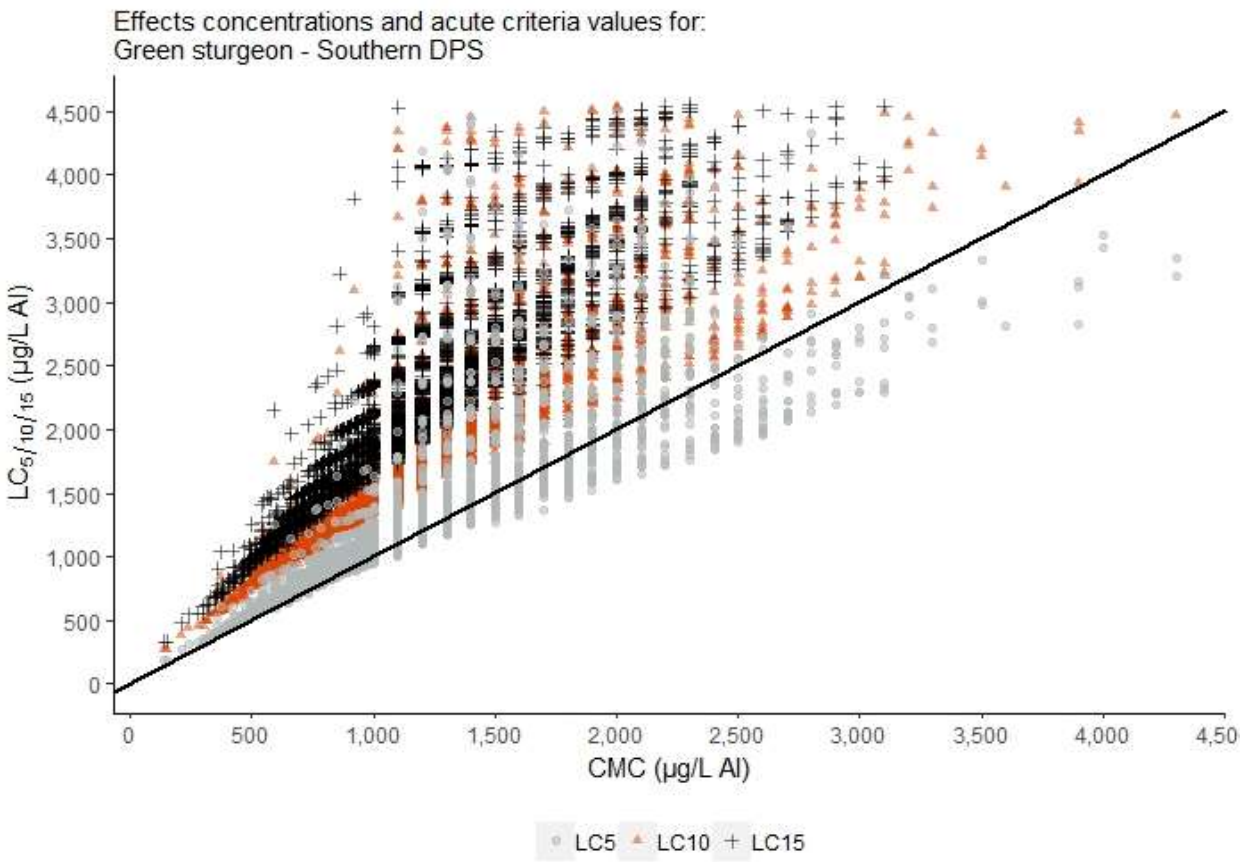

Figure 5-33. Paired green sturgeon acute effect concentrations and acute criterion values in Oregon ecoregions containing the green sturgeon Southern Distinct Population Segment. Note, 2.96% ($n_{removed}$ = 423) of available data points were removed from Figure 2-31 as the Y axis was scaled down (Y axis maximum = 15,781) to aid in data visualization.

Table 5-22. The number and percentage of occurrences where the acute criterion magnitude exceed green sturgeon LC_5, LC_{10}, and LC_{15} values for the Southern Distinct Population Segment (DPS). Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing the range of the Southern DPS. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (mortality) at each effects concentration.

Distinct Population Segment	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}			CMC > LC_{15}		
Name		n	%	AEL	N	%	AEL	n	%	AEL
Southern DPS	4,768	779	16.34	0.82	0	0	0	0	0	0

5.2.6.1.5 Green Sturgeon: Tier I Acute Assessment Conclusion and Additional Assessment

The acute aluminum CMC was greater than the green sturgeon acute low effect threshold (i.e., LC_5) in 16.34% of water chemistry scenarios, but never exceeded the green sturgeon LC_{10} (Table 5-22). The estimates of the green sturgeon acute low effect threshold (i.e., LC_5), LC_{10}, and LC_{15} were considered relatively conservative because the green sturgeon LC_{50} value was transformed to LC_5, LC_{10}, and LC_{15} estimates through adjustment factors based on invertebrate taxa only (see Section 5.1.2.1.1).

For the green sturgeon and other non-salmonid vertebrates, adjustment factors based on invertebrates only were used initially because the only quantitatively-acceptable curves available were from acute invertebrate tests. Application of invertebrate-based adjustment factors to non-salmonid vertebrates was conservative because invertebrate-based adjustment factors were greater than the best estimates of LC_{50}:LC_5, LC_{50}:LC_{10}, and LC_{50}:LC_{15} adjustment factors based on vertebrate species (i.e., Salmonidae-based adjustment factors based on qualitatively-acceptable C-R curves; see Section 5.2.1.1.2). However, applying invertebrate-based surrogacy to represent green sturgeon species solely for the sake of conservatism may be inappropriate considering the physiological differences between vertebrates and invertebrates. Therefore, EPA also calculated green sturgeon LC_5, LC_{10}, and LC_{15} by applying the Salmonidae-based adjustment factors to the representative green sturgeon LC_{50} value:

- *LC_{50}/(Salmonidae LC_{50}:LC_5 adjustment factor) =* ***LC_5*** *= 3,593 µg/L / 1.967 =* ***1,826 µg/L***
- *LC_{50}/(Salmonidae LC_{50}:LC_{10} adjustment factor) =* ***LC_{10}*** *= 3,593 µg/L / 1.638 =* ***2,193 µg/L***
- *LC_{50}/(Salmonidae LC_{50}:LC_{15} adjustment factor) =* ***LC_{15}*** *= 3,593 µg/L / 1.466 =* ***2,451 µg/L***

Although these green sturgeon LC_5, LC_{10}, and LC_{15} are not based on conservative assumptions, they are expected to better represent the within-species variation in sensitivity (i.e., shape and slope of the concentration response curve) of green sturgeon because they are based on vertebrate responses. These refined green sturgeon acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate an acute criterion magnitude and corresponding refined green sturgeon acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for each set of paired water chemistry data collected within level-III Oregon ecoregions that contain the green sturgeon Southern DPS.

The refined LC_5, LC_{10}, and LC_{15} estimates for green sturgeon were always greater corresponding acute criterion magnitude (Figure 5-34), suggesting the use of invertebrate-based adjustment factors may overestimate green sturgeon sensitivity in an attempt to account for data limitations. Although the refined LC_5, LC_{10}, and LC_{15} estimates for green sturgeon are based on a class-level adjustment factors, and the appropriateness of surrogacy beyond the order remained unclear, green sturgeon were concluded to be tolerant to acute Al exposures because; 1) only 16.34% of acute criterion magnitudes exceeded the green sturgeon LC_5 value that was based on the conservative LC_{50}:LC_5 adjustment factor calculated from invertebrates (Figure 5-33) and 2) criterion magnitudes were always less than the green sturgeon LC_5 values that was based on the salmonidae LC_{50}:LC_5 adjustment factor. The LC_5 range was 383 to 18,140 µg/L and the CMC range was 140 to 4,300 µg/L. As a result, consideration of a refined exposure assessment through a weight of evidence approach was required for green sturgeon based on the acute criterion. However, EPA notes how apparent sensitivity of green sturgeon to Al is somewhat

dependent on the choice of the LC_{50}:LC_5 adjustment factor, which is due to subtleties in choices made in the selection of the best fitting most representative concentration-response curves.

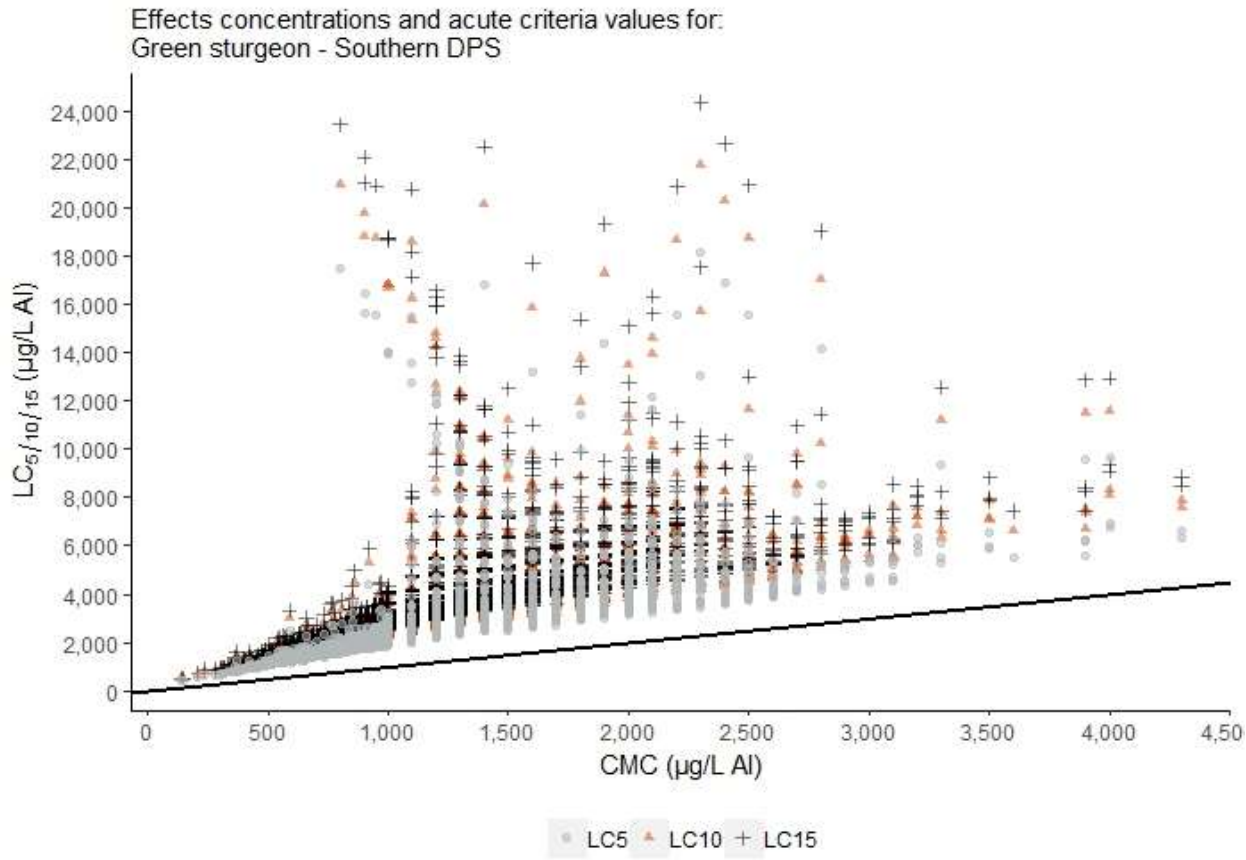

Figure 5-34. Paired green sturgeon acute effect concentrations (calculated using the salmonidae adjustment factors) and acute criterion values in Oregon ecoregions containing the green sturgeon Southern Distinct Population Segment.

5.2.6.2 Green Sturgeon Aluminum Chronic Effects Assessment

5.2.6.2.1 Identifying Green Sturgeon Chronic Aluminum Data

Chronic aluminum toxicity data were not available for green sturgeon or any other species within the Order Acipenseriformes. As a result, the *Acipenser* GMAV (see Section 5.2.6.1.1) that was representative of green sturgeon was transformed to represent a chronic toxicity value (i.e., EC_{20}). Specifically, the *Acipenser* GMAV (i.e., LC_{50} = 3,593 µg/L total aluminum, normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) was divided by an acute-to-chronic ratio (ACR). An ACR is intended to broadly relate a species acute effect concentration to an estimated chronic effect concentration. All acute toxicity tests (Appendix A.1) that had paired chronic toxicity tests (Appendix A.2) conducted using the same species in the same lab are compiled in Table 5-23. The vertebrate-specific ACR (5.651) was based on a single ACR from Kimball (1978) for the fathead minnow, *Pimephales promelas* (Table 5-23). USEPA (2018) states, "*Kimball (1978) conducted an early life stage test using fathead minnow (Pimephales.*

promelas) fertilized eggs (16 to 40-hr old) in flowing hard well water. Six treatments of aluminum sulfate plus control replicated four times were used to expose fish for 28 days posthatch, and aluminum concentrations were measured three times per week during the study. Biomass was more sensitive to the aluminum exposures than percent hatchability, growth and survival, with a resulting EC_{20} of 6,194 µg/L, or 2,690 µg/L when normalized." The fathead minnow ACR (and by translation the vertebrate ACR for aluminum) numerator was based on an LC_{50} value and the denominator was based on an EC_{20} value. Therefore, application of the vertebrate ACR to the *Acipenser* GMAV results in an estimated *Acipenser* GMCV that was equivalent to an EC_{20}:

- *Acipenser GMAV / Vertebrate ACR =* ***EC_{20}*** *= 3,593 µg/L / 5.651 = 635.8* ***µg/L***

The *Acipenser* GMCV was 635.8 µg/L total aluminum (normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) and represents green sturgeon sensitivity to chronic Al exposures.

Table 5-23. Acute-to-Chronic Ratios for Aluminum in Freshwater from USEPA (2018).

Data Type	Order	Family	Species	Endpoint	Reference	Effect Concentration (µg/L)	Test-specific ACR	Species ACR	Invert. / Vert. ACR
Chronic	Unionoida	Unionidae	Fatmucket, *Lampsilis siliquoidea*	Biomass	(Wang et al., 2016; Wang et al., 2018)	169.00	>37.29	>37.29	9.089
Acute				LC_{50}	(Wang et al., 2016; Wang et al., 2018)	>6,302			
Chronic	Diplostraca	Daphniidae	Cladoceran, *Ceriodaphnia dubia*	Reproduction - young/starting adult	(McCauley, Brooke, Call, & Lindberg, 1986)	1,780	1.067	1.427	
Acute				LC_{50}	(McCauley et al., 1986)	1,900			
Chronic	Diplostraca	Daphniidae	Cladoceran, *Ceriodaphnia dubia*	MATC (reproduction - young/starting adult)	(McCauley et al., 1986)	<1,100	<1.364		
Acute				LC_{50}	(McCauley et al., 1986)	1,500			
Chronic	Diplostraca	Daphniidae	Cladoceran, *Ceriodaphnia dubia*	Reproduction - young/female	(ENSR, 1992b)	1,557	0.4623		
Acute				LC_{50}	(ENSR, 1992a)	720			
Chronic	Diplostraca	Daphniidae	Cladoceran, *Ceriodaphnia dubia*	Reproduction - young/female	(ENSR, 1992b)	808.7	2.325		
Acute				LC_{50}	(ENSR, 1992a)	1,880			
Chronic	Diplostraca	Daphniidae	Cladoceran, *Ceriodaphnia dubia*	Reproduction - young/female	(ENSR, 1992b)	647.2	3.785		
Acute				LC_{50}	(ENSR, 1992a)	2,450			
Chronic	Diplostraca	Daphniidae	Cladoceran, *Ceriodaphnia dubia*	Reproduction - young/female	(ENSR, 1992b)	683.6	>145.7[a]		
Acute				LC_{50}	(ENSR, 1992a)	>99,600			
Chronic	Amphipoda	Hyalellidae	Amphipod, *Hyalella azteca*	Biomass	(Wang et al., 2016; Wang et al., 2018)	425	>14.11	>14.11	
Acute				LC_{50}	(Wang et al., 2016; Wang et al., 2018)	>5,997			
Chronic	Cypriniformes	Cyprinidae	Fathead minnow, *Pimephales promelas*	Biomass	(Kimball, 1978)	6,194	5.651	5.651	5.651
Acute				LC_{50}	(Kimball, 1978)	35,000[b]			

[a] Not used to calculate *C. dubia* ACR because value was considered anomalous considering the remaining five *C. dubia* ACR values all range from 0.4623 to 3.785. Stephen at el. (1985) cautions against the use of ACRs that differ by more than a factor of ten, stating, "*derivation of a criterion might not be possible if the available acute-chronic ratios vary by more than a factor of ten with no apparent pattern*."

[b] Acute test fed.

5.2.6.2.2 Deriving Green Sturgeon Chronic Adjustment Factors

Because no raw empirical chronic toxicity data were available for fish species in the same order as the green sturgeon (Acipenseriformes), no chronic order-level TAF could be calculated. As a result, EPA obtained and analyzed C-R data for all chronic tests in the EPA 2018 chronic aluminum dataset where such data were reported or could be obtained to derive a chronic vertebrate TAF or chronic MAF, if necessary (i.e., if the vertebrate and invertebrate chronic TAFs do not differ from one another). As discussed in Section 5.1.2.1.1 only one quantitatively acceptable EC_{20}:EC_5 ratio for vertebrates was available, not permitting a statistical test to determine if vertebrate and invertebrate TAFs differ. Moreover, no qualitatively acceptable models (up to the order level) exist that can be used to estimate minimum effect concentrations (as in the case of the salmonids). Thus, the chronic MAF was used to estimate green sturgeon chronic low effect concentrations (see Table 5-4). The chronic MAF was 1.403, calculated as the geometric mean of genus-level EC_{20}:EC_5 ratios of 1.168 and 1.685 for fathead minnow and a cladoceran (*Ceriodaphnia dubia*), respectively. The chronic MAF (1.403) was more conservative than the chronic fathead minnow TAF (1.168, also serves as the vertebrate TAF since no other vertebrate TAFs were available; see Table 5-4 and Appendix B.2). The corresponding EC_{20}:EC_{10} and EC_{20}:EC_{15} mean adjustment factors were 1.200 and 1.083, respectively (see Appendix C.3 and Appendix C.4).

5.2.6.2.3 Calculating Green Sturgeon Chronic Aluminum Minimum Effect Threshold

The chronic MAF (i.e., EC_{20}:EC_5 adjustment factor) as well as the EC_{20}:EC_{10} and EC_{20}:EC_{15} adjustment factors were applied, as a denominator, to the *Acipenser* GMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the green sturgeon:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) = **EC_5** = 635.8 µg/L / 1.403 = **453.2 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) = **EC_{10}** = 635.8 µg/L / 1.200 = **529.8 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) = **EC_{15}** = 635.8 µg/L / 1.083 = **586.9 µg/L***

5.2.6.2.4 Green Sturgeon: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and vertebrate species sensitivity across water chemistry, green sturgeon chronic EC_5, EC_{10}, and EC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding green sturgeon chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and green sturgeon chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were compared to the green sturgeon Southern DPS. The comparison between chronic effect threshold data and criterion magnitudes for the green sturgeon Southern DPS only considered the specific water quality data that were collected from Oregon level-III ecoregions within the green sturgeon Southern DPS. To visually display the comparison, criterion magnitudes (X-axis) and corresponding chronic effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line for the green sturgeon southern DPS (see Figure 5-35). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect

threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and green sturgeon Southern DPS chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) are reported in Table 5-24.

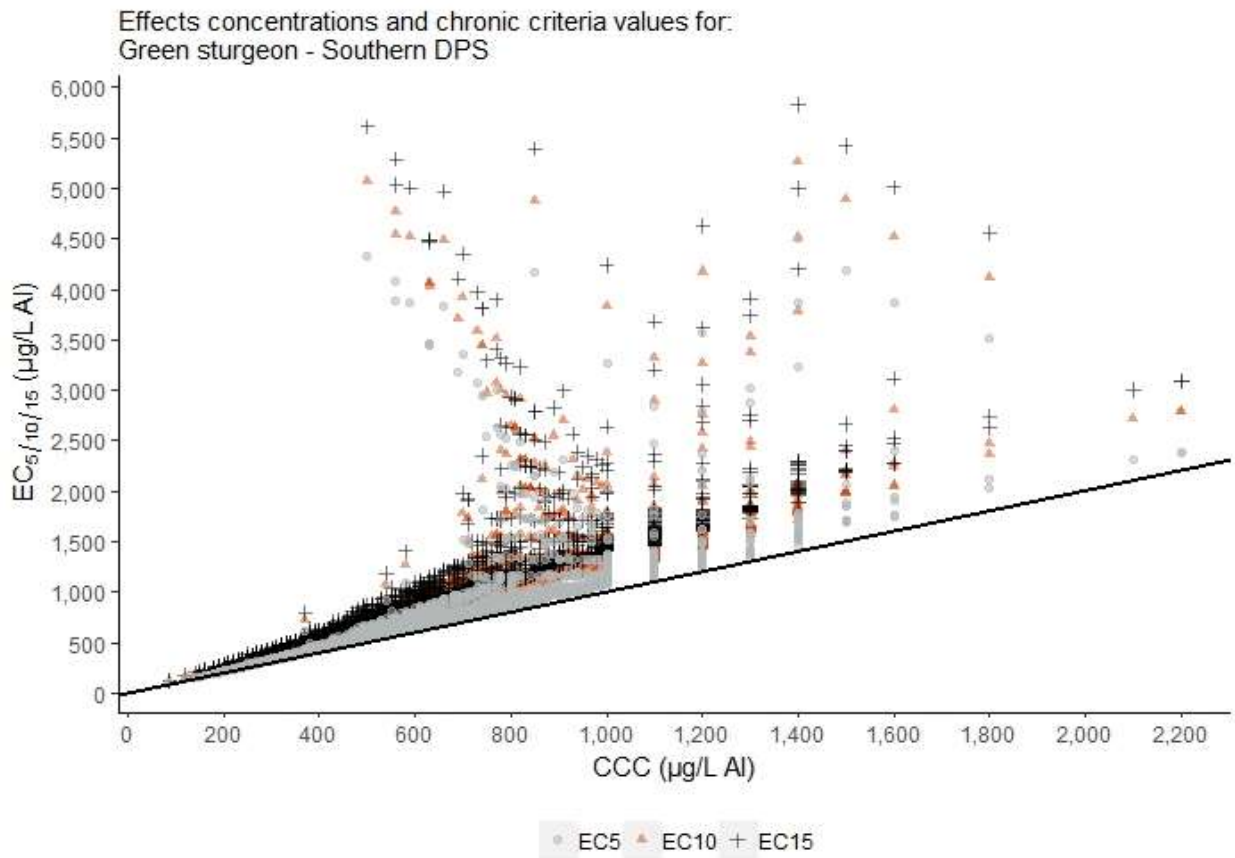

Figure 5-35. Paired green sturgeon chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the green sturgeon Southern Distinct Population Segment.

Table 5-24. The number and percentage of occurrences where the chronic criterion magnitude exceed green sturgeon EC_5, EC_{10}, and EC_{15} values for the Southern Distinct Population Segment (DPS). Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing the range of the Southern DPS. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (growth/biomass) at each effects concentration. AEL for each ECx value is 0%.

Distinct Population Segment	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}		CCC > EC_{15}	
Name		n	%	AEL	N	%	n	%
Southern DPS	4,768	0	0	0	0	0	0	0

5.2.6.2.5 Green Sturgeon: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC was always less than the green sturgeon Southern DPS chronic low effect threshold (i.e., EC_5; Table 5-24). As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for green sturgeon based on the chronic criterion; however, such considerations for the green sturgeon remained, due to acute sensitivity to Al (see Section 5.2.6.1.5).

5.2.7 Pacific eulachon (Thaleichthys pacificus)

5.2.7.1 Pacific eulachon Acute Aluminum Effects Assessment

5.2.7.1.1 Identifying Pacific eulachon Acute Aluminum Data

Acute aluminum toxicity data were not available for the Pacific eulachon or any other species within the Order Osmeriformes. Furthermore, no web-ICE models were available for the Pacific eulachon, *Thaleichthys* species, or members of the Family Osmeridae. As a result, all salmonid data were used to derive a salmoniformes OMAV that served as the acute toxicity value (i.e., LC_{50}) representative of Pacific eulachon. The salmoniformes OMAV was used to represent Pacific eulachon acute sensitivity to aluminum because;

1. salmonid data have historically been used for effects assessments involving Pacific eulachon (NOAA 2012);
2. previous assessments indicated members the same class may have pronounced differences in sensitivity to chemical exposures (see Section 5.1.2.1.1) and the derivation of a class-level toxicity value (i.e., Actinopterygii mean acute value) may not produce scientifically-defensible conclusions and;
3. Salmonid species are relatively sensitive to aluminum exposures and the application of the salmoniformes OMAV to represent Pacific eulachon was expected to provide a reasonable degree of conservatism to account for data limitation associated with Pacific eulachon or closely-related surrogate species.

Aluminum toxicity data from three salmonid species, across three genera and one family, were used to calculate a Salmoniformes order mean acute value (OMAV) that was used to represent the sensitivity of Pacific eulachon to acute aluminum exposure (Table 5-25), even though Pacific eulachon is not a member of the Order Salmoniformes. Although, salmonids and Pacific eulachon are both members of the Class Actinopterygii, use of the salmoniformes OMAV to represent Pacific eulachon was not considered application of class-level surrogacy because additional acute toxicity data from other members of the Class Actinopterygii were available, but were not considered because salmonids were among the most sensitive taxa to aluminum exposures (Note: In reference conditions [pH = 7.0, hardness = 100 mg/L, DOC = 1.0 mg/L], five genera within the Class Actinopterygii ranked in the 11 most sensitive genera, with Oncorhynchus, Salmo, and Salvelinus GMACs ranking as the third, sixth, and eleventh most sensitive, respectively. The other genera within the Class Actinopterygii that ranked in the 11 most sensitive genera were Micropterus, which was the second most sensitive genera with a GMAV of 2,988 µg/L, and Poecilia, which ranked as the seventh most sensitive with a GMAV of 9,061 µg/L).

The Salmoniformes OMAV was specifically calculated as the geometric mean of three SMAVs; one for rainbow trout based on six LC_{50}s, one for Atlantic salmon based on two LC_{50}s, and one for brook trout based on three LC_{50}s (Appendix A.1). The Salmoniformes OMAV, that was used to represent Pacific eulachon, was 8,150 µg/L total aluminum (normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L).

Table 5-25. Data used to calculate the estimated Salmoniformes Order Mean Acute Value representative of the acute aluminum sensitivity of Pacific eulachon.

Order	Family	Species	SMAV (µg/L)[a]	GMAV (µg/L)[a]	FMAV (µg/L)[a]	OMAV (µg/L)[a]
Salmoniformes	Salmonidae	Rainbow trout, *Oncorhynchus mykiss*	3,312	3,312	8,150	8,150
Salmoniformes	Salmonidae	Atlantic salmon, *Salmo salar*	8,642	8,642		
Salmoniformes	Salmonidae	Brook trout, *Salvelinus fontinalis*	18,913	18,913		

[a] All values expressed as total aluminum, normalized to pH 7, DOC of 1 mg/L and 100 mg/L hardness as $CaCO_3$; normalized using MLR equations identified in (USEPA, 2018) .
N/A: not available.

5.2.7.1.2 Deriving Acute Adjustment Factors

Since no acute aluminum toxicity data were available for Pacific eulachon or other members of the Order Osmeriformes, no order-level TAF could be derived. Because of these limitations, and because data for Order Salmoniformes has been used for effects assessments involving Pacific eulachon, EPA relied on the two qualitatively acceptable brook trout tests described in Section 5.2.1.1.2 (see Table 5-8) to derive LC_{50}:LC_5, LC_{50}:LC_{10}, and LC_{50}:LC_{15} ratios. The LC_{50}:LC_5, LC_{50}:LC_{10}, and LC_{50}:LC_{15} ratios that are representative of Pacific eulachon are 1.967, 1.638, and 1.466, respectively.

5.2.7.1.3 Calculating Pacific Eulachon Acute Aluminum Effect Threshold Values

The LC_{50}:LC_5, LC_{50}:LC_{10}, and LC_{50}:LC_{15} ratios that are representative of Pacific eulachon were applied, as a denominator, to the Salmoniformes OMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value representative of the Pacific eulachon:

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) = **LC_5** = 8,150 µg/L / 1.967 = **4,142 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) = **LC_{10}** = 8,150 µg/L / 1.638 = **4,976 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) = **LC_{15}** = 8,150 µg/L / 1.466 = **5,561 µg/L***

5.2.7.1.4 Pacific Eulachon: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude and vertebrate species sensitivity across water chemistry, Pacific eulachon acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate an

acute criterion magnitude and corresponding Pacific eulachon acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and Pacific eulachon acute effect threshold data (i.e., LC_5, LC_{10}, and LC_{15}) were compared to the Pacific eulachon Southern DPS. The comparison between acute effect threshold data and criterion magnitudes for the Pacific eulachon Southern DPS only considered the specific water quality data that were collected from Oregon level-III ecoregions within the Pacific eulachon Southern DPS. To visually display the comparison, criterion magnitudes (X-axis) and corresponding acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line for the Pacific eulachon Southern DPS (see Figure 5-36). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, LC_{10}, or LC_{15}) and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and Pacific eulachon Southern DPS acute effect threshold data (i.e., LC_5, LC_{10}, LC_{15}) are reported in Table 5-26.

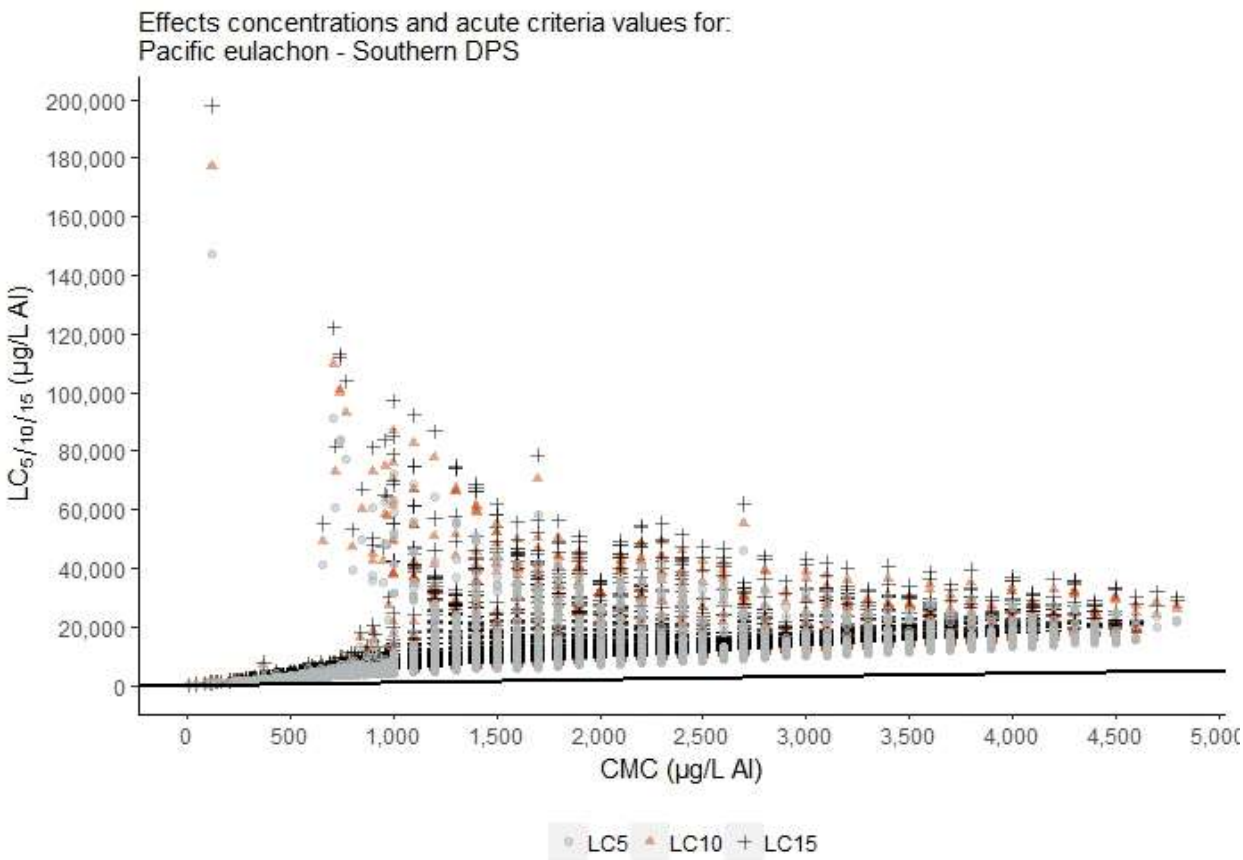

Figure 5-36. Paired Pacific eulachon acute effect concentrations and acute criterion values in Oregon ecoregions containing the Pacific eulachon Southern Distinct Population Segment.

Table 5-26. The number and percentage of occurrences where the acute criterion magnitude exceed Pacific eulachon LC_5, LC_{10}, and LC_{15} values for the Southern Distinct Population Segment (DPS). Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing the range of the Southern DPS. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (mortality) at each effects concentration. AEL for each ECx value is 0%.

Distinct Population Segment	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}		CMC > LC_{15}	
Name		n	%	AEL	N	%	n	%
Southern DPS	13,973	0	0	0	0	0	0	0

5.2.7.1.5 Pacific Eulachon: Tier I Acute Assessment Conclusion

The acute aluminum CMC was less than the Pacific eulachon acute low effect threshold (i.e., LC_5) in all water chemistry scenarios (Table 5-26). The LC_5 range was 89 to 147,408 µg/L and the CMC range was 4.8 to 4,800 µg/L. As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for Pacific eulachon based on the acute criterion.

5.2.7.2 Pacific Eulachon Aluminum Chronic Effects Assessment

5.2.7.2.1 Identifying Pacific Eulachon Chronic Aluminum Data

Chronic aluminum toxicity data are not available for Pacific eulachon or other species within the Order Osmeriformes. Similar to the acute Pacific eulachon Tier 1 Assessment, EPA relied on all available chronic data for species within the Order Salmoniformes to provide an Order Mean Chronic Value (OMCV) that represents Pacific eulachon. The Salmoniformes OMCV was used to represent Pacific eulachon sensitivity to chronic aluminum exposures because salmonids have been historically used as a representative surrogate for eulachon (NOAA Fisheries, 2012) and salmonids are among the most sensitive genera to chronic aluminum exposures, which provided a reasonable degree of conservatism to account for eulachon data limitations (see Section 5.2.7.1.1) The Salmoniformes OMCV was the geometric mean of two SMCVs; the Atlantic salmon SMCV based on an ELS test, the brook trout SMCV based on the geometric mean of two ELS tests (see Appendix A.2 and Table 5-27). The Salmoniformes OMCV which was representative of Pacific eulachon was 526.5 µg/L total aluminum (normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L).

Table 5-27. Data used to calculate the Salmoniformes Order Mean Chronic Value representative of the chronic aluminum sensitivity of Pacific eulachon.

Order	Family	Species	SMCV (µg/L)[a]	GMCV (µg/L)[a]	FMCV (µg/L)[a]	OMCV (µg/L)[a]
Salmoniformes	Salmonidae	Atlantic salmon, *Salmo salar*	434.4	434.4	526.5	526.5
Salmoniformes	Salmonidae	Brook trout, *Salvelinus fontinalis*	638.2	638.2		

[a] All values expressed as total aluminum, normalized to pH 7, DOC of 1 mg/L and 100 mg/L hardness as $CaCO_3$; normalized using MLR equations identified in (USEPA, 2018).
N/A: not available.

5.2.7.2.2 Deriving EC_{20} to EC_5 Chronic Adjustment Factor

Since no chronic aluminum toxicity data are available for Pacific eulachon or other members of the Order Osmeriformes, no order-level TAF could be derived for the Order Osmeriformes. Because of these limitations, and because data for Order Salmoniformes has been used for effects assessments involving Pacific eulachon, EPA relied on the qualitatively acceptable brook trout test described in Section 5.2.1.2.2 (see Table 5-12) to derive EC_{20}:EC_5, EC_{20}:EC_{10}, and EC_{20}:EC_{15} ratios. The EC_{20}:EC_5, EC_{20}:EC_{10}, and EC_{20}:EC_{15} ratios that are representative of Pacific eulachon are 1.696, 1.316, and 1.125, respectively.

5.2.7.2.3 Calculating Pacific Eulachon Chronic Aluminum Minimum Effect Threshold Values

The EC_{20}:EC_5, EC_{20}:EC_{10}, and EC_{20}:EC_{15} ratios that are representative of Pacific eulachon were applied, as a denominator, to the Salmoniformes OMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the Pacific eulachon:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) = **EC_5** = 526.5 µg/L / 1.696 = **310.4 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) = **EC_{10}** = 526.5 µg/L / 1.316 = **400.0 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) = **EC_{15}** = 526.5 µg/L / 1.125 = **467.9 µg/L***

5.2.7.2.4 Pacific Eulachon: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and vertebrate species sensitivity across water chemistry, Pacific eulachon chronic EC_5, EC_{10}, and EC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding Pacific eulachon chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and Pacific eulachon chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were compared to the Pacific eulachon Southern DPS. The comparison between chronic effect threshold data and criterion magnitudes for the Pacific eulachon Southern DPS only considered the specific water quality data that were collected from Oregon level-III ecoregions within the Pacific eulachon Southern DPS. To visually display the comparison, criterion magnitudes (X-axis) and corresponding chronic effect concentrations (Y-axis) were

plotted over a linear (i.e., 1:1) line for the Pacific eulachon southern DPS (see Figure 5-37). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and Pacific eulachon Southern DPS chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) are reported in Table 5-28.

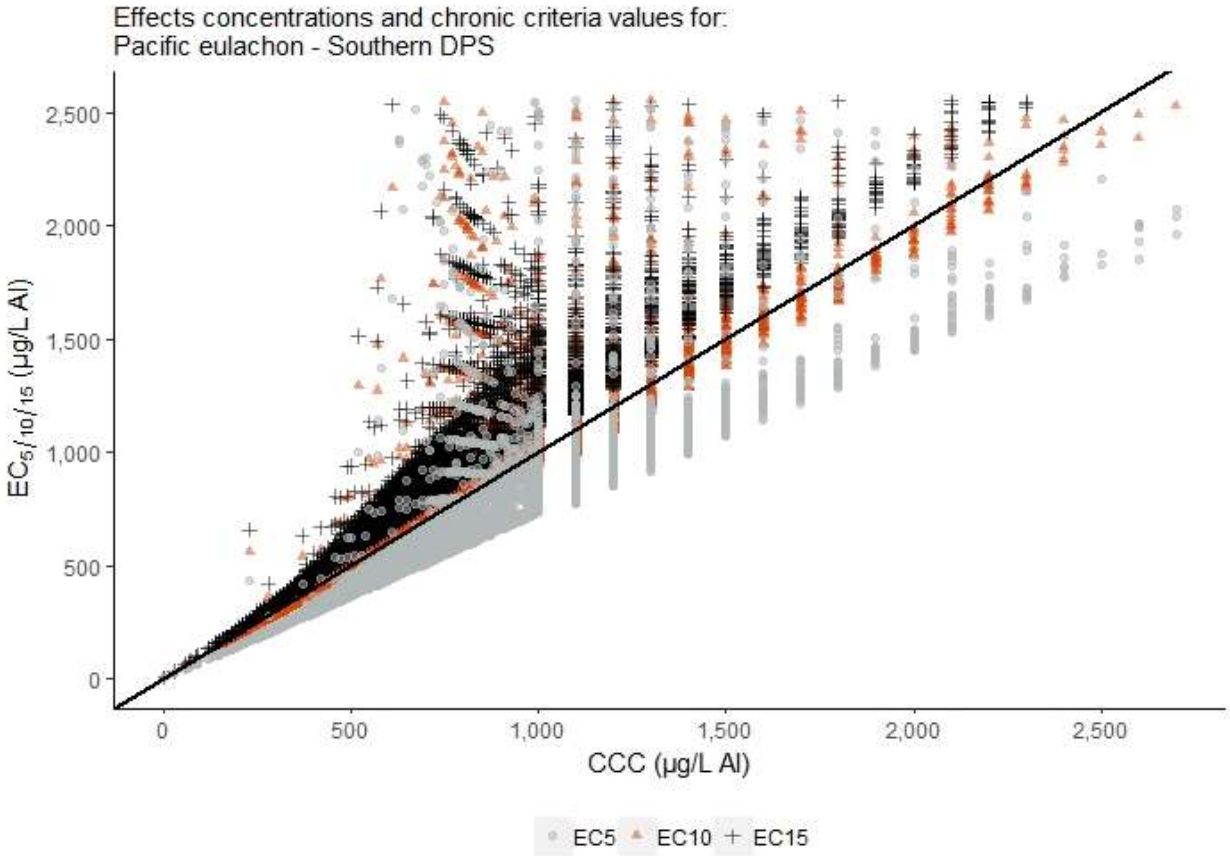

Figure 5-37. Paired Pacific eulachon chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the Pacific eulachon Southern Distinct Population Segment. Note, 1.27% ($n_{removed}$ = 531) of available data points were removed from Figure 5-36 as the Y axis was scaled down (Y axis maximum = 16,649) to aid in data visualization.

Table 5-28. The number and percentage of occurrences where the chronic criterion magnitude exceed Pacific eulachon EC_5, EC_{10}, and EC_{15} values for the Southern Distinct Population Segment (DPS). Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing the range of the Southern DPS. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (growth/biomass) at each effects concentration.

Distinct Population Segment	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}			CCC > EC_{15}		
Name		n	%	AEL	N	%	AEL	n	%	AEL
Southern DPS	13,973	13,232	94.70	4.74	8513	60.92	6.09	0	0	0

5.2.7.2.5 Pacific Eulachon: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC was greater than the Pacific eulachon EC_5 in 94.70% of water chemistry scenarios and exceeded the Pacific eulachon EC_{10} in 60.92% of water chemistry scenarios. The chronic Al CCC never exceeded the Pacific eulachon EC_{15} (Table 5-28). Because the CCC exceeded the Pacific eulachon chronic low effect threshold, consideration of a refined exposure assessment through a weight of evidence approach was required for Pacific eulachon based on the chronic criterion.

5.2.8 Lahontan Cutthroat Trout (Oncorhynchus clarki henshawi)

5.2.8.1 Lahontan Cutthroat Trout Acute Aluminum Effects Assessment

5.2.8.1.1 Identifying Lahontan Cutthroat Trout Acute Aluminum Data

Species-level acute toxicity data were not available for Lahontan cutthroat trout. The only acute toxicity data available for members of the genus *Oncorhynchus* were based on rainbow trout (see Appendix A.1). The rainbow trout species mean acute value (SMAV; 3,312 µg/L total aluminum, normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) was therefore used to represent the sensitivity of Lahontan cutthroat trout to acute aluminum exposures (see Section 5.2.1.1.1 and Table 5-6).

5.2.8.1.2 Deriving Lahontan Cutthroat Trout Acute Adjustment Factors

No quantitatively acceptable acute TAFs were available for Lahontan cutthroat trout or any other vertebrate. Qualitatively acceptable C-R models from two acute brook trout tests (see Al-Acute 66 and Al-Acute-68 in Appendix B.1) were, therefore, used to derive a family-level TAF (i.e., LC_{50}:LC_5 adjustment factor) of 1.967, representative of fish in the Family Salmonidae, including Lahontan cutthroat trout (see Section 5.2.1.1.2 and Table 5-8). Al-Acute-66 and Al-Acute-68 C-R models (see Appendix B.1) were also used to calculate LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors of 1.638 and 1.466, respectively that are representative of Lahontan cutthroat trout.

5.2.8.1.3 Calculating Lahontan Cutthroat Trout Acute Aluminum Minimum Effect Threshold Values

The Salmonidae family-level TAF (i.e., LC_{50}:LC_5 adjustment factor) as well as the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were applied, as a denominator, to the *Oncorhynchus* GMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value representative of the Lahontan cutthroat trout:

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) =* ***LC_5*** *= 3,312 µg/L / 1.967 =* ***1,683 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) =* ***LC_{10}*** *= 3,312 µg/L / 1.638 =* ***2,022 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) =* ***LC_{15}*** *= 3,312 µg/L / 1.466 =* ***2,260 µg/L***

5.2.8.1.4 Lahontan Cutthroat Trout: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude and vertebrate species sensitivity across water chemistry, Lahontan cutthroat trout acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate an acute criterion magnitude and corresponding Lahontan cutthroat trout acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and Lahontan cutthroat trout acute effect threshold data (i.e., LC_5, LC_{10}, and LC_{15}) were compared to the specific water quality data that were collected from level-III ecoregions within the native range of the Lahontan cutthroat trout. To visually display the comparison, criterion magnitudes (X-axis) and corresponding Lahontan cutthroat trout acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-38). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, LC_{10}, or LC_{15}) and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and Lahontan cutthroat trout acute effect threshold data (i.e., LC_5, LC_{10}, LC_{15}) are reported in Table 5-29.

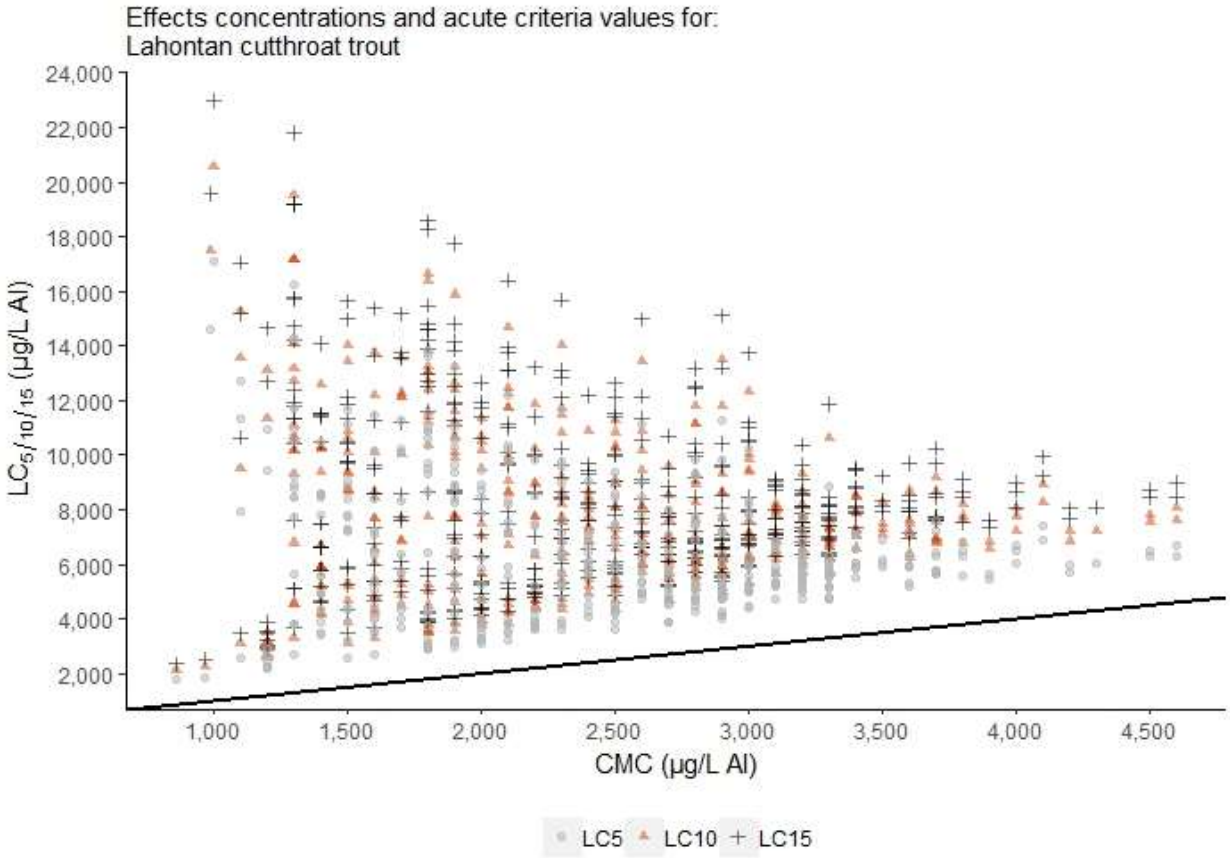

Figure 5-38. Paired Lahontan cutthroat trout acute effect concentrations and acute criterion values in Oregon ecoregions containing Lahontan cutthroat trout.

Table 5-29. The number and percentage of occurrences where the acute criterion magnitude exceed Lahontan cutthroat trout LC_5, LC_{10}, and LC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing Lahontan cutthroat trout range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (mortality) at each effects concentration. AEL for each ECx value is 0%.

Species	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}		CMC > LC_{15}	
Name		n	%	AEL	N	%	n	%
Lahontan cutthroat trout	443	0	0	0	0	0	0	0

5.2.8.1.5 Lahontan Cutthroat Trout: Tier I Acute Assessment Conclusion

The acute aluminum CMC was less than the Lahontan cutthroat trout acute low effect threshold (i.e., LC_5) in all water chemistry scenarios in Oregon ecoregions within the Lahontan cutthroat trout range (**Table 5-29**). The LC_5 range was 1,756 to 17,107 µg/L and the CMC range was 860 to 4,600 µg/L. As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for Lahontan cutthroat trout based on the acute criterion.

5.2.8.2 Lahontan Cutthroat Trout Chronic Aluminum Effects Assessment

5.2.8.2.1 Identifying Lahontan Cutthroat Trout Chronic Aluminum Data

Species-level chronic aluminum toxicity data were not available for Lahontan cutthroat trout or any other members within the genus *Oncorhynchus*. Therefore, chronic toxicity data for members of the Family Salmonidae were used to estimate the Salmonidae FMCV of 526.5 µg/L (total aluminum pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) that was representative of Lahontan cutthroat trout (see Section 5.2.1.2.1 and Table 5-10). EPA also evaluated qualitatively acceptable chronic toxicity data (Appendix H of USEPA (2018)) from several tests with early-life stages of golden, cutthroat, and rainbow trout to provide comparative information (Table 5-11 provides a summary of these tests). Overall, results of these qualitatively acceptable chronic toxicity tests appear to support the Salmonidae FMCV; however, the range in normalized no effect and effect concentrations, as well as endpoint variability, also indicate the potential for epistemic uncertainty and natural variability that can confound estimates of Lahontan cutthroat trout (and other *Oncorhynchus* sp.) sensitivity to chronic aluminum exposures.

5.2.8.2.2 Deriving Lahontan Cutthroat Trout Chronic Adjustment Factors

No quantitatively acceptable chronic TAFs were available for Lahontan cutthroat trout or any other member of the Order Salmoniformes. The only quantitatively acceptable C-R model available for a vertebrate species was based on the fathead minnow. As a result, a family-level TAF was calculated from a brook trout EC_{20}:EC_5 ratio that was originally determined to be a qualitatively acceptable C-R curve fit (see Appendix B.2, curve Al-Chronic-9 and Section 5.2.1.2.2). The chronic salmonid family-level TAF (i.e., EC_{20}:EC_5) was 1.696 and the corresponding EC_{20}:EC_{10} ratio and the EC_{20}:EC_{15} ratio from curve Al-Chronic-9 was 1.316 and 1.125, respectively (see Section 5.2.1.2.2 and Table 5-12).

5.2.8.2.3 Calculating Lahontan Cutthroat Trout Chronic Aluminum Minimum Effect Threshold Values

The Salmonidae family-level TAF (i.e., EC_{20}:EC_5 adjustment factor) as well as the EC_{20}:EC_{10} and EC_{20}:EC_{15} adjustment factors were applied, as a denominator, to the Salmonidae FMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the Lahontan cutthroat trout:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) = **EC_5** = 526.5 µg/L / 1.696 = **310.4 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) = **EC_{10}** = 526.5 µg/L / 1.316 = **400.0 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) = **EC_{15}** = 526.5 µg/L / 1.125 = **467.9 µg/L***

5.2.8.2.4 Lahontan Cutthroat Trout: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and vertebrate species sensitivity across water chemistry, Lahontan cutthroat trout chronic EC_5, EC_{10}, and EC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding Lahontan cutthroat trout chronic low effect threshold values (i.e., EC_5, EC_{10}, EC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and Lahontan cutthroat trout chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were compared to water quality data that were collected from level-III ecoregions within the range of the Lahontan cutthroat trout. To visually display the comparison, criterion magnitudes (X-axis) and corresponding Lahontan cutthroat trout chronic effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-39). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and Lahontan cutthroat trout chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) are reported in Table 5-30.

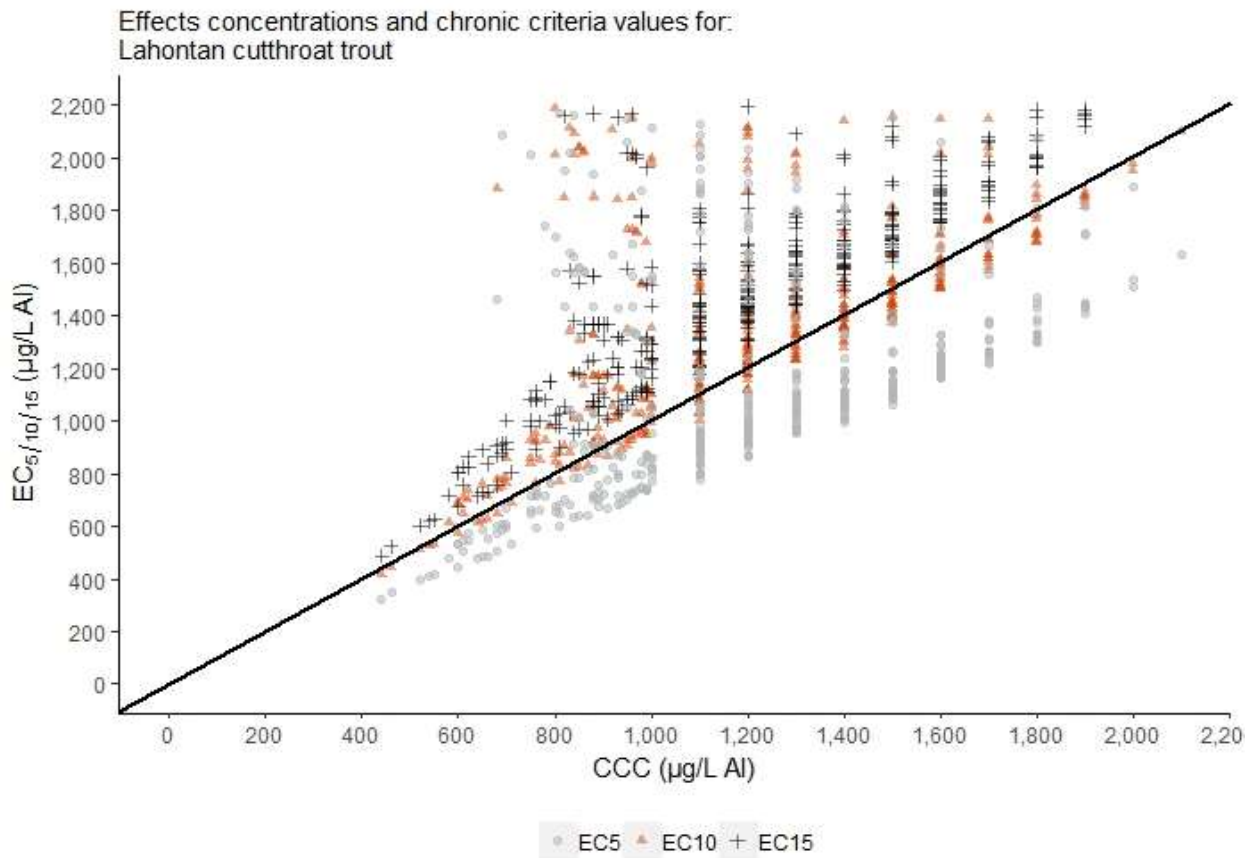

Figure 5-39. Paired Lahontan cutthroat trout chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the Lahontan cutthroat trout. Note, 12.34% ($n_{removed}$ = 164) of available data points were removed from Figure 5-38 as the Y axis was scaled down (Y axis maximum = 4,754) to aid in data visualization.

Table 5-30. The number and percentage of occurrences where the chronic criterion magnitude exceed Lahontan cutthroat trout EC_5, EC_{10}, and EC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing Lahontan cutthroat trout range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (growth/biomass) at each effects concentration.

Species	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}			CCC > EC_{15}		
Name		n	%	AEL	N	%	AEL	n	%	AEL
Lahontan cutthroat trout	443	323	72.91	3.65	134	30.25	3.03	0	0	0

5.2.8.2.5 Lahontan Cutthroat Trout: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC, across the range of ecoregions that contain the Lahontan cutthroat trout range, was greater than the Lahontan cutthroat trout chronic low effect threshold (i.e., EC_5) in 72.91% of water chemistry scenarios and exceeded the Lahontan cutthroat trout EC_{10} in 30.25% of water chemistry scenarios. The chronic Al CCC never exceeded the Lahontan cutthroat trout EC_{15} (Table 5-30). Because the CCC exceeded the Lahontan cutthroat trout chronic low effect threshold, consideration of a refined exposure assessment through a weight of evidence approach was required for Lahontan cutthroat trout based on the chronic criterion.

5.2.9 Bull Trout (Salvelinus confluentus)

5.2.9.1 Bull Trout Acute Aluminum Effects Assessment

5.2.9.1.1 Identifying Bull Trout Acute Aluminum Data

Species-level acute toxicity data were not available for bull trout. The only acute toxicity data available for members of the genus *Salvelinus* were based on brook trout (see Appendix A.1). The brook trout species mean acute value (SMAV; 18,913 µg/L total aluminum, normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) was therefore used to represent the sensitivity of bull trout to acute aluminum exposures (Table 5-31).

Table 5-31. Data used to calculate the *Salvelinus* Genus Mean Acute Value representative of the acute aluminum sensitivity of bull trout.

Species	Citation	LC_{50} (µg/L)[a]	SMAV (µg/L)[a]	GMAV (µg/L)[a]
Bull trout, *Salvelinus confluentus*	N/A	N/A	N/A	18,913
Brook trout, *Salvelinus fontinalis*	Tandjung 1982	30,038	18,913	
Brook trout, *Salvelinus fontinalis*	Tandjung 1982	24,514		
Brook trout, *Salvelinus fontinalis*	Tandjung 1982	9,187		

[a] All values expressed as total aluminum, normalized to pH 7, DOC of 1 mg/L and 100 mg/L hardness as $CaCO_3$; normalized using MLR equations identified in (USEPA, 2018).
N/A: not available.

5.2.9.1.2 Deriving Bull Trout Acute Adjustment Factors

No quantitatively acceptable acute TAFs were available for bull trout or any other vertebrate. Qualitatively acceptable C-R models from two acute brook trout tests (see Al-Acute 66 and Al-Acute-68 in Appendix B.1) were, therefore, used to derive a genus-level TAF (i.e., LC_{50}:LC_5 adjustment factor) of 1.967, representative of fish in the genus *Salvelinus*, including bull trout (see Section 5.2.1.1.2 and Table 5-8). Al-Acute-66 and Al-Acute-68 C-R models (see Appendix B.1) were also used to calculate LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors of 1.638 and 1.466, respectively that are representative of bull trout.

5.2.9.1.3 Calculating Bull Trout Acute Aluminum Minimum Effect Threshold Values

The *Salvelinus* genus-level TAF (i.e., LC_{50}:LC_5 adjustment factor) as well as the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were applied, as a denominator, to the *Salvelinus* GMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value representative of the bull trout:

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) =* ***LC_5*** *= 18,913 µg/L / 1.967 =* ***9,613 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) =* ***LC_{10}*** *= 18,913 µg/L / 1.638 =* ***11,547 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) =* ***LC_{15}*** *= 18,913 µg/L / 1.466 =* ***12,905 µg/L***

5.2.9.1.4 Bull Trout: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude and vertebrate species sensitivity across water chemistry, bull trout acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate an acute criterion magnitude and corresponding bull trout acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and bull trout acute effect threshold data (i.e., LC_5, LC_{10}, and LC_{15}) were compared to the specific water quality data that were collected from level-III ecoregions within the range of the bull trout. To visually display the comparison, criterion magnitudes (X-axis) and corresponding bull trout acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-40). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, LC_{10}, or LC_{15}) and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and bull trout acute effect threshold data (i.e., LC_5, LC_{10}, LC_{15}) are reported in Table 5-32.

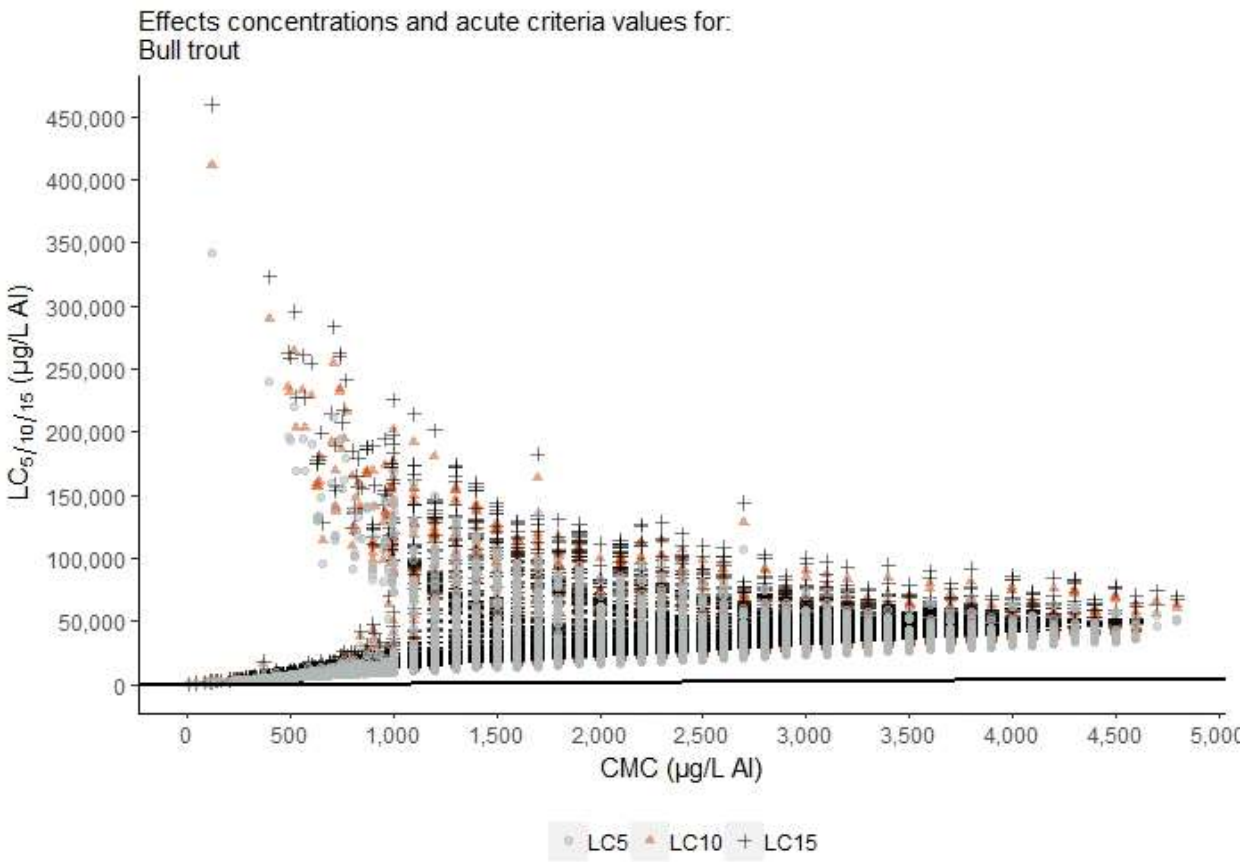

Figure 5-40. Paired bull trout acute effect concentrations and acute criterion values in Oregon ecoregions containing the bull trout.

Table 5-32. The number and percentage of occurrences where the acute criterion magnitude exceed bull trout LC_5, LC_{10}, and LC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing bull trout range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (mortality) at each effects concentration. AEL for each ECx value is 0%.

Species	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}		CMC > LC_{15}	
Name		n	%	AEL	N	%	n	%
Bull trout	17,836	0	0	0	0	0	0	0

5.2.9.1.5 Bull Trout: Tier I Acute Assessment Conclusion

The acute aluminum CMC was less than the bull trout acute low effect threshold (i.e., LC_5) in all water chemistry scenarios in Oregon ecoregions within the bull trout range (Table 5-32). The LC_5 range was 206 to 342,706 µg/L and the CMC range was 4.8 to 4,800 µg/L. As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for bull trout based on the acute criterion.

5.2.9.2 Bull Trout Chronic Aluminum Effects Assessment

5.2.9.2.1 Identifying Bull Trout Chronic Aluminum Data

Species-level chronic toxicity data were not available for bull trout. The only chronic toxicity data available for members of the genus *Salvelinus* were based on brook trout (see Appendix A.2). The brook trout species mean chronic value (SMCV; 638.2 µg/L total aluminum, normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) was based on two 60-day chronic tests with the most-sensitive life stage (eyed eggs) with a biomass endpoint (see Table 5-33). The brook trout SMCV was used to represent the sensitivity of bull trout to chronic aluminum exposures.

Table 5-33. Data used to calculate the *Salvelinus* Genus Mean Chronic Value representative of the chronic aluminum sensitivity of bull trout.

Species	Citation	EC_{20} (µg/L)[a]	SMCV (µg/L)[a]	GMCV (µg/L)[a]
Bull trout, *Salvelinus confluentus*	N/A	N/A	N/A	638.2
Brook trout, *Salvelinus fontinalis*	Cleveland et al. 1989	378.7	638.2	
Brook trout, *Salvelinus fontinalis*	Cleveland et al. 1989	1,076		

[a] All values expressed as total aluminum, normalized to pH 7, DOC of 1 mg/L and 100 mg/L hardness as $CaCO_3$; normalized using MLR equations identified in (USEPA, 2018).
N/A: not available.

5.2.9.2.2 Deriving Bull Trout Chronic Adjustment Factors

No quantitatively acceptable chronic TAFs were available for bull trout or any other member of the Order Salmoniformes. The only quantitatively acceptable C-R model available for a vertebrate species was based on the fathead minnow. As a result, a genus-level TAF was calculated from a brook trout EC_{20}:EC_5 ratio that was originally determined to be based on a qualitatively acceptable C-R curve fit (see Appendix B.2, curve Al-Chronic-9 and Section 5.2.1.2.2). The chronic salmonid family-level TAF (i.e., EC_{20}:EC_5) was 1.696 and the corresponding EC_{20}:EC_{10} ratio and the EC_{20}:EC_{15} ratio from curve Al-Chronic-9 was 1.316 and 1.125, respectively (see Section 5.2.1.2.2 and Table 5-12).

5.2.9.2.3 Calculating Bull Trout Chronic Aluminum Minimum Effect Threshold Values

The *Salvelinus* genus-level TAF (i.e., EC_{20}:EC_5 adjustment factor) as well as the EC_{20}:EC_{10} and EC_{20}:EC_{15} adjustment factors were applied, as a denominator, to the *Salvelinus* GMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the bull trout:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) = **EC_5** = 638.2 µg/L / 1.696 =* ***376.3 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) = **EC_{10}** = 638.2 µg/L / 1.316 =* ***484.8 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) = **EC_{15}** = 638.2 µg/L / 1.125 =* ***567.1 µg/L***

5.2.9.2.4 Bull Trout: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and vertebrate species sensitivity across water chemistry, bull trout chronic EC_5, EC_{10}, and EC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding bull trout chronic low effect threshold values (i.e., EC_5, EC_{10}, EC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and bull trout chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were compared to water quality data that were collected from level-III ecoregions within the range of the bull trout. To visually display the comparison, criterion magnitudes (X-axis) and corresponding bull trout chronic effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-41). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and bull trout chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) are reported in Table 5-34.

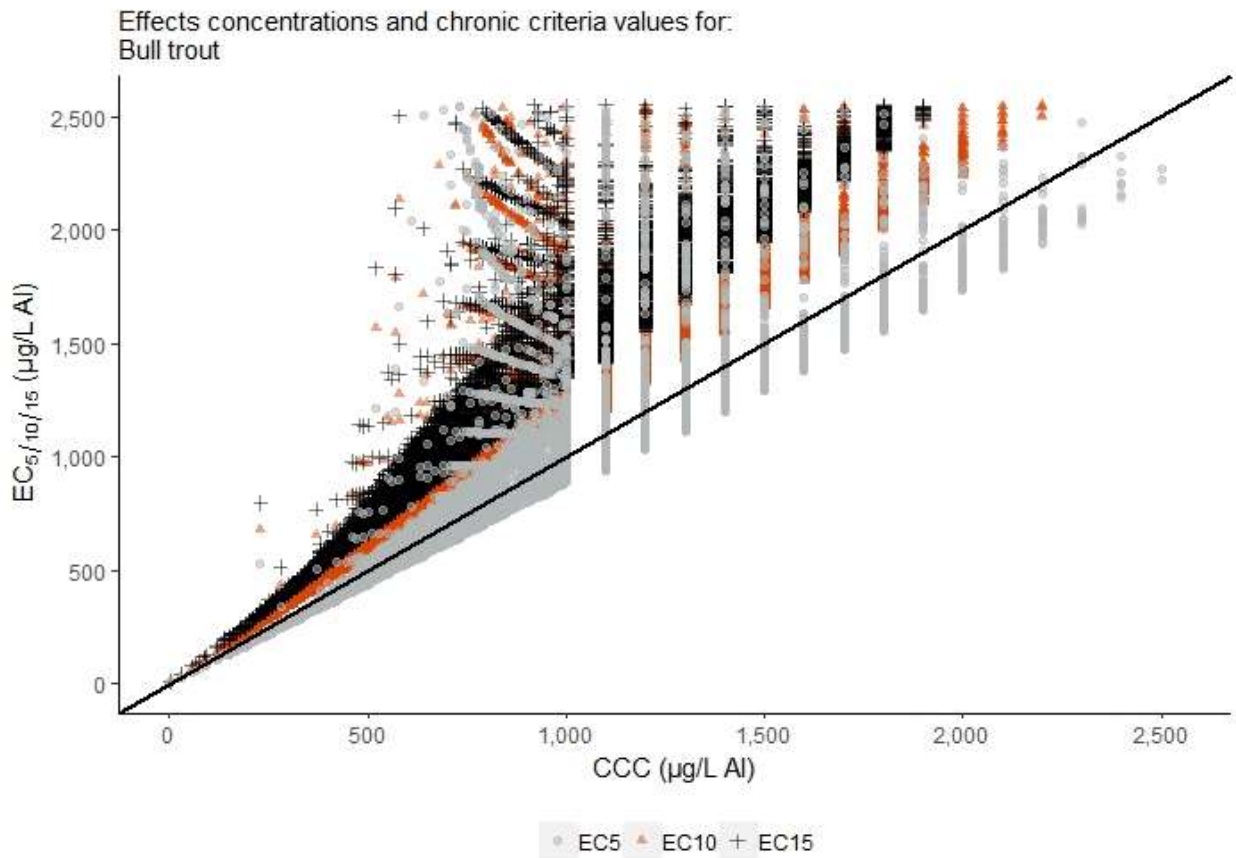

Figure 5-41. Paired bull trout chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the bull trout. Note, 4.08% ($n_{removed}$ = 2,183) of available data points were removed from Figure 5-40 as the Y axis was scaled down (Y axis maximum = 20,180) to aid in data visualization.

Table 5-34. The number and percentage of occurrences where the chronic criterion magnitude exceed bull trout EC_5, EC_{10}, and EC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing bull trout range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (growth/biomass) at each effects concentration.

Species	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}			CCC > EC_{15}		
Name		n	%	AEL	N	%	AEL	n	%	AEL
Bull trout	17,836	12,958	72.65	3.63	0	0	0	0	0	0

5.2.9.2.5 Bull Trout: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC, across the range of ecoregions that contain the bull trout, was greater than the bull trout chronic low effect threshold (i.e., EC_5) under 72.65% of water chemistry scenarios. The chronic Al CCC never exceeded the bull trout EC_{10} or EC_{15} (Table

5-34). Because the CCC exceeded the bull trout chronic low effect threshold, consideration of a refined exposure assessment through a weight of evidence approach was required for bull trout based on the chronic criterion.

5.2.10 Warner Sucker (Catostomus warnerensis)

5.2.10.1 Warner Sucker Acute Aluminum Effects Assessment

5.2.10.1.1 Identifying Warner Sucker Acute Aluminum Data

Species-level acute toxicity data were not available for Warner sucker or any other members of the Family Catostomidae. Acute toxicity data available were available for members of the Order Cypriniformes (see Appendix A.1). The Cypriniformes order mean acute value (OMAV; >21,937 µg/L total aluminum, normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) was based directly on the Cyprinidae FMAV, which was calculated from two SMAVs (representing two different genera; see Table 5-35). The Rio Grande silvery minnow SMAV was based on one acute toxicity test with 3-5 days-post hatch (dph) larva, while the fathead minnow SMAV was based on the geometric mean of three acute toxicity tests (Call et al. 1984 tested 32-33 dph juvenile fathead minnow in two separate tests; Buhl (2002) tested 4-6 dph larva fathead minnow; see Appendix A.1). All four individual toxicity tests produced non-definitive LC_{50} values (i.e., "greater than" LC_{50} values) because the highest test concentrations used in these tests did not result in ≥50% mortality. Therefore, the Cypriniformes OMAV that was representative of Warner sucker may have resulted in a conservative LC_{50} estimate.

Table 5-35. Data used to calculate the Cypriniformes Order Mean Acute Value representative of the acute aluminum sensitivity of Warner sucker.

Order	Family	Species	SMAV (µg/L)[a]	GMAV (µg/L)[a]	FMAV (µg/L)[a]	OMAV (µg/L)[a]
Cypriniformes	Catostomidae	Warner sucker, *Catostomus warnerensis*	N/A	N/A	N/A	>21,937
Cypriniformes	Cyprinidae	Rio Grande silvery minnow, *Hybognathus amarus*	>21,779	>21,779	>21,937	
Cypriniformes	Cyprinidae	Fathead minnow, *Pimephales promelas*	>22,095	>22,095		

[a] All values expressed as total aluminum, normalized to pH 7, DOC of 1 mg/L and 100 mg/L hardness as $CaCO_3$; normalized using MLR equations identified in (USEPA, 2018).
N/A: not available.

5.2.10.1.2 Deriving Warner Sucker Acute Adjustment Factors

No quantitatively acceptable acute TAFs were available for Warner sucker or any other vertebrate species (see Table 5-4). As a result, all genus-level TAFs (based on quantitatively acceptable C-R curves) were averaged (geometric mean; n = 2) to calculate the acute MAF, which serves as a generic, non-taxa-specific adjustment factor meant to broadly relate acute LC_{50} values to lower level effects concentrations (i.e., LC_5). Because no quantitatively acceptable TAFs were available for vertebrate species, the acute MAF is based on invertebrate species only

(see Table 5-4). Unlike, salmonids no qualitatively acceptable acute C-R models for fishes in the Order Cypriniformes were available to be considered in the development of a TAF based on qualitatively acceptable data (see Appendix B.1).

The acute MAF was calculated as the geometric mean of the *Physa* genus-level TAF (3.597) and the *Ceriodaphnia* genus-level TAF (4.189) and is 3.881 (Table 5-3). The acute C-R curves used to derive the acute MAF (i.e., LC_{50}:LC_5) were also used to calculate LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors of 2.787 and 2.261, respectively (see Appendix C.1 and Appendix C.2). Similar to the acute MAF (i.e., LC_{50}:LC_5), the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were intended to provide a mechanism for relating acute effect concentrations (i.e., LC_{50}) to lower-level effect concentrations (i.e., LC_{15} and LC_{10} values, respectively) across a broad range of taxa.

5.2.10.1.3 Calculating Warner Sucker Acute Aluminum Minimum Effect Threshold Values

The acute MAF (i.e., LC_{50}:LC_5 adjustment factor) as well as the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were applied, as a denominator, to the Cypriniformes OMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value representative of the Warner sucker:

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) = **LC_5** = 21,937 µg/L / 3.881 = **5,652 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) = **LC_{10}** = 21,937 µg/L / 2.787 = **7,870 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) = **LC_{15}** = 21,937 µg/L / 2.261 = **9,702 µg/L***

5.2.10.1.4 Warner Sucker: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude and vertebrate species sensitivity across water chemistry, Warner sucker acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate an acute criterion magnitude and corresponding Warner sucker acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and Warner sucker acute effect threshold data (i.e., LC_5, LC_{10}, and LC_{15}) were compared to the specific water quality data that were collected from level-III ecoregions within the range of the Warner sucker. To visually display the comparison, criterion magnitudes (X-axis) and corresponding Warner sucker acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-42). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, LC_{10}, or LC_{15}) and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and Warner sucker acute effect threshold data (i.e., LC_5, LC_{10}, LC_{15}) are reported in Table 5-36.

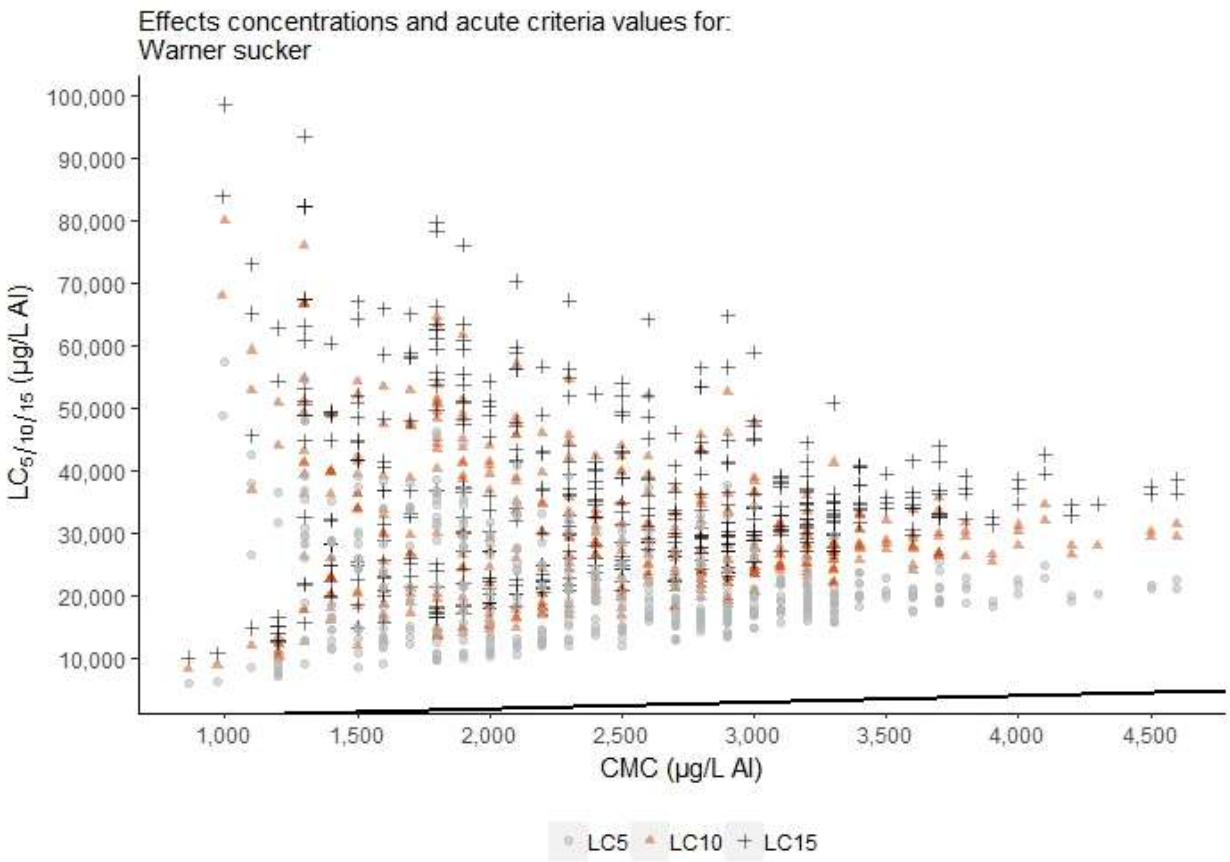

Figure 5-42. Paired Warner sucker acute effect concentrations and acute criterion values in Oregon ecoregions containing the Warner sucker.

Table 5-36. The number and percentage of occurrences where the acute criterion magnitude exceed Warner sucker LC_5, LC_{10}, and LC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing Warner sucker range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (mortality) at each effects concentration. AEL for each LCx value is 0%.

Species	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}		CMC > LC_{15}	
Name		n	%	AEL	N	%	n	%
Warner sucker	443	0	0	0	0	0	0	0

5.2.10.1.5 Warner Sucker: Tier I Acute Assessment Conclusion

The acute aluminum CMC was less than the Warner sucker acute low effect threshold (i.e., LC_5) in all water chemistry scenarios in Oregon ecoregions within the Warner sucker range (Table 5-36). The LC_5 range was 5,897 to 57,432 µg/L and the CMC range was 860 to 4,600 µg/L. As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for Warner sucker based on the acute criterion.

5.2.10.2 Warner Sucker Chronic Aluminum Effects Assessment

5.2.10.2.1 Identifying Warner Sucker Chronic Aluminum Data

Species-level chronic toxicity data were not available for Warner sucker or any other members of the Family Catostomidae. Two species within the Order Cypriniformes had chronic Al toxicity data to derive a Cypriniformes order mean chronic value (OMCV). The Cypriniformes OMCV is based directly on the Cyprinidae FMCV, which was calculated from the *Danio* GMCV and the *Pimephales* GMCV. The *Danio* GMCV is based directly on the zebrafish SMCV, which was determined from a single early life stage (<36 hours post hatch [hph]) test that exposed embryos to Al for 33 days with a biomass endpoint (OSU (2013) also referred to as Cardwell et al. (2018); see Appendix A.2). The *Pimephales* GMCV is based directly on the fathead minnow SMCV, which was calculated as the geometric mean of two fathead minnow tests. Kimball (1978) exposed fathead minnow to Al for 28 days and reported a biomass-based EC_{20} of 2,690 µg/L total aluminum (normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L; see Appendix A.2). OSU (2012); also referred to as Cardwell et al. (2018), exposed fathead minnow embryos (<24 hph) to Al for 33 days and reported a 20% reduction in fry survival occurring at a concentration of 2,154 µg/L total aluminum (normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L; see Appendix A.2). Together, these chronic tests resulted in a Cypriniformes OMCV of 1,797 µg/L total aluminum (normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) that was representative of the Warner sucker (Table 5-37).

Table 5-37. Data used to calculate the Cypriniformes Order Mean Chronic Value representative of the chronic aluminum sensitivity of Warner sucker.

Order	Family	Species	SMCV (µg/L)[a]	GMCV (µg/L)[a]	FMCV (µg/L)[a]	OMCV (µg/L)[a]
Cypriniformes	Catostomidae	Warner sucker, *Catostomus Warnerensis*	N/A	N/A	1,797	1,797
Cypriniformes	Cyprinidae	Zebrafish, *Danio rerio*	1,342	1,342		
Cypriniformes	Cyprinidae	Fathead minnow, *Pimephales promelas*	2,407	2,407		

[a] All values expressed as total aluminum, normalized to pH 7, DOC of 1 mg/L and 100 mg/L hardness as $CaCO_3$; normalized using MLR equations identified in (USEPA, 2018).
N/A: not available.

5.2.10.2.2 Deriving Warner Sucker Chronic Adjustment Factors

Of the three chronic tests used to calculate the OMCV, only the fathead minnow test reported by Kimball (1978) included C-R data that resulted in an acceptable C-R model (Al-Chronic 24; see Appendix B.2). The C-R model generated from the C-R data from the other fathead minnow test (OSU, 2012) was determined to be qualitatively acceptable because of a poor fit and significant variation in responses at low treatment concentrations (Al-Chronic 31). The C-R model generated from the zebrafish test (OSU, 2013) was determined to be unacceptable because of poor fit and resultant in 95% confidence bounds that extended into the negative number range (Al-Chronic 33; see Appendix B.2). The fathead minnow EC_{20}:EC_5 ratio (1.168) served as the Cypriniformes order-level TAF, representative of the Warner sucker (Al-Chronic 24; see Appendix B.2). The EC_{20}:EC_{15} (1.042) ratio and EC_{20}:EC_{10} ratio (1.094) from the same fathead minnow test served as an adjustment factors for transforming chronic EC_{20} values into lower-level effect concentrations (i.e., EC_{15} and EC_{10}, respectively) for members of the Order Cypriniformes, including Warner sucker (see AL-Chronic 24; Appendix C.3 and Appendix C.4).

5.2.10.2.3 Calculating Warner Sucker Chronic Aluminum Minimum Effect Threshold Values

The Cypriniformes order-level TAF (i.e., EC_{20}:EC_5 adjustment factor) as well as the EC_{20}:EC_{10} and EC_{20}:EC_{15} adjustment factors were applied, as a denominator, to the Cypriniformes OMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the Warner sucker:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) = **EC_5** = 1,797 µg/L / 1.168 = **1,539 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) = **EC_{10}** = 1,797 µg/L / 1.093 = **1,643 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) = **EC_{15}** = 1,797 µg/L / 1.042 = **1,725 µg/L***

5.2.10.2.4 Warner Sucker: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and vertebrate species sensitivity across water chemistry, Warner sucker chronic EC_5, EC_{10}, and EC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding Warner sucker chronic low effect threshold values (i.e., EC_5, EC_{10}, EC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and Warner sucker chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were compared to water quality data that were collected from level-III ecoregions within the range of the Warner sucker. To visually display the comparison, criterion magnitudes (X-axis) and corresponding Warner sucker chronic effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-43). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and Warner sucker chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) are reported in Table 5-38.

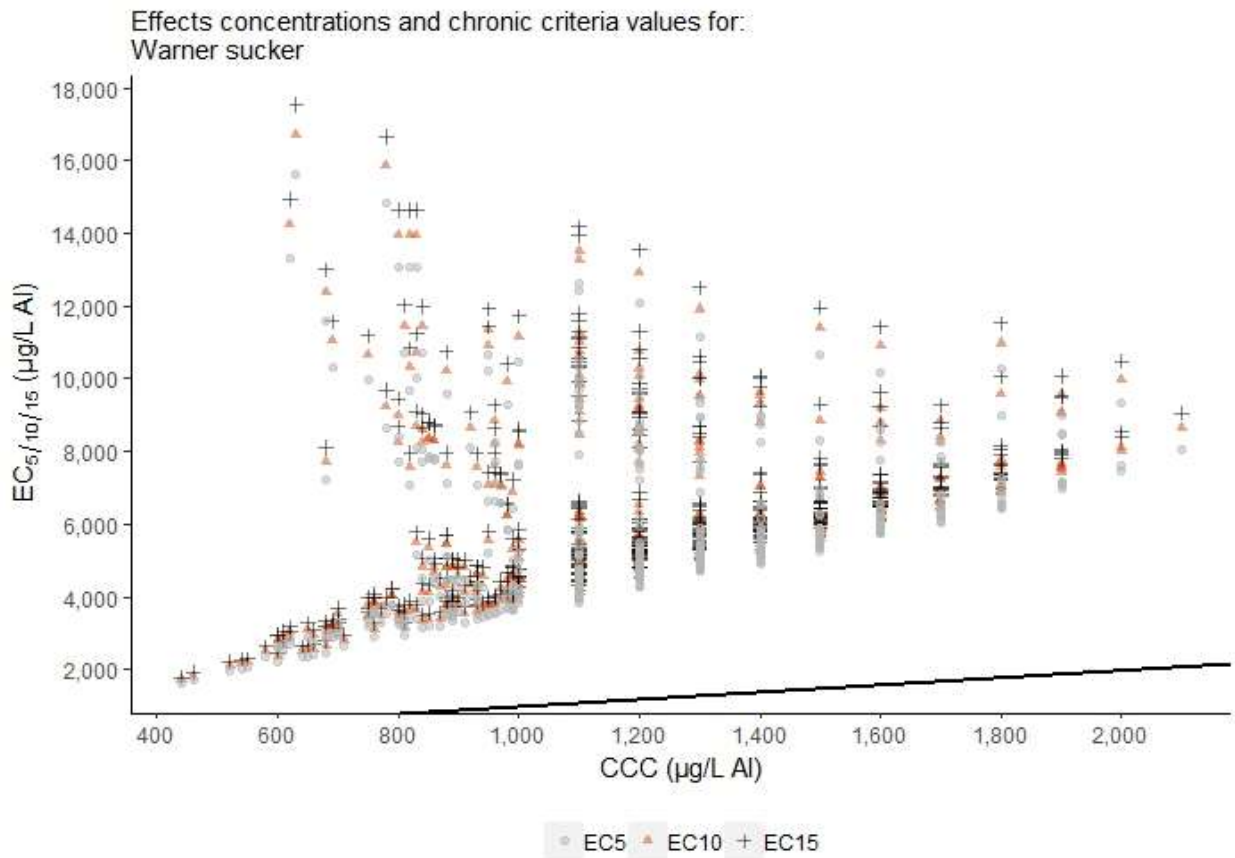

Figure 5-43. Paired Warner sucker chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the Warner sucker.

Table 5-38. The number and percentage of occurrences where the chronic criterion magnitude exceed Warner sucker EC_5, EC_{10}, and EC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing Warner sucker range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (growth/biomass) at each effects concentration. AEL for each ECx value is 0%.

Species	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}		CCC > EC_{15}	
Name		n	%	AEL	N	%	n	%
Warner sucker	443	0	0	0	0	0	0	0

5.2.10.2.5 Warner Sucker: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC, across the range of ecoregions within the Warner sucker range, was less than the Warner sucker chronic low effect threshold (i.e., EC_5) in all water chemistry scenarios. The chronic Al CCC never exceeded the Warner sucker EC_{10} or EC_{15} (Table 5-38). As

a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for Warner sucker based on the chronic criterion.

5.2.11 Lost River Sucker (Deltistes luxatus)

5.2.11.1 Lost River Sucker Acute Aluminum Effects Assessment

5.2.11.1.1 Identifying Lost River Sucker Acute Aluminum Data

Species-level acute toxicity data were not available for Lost River sucker or any other members of the Family Catostomidae. Acute toxicity data available were available for members of the Order Cypriniformes to calculate a Cypriniformes OMAV (see Appendix A.1). The Cypriniformes OMAV was >21,937 µg/L total aluminum (normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) and was representative of Lost River sucker (see Section 5.2.10.1.1 and Table 5-35).

5.2.11.1.2 Deriving Lost River Sucker Acute Adjustment Factors

No quantitatively acceptable acute TAFs were available for Lost River sucker or any other vertebrate species (see Table 5-3). As a result, the acute MAF (3.881) was selected as the adjustment factor to relate the Lost River sucker acute LC_{50} value to a 5% effect level (see Section 5.2.10.1.2. The same acute C-R curves used to derive the acute MAF (i.e., LC_{50}:LC_5) were also used to calculate LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors of 2.787 and 2.261, respectively (see Appendix C.1 and Appendix C.2). Similar to the acute MAF (i.e., LC_{50}:LC_5), the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were intended to provide a mechanism for relating acute effect concentrations (i.e., LC_{50}) to lower-level effect concentrations (i.e., LC_{15} and LC_{10} values, respectively) across a broad range of taxa (see Section 5.2.10.1.2).

5.2.11.1.3 Calculating Lost River Sucker Acute Aluminum Minimum Effect Threshold Values

The acute MAF (i.e., LC_{50}:LC_5 adjustment factor) as well as the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were applied, as a denominator, to the Cypriniformes OMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value representative of the Lost River sucker:

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) = **LC_5** = 21,937 µg/L / 3.881 = **5,652 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) = **LC_{10}** = 21,937 µg/L / 2.787 = **7,870 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) = **LC_{15}** = 21,937 µg/L / 2.261 = **9,702 µg/L***

5.2.11.1.4 Lost River Sucker: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude and vertebrate species sensitivity across water chemistry, Lost River sucker acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate

an acute criterion magnitude and corresponding Lost River sucker acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and Lost River sucker acute effect threshold data (i.e., LC_5, LC_{10}, and LC_{15}) were compared to the specific water quality data that were collected from level-III ecoregions within the range of the Lost River sucker. To visually display the comparison, criterion magnitudes (X-axis) and corresponding Lost River sucker acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-44). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, LC_{10}, or LC_{15}) and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and Lost River sucker acute effect threshold data (i.e., LC_5, LC_{10}, LC_{15}) are reported in Table 5-39.

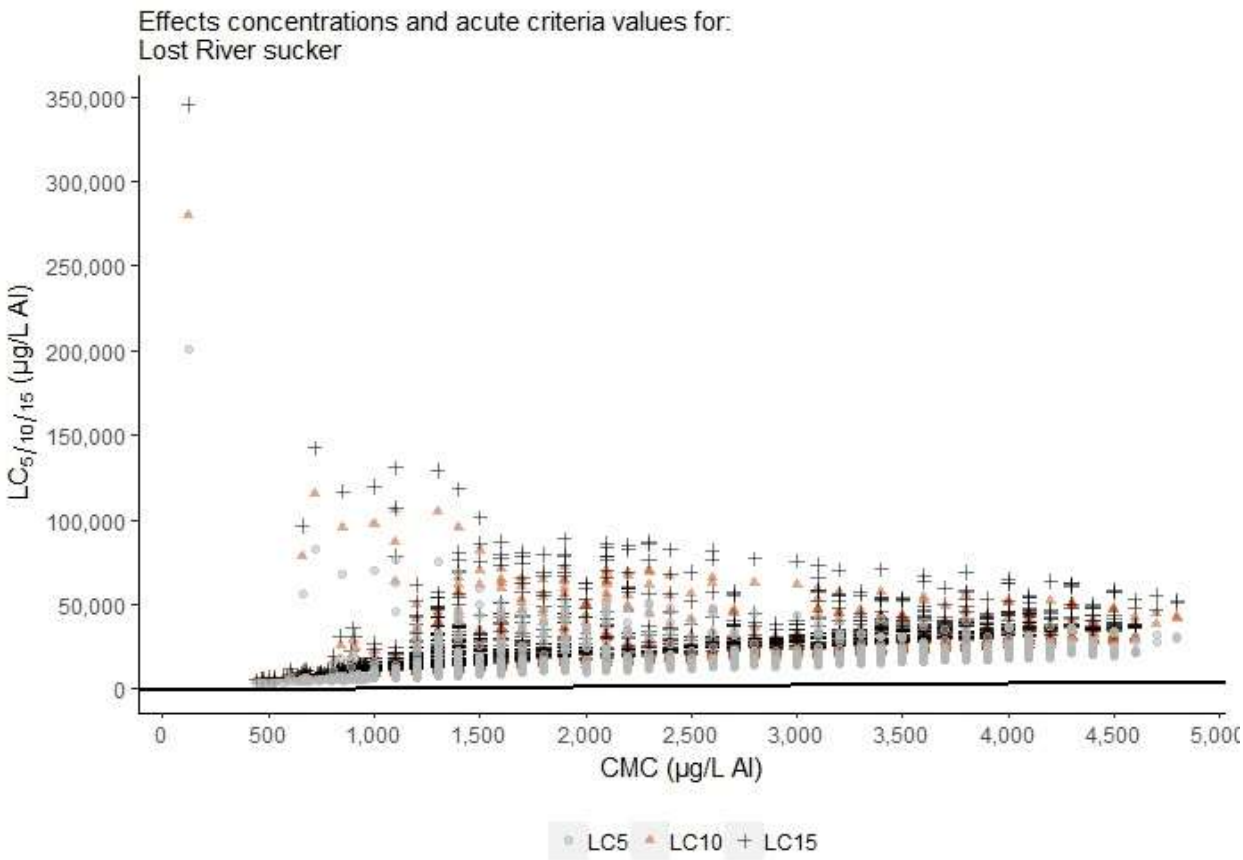

Figure 5-44. Paired Lost River sucker acute effect concentrations and acute criterion values in Oregon ecoregions containing the Lost River sucker.

Table 5-39. The number and percentage of occurrences where the acute criterion magnitude exceed Lost River sucker LC_5, LC_{10}, and LC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing Lost River sucker range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (mortality) at each effects concentration. AEL for each LCx value is 0%.

Species	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}		CMC > LC_{15}	
Name		n	%	AEL	N	%	n	%
Lost River sucker	1,830	0	0	0	0	0	0	0

5.2.11.1.5 Lost River Sucker: Tier I Acute Assessment Conclusion

The acute aluminum CMC was less than the Lost River sucker acute low effect threshold (i.e., LC_5) under all water chemistry scenarios in Oregon ecoregions within the Lost River sucker range (Table 5-39). The LC_5 range was 3,067 to 201,115 µg/L and the CMC range was 120 to 4,800 µg/L. As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for Lost River sucker based on the acute criterion.

5.2.11.2 Lost River Sucker Chronic Aluminum Effects Assessment

5.2.11.2.1 Identifying Lost River Sucker Chronic Aluminum Data

Species-level chronic toxicity data were not available for Lost River sucker or any other members of the Family Catostomidae. Two species within the Order Cypriniformes had chronic Al toxicity data to derive a Cypriniformes OMCV. The Cypriniformes OMCV is 1,797 µg/L total aluminum (normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) and was representative of the Lost River sucker (see Section 5.2.10.2.1 and Table 5-37).

5.2.11.2.2 Deriving Lost River Sucker Chronic Adjustment Factors

The only quantitatively acceptable C-R model from the available studies used to determine the Cypriniformes OMCV produced an EC_{20}:EC_5 ratio of 1.168 that served as the Cypriniformes order-level TAF, representative of the Lost River sucker (see Section 5.2.10.2.2). The EC_{20}:EC_{15} (1.042) ratio and EC_{20}:EC_{10} ratio (1.093) from the same fathead minnow test also served as adjustment factors for transforming chronic EC_{20} values into lower-level effect concentrations (i.e., EC_{15} and EC_{10}, respectively) for members of the Order Cypriniformes, including Lost River sucker (see Section 5.2.10.2.2).

5.2.11.2.3 Calculating Lost River Sucker Chronic Aluminum Minimum Effect Threshold Values

The Cypriniformes order-level TAF (i.e., EC_{20}:EC_5 adjustment factor) as well as the EC_{20}:EC_{10} and EC_{20}:EC_{15} adjustment factors were applied, as a denominator, to the Cypriniformes OMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the Lost River sucker:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) = **EC_5** = 1,797 µg/L / 1.168 =* ***1,539 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) = **EC_{10}** = 1,797 µg/L / 1.093 =* ***1,643 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) = **EC_{15}** = 1,797 µg/L / 1.042 =* ***1,725 µg/L***

5.2.11.2.4 Lost River Sucker: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and vertebrate species sensitivity across water chemistry, Lost River sucker chronic EC_5, EC_{10}, and EC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding Lost River sucker chronic low effect threshold values (i.e., EC_5, EC_{10}, EC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and Lost River sucker chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were compared to water quality data that were collected from level-III ecoregions within the range of the Lost River sucker. To visually display the comparison, criterion magnitudes (X-axis) and corresponding Lost River sucker chronic effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-45). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and Lost River sucker chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) are reported in Table 5-40.

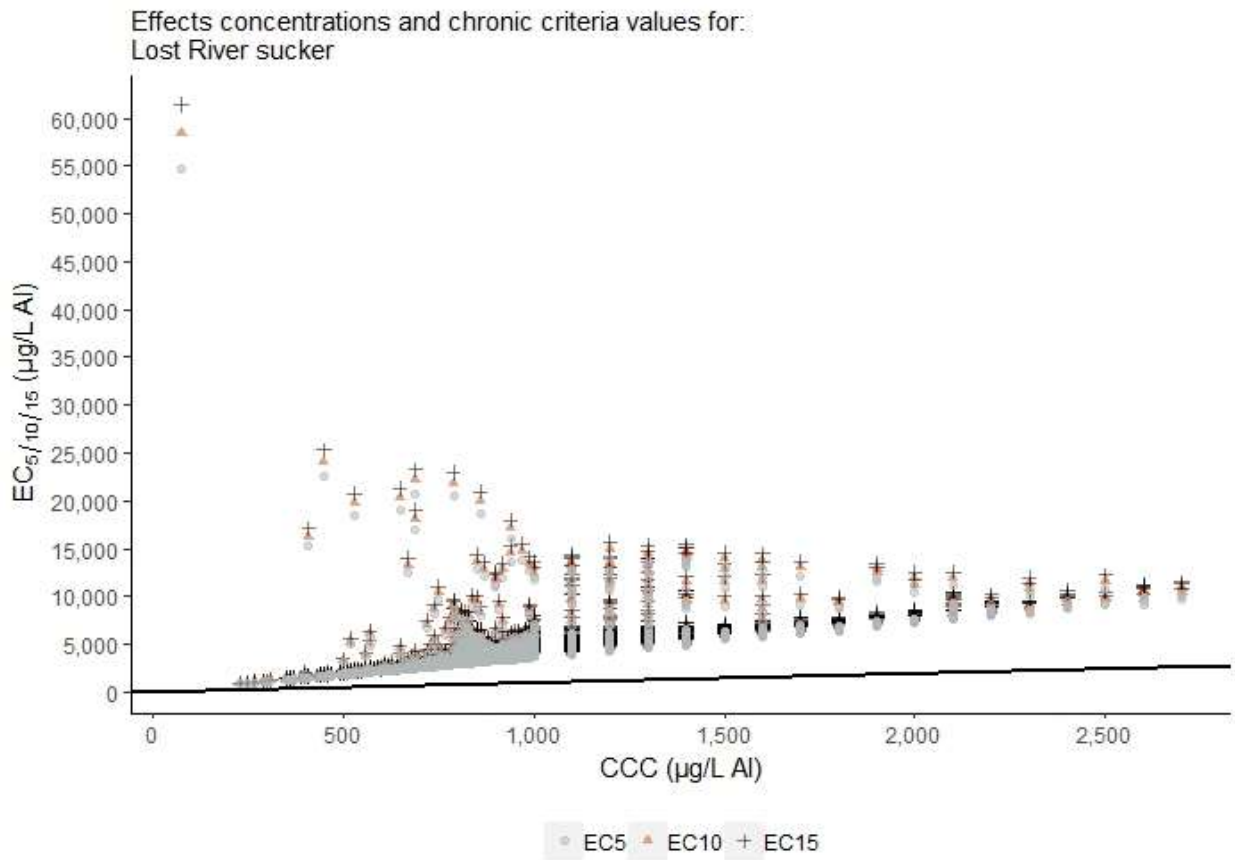

Figure 5-45. Paired Lost River sucker chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the Lost River sucker.

Table 5-40. The number and percentage of occurrences where the chronic criterion magnitude exceed Lost River sucker EC_5, EC_{10}, and EC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing Lost River sucker range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (growth/biomass) at each effects concentration. AEL for each ECx value is 0%.

Species	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}		CCC > EC_{15}	
Name		n	%	AEL	N	%	n	%
Lost River sucker	1,830	0	0	0	0	0	0	0

5.2.11.2.5 Lost River Sucker: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC, across the range of ecoregions within the Lost River sucker range, was less than the Lost River sucker chronic low effect threshold (i.e., EC_5) in all water chemistry scenarios. The chronic Al CCC never exceeded the Lost River sucker EC_{10} or EC_{15} (Table 5-40). As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for Lost River sucker based on the chronic criterion.

5.2.12 Shortnose Sucker (Chasmistes brevirostris)

5.2.12.1 Shortnose Sucker Acute Aluminum Effects Assessment

5.2.12.1.1 Identifying Shortnose Sucker Acute Aluminum Data

Species-level acute toxicity data were not available for shortnose sucker or any other members of the Family Catostomidae. Acute toxicity data available were available for members of the Order Cypriniformes to calculate a Cypriniformes OMAV (see Appendix A.1). The Cypriniformes OMAV was >21,937 µg/L total aluminum (normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) and was representative of shortnose sucker (see Section 5.2.10.1.1 and Table 5-35).

5.2.12.1.2 Deriving Shortnose Sucker Acute Adjustment Factors

No quantitatively acceptable acute TAFs were available for shortnose sucker or any other vertebrate species (see Table 5-3). As a result, the acute MAF (3.881) was selected as the adjustment factor to relate the shortnose sucker acute LC_{50} value to a 5% effect level (see Section 5.2.10.1.2). The same acute C-R curves used to derive the acute MAF (i.e., LC_{50}:LC_5) were also used to calculate LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors of 2.787 and 2.261, respectively (see Appendix C.1 and Appendix C.2). Similar to the acute MAF, the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were intended to provide a mechanism for relating acute effect concentrations (i.e., LC_{50}) to lower-level effect concentrations (i.e., LC_{15} and LC_{10} values, respectively) across a broad range of taxa (see Section 5.2.10.1.2).

5.2.12.1.3 Calculating Shortnose Sucker Acute Aluminum Minimum Effect Threshold Values

The acute MAF (i.e., LC_{50}:LC_5 adjustment factor) as well as the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were applied, as a denominator, to the Cypriniformes OMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value representative of the shortnose sucker:

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) = **LC_5** = 21,937 µg/L / 3.881 = **5,652 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) = **LC_{10}** = 21,937 µg/L / 2.787 = **7,871 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) = **LC_{15}** = 21,937 µg/L / 2.261 = **9,702 µg/L***

5.2.12.1.4 Shortnose Sucker: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude and vertebrate species sensitivity across water chemistry, shortnose sucker acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate an acute criterion magnitude and corresponding shortnose sucker acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and shortnose sucker acute effect threshold data (i.e., LC_5, LC_{10}, and LC_{15}) were compared to the specific water quality data that were collected from level-III ecoregions within the range of the shortnose sucker. To visually display the comparison, criterion magnitudes (X-axis) and corresponding shortnose sucker acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-46). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, LC_{10}, or LC_{15}) and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and shortnose sucker acute effect threshold data (i.e., LC_5, LC_{10}, LC_{15}) are reported in Table 5-41.

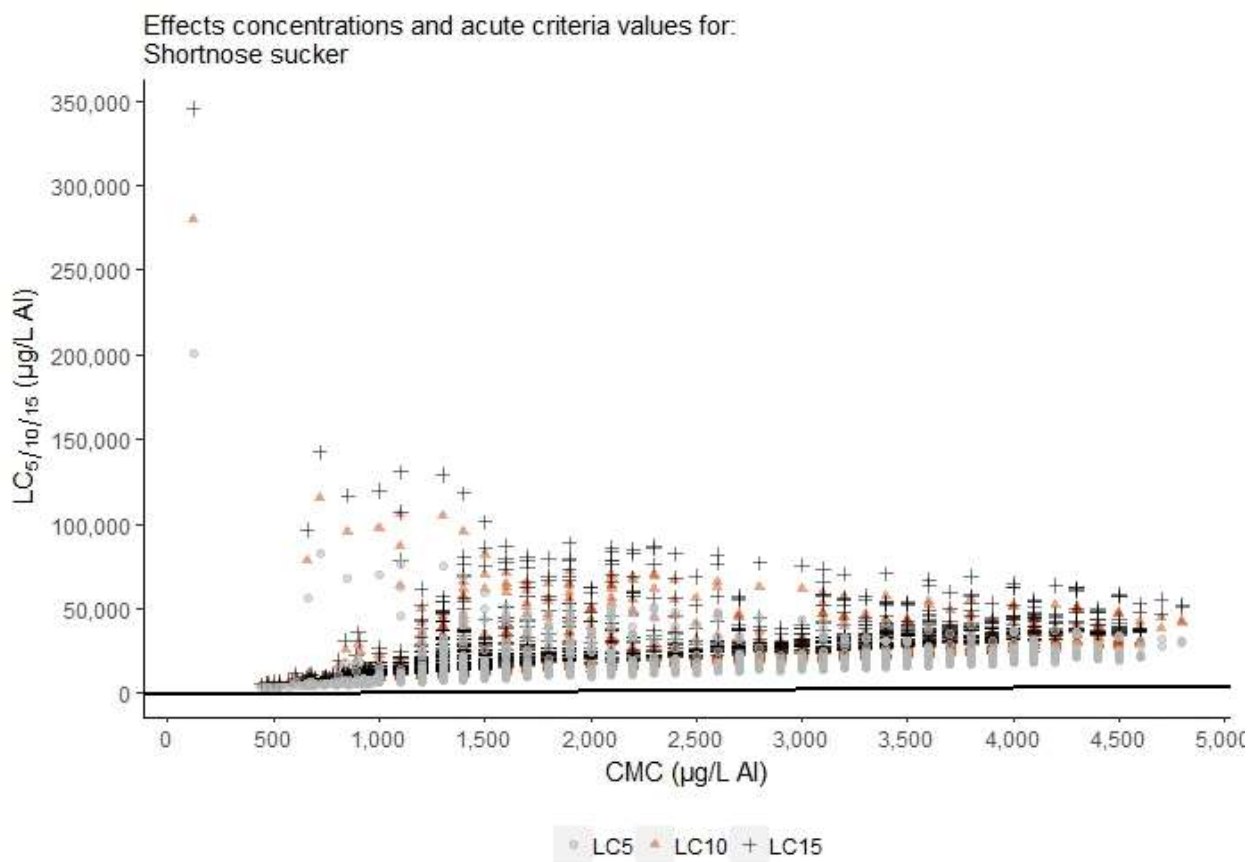

Figure 5-46. Paired shortnose sucker acute effect concentrations and acute criterion values in Oregon ecoregions containing the shortnose sucker.

Table 5-41. The number and percentage of occurrences where the acute criterion magnitude exceed shortnose sucker LC_5, LC_{10}, and LC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing shortnose sucker range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (mortality) at each effects concentration. AEL for each LCx value is 0%.

Species	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}		CMC > LC_{15}	
Name		n	%	AEL	N	%	n	%
Shortnose sucker	1,830	0	0	0	0	0	0	0

5.2.12.1.5 Shortnose Sucker: Tier I Acute Assessment Conclusion

The acute aluminum CMC was less than the shortnose sucker acute low effect threshold (i.e., LC_5) under all water chemistry scenarios in Oregon ecoregions within the shortnose sucker range (Table 5-41). The LC_5 range was 3,067 to 201,115 µg/L and the CMC range was 120 to 4,800 µg/L. As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for shortnose sucker based on the acute criterion.

5.2.12.2 Shortnose Sucker Chronic Aluminum Effects Assessment

5.2.12.2.1 Identifying Shortnose Sucker Chronic Aluminum Data

Species-level chronic toxicity data were not available for shortnose sucker or any other members of the Family Catostomidae. Two species within the Order Cypriniformes had chronic Al toxicity data to derive a Cypriniformes OMCV. The Cypriniformes OMCV is 1,797 µg/L total aluminum (normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) and was representative of the shortnose sucker (see Section 5.2.10.2.1 and Table 5-37).

5.2.12.2.2 Deriving Shortnose Sucker Chronic Adjustment Factors

The only quantitatively acceptable C-R model from the available studies used to determine the Cypriniformes OMCV produced an EC_{20}:EC_5 ratio of 1.168 that served as the Cypriniformes order-level TAF, representative of the shortnose sucker (see Section 5.2.10.2.2). The EC_{20}:EC_{15} (1.042) ratio and EC_{20}:EC_{10} ratio (1.093) from the same fathead minnow test also served as adjustment factors for transforming chronic EC_{20} values into lower-level effect concentrations (i.e., EC_{15} and EC_{10}, respectively) for members of the Order Cypriniformes, including shortnose sucker (see Section 5.2.10.2.2).

5.2.12.2.3 Calculating Shortnose Sucker Chronic Aluminum Minimum Effect Threshold Values

The Cypriniformes order-level TAF (i.e., EC_{20}:EC_5 adjustment factor) as well as the EC_{20}:EC_{10} and EC_{20}:EC_{15} adjustment factors were applied, as a denominator, to the Cypriniformes OMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the shortnose sucker:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) = **EC_5** = 1,797 µg/L / 1.168 =* ***1,539 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) = **EC_{10}** = 1,797 µg/L / 1.093 =* ***1,643 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) = **EC_{15}** = 1,797 µg/L / 1.042 =* ***1,725 µg/L***

5.2.12.2.4 Shortnose Sucker: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and vertebrate species sensitivity across water chemistry, shortnose sucker chronic EC_5, EC_{10}, and EC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding shortnose sucker chronic low effect threshold values (i.e., EC_5, EC_{10}, EC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and shortnose sucker chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were compared to water quality data that were collected from level-III ecoregions within the range of the shortnose sucker. To visually display the comparison, criterion magnitudes (X-axis) and corresponding shortnose sucker chronic effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-47). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and shortnose sucker chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) are reported in Table 5-42.

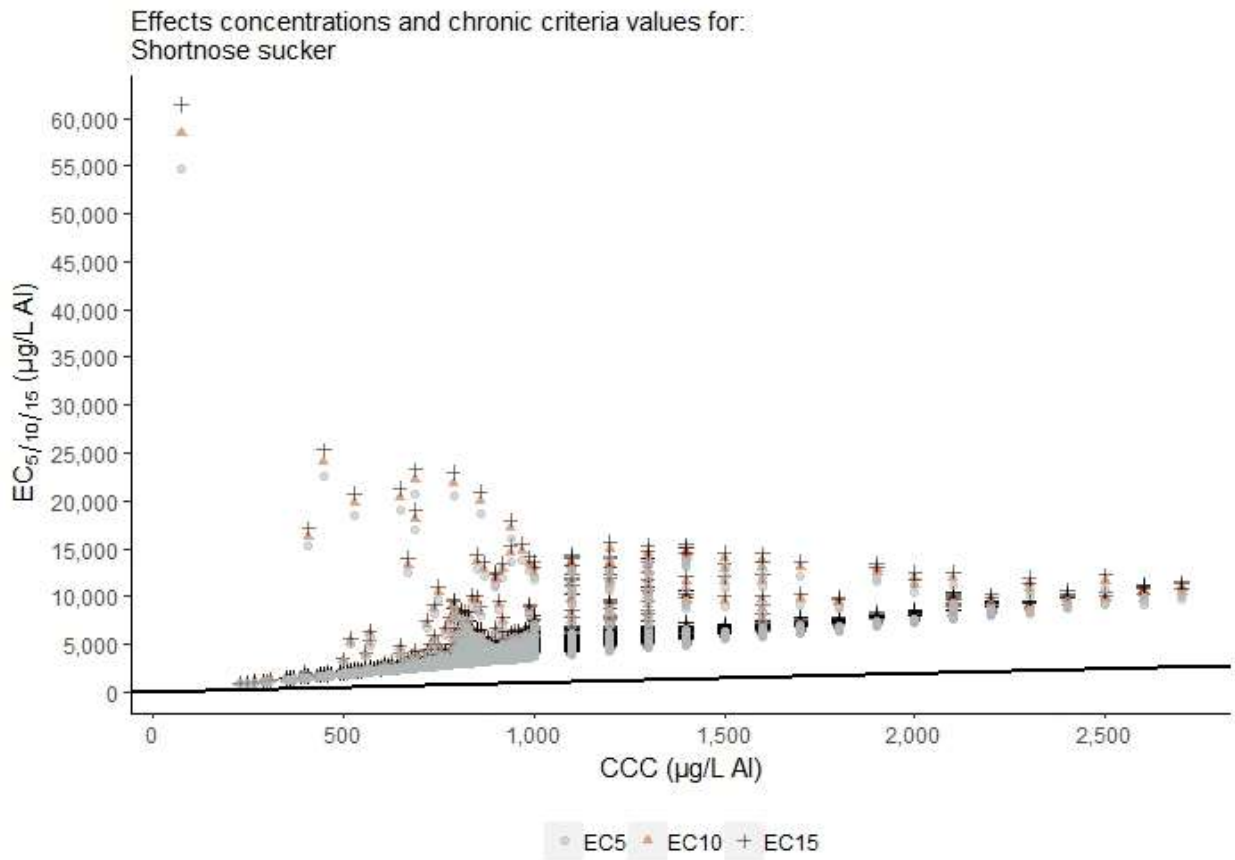

Figure 5-47. Paired shortnose sucker chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the shortnose sucker.

Table 5-42. The number and percentage of occurrences where the chronic criterion magnitude exceed shortnose sucker EC_5, EC_{10}, and EC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing shortnose sucker range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (growth/biomass) at each effects concentration. AEL for each ECx value is 0%.

Species	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}		CCC > EC_{15}	
Name		n	%	AEL	N	%	n	%
Shortnose sucker	1,830	0	0	0	0	0	0	0

5.2.12.2.5 Shortnose Sucker: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC, across the range of ecoregions that contain shortnose sucker range, was greater than the shortnose sucker chronic low effect threshold (i.e., EC_5) under all water chemistry scenarios. The chronic Al CCC never exceeded the shortnose sucker EC_{10} or EC_{15}

(Table 5-42). As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for shortnose sucker based on the chronic criterion.

5.2.13 Borax Lake Chub (Gila boraxobius)

5.2.13.1 Borax Lake Chub Acute Aluminum Effects Assessment

5.2.13.1.1 Identifying Borax Lake Chub Acute Aluminum Data

Species level acute toxicity data were not available for Borax Lake chub or any other members of the Genus *Gila*. Acute toxicity data available were available for members of the Family Cyprinidae (see Appendix A.1). The Cyprinidae family mean acute value (FMAV; >21,937 µg/L total aluminum, normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) was based on two SMAVs (representing two different genera; see Table 5-43). The Rio Grande silvery minnow SMAV was based on one acute toxicity test with 3-5 days-post hatch (dph) larva, while the fathead minnow SMAV was based on the geometric mean of three acute toxicity tests (Call et al. (1984) tested 32-33 dph juvenile fathead minnow in two separate tests; Buhl (2002) tested 4-6 dph larva fathead minnow; see Appendix A.1). All four individual toxicity tests produced non-definitive LC_{50} values (i.e., "greater than" LC_{50} values) because the highest test concentrations used in these tests did not result in ≥50% mortality. Therefore, the Cyprinidae FMAV that was representative of Borax Lake chub may have resulted in a conservative LC_{50} estimate.

Table 5-43. Data used to calculate the Cyprinidae Family Mean Acute Value representative of the acute aluminum sensitivity of Borax Lake chub.

Family	Species	SMAV (µg/L)[a]	GMAV (µg/L)[a]	FMAV (µg/L)[a]
Cyprinidae	Borax Lake chub, *Gila boraxobius*	N/A	N/A	>21,937
Cyprinidae	Rio Grande silvery minnow, *Hybognathus amarus*	>21,779	>21,779	
Cyprinidae	Fathead minnow, *Pimephales promelas*	>22,095	>22,095	

[a] All values expressed as total aluminum, normalized to pH 7, DOC of 1 mg/L and 100 mg/L hardness as $CaCO_3$; normalized using MLR equations identified in (USEPA, 2018).
N/A: not available.

5.2.13.1.2 Deriving Borax Lake Chub Acute Adjustment Factors

No quantitatively acceptable acute TAFs were available for Borax Lake chub or any other vertebrate species (see Table 5-3 As a result, all genus-level TAFs (based on quantitatively acceptable C-R curves) were averaged (geometric mean; n = 2) to calculate the acute MAF. The acute MAF serves as a generic, non-taxa-specific adjustment factor meant to broadly relate acute LC_{50} values to lower level effects concentrations (i.e., LC_5). Because no quantitatively acceptable TAFs were available for vertebrate species, the acute MAF is based on invertebrate species only (see Table 5-4). Unlike, salmonids no qualitatively acceptable acute C-R models for

fishes in the Order Cypriniformes were available to be considered the development of a TAF based on qualitatively acceptable data (see Table 5-6 and Appendix B.1).

The acute MAF was calculated as the geometric mean of the *Physa* genus-level TAF (i.e., 3.597) and the *Ceriodaphnia* genus-level TAF (i.e., 4.189) and is 3.881 (Table 5-3). The acute C-R curves used to derive the acute MAF (i.e., LC_{50}:LC_5) were also used to calculate LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors of 2.787 and 2.261, respectively (see Appendix C.1 and Appendix C.2). Similar to the acute MAF (i.e., LC_{50}:LC_5), the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were intended to provide a mechanism for relating acute effect concentrations (i.e., LC_{50}) to lower-level effect concentrations (i.e., LC_{10} and LC_{15} values, respectively) across a broad range of taxa.

5.2.13.1.3 Calculating Borax Lake Chub Acute Aluminum Minimum Effect Threshold Values

The acute MAF (i.e., LC_{50}:LC_5 adjustment factor) as well as the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were applied, as a denominator, to the Cyprinidae FMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value representative of the Borax Lake chub:

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) = **LC_5** = 21,937 µg/L / 3.881 = **5,652 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) = **LC_{10}** = 21,937 µg/L / 2.787 = **7,870 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) = **LC_{15}** = 21,937 µg/L / 2.261 = **9,702 µg/L***

5.2.13.1.4 Borax Lake Chub: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude and vertebrate species sensitivity across water chemistry, Borax Lake chub acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate an acute criterion magnitude and corresponding Borax Lake chub acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and Borax Lake chub acute effect threshold data (i.e., LC_5, LC_{10}, and LC_{15}) were compared to the specific water quality data that were collected from level-III ecoregions within the range of the Borax Lake chub. To visually display the comparison, criterion magnitudes (X-axis) and corresponding Borax Lake chub acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-48). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, LC_{10}, or LC_{15}) and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and Borax Lake chub acute effect threshold data (i.e., LC_5, LC_{10}, LC_{15}) are reported in Table 5-44.

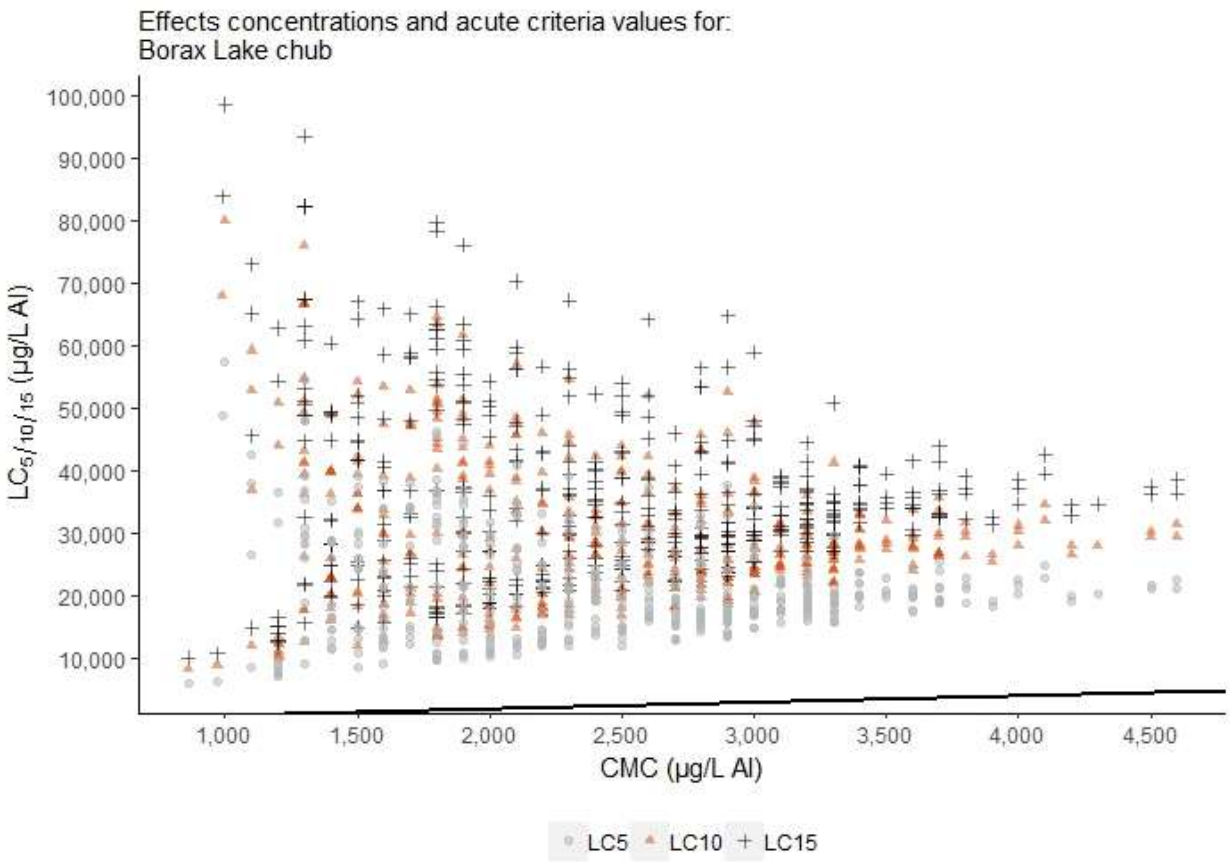

Figure 5-48. Paired Borax Lake chub acute effect concentrations and acute criterion values in Oregon ecoregions containing the Borax Lake chub.

Table 5-44. The number and percentage of occurrences where the acute criterion magnitude exceed Borax Lake chub LC_5, LC_{10}, and LC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing Borax Lake chub range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (mortality) at each effects concentration. AEL for each LCx value is 0%.

Species	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}		CMC > LC_{15}	
Name		n	%	AEL	n	%	n	%
Borax Lake chub	443	0	0	0	0	0	0	0

5.2.13.1.5 Borax Lake Chub: Tier I Acute Assessment Conclusion

The acute aluminum CMC was less than the Borax Lake chub acute low effect threshold (i.e., LC_5) in all water chemistry scenarios in Oregon ecoregions within the Borax Lake chub range (Table 5-44). The LC_5 range was 5,897 to 57,432 µg/L and the CMC range was 860 to 4,600

µg/L. As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for Borax Lake chub based on the acute criterion.

5.2.13.2 Borax Lake Chub Chronic Aluminum Effects Assessment

5.2.13.2.1 Identifying Borax Lake Chub Chronic Aluminum Data

Species-level chronic toxicity data were not available for Borax Lake chub or any other members of the Genus *Gila*. Two species within the Family Cyprinidae had chronic Al toxicity data to derive a Cyprinidae family mean chronic value (FMCV). The Cyprinidae FMCV was calculated from the *Danio* GMCV and the *Pimephales* GMCV. The *Danio* GMCV is based directly on the zebrafish SMCV, which was determined from a single early life stage (<36 hours post hatch [hph]) test that exposed embryos to Al for 33 days with a biomass endpoint (OSU (2013); also referred to as Cardwell et al. (2018); see Appendix A.2). The *Pimephales* GMCV is based directly on the fathead minnow SMCV, which was calculated as the geometric mean of two fathead minnow tests. Kimball (1978) exposed fathead minnow to Al for 28 days and reported a biomass-based EC_{20} of 2,690 µg/L total aluminum (normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L; see Appendix A.2). OSU (2012); also referred to as Cardwell et al. (2018), exposed fathead minnow embryos (<24 hph) to Al for 33 days and reported a 20% reduction in fry survival occurring at a concentration of 2,154 µg/L total aluminum (normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L; see Appendix A.2). Together, these chronic tests resulted in a Cyprinidae FMCV of 1,797 µg/L total aluminum (normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) that was representative of the Borax Lake chub (Table 5-45).

Table 5-45. Data used to calculate the Cyprinidae Family Mean Chronic Value representative of the chronic aluminum sensitivity of Borax Lake chub.

Family	Species	SMCV (µg/L)[a]	GMCV (µg/L)[a]	FMCV (µg/L)[a]
Cyprinidae	Borax Lake chub, *Gila boraxobius*	N/A	N/A	1,797
Cyprinidae	Zebrafish, *Danio rerio*	1,342	1,342	
Cyprinidae	Fathead minnow, *Pimephales promelas*	2,407	2,407	

[a] All values expressed as total aluminum, normalized to pH 7, DOC of 1 mg/L and 100 mg/L hardness as $CaCO_3$; normalized using MLR equations identified in (USEPA, 2018).
N/A: not available.

5.2.13.2.2 Deriving Borax Lake Chub Chronic Adjustment Factors

Of the three chronic tests used to calculate the FMCV, only the fathead minnow test reported by Kimball (1978) included C-R data that resulted in a quantitatively acceptable C-R model (Al-Chronic 24; see Appendix B.2). The model generated from the C-R data for the other fathead minnow test (OSU, 2012)was determined to be qualitatively acceptable because of a poor fit and significant variation in responses at low treatment concentrations (Al-Chronic 31). The C-R model generated from the zebrafish test (OSU, 2013) was determined to be unacceptable because

of poor fit and resultant in 95% confidence bounds that extended into the negative number range (Al-Chronic 33; see Appendix B.2). The fathead minnow EC_{20}:EC_5 ratio (1.168) served as the Cyprinidae family-level TAF, representative of the Borax Lake chub (Al-Chronic 24; see Appendix B.2). The EC_{20}:EC_{15} (1.042) ratio and EC_{20}:EC_{10} ratio (1.093) from the same fathead minnow test served as adjustment factors for transforming chronic EC_{20} values into lower-level effect concentrations (i.e., EC_{15} and EC_{10}, respectively) for members of the Family Cyprinidae, including the Borax Lake chub (see AL-Chronic 24; Appendix C.3 and Appendix C.4).

5.2.13.2.3 Calculating Borax Lake Chub Chronic Aluminum Minimum Effect Threshold Values

The Cyprinidae family-level TAF (i.e., EC_{20}:EC_5 adjustment factor) as well as the EC_{20}:EC_{10} and EC_{20}:EC_{15} adjustment factors were applied, as a denominator, to the Cyprinidae FMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the Borax Lake chub:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) = **EC_5** = 1,797 µg/L / 1.168 = **1,539 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) = **EC_{10}** = 1,797 µg/L / 1.093 = **1,643 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) = **EC_{15}** = 1,797 µg/L / 1.042 = **1,725 µg/L***

5.2.13.2.4 Borax Lake Chub: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and vertebrate species sensitivity across water chemistry, Borax Lake chub chronic EC_5, EC_{10}, and EC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding Borax Lake chub chronic low effect threshold values (i.e., EC_5, EC_{10}, EC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and Borax Lake chub chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were compared to water quality data that were collected from level-III ecoregions within the range of the Borax Lake chub. To visually display the comparison, criterion magnitudes (X-axis) and corresponding Borax Lake chub chronic effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-49). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and Borax Lake chub chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) are reported in Table 5-46.

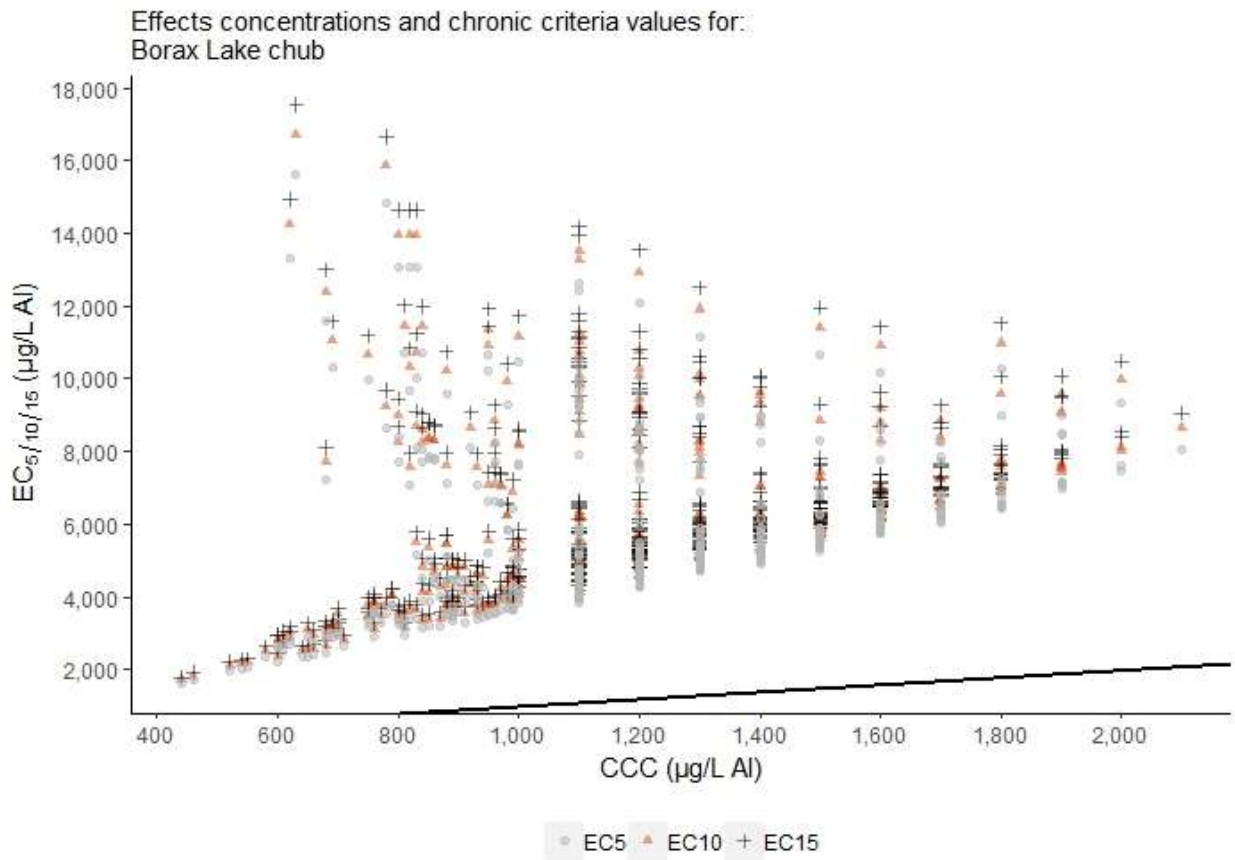

Figure 5-49. Paired Borax Lake chub chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the Borax Lake chub.

Table 5-46. The number and percentage of occurrences where the chronic criterion magnitude exceed Borax Lake chub EC_5, EC_{10}, and EC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing Borax Lake chub range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (growth/biomass) at each effects concentration. AEL for each ECx value is 0%.

Species	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}		CCC > EC_{15}	
Name		n	%	AEL	n	%	n	%
Borax Lake chub	443	0	0	0	0	0	0	0

5.2.13.2.5 Borax Lake Chub: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC, across the range of ecoregions that contain the Borax Lake chub range, was less than the Borax Lake chub chronic low effect threshold (i.e., EC_5) in all water chemistry. The chronic Al CCC never exceeded the Borax Lake chub EC_{10} or EC_{15} (Table 5-46).

As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for Borax Lake chub based on the chronic criterion.

5.2.14 Hutton Tui Chub (Gila bicolor ssp.)

5.2.14.1 Hutton Tui Chub Acute Aluminum Effects Assessment

5.2.14.1.1 Identifying Hutton Tui Chub Acute Aluminum Data

Species-level acute toxicity data were not available for Hutton tui chub or any other members of the Genus *Gila*. Acute toxicity data available were available for members of the Family Cyprinidae to calculate a Cyprinidae FMAV (see Appendix A.1). The Cyprinidae FMAV was >21,937 µg/L total aluminum (normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) and was representative of Hutton tui chub (see Section 5.2.13.1.1 and Table 5-43).

5.2.14.1.2 Deriving Hutton Tui Chub Acute Adjustment Factors

No quantitatively acceptable acute TAFs were available for Hutton tui chub or any other vertebrate species (see Table 5-3). As a result, the MAF (3.881) was selected as the adjustment factor to relate the Hutton tui chub acute LC_{50} value to a 5% effect level (see Section 5.2.13.1.2). The same acute C-R curves used to derive the acute MAF (i.e., LC_{50}:LC_5) were also used to calculate LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors of 2.787 and 2.261, respectively (see Appendix C.1 and Appendix C.2). Similar to the acute MAF, the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were intended to provide a mechanism for relating acute effect concentrations (i.e., LC_{50}) to lower-level effect concentrations (i.e., LC_{10} and LC_{15} values, respectively) across a broad range of taxa (see Section 5.2.13.1.2).

5.2.14.1.3 Calculating Hutton Tui Chub Acute Aluminum Minimum Effect Threshold Values

The acute MAF (i.e., LC_{50}:LC_5 adjustment factor) as well as the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were applied, as a denominator, to the Cyprinidae FMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value representative of the Hutton tui chub:

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) =* ***LC_5*** *= 21,937 µg/L / 3.881 =* ***5,652 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) =* ***LC_{10}*** *= 21,937 µg/L / 2.787 =* ***7,870 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) =* ***LC_{15}*** *= 21,937 µg/L / 2.261 =* ***9,702 µg/L***

5.2.14.1.4 Hutton Tui Chub: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude and vertebrate species sensitivity across water chemistry, Hutton tui chub acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate an

acute criterion magnitude and corresponding Hutton tui chub acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and Hutton tui chub acute effect threshold data (i.e., LC_5, LC_{10}, and LC_{15}) were compared to the specific water quality data that were collected from level-III ecoregions within the range of the Hutton tui chub. To visually display the comparison, criterion magnitudes (X-axis) and corresponding Hutton tui chub acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-50). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, LC_{10}, or LC_{15}) and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and Hutton tui chub acute effect threshold data (i.e., LC_5, LC_{10}, LC_{15}) are reported in Table 5-47.

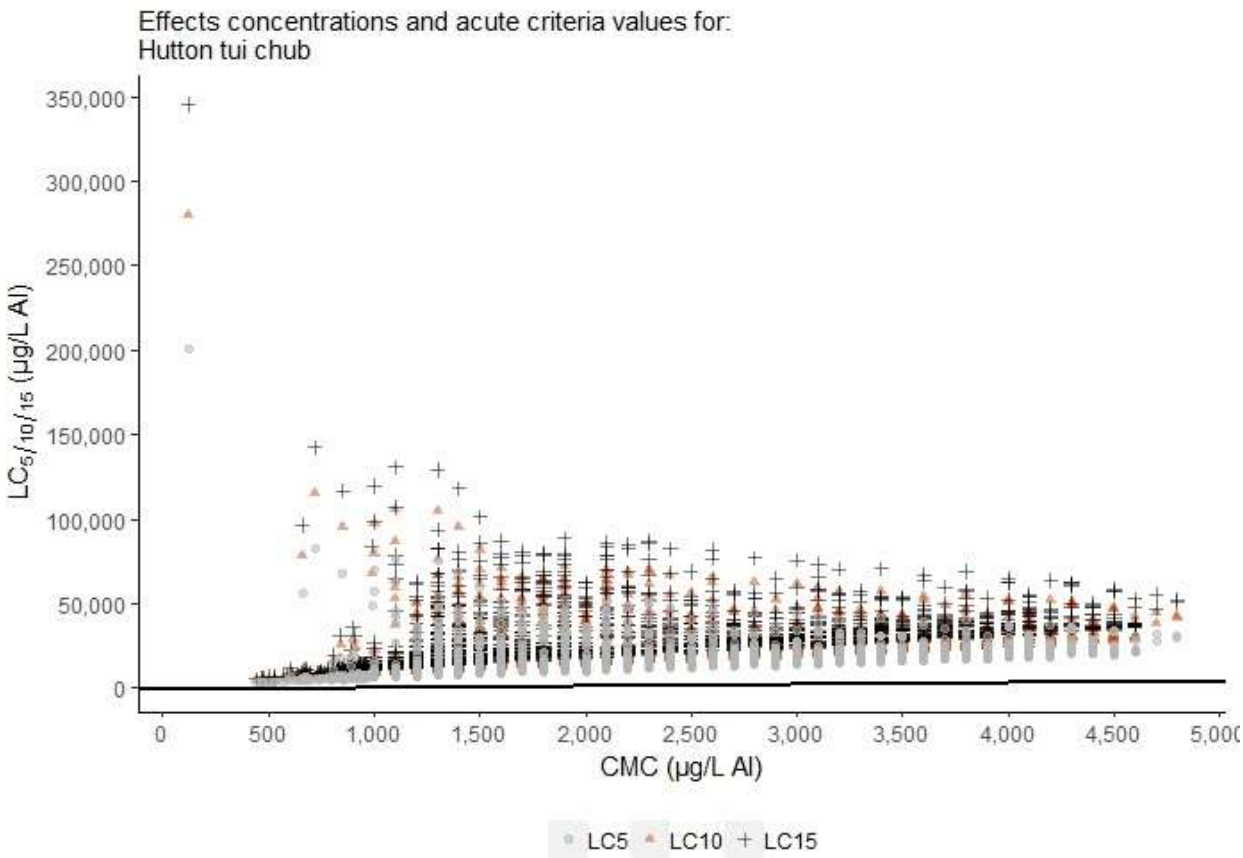

Figure 5-50. Paired Hutton tui chub acute effect concentrations and acute criterion values in Oregon ecoregions containing the Hutton tui chub.

Table 5-47. The number and percentage of occurrences where the acute criterion magnitude exceed Hutton tui chub LC_5, LC_{10}, and LC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing Hutton tui chub range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (mortality) at each effects concentration. AEL for each LCx value is 0%.

Species	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}		CMC > LC_{15}	
Name		n	%	AEL	n	%	n	%
Hutton tui chub	2,273	0	0	0	0	0	0	0

5.2.14.1.5 Hutton Tui Chub: Tier I Acute Assessment Conclusion

The acute aluminum CMC was less than the Hutton tui chub acute low effect threshold (i.e., LC_5) in all water chemistry in Oregon ecoregions that overlap with Hutton tui chub range (Table 5-47). The LC_5 range was 3,067 to 201,115 µg/L and the CMC range was 120 to 4,800 µg/L. As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for Hutton tui chub based on the acute criterion.

5.2.14.2 Hutton Tui Chub Chronic Aluminum Effects Assessment

5.2.14.2.1 Identifying Hutton Tui Chub Chronic Aluminum Data

Species-level chronic toxicity data were not available for Hutton tui chub or any other members of the Genus *Gila*. Two species within the Family Cyprinidae had chronic Al toxicity data to derive a Cyprinidae FMCV. The Cyprinidae FMCV is 1,797 µg/L total aluminum (normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) and was representative of the Hutton tui chub (see Section 5.2.13.2.1 and Table 5-45).

5.2.14.2.2 Deriving Hutton Tui Chub Chronic Adjustment Factors

The only quantitatively acceptable C-R model from the available studies used to determine the Cyprinidae FMCV produced an EC_{20}:EC_5 ratio of 1.168 that served as the Cyprinidae family-level TAF, representative of the Hutton tui chub (see Section 5.2.13.2.2). The EC_{20}:EC_{15} (1.042) ratio and EC_{20}:EC_{10} ratio (1.093) from the same fathead minnow test also served as adjustment factors for transforming chronic EC_{20} values into lower-level effect concentrations (i.e., EC_{15} and EC_{10}, respectively) for members of the Family Cyprinidae, including Hutton tui chub (see Section 5.2.13.2.2).

5.2.14.2.3 Calculating Hutton Tui Chub Chronic Aluminum Minimum Effect Threshold Values

The Cyprinidae family-level TAF (i.e., EC_{20}:EC_5 adjustment factor) as well as the EC_{20}:EC_{10} and EC_{20}:EC_{15} adjustment factors were applied, as a denominator, to the Cyprinidae FMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the Hutton tui chub:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) = **EC_5** = 1,797 µg/L / 1.168 =* ***1,539 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) = **EC_{10}** = 1,797 µg/L / 1.093 =* ***1,643 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) = **EC_{15}** = 1,797 µg/L / 1.042 =* ***1,725 µg/L***

5.2.14.2.4 Hutton Tui Chub: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and vertebrate species sensitivity across water chemistry, Hutton tui chub chronic EC_5, EC_{10}, and EC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding Hutton tui chub chronic low effect threshold values (i.e., EC_5, EC_{10}, EC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and Hutton tui chub chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were compared to water quality data that were collected from level-III ecoregions within the range of the Hutton tui chub. To visually display the comparison, criterion magnitudes (X-axis) and corresponding Hutton tui chub chronic effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-51). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and Hutton tui chub chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) are reported in Table 5-48.

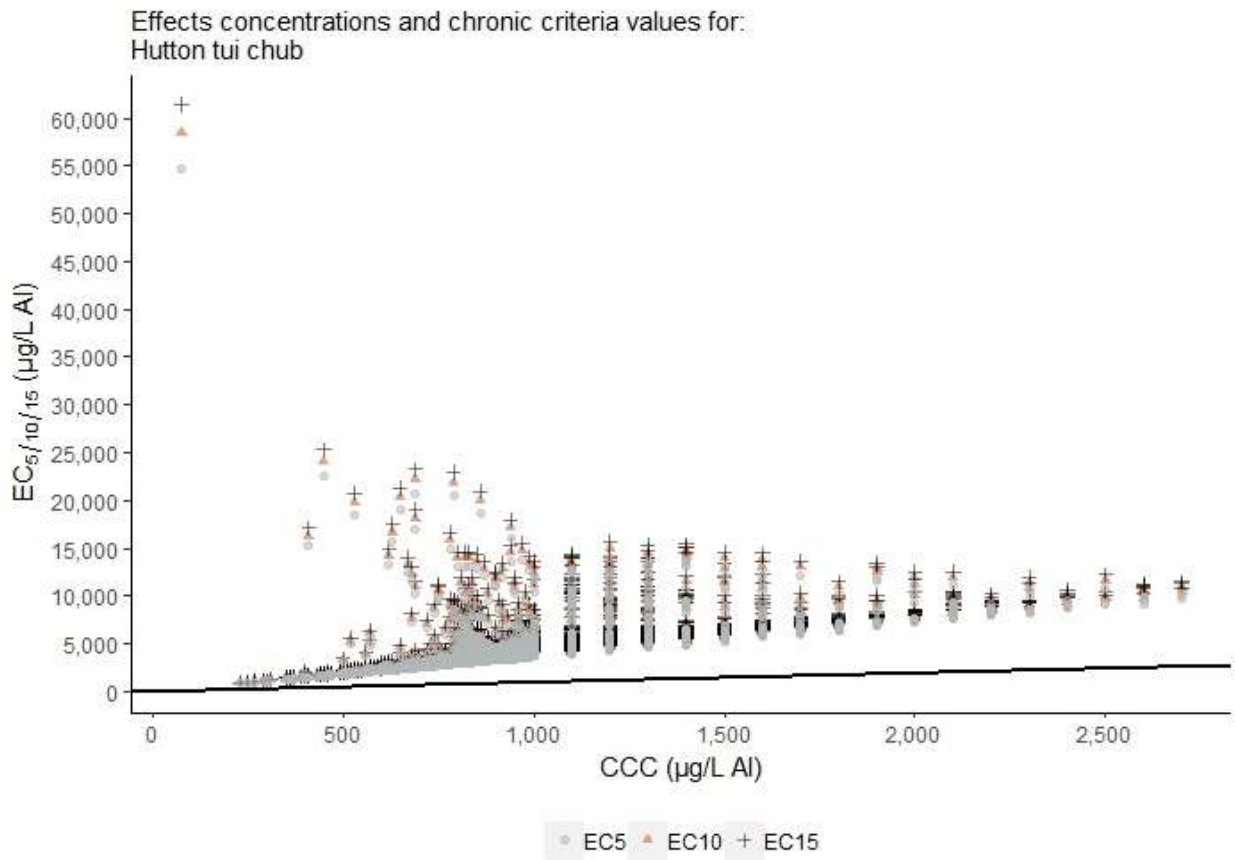

Figure 5-51. Paired Hutton tui chub chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the Hutton tui chub.

Table 5-48. The number and percentage of occurrences where the chronic criterion magnitude exceed Hutton tui chub EC_5, EC_{10}, and EC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing Hutton tui chub range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (mortality) at each effects concentration. AEL for each LCx value is 0%.

Species	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}		CCC > EC_{15}	
Name		n	%	AEL	n	%	n	%
Hutton tui chub	2,273	0	0	0	0	0	0	0

5.2.14.2.5 Hutton Tui Chub: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC, across the range of ecoregions that contain Hutton tui chub range, was less than the Hutton tui chub chronic low effect threshold (i.e., EC_5) in all water chemistry. The chronic Al CCC never exceeded the Hutton tui chub EC_{10} or EC_{15} (Table 5-48). As a result,

consideration of a refined exposure assessment through a weight of evidence approach was not required for Hutton tui chub based on the chronic criterion.

5.2.15 Foskett Speckled Dace (Rhinichthys osculus spp.)

5.2.15.1 Foskett Speckled Dace Acute Aluminum Effects Assessment

5.2.15.1.1 Identifying Foskett Speckled Dace Acute Aluminum Data

Species-level acute toxicity data were not available for Foskett speckled dace or any other members of the Genus *Gila*. Acute toxicity data available were available for members of the Family Cyprinidae to calculate a Cyprinidae FMAV (see Appendix A.1). The Cyprinidae FMAV was >21,937 µg/L total aluminum (normalized to pH 7, total hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) and was representative of Foskett speckled dace (see Section 5.2.13.1.1 and Table 5-43).

5.2.15.1.2 Deriving Foskett Speckled Dace Acute Adjustment Factors

No quantitatively acceptable acute TAFs were available for Foskett speckled dace or any other vertebrate species (see Table 5-3). As a result, the acute MAF (3.881) was selected as the adjustment factor to relate the Foskett speckled dace acute LC_{50} value to a 5% effect level (see Section 5.2.13.1.2). The same acute C-R curves used to derive the acute MAF (i.e., LC_{50}:LC_5) were also used to calculate LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors of 2.787 and 2.261, respectively (see Appendix C.1 and Appendix C.2). Similar to the acute MAF (i.e., LC_{50}:LC_5), the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were intended to provide a mechanism for relating acute effect concentrations (i.e., LC_{50}) to lower-level effect concentrations (i.e., LC_{10} and LC_{15} values, respectively) across a broad range of taxa (see Section 5.2.13.1.2).

5.2.15.1.3 Calculating Foskett Speckled Dace Acute Aluminum Minimum Effect Threshold Values

The acute MAF (i.e., LC_{50}:LC_5 adjustment factor) as well as the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were applied, as a denominator, to the Cyprinidae FMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value representative of the Foskett speckled dace:

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) = **LC_5** = 21,937 µg/L / 3.881 = **5,652 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) = **LC_{10}** = 21,937 µg/L / 2.787 = **7,870 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) = **LC_{15}** = 21,937 µg/L / 2.261 = **9,702 µg/L***

5.2.15.1.4 Foskett Speckled Dace: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude and vertebrate species sensitivity across water chemistry, Foskett speckled dace acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time

to calculate an acute criterion magnitude and corresponding Foskett speckled dace acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and Foskett speckled dace acute effect threshold data (i.e., LC_5, LC_{10}, LC_{15}) were compared to the specific water quality data that were collected from level-III ecoregions within the range of the Foskett speckled dace. To visually display the comparison, criterion magnitudes (X-axis) and corresponding Foskett speckled dace acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-52). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, LC_{10}, or LC_{15}) and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and Foskett speckled dace acute effect threshold data (i.e., LC_5, LC_{10}, LC_{15}) are reported in Table 5-49.

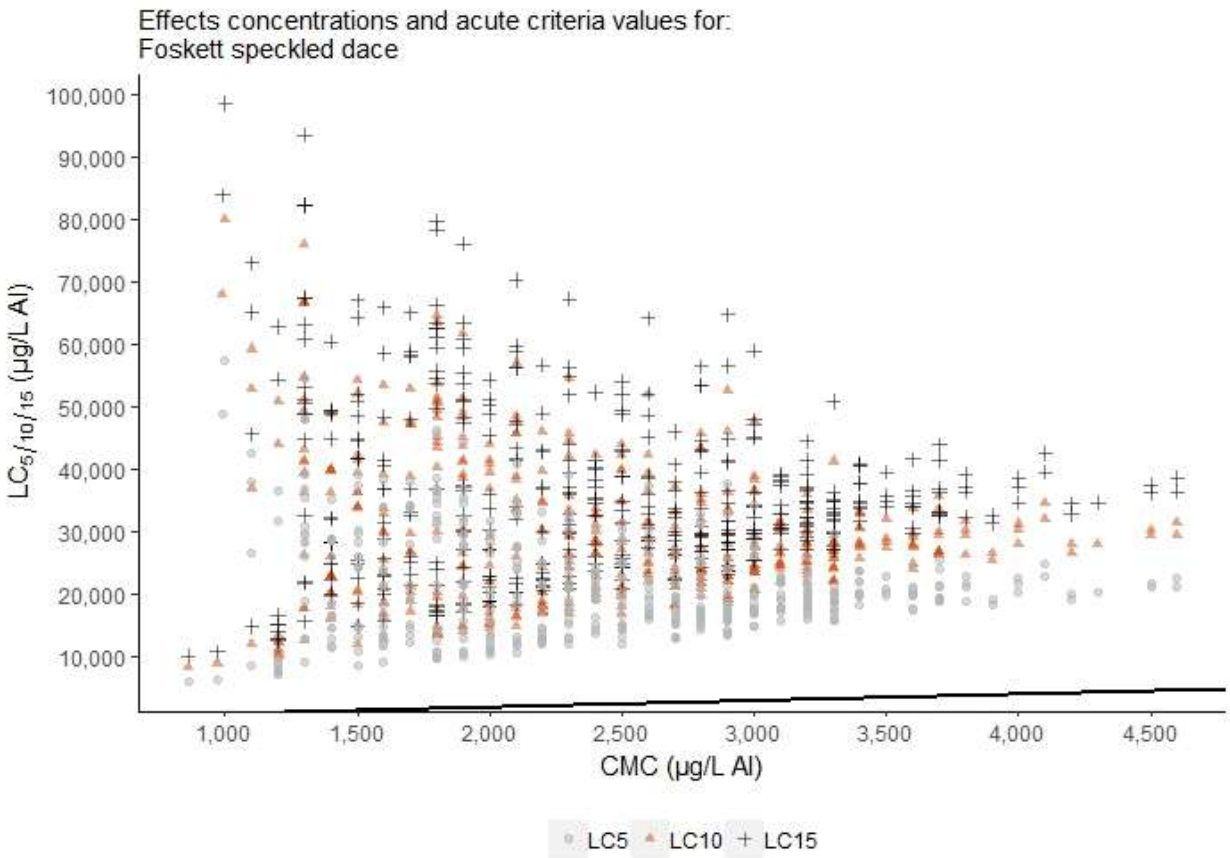

Figure 5-52. Paired Foskett speckled dace acute effect concentrations and acute criterion values in Oregon ecoregions containing the Foskett speckled dace.

Table 5-49. The number and percentage of occurrences where the acute criterion magnitude exceed Foskett speckled dace LC_5, LC_{10}, and LC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing Foskett speckled dace range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (mortality) at each effects concentration. AEL for each LCx value is 0%.

Species	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}		CMC > LC_{15}	
Name		n	%	AEL	n	%	n	%
Foskett speckled dace	443	0	0	0	0	0	0	0

5.2.15.1.5 Foskett Speckled Dace: Tier I Acute Assessment Conclusion

The acute aluminum CMC was less than the Foskett speckled dace acute low effect threshold (i.e., LC_5) in all water chemistry scenarios in Oregon ecoregions within the Foskett speckled dace range (Table 5-49). The LC_5 range was 5,897 to 57,432 µg/L and the CMC range was 860 to 4,600 µg/L. As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for Foskett speckled dace based on the acute criterion.

5.2.15.2 Foskett Speckled Dace Chronic Aluminum Effects Assessment

5.2.15.2.1 Identifying Foskett Speckled Dace Chronic Aluminum Data

Species-level chronic toxicity data were not available for Foskett speckled dace or any other members of the Genus *Gila*. Two species within the Family Cyprinidae had chronic Al toxicity data to derive a Cyprinidae FMCV. The Cyprinidae FMCV is 1,797 µg/L total aluminum (normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) and was representative of the Foskett speckled dace (see Section 5.2.13.2.1 and Table 5-45).

5.2.15.2.2 Deriving Foskett Speckled Dace Chronic Adjustment Factors

The only quantitatively acceptable C-R model from the available studies used to determine the Cyprinidae FMCV produced an EC_{20}:EC_5 ratio of 1.168 that served as the Cyprinidae family-level TAF, representative of the Foskett speckled dace (see Section 5.2.13.2.2). The EC_{20}:EC_{15} (1.042) ratio and EC_{20}:EC_{10} ratio (1.093) from the same fathead minnow test also served as adjustment factors for transforming chronic EC_{20} values into lower-level effect concentrations (i.e., EC_{15} and EC_{10}, respectively) for members of the family Cyprinidae, including the Foskett speckled dace (see Section 5.2.13.2.2).

5.2.15.2.3 Calculating Foskett Speckled Dace Chronic Aluminum Minimum Effect Threshold Values

The Cyprinidae family-level TAF (i.e., EC_{20}:EC_5 adjustment factor) as well as the EC_{20}:EC_{10} and EC_{20}:EC_{15} adjustment factors were applied, as a denominator, to the Cyprinidae FMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the Foskett speckled dace:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) =* ***EC_5*** *= 1,797 µg/L / 1.168 =* ***1,539 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) =* ***EC_{10}*** *= 1,797 µg/L / 1.093 =* ***1,643 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) =* ***EC_{15}*** *= 1,797 µg/L /1.042 =* ***1,725 µg/L***

5.2.15.2.4 Foskett Speckled Dace: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and vertebrate species sensitivity across water chemistry, Foskett speckled dace chronic EC_5, EC_{10}, and EC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding Foskett speckled dace chronic low effect threshold values (i.e., EC_5, EC_{10}, EC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and Foskett speckled dace chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were compared to water quality data that were collected from level-III ecoregions within the range of the Foskett speckled dace. To visually display the comparison, criterion magnitudes (X-axis) and corresponding Foskett speckled dace chronic effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-53). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and Foskett speckled dace chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) are reported in Table 5-50.

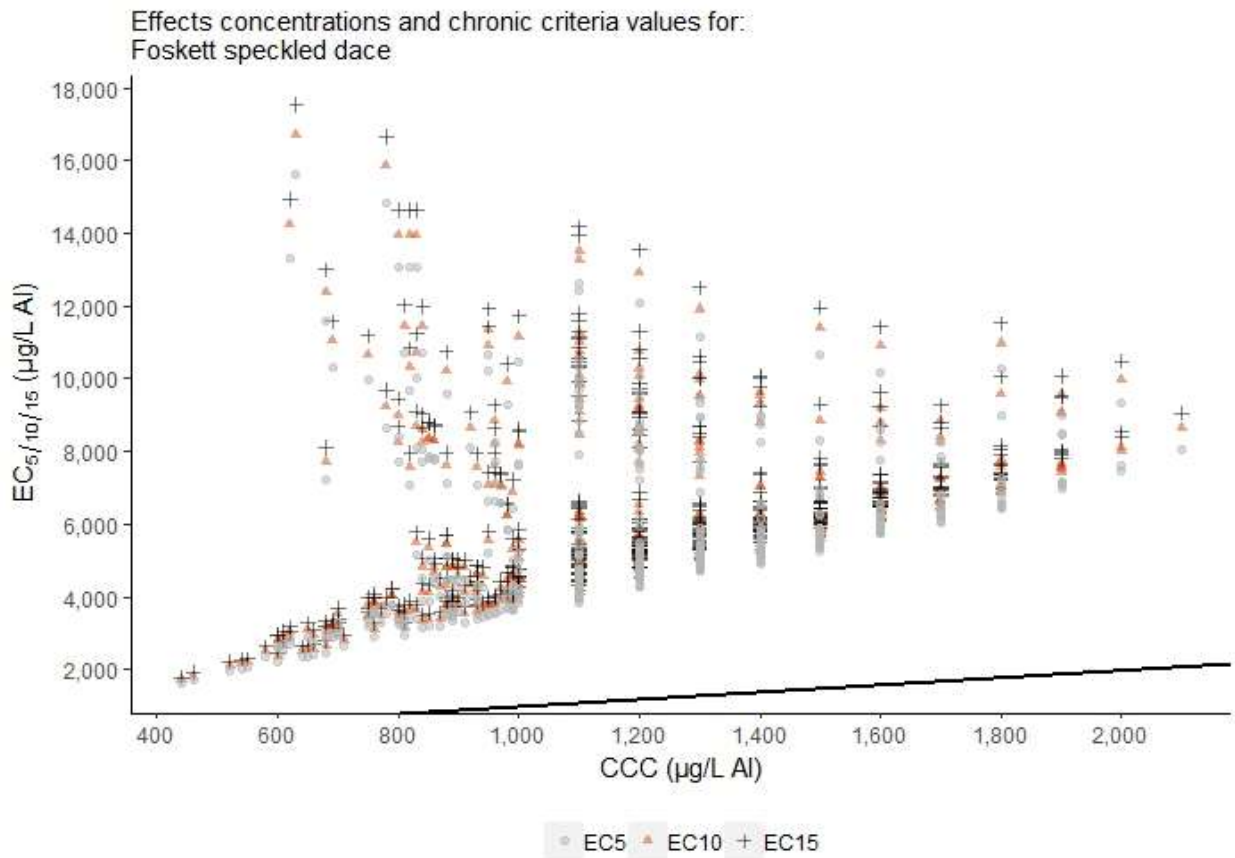

Figure 5-53. Paired Foskett speckled dace chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the Foskett speckled dace.

Table 5-50. The number and percentage of occurrences where the chronic criterion magnitude exceed Foskett speckled dace EC_5, EC_{10}, and EC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing Foskett speckled dace range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (growth/biomass) at each effects concentration. AEL for each ECx value is 0%.

Species	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}		CCC > EC_{15}	
Name		n	%	AEL	n	%	n	%
Foskett speckled dace	443	0	0	0	0	0	0	0

5.2.15.2.5 Foskett Speckled Dace: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC, across the range of ecoregions within the Foskett speckled dace range, was less than the Foskett speckled dace chronic low effect threshold (i.e., EC_5) in all water chemistry scenarios. The chronic Al CCC never exceeded the Foskett speckled dace EC_{10} or

EC_{15} (Table 5-50). As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for Foskett speckled dace based on the chronic criterion.

5.2.16 Vernal Pool Fairy Shrimp (Branchinecta lynchi)

5.2.16.1 Vernal Pool Fairy Shrimp Aluminum Acute Effects Assessment

5.2.16.1.1 Identifying Vernal Pool Fairy Shrimp Acute Aluminum Data

Acute aluminum toxicity data from the EPA's 2018 aluminum aquatic life criteria document were not available for any species within the Order Anostraca (see Appendix A.1). Therefore, a representative acute toxicity value was estimated from the traditional module of EPA's web-based Interspecies Correlation Estimation (web-ICE) program[40]. Web-ICE contained a vernal pool fairy shrimp-specific model, capable of predicting acute Al toxicity to the vernal pool fairly shrimp from as many as 14 surrogate species. Of the 14 surrogate species, only *Lampsilis siliquoidea*, *Daphnia magna*, and *Ceriodaphnia dubia* had quantitatively acceptable SMAVs reported in Appendix A of USEPA (2018) to be used as input data (see Appendix A.1). The *D. magna* (surrogate)-to-*B. lynchi* (predicted) ICE model was determined to be most robust ICE model based on model selection guidance reported by Willming et al. (2016); see Appendix D.1 for ICE model parameters).

The *D. magna* SMAV (2,944 µg/L total aluminum, normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) was input into the *D. magna* (surrogate)-to-*B. lynchi* (predicted) ICE model to determine a vernal pool fairly shrimp SMAV of 2,728 µg/L. Because the input toxicity value (i.e., *D. magna* SMAV) was normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L, the resultant output toxicity value was representative of vernal pool fairy shrimp acute sensitivity to Al in that same water chemistry.

5.2.16.1.2 Deriving Vernal Pool Fairy Shrimp Acute Adjustment Factors

Because no raw empirical acute toxicity data were available for species in the same order as the vernal pool fairy shrimp (Anostraca), no acute order-level TAF could be calculated. As a result, EPA obtained and analyzed C-R data for all acute tests in the (USEPA, 2018) acute aluminum dataset where such data were reported or could be obtained to derive an acute invertebrate TAF or acute MAF, if necessary (i.e., if the vertebrate and invertebrate acute TAFs do not differ from one another). As discussed in Section 5.1.2.1.1, no quantitatively acceptable LC_{50}:LC_5 ratios for vertebrates were available. Thus, the acute invertebrate TAF is the same as the acute MAF. The acute MAF was used to estimate vernal pool fairy shrimp low acute effect concentrations (see Table 5-3 and Appendix B.1). The acute MAF (LC_{50}:LC_5) was 3.881 (Table 5-3) and the corresponding LC_{50}:LC_{10} and LC_{50}:LC_{15} mean adjustment factors are 2.787 and 2.261, respectively (see Appendix C.1 and Appendix C.2).

[40] Web-ICE, operated using the standard Aquatic module; https://www3.epa.gov/webice/

5.2.16.1.3 Calculating Vernal Pool Fairy Shrimp Acute Aluminum Minimum Effect Threshold Values

The MAF (i.e., LC_{50}:LC_5 adjustment factor) as well as the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were applied, as a denominator, to the vernal pool fairy shrimp SMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for the vernal pool fairy shrimp:

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) = **LC_5** = 2,728 µg/L / 3.881 = **702.9 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) = **LC_{10}** = 2,728 µg/L / 2.787 = **978.8 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) = **LC_{15}** = 2,278 µg/L / 2.261 = **1,207 µg/L***

5.2.16.1.4 Vernal Pool Fairy Shrimp: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude and invertebrate species sensitivity across water chemistry, vernal pool fairy shrimp LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate an acute criterion magnitude and corresponding vernal pool fairy shrimp acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and vernal pool fairy shrimp acute effect threshold data (i.e., LC_5, LC_{10}, and LC_{15}) were compared to the specific water quality data that were collected from level-III ecoregions containing the range of the vernal pool fairy shrimp. To visually display the comparison, criterion magnitudes (X-axis) and corresponding vernal pool fairy shrimp acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-54). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, LC_{10}, or LC_{15}) and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and vernal pool fairy shrimp acute effect threshold data (i.e., LC_5, LC_{10}, LC_{15}) are reported in Table 5-51.

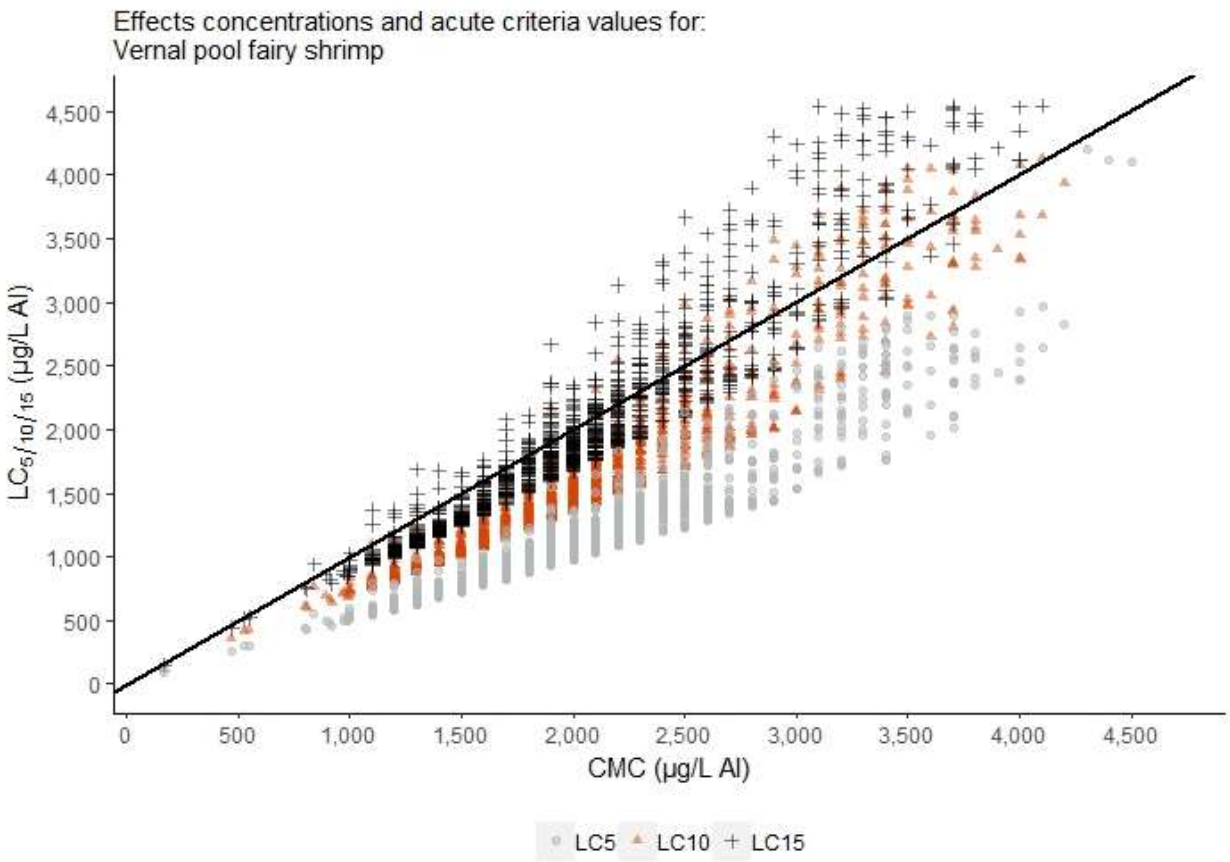

Figure 5-54. Paired vernal pool fairy shrimp acute effect concentrations and acute criterion values in Oregon ecoregions containing the vernal pool fairy shrimp. Note, 0.95% ($n_{removed}$ = 41) of available data points were removed from Figure 5-53 as the Y axis was scaled down (Y axis maximum = 14,422) to aid in data visualization.

Table 5-51. The number and percentage of occurrences where the acute criterion magnitude exceed vernal pool fairy shrimp LC_5, LC_{10}, and LC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing vernal pool fairy shrimp range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (mortality) at each effects concentration.

Species	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}			CMC > LC_{15}		
Name		n	%	AEL	n	%	AEL	n	%	AEL
Vernal pool fairy shrimp	1,438	1,432	99.58	4.80	1,355	94.23	9.42	1,041	72.39	10.86

5.2.16.1.5 Vernal Pool Fairy Shrimp: Tier I Acute Assessment Conclusion

The acute aluminum CMC was greater than the vernal pool fairy shrimp acute low effect threshold (i.e., LC_5) in 99.58% of water chemistry scenarios and greater than the vernal pool fairy shrimp LC_{10} and LC_{15} in 94.23% and 72.39% of exposure conditions (Table 5-51). The LC_5 range was 85 to 8,402 µg/L and the CMC range was 170 to 4,700 µg/L. Because the CMC exceeded the vernal pool fairy shrimp acute low effect threshold, consideration of a refined exposure assessment through a weight of evidence approach was required for vernal pool fairy shrimp based on the acute criterion. Chronic sensitivity of vernal pool fairy shrimp to Al at the chronic criterion magnitude was also assessed before considering vernal pool fairy shrimp in a refined exposure assessment.

5.2.16.2 Vernal Pool Fairy Shrimp Aluminum Chronic Effects Assessment

5.2.16.2.1 Identifying Vernal Pool Fairy Shrimp Chronic Aluminum Data

Chronic aluminum toxicity data were not available for vernal pool fairy shrimp or any other species within the Order Anostraca (see Appendix A.2). As a result, the web-ICE predicted vernal pool fairy shrimp SMAV (see Section 5.2.6.2.1) was transformed to represent a chronic toxicity value (i.e., EC_{20}). Specifically, the vernal pool fairy shrimp SMAV (i.e., LC_{50} = 2,728 µg/L total aluminum, normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) was divided by an acute-to-chronic ratio (ACR).

An ACR is intended to broadly relate a species acute effect concentration to an estimated chronic effect concentration. All acute toxicity tests (Appendix A.1) that had paired chronic toxicity tests (Appendix A.2) conducted using the same species in the same lab are compiled in Table 5-23 (see Section 5.2.6.2.1). The invertebrate-specific ACR (9.089) was based on eight ACRs total, representing two crustacean species (*C. dubia* and *H. azteca*) and a mussel species (*Lampsilis siliquoidea*; Table 2-23). The two crustacean species had calculated ACRs that nearly fell within a factor of 10 of one another. *Ceriodaphnia dubia* had a species-specific ACR of 1.427 and *H. azteca* had a species-specific ACR of >14.11. While the range of ACRs between the two crustaceans was large and may have even exceeded a factor of ten[41], the *L. siliquoidea* had a species-specific ACR of >37.29, which was relatively larger than both crustacean ACRs. (Stephen et al., 1985) advises risk assessors to select appropriate ACRs with caution when the range of ACRs exceeds a factor of ten. Consequently, EPA chose to exclude the *L. siliquoidea* ACR from the invertebrate ACR calculations for the assessment of vernal pool fairy shrimp because; 1) the vernal pool fairy shrimp is a crustacean like *C. dubia* and *H. azteca* and, therefore, expected to respond to toxicant exposures more similarly to these species than a mussel species and 2) the two crustaceans ACRs were reasonably similar to one another relative to the *L. siliquoidea* ACR, based on rank order (i.e., when ranking invertebrate species ACRs from lowest to highest, the two crustacean ACRs ranked next to one another, while the *L. siliquoidea* ACR was the third highest; had the *L. siliquoidea* ACR fallen in-between the two

[41] *Ceriodaphnia dubia* had a species-specific ACR of 1.427 and *H. azteca* had a species specific ACR of >14.11 (see **Table 2-17**). ACRs between *Ceriodaphnia dubia* and *H. azteca* may exceed a factor of ten because the *H. azteca* ACR was expressed as a "greater than" value because the numerator (acute value) of this ACR was based on a test that did not result in ≥50% mortality in the highest concentrations tested (Wang et al., 2016; Wang et al., 2018).

crustacean ACRs, it would have not been excluded from the invertebrate ACR in this assessment). Taken together as a geometric mean, the *Ceriodaphnia dubia* ACR (1.427) and *H. azteca* ACR of (>14.11) results in an ACR of 4.487 that was used to relate to vernal pool fairy shrimp acute (LC_{50}) response to an aluminum chronic (EC_{20}) response.

- *B. lynchi SMAV / Invertebrate ACR*[42] *=* ***EC_{20}*** *= 2,728 µg/L / 4.487 = 608.0* ***µg/L***

The vernal pool fairy shrimp SMCV was 608.0 µg/L total aluminum (normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L) and represents vernal pool fairy shrimp sensitivity to chronic Al exposures.

5.2.16.2.2 Deriving Vernal Pool Fairy Shrimp Chronic Adjustment Factors

Because no raw empirical chronic toxicity data were available for species in the same order as the vernal pool fairy shrimp (Anostraca), no chronic order-level TAF could be calculated. As a result, EPA obtained and analyzed C-R data for all chronic tests in the (USEPA, 2018) chronic aluminum dataset where such data were reported or could be obtained to derive a chronic invertebrate TAF or chronic MAF, if necessary (i.e., if the vertebrate and invertebrate chronic TAFs do not differ from one another). As discussed in Section 5.1.2.1.1, only one quantitatively acceptable EC_{20}:EC_5 ratio for vertebrates was available, not permitting a statistical test to determine if vertebrate and invertebrate TAFs differ. Moreover, no qualitatively acceptable models (up to the order level) exist that could be used to estimate minimum effect concentrations (as was the case of the salmonids). Thus, the chronic MAF was used to estimate vernal pool fairy shrimp chronic low threshold effect concentrations (see Table 5-4. The chronic MAF was 1.403, calculated as the geometric mean of genus-level EC_{20}:EC_5 ratios of 1.168 and 1.685 for fathead minnows and the cladoceran (*Ceriodaphnia dubia*), respectively. The corresponding EC_{20}:EC_{10} and EC_{20}:EC_{15} mean adjustment factors were 1.200 and 1.083, respectively (see Appendix C.3 and Appendix C.4).

5.2.16.2.3 Calculating Vernal Pool Fairy Shrimp Chronic Aluminum Minimum Effect Threshold

The chronic MAF (i.e., EC_{20}:EC_5 adjustment factor) as well as the EC_{20}:EC_{10} and EC_{20}:EC_{15} adjustment factors were applied, as a denominator, to the vernal pool fairy shrimp SMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the vernal pool fairy shrimp:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) =* ***EC_5*** *= 608.0 µg/L / 1.403 =* ***433.4 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) =* ***EC_{10}*** *= 608.0 µg/L / 1.200 =* ***506.7 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) =* ***EC_{15}*** *= 608.0 µg/L / 1.083 =* ***561.3 µg/L***

5.2.16.2.4 Vernal Pool Fairy Shrimp: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and invertebrate species sensitivity across water chemistry, vernal pool fairy shrimp chronic EC_5, EC_{10}, and EC_{15} values

[42] Calculated excluding the *L. siliquoidea* ACR.

were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding vernal pool fairy shrimp chronic low effect threshold values (i.e., EC_5, EC_{10}, EC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and vernal pool fairy shrimp chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were compared to water quality data that were collected from level-III ecoregions containing the range of the vernal pool fairy shrimp. To visually display the comparison, criterion magnitudes (X-axis) and corresponding vernal pool fairy shrimp chronic effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-55). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and vernal pool fairy shrimp chronic effect threshold data (i.e., EC_5, EC_{10}, EC_{15}) are reported in Table 5-52.

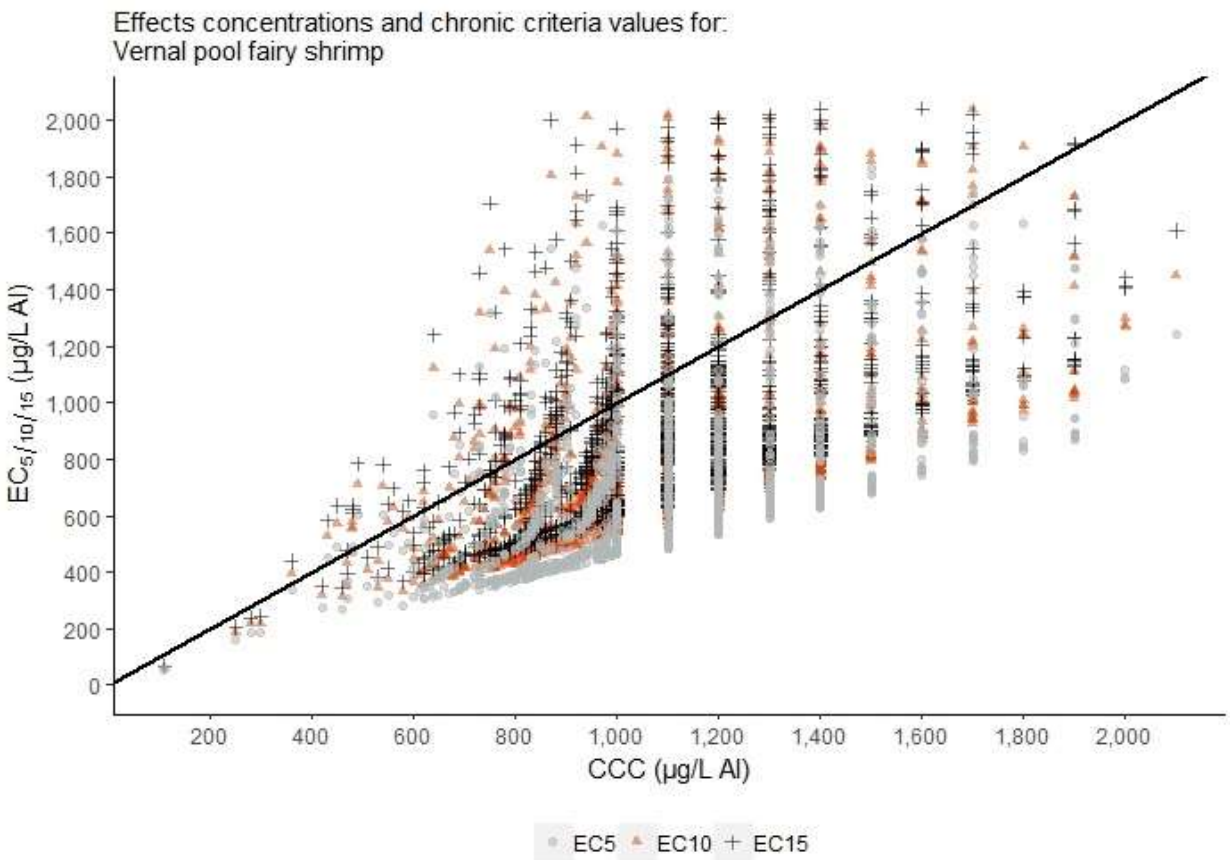

Figure 5-55. Paired vernal pool fairy shrimp chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the vernal pool fairy shrimp. Note, 1.53% ($n_{removed}$ = 66) of available data points were removed from Figure 5-54 as the Y axis was scaled down (Y axis maximum = 6,708) to aid in data visualization.

Table 5-52. The number and percentage of occurrences where the chronic criterion magnitude exceed vernal pool fairy shrimp EC_5, EC_{10}, and EC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing vernal pool fairy shrimp range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (growth/biomass) at each effects concentration.

Species	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}			CCC > EC_{15}		
Name		n	%	AEL	n	%	AEL	n	%	AEL
Vernal pool fairy shrimp	1,438	1,298	90.26	4.51	1,187	82.55	8.26	1,089	75.73	11.36

5.2.16.2.5 Vernal Pool Fairy Shrimp: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC was greater than the vernal pool fairy shrimp acute low effect threshold (i.e., EC_5) in 90.26% of water chemistry scenarios and greater than the vernal pool fairy shrimp EC_{10} and EC_{15} in 82.55% and 75.73% of exposure conditions, respectively (Table 5-52). Because the CCC exceeded the vernal pool fairy shrimp chronic low effect threshold, consideration, a refined exposure assessment through a weight of evidence approach was required for vernal pool fairy shrimp based on the chronic criterion. Moreover, sensitivity of vernal pool fairy shrimp to acute Al exposures (see Section 5.2.16.1.5) also indicated a refined exposure assessment should have been considered through a weight of evidence approach to inform a final effect determination.

5.2.17 Oregon Spotted Frog (Rana pretiosa)

5.2.17.1 Oregon Spotted Frog Acute Aluminum Effects Assessment

5.2.17.1.1 Identifying Oregon Spotted Frog Acute Aluminum Data

Species-level acute toxicity data were not available for Oregon spotted frog or any other members of the Family Ranidae. Acute toxicity data were available for members of the Order Anura (see Appendix A.1). The Anura OMAV is based on a single green tree frog (*Hyla cinerea*; life stage = <1 dph tadpole) toxicity test that produced a non-definitive LC_{50} value of >18,563 µg/L total aluminum (normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L; Jung and Jagoe 1995).

5.2.17.1.2 Deriving Oregon Spotted Frog Acute Adjustment Factors

The green tree frog acute toxicity test (Jung and Jagoe 1995) did not report concentration-response data. No quantitatively acceptable acute TAFs were available for Oregon spotted frog or any other vertebrate species (see Table 5-4). Therefore, all genus-level TAFs (based on quantitatively acceptable C-R curves) were averaged (geometric mean; n = 2) to calculate the

acute MAF, which serves as a generic, non-taxa-specific adjustment factor meant to broadly relate acute LC_{50} values to lower level effects concentrations (i.e., LC_5). Because no quantitatively acceptable TAFs were available for vertebrate species, the acute MAF is based on invertebrate species only (see Table 5-4). Unlike, salmonids no qualitatively acceptable acute C-R models for species in the Order Anura were available to be considered towards the development of a TAF based on qualitatively acceptable data (see Table 5-8 and Appendix B.1).

The acute MAF was calculated as the geometric mean of the *Physa* genus-level TAF (i.e., 3.597) and the *Ceriodaphnia* genus-level TAF (i.e., 4.189) and is 3.881 (Table 5-3). The acute C-R curves used to derive the acute MAF (i.e., LC_{50}:LC_5) were also used to calculate LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors of 2.787 and 2.261, respectively (see Appendix C.1 and Appendix C.2). Similar to the acute MAF (i.e., LC_{50}:LC_5), the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were intended to provide a mechanism for relating acute effect concentrations (i.e., LC_{50}) to lower-level effect concentrations (i.e., LC_{10} and LC_{15} values, respectively) across a broad range of taxa.

5.2.17.1.3 Calculating Oregon Spotted Frog Acute Aluminum Minimum Effect Threshold Values

The acute MAF (i.e., LC_{50}:LC_5 adjustment factor) as well as the LC_{50}:LC_{10} and LC_{50}:LC_{15} adjustment factors were applied, as a denominator, to the Anura OMAV to calculate an acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value representative of the Oregon spotted frog:

- *LC_{50}/(LC_{50}:LC_5 adjustment factor) = **LC_5** = 18,563 µg/L / 3.881 =* ***4,782 µg/L***
- *LC_{50}/(LC_{50}:LC_{10} adjustment factor) = **LC_{10}** = 18,563 µg/L / 2.787 =* ***6,660 µg/L***
- *LC_{50}/(LC_{50}:LC_{15} adjustment factor) = **LC_{15}** = 18,563 µg/L / 2.261 =* ***8,210 µg/L***

5.2.17.1.4 Oregon Spotted Frog: Assessing Acute Responses Across Oregon Water Chemistries

To account for the changes in the acute Al criterion magnitude and vertebrate species sensitivity across water chemistry, Oregon spotted frog acute LC_5, LC_{10}, and LC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate an acute criterion magnitude and corresponding Oregon spotted frog acute low effect threshold (i.e., LC_5), LC_{10} value, and LC_{15} value for each set of paired water chemistry data (n = 19,274).

Paired acute criterion magnitudes and Oregon spotted frog acute effect threshold data (i.e., LC_5, LC_{10}, and LC_{15}) were compared to the specific water quality data that were collected from level-III ecoregions containing the range of the Oregon spotted frog. To visually display the comparison, criterion magnitudes (X-axis) and corresponding Oregon spotted frog acute effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-56). Data points above the linear line indicated scenarios where the CMC was less than the corresponding acute effect threshold data (i.e., LC_5, LC_{10}, or LC_{15}) and data points below the 1:1 line indicated scenarios where the CMC was greater than the corresponding acute effect threshold data. Direct numeric comparisons between the acute criterion magnitudes and Oregon spotted frog acute effect threshold data (i.e., LC_5, LC_{10}, and LC_{15}) are reported in Table 5-53.

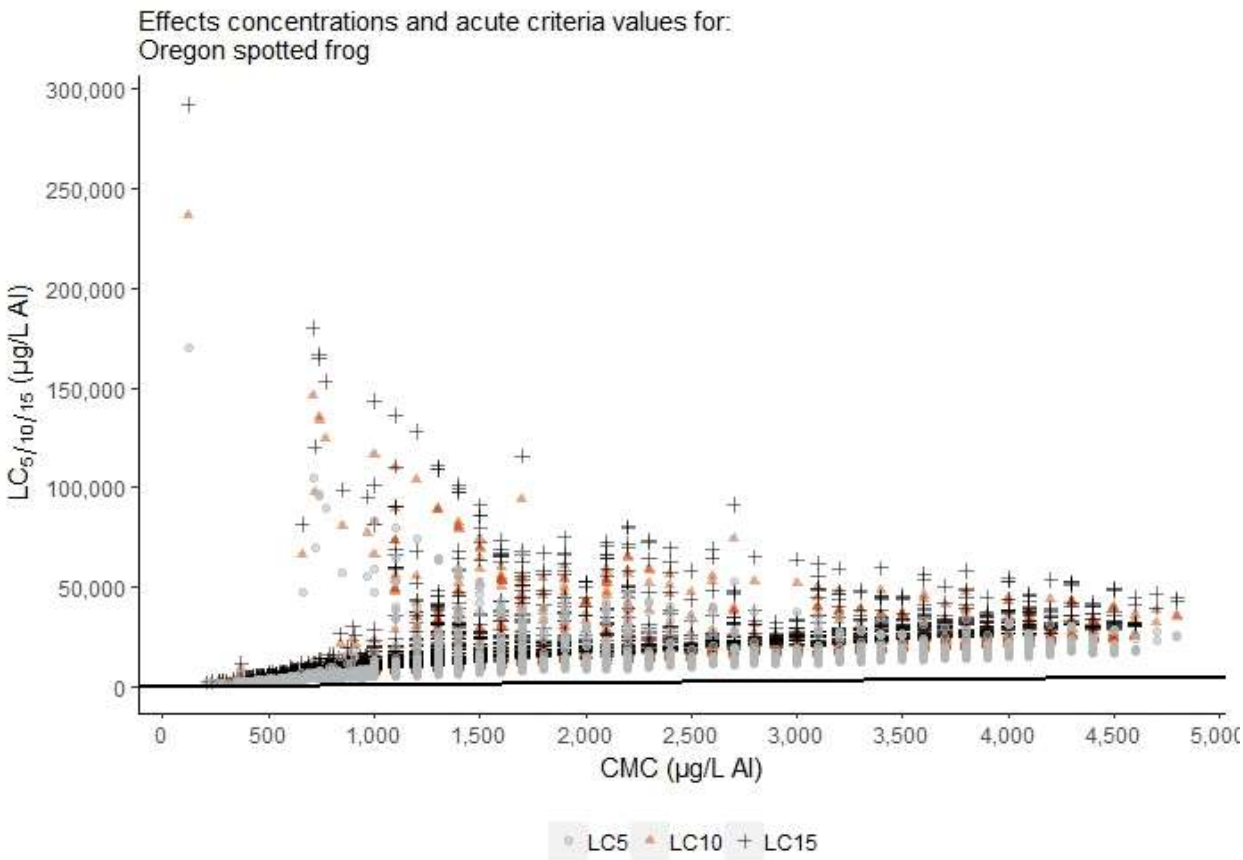

Figure 5-56. Paired Oregon spotted frog acute effect concentrations and acute criterion values in Oregon ecoregions containing the Oregon spotted frog.

Table 5-53. The number and percentage of occurrences where the acute criterion magnitude exceed Oregon spotted frog LC_5, LC_{10}, and LC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing Oregon spotted frog range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (mortality) at each effects concentration. AEL for each LCx value is 0%.

Species	Paired Hardness, pH, DOC Samples (n)	CMC > LC_5			CMC > LC_{10}		CMC > LC_{15}	
Name		n	%	AEL	n	%	n	%
Oregon spotted frog	2,880	0	0	0	0	0	0	0

5.2.17.1.5 Oregon Spotted Frog: Tier I Acute Assessment Conclusion

The acute aluminum CMC was less than the Oregon spotted frog acute low effect threshold (i.e., LC_5) in all water chemistry scenarios in Oregon ecoregions within the Oregon spotted frog range (Table 5-51). The LC_5 range was 1,385 to 170,182 µg/L and the CMC range was 120 to 4,800

µg/L. As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for Oregon spotted frog based on the acute criterion.

5.2.17.2 Oregon Spotted Frog Chronic Aluminum Effects Assessment

5.2.17.2.1 Identifying Oregon Spotted Frog Chronic Aluminum Data

Species-level chronic toxicity data were not available for Oregon spotted frog or any other members of the Genus *Rana*. One chronic aluminum test with the wood frog (*Lithobates sylvatica*) was available to serve as the Ranidae FMCV. No other chronic toxicity tests were available for species within in the Family Ranidae. The wood frog (*Lithobates sylvatica*) test produced a non-definitive maximum acceptable toxicant concentration (MATC) of >10,684 µg/L total aluminum (normalized to pH 7, hardness of 100 mg/L as $CaCO_3$ and DOC of 1 mg/L; Peles 2013; endpoint = complete metamorphosis; duration = 43-102 days; life stage = larva, Gosner stage 25) (see Appendix A.2).

An MATC differs from a point estimate (e.g., EC_{20}) as an MATC is the geometric mean of the lowest effect and no effect treatment concentrations used in the test. In criteria derivation and effects assessments, EPA relies on point estimates to the greatest extent possible; however, when data are limited, results from hypothesis-based testing can be used to inform conclusions in a qualitative manner, particularly for tolerant species. For example, the wood frog chronic toxicity test reported by (Peles, 2013) was used qualitatively in the 2018 aluminum aquatic life document to fulfill an otherwise missing minimum data requirement (USEPA, 2018). Although the MATC introduced uncertainty into this assessment, EPA considered it adequately representative of an EC_{20} since the effect concentration indicated that harm to the species would be minimal because the MATC strongly indicated the wood frog (and via surrogacy, the Oregon spotted frog) were a relatively tolerant species.

5.2.17.2.2 Deriving Oregon Spotted Frog Chronic Adjustment Factors

(Peles, 2013) did not contain adequate information for C-R modeling and no other chronic toxicity tests with members of the Order Anura were available to calculate a chronic order-level TAF. As a result, EPA obtained and analyzed C-R data for all chronic tests in the (USEPA, 2018) chronic aluminum dataset where such data were reported or could be obtained to derive a chronic vertebrate TAF or chronic MAF, if necessary (i.e., if the vertebrate and invertebrate chronic TAFs do not differ from one another). As discussed in Section 5.1.2.1.1, only one quantitatively acceptable EC_{20}:EC_5 ratio for vertebrates was available, not permitting a statistical test to determine if vertebrate and invertebrate TAFs differ. Moreover, no qualitatively acceptable models (up to the order level) exist that can be used to estimate minimum effect concentrations (as in the case of the salmonids). Thus, the chronic MAF was used to estimate Oregon spotted frog chronic low effect concentrations (see Table 5-4). The chronic MAF was 1.403, calculated as the geometric mean of genus-level EC_{20}:EC_5 ratios of 1.168 and 1.685 for fathead minnow and a cladoceran (*Ceriodaphnia dubia*), respectively. The chronic MAF (1.403) was more conservative than the chronic fathead minnow species-level TAF (1.168, also serves as the vertebrate TAF since no other vertebrate TAFs were available; see Table 5-4 and Appendix B.2). The corresponding EC_{20}:EC_{10} and EC_{20}:EC_{15} mean adjustment factors were 1.200 and 1.083, respectively (see Appendix C.3 and Appendix C.4).

5.2.17.2.3 Calculating Oregon Spotted Frog Chronic Aluminum Minimum Effect Threshold

The chronic MAF (i.e., EC_{20}:EC_5 adjustment factor) as well as the EC_{20}:EC_{10} and EC_{20}:EC_{15} adjustment factors were applied, as a denominator, to the Ranidae FMCV to calculate a chronic low effect threshold (i.e., EC_5), EC_{10} value, and EC_{15} value representative of the green sturgeon:

- *EC_{20}/(EC_{20}:EC_5 adjustment factor) = **EC_5** = 10,684 µg/L / 1.403 =* ***7,615 µg/L***
- *EC_{20}/(EC_{20}:EC_{10} adjustment factor) = **EC_{10}** = 10,684 µg/L / 1.200 =* ***8,904 µg/L***
- *EC_{20}/(EC_{20}:EC_{15} adjustment factor) = **EC_{15}** = 10,684 µg/L / 1.083 =* ***9,862 µg/L***

5.2.17.2.4 Oregon Spotted Frog: Assessing Chronic Responses Across Oregon Water Chemistries

To account for the changes in the chronic Al criterion magnitude and vertebrate species sensitivity across water chemistry, Oregon spotted frog chronic EC_5, EC_{10}, and EC_{15} values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time to calculate a chronic criterion magnitude and corresponding Oregon spotted frog chronic low effect threshold values (i.e., EC_5, EC_{10}, EC_{15}) for each set of paired water chemistry data (n = 19,274).

Paired chronic criterion magnitudes and Oregon spotted frog chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) were compared to water quality data that were collected from level-III ecoregions containing the range of the Oregon spotted frog. To visually display the comparison, criterion magnitudes (X-axis) and corresponding Oregon spotted frog chronic effect concentrations (Y-axis) were plotted over a linear (i.e., 1:1) line (see Figure 5-57). Data points above the linear line indicated scenarios where the CCC was less than the corresponding chronic effect threshold data (i.e., EC_5, EC_{10}, or EC_{15}) and data points below the 1:1 line indicated scenarios where the CCC was greater than the corresponding chronic effect threshold data. Direct numeric comparisons between the chronic criterion magnitudes and Oregon spotted frog chronic effect threshold data (i.e., EC_5, EC_{10}, and EC_{15}) are reported in Table 5-54.

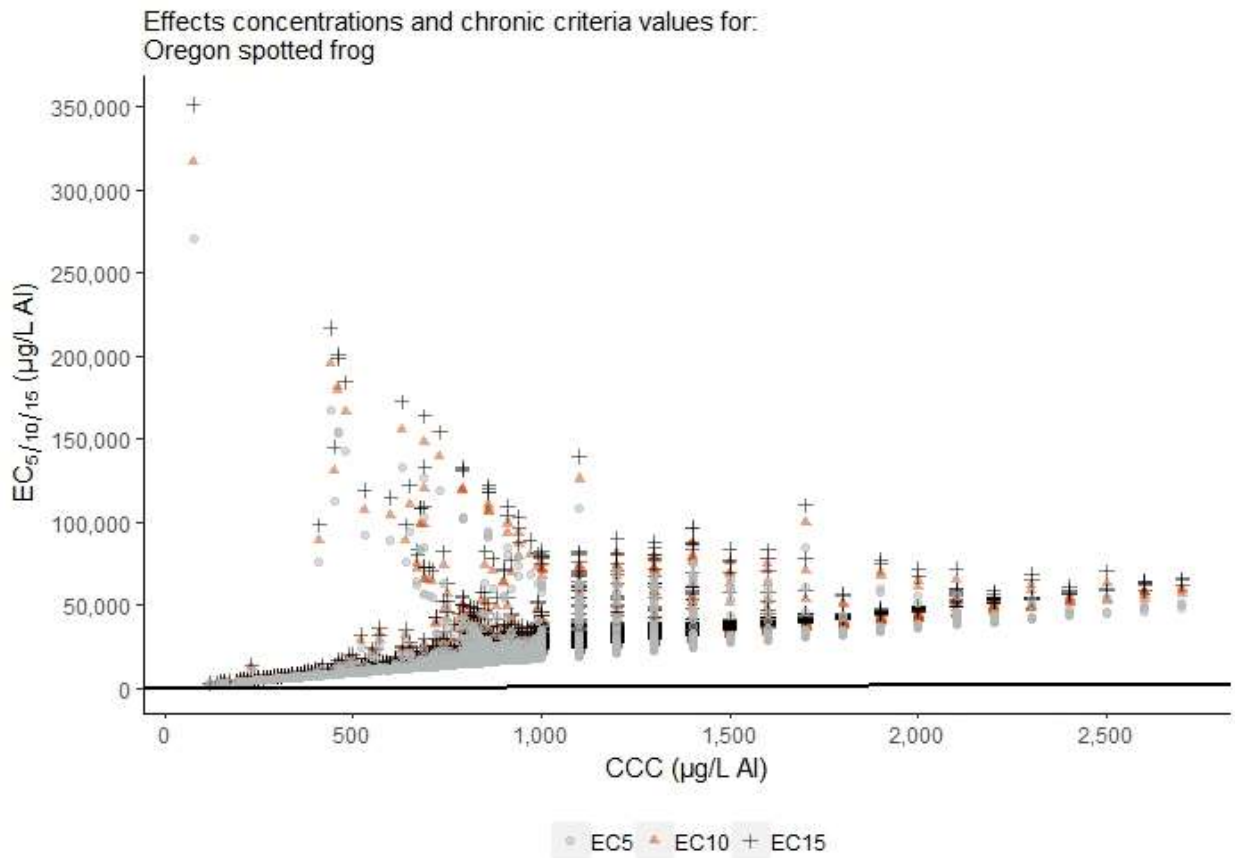

Figure 5-57. Paired Oregon spotted frog chronic effect concentrations and chronic criterion values in Oregon ecoregions containing the Oregon spotted frog.

Table 5-54. The number and percentage of occurrences where the chronic criterion magnitude exceed Oregon spotted frog EC_5, EC_{10}, and EC_{15} values. Comparisons are based only on paired pH, dissolved organic carbon, and hardness data from level-III ecoregions containing Oregon spotted frog range. Aggregate Effect Levels (AEL) are also shown, defined as the percent of an exposed population that would experience adverse effects (growth/biomass) at each effects concentration. AEL for each ECx value is 0%.

Species	Paired Hardness, pH, DOC Samples (n)	CCC > EC_5			CCC > EC_{10}		CCC > EC_{15}	
Name		n	%	AEL	n	%	n	%
Oregon spotted frog	2,880	0	0	0	0	0	0	0

5.2.17.2.5 Oregon Spotted Frog: Tier I Chronic Assessment Conclusion

The chronic aluminum CCC, across the range of ecoregions within the Oregon spotted frog range, was less than the Oregon spotted frog chronic low effect threshold (i.e., EC_5) in all water chemistry scenarios. The chronic Al CCC never exceeded the Oregon spotted frog EC_{10} or EC_{15}

(Table 5-54). As a result, consideration of a refined exposure assessment through a weight of evidence approach was not required for Oregon spotted frog based on the chronic criterion.

5.3 Refined Exposure Assessment: LOE 2 Assessment

5.3.1 LOE 2: Refined Exposure Assessment

As described in the weight of evidence methodology overview (Section 5.1), LOE 2 deals with the exposure potential of a species to Al in locations where the Al criteria may be used. Potential uses include derivation of protective effluent permit limits, assessments for impaired waters, total maximum daily load determinations, and site cleanup activities. The results of LOE 2 may help the Services determine the likelihood that T/E species would experience exposures to Al in circumstances that would be subject to the proposed Al water quality criteria. In Section 4 (Environmental Baseline) of this BE, EPA reviewed the factors associated with potential Al releases into the environment and how these activities overlapped with locations where T/E species may occur. Through this process, EPA gained a better understanding of the exposure potential to Al. The analyses presented herein are intended to better contextualize the analysis conducted for LOE 1. In other words, EPA addresses the question: in what locations in Oregon might species exposures to Al more likely reach criteria concentration levels? As this analysis is subject to a fair amount of uncertainty, EPA does not solely rely on conclusions made in LOE 2 for its final effects determinations. Given the uncertainty of the available information and how it applies to EPA's proposed action and given that the proposed action does not include implementation activities, EPA assigned a relative weighting of evidence of 0.10 to LOE 2 in its final weighing of the evidence in Section 6. Calculations in Excel can be found in Attachment 1, LOE 2 Calculations.

5.3.1.1 Impaired Waters Analysis to Determine the Land Use Score for Each Species

EPA conducted an analysis to determine the potential for a water in Oregon to be identified as impaired as a result of exceeding the proposed aluminum criteria. This analysis relies on two documents published in the Federal Rule docket (USEPA, 2019b).

- Economic Analysis[43]
- Default Ecoregional Aluminum Criteria Values[44] (Table 5-55).

Data sources used for this analysis included:

- Criteria input values (pH, DOC, hardness), as described in Section 4 of this BE as well as EPA-HQ-OW-2016-0694-0114[2].
- Aluminum (Al) data acquired from ODEQ and USGS with sampling dates that spanned August 2000 to May 2017. Al data were acquired as dissolved or total recoverable, but

[43] Economic Analysis For The Proposed Rule: Aquatic Life Criteria For Aluminum In Oregon. April 2019. EPA-HQ-OW-2016-0694-0125.

[44] Analysis of the Protectiveness of Default Ecoregional Aluminum Criteria Values. April 2019. EPA-HQ-OW-2016-0694-0114.

only total recoverable concentrations were used because the Al criteria are based on this fraction and because no consistent dissolved to total Al translator (ranged from approximately 1 to >1000) could be identified in the dataset. Some monitoring stations had multiple Al samples reported in one day. In those cases, EPA took the highest available concentration for that day and used that concentration as a single observation.

To determine potential impairments, ambient aluminum concentrations were compared to suggested ecoregional default chronic criteria values calculated as the 10th centile of criteria values within an ecoregion. This procedure was done instead of using criteria values calculated on the day of the Al sample because most Al samples did not also have associated water chemistry data for the criteria calculation. Samples in which Al exceeded the default criteria value on two or more occasions were determined to indicate a potential impairment per ODEQ's 303(d) listing policy. All other samples were determined to indicate no potential impairment. The results of the analysis can be found in Figure 5-58 as well as Table 5-55. Note that the "impairment uncertain" category is not used in this BE but is shown in Figure 5-58.

Table 5-55. Suggested ecoregional default criteria values for Oregon as presented in (USEPA, 2019b).

Ecoregion	CCC	CMC
Coast Range	350	680
Willamette Valley	440	870
Cascades	350	600
East Cascades	600	1,100
Columbia Plateau	840	1,400
Blue Mountains	780	1,300
Snake River Plain	1,200	3,000
Klamath Mountains	780	1,300
Northern Basin and Range	790	1,400

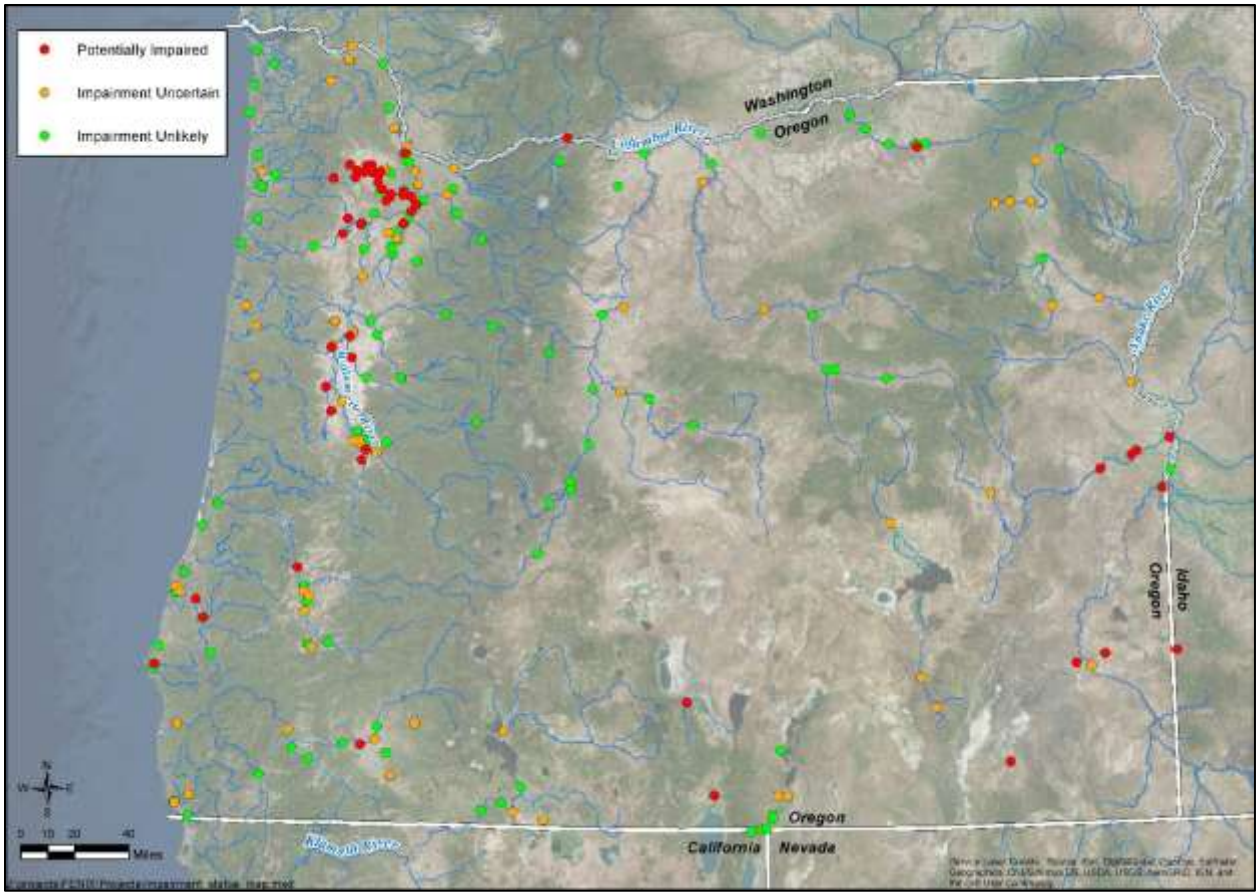

Figure 5-58. Potential Al impairments by Al sampling location. Note that EPA did not consider the *impairment uncertain* condition for this BE.

Table 5-56. Summary of potential and unlikely Al impairments by ecoregion.

Ecoregion1			
Code	**Name**	**Potentially Impaired**	**Impairment Unlikely**
1	Coast Range	3	31
3	Willamette Valley	36	48
4	Cascades	0	5
9	Eastern Cascade Slopes	3	12
10	Columbia Plateau	1	10
11	Blue Mountains	0	14
12	Snake River Plain	5	1
78	Klamath Mountains	2	13
80	Northern Basin & Range	4	6
Total		**54**	**140**

1, Ecoregions correspond to Level III Ecoregions of the Continental United States from the EPA

EPA used the ecoregion-based results to estimate effects of the action (i.e., to determine the *impairment score* used to calculate the LOE 2 Score) on listed entities in the following manner:

1. The number of potential impairments as in Table 5-55. were used to calculate an impairment score that was scaled to fall between 0 and 1. Scaling was based on the Willamette Valley's 36 potential impairments. Thus, the Willamette Valley scored a 1.0 while the Snake River Plain scored a 0.14 (5÷36) based on its 5 potential impairments (see Table 5-55), and so on for each ecoregion.
2. Species range was mapped as in Section 4 and as shown in Table 4-8 in order to determine in which ecoregions the species may occur.
3. The impairment score for each ecoregion in which a species may occur was averaged to determine a species-specific impairment score.

The results of this process can be found in Table 5-56.

5.3.2 Land Use Analysis to Determine the Land Use Score for Each Species

EPA next conducted an analysis based on the land use analysis described in Section 4. This analysis was used to determine the potential for non-point sources of Al to be released into waters in which a species may occur. EPA's assumption was that the higher the potential for Al releases, the more likely the Al criteria may be used to assess the potential for Al toxicity at a site. In such locations, it is important to understand whether achieving Al concentrations compliant with the Al criteria would be protective for listed entities. In other words, these would be locations where the Al criteria may more likely be used. Locations with a lower potential for non-point Al sources may be of lower concern for Al toxicity to a listed species. The process of determining the Land Use Score was as follows:

1. The Land Use Score for an ecoregion was determined by
    a. Focusing on the land use that was significantly ($P<0.05$) related (Pearson r correlation coefficient) to Al concentrations in an ecoregion. Only agriculture and forestry were considered because these are the only two land uses which were statistically significant across the state. EPA used this criterion because of the small number of samples within each ecoregion. See Tables 5-56 and 5-57.
    b. If the correlation between percent forestry or percent agriculture and Al concentration was significant in an ecoregion, the percent of each land use was identified.
    c. The land use score was determined to be either 0, 1, or 2 depending on if the statistically significant land use was 0%, 1-50%, or 51-100% within an ecoregion, respectively (See Table 5-56 for percentages by ecoregion).
2. Species range was mapped as in Section 4 and as shown in Table 4-7 in order to determine in which ecoregions the species may occur.
3. The land use score for each ecoregion in which a species may occur was averaged to determine a species-specific land use score.

The results of this process can be found in Table 5-57.

5.3.3 Potential Discharger Analysis to Determine the Discharger Score for Each Species

EPA used information compiled in Section 4 for this piece of the effects analysis. As described in Section 4, the potential for a facility to discharge Al was based on processes that may include Al. EPA notes that the process of identifying a potential facility that would release Al is speculative because there was a lack of facility specific data. Additionally, the potential to discharge Al does not in any way suggest a reasonable potential to exceed proposed Oregon aluminum water quality criteria.

The facilities with the potential to release Al into surface waters included:

- Anodizing
- Municipal waste water treatment plants
- Industrial
- Drinking water

The information gained from this component of the effects analysis was used to understand the potential for the Al criteria to be used and thus be relevant to a listed entity. An underlying assumption was that with a higher number of potential Al discharging facilities, a species may be more likely exposed to Al at concentrations approaching the Al criteria values. That is, a higher number of potential discharges in locations in which species may occur was considered a risk factor for Al exposure for that species. Therefore, EPA determine a Discharger Score as follows:

1. The number of potential dischargers in an ecoregion
2. The discharger score for each ecoregion in which a species may occur was averaged to determine a species-specific discharger score

The results of this analysis can be found in Table 5-58

Although not a formal part of the effects analysis in Chapter 5, please note that EPA has provided a hypothetical Discharger Assessment in Appendix F to better understand the frequency above which an in-stream Al concentration may exceed a species low effects threshold (e.g., EC05). That is, a hypothetical discharger assessment was completed to fully analyze how the aluminum aquatic life criteria (USEPA, 2018) might be incorporated into a regulated discharger's permit through the National Pollutant Discharge Elimination System (NPDES), where a facility discharge is found to have reasonable potential to cause or contribute to an excursion above the aluminum criteria, and how that permitting process may affect a listed species. The assessment is considered highly speculative because EPA lacked site-specific data and made calculation assumptions that may not represent choices made by Oregon DEQ. However, the hypothetical Discharger Assessment can be considered illustrative of what may occur in Oregon if a facility requires Al effluent limits.

5.3.4 Determining an LOE 2 Score for Each Species

EPA summed the impairment score, land use score and discharger scores for an overall LOE 2 Score from 0 to 4 as shown in Table 5-57 for ecoregions and 5-58 for species. In Section 5.1, the species LOE 2 Score was multiplied by the weighting factor as part of the WOE Score calculation for that species.

Table 5-57. Determination of Impairment Score, Land Use Score and Discharger Score for LOE 2 Score calculation on an ecoregional basis.

Ecoregion	Ecoregion Code	Potential impairments	Impairment Score[1]	Land Use Score[2]	Comment	Potential Dischargers	Discharger Score[3]	LOE 2 Score[4]
Blue Mtns	11	0	**0.00**	**0**		1	**0.03**	0.03
Cascades	4	0	**0.00**	**0**		3	**0.09**	0.09
Coast Range	1	3	**0.08**	**2**	88% forest	15	**0.47**	2.55
Columbia Plateau	10	1	**0.03**	**0**	3% developed	3	**0.09**	0.12
Eastern Cascades	9	3	**0.08**	**0**	0.90% barren	1	**0.03**	0.11
Klamath Mtns	78	2	**0.06**	**0**		19	**0.59**	0.65
Northern Basin	80	4	**0.11**	**2**	91% forest	0	**0.00**	2.11
Snake River	12	5	**0.14**	**1**	32% Ag	1	**0.03**	1.17
Willamette Valley	3	36	**1.00**	**1**	43% Ag	32	**1.00**	3.00

Notes:

1, Potential impairments found when total recoverable aluminum concentration > default ecoregional criteria on two or more occasions at the same location. Impairment score was scaled to the ecoregion with the highest number of potential impairments (Willamette Valley). Scores ranged from 0-1.

2, Land use score was 0 when no land use type correlated with Al concentrations in an ecoregion; 1 if a correlation was present and the land use percentage was 1-50%; 2 if a correlation was present and the land use percentage was 51-100%.

3, Discharger score was the number of potential Al dischargers in an ecoregion scaled to the ecoregion with the highest number of potential Al discharging facilities (Willamette Valley). Scores ranged from 0-1.

4, LOE 2 Score was the sum of each of the other scores and could range from 0-4.

Table 5-58. Determination of an LOE 2 Score for each species. The calculation of each score and the LOE 2 Score is described in the main body of the text.

Species	ESU/DPS	Ecoregion Code[1]	Impairment Score (0-1)	Land Use Score (0-2)	Discharger Score (0-1)	LOE Score (0-4)
Green sturgeon		1	0.08	2.00	1.00	3
Eulachon		1,3,4,9, CR	0.29	0.75	0.40	1
Coho	Lower Columbia River	1,3,4,9,CR	0.29	0.75	0.40	1
	Oregon Coast	1,78	0.07	1.00	0.53	2
	Southern Oregon/Northern California Coast	1,78,4	0.05	0.67	0.39	1
Chum	Columbia River	1,3,4,9,10,CR	0.24	0.60	0.34	1
Chinook	Upper Columbia River spring	1,3,4,9,10,CR	0.24	0.60	0.34	1
	Snake River spring/summer	1,3,4,9,10,11,CR	0.20	0.50	0.29	1
	Snake River fall	1,3,4,9,10,11,CR	0.20	0.50	0.29	1
	Upper Willamette River	1,3,4,CR	0.36	1.00	0.52	2
	Lower Columbia	1,3,4,9,CR	0.29	0.75	0.40	1
Sockeye	Snake River	1,3,4,9,10,11,CR	0.22	0.83	0.27	1
Steelhead	Upper Columbia	1,3,4,9.10,CR	0.24	0.60	0.34	1
	Snake River Basin	1,3,4,9,10,11,CR	0.20	0.50	0.29	1
	Middle Columbia	1,3,4,9,10,11,CR	0.20	0.50	0.29	1
	Upper Willamette River	1,3,4,CR	0.36	1.00	0.52	2
	Lower Columbia	1,3,4,9,CR	0.29	0.75	0.40	1
Oregon spotted frog		4,9	0.04	0.00	0.06	0
Warner Sucker		80	0.11	2.00	0.00	2
Shortnose Sucker		9	0.08	0.00	0.03	0
Lost River Sucker		9	0.08	0.00	0.03	0
Hutton Tui Chub		9,80	0.10	1.00	0.02	1
Borax Lake Chub		80	0.11	2.00	0.00	2
Lahontan cutthroat trout		80	0.11	2.00	0.00	2

Species	ESU/DPS	Ecoregion Code[1]	Impairment Score (0-1)	Land Use Score (0-2)	Discharger Score (0-1)	LOE Score (0-4)
Foskett speckled dace		80	0.11	2.00	0.00	2
Bull trout		1,3,4,9,10,11,12,80,CR	0.18	0.75	0.22	1
Vernal Pool Fairy Shrimp		78	0.06	0.00	0.59	1

Notes:
1, Ecoregion codes are:
1 Coast Range
3 Willamette Valley
4 Cascades
9 Eastern Cascade Slopes
10 Columbia Plateau
11 Blue Mountains
12 Snake River Plain
78 Klamath Mountains
80 Northern Basin & Range
CR Columbia River (not an ecoregion)

5.4 Hypothetical Discharger Assessments: See Appendix F

5.5 Indirect Effects: Tier 2/Line of Evidence 3 Assessment

Following assessment of direct biological effects, EPA considered and assessed potential indirect biological effects of the water quality standard action on listed aquatic organisms in Oregon. To assess potential indirect effects, EPA considered potential effects to listed species diets and feeding behaviors. For each listed species a chronic aluminum species sensitivity distribution (SSD) was created that only contained prey items, or reasonable surrogate prey items, for the species of concern (e.g., *Lampsilis* was a surrogate for other mollusks). Specifically, EPA removed non-prey items from the 2018 aluminum chronic toxicity sensitivity distribution and calculated a water chemistry specific HC_5 value (representing 20% chronic effects to the 5th centile of sensitive genera under long-term exposure scenarios) from the listed species-prey item distribution (i.e., listed species-prey-item-specific SSD) in each of the 19,274 water chemistry samples where pH, DOC, and hardness were collected concurrently. Therefore, for each listed species the genus mean chronic values (GMCVs) comprising their prey-item-specific SSD were all renormalized for each water chemistry scenario and ranked based on relative genera sensitivity to the specific water chemistry to create a unique listed 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).

Individual listed species prey item-by-water-chemistry-specific HC_5 values were then compared to corresponding chronic criterion magnitudes to determine the number and percentage of time prey-item-specific HC_5 values were less than criteria. Comparisons between prey-item specific HC_5 values and chronic criterion magnitudes were only based on water chemistry collected from Level-III Ecoregions, within Oregon, where the particular listed species occurred. This is consistent with the approach outlined in Section 5.1.2 for assessing direct biological acute and chronic effect concentrations across Oregon water chemistries.

For many of the listed species assessed, prey-item-specific HC_5 values were less than corresponding criterion concentrations in a subset of Oregon water chemistries, which may initially be counter-intuitive since the HC_5 values were calculated from the same SSD as the chronic criterion with non-prey item genera removed. That is, no GMCVs were added to the prey-item specific SSDs that were more sensitive than the GMCVs in the chronic criterion SSD; however, there were many instances where HC_5 values were less than the corresponding criterion magnitude even though both were based on the same underlying data and calculation procedure to extrapolate an HC_5 value (US EPA 1985). Therefore, removing non-prey-item GMCVs reduced the "n" used to calculate the HC_5, which in turn, reduced the HC_5 value relative to the chronic criterion that was calculated with a greater "n."[45] Results of the prey-item-specific HC_5

[45] To demonstrate the effect of reducing the "n" in determining the HC_5, EPA reran the 304(a) chronic aquatic life criteria calculator (which uses an "n" of 13) for all 19,274 water chemistry scenarios in Oregon along with a revised calculator that forced the chronic criterion calculator to assume only 12 Genus Mean Chronic Values (GMCV) were in the chronic Species Sensitivity Distribution (SSD). The criteria outputs were compared between the chronic magnitude calculated assuming n = 12 and n = 13. The 304(a) CCC (i.e., n = 13) was greater than the revised CCC (assume n = 12) in 12,605 instances out of 19,274 and equaled the 304(a) CCC (i.e., n = 13) in the remaining 6,669 water chemistry scenarios.

comparisons to corresponding criterion magnitudes were influenced primarily by the "n" in the prey-item-specific SSD, rather than prey item sensitivity itself. Therefore, the most sensitive GMCV from each prey-item specific HC_5 calculated was also compared to the corresponding criterion magnitude to determine if the most sensitive prey-item specific GMCV was ever less than the chronic criterion in all waters assessed. 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 "n".

In some cases, the analysis included consideration of other factors such as whether a species exhibits generalized versus specific feeding behaviors, if prey items reflect K or r life history strategies, and qualitative considerations of relative taxa sensitivity for species that relied on plant-based food resources. Furthermore, the indirect effects analysis focused on chronic effects to listed species prey items because most listed species prey items 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. Lastly, we note that the indirect effects analyses provided below all adhere to what is described above except for the analysis for SRKW (Section 5.5.18). SRKW generally consume Chinook salmon only, making the prey-based SSD/HC_5 approach irrelevant.

Figure 5-58 removed for this versionTable 5-57 removed for this version

5.5.1 Vernal Pool Fairy Shrimp Aluminum Indirect Effects Assessment: Freshwater

The vernal pool fairy shrimp occurs primarily in vernal pools and seasonal wetlands that fill with water during fall and winter rains and dry up in spring and summer. The species does not inhabit the majority of these vernal pools within or between complexes at any certain time, and different pools may provide habitat over alternative years based on climatic conditions and the variation of those conditions. The vernal pool fairy shrimp consumes algae and plankton by scraping and straining them from surfaces within the vernal pool. In addition, the vernal pool fairy shrimp also feeds on a diet of bacteria, protozoa, rotifers, and small amounts of detritus (Pennak, 1989)[46]

As discussed in EPA's 2018 aluminum criteria document (USEPA, 2018), freshwater and estuarine/marine algae and plants were not more sensitive than vertebrates and invertebrate animal taxa to Al exposures, so plant criteria were not developed. EPA (2018), states, "*the effect levels observed* [to plants] *are similar to the available animal data, and the recommended criteria should therefore be protective for algae and aquatic plants*." Because plants were not included in the chronic Al criterion SSD, a vernal pool fairy shrimp prey-item-specific HC_5 was not derived. However, the acute and chronic Al criteria, based on animal responses, were expected to be protective of plants. EPA (2018) states:

> *Aquatic plant data are not used to derive the criteria for aluminum. However, a summary of available data is presented below. For freshwater algae, aluminum effect*

[46] Here and throughout the indirect effect assessment document, many references used to describe the status of the species also provided necessary dietary information used in the indirect effect assessment. Readers are encouraged to see **Section 3**, *status of the species*, for additional information and species descriptions.

concentrations ranged from 50 µg/L to 6,477 µg/L, with most effect levels below 1,000 µg/L (Appendix E I [of USEPA (2018)] *Acceptable Toxicity Data of Aluminum to Freshwater Aquatic Plants). Studies for freshwater macrophytes are limited, but available data suggest freshwater macrophytes are more tolerant to aluminum than freshwater algae. The effect concentration for Eurasian watermilfoil is 2,500 µg/L based on root weight (Stanley, 1974), which is near the upper range of freshwater algae sensitivities. Several 3-day tests with the green alga Pseudokirchneriella subcapitata at pH 6, 7 and 8 across a range of total hardness and DOC concentrations revealed that both an increase in pH, total hardness and DOC reduced the toxicity of aluminum (European Aluminum Association 2009). DeForest, Brix, Tear, and Adams (2018) used these 27 toxicity tests (as summarized in Robert W Gensemer et al. (2018)) to develop a MLR model to explain the effects of water chemistry on algal toxicity. The MLR model developed was:*

$$P.\ subcapitata\ EC_{20} = e[-61.952+[1.678\times\ln(DOC)]+[4.007\times\ln(hard)]+(17.019\times pH)-(1.020\times pH2)-[0.204\times pH:\ln(DOC)]-[0.556\times pH:\ln(hard)]]$$

The MLR model for P. subcapitata was within a factor of two for 100% of the predicted versus observed values (DeForest et al. 2018a). Most of the acceptable toxicity data for freshwater aquatic plants (Appendix E [of USEPA (2018)]*) did not report all three water quality parameters (i.e., pH, total hardness and DOC) preventing the use of applying the alga based MLR equation to the data. The EPA contacted authors and in limited cases, the authors were able to provide rough estimates of some of the missing information. Normalized lowest observed effect concentrations (LOECs) for the twenty-one-day tests as reported by Pillsbury and Kingston (1990) were 3,482 µg/L, while normalized 4-day* EC_{50}*s for P. subcapitata were 620 and 1,067 µg/L (Call et al., 1984). These values are above the chronic criterion at the same test conditions, suggesting that the criteria developed using aquatic animals will also be protective of aquatic plants. This was also observed when normalizing the 3-day P. subcapitata test in Appendix H [*of USEPA (2018)] *(Other Data on Effects of Aluminum to Freshwater Aquatic Organisms) with normalized effect concentrations ranging from 161 to 5,113 µg/L. The geometric mean of these values was 1,653 µg/L (Note: 77 these tests were excluded from the acceptable table due to the insufficient test duration, less than 4 days). In contrast to other freshwater plants, duckweed is highly tolerant to aluminum, with an effect concentration based on reduced growth of greater than 45,700 µg/L (Call et al., 1984). For the one acceptable study of a saltwater plant (Seagrass, Halophila stipulacea), less than 50% mortality of teeth cells was observed at 26.98 µg/L, and more than 50% mortality of teeth cells observed at 269.8 µg/L (Malea & Haritonidis, 1996). In a shorter duration study, the saltwater algal species, Dunaliella tertiolecta, also exhibited sensitivity to aluminum, but the effect concentration was higher at 18,160 µg/L (Appendix I* [of USEPA (2018)] *Other Data on Effects of Aluminum to Estuarine/Marine Aquatic Organisms). Although aquatic plant data are not normalized using the alga based MLR equation, the effect levels observed are similar to the available animal data, and the recommended criteria should therefore be protective for algae and aquatic plants.*

Overall, vernal pool fairy shrimp plant-based food resources are not anticipated to be sensitive to Al at acute and chronic criteria concentrations. If aluminum were to exist at criteria concentrations indefinitely in Oregon fresh waters (which is not the intent of the action considering the full definition of criteria include magnitude, duration, and frequency), it is unlikely that vernal pool fairy shrimp would experience indirect effects.

5.5.2 Green Sturgeon Aluminum Indirect Effects Assessment: Freshwater

The green sturgeon is a bottom-dwelling species, mostly seen from inshore waters to 200 feet, primarily in the seawater and mixing zones of bays and estuaries. In estuaries, they concentrate in deep areas with soft sediments and move into intertidal areas to feed at high tides. Adults may travel tens of miles upstream to spawn in rivers and require good water quality and specific temperatures to spawn and hatch their eggs. Both juveniles and adults move extensively along the Pacific coast to take advantage of scattered food resources. Juveniles remain in freshwater for one to four years before heading to more estuarine waters, where they remain for up to four to six years, during which they can migrate considerable distances along the coast as they grow larger.

Green sturgeon is an opportunistic predator and will consume a variety of available prey types. These fish feed by using an elongated mouth appendage that sucks food and sediment from the sediment surface. 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*), and polychaetes (Dumbauld, Holden, & Langness, 2008b; NMFS, 2018b). 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, 2018b; Radtke, 1966). Diet for the larval stage of green sturgeon is largely unknown; however, it is assumed to be similar to that of the white sturgeon which would include insect larvae, oligochaetes, and decapods (NMFS, 2009a). After spawning adults likely feed on benthic prey species such as lamprey ammocoetes and crayfish. Adult green sturgeon and near adult green sturgeon are found in bays and coastal regions, with diets consisting of shrimp, clams, crabs, and benthic fish. Recent evaluations of subadult green sturgeon captured in the California halibut trawl fishery suggest their diet consists of right-eyed flatfish, shrimp, bivalves, and crabs (NMFS, 2018b).

Table 5-59 is the USEPA (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. 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).

Table 5-59. Green sturgeon chronic species-prey item sensitivity distribution in reference water conditions.

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

Salmonids are the two most sensitive genera in most Oregon waters (Table 5-58) and do not serve as green sturgeon prey items. Consequently, green sturgeon prey items are expected to be adequately protected by the chronic criterion across Oregon water chemistries. To account for changes in species prey base sensitivity relative to the chronic Al criterion magnitude across water chemistries, green sturgeon prey-item-specific HC_5 values and genus mean chronic values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time for each set of paired water chemistry data available (n = 19,274). Comparisons were then limited to water quality data collected for level-III ecoregions overlapping the green sturgeon Southern DPS (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 genus mean chronic value 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 will not be significantly affected or reduced.

Moreover, green sturgeon is a broad opportunistic feeder that relies on a range of available food sources and would, therefore, not experience a large loss of prey availability if one particular prey item or taxa were reduced. If aluminum were to exist at criteria concentrations indefinitely in Oregon fresh waters (which is not the intent of the action considering the full definition of criteria including magnitude, duration, and frequency), it is highly unlikely that green sturgeon would be indirectly affected because larval and juvenile green sturgeon rely predominantly on aluminum-tolerant prey species (e.g., benthic invertebrates) as food sources.

5.5.3 Lahontan Cutthroat Trout Aluminum Indirect Effects Assessment: Freshwater

The Lahontan cutthroat trout is a subspecies of cutthroat trout species and exists primarily in streams of the Lahontan and Coyote Lake Basins in southeastern Oregon. Fluvial Lahontan cutthroat trout are opportunistic feeders that eat a variety of foods including aquatic insects, zooplankton, and some terrestrial forms of food (TRBT, 2003). In general, the diet of small individuals is dominated by invertebrates, and the diet of larger individuals is composed

primarily of fish. Lacustrine forms of Lahontan cutthroat trout eat zooplankton and a variety of fish species including tui chub (*Gila bicolor*), Lahontan redside shiners (*Richardsonius egregius*), speckled dace (*Rhinichthys osculus*), and Tahoe suckers (*Catostomus tahoensis*) (TRBT, 2003).

Table 5-60 is the USEPA (2018) aluminum chronic criterion dataset modified to reflect species or surrogates that represent possible Lahontan cutthroat trout prey items (juvenile emphasis). 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).

Table 5-60. Lahontan cutthroat trout chronic species-prey item sensitivity distribution in reference water conditions (pH = 7, DOC = 1 mg/L, total hardness = 100 mg/L).

Rank	GMCV (µg/L)	Species
8	20,514	Oligochaete, *Aeolosoma sp.*
7	5,099	Midge, *Chironomus riparius*
6	3,539	Rotifer, *Brachionus calyciflorus*
5	2,407	Fathead minnow, *Pimephales promelas*
4	1,387	Amphipod, *Hyalella azteca*
3	1,342	Zebrafish, *Danio rerio*
2	1,181	Cladoceran, *Ceriodaphnia dubia*
1	985.3	Cladoceran, *Daphnia magna*

Salmonids are the two most chronic sensitive genera in most Oregon waters (Table 5-58) and do not serve as juvenile Lahontan cutthroat trout prey items. Consequently, Lahontan cutthroat trout prey items (primarily aquatic insects and zooplankton) are expected to be adequately protected by the chronic criterion across Oregon water chemistries. To account for changes in species prey base sensitivity relative to the chronic Al criterion magnitude across water chemistries, Lahontan cutthroat trout prey-item-specific HC_5 values and genus mean chronic values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time for each set of paired water chemistry data available (n = 19,274). Comparisons were then limited to water quality data collected for level-III ecoregions overlapping the Lahontan cutthroat trout range (n = 443). The chronic aluminum CCC exceeded the Lahontan cutthroat trout prey-item HC_5 in 50.79% of water chemistry scenarios. The most sensitive prey-item's genus mean chronic value was greater than the CCC in 100% of water chemistries evaluated. Considering the most sensitive prey item GMCV was always greater than the criterion, Lahontan cutthroat trout prey resources will not be significantly affected or reduced. If aluminum were to exist at criteria concentrations indefinitely in Oregon fresh waters (which is not the intent of the action considering the full definition of criteria including magnitude, duration, and frequency), it is unlikely that Lahontan cutthroat trout would be indirectly affected because their prey are aluminum-tolerant food sources.

5.5.4 Chum Salmon Aluminum Indirect Effects Assessment: Freshwater

Chum salmon are an anadromous fish species that only inhabit fresh water during a short juvenile stage after hatching in their natal streams. These fish quickly migrate into the marine environment after hatching where they congregate in schools. Prior to migration, chum salmon fry feed on small invertebrates and crustaceans (Behnke, 2010). As juveniles, chum salmon feed broadly on benthic invertebrates, including insects, crustaceans, and mollusks as primary food sources. Adult chum salmon exhibit opportunistic feeding on invertebrates and fishes. They preferentially feed on larger zooplankton over small zooplankton (Higgs, Macdonald, Levings, & Dosanjh, 2010). Because of their larger stomach, chum salmon are better able to utilize gelatinous zooplankton more efficiently than other species (NPFMC, 2012). As with all anadromous salmonids, the adult salmon will rarely eat as they migrate back to fresh water. Therefore, their diet in freshwater will only consist of invertebrate items consumed by fish up to and including the juvenile stage.

Table 5-61 is the USEPA 2018 aluminum chronic criterion dataset modified to reflect species or surrogates that represent possible juvenile chum salmon prey items. 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).

Table 5-61. Chum salmon chronic species-prey item sensitivity distribution in reference water conditions (pH = 7, DOC = 1 mg/L, total hardness = 100 mg/L).

Rank	GMCV (µg/L)	Species
8	20,514	Oligochaete, *Aeolosoma sp.*
7	5,099	Midge, *Chironomus riparius*
6	3,539	Rotifer, *Brachionus calyciflorus*
5	3,119	Great pond snail, *Lymnaea stagnalis*
4	1,387	Amphipod, *Hyalella azteca*
3	1,181	Cladoceran, *Ceriodaphnia dubia*
2	1,026	Fatmucket, *Lampsilis siliquoidea*
1	985.3	Cladoceran, *Daphnia magna*

Salmonids were the two most chronic sensitive genera in most Oregon waters (Table 5-58) and do not serve as juvenile chum salmon primary prey items. Consequently, chum salmon prey items are expected to be adequately protected by the chronic criterion across Oregon water chemistries. To account for changes in species prey base sensitivity relative to the chronic Al criterion magnitude across water chemistries, chum salmon prey-item-specific HC_5 values and genus mean chronic values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time for each set of paired water chemistry data available (n = 19,274). Comparisons were then limited to water quality data collected for level-III ecoregions overlapping chum salmon range (n = 14,735). The chronic aluminum CCC exceeded the chum salmon prey-item HC_5 in 19.86% of water chemistry scenarios. The most sensitive prey-item's genus mean chronic value was greater than the CCC in 100% of water chemistries evaluated. Considering the most sensitive prey item GMCV was always greater than the criterion, chum

salmon prey resources will not be significantly affected or reduced. Since salmonids themselves are the most sensitive species in many of Oregon's waters, indirect effects of the approval action are less significant, since prey items are less sensitive than salmonids to aluminum toxicity. As a result, EPA approval of Oregon freshwater acute and chronic aluminum standards is not expected to affect chum salmon through indirect effects.

5.5.5 Coho Salmon Aluminum Indirect Effects Assessment: Freshwater

Coho salmon are anadromous fish species but can also live in freshwater rivers and streams, lakes, estuaries and marine environments, depending on life history stage. As juveniles, coho salmon are opportunistic and feed broadly on benthic invertebrates, including insects, crustaceans, and mollusks as primary food sources. Seasonal rainfall plays a factor in the diet of fry in riverine and pond habitats, with large numbers of terrestrial insects (springtails, Isotomidae and Entomobryidae) utilized as prey in early December after heavy rain, and feeding more on benthic invertebrates, such as taeniopterygid nymphs and simuliid larvae in the creek, and chironomid larvae in creek and ponds, in January when rainfall was low (Minakawa & Kraft, 1999). Juvenile coho salmon are opportunistic, however, and have also been shown to feed on adult salmonid carcasses in the winter months (Bilby, Fransen, & Bisson, 1996), as reported in (Sutherland, 2005). As the fish grow larger, they become increasingly more piscivorous, feeding on smaller pelagic marine fishes (State of California, 2004).

Table 5-62 is the USEPA 2018 aluminum chronic criterion dataset modified to reflect species or surrogates that represent possible juvenile coho salmon prey items. 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).

Table 5-62. Coho salmon chronic species-prey item sensitivity distribution in reference water conditions (pH = 7, DOC = 1 mg/L, total hardness = 100 mg/L).

Rank	GMCV (µg/L)	Species
12	20,514	Oligochaete, *Aeolosoma sp.*
11	5,099	Midge, *Chironomus riparius*
10	3,539	Rotifer, *Brachionus calyciflorus*
9	3,119	Great pond snail, *Lymnaea stagnalis*
8	2,407	Fathead minnow, *Pimephales promelas*
7	1,387	Amphipod, *Hyalella azteca*
6	1,342	Zebrafish, *Danio rerio*
5	1,181	Cladoceran, *Ceriodaphnia dubia*
4	1,026	Fatmucket, *Lampsilis siliquoidea*
3	985.3	Cladoceran, *Daphnia magna*
2	638.2	Brook trout, *Salvelinus fontinalis*
1	434.4	Atlantic salmon, *Salmo salar*

Salmonids were the two most chronic sensitive genera in most Oregon waters (Table 5-58) and have potential (i.e., secondarily, over winter) to serve as a juvenile coho salmon prey item. Consequently, coho salmon prey items could be reduced (e.g., adult coho salmon carcasses) by long-term Al exposures. Young and juvenile salmon are generally more sensitive to aluminum than adults and adult fish are not expected to be affected by aluminum at the CCC. Furthermore, aluminum at the chronic criterion concentration is not anticipated to exist continuously across in Oregon fresh waters.

To account for changes in species prey base sensitivity relative to the chronic Al criterion magnitude across water chemistries, coho salmon prey-item-specific HC_5 values and genus mean chronic values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time for each set of paired water chemistry data available (n = 19,274). Comparisons were then limited to water quality data collected for level-III ecoregions overlapping coho salmon range (n = 15,411). The chronic aluminum CCC exceeded the coho salmon prey-item HC_5 in 94.45% of water chemistry scenarios. The most sensitive prey-item's genus mean chronic value was greater than the CCC in 99.29% of water chemistries evaluated. The low percent of CCC values that are lower than the corresponding lowest GMCVs is a result of rounding differences. For example, US EPA (1985) states intermediate calculations, such as the calculation of GMCVs, should be rounded to four significant digits while the chronic criterion magnitude is only rounded to two significant digits. Considering the most sensitive prey item GMCV was almost always (i.e., 99.29%) greater than the criterion, coho salmon prey resources will not be significantly affected or reduced. The majority of coho salmon prey items are insensitive to chronic Al exposures because coho salmon are broad opportunistic feeders. As a result, EPA approval of Oregon freshwater acute and chronic aluminum standards is not expected to affect coho salmon through indirect effects.

5.5.6 Sockeye Salmon Aluminum Indirect Effects Assessment: Freshwater

Sockeye salmon are generally anadromous, but distinct populations of non-anadromous sockeye salmon also exist; these fish are commonly referred to as kokanee (*O. nerka* Kennerlyi) or silver trout (Wydoski & Whitney, 2003). They display more life history diversity than all other members of the Oncorhynchus genus (Burgner, 1991).

Juvenile sockeye salmon generally feed on plankton (such as ostracods, cladocerans, and copepods), benthic amphipods, and insects before they migrate to the ocean. In the ocean, they continue to feed on plankton but also prey upon larval and small adult fishes (such as sand lance), and occasionally squid (ADFG, 1994).

Table 5-63 is the USEPA 2018 aluminum chronic criterion dataset modified to reflect species or surrogates that represent possible juvenile sockeye salmon prey items. 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).

Table 5-63. Sockeye salmon chronic species-prey item sensitivity distribution in reference water conditions (pH = 7, DOC = 1 mg/L, total hardness = 100 mg/L).

Rank	GMCV (µg/L)	Species
8	20,514	Oligochaete, *Aeolosoma sp.*
7	5,099	Midge, *Chironomus riparius*
6	3,539	Rotifer, *Brachionus calyciflorus*
5	3,119	Great pond snail, *Lymnaea stagnalis*
4	1,387	Amphipod, *Hyalella azteca*
3	1,181	Cladoceran, *Ceriodaphnia dubia*
2	1,026	Fatmucket, *Lampsilis siliquoidea*
1	985.3	Cladoceran, *Daphnia magna*

Salmonids were the two most chronic sensitive genera in most Oregon waters (Table 5-58) and do not serve as juvenile sockeye salmon primary prey items nor do they need to serve as surrogates for other fish-based prey items. Consequently, sockeye salmon prey items are expected to be adequately protected by the chronic criterion across Oregon water chemistries. To account for changes in species prey base sensitivity relative to the chronic Al criterion magnitude across water chemistries, sockeye salmon prey-item-specific HC_5 values and genus mean chronic values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time for each set of paired water chemistry data available (n = 19,274). Comparisons were then limited to water quality data collected for level-III ecoregions overlapping sockeye salmon range (n = 16,865). The chronic aluminum CCC exceeded the sockeye salmon prey-item HC_5 in 22.19% of water chemistry scenarios. The most sensitive prey-item's genus mean chronic value was greater than the CCC in 100% of water chemistries evaluated. Considering the most sensitive prey item GMCV was always greater than the criterion, sockeye salmon prey resources will not be significantly affected or reduced. As a result, EPA approval of Oregon freshwater acute and chronic aluminum standards is not expected to affect sockeye salmon through indirect effects.

5.5.7 Chinook Salmon Aluminum Indirect Effects Assessment: Freshwater

Chinook salmon, also called king salmon, are the largest and least abundant species of Pacific salmon. They are anadromous, and as juveniles feed on insect larvae and zooplankton when they are in freshwater. Juveniles feed on terrestrial and aquatic invertebrates. As the Chinook salmon begin to age and mature and enter marine waters their diet changes and they begin to eat epipelagic fish such as herring, sand lance, smelt, and, anchovy along with shrimp and squid (Scott & Crossman, 1973).

Table 5-64 is the USEPA 2018 aluminum chronic criterion dataset modified to reflect species or surrogates that represent possible juvenile chinook salmon prey items. 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).

Table 5-64. Chinook salmon chronic species-prey item sensitivity distribution in reference water conditions (pH = 7, DOC = 1 mg/L, total hardness = 100 mg/L).

Rank	GMCV (µg/L)	Species
8	20,514	Oligochaete, *Aeolosoma sp.*
7	5,099	Midge, *Chironomus riparius*
6	3,539	Rotifer, *Brachionus calyciflorus*
5	3,119	Great pond snail, *Lymnaea stagnalis*
4	1,387	Amphipod, *Hyalella azteca*
3	1,181	Cladoceran, *Ceriodaphnia dubia*
2	1,026	Fatmucket, *Lampsilis siliquoidea*
1	985.3	Cladoceran, *Daphnia magna*

Since salmonids themselves are the most sensitive species in many of Oregon's waters, indirect effects of the approval action are less significant, since prey items are less sensitive than salmonids to aluminum toxicity. To account for changes in species prey base sensitivity relative to the chronic Al criterion magnitude across water chemistries, Chinook salmon prey-item-specific HC_5 values and genus mean chronic values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time for each set of paired water chemistry data available (n = 19,274). Comparisons were then limited to water quality data collected for level-III ecoregions overlapping Chinook salmon range (n = 16,865). The chronic aluminum CCC exceeded the Chinook salmon prey-item HC_5 in 22.19% of water chemistry scenarios. The most sensitive prey-item's genus mean chronic value was greater than the CCC in 100% of water chemistries evaluated. Considering the most sensitive prey item GMCV was always greater than the criterion, Chinook salmon prey resources will not be significantly affected or reduced. As a result, EPA approval of Oregon freshwater acute and chronic aluminum standards is not expected to affect Chinook salmon through indirect effects.

5.5.8 Steelhead Aluminum Indirect Effects Assessment: Freshwater

Steelhead are anadromous fish species but can also live in freshwater rivers and streams, lakes, estuaries, and marine environments, depending on life history stage. As juveniles, steelhead feed broadly on benthic invertebrates, including insects, crustaceans, and mollusks as primary food sources, with dietary composition expanding with age to include small fish and thus becoming top predators. They may also eat snails, plankton, and leeches as adult fish (Mueller and Staley 2000).

Table 5-65 is the USEPA 2018 aluminum chronic criterion dataset modified to reflect species or surrogates that represent possible steelhead prey items. 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).

Since salmonids themselves are the most sensitive species in many of Oregon's waters, indirect effects of the approval action are less significant, since prey items are less sensitive than salmonids to aluminum toxicity. To account for changes in species prey base sensitivity relative to the chronic Al criterion magnitude across water chemistries, steelhead prey-item-specific HC_5 values and genus mean chronic values were renormalized to Oregon-specific hardness, pH, and

DOC measurements paired in space and time for each set of paired water chemistry data available (n = 19,274). Comparisons were then limited to water quality data collected for level-III ecoregions overlapping steelhead range (n = 16,865). The chronic aluminum CCC exceeded the steelhead prey-item HC_5 in 20.09% of water chemistry scenarios. The most sensitive prey-item's genus mean chronic value was greater than the CCC in 100% of water chemistries evaluated. Considering the most sensitive prey item GMCV was always greater than the criterion, steelhead prey resources will not be significantly affected or reduced. As a result, EPA approval of Oregon freshwater acute and chronic aluminum standards is not expected to affect steelhead through indirect effects.

Table 5-65. Steelhead chronic species-prey item sensitivity distribution in reference water conditions (pH = 7, DOC = 1 mg/L, total hardness = 100 mg/L).

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

5.5.9 Bull Trout Aluminum Indirect Effects Assessment: Freshwater

Bull trout are most common in high mountainous areas where snowfields and glaciers are present. They mainly occur in deep pools of large, cold rivers and lakes. Juvenile bull trout, during their first year of life, feed primarily on small aquatic invertebrates (Stewart, Mochnacz, Sawatzky, Carmichael, & Reist, 2007). They will also intake annelids, mollusks, crustaceans such as amphipods, cladocerans, and mysids, as well as fish species that are smaller in size. Bull trout become increasingly piscivorous with increasing size, and adult bull trout diet consists primarily of other fish species. Bull trout have been known to feed on whitefish, sculpins, darters, and other species of trout and salmon. They may also partake in the occasional consumption of small birds, such as ducklings, as well as smaller mammals such as shrews and mice (Stewart et al., 2007).

Table 5-66 is the USEPA 2018 aluminum chronic criterion dataset modified to reflect species or surrogates that represent possible bull trout prey items. 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).

Salmonids were the two most chronic sensitive genera in most Oregon waters (Table 5-58) and have potential to serve as a bull trout prey item. Consequently, bull trout prey items could be reduced by long-term Al exposures.. To account for changes in species prey base sensitivity

relative to the chronic Al criterion magnitude across water chemistries, bull trout prey-item-specific HC_5 values and genus mean chronic values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time for each set of paired water chemistry data available (n = 19,274). Comparisons were then limited to water quality data collected for level-III ecoregions overlapping bull trout range (n =17,836). The chronic aluminum CCC exceeded the bull trout prey-item HC_5 in 92.35% of water chemistry scenarios. The most sensitive prey-item's genus mean chronic value was greater than the CCC in 99.44% of water chemistries evaluated. The low percent of CCC values that are lower than the corresponding lowest GMCVs is a result of rounding differences. For example, US EPA (1985) states intermediate calculations, such as the calculation of GMCVs, should be rounded to four significant digits while the chronic criterion magnitude is only rounded to two significant digits. Considering the most sensitive prey item GMCV was almost always (i.e., 99.44%) greater than the criterion, bull trout prey resources will not be sufficiently affected or reduced. Furthermore, aluminum at the chronic criterion concentration is not anticipated to exist continuously across in Oregon fresh waters given how the criteria may be implemented into permits (see Appendix F, for example) and bull trout are broad opportunistic feeders. As a result, EPA approval of Oregon freshwater acute and chronic aluminum standards is not expected to affect bull trout through indirect effects.

Table 5-66. Bull trout chronic species-prey item sensitivity distribution in reference water conditions (pH = 7, DOC = 1 mg/L, total hardness = 100 mg/L).

Rank	GMCV (µg/L)	Species
12	20,514	Oligochaete, *Aeolosoma sp.*
11	5,099	Midge, *Chironomus riparius*
10	3,539	Rotifer, *Brachionus calyciflorus*
9	3,119	Great pond snail, *Lymnaea stagnalis*
8	2,407	Fathead minnow, *Pimephales promelas*
7	1,387	Amphipod, *Hyalella azteca*
6	1,342	Zebrafish, *Danio rerio*
5	1,181	Cladoceran, *Ceriodaphnia dubia*
4	1,026	Fatmucket, *Lampsilis siliquoidea*
3	985.3	Cladoceran, *Daphnia magna*
2	638.2	Brook trout, *Salvelinus fontinalis*
1	434.4	Atlantic salmon, *Salmo salar*

5.5.10 Warner Sucker Aluminum Indirect Effects Assessment: Freshwater

The Warner sucker inhabits lotic freshwaters where larvae rely primarily on planktonic crustaceans. As they become older, they begin to rely upon diatoms, filamentous algae, and detritus (USFWS, 1998).

Table 5-67. is the USEPA 2018 aluminum chronic criterion dataset modified to reflect species or surrogates that represent possible Warner sucker prey items. 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).

Table 5-67. Warner sucker chronic species-prey item sensitivity distribution in reference water conditions (pH = 7, DOC = 1 mg/L, total hardness = 100 mg/L).

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

Salmonids are the two most chronic sensitive genera in most Oregon waters (Table 5-58) and do not serve as Warner sucker prey items. Consequently, Warner sucker prey items are expected to be adequately protected by the chronic criterion across Oregon water chemistries. To account for changes in species prey base sensitivity relative to the chronic Al criterion magnitude across water chemistries, Warner sucker prey-item-specific HC_5 values and genus mean chronic values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time for each set of paired water chemistry data available (n = 19,274). Comparisons were then limited to water quality data collected for level-III ecoregions overlapping the Warner sucker range (n = 443). The chronic aluminum CCC exceeded the Warner sucker prey-item HC_5 in 69.75% of water chemistry scenarios. The most sensitive prey-item's genus mean chronic value was greater than the CCC in 100% of water chemistries evaluated. Considering the most sensitive prey item GMCV was always greater than the criterion, Warner sucker prey resources will not be significantly affected or reduced. Since Warner sucker is a generalized feeder that relies on a range of available food sources, they are not expected to experience a large loss of prey availability if one particular prey item or taxa were reduced. As a result, EPA approval of Oregon freshwater acute and chronic aluminum standards is not expected to affect the Warner sucker through indirect effects.

5.5.11 Shortnose Sucker Aluminum Indirect Effects Assessment: Freshwater

The shortnose sucker inhabits lotic aquatic habitat. It broadly relies on organic debris, microscopic animals, algae or diatoms, and aquatic insects as primary food sources (USFWS, 2012c).

Table 5-68 is the USEPA 2018 aluminum chronic criterion dataset modified to reflect species or surrogates that represent possible shortnose sucker prey items. 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).

Table 5-68. Shortnose sucker chronic species-prey item sensitivity distribution in reference water conditions (pH = 7, DOC = 1 mg/L, total hardness = 100 mg/L).

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

Salmonids are the two most chronic sensitive genera in most Oregon waters (Table 5-58) and do not serve as shortnose sucker prey items. Consequently, shortnose sucker prey items are expected to be adequately protected by the chronic criterion across Oregon water chemistries. To account for changes in species prey base sensitivity relative to the chronic Al criterion magnitude across water chemistries, shortnose sucker prey-item-specific HC_5 values and genus mean chronic values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time for each set of paired water chemistry data available (n = 19,274). Comparisons were then limited to water quality data collected for level-III ecoregions overlapping the shortnose sucker range (n = 1,830). The chronic aluminum CCC exceeded the shortnose sucker prey-item HC_5 in 71.64% of water chemistry scenarios. The most sensitive prey-item's genus mean chronic value was greater than the CCC in 100% of water chemistries evaluated. Considering the most sensitive prey item GMCV was always greater than the criterion, shortnose sucker prey resources will not be significantly affected or reduced. Since the shortnose sucker is a generalized feeder that relies on a range of available food sources, shortnose sturgeon are not expected to experience a large loss of prey availability if one particular prey item or taxa were reduced. As a result, EPA approval of Oregon freshwater acute and chronic aluminum standards is not expected to affect the shortnose sucker through indirect effects.

5.5.12 Lost River Sucker Aluminum Indirect Effects Assessment: Freshwater

The Lost River sucker (with the exception of spawning) inhabits lentic environments. Like the other sucker species evaluated in this assessment, Lost River sucker broadly rely on organic debris, microscopic animals, algae or diatoms, and aquatic insects as primary food sources (USFWS, 2012c).

Table 5-69 is the USEPA 2018 aluminum chronic criterion dataset modified to reflect species or surrogates that represent possible Lost River sucker prey items. 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).

Table 5-69. Lost River sucker chronic species-prey item sensitivity distribution in reference water conditions (pH = 7, DOC = 1 mg/L, total hardness = 100 mg/L).

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

Salmonids are the two most chronic sensitive genera in most Oregon waters (Table 5-58) and do not serve as Lost River sucker prey items. Consequently, Lost River sucker prey items are expected to be adequately protected by the chronic criterion across Oregon water chemistries. To account for changes in species prey base sensitivity relative to the chronic Al criterion magnitude across water chemistries, Lost River sucker prey-item-specific HC_5 values and genus mean chronic values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time for each set of paired water chemistry data available (n = 19,274). Comparisons were then limited to water quality data collected for level-III ecoregions overlapping the Lost River sucker range (n = 1,830). The chronic aluminum CCC exceeded the Lost River sucker prey-item HC_5 in 71.64% of water chemistry scenarios. The most sensitive prey-item's genus mean chronic value was greater than the CCC in 100% of water chemistries evaluated. Considering the most sensitive prey item GMCV was always greater than the criterion, Lost River sucker prey resources will not be significantly affected or reduced. Since the Lost River sucker is a generalized feeder that relies on a range of available food sources, Lost River sucker are not expected to experience a large loss of prey availability if one particular prey item or taxa were reduced. As a result, EPA approval of Oregon freshwater acute and chronic aluminum standards is not expected to affect the Lost River sucker through indirect effects.

5.5.13 Foskett Speckled Dace Aluminum Indirect Effects Assessment: Freshwater

Foskett speckled dace in Oregon inhabit Foskett and Dace springs, small desert springs on the west side of Coleman Lake in Lake County, Oregon in the Warner Basin. Speckled dace in general are omnivorous, feeding on detritus, filamentous algae, aquatic insects, and eggs of other fish (Matthews & Moseley, 1990; McPhail & Baxter, 1996).

Table 5-70 is the USEPA 2018 aluminum chronic criterion dataset modified to reflect species or surrogates that represent possible Foskett speckled dace prey items. 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).

Table 5-70. Foskett speckled dace chronic species-prey item sensitivity distribution in reference water conditions (pH = 7, DOC = 1 mg/L, total hardness = 100 mg/L).

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

Salmonids are the two most chronic sensitive genera in most Oregon waters (Table 5-58) and do not serve as Foskett speckled dace prey items. Consequently, Foskett speckled dace prey items are expected to be adequately protected by the chronic criterion across Oregon water chemistries. To account for changes in species prey base sensitivity relative to the chronic Al criterion magnitude across water chemistries, Foskett speckled dace prey-item-specific HC_5 values and genus mean chronic values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time for each set of paired water chemistry data available (n = 19,274). Comparisons were then limited to water quality data collected for level-III ecoregions overlapping the Foskett speckled dace range (n = 443). The chronic aluminum CCC exceeded the Foskett speckled dace prey-item HC_5 in 69.75% of water chemistry scenarios. The most sensitive prey-item's genus mean chronic value was greater than the CCC in 100% of water chemistries evaluated. Considering the most sensitive prey item GMCV was always greater than the criterion, Foskett speckled dace prey resources will not be significantly affected or reduced. Since the Foskett speckled dace is a generalized feeder that relies on a range of available food sources, Foskett speckled dace are not expected to experience a large loss of prey availability if one particular prey item or taxa were reduced. As a result, EPA approval of Oregon freshwater acute and chronic aluminum standards is not expected to affect the Foskett speckled dace through indirect effects.

5.5.14 Hutton Tui Chub Aluminum Indirect Effects Assessment: Freshwater

The Hutton tui chub is the only fish found in the Alkali Subbasin in southwestern Oregon. As of 1996, the distribution of the Hutton tui chub is thought to include only a single spring in the Alkali subbasin: Hutton Spring. Attempts to find this unnamed spring in 1996 were unsuccessful and this population may have been lost.

There is very little information regarding the ecology of the Hutton tui chub. Hutton tui chub is expected to be omnivorous with a majority of food eaten being filamentous algae (USFWS, 1998). As discussed in EPA's 2018 aluminum criteria document, freshwater and estuarine/marine algae and plants were not more sensitive than vertebrates and invertebrate animal taxa to Al exposures, so plant criteria were not developed. Because plants were not included in the chronic Al criterion SSD, and the hutton tui chub is thought to rely heavily on plants as a food resource, Hutton tui chub prey-item-specific HC_5's were not derived. Overall, plants are not more sensitive than vertebrate and invertebrate taxa used to derive the criteria (see **Section 5.5.1**) Given the limited range of the Hutton tui chub it is unlikely Hutton tui chub prey

items will be meaningfully exposed to Al dischargers. Hutton tui chub does not feed on salmonids, which represented the most sensitive genera to chronic Al exposures in most Oregon waters. Consequently, EPA approval of the Oregon freshwater acute and chronic aluminum standards is not expected to affect the Hutton tui chub through indirect effects.

5.5.15 Borax Lake Chub Aluminum Indirect Effects Assessment: Freshwater

The Borax Lake chub resides in Borax Lake. Borax Lake is a natural lake fed from the waters of several thermal springs and is perched atop large sodium-borate deposits in the Alvord Desert. It is small and shallow, about 4.1 hectares (10 acres) in size. Spring inputs near the bottom of a deep vent, 32 meters below the surface, range from 40 to 148°C (104 to 300°F). Surface water temperatures typically range from 16 to 38°C (61 to 100°F), but fluctuations occur and temperatures occasionally exceed 38°C (100°F) (USFWS, 2012a).

The Borax Lake chub is an opportunistic omnivore (J. E. Williams & Williams, 1980). Juveniles and adults eat essentially the same food resources, including aquatic and terrestrial insects, algae, mollusks and mollusk eggs, aquatic worms, fish scales, spiders, and seeds.

Table 5-71 reflects the USEPA 2018 aluminum chronic criterion dataset modified to reflect species or surrogates that represent possible Borax Lake chub prey items. 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).

Table 5-71. Borax Lake chub chronic species-prey item sensitivity distribution in reference water conditions (pH = 7, DOC = 1 mg/L, total hardness = 100 mg/L).

Rank	GMCV (µg/L)	Species
8	20,514	Oligochaete, *Aeolosoma sp.*
7	5,099	Midge, *Chironomus riparius*
6	3,539	Rotifer, *Brachionus calyciflorus*
5	3,119	Great pond snail, *Lymnaea stagnalis*
4	1,387	Amphipod, *Hyalella azteca*
3	1,181	Cladoceran, *Ceriodaphnia dubia*
2	1,026	Fatmucket, *Lampsilis siliquoidea*
1	985.3	Cladoceran, *Daphnia magna*

Salmonids are the two most chronic sensitive genera in most Oregon waters (Table 5-58) and do not serve as Borax Lake chub prey items. Consequently, Borax Lake chub prey items are expected to be adequately protected by the chronic criterion across Oregon water chemistries. If aluminum were to exist at criteria concentrations indefinitely in Oregon fresh waters (which is not the intent of the action considering the full definition of criteria including magnitude, duration, and frequency), it is highly unlikely that Borax Lake chub would be indirectly affected because their prey are aluminum-tolerant food sources.

To account for changes in species prey base sensitivity relative to the chronic Al criterion magnitude across water chemistries, Borax Lake chub prey-item-specific HC_5 values and genus mean chronic values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time for each set of paired water chemistry data available (n = 19,274). Comparisons were then limited to water quality data collected for level-III ecoregions overlapping the Borax Lake chub range (n = 443). The chronic aluminum CCC exceeded the Borax Lake chub prey-item HC_5 in 41.53% of water chemistry scenarios. The most sensitive prey-item's genus mean chronic value was greater than the CCC in 100% of water chemistries evaluated. Considering the most sensitive prey item GMCV was always greater than the criterion, Borax Lake chub prey resources will not be significantly affected or reduced. Since Borax Lake chub are generalized feeders that rely on a range of available food sources, they are not expected to experience a large loss of prey availability if one particular prey item or taxa were reduced. Consequently, EPA approval of the Oregon freshwater acute and chronic aluminum standards is not expected to affect the Borax Lake chub through indirect effects.

5.5.16 Pacific eulachon Aluminum Indirect Effects Assessment: Freshwater

Pacific eulachon are an anadromous, short-lived, high-fecundity, high-mortality forage fish. Eulachon spawn in freshwater streams. As the spawning season approaches, eulachon gather in large schools off the mouths of their spawning streams and rivers. Also like the anadromous salmonids, the majority of adult eulachon die soon after spawning.

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 studies have shown that their stomachs are not very full suggesting they do not actively feed during that time of the fall. Larval stages of the Pacific eulachon eat phytoplankton, copepods, copepod eggs, mysids, barnacle larvae, worm larvae, and eulachon larvae (WDFW & ODFW, 2001), as reported in (Willson et al., 2006).

Table 5-72 reflects the USEPA 2018 aluminum chronic criterion dataset modified to reflect species or surrogates that represent possible Pacific eulachon prey items. 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).

Table 5-72. Pacific eulachon chronic species-prey item sensitivity distribution in reference water conditions (pH = 7, DOC = 1 mg/L, total hardness = 100 mg/L).

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

Salmonids are the two most chronic sensitive genera in most Oregon waters (Table 5-58) and do not serve as Pacific eulachon prey items. Consequently, Pacific eulachon prey items are expected to be adequately protected by the chronic criterion across Oregon water chemistries. To account for changes in species prey base sensitivity relative to the chronic Al criterion magnitude across water chemistries, Pacific eulachon prey-item-specific HC_5 values and genus mean chronic values were renormalized to Oregon-specific hardness, pH, and DOC measurements paired in space and time for each set of paired water chemistry data available (n = 19,274). Comparisons were then limited to water quality data collected for level-III ecoregions overlapping the Pacific eulachon range (n = 13,973). The chronic aluminum CCC exceeded the Pacific eulachon prey-item HC_5 in 76.93% of water chemistry scenarios. The most sensitive prey-item's genus mean chronic value was greater than the CCC in 100% of water chemistries evaluated. Considering the most sensitive prey item GMCV was always greater than the criterion, Pacific eulachon prey resources will not be significantly affected or reduced. Since the Pacific eulachon is a generalized feeder that relies on a range of available food sources, Pacific eulachon are not expected to experience a large loss of prey availability if one particular prey item or taxa were reduced. As a result, EPA approval of Oregon freshwater acute and chronic aluminum standards is not expected to affect the Pacific eulachon through indirect effects.

5.5.17 Oregon Spotted Frog Aluminum Indirect Effects Assessment: Freshwater

The Oregon spotted frog is an aquatic native frog in the Pacific Northwest. It is found in or near perennial water bodies, including zones of shallow water and abundant emergent or floating aquatic plants (Hallock, 2013). The Oregon spotted frog prefer fairly large, warm marshes that can support a population to persist despite high predation rates and sporadic reproductive failures.

Oregon spotted frog tadpoles are predominately herbivorous, feeding on algae, decaying vegetation, and detritus. After they undergo metamorphosis, they become opportunistic predators (Licht, 1974), as reported in (Hallock, 2013). Insects [terrestrial] were observed as being the primary food source with the types of insects growing larger as the frog did the same. It has also been observed that the adults will occasionally prey upon other frog species such as newly metamorphosized Northern Red-Legged frogs and juvenile Western Toads (Hallock, 2013).

As discussed in EPA's 2018 aluminum criteria document, freshwater and estuarine/marine algae and plants were not more sensitive than vertebrates and invertebrate animal taxa to Al exposures, so plant criteria were not developed. (USEPA, 2018), states, "*the effect levels observed* [to plants] *are similar to the available animal data, and the recommended criteria should therefore be protective for algae and aquatic plants*" (see Section 5.5.1) Because plants were not included in the chronic Al criterion SSD, an Oregon spotted frog prey-item-specific HC_5 was not derived. Oregon spotted frog does not feed on salmonids, which represented the most sensitive genera to chronic Al exposures in most Oregon waters. Furthermore, the Oregon spotted frog is largely a generalist feeder, often times relying on terrestrial-based food resources that will experience no meaningful exposure to Al. Consequently, EPA approval of the Oregon freshwater acute and chronic aluminum standards is not expected to affect the Oregon spotted frog through indirect effects.

5.5.18 Southern Resident Killer Whale Aluminum Indirect Effects Assessment

As discussed in Section 3.17, the Southern Resident Killer Whale (SRKW) primarily consumes chinook salmon (>85% of their diet). Although other salmonid species aren't considered in this assessment, the conclusions are not affected given that all salmonids would be similarly affected by the proposed action. Therefore, this analysis is focused on the effects of the action on chinook prey biomass available for SRKW, as reduction of salmonid swim-up fry biomass was the most sensitive endpoint to Al exposure (see Section 5.2). The premise for the analysis in this section is that lost chinook biomass can translate to reduced prey biomass (i.e., calories per fish) which may adversely affect SRKW, given existing problems with this species' ability to fulfill its nutritional requirements. Below, EPA estimates lost chinook biomass relevant to SRKW diet.

Due to limited data, it is difficult to quantify the fraction of the SRKW diet that is composed of Oregon-reared Chinook. However, recent information suggests that approximately 55% of the fall diet of SRKW comes from Columbia River Chinook (Ward, Hanson, Ford, & Emmons, 2019 , unpublished data). EPA notes that it assumes that all of these salmon would be affected (exposed to Al at CCC concentrations that are greater than Chinook EC_5 concentrations and experience reductions in growth) by the action, although that is a conservative assumption. Therefore, in coordination with NOAA, EPA developed the following means of estimating the indirect effect of the proposed action on SRKW in terms of lost biomass to their primary prey item, Chinook salmon.

Lost Salmon Biomass (B) = X * Y * Z

Where,

B = % reduction in aggregate salmonid biomass

X = % reduction in salmon biomass when CCC > $EC_{05,\ chinook}$

Y = % Oregon water sample locations in which CCC > $EC_{05,\ chinook}$

Z = % of salmonid population relevant to SRKW diet

As an example, the indirect effect on SRKW by direct effects on Lower Columbia River chinook salmon ESU is shown here (See Table 5-17 of the BE):

B = 0.05 * 0.9470 * 0.55

B = 2.60%.

The table below provides the above information, analysis, and results for each Chinook Salmon ESU that comes into contact with the Columbia River Basin.

Table 5-73. Estimated percent lost prey biomass (*B*) available for SRKW that use Chinook salmon affected by Oregon waters given the percent reduction in salmon biomass when CCC>EC05 (*X*), percent Oregon water sample locations that CCC>EC05 (*Y*), and percent of the salmonid population that constitutes SRKW prey (*Z*). *B = X*Y*Z*.

Chinook Salmon ESU	X (%)	Y (%)	Z (%)	B (%)
Lower Columbia River	5.0	94.70	55.0	2.60
Snake River (fall)	5.0	92.08	55.0	2.53
Snake River (spring/summer)	5.0	92.08	55.0	2.53
Upper Columbia River (spring)	5.0	93.45	55.0	2.57
Upper Willamette River	5.0	96.47	55.0	2.65

The estimated lost prey biomass shown in Table 5-73 indicates that the action has the potential to adversely affect food resources for the SRKW. NOAA may use *B* to estimate lost salmonid individuals as a result of the proposed action. NOAA may estimate the number of lost salmon individuals using its population estimates and the estimated 2-3% reduction in aggregate chinook salmon biomass provided by EPA.

5.6 Mixtures Toxicity

EPA recognizes the potential for pollutants to co-occur in space and time and therefore potentially expose listed species to more than one pollutant at a time. Once Oregon has Al criteria in place, as a result of the proposed action, and if Al discharges occur in Oregon, there is a potential for more than one pollutant to be present in such a discharge. Thus, listed species in Oregon may be exposed to Al along with other pollutants downstream of a discharger. However, it is not possible to confidently know in advance what chemicals may co-occur, their concentrations, and how their proportions may vary in space and time. By extension, it is not possible to confidently predict with any certainty how listed species may be exposed to a given mixture. Furthermore, EPA does not have existing procedures to derive criteria for mixtures, *a priori.* Hence, EPA did not analyze the effects of mixtures toxicity for this BE.

However, it is noted that ODEQ conducts whole effluent toxicity (WET) testing in certain circumstances. WET testing accounts for the mixtures toxicity in a way that is relevant to the discharge and location and that can be used as a way of adjusting discharger limits in light of observed toxicity. EPA provides the following information regarding ODEQ's procedures for WET testing and mixing zones to help understand how mixtures may be dealt with in the context of Oregon's implementation programs.

- Section 4.7 of this BE provides language from Oregon's mixing zone policy
    - ODEQ 340-041-0053[47]
- Permits and Fact Sheets for two discharge permits in Oregon that include Al and WET testing requirements

[47] Department of Environmental Quality Chapter 340 Division 41. WATER QUALITY STANDARDS: BENEFICIAL USES, POLICIES, AND CRITERIA FOR OREGON. 340-041-0053, Mixing Zones

    - Northwest Aluminum Specialties (sometimes referred to as SAPA Extrusions), Permit No. 101759, File No. 53166, EPA Reference No. OR000170-8.
    - Fujimi Corporation, Permit No. 103033, File No. 107178, EPA Reference No. OR 004033-9
    - A third permit was recently (December 2019) found by EPA
        - Georgia-Pacific Wauna Mill, Permit No. 101172, File No. 21328, EPA Reference No. OR0000795.
- WET testing procedures
    - ODEQ Internal Management Directive, Reasonable Potential Analysis Process for Toxic Pollutants, Version 3.1.[48]

EPA then used this information to identify provisions and procedures intended to protect sensitive species from hazardous exposure to chemical mixtures. These are listed below.

- Appendix E[48] describes how WET testing procedures are to occur and how permit writers are to integrate the findings into permits to address the toxicity of the worst actor in the context of an environmental mixture. Toxicity Identification/Reduction Evaluation Plan procedures are defined in Appendix E as a way of addressing the worst actor(s).
- Provisions in the Mixing Zone Policy by Paragraph (See Section 4.7)
    - 2(a)(A)
    - 2(b)(A)
    - 2(c)(C)
    - 2(f)
- Two (Northwest Al Specialties and Georgia-Pacific Wauna Mill) of three permits listed above call for WET testing in the context of an Al discharge.

Given the above information, EPA has determined that Oregon's procedures for implementing mixing zones and conducting WET testing together provide an approach to addressing mixtures toxicity with respect to the implementation of the Al criteria. The information in this Section (5.6) and referenced in this section provides sufficient specificity for the evaluation of procedures to address mixtures toxicity in light of the proposed action.

5.7 Final Statement

The synthesis of the information presented above is provided in Chapter 6, Effects Determinations, using the tiering or weight of evidence approach (Section 5.1.1) depending on the sensitivity of the organism to aluminum exposure as evaluated in Tier 1/LOE 1).

[48] https://www.oregon.gov/deq/Filtered%20Library/rpaIMD.pdf

6. Effects Determinations, Cumulative Effects, Interdependent/Interrelated Actions

6.1 Synthesizing the available information for the final effects determinations

As described in Chapter 5 of this BE, EPA has considered multiple lines of evidence in order to make a final effects determination for each species. Figure 6-1 provides a schematic of how a hybrid of tiering and a weighing of the evidence was used to make effects determinations for each species. The final effects determinations are summarized in Section 6.2 and shown in Tables 6-2 and 6-3.

The species-level effects determinations were made in consideration of effects on species by both the CMC and CCC. Only one effects determination was made per species even if both the CMC and CCC would adversely affect a species. A tiered approach was used for species that were found to be insensitive to Al via direct exposure/effects. That is, for Tier 1, when a species acute or chronic toxicity value was greater than the CMC or CCC, for all Oregon water chemistries, a species was considered insensitive to Al for the purposes of this BE. A Tier 2 analysis was then utilized to determine if the species would be indirectly affected by the proposed action by way of a reduced prey base. If both Tiers 1 and 2 were negative, NLAA was determined. If, however, Tier 1 was negative but Tier 2 found effects to a species prey base (i.e., HC5 < CCC 100% of the time) then LAA was determined.

If a species was found to be sensitive during the Tier 1 analysis (Direct Effects), the Weight of Evidence approach was applied, and Tier 1 was then labeled Line of Evidence (LOE) 1, and the weighted analysis was performed. Subsequent LOEs through LOE 3 were evaluated according to the weighting process described in Section 5.1.1 and shown in Figure 6-1. The calculations for the percent evidence for LAA for each species and CMC/CCC values (across representative Oregon water chemistries) are summarized in Attachment 2. Determinations of NLAA or LAA were made and given a ranking of Very Low to High, depending on the percent of evidence for LAA (Table 6-1).

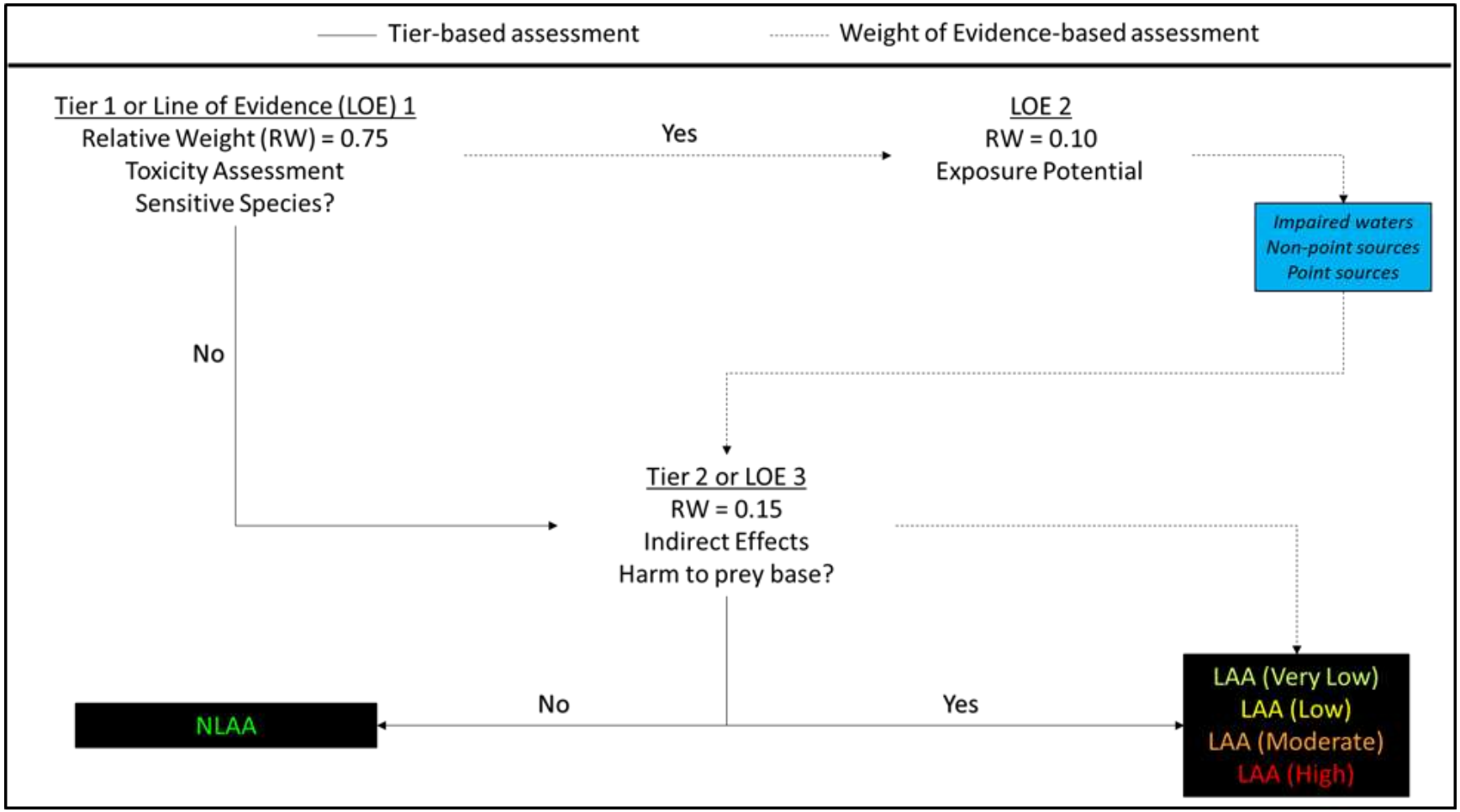

Figure 6-1. Effects determination schematic illustrating the hybrid approach used by EPA. Tiering was used for non-sensitive species whereas a weight of evidence approach was used for sensitive species.

Table 6-1. Effects determination criteria for species assessed with the Tiering (Tier 2, Indirect Effects) or Weight of Evidence (WOE) approach.

Effects Determination	Level of Evidence for LAA	[1]Tier 2 (Indirect Effects)	WOE Score Evidence for LAA
NLAA	None	0-5	0
LAA	Very Low	4	1-25
LAA	Low		26-50
LAA	Moderate		51-75
LAA	High		76-100

Notes:
1, Effects determinations for Tier 2 evaluated species (species insensitive under the Tier 1 evaluation process) were simply NLAA or LAA.

6.2 Effects Determinations for each Species/ESU/DPS subject to this consultation

The discussions below apply to both the CMC and CCC even when EPA refers to only the CCC in its descriptions. However, the CMC is addressed when adverse effects were detected for a species.

6.2.1 Oregon spotted frog (*Rana pretiosa*)

6.2.1.1 Direct and Indirect Effects Determinations for Oregon spotted frog (Rana pretiosa)

Aluminum CCC values, across the range of water chemistry conditions in ecoregions within which the Oregon spotted frog occurs, never exceeded the spotted frog chronic low effect threshold (i.e., EC_5) (Tier 1; Chapter 5). Indirect adverse effects are not expected for Oregon spotted frog (HC5 < CCC) (Tier 2; Chapter 5). Therefore, the EPA has determined that the Agency's proposed action **may affect but is not likely to adversely affect Oregon spotted frog**.

6.2.1.2 Effects to Critical Habitat for Oregon spotted frog (Rana pretiosa)

As described in Chapter 3, PBFs for all areas designated as critical habitat for Oregon spotted frog that are most relevant to the Agency's action include increased sedimentation, increased water temperatures, reduced water quality, and vegetation changes resulting from the timing and intensity of livestock grazing (or, in some instances, removal of livestock grazing at locations where it maintains early seral stage habitat essential for breeding) and inadequate existing regulatory mechanisms that result in significant negative impacts such as habitat loss and modification.

As described in Section 6.2.1.1, the Agency's action may affect but is not likely to adversely affect Oregon spotted frog PBFs for reduced water quality/negative impacts to

habitat/modification (direct effects), likewise the Agency's action may affect but is not likely to adversely affect designated critical habitat for Oregon spotted frog.

6.2.2 Warner sucker (Catostomus warnerensis)

6.2.2.1 Direct and Indirect Effects Determinations for Warner sucker (Catostomus warnerensis)

Aluminum CCC values, across the range of water chemistry conditions in ecoregions within which Warner Sucker may occur, never exceeded the sucker chronic low effect threshold (i.e., EC_5) (Tier 1; Chapter 5). Indirect adverse effects are not expected for Warner sucker (HC5 < CCC) (Tier 2; Chapter 5). Therefore, the EPA has determined that the Agency's proposed action **may affect but is not likely to adversely affect the Warner Sucker**.

6.2.2.2 Effects to Critical Habitat for Warner sucker (Catostomus warnerensis)

As described in Chapter 3, PBFs for all areas designated as critical habitat for Warner Sucker include streams 15 feet to 80 feet wide with gravel-bottom shoal and riffle areas with intervening pools. Streams should have clean, unpolluted flowing water and a stable riparian zone. The streams should support a variety of aquatic insects, crustaceans, and other small invertebrates for food.

As described in Section 6.2.2.1, the Agency's action may affect but is not likely to adversely affect Warner sucker PBFs for water quality (direct effects) and prey availability (indirect effects), likewise the Agency's action **may affect but is not likely to adversely affect designated critical habitat for Warner sucker**.

6.2.3 Shortnose sucker (Chasmistes brevirostris)

6.2.3.1 Direct and Indirect Effects Determinations for Shortnose sucker (Chasmistes brevirostris)

Aluminum CCC values, across the range of water chemistry conditions in ecoregions within which Shortnose Sucker may occur, never exceeded the sucker chronic low effect threshold (i.e., EC_5) (Tier 1; Chapter 5). Indirect adverse effects are not expected for Shortnose sucker (HC5 < CCC) (Tier 2; Chapter 5). Therefore, the EPA has determined that the Agency's action **may affect but is not likely to adversely affect the Shortnose Sucker**.

6.2.3.2 Effects to Critical Habitat for Shortnose sucker (Chasmistes brevirostris)

As described in Chapter 3, the PBFs for shortnose sucker includes sufficient water quantity, complexity, depth, quality, and connectivity for survival including multiple water depths for various life stages, water temperatures <28°C (82.4°F), water pH <9.75, dissolved oxygen concentrations > 4 milligrams per liter, low levels of microcystin, and unionized ammonia <0.5 milligrams per liter. Natural (or like-natural) flow regimes are another PBF for sucker

critical habitat. Spawning and rearing habitats must have suitable substrates (e.g., gravel and cobble) in <1.3 m (4.3 feet) of water with an adequate stream velocity and emergent vegetation. Critical habitats must also provide an adequate supply of food items (e.g., insects and crustaceans).

As described in Section 6.2.3.1, the Agency's action may affect but is not likely to adversely affect Shortnose sucker PBFs for water quality (direct effects) and prey availability (indirect effects), likewise the Agency's action **may affect but is not likely to adversely affect designated critical habitat for Shortnose sucker**.

6.2.4 Lost River sucker (*Deltistes luxatus*)

6.2.4.1 Direct and Indirect Effects Determinations for Lost River sucker (Deltistes luxatus)

Aluminum CCC values, across the range of water chemistry conditions in ecoregions within which Lost River Sucker may occur, never exceeded the sucker chronic low effect threshold (i.e., EC_5) (Tier 1; Chapter 5). Indirect adverse effects are not expected for Lost River sucker (HC5 < CCC) (Tier 2; Chapter 5). Therefore, the EPA has determined that the Agency's proposed action may affect but is not likely to adversely affect the Lost River Sucker.

6.2.4.2 Effects to Critical Habitat for Lost River sucker (Deltistes luxatus)

As described in Chapter 3, the PBFs for Lost River sucker includes sufficient water quantity, complexity, depth, quality, and connectivity for survival including multiple water depths for various life stages, water temperatures <28°C (82.4°F), water pH <9.75, dissolved oxygen concentrations > 4 milligrams per liter, low levels of microcystin, and unionized ammonia <0.5 milligrams per liter. Natural (or like-natural) flow regimes are another PBF for sucker critical habitat. Spawning and rearing habitats must have suitable substrates (e.g., gravel and cobble) in <1.3 m (4.3 feet) of water with an adequate stream velocity and emergent vegetation. Critical habitats must also provide an adequate supply of food items (e.g., insects and crustaceans).

As described in Section 6.2.4.1, the Agency's action may affect but is not likely to adversely affect Lost River sucker PBFs for water quality (direct effects) and prey availability (indirect effects), likewise the Agency's action **may affect but is not likely to adversely affect designated critical habitat for Lost River sucker**.

6.2.5 Hutton tui chub (*Gila bicolor ssp.*)

6.2.5.1 Direct and Indirect Effects Determinations for Hutton tui chub (Gila bicolor ssp.)

Aluminum CCC values, across the range of water chemistry conditions in ecoregions within which Hutton tui chub may occur, never exceeded the chub chronic low effect threshold (i.e., EC_5) (Tier 1; Chapter 5). Indirect adverse effects are not expected for Hutton tui chub (HC5 <

CCC) (Tier 2; Chapter 5). Therefore, the EPA has determined that the Agency's proposed action **may affect but is not likely to adversely affect the Hutton tui chub**.

6.2.5.2 Effects to Critical Habitat for Hutton tui chub (Gila bicolor ssp.)

Not applicable (not designated in action area).

6.2.6 Borax Lake chub (Gila boraxobius)

6.2.6.1 Direct and Indirect Effects Determinations for Borax Lake chub (Gila boraxobius)

Aluminum CCC values, across the range of water chemistry conditions in ecoregions within which Borax Lake chub may occur, never exceeded the chub chronic low effect threshold (i.e., EC_5) (Tier 1; Chapter 5). Indirect adverse effects are not expected for Borax Lake chub (HC5 < CCC) (Tier 2; Chapter 5). Therefore, the EPA has determined that the Agency's action **may affect but is not likely to adversely affect the Borax Lake chub**.

6.2.6.2 Effects to Critical Habitat for Borax Lake chub (Gila boraxobius)

As described in Chapter 3, the PBFs of designated critical habitat for the Borax Lake chub that are most relevant to the Agency's action are the aquatic and terrestrial food organisms of this ecosystem.

As described in Section 6.2.6.1, the Agency's action may affect but is not likely to adversely affect Borax Lake chub PBFs for water quality (direct effects) and prey availability (indirect effects) (Section 6.2.6.1), likewise the EPA has determined that the Agency's action **may affect but is not likely to adversely affect designated critical habitat for Borax Lake chub**.

6.2.7 Lahontan cutthroat trout (Oncorhynchus clarki henshawi)

6.2.7.1 Direct and Indirect Effects Determinations for Lahontan cutthroat trout (Oncorhynchus clarki henshawi)

Aluminum CCC values, across the range of water chemistry conditions in ecoregions within which Lahontan cutthroat trout may occur, were greater than the Lahontan cutthroat trout chronic low effect threshold (i.e., EC_5) for 72.91% of the conditions (Tier 1; Chapter 5). In other words, 72.91% of the time/locations analyzed, the proposed chronic criterion would result in adverse effects for 5% of Lahontan cutthroat trout (Tier 1, LOE 1). Indirect adverse effects are not expected for Lahontan cutthroat trout (HC5 < CCC) (Tier 2, LOE 3). After examining potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect the Lahontan cutthroat trout at the Moderate LAA level**, given the WOE Score of 61.

6.2.7.2 Effects to Critical Habitat for Lahontan cutthroat trout (Oncorhynchus clarki henshawi)

Not applicable (not designated in action area).

6.2.8 Foskett speckled dace (*Rhinichthys osculus ssp*.)

6.2.8.1 Direct and Indirect Effects Determinations for Foskett speckled dace (Rhinichthys osculus ssp.)

Aluminum CCC values, across the range of water chemistry conditions in ecoregions within which Fossett speckled dace may occur, never exceeded the dace chronic low effect threshold (i.e., EC_5) (Tier 1; Chapter 5). Indirect adverse effects are not expected for Fossett speckled dace (HC5 < CCC) (Tier 2; Chapter 5). Therefore, the EPA has determined that the Agency's action may affect but is not likely to adversely affect the Fossett speckled dace.

6.2.8.2 Effects to Critical Habitat for Foskett speckled dace (Rhinichthys osculus ssp.)

Not applicable (not designated in action area).

6.2.9 Bull trout (Salvelinus confluentus)

6.2.9.1 Direct and Indirect Effects Determinations for Bull trout (Salvelinus confluentus)

Aluminum CCC values, across the range of water chemistry conditions in ecoregions within which Bull trout may occur, were greater than the bull trout chronic low effect threshold (i.e., EC_5) under 72.65% of exposure conditions (Tier 1; Chapter 5). In other words, 72.65% of the time/locations analyzed, the proposed chronic criterion would result in adverse effects for 5% of Bull trout (Tier 1, LOE 1). In addition, indirect adverse effects are expected for Bull trout (Tier 2, LOE 3) (HC5 of 370 μg/L compared to 380 μg/L, the aluminum CCC in reference waters). After examining potential exposures to all sources (LOE 2) the EPA has determined that the action is **likely to adversely affect the Bull trout at the Moderate LAA level**, given the WOE Score of 63.

6.2.9.2 Effects to Critical Habitat for Bull trout (Salvelinus confluentus)

As described in Chapter 3, the PBFs for Bull trout critical habitat that are most relevant to the Agency's action include migration habitats with minimal physical, biological, or water quality impediments between spawning, rearing, overwintering, and freshwater and marine foraging habitats, including but not limited to permanent, partial, intermittent, or seasonal barriers; an abundance of food, including terrestrial organisms of riparian origin, aquatic macroinvertebrates, and forage fish; and, sufficient water quality and quantity to sustain normal reproduction, growth, and survival.

As described in Section 6.2.9.1, Bull trout are likely to be adversely affected by the Agency's action, including impacts to water quality with implications for the growth, reproduction, and survival of Bull trout. These adverse effects include likely impacts to migration, spawning, rearing, overwintering, and foraging habitats. In addition, there are likely adverse effects for the prey of Bull trout (abundance of food). Therefore, the EPA has determined that the Agency's action is **likely to adversely affect the designated critical habitat for Bull trout**.

6.2.10 Vernal pool fairy shrimp (*Branchinecta lynchi*)

6.2.10.1 Direct and Indirect Effects Determinations for Vernal pool fairy shrimp (Branchinecta lynchi)

Aluminum CCC values were greater than the vernal pool fairy shrimp chronic low effect threshold (i.e., EC_5) under 90.26% of exposure conditions (Tier 1; Chapter 5). In other words, 90.26% of the time/locations analyzed, the proposed chronic criterion would result in adverse effects for 5% of Vernal pool fairy shrimp (Tier 1, LOE 1). Likewise, acute aluminum criteria values (CMC), across the range of water chemistry conditions in ecoregions within which Vernal pool fairy shrimp may occur, were estimated to exceed the Vernal pool fairy shrimp acute low effect threshold (i.e., LC_5) 99.58% of the time. In other words, 99.58% of the time/locations analyzed, the proposed acute criterion would result in acute adverse effects for 5% of Vernal pool fairy shrimp (Tier 1, LOE 1). Indirect adverse effects are not expected for Vernal pool fairy shrimp (HC5 < CCC) (Tier 2, LOE 3). After examining potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect the Vernal pool fairy shrimp at the High LAA level,** given the WOE Score of 78.

6.2.10.2 Effects to Critical Habitat for Vernal pool fairy shrimp (Branchinecta lynchi)

As described in Chapter 3, the PBF for Vernal pool fairy shrimp designated critical habitat that are most relevant for the Agency's action includes depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water for a minimum of 18 days, in all but the driest years; thereby providing adequate water for incubation, maturation, and reproduction.

As described in Section 6.2.10.1, Vernal pool fairy shrimp are likely to be adversely affected by the Agency's action, including impacts to water quality with implications for the growth, reproduction, and survival of Vernal pool fairy shrimp. The EPA has determined that the Agency's action is **likely to adversely affect the designated critical habitat for Vernal pool fairy shrimp**.

6.2.11 Green Sturgeon (Acipenser medirostris)

6.2.11.1 Direct and Indirect Effects Determinations for Green Sturgeon (Acipenser medirostris)

Toxicity information specific for green sturgeon was not available for Al. Therefore, the effects determinations are based on best estimates of toxicity as described in Section 5.2.6. Briefly, an LC_5 was used to compare against the CMC. For green sturgeon, the acute toxicity value (from which an LC_5 was calculated using concentration-response curves) was estimated for *Acipenser* species with a web-ICE model using toxicity data from rainbow trout. Chronic toxicity was determined by comparing EC_5 values to CCCs. For green sturgeon, the chronic toxicity value (from which an EC_5 was calculated using concentration-response curves) was based on a transformation using an acute to chronic ratio (ACR). The ACR was calculated from a fathead minnow study in which reduced biomass of exposed hatchlings was the chronic effect (Section 5.2.6.1). Therefore, when a green sturgeon EC_5 in one water chemistry is less than the corresponding CCC, EPA expects a greater than insignificant (i.e., > 5%) reduction to green sturgeon biomass in that situation.

Aluminum CCC values, across the range of water chemistry conditions in ecoregions within which Green sturgeon may occur, never exceeded the Green sturgeon chronic low effect threshold (i.e., EC_5) (Tier 1; Chapter 5). However, acute aluminum criteria values (CMC), across the range of water chemistry conditions in ecoregions within which Green sturgeon may occur, were estimated to exceed the Green sturgeon acute low effect threshold (i.e., LC_5) 16.34% of the time. In other words, for 16.34% of the time/locations analyzed, the proposed acute criterion would result in mortality for 5% of Green sturgeon (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 0.82% (0.1634 * 0.05) increase in mortality across the green sturgeon population in Oregon if exposed to Al at CMC concentrations. Recall that LC_5 values are an outer limit of what can be reliably statistically quantified and therefore EPA considers 5th centile (See Section 5.1.2.1.1). Indirect adverse effects are not expected for Green sturgeon because although the prey based HC5 was less than the CCC for 10.91% of relevant water chemistries, this finding represents an insignificant (<1%) effect to the prey base (i.e., 10.91% * 5% * 20% = 0.11% reduction in prey, See Section 5.1.1.2) (Tier 2, LOE 3). After examining potential exposures to all sources (LOE 2), the EPA has determined that the action (CMC criteria) is **likely to adversely affect the Green sturgeon at the Low LAA level**, given the WOE Score of 26 (CCC criteria WOE Score was 0).

6.2.11.2 Effects to Critical Habitat for Green Sturgeon (Acipenser medirostris)

As described in Chapter 3.0, the following PBFs are most relevant to the Agency's action for freshwater and estuarine green sturgeon habitats:

- Abundant prey for all life stages; and
- Suitable water quality (i.e., temperature, salinity, dissolved oxygen, and "other chemical characteristics") for normal behavior, growth, and viability of all life stages;

As described in Section 6.2.11.1, the Agency's action is likely to adversely affect water quality for the survival, growth and viability of of Green Sturgeon (assessed via the direct effects

analysis). Furthermore, indirect effects of the action on green sturgeon prey may be affected but not at a level expected to induce a significant reduction in prey availability (Section 5.5.2). Therefore, given the expected direct effects to green sturgeon through impacts on water quality (Al concentrations > EC_5 for sturgeon), the EPA has determined that the Agency's action is **likely to adversely affect the designated critical habitat for Green Sturgeon**.

6.2.12 Eulachon (Thaleichthys pacificus)

6.2.12.1 Direct and Indirect Effects Determinations for Eulachon (Thaleichthys pacificus)

Toxicity information specific for eulachon was not available for Al. Therefore, the effects determinations are based on best estimates of toxicity as described in Section 5.2.7. Briefly, an LC_5 was used to compare against the CMC. For eulachon, the acute toxicity value (from which an LC_5 was calculated using concentration-response curves) was estimated from a Salmoniformes acute value using toxicity data from rainbow trout, Atlantic salmon, and brook trout. Chronic toxicity was determined by comparing EC_5 values to CCCs. For eulachon, the chronic toxicity value (from which an EC_5 was calculated using concentration-response curves) was based on three ELS tests in which the biomass of Atlantic salmon and brook trout was reduced (Section 5.2.7.2.1). Therefore, when a eulachon EC_5 in a given water chemistry is less than the corresponding CCC, EPA expects a greater than insignificant (i.e., > 5%) reduction to eulachon biomass in that situation.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1. Aluminum CCC values were greater than the Pacific eulachon EC_5 under 94.70% of exposure conditions. In other words, in 94.70% of the water chemistry conditions relevant to eulachon, the proposed chronic criterion would result in a 5% reduction in eulachon biomass (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 4.74% (0.9470 * 0.05) decrease in biomass across the eulachon population in Oregon if exposed at CCC concentrations. Insignificant indirect adverse effects are expected for the eulachon given that 76.93% of the prey-based HC5 values were less than the corresponding CCC (Tier 2, LOE 3). This finding is insignificant because it represents a less than 1% decrease in eulachon prey availability (i.e., 76.93% * 5% * 20% = 0.77%). After examining potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect eulachon ESU at the High LAA level**, given the WOE Score of 78.

6.2.12.2 Effects to Critical Habitat for Eulachon (Thaleichthys pacificus)

As described in Chapter 3, the PBFs for eulachon for freshwater spawning and incubation sites include water flow, quality, and temperature conditions; spawning and incubation substrates; and migratory access. PBFs for freshwater and estuarine migration corridors include waters free of obstruction; specific water flow, quality, and temperature conditions (for supporting larval and

adult mobility); and abundant prey items (for supporting larval feeding after the yolk sac is depleted).

The PBFs that may be affected by this action include 1) water quality for freshwater spawning and incubation sites, 2) water quality for freshwater and estuarine migration corridors, and 3) abundant prey items. As described in Chapter 6.2.12.1., this action is likely to adversely affect the growth, health, migration, and propagation of eulachon through reductions in fry biomass (assessed via the direct effects analysis). Therefore, the EPA has determined that the Agency's action is **likely to adversely affect the designated critical habitat for eulachon**.

6.2.13 Southern Resident Killer Whales (*Orcinus Orca*) (SRKW)

As established in Chapter 3, SRKW predate upon threatened and endangered species present in the action area, particularly Chinook salmon.

6.2.13.1 Direct and Indirect Effects Determinations for Southern Resident Killer Whales (Orcinus Orca)

Because this action is unlikely to affect water quality outside of the freshwaters of Oregon, including marine and estuarine waters where SRKW reside, there are no known direct effects of this action for the SRKW ESU. Furthermore, even if Al from freshwater sources entered marine waters where SRKW occur, oral exposure to contaminated water is not expected because marine mammals do not drink marine water to any significant degree (Ortiz, 2001). However, due to the adverse effects to SRKW prey species as described in Section 5.2.4 and 5.5.18 there are likely to be indirect adverse effects of this action on SRKW. As discussed in Section 6.2.16, Upper Columbia River and Lower Columbia River spring-run and Snake River fall and spring/summer run Chinook are likely to be adversely affected by this action. Because these Chinook ESUs comprise a large proportion of the diet of SRKW (per Chapter 3), this action is likewise likely to adversely affect SRKW. Approximately 2.5% of SRKW prey biomass could be lost as a consequence of this action. Furthermore, reductions in growth of juvenile salmonids has been found to translate to reductions in their survival (Mebane & Arthaud, 2010), which may affect prey abundance through that mechanism; but data are not available for EPA to reliably quantify this effect. As such, EPA has made its effects determination based on reductions to SRKW prey biomass and the indirect effect that may stem from such a reduction. Therefore, the agency's action is **likely to adversely affect Southern Resident Killer Whales.**

6.2.13.2 Effects to Critical Habitat for Southern Resident Killer Whales (Orcinus Orca)

6.2.13.2.1 Designated Critical Habitat (Puget Sound and Juan de Fuca Strait) Effects Determination

Because the "quantity and quality of prey" PBF for SRKW is likely to be adversely affected per Section 6.2.13.1, the Agency finds that the DCH for SRKW is likely to be adversely affected.

6.2.13.2.2 Proposed Critical Habitat (Eastern Pacific Coast) Effects Determination (Conference Opinion)

Because the "quantity and quality of prey" PBF for SRKW is likely to be adversely affected per Section 6.2.13.1, the Agency's action is **likely to be adversely affected that the proposed critical habitat for SRKW**.

6.2.14 Coho Salmon (Oncorhynchus kisutch)

6.2.14.1 Direct and Indirect Effects Determinations for Coho Salmon (Oncorhynchus kisutch)

6.2.14.1.1 Lower Columbia Coho ESU

Toxicity information specific for coho salmon was not available for Al. However, high quality acute toxicity data for *Oncorhynchus* were available and used for the coho assessment (See Section 5.2.2.1). The acute toxicity value was converted to an LC5 using concentration-response curves. Concentration-response data from two brook trout tests were used for LC5 estimation, as no data specific to coho were available. Then, as with all acute toxicity (mortality) assessments, the LC_5 was used to compare against the CMC.

Chronic toxicity was determined by comparing EC_5 values to CCCs. For coho, the chronic toxicity value (from which an EC_5 was calculated using concentration-response curves) was based on three ELS tests in which the biomass of Atlantic salmon and brook trout was reduced (Section 5.2.1.2.1). Therefore, when a coho EC_5 in a given water chemistry is less than the corresponding CCC, EPA expects a greater than insignificant (i.e., > 5%) reduction to coho biomass in that situation.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1. Aluminum CCC values were greater than the coho EC_5 under 94.70% of exposure conditions. In other words, in 94.70% of the water chemistry conditions relevant to coho Lower Columbia River, the proposed chronic criterion would result in a 5% reduction in coho biomass (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 4.74% (0.9470 * 0.05) decrease in biomass across the coho Lower Columbia River ESU in Oregon if exposed at CCC concentrations. Insignificant indirect adverse effects are expected for the coho given that 94.45% of the prey-based HC5 values and just 0.71% of the lowest GMCV in the prey-based SSD were less than the corresponding CCC (Tier 2, LOE 3). This finding is insignificant because it represents a less than 1% decrease in eulachon prey availability (i.e., 94.45% * 5% * 20% = 0.94%). Further supporting a finding of an insignificant indirect effect of the action is that the GMCV of the most sensitive prey item was higher than the CCC more than 99% of the time.

After examining direct and indirect effects and all potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect coho Lower Columbia River ESU at the High LAA level**, given the WOE Score of 78.

6.2.14.1.2 Oregon Coast ESU

See Section 6.2.14.1.1 coho Lower Columbia River ESU for toxicity and biological information relevant to coho Oregon Coast ESU as the same data were used for both ESUs.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1. Aluminum CCC values were greater than the coho EC_5 under 94.49% of exposure conditions. In other words, in 94.49% of the water chemistry conditions relevant to coho Oregon Coast, the proposed chronic criterion would result in a 5% reduction in coho biomass (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 4.72% (0.9449 * 0.05) decrease in biomass across the coho Oregon Coast ESU in Oregon if exposed at CCC concentrations. Insignificant indirect adverse effects are expected for the coho given that 94.45% of the prey-based HC5 values and just 0.71% of the lowest GMCV in the prey-based SSD were less than the corresponding CCC (Tier 2, LOE 3). This finding is insignificant because it represents a less than 1% decrease in eulachon prey availability (i.e., 94.45% * 5% * 20% = 0.94%). Further supporting a finding of an insignificant indirect effect of the action is that the GMCV of the most sensitive prey item was higher than the CCC more than 99% of the time.

After examining direct and indirect effects and potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect Oregon Coast coho salmon at the High LAA level**, given the WOE Score of 80.

6.2.14.1.3 Southern Oregon Northern California Coast ESU

See Section 6.2.14.1.1 coho Lower Columbia River ESU for toxicity and biological information relevant to coho Southern Oregon Northern California ESU as the same data were used for both ESUs.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1. Aluminum CCC values were greater than the coho EC_5 under 93.73% of exposure conditions. In other words, in 93.73% of the water chemistry conditions relevant to this coho ESU, the proposed chronic criterion would result in a 5% reduction in coho biomass (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 4.68% (0.9373 * 0.05) decrease in biomass

across the coho Oregon Coast ESU in Oregon if exposed at CCC concentrations. Insignificant indirect adverse effects are expected for the coho given that 94.45% of the prey-based HC5 values and just 0.71% of the lowest GMCV in the prey-based SSD were less than the corresponding CCC (Tier 2, LOE 3). This finding is insignificant because it represents a less than 1% decrease in eulachon prey availability (i.e., 94.45% * 5% * 20% = 0.94%). Further supporting a finding of an insignificant indirect effect of the action is that the GMCV of the most sensitive prey item was higher than the CCC more than 99% of the time.

After examining direct and indirect effects and potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect Southern Oregon Northern California Coast coho salmon at the High LAA level,** given the WOE Score of 78.

6.2.14.2 Effects to Critical Habitat for Coho Salmon (Oncorhynchus kisutch)

The PBFs for coho salmon are the same for all ESUs, and they are described in Table 3-1. The PBFs that may be affected by the Agency's action include water quality protective of freshwater spawning, freshwater rearing, and freshwater migration. As described in Chapter 6.2.14.1, all coho ESUs are likely to be adversely affected by the Agency's action through impacts to water quality for freshwater spawning, rearing and migration (assessed via the direct effects analysis). Therefore, the EPA has determined that the Agency's action is **likely to adversely affect the designated critical habitat for all coho ESUs with designated critical habitat in the Action Area**.

6.2.15 Columbia River Chum Salmon (*Oncorhynchus keta*)

6.2.15.1 Direct and Indirect Effects Determinations for Chum Salmon (Oncorhynchus keta)

See Section 6.2.14.1.1 coho Lower Columbia River ESU for toxicity and biological information relevant to Columbia River chum ESU as the same data were used for both ESUs.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1. Aluminum CCC values were greater than the chum EC_5 under 93.46% of exposure conditions. In other words, in 93.46% of the water chemistry conditions relevant to this chum ESU, the proposed chronic criterion would result in a 5% reduction in chum biomass (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 4.70% (0.9346 * 0.05) decrease in biomass across the Columbia River chum ESU in Oregon if exposed at CCC concentrations. Insignificant indirect adverse effects are expected for the chum given that 19.86% of the prey-based HC5 values and just 0% of the lowest GMCV in the prey-based SSD were less than the corresponding CCC (Tier 2, LOE 3). This finding is insignificant because it represents a less than 1% decrease in eulachon prey availability (i.e., 19.86% * 5% * 20% = 0.20%). Further supporting a finding of an insignificant indirect effect of the action is that the GMCV of the most sensitive prey item was higher than the CCC 100% of the time.

After examining direct and indirect effects and after examining potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect chum salmon at the High LAA level,** given the WOE Score of 78.

6.2.15.2 Effects to Critical Habitat for Chum Salmon (Oncorhynchus keta)

The PBFs for chum salmon are described in Table 3-1. The PBFs that may be affected by the Agency's action include water quality protective of freshwater spawning, freshwater rearing, and freshwater migration. As described in Section 6.2.16.1, chum salmon are likely to be adversely affected by the Agency's action through impacts to water quality for freshwater spawning, rearing and migration (assessed via the direct effects analysis). Therefore, the EPA has determined that the Agency's action is **likely to adversely affect the designated critical habitat for chum salmon**.

6.2.16 Chinook Salmon (Oncorhynchus tshawytscha)

6.2.16.1 Direct and Indirect Effects Determinations for Chinook Salmon (Oncorhynchus tshawytscha)

6.2.16.1.1 Upper Columbia River Spring-Run ESU

See Section 6.2.14.1.1 coho Lower Columbia River ESU for toxicity and biological information relevant to Upper Columbia River (spring) chinook ESU as the same data were used for both ESUs.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1. Aluminum CCC values were greater than the chinook EC_5 under 93.46% of exposure conditions. In other words, in 93.46% of the water chemistry conditions relevant to this chinook ESU, the proposed chronic criterion would result in a 5% reduction in chinook biomass (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 4.70% (0.9346 * 0.05) decrease in biomass across this chinook ESU in Oregon if exposed at CCC concentrations. Insignificant indirect adverse effects are expected for the chinook given that 22.19% of the prey-based HC5 values and 0% of the lowest GMCV in the prey-based SSD were less than the corresponding CCC (Tier 2, LOE 3). This finding is insignificant because it represents a less than 1% decrease in chinook prey availability (i.e., 22.19% * 5% * 20% = 0.22%). Further supporting a finding of an insignificant indirect effect of the action is that the GMCV of the most sensitive prey item was higher than the CCC 100% of the time.

After examining direct and indirect effects and potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect the UCR spring-run Chinook salmon ESU at the High LAA level,** given the WOE Score of 78.

6.2.16.1.2 Lower Columbia River ESU

See Section 6.2.14.1.1 coho Lower Columbia River ESU for toxicity and biological information relevant to Lower Columbia River chinook ESU as the same data were used for both ESUs.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1. Aluminum CCC values were greater than the chinook EC_5 under 94.70% of exposure conditions. In other words, in 94.70% of the water chemistry conditions relevant to this chinook ESU, the proposed chronic criterion would result in a 5% reduction in chinook biomass (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 4.73% (0.9470 * 0.05) decrease in biomass across this chinook ESU in Oregon if exposed at CCC concentrations. Insignificant indirect adverse effects are expected for the chinook given that 22.19% of the prey-based HC5 values and 0% of the lowest GMCV in the prey-based SSD were less than the corresponding CCC (Tier 2, LOE 3). This finding is insignificant because it represents a less than 1% decrease in chinook prey availability (i.e., 22.19% * 5% * 20% = 0.22%). Further supporting a finding of an insignificant indirect effect of the action is that the GMCV of the most sensitive prey item was higher than the CCC 100% of the time.

After examining direct and indirect effects and potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect the LCR Chinook salmon ESU at the High LAA level,** given the WOE Score of 78.

6.2.16.1.3 Snake River Spring/Summer-Run ESU

See Section 6.2.14.1.1 coho Lower Columbia River ESU for toxicity and biological information relevant to Snake River (spring/summer) chinook ESU as the same data were used for both ESUs.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1. Aluminum CCC values were greater than the chinook EC_5 under 92.08% of exposure conditions. In other words, in 94.70% of the water chemistry conditions relevant to this chinook ESU, the proposed chronic criterion would result in a 5% reduction in chinook biomass (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 4.60% (0.9208 * 0.05) decrease in biomass across this chinook ESU in Oregon if exposed at CCC concentrations. Insignificant indirect adverse effects are expected for the chinook given that 22.19% of the prey-based HC5 values and 0% of the lowest GMCV in the prey-based SSD were less than the corresponding CCC (Tier 2, LOE 3). This finding is insignificant because it represents a less than 1% decrease in chinook prey availability (i.e., 22.19% * 5% * 20% = 0.22%). Further supporting a finding of an insignificant indirect effect of the action is that the GMCV of the most sensitive prey item was higher than the CCC 100% of the time.

After examining direct and indirect effects and potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect the SR Spring/Summer-run ESU at the High LAA level,** given the WOE Score of 78.

6.2.16.1.4 Snake River Fall-Run ESU

See Section 6.2.14.1.1 coho Lower Columbia River ESU for toxicity and biological information relevant to Snake River (fall) chinook ESU as the same data were used for both ESUs.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1. Aluminum CCC values were greater than the chinook EC_5 under 92.08% of exposure conditions. In other words, in 94.70% of the water chemistry conditions relevant to this chinook ESU, the proposed chronic criterion would result in a 5% reduction in chinook biomass (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 4.60% (0.9208 * 0.05) decrease in biomass across this chinook ESU in Oregon if exposed at CCC concentrations. Insignificant indirect adverse effects are expected for the chinook given that 22.19% of the prey-based HC5 values and 0% of the lowest GMCV in the prey-based SSD were less than the corresponding CCC (Tier 2, LOE 3). This finding is insignificant because it represents a less than 1% decrease in chinook prey availability (i.e., 22.19% * 5% * 20% = 0.22%). Further supporting a finding of an insignificant indirect effect of the action is that the GMCV of the most sensitive prey item was higher than the CCC 100% of the time.

After examining direct and indirect effects and potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect the SR Fall-run Chinook salmon ESU at the High LAA level,** given the WOE Score of 78.

6.2.16.1.5 Upper Willamette River ESU

See Section 6.2.14.1.1 coho Lower Columbia River ESU for toxicity and biological information relevant to Upper Willamette River chinook ESU as the same data were used for both ESUs.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1. Aluminum CCC values were greater than the chinook EC_5 under 96.47% of exposure conditions. In other words, in 94.70% of the water chemistry conditions relevant to this chinook ESU, the proposed chronic criterion would result in a 5% reduction in chinook biomass (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 4.82% (0.9647 * 0.05) decrease in biomass across this chinook ESU in Oregon if exposed at CCC concentrations. Insignificant indirect adverse effects are expected for the chinook given that 22.19% of the prey-based HC5 values and 0% of the lowest GMCV in the prey-based SSD were less than the corresponding CCC (Tier 2, LOE 3). This finding is insignificant because it represents a less than 1% decrease in chinook prey availability (i.e., 22.19% * 5% * 20% = 0.22%). Further supporting a finding of

an insignificant indirect effect of the action is that the GMCV of the most sensitive prey item was higher than the CCC 100% of the time.

After examining direct and indirect effects and potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect the Upper Willamette Chinook salmon ESU at the High LAA level**, given the WOE Score of 80.

6.2.16.2 Effects to Critical Habitat for Chinook Salmon (Oncorhynchus tshawytscha)

The PBFs for Chinook are the same for all ESUs, and they are described in Table 3-1. The PBFs that may be affected by the Agency's action include water quality protective of freshwater spawning, freshwater rearing, and freshwater migration. As described in Section 6.2.16.1, all Chinook ESUs are likely to be adversely affected by the Agency's action through impacts to water quality for freshwater spawning, rearing and migration (assessed via the direct effects analysis). Therefore, the EPA has determined that the Agency's action is **likely to adversely affect the designated critical habitat for all Chinook ESUs**.

6.2.17 Sockeye Salmon (Oncorhynchus nerka)

6.2.17.1 Direct and Indirect Effects Determinations for Sockeye Salmon (Oncorhynchus nerka)

See Section 6.2.14.1.1 coho Lower Columbia River ESU for toxicity and biological information relevant to sockeye salmon ESU as the same data were used for both ESUs.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1. Aluminum CCC values were greater than the sockeye EC_5 under 92.08% of exposure conditions. In other words, in 94.70% of the water chemistry conditions relevant to sockeye, the proposed chronic criterion would result in a 5% reduction in sockeye biomass (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 4.60% (0.9208 * 0.05) decrease in biomass across sockeye locations in Oregon if exposed at CCC concentrations. Insignificant indirect adverse effects are expected for the sockeye given that 22.19% of the prey-based HC5 values and 0% of the lowest GMCV in the prey-based SSD were less than the corresponding CCC (Tier 2, LOE 3). This finding is insignificant because it represents a less than 1% decrease in sockeye prey availability (i.e., 22.19% * 5% * 20% = 0.22%). Further supporting a finding of an insignificant indirect effect of the action is that the GMCV of the most sensitive prey item was higher than the CCC 100% of the time.

After examining direct and indirect effects and potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect the sockeye salmon ESU at the High LAA level**, given the WOE Score of 78.

6.2.17.2 Effects to Critical Habitat for Sockeye Salmon (Oncorhynchus nerka)

The PBFs for Sockeye are described in Table 3-1. The PBFs that may be affected by the Agency's action include water quality protective of freshwater spawning, freshwater rearing, and freshwater migration. As described in Section 6.2.17.1, sockeye are likely to be adversely affected by the Agency's action through impacts to water quality for freshwater spawning, rearing and migration (assessed via the direct effects analysis). Therefore, the EPA has determined that the Agency's action is **likely to adversely affect the designated critical habitat for sockeye salmon**.

6.2.18 Steelhead (Oncorhynchus mykiss)

6.2.18.1 Direct and Indirect Effects Determinations for Steelhead (Oncorhynchus mykiss)

6.2.18.1.1 Upper Columbia River ESU

See Section 6.2.14.1.1 coho Lower Columbia River ESU for toxicity and biological information relevant to Upper Columbia River steelhead ESU as the same data were used for both ESUs.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1. Aluminum CCC values were greater than the steelhead EC_5 under 93.46% of exposure conditions. In other words, in 93.46% of the water chemistry conditions relevant to this steelhead ESU, the proposed chronic criterion would result in a 5% reduction in steelhead biomass (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 4.67% (0.9346 * 0.05) decrease in biomass across this steelhead ESU in Oregon if exposed at CCC concentrations. Insignificant indirect adverse effects are expected for the chinook given that 20.09% of the prey-based HC5 values and 0% of the lowest GMCV in the prey-based SSD were less than the corresponding CCC (Tier 2, LOE 3). This finding is insignificant because it represents a less than 1% decrease in steelhead prey availability (i.e., 20.09% * 5% * 20% = 0.20%). Further supporting a finding of an insignificant indirect effect of the action is that the GMCV of the most sensitive prey item was higher than the CCC 100% of the time.

After examining direct and indirect effects and potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect the Upper Columbia Steelhead ESU at the High LAA level**, given the WOE Score of 78.

6.2.18.1.2 Middle Columbia River ESU

See Section 6.2.14.1.1 coho Lower Columbia River ESU for toxicity and biological information relevant to Middle Columbia River steelhead ESU as the same data were used for both ESUs.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% *

5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1. Aluminum CCC values were greater than the steelhead EC_5 under 92.08% of exposure conditions. In other words, in 92.08% of the water chemistry conditions relevant to this steelhead ESU, the proposed chronic criterion would result in a 5% reduction in steelhead biomass (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 4.60% (0.9208 * 0.05) decrease in biomass across this steelhead ESU in Oregon if exposed at CCC concentrations. Insignificant indirect adverse effects are expected for the steelhead given that 20.09% of the prey-based HC5 values and 0% of the lowest GMCV in the prey-based SSD were less than the corresponding CCC (Tier 2, LOE 3). This finding is insignificant because it represents a less than 1% decrease in steelhead prey availability (i.e., 20.09% * 5% * 20% = 0.20%). Further supporting a finding of an insignificant indirect effect of the action is that the GMCV of the most sensitive prey item was higher than the CCC 100% of the time.

After examining direct and indirect effects and potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect the Middle Columbia Steelhead ESU at the High LAA,** level given the WOE Score of 78.

6.2.18.1.3 Lower Columbia River ESU

See Section 6.2.14.1.1 coho Lower Columbia River ESU for toxicity and biological information relevant to Lower Columbia River steelhead ESU as the same data were used for both ESUs.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1. Aluminum CCC values were greater than the steelhead EC_5 under 94.70% of exposure conditions. In other words, in 94.70% of the water chemistry conditions relevant to this steelhead ESU, the proposed chronic criterion would result in a 5% reduction in steelhead biomass (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 4.73% (0.9470 * 0.05) decrease in biomass across this steelhead ESU in Oregon if exposed at CCC concentrations. Insignificant indirect adverse effects are expected for the steelhead given that 20.09% of the prey-based HC5 values and 0% of the lowest GMCV in the prey-based SSD were less than the corresponding CCC (Tier 2, LOE 3). This finding is insignificant because it represents a less than 1% decrease in steelhead prey availability (i.e., 20.09% * 5% * 20% = 0.20%). Further supporting a finding of an insignificant indirect effect of the action is that the GMCV of the most sensitive prey item was higher than the CCC 100% of the time.

After examining direct and indirect effects and examining exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect the Lower Columbia Steelhead ESU at the High LAA level,** given the WOE Score of 78.

6.2.18.1.4 Snake River ESU

See Section 6.2.14.1.1 coho Lower Columbia River ESU for toxicity and biological information relevant to Snake Basin River steelhead ESU as the same data were used for both ESUs.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1. Aluminum CCC values were greater than the steelhead EC_5 under 92.08% of exposure conditions. In other words, in 92.08% of the water chemistry conditions relevant to this steelhead ESU, the proposed chronic criterion would result in a 5% reduction in steelhead biomass (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 4.60% (0.9208 * 0.05) decrease in biomass across this steelhead ESU in Oregon if exposed at CCC concentrations. Insignificant indirect adverse effects are expected for the steelhead given that 20.09% of the prey-based HC5 values and 0% of the lowest GMCV in the prey-based SSD were less than the corresponding CCC (Tier 2, LOE 3). This finding is insignificant because it represents a less than 1% decrease in steelhead prey availability (i.e., 20.09% * 5% * 20% = 0.20%). Further supporting a finding of an insignificant indirect effect of the action is that the GMCV of the most sensitive prey item was higher than the CCC 100% of the time.

After examining direct and indirect effects and potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect the Snake River Steelhead ESU at the High LAA level,** given the WOE Score of 78.

6.2.18.1.5 Upper Willamette River ESU

See Section 6.2.14.1.1 coho Lower Columbia River ESU for toxicity and biological information relevant to Upper Willamette River steelhead ESU as the same data were used for both ESUs.

Aluminum CMC values never exceeded LC_5 values across Oregon water chemistries and so mortality effects are expected to be low. That is, the action would result in a 0% chance (0% * 5% = 0%) of causing mortality that exceeds a level that is measurable (i.e., LC5 is a minimum statistically detectable mortality rate in a toxicity test, See Section 5.1.2.1.1. Aluminum CCC values were greater than the steelhead EC_5 under 96.47% of exposure conditions. In other words, in 96.47% of the water chemistry conditions relevant to this steelhead ESU, the proposed chronic criterion would result in a 5% reduction in steelhead biomass (Tier 1, LOE 1). It may also be stated from this analysis that the action would result in a 4.82% (0.9647 * 0.05) decrease in biomass across this steelhead ESU in Oregon if exposed at CCC concentrations. Insignificant indirect adverse effects are expected for the steelhead given that 20.09% of the prey-based HC5 values and 0% of the lowest GMCV in the prey-based SSD were less than the corresponding CCC (Tier 2, LOE 3). This finding is insignificant because it represents a less than 1% decrease in steelhead prey availability (i.e., 20.09% * 5% * 20% = 0.20%). Further supporting a finding of an insignificant indirect effect of the action is that the GMCV of the most sensitive prey item was higher than the CCC 100% of the time.

After examining direct and indirect effects and potential exposures to all sources (LOE 2), the EPA has determined that the action is **likely to adversely affect the Upper Willamette River Steelhead ESU at the High LAA level,** given the WOE Score of 80.

6.2.18.2 Effects to Critical Habitat for Steelhead (Oncorhynchus mykiss)

The PBFs for Steelhead are the same for all ESUs, and they are described in Table 3-1. The PBFs that may be affected by the Agency's action include water quality protective of freshwater spawning, freshwater rearing, and freshwater migration. As described in Chapter 6.2.18.1, all Steelhead ESUs are likely to be adversely affected by the Agency's action through impacts to water quality for freshwater spawning, rearing and migration (assessed via the direct effects analysis). Therefore, the EPA has determined that the Agency's action is **likely to adversely affect the designated critical habitat for all Steelhead ESUs**.

Table 6-2. Species effects determinations for the CMC.

	Relative Weights	Max Points	Green sturgeon	Eulachon	Coho-LCR	Coho-OC	Coho-SONCC	Chum	Chinook-UCRS	Chinook-SRSS	Chinook-SRF	Chinook-UWR	Chinook-LC	Sockeye	Steelhead-UCR	Steelhead-SRB	Steelhead-MC	Steelhead-UWR	Steelhead-LCR	SRKW	OR spotted frog	Warner sucker	Shortnose sucker	Lost River sucker	Hutton tui chub	Borax lake chub	Lahontan	Fockett	Bull trout	VPFS
LOE1	0.75	4	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
LOE2	0.10	4	3																											1
LOE3	0.15	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0
Total points		12	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	5
Score		4	1.05	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.6	0	0	0	0	0	0	0	0	0	3.1
Possible Score		4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4
WOE Score		100	26	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	15	0	0	0	0	0	0	0	0	0	78
Interpretation		High	Low	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	Very Low	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	High

Table 6-2. Species effects determinations for the CCC.

	Relative Weights	Max Points	Green sturgeon	Eulachon	Coho-LCR	Coho-OC	Coho-SONCC	Chum	Chinook-UCRS	Chinook-SRSS	Chinook-SRF	Chinook-UWR	Chinook-LC	Sockeye	Steelhead-UCR	Steelhead-SRB	Steelhead-MC	Steelhead-UWR	Steelhead-LCR	SRKW	OR spotted frog	Warner sucker	Shortnose sucker	Lost River sucker	Hutton tui chub	Borax lake chub	Lahontan	Foskett	Bull trout	VPFS
LOE1	0.75	4	0	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	0	0	0	0	0	0	0	3	0	3	4
LOE2	0.10	4		1	1	2	1	1	1	1	1	2	1	1	1	1	1	2	1								2		1	1
LOE3	0.15	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0
Total points		12	0	5	5	6	5	5	5	5	5	6	5	5	5	5	5	6	5	4	0	0	0	0	0	0	5	0	4	5
Score		4	0	3.1	3.1	3.2	3.1	3.1	3.1	3.1	3.1	3.2	3.1	3.1	3.1	3.1	3.1	3.2	3.1	0.6	0	0	0	0	0	0	2.45	0	2.35	3.1
Possible Score		4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4	4
WOE Score		100	0	78	78	80	78	78	78	78	78	80	78	78	78	78	78	80	78	15	0	0	0	0	0	0	61	0	59	78
Interpretation		High	NLAA	High	High	High	High	High	High	High	High	High	High	High	High	High	High	High	High	Very Low	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	Moderate	NLAA	Moderate	High

6.3 Cumulative Effects and Interdependent/Interrelated Actions, and Ongoing Environmental Conditions

6.3.1 Cumulative Effects

Cumulative effects include the effects of future state, tribal, local, or private actions on endangered or threatened species or critical habitat that are reasonably certain to occur in the action area considered in this biological evaluation. Future federal actions or actions on federal lands that are not related to the proposed action are not considered in this chapter. Future anticipated non - federal actions that may occur in or near surface waters in the State of Oregon include timber harvest, grazing, mining, agriculture, urban development, municipal and industrial wastewater discharges, road building, sand and gravel operations, introduction of nonnative fishes, off-road vehicle use, fishing, hiking, and camping. These non-federal actions are likely to continue having adverse effects on the endangered and threatened species, and their habitat.

There are also non - federal actions likely to occur in or near surface waters in the State of Oregon that are likely to have beneficial effects on the endangered and threatened species. These include implementation of water conservation and re-use programs, toxic reduction and pollution prevention programs, riparian improvement measures, best management practices associated with timber harvest, grazing, agricultural activities, urban development, road building and abandonment, recreational activities, and other non-point source pollution controls.

6.3.2 Interdependent/Interrelated Actions (consequences of the action)

Interdependent actions are defined as actions with no independent use apart from the proposed action. Interrelated actions include those that are part of a larger action and depend on the larger action for justification.

In recent comments to EPA (NOAA letter to EPA dated October 19, 2019), NMFS raised a concern that CWA implementation, i.e., establishment of NPDES permitted mixing zones, are potentially interrelated or interdependent upon the Agency's action. However, considering that ODEQ already has the discretionary authority to grant mixing zones, subject to the regulatory restrictions at OAR 310-041-0053 and ODEQ guidance, the criteria approval is not the but for cause of any future mixing zone authorization. The fact that mixing zone authorizations are discretionary state permitting decisions that are based on many factors also raises questions about whether future authorizations are reasonably certain to occur. The EPA has provided qualitative assessments of mixing zones relying on relevant available information consistent with 50 C.F.R. 402.14(c)(6) to NMFS (Appendix F) as baseline context for NMFS's reference in this consultation. However, the Agency has determined that there are no interdependent or interrelated actions expected as a result of the proposed criteria for aluminum.

6.3.3 Climate Change

Climate change has and will continue to result in broad-ranging effects on Oregon freshwaters, with implications for the ability of threatened and endangered species to survive and propagate. Observed and predicted impacts to NW habitats include:

- Warmer air and water temperatures;
- More frequent and severe extreme weather events (e.g., heavy rainfall, flooding, high temperatures, and drought) and wildfires;
- Increasing winter temperature shifting snowfall to rain and timing of melting snow and ice to earlier in the season;
- Rising sea levels; and
- Increased marine salinity and reduced marine pH (acidification)

These impacts are predicted to continue (and in some cases accelerate or under certain cicumstances), which may substantially affect the ESA-listed species and their habitats considered in this BEA.

The changing climate is expected to affect species and habitats in many different ways. For example, the distribution of shoreline-dependent species will shift as a consequence of rising sea levels. The California Current Large Marine Ecosystem will see northern shifting isotherms, which will shift species distributions as well as increase ocean stratification, impeding nutrient transport and plankton production (NOAA Fisheries, 2016a).

Migration patterns of salmon are also expected to be affected by changing water temperatures (WDoE, 2012). As a result of increasing typhoon frequency, changing water temperatures, and increasing oceanic salinity (each related to climate change), green sea turtles (Chelonia mydas) may experience threats, including (but not limited to), nest failure and an unstable prey base (Duarte, 2002; Short & Neckles, 1999).

The reduction of prey and alteration of food webs caused by climate change and ocean acidification have the potential to impact ESA-listed species. For example, survival of emigrating fall-run LCR coho salmon (Oncorhynchus kisutch) smolts and the abundance of LCR Chinook salmon (O. tshawytscha) have both been impacted by reduced prey and altered food webs (related to climate change) (NOAA Fisheries, 2015b). Climate change is also cited as the most serious threat to the southern distinct population segment (DPS) of Pacific eulachon (Thaleichthys pacificus), which are dependent on cool water habitat and the invertebrate communities therein (Gustafson et al., 2010; Willson et al., 2006).

It is predicted that changes in winter precipitation regimes from snow to rain for many watersheds will occur, further taxing threatened and endangered salmonids that rely on summer snow-melt for cooling during migration. In addition to changes in flow regimes, warmer stream temperatures projected for 2040 and 2080 (USGCRP, 2018) for the Pacific Northwest indicate that thermal stress will continue to threaten the continued existence of coldwater fish and the threatened and endangered species that rely on coldwater fish to survive. Lower flows are also projected for many river systems, with less dilution available for toxic pollutants at certain times of year. The combination of stressors is likely to result in further adverse effects for threatened and endangered species sensitive to reduced cold water availability, reduced prey quality and quantity, as well as the increased likelihood of exposure to excess toxic pollutant concentrations.

6.3.4 Implementation of water quality criteria

In addition to aluminum, there are many water quality criteria for toxic pollutants that are not considered herein that may have synergistic or antagonistic effects with aluminum concentrations allowable by the Agency's action. Due to a paucity of data, it is difficult to ascertain what effects such mixtures may have on threatened and endangered species. Likewise, many toxic pollutants, including for example personal care products and emerging contaminants, remain unregulated at the state and federal level and could likewise cumulatively adversely affect threatened and endangered species. See also Section 5.6 for information regarding the mixtures toxicity.

6.3.5 Hatcheries

As mentioned in Chapter 3, hatcheries are one means by which certain T&E species populations have increased. However, genetic diversity may suffer with hatchery-dominated populations in the environment, further weakening the natural species diversity and ability of populations to survive abrupt changes in environmental conditions. For example, the high proportion of hatchery fish as juveniles could result in limiting factors of competition with wild fish for habitat, food, and other resources, while the high proportion of hatchery-origin spawners could result in limiting factors of genetic change, loss of fitness, competition among spawners for resources, including spawning areas. Particularly the second factor may cumulatively interact with the direct and indirect effects of this action by rendering wild spawning fish at a competitive disadvantage while undergoing additional stressors.

6.3.6 Hydrosystem

As described in Chapter 4, the emplacement and continued operation of the hydrosystem has adversely affected many threatened and endangered species, particularly in the Willamette, Snake, and Columbia Rivers. Both in impeding in-migration and out-migration, changing thermal regimes and flows, and affecting water quality including turbidity, toxics fluxes, total dissolved gas concentrations, and dissolved oxygen concentrations, current hydrosystem impacts may cumulatively affect T&E species that are present.

Recent planning and management efforts (including a 2019 Biological Opinion, and forthcoming EIS for the Federal Columbia River Power System, and implementation of the activities required pursuant to the 401certification for the Hells Canyon Complex) are carefully examining and implementing ways to mitigate some of the continued operational impacts on threatened and endangered species. Should those management strategies that are found to be effective be put into practice over the next few decades, improvement could be seen in conditions for T&E species in these rivers affected by hydrosystem operations.

6.4 Conservative Assumptions of the Analysis

6.4.1 Conservatisms Applied to Anadromous Salmonid Chronic Effect Assessments

The screening level toxicity assessment indicated anadromous salmon may be sensitive to chronic exposures; however, the screening level assessment methodology applied to the chum salmon was built on several conservative assumptions. These same conservatisms also apply to the chronic effect assessment methodologies of other anadromous salmonids occuring in Oregon.

The first conservatism was the preferential use of biomass as a chronic endpoint. Biomass is calculated based on both chronic effects to long-term survival and growth and often represents a sensitive endpoint. However, the link between biomass effects and population-level effects is unclear.

The second conservatism was the use of 60-day continuous exposure toxicity tests to derive the chronic criterion magnitude, even though the chronic criterion duration is 4 days. Chronic effect levels based on 60-day exposures are often significantly lower (i.e., appear more sensitive) than effect levels based the criterion duration of 4 days. Although the duration component functions as the averaging period for which a chronic magnitude should be calculated instead of a specific exposure duration, its application in National Pollutant Discharge Elimination System (NPDES) permit limits ensures dischargers operate assuming low flow conditions in receiving waters (see Appendix F). Therefore, the duration component of criteria is implemented in a manner that limits extended exposure duration in receiving streams. As a result, assuming a 60-day continuous exposure conditions across all Oregon freshwaters is inherently conservative given how criteria are implemented in permit limits. Nevertheless, such an assumption may be appropriate when:

(1) latent toxicity and/or time-to-effect data are unavailable to evaluate chronic responses at a 4-day resolution;
(2) time-to-effect data indicate toxic responses may occur during short exposure periods (e.g., 4 days) at the criterion magnitude, and/or;
(3) chronic responses evaluated during a recovery period (i.e., test organisms moved to clean/control waters and further observed) suggest latent toxicity can occur after a brief exposure even though no response was observed during the exposure period.

6.4.2 Considering Criteria Duration in Anadromous Salmonid Chronic Effect Assessments

The three chronic toxicity tests used to determine sensitivity of anadromous salmonids to chronic aluminum exposures (Cleveland et al. 1989; McKee et al. 1989[49]) were further evaluated to determine if salmonid survival, length, and weight were affected by exposure duration in order to determine if assuming 60-day continuous exposures in the chronic effect assessment was

[49] Buckler et al. (1995) published data from the same toxicity test as McKee et al. (1989), but reported length and survival, while McKee et al. (1989) reported survival and weight. For simplicity, all results from these two publications are referred to as McKee et al. (1989).

inappropriate based on toxicological responses. The specific objective was to plot chronic effect concentrations (as survival, weight, and length), assessed over time, to determine if salmonids were more tolerant to chronic Al exposures at shorter exposures durations more consistent with the criterion duration. Although, the chronic Al effect assessment for salmonids relied on biomass, assessment of salmonid responses over various time intervals relied only on the specific endpoints reported to show the effect of exposure duration (rather than the effect of Al concentration) on salmonid sensitivity measured as various endpoints.

In two tests, Cleveland et al. (1989) measured chronic brook trout survival and length at 15, 30, 45, and 60 days, and reported weight at days 45 and 60. McKee et al. (1989) reported Atlantic salmon survival and weight (dry weight and wet weight) at 15, 30, and 60 days and length at 30, 45, and 60 days. Raw C-R data and corresponding C-R curve score (see Appendix E for C-R analysis methodology) for each individual endpoint by time interval combination from Cleveland et al. (1989) and McKee et al. (1989) are reported in Table 6-4 (See **Supplemental Information B** for individual C-R models).

Table 6-3. All C-R data for survival, length, and weight-based endpoints used to calculate chronic response to members of the Family Salmoniformes. Reported EC_{15}, EC_{10}, and EC_5 values have been normalized to reference conditions of pH = 7.0, hardness = 100 mg/L, DOC = 1 mg/L. See Appendix E for model scoring methodology. Note, in many cases, particularly at brief exposure durations (e.g., < 30 days), no C-R relationship was exhibited which resulted in poor model fits and resultant effect concentrations that are beyond the bounds of the test concentrations. See Supplemental Information B for individual C-R models.

Test Conc. (µg/L)	Response	Exposure Duration (days)	Endpoint	Species	Citation	C-R curve label	Curve acceptability (score)	EC_{15}[a] (µg/L)	EC_{10}[a] (µg/L)	EC_5[a] (µg/L)
8	98.2	15	survival (%)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8A1	2	653.005	321.266	122.559
29	97									
68	99.2									
142	96									
292	97.7									
8	98	30	survival (%)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8A2	2	249.435	223.269	186.677
29	96.5									
68	99									
142	96									
292	73									
8	98	45	survival (%)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8A3	1	175.951	162.236	145.299
29	96.5									
68	99									
142	93.7									
292	42.3									
8	92.5	60	survival (%)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8A4	1	195.226	178.59	154.034
29	96.5									
68	93.2									
142	91									
292	29.7									

Test Conc. (µg/L)	Response	Exposure Duration (days)	Endpoint	Species	Citation	C-R curve label	Curve acceptability (score)	EC_{15}[a] (µg/L)	EC_{10}[a] (µg/L)	EC_{5}[a] (µg/L)
8	14.5	15	length (mm)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8B1	3	54.579	50.589	45.631
29	14.3									
68	15.2									
142	16.4									
292	16.7									
8	14.3	30	length (mm)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8B2	3	321.564	260.82	205.583
29	14.5									
68	14.9									
142	14									
292	12.2									
8	27.8	45	length (mm)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8B3	2	259.107	230.195	193.386
29	28									
68	28									
142	27.6									
292	21.8									
8	33	60	length (mm)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8B4	1	250.418	215.605	175.931
29	32.4									
68	33.1									
142	32.2									
292	25.9									
8	170	45	weight (mg)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8C1	1	172.38	158.091	142.646
29	176									
68	180									
142	160									
292	77									

Test Conc. (µg/L)	Response	Exposure Duration (days)	Endpoint	Species	Citation	C-R curve label	Curve acceptability (score)	EC_{15}[a] (µg/L)	EC_{10}[a] (µg/L)	EC_{5}[a] (µg/L)
8	303	60	weight (mg)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8C2	3	116.207	91.115	60.831
29	279									
68	270									
142	237									
292	125									
4	98	15	survival (%)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9A1	3	6069.869	3530.965	1435.874
57	97.2									
88	99.2									
169	97.7									
350	96.2									
4	98	30	survival (%)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9A2	3	854.077	440.877	151.521
57	95.7									
88	99									
169	97									
350	91.5									
4	98	45	survival (%)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9A3	2	265.028	242.881	215.727
57	95.7									
88	99									
169	97									
350	62.5									
4	95.5	60	survival (%)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9A4	2	293.718	279.001	256.773
57	93.2									
88	96.5									
169	97									
350	51.5									

Test Conc. (µg/L)	Response	Exposure Duration (days)	Endpoint	Species	Citation	C-R curve label	Curve acceptability (score)	EC_{15}[a] (µg/L)	EC_{10}[a] (µg/L)	EC_{5}[a] (µg/L)
4	15.7	15	length (mm)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9B1	3	146.093	133.506	115.425
57	16.1									
88	16.3									
169	15.8									
350	15.3									
4	20.4	30	length (mm)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9B2	1	61.674	53.652	44.4
57	19.5									
88	18.2									
169	15.9									
350	14.4									
4	29.5	45	length (mm)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9B3	1	312.961	247.387	167.41
57	29.2									
88	28.9									
169	28									
350	24.1									
4	35.3	60	length (mm)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9B4	1	116.745	96.297	69.938
57	35.1									
88	34.8									
169	33.4									
350	30.1									
4	224	45	weight (mg)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9C1	1	144.811	114.765	78.83
57	224									
88	210									
169	182									
350	112									

Test Conc. (µg/L)	Response	Exposure Duration (days)	Endpoint	Species	Citation	C-R curve label	Curve acceptability (score)	EC_{15}[a] (µg/L)	EC_{10}[a] (µg/L)	EC_{5}[a] (µg/L)
4	376	60	weight (mg)	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9C2	1	105.378	92.065	73.565
57	372									
88	361									
169	286									
350	204									
3	100	15	survival (%)	*Salmo salar*	McKee et al. 1989	Al chronic 26A1	3	1522.629	941.841	424.126
33	100									
71	96									
124	100									
264	98									
3	100	30	survival (%)	*Salmo salar*	McKee et al. 1989	Al chronic 26A2	2	289.145	194.221	100.292
33	98									
71	95									
124	97									
264	85									
3	96	60	survival (%)	*Salmo salar*	McKee et al. 1989	Al chronic 26A3	1	134.471	113.825	86.299
33	95									
71	91									
124	85									
264	37									
3	103.8	15	wet weight (mg)	*Salmo salar*	McKee et al. 1989	Al chronic 26B1	3	129.087	99.869	70.472
33	101.9									
71	105									
124	103.1									
264	105.6									

Test Conc. (µg/L)	Response	Exposure Duration (days)	Endpoint	Species	Citation	C-R curve label	Curve acceptability (score)	EC_{15}[a] (µg/L)	EC_{10}[a] (µg/L)	EC_{5}[a] (µg/L)
3	85	30	wet weight (mg)	*Salmo salar*	McKee et al. 1989	Al chronic 26B2	3	85.966	84.134	82.071
33	88.8									
71	91.3									
124	70									
264	69.3									
3	89.4	60	wet weight (mg)	*Salmo salar*	McKee et al. 1989	Al chronic 26B3	3	1.775	0.355	0.024
33	75									
71	68.1									
124	66.9									
264	56.7									
3	21	15	dry weight (mg)	*Salmo salar*	McKee et al. 1989	Al chronic 26C1	3	23.099	20.664	17.174
33	24.7									
71	29.7									
124	29.9									
264	26.6									
3	14.5	30	dry weight (mg)	*Salmo salar*	McKee et al. 1989	Al chronic 26C2	3	130.411	107.082	81.927
33	17.7									
71	15.6									
124	13.8									
264	9.9									
3	14.1	60	dry weight (mg)	*Salmo salar*	McKee et al. 1989	Al chronic 26C3	3	Not Calculable[b]	Not Calculable[b]	Not Calculable[b]
33	14.2									
71	13.6									
124	11.7									
264	16.1									

Test Conc. (µg/L)	Response	Exposure Duration (days)	Endpoint	Species	Citation	C-R curve label	Curve acceptability (score)	EC_{15}[a] (µg/L)	EC_{10}[a] (µg/L)	EC_{5}[a] (µg/L)
3	17.9	30	length (mm)	*Salmo salar*	McKee et al. 1989 / Buckler et al. 1995	Al chronic 26D1	3	45.093	42.41	39.017
33	18.1									
71	17.3									
124	17.1									
264	16.9									
3	23	45	length (mm)	*Salmo salar*	McKee et al. 1989 / Buckler et al. 1995	Al chronic 26D2	3	1.918	1.607	1.208
33	21.8									
71	21.4									
124	21.2									
264	22.6									
3	25.1	60	length (mm)	*Salmo salar*	McKee et al. 1989 / Buckler et al. 1995	Al chronic 26D3	1	181.145	111.795	50.153
33	24.3									
71	23.3									
124	22.6									
264	20									

[a] All values expressed as total aluminum, normalized to pH 7, DOC of 1 mg/L and 100 mg/L hardness as $CaCO_3$; normalized using MLR equations identified in USEPA (2018).

[b] Not calculable because data do not exhibit a concentration response relationship.

For each study by endpoint combination, an unnormalized EC_{20} value was also calculated at each reported time interval. Unnormalized EC_{20} values were selected for assessment of criterion duration because the chronic 20% effect level is preferred in criteria derivation and represents an effect threshold that has less uncertainty than lower effect levels (e.g., EC_5, EC_{10}). EC_{20} values also remained unnormalized because it was not required to inform conclusions and eliminated uncertainties produced through the Multi Linear Regression normalization process. Unnormalized EC_{20} values for each time interval by endpoint combination are reported in Table 6-5 (see **Supplemental Information B** and Table 6-4 for raw C-R data).

Within each toxicity test, EC_{20} values for individual endpoints (e.g., survival, length, weight) were plotted on the Y-axis relative to the corresponding exposure time on the X-axis to create time-to-effect plots. Functions were not fit to time-to-effect plots to create time-to-effect curves because growth endpoints did not typically exhibit time-dependent sensitivity and/or chronic responses were not quantified at an adequate temporal resolution to inform robust time-to-effect curves. Time-to-effect plots are displayed in Figures 6-2, 6-3, and 6-4.

Table 6-4. Chronic EC_{20} values from each time interval by endpoint combination for two early life stage brook trout tests reported by Cleveland et al. (1989) and an early life stage Atlantic salmon test reported by McKee et. al. (1989). Raw data used to derive EC_{20} values are reported in Table 6-4 and C-R curves are reported in Supplemental Information B.

Exposure Duration (days)	Endpoint	Species	Citation	C-R curve label	DOC (mg/L)	Hardness (mg/L)	pH	EC20 (µg/L)
15	survival	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8A1	1.8	12.8	5.65	1,192.50
30	survival	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8A2	1.8	12.8	5.65	271.14
45	survival	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8A3	1.8	12.8	5.65	188.5
60	survival	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8A4	1.8	12.8	5.65	208.36
15	length	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8B1	1.8	12.8	5.65	58.21
30	length	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8B2	1.8	12.8	5.65	390.93
45	length	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8B3	1.8	12.8	5.65	284.39
60	length	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8B4	1.8	12.8	5.65	284.35
45	weight	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8C1	1.8	12.8	5.65	186.15
60	weight	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 8C2	1.8	12.8	5.65	138.83
15	survival	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9A1	1.9	12.3	6.55	9,021.57
30	survival	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9A2	1.9	12.3	6.55	1,405.11
45	survival	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9A3	1.9	12.3	6.55	285.4
60	survival	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9A4	1.9	12.3	6.55	305.3
15	length	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9B1	1.9	12.3	6.55	156.34
30	length	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9B2	1.9	12.3	6.55	69.41
45	length	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9B3	1.9	12.3	6.55	371.69
60	length	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9B4	1.9	12.3	6.55	134.4
45	weight	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9C1	1.9	12.3	6.55	172.51
60	weight	*Salvelinus fontinalis*	Cleveland et al. 1989	Al chronic 9C2	1.9	12.3	6.55	116.32
15	survival	*Salmo salar*	McKee et al. 1989	Al chronic 26A1	1.8	12.7	5.7	2,163.66
30	survival	*Salmo salar*	McKee et al. 1989	Al chronic 26A2	1.8	12.7	5.7	386.83
60	survival	*Salmo salar*	McKee et al. 1989	Al chronic 26A3	1.8	12.7	5.7	151.9
15	wet weight	*Salmo salar*	McKee et al. 1989	Al chronic 26B1	1.8	12.7	5.7	160.51
30	wet weight	*Salmo salar*	McKee et al. 1989	Al chronic 26B2	1.8	12.7	5.7	87.02
60	wet weight	*Salmo salar*	McKee et al. 1989	Al chronic 26B3	1.8	12.7	5.7	5.75
15	dry weight	*Salmo salar*	McKee et al. 1989	Al chronic 26C1	1.8	12.7	5.7	25.06
30	dry weight	*Salmo salar*	McKee et al. 1989	Al chronic 26C2	1.8	12.7	5.7	154.166
60	dry weight	*Salmo salar*	McKee et al. 1989	Al chronic 26C3	1.8	12.7	5.7	Not Calculable[a]
30	length	*Salmo salar*	McKee et al. 1989 / Buckler et al. 1995	Al chronic 26D1	1.8	12.7	5.7	47.5
45	length	*Salmo salar*	McKee et al. 1989 / Buckler et al. 1995	Al chronic 26D2	1.8	12.7	5.7	2.19
60	length	*Salmo salar*	McKee et al. 1989 / Buckler et al. 1995	Al chronic 26D3	1.8	12.7	5.7	257.83

[a] Not calculable because data do not exhibit a concentration response relationship.

6.4.3 Considering Duration: Survival over Time

Across all three tests survival-based EC_{20} values decreased sharply between 15 and 30 days before leveling off from 30 to 60 days (see panel A in Figures 6-2, 6-3, and 6-4). The leveling off of survival-based EC_{20} values in all three tests from days 30 to 60 indicated survivors after 30 days generally remained alive regardless of exposure duration, likely because these were among the more tolerant individuals within the population. Increased mortality between 15 and 30 days could have been the result, (1) an exposure duration of > 15 days was required to elicit toxic effects or (2) exposed individuals began to swim up from eyed eggs sometime after 15 days leading to the introduction of a sensitive life stage.

If salmonids required a 15 to 30-day Al exposure to fully display mortality, it is unclear whether mortality events were the result of exposures for greater than 15 days or toxicological effects of Al occurred within the first 15 days but were only observable as latent after 15 days of exposure. Alternatively, salmonid deaths occuring between 15 and 30 days could reflect the time when eyed eggs start to swim up and begin exogenous feeding as fry, indicating sensitive life stages may experience acute effects to in a relatively short time period[50]. Cleveland et al. (1989) did not explicitly report changes in life stages over time but referred to test organisms at 15 days as "larvae," while at 30 – 60 days referred to test organisms as "juveniles," suggesting changes in life stages may have occurred between 15 and 30 days. Moreover, Cleveland et al. (1989) also measured behavioral responses, including feeding behavior, after 30 and 60 days of exposure, further suggesting test organisms had begun exogenous feeding within the first 30 days of exposure. Contrary to Cleveland et al. (1989), McKee et al. (1989; reported in Buckler et al. 1995), indicated Atlantic salmon in their early life stage test did not begin exogenous feed until "near the end of the studies" due to the relatively cold test water temperatures (i.e., 8°C). Therefore, the mechanism producing sensitivity differences (based on mortality) between 15 and 30 days in this particular toxicity test may represent a required exposure of at least 15-days, in combination with, the initiation of physiological changes associated with the onset of swim up or some other unknown confounding factor.

[50] This discussion focuses on exposure duration at Al concentrations higher than the chronic criterion. The intent is to determine how exposure time influences salmonid responses, regardless of sensitivity relative to the chronic criterion magnitude.

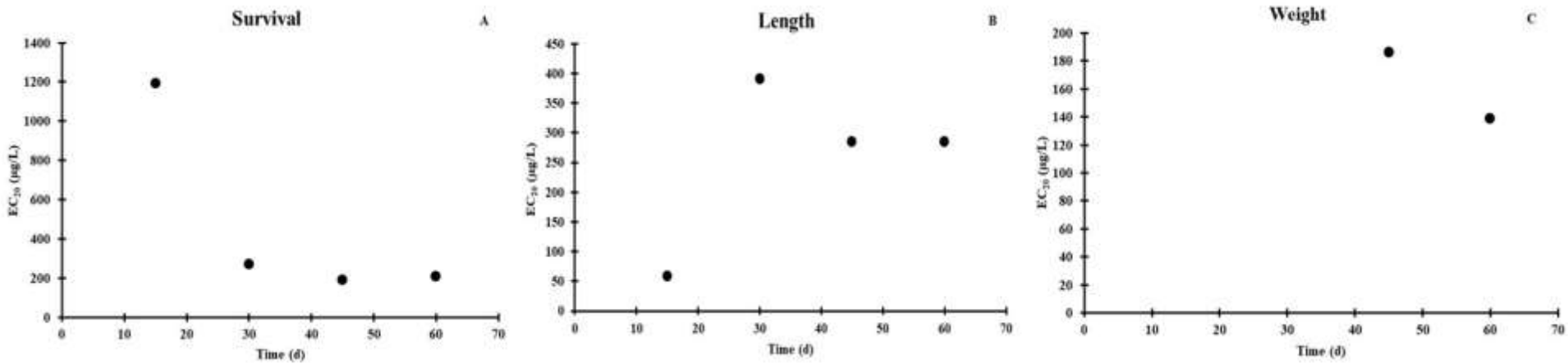

Figure 6-2. Early life stage brook trout time-to-effect plots constructed from survival and growth measurements over time (Cleveland et al. 1989). Panel A shows survival-based EC_{20} values at 15, 30, 45, and 60 days (C-R Curves 8A1 – 8A4). Panel B shows length-based EC_{20} values at 15, 30, 45, and 60 days (C-R Curves 8B1 – 8B4). Panel C shows weight-based EC_{20} values at 45 and 60 days (C-R Curves 8AC1 and 8C2).

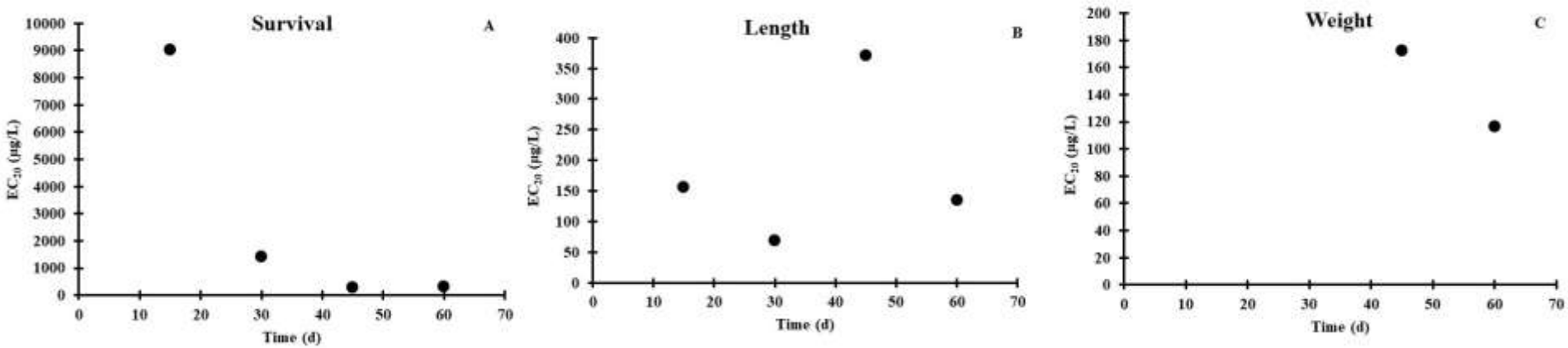

Figure 6-3. Early life stage brook trout time-to-effect plots constructed from survival and growth measurements over time (Cleveland et al. 1989). Panel A shows survival-based EC_{20} values at 15, 30, 45, and 60 days (C-R Curves 9A1 – 9A4). Panel B shows length-based EC_{20} values at 15, 30, 45, and 60 days (C-R Curves 9B1 – 9B4). Panel C shows weight-based EC_{20} values at 45 and 60 days (C-R Curves 9C1 and 9C2).

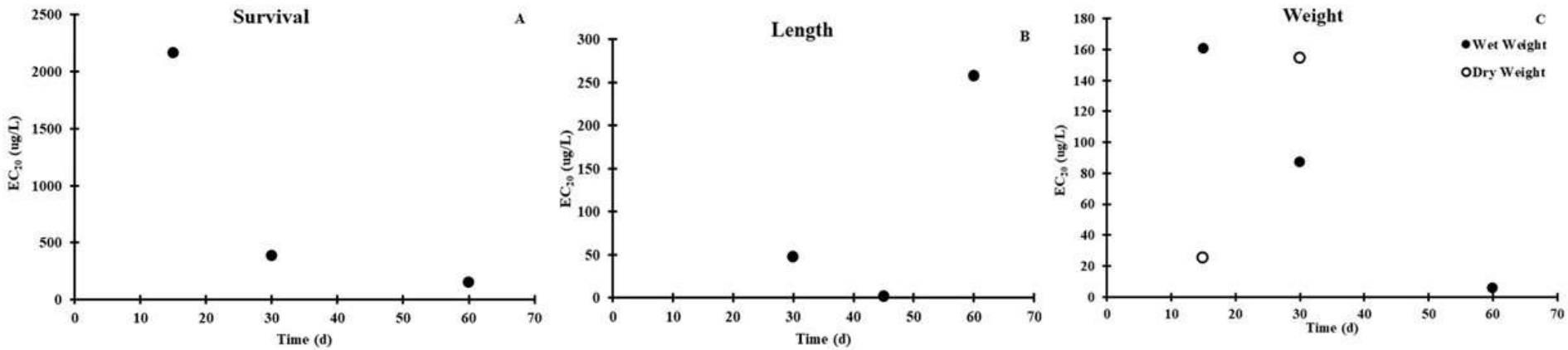

Figure 6-4. Early life stage Atlantic Salmon time-to-effect plots constructed from survival and growth measurements over time (McKee et al. 1989). Panel A shows survival-based EC_{20} values at 15, 30, and 60 days (C-R Curves 26A1 – 26A3). Panel B shows length-based EC_{20} values at 30, 45, and 60 days (C-R Curves 26D1 – 26D3). Panel C shows wet-weight-based EC_{20} values at 15, 30, and 60 days (C-R Curves 26B1 and 26B3). Panel C also shows dry-weight-based EC20 values at 15 and 30 days (C-R Curves 26C1 and 26C2; 60-day dry weight effect concentrations from C-R curve 26C3 were not calculable because of a concentration-response relationship was not observed).

6.4.4 Considering Duration: Length over Time

Across the brook trout studies, length appears to be relatively sensitive at 15 days of exposures (Note: McKee et al. [1989] did not report length measurements at 15 days), with Figure 6-2 (see panel A and B) showing a length-based time-to-effect plot that is nearly inverted from the mortality-based time-to-effect plots. The inversion between length and survival at 15 days may suggest the relatively sensitive individuals are capable of surviving the first 15 days of exposure at the expense of reduced length, before dying between 15 and 30 days. Beyond, Figure 6-2 (panels A and B), length-based time-to-effect plots for the remaining salmonid tests (Figure 6-3 and Figure 6-4) indicate no clear changes in length-based sensitivity over time.

6.4.5 Considering Duration: Weight over Time

Cleveland et al (1989) only reported weight-based responses of brook trout after 45 and 60 days of exposure. Weight was not reported with enough temporal resolution to fully inform conclusions (panel C in **Figure 6-2** and **6-3**). Wet-weight-based Atlantic salmon sensitivity increased linearly over time, suggesting it may be inappropriate to assume weight-based sensitivity at 60 days is representative of weight-based sensitivity at shorter, more environmentally-relevant exposure durations. Nevertheless, Atlantic salmon dry-weight-based sensitivity decreased from 15 to 30 days and exhibited no C-R relationship at 60 days, confounding any conclusions for Atlantic salmon based on wet weight (panel C in Figure 6-4).

6.4.6 Considering Duration: Growth and Physiological Factors

Observed salmonid sensitivity to chronic Al exposures is further supported by physiological considerations associated with the growth of early life stages. Published data (Delonay, 1991; Wood et al., 1990)

suggest larval salmonids are more sensitive to chronic Al exposure at the swim up stage (after larvae begin exogenous feeding) when the fish are rapidly changing physiologically, which suggests a specific time-to-effect duration (e.g., 15 days) is not required to produce chronic effects. Confounding the issue of post-hatch life stage sensitivity patterns with early life stage salmonid (ELS) studies of Al (reviewed by Ingersoll et al. 1990), are the variety of water quality conditions used in toxicity tests. The discrepancies in response of ELS salmonids to chronic Al toxicity are likely the result of differences in the water temperature used for rearing and Al exposure in the assays, which affects development rates of ELS salmonids, and the balance of Al concentration relative to hydrogen ion (H^+) concentration as the primary toxic agent (Wood et al., 1990). ELS salmonid sensitivity to Al exposure generally increased with age (most sensitive = swim-up fry > yolk-sac fry > eyed eggs = fertilized eggs = most tolerant), whereas the opposite was true for pH (Ingersoll et al. 1990). Below, two studies are summarized directly addressing, (1) whether a 15-day exposure duration was the time required for Al toxicity to manifest in larval, post-hatch salmonids and (2) if there was a difference in sensitivity to chronic Al exposure amongst different developmental stages of ELS salmonids.

<u>DeLonay et al. (1991) - Sensitivity of Early-life Stage Golden Trout to Low pH and Elevated Aluminum:</u> DeLonay et al. (1991) exposed ELS golden trout (*Oncorhynchus aguabonita aguabonita*) to acid and Al to determine sensitivity to conditions simulating an episodic pH

depression. Freshly fertilized eggs, yolk-sac fry (alevins), and swim-up larvae were exposed for 7 days to one of 12 combinations of pH and Al, and surviving fish were held to 40 day post-hatch to determine the effect of exposure on subsequent survival and recovery. Nominal and measured pH of the 12 exposure combinations differed by no more than 0.15 units and measured total Al concentrations were within 6% of nominal values. Focusing only on pH 5.0 and 5.5 treatments[51], golden trout exposed for 7 days as swim-up larvae were more sensitive to Al than yolk-sac fry (Figure 6-5). Survival was significantly reduced when swim-up larvae were exposed to total Al concentrations ≥ 100 µg/L, and complete mortality occurred at 300 µg/L. Most mortality occurred during the 7-d exposure and little or no mortality occurred during the recovery period once the exposed life stage had attained the next development stage. Coincident with these results, golden trout exposed at pH 5.5 and 100 µg total Al/L exhibited a significant (> 50%) reduction in body length and weight at 40 days post-hatch compared to fish at pH 5.5 without Al. DeLonay et al. (1991) demonstrated that Al exposure less than 15 days affects ELS salmonids, and, that sensitivity of golden trout to Al changes with age and developmental stage, such that swim-up larvae were more sensitive than yolk-sac larvae.

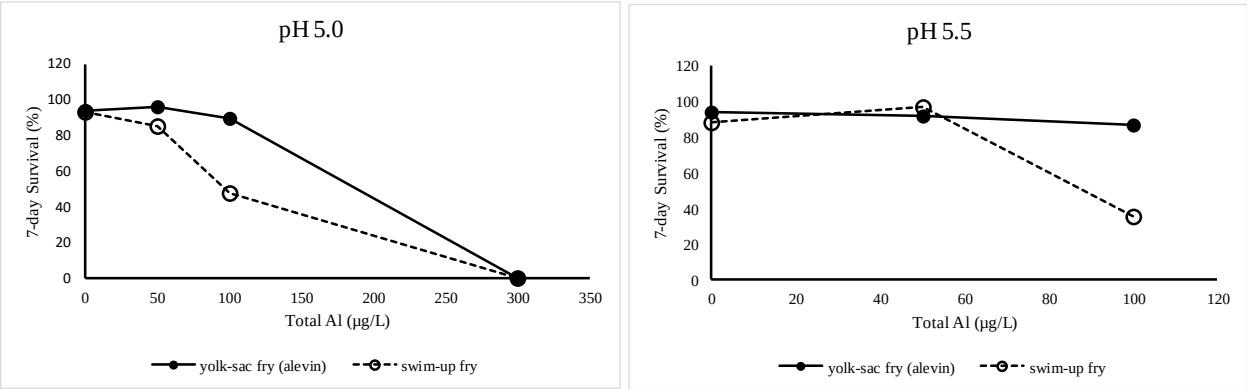

Figure 6-5. Seven-day percent survival of golden trout exposed to various Al concentrations at pH 5.0 and 5.5.

<u>Wood et al. (1990) - Whole Body Ions of Brook Trout Alevins: Responses of Yolk-Sac and Swim-up Stages to Water Acidity, Calcium, and Aluminum, and Recovery Effects:</u> Wood et al. (1990) raised brook trout from fertilization (day 0) in flowing soft water (pH = 6.5, Ca = 2 mg/L, Al = 0 µg/L), and exposed them to different combinations of pH/Ca/Al at different time intervals post fertilization. Specifically, Wood et al. (1990) exposed individuals 49 days after fertilization (yolk-sac stage, 2 d post-hatch) and 70 days after fertilization (swim-up stage) to test waters for 21 days and then followed exposures with a 20-day recovery time. Ignoring effects from pH alone or the complex combination of pH, Ca and Al, Wood et al. (1990) demonstrated Al (37-111 µg/L total Al) raised most body ions above control values (a positive or stimulatory Al effect), while greater Al concentrations lowered them (negative Al effect). Importantly, swim-up fry were more sensitive to Al than yolk-sac fry, showing pronounced developmental inhibition (lower body weight, higher Mg) under adverse conditions. Al concentrations of 37 and 111 µg/L

[51] Because pH alone at a level of 4.5 without Al amendment significantly reduced percent survival of fertilized eggs, yolk-sac, and swim-up fry compared to pH alone at levels of 5.0, 5.5, 6.0 and 6.5, and, because the only concentration of Al tested (50 µg/L) at pH 6.0 and 6.5 did not affect survival of any life stage compared to controls.

at pH 4.8 and 5.2 affected yolk-sac fry whole body Na, CI, K, and Ca concentrations, but had a negligible influence on either Mg concentrations or body weight. Thus, in terms of survival, body weight, development, and the observation of persistence deleterious ionoregulatory effects during recovery, the swim-up stage was more sensitive.

The authors speculated that increased sensitivity of swim-up larvae to Al related to the increased importance of the gills in the swim-up stage for both ion uptake and respiratory gas exchange, where aluminum's the toxic mode of action is likely the same as in adult salmonids - a combination of inhibition of active ion influx through the "chloride cells" and stimulation of passive efflux through paracellular channels. The key difference between yolk-sac and swim-up larvae is reliance on the vitelline versus the branchial membranes as the principal exchange sites for both ion uptake and respiratory gas exchange. Associated with the transition from vitelline to branchial membranes as the principal exchange site is an increasing reliance on the external environment and a decreasing reliance on yolk-sac stores for both minerals, nutrients, and energy for further development.

Given the study design, Wood et al. (1990) was able to demonstrate differential responses when testing larvae at yolk-sac versus swim-up stages. Contrary to Wood et al. (1990), study designs by Cleveland et al. (1989) and McKee et al. (1989) were designed in a manner where apical endpoints, such as growth and survival, were only evaluated at discrete intervals (e.g., 15, 30, 45, 60 days) from fertilization through swim-up stage. As a result, these studies were not able to clearly discern the clear negative effect of Al on swim-up larvae.

6.4.7 Considering Duration: Conclusions

Based on the limited studies available, salmonid sensitivity based on mortality was influenced by time, with salmonids becoming more sensitive between 15 and 30 days into the exposures. Survival-based sensitivity did not appear to be influenced by time between 30 and 60 days. It is unclear, if time-dependent mortality events were the result of the time-to-effect exceeding 15 days, the presence of sensitive life stages after 15 days, or a combination of the two. Growth-based salmonid sensitivity (i.e., length and weight) displayed no clear changes over time with no consistent patterns observed across studies. Physiological considerations suggest salmonids may be sensitive to brief aluminum exposures during the swim-up stage. Therefore, basing chronic salmonid sensitivity to Al on 60-day effect concentrations provided a reasonable estimate of expected responses at shorter exposure periods and captured the brief time intervals when salmonid life stages may be particularly sensitive to chronic Al exposures.

7. Summary and Conclusions

Table 7-1. Summary of Effects Determinations

A. USFWS Species Purview

Species Name	Status (T=threatened; E=endangered)	Designated Critical Habitat in Action Area?	Species Effects Determination	Critical Habitat Effects Determination
Oregon spotted frog (*Rana pretiosa*)	T	Y	NLAA[1,2]	NLAA
Warner sucker (*Catostomus warnerensis*)	T	Y	NLAA[1,2]	NLAA
Shortnose sucker (C*hasmistes brevirostris*)	E	Y	NLAA[1,2]	NLAA
Lost River sucker (*Deltistes luxatus*)	E	Y	NLAA[1,2]	NLAA
Hutton tui chub (*Gila bicolor ssp.*)	T	N	NLAA[1,2]	NLAA
Borax Lake chub (*Gila boraxobius*)	E, Proposed for Delisting	Y	NLAA[1,2]	NLAA
Lahontan cutthroat trout (*Oncorhynchus clarki henshawi*)	T	N	**LAA**[2]	NLAA
Foskett speckled dace (*Rhinichthys osculus ssp*.)	T, Proposed for Delisting	N	NLAA[1,2]	NLAA
Bull trout (*Salvelinus confluentus*)	T	Y	**LAA**[2]	**LAA**
Vernal pool fairy shrimp (*Branchinecta lynchi*)	T	Y	**LAA**[1,2]	**LAA**

Notes:
1, Based on CMC analyses
2, Based on CCC analyses

B. NMFS Species Purview

Species Name	DPS/ESU	Status (T=threatened; E=endangered)	Designated Critical Habitat in Action Area?	Species Effects Determination	Critical Habitat Effects Determination
Green Sturgeon (*Acipenser medirostris*)	Southern DPS	T	Y	**LAA**[1]	**LAA**
Eulachon (*Thaleichthys pacificus*)	Southern DPS	T	Y	**LAA**[2]	**LAA**
Southern Resident Killer Whales (*Orcinus Orca*)	Puget Sound	E	N (Y)[3]	**LAA**[2]	**LAA**
Coho Salmon (*Oncorhynchus kisutch*)	Lower Columbia River	T	Y	**LAA**[2]	**LAA**
	Oregon Coast	T	Y	**LAA**[2]	**LAA**
	Southern Oregon/Northern California Coast	T	Y	**LAA**[2]	**LAA**
Chum Salmon (*Oncorhynchus keta*)	Columbia River	T	Y	**LAA**[2]	**LAA**
Chinook Salmon (*Oncorhynchus tshawytscha*)	Upper Columbia River, spring-run	E	Y	**LAA**[2]	**LAA**
	Snake River spring/summer-run	T	Y	**LAA**[2]	**LAA**
	Snake River Fall-run	T	Y	**LAA**[2]	**LAA**
	Upper Willamette River	T	Y	**LAA**[2]	**LAA**
	Lower Columbia	T	Y	**LAA**[2]	**LAA**
Sockeye Salmon (*Oncorhynchus nerka*)	Snake River	E	Y	**LAA**[2]	**LAA**

Species Name	DPS/ESU	Status (T=threatened; E=endangered)	Designated Critical Habitat in Action Area?	Species Effects Determination	Critical Habitat Effects Determination
Steelhead (*Oncorhynchus mykiss*)	Upper Columbia River	T	Y	**LAA**[2]	**LAA**
	Snake River Basin	T	Y	**LAA**[2]	**LAA**
	Middle Columbia River	T	Y	**LAA**[2]	**LAA**
	Upper Willamette River	T	Y	**LAA**[2]	**LAA**
	Lower Columbia River	T	Y	**LAA**[2]	**LAA**

Notes:
1, Based on CMC analyses
2, Based on CCC analyses
3, Relevant to the Conference Opinion

8. Essential Fish Habitat

In this section, Essential Fish Habitat (EFH) is assessed for potential adverse impacts from the USEPA's proposed aluminum water quality criteria for freshwaters in Oregon.

8.1 Background

The Magnuson-Stevens Fishery Conservation and Management Act (MSA), as amended by the Sustainable Fisheries Act of 1996 (Public Law 104-267), requires federal agencies to consult with NOAA Fisheries on activities that may adversely affect EFH. According to the Magnuson-Stevens Fishery Conservation and Management Act (MSA§3), EFH means those waters and substrate necessary to fish for spawning, breeding, feeding, or growth and maturity. For the purpose of interpreting this definition of EFH: "waters" include aquatic areas and their associated physical, chemical, and biological properties that are used by fish; "substrate" includes sediment, hard bottom , structures underlying the waters, and associated biological communities; "necessary" means the habitat required to support a sustainable fishery and the managed species' contribution to a healthy ecosystem; and "spawning, breeding, feeding, and growth to maturity" covers a species' full life cycle ("Magnuson-Stevens Act Provisions," 1976b). "Adverse effect" means any impact which reduces quality and/or quantity of EFH, and may include direct (e.g. physical disruption), indirect (e.g. loss of prey), site-specific or habitat-wide impacts, including individual, cumulative, or synergistic consequences of actions ("Magnuson-Stevens Act Provisions," 1976a).

Pursuant to the MSA, the Pacific Fisheries Management Council (PFMC) has designated EFH for three species of federally-managed Pacific salmon: Chinook (*Oncorhynchus tshawytscha*); coho (O. *kisutch*); and Puget Sound pink salmon (O. *gorbuscha*) (PFMC, 2000). Freshwater EFH for Pacific salmon includes all those streams, lakes, ponds, wetlands, and other water bodies currently, or historically accessible to salmon in Washington, Oregon, Idaho, and California, except areas upstream of certain impassable man-made barriers (as identified by PFMC (2000)), and longstanding, naturally-impassable barriers (i.e. natural waterfalls in existence for several hundred years).

The objective of this EFH assessment is to determine if the proposed action may "adversely affect" designated EFH for relevant commercially or federally managed fisheries species within the proposed action area. It also describes conservation measures proposed to avoid, minimize or otherwise offset potential adverse effects to designated EFH resulting from the proposed action.

USEPA reviewed the NMFS information and (www.fisheries.noaa.gov/resource/map/essential-fish-habitat-mapper) to determine if the Action Area for this BE overlaps with EFH. In this case this overlap would be restricted to the EFH species that use freshwater habitats—Chinook, Pink salmon, and Coho salmon, since the proposed water quality criteria for aluminum for freshwaters in Oregon is not relevant to the Puget Sound (or other marine waters). The USEPA made the following conclusions:

- Pink Salmon. Oregon freshwaters are not within the distribution of O. *gorbusha*.

- Chinook salmon. Oregon freshwaters have been designated as EFH for Chinook. As described in Chapter 3 of the BE, Chinook salmon are present and use the Action Area as a migration corridor, spawning habitat, rearing habitat, and smolt out-migration habitat.
- Coho salmon. Oregon freshwaters have been designated as EFH for Coho. As described in Chapter 3 of the BE, coho salmon are present and use the Action Area as a migration corridor, spawning habitat, rearing habitat, and smolt out-migration habitat.

8.2 Description of the Project/Proposed Activity

The activity under consideration for this EFH assessment is identical to the description contained in this Biological Evaluation (BE) for this proposed action, as described in Chapters 1 and 2 of the BE.

Water quality is an important component of EFH. The potential effects of this action on EFH within the Action Area are the same as those described for fish species of concern in Chapters 5 and 6. A summary of the determinations made for ESA listed species is found in Chapter 7. The USEPA has performed an assessment of how this action may affect the water quality (total recoverable Al concentrations) that Chinook ESUs, including Snake River spring/summer-run and fall-run Chinook and Columbia River spring/summer and fall-run Chinook, could potentially be exposed to in freshwaters of Oregon. A summary of the determinations made for ESA-listed species is found in the BE. Using the information presented in the BE, the USEPA has determined that promulgation of the aluminum water quality criteria for Oregon freshwaters is likely to adversely affect Columbia and Snake River Chinook salmon ESUs, and coho salmon ESUs, and their critical habitat. Therefore, this action is **likely to adversely affect coho salmon EFH and Chinook salmon EFH** in this area.

8.3 EFH Conservation Measures and Conclusion

The USEPA concludes that the proposed action is likely to adversely affect EFH for Chinook salmon and coho salmon but will not affect EFH for Puget Sound pink salmon.

9. References

ADFG. (1994). Sockeye salmon. Retrieved from https://www.adfg.alaska.gov/static/education/wns/sockeye_salmon.pdf

Al-Humaidhi, A. W., Bellman, M. A., Jannot, J., & Majewski, J. (2010). *Observed and Estimated Total Bycatch of Green Sturgeon in the 2002-2008 U.S. West Coast Groundfish Fisheries*. Retrieved from Seattle, WA:

Alvarez, D., Perkins, S., Nilsen, E., & Morace, J. (2014). Spatial and temporal trends in occurrence of emerging and legacy contaminants in the Lower Columbia River 2008-2010. *Science of the Total Environment, 484*, 322-330.

ATSDR. (2008). *Toxicological Profile for Aluminum.* Retrieved from Atlanta, GA:

Au, W., Horne, J., & Jones, C. (2010). Basis of acoustic discrimination of Chinook salmon from other salmons by echolocating Orcinus orca. *The Journal of the Acoustical Society of America, 128*, 2225-2232. doi:10.1121/1.3473697

Bain, D. (1990). Examining the validity of inferences drawn from photo-identification data, with special reference to studies of the killer whale (Orcinus orca) in British Columbia. In P. S. Hammond, S. A. Mizroch, & D. G. P (Eds.), *Report of the International Whaling Commission. Special Issue 12: Individual Recognition Of Cetaceans: Use Of Photo-Identification And Other Techniques to Estimate Population Parameters (SC/A88/ID33)* (pp. 93-100). La Jolla, CA: International Whaling Commission.

Baird, R. W. (2000). The killer whale: Foraging specializations and group hunting. In J. Mann, R. C. Connor, P. L. Tyack, & H. Whitehead (Eds.), *Cetacean Societies: Field Studies of Dolphins and Whales* (pp. 127-153). Chicago, IL: University of Chicago Press.

Baird, R. W., Hanson, M. B., & Dill, L. M. (2005). Factors influencing the diving behaviour of fish-eating killer whales: sex differences and diel and interannual variation in diving rates. *Canadian Journal of Zoology, 83*, 257-267.

Banish, N. P., Adams, B. J., Shively, R. S., Mazur, M. M., Beauchamp, D. A., & Wood, T. M. (2009). Distribution and habitat associations of radio-tagged adult Lost River suckers and shortnose suckers in Upper Klamath Lake, Oregon. *Transactions of the American Fisheries Society, 138*, 153-168.

Barrett-Lennard, L. G., & Heise, K. A. (2007). The natural history and ecology of killer whales. In J. Estes, D. P. DeMaster, D. F. Doak, T. M. Williams, & R. L. Brownell (Eds.), *Whales, Whaling, and Ocean Ecosystems* (pp. 161-171). Oakland, CA: University of California Press.

Battin, J., Wiley, M. W., Ruckelshaus, M. H., Palmer, R. N., Korb, E., Bartz, K. K., & Imaki, H. (2007). Projected impacts of climate change on salmon habitat restoration. *Proceedings of the National Academy of Sciences, 104*(16), 6720-6725. doi:10.1073/pnas.0701685104

Baxter, J. S., Taylor, E. B., Devlin, R. H., Hagen, J., & McPhail, J. D. (1997). Evidence for natural hybridization between Dolly Varden (Salvelinus malma) and bull trout (Salvelinus confluentus) in a northcentral British Columbia watershed. *Canadian Journal of Fisheries and Aquatic Sciences, 54*(2), 421-429. doi:10.1139/f96-289

Beamish, R. J., & Bouillon, D. R. (1993). Pacific salmon production trends in relation to climate. *Canadian Journal of Fisheries and Aquatic Sciences, 50*, 1002-1016.

Beauchamp, D., & Van Tassell, J. (2001). Modeling Seasonal Trophic Interactions of Adfluvial Bull Trout in Lake Billy Chinook, Oregon. *Transactions of the American Fisheries Society, 130*, 204-216. doi:10.1577/1548-8659(2001)130<0204:MSTIOA>2.0.CO;2

Behnke, R. (2010). *Trout and Salmon of North America*. New York, NY: Simon and Shuster.
Belk, D. (1977). Evolution of Egg Size Strategies in Fairy Shrimps. *The Southwestern Naturalist, 22*(1), 99-105. doi:10.2307/3670467
Bell, E., & Duffy, W. G. (2007). Previously undocumented two-year freshwater residency of juvenile coho salmon in Prairie Creek, California. *Transactions of the American Fisheries Society, 136*, 966-970.
Bigg, M. A., Olesiuk, P., Ellis, G. M., Ford, J., & Balcomb, K. C. (1990). Social organization and genealogy of resident killer whales (Orcinus orca) in the coastal waters of British Columbia and Washington State. In P. S. Hammond, S. A. Mizroch, & D. G. P (Eds.), *Report of the International Whaling Commission. Special Issue 12: Individual Recognition Of Cetaceans: Use Of Photo-Identification And Other Techniques to Estimate Population Parameters (SC/A88/ID33)* (pp. 383-405). La Jolla, CA: International Whaling Commission.
Bilby, R. E., Fransen, B. R., & Bisson, P. A. (1996). Incorporation of nitrogen and carbon from spawning coho salmon into the trophic system of small streams: evidence from stable isotopes. *Canadian Journal of Fisheries and Aquatic Sciences, 53*(1), 164-173. doi:10.1139/f95-159
Bills, F. T. (1978). *Taxonomic status of the isolated populations of tui chub referred to Gila bicolor oregonensis (Snyder).* (Master of Science), Oregon State University, Corvallis, OR.
Birchall, J. D., Exley, C., Chappell, J. S., & Phillips, M. J. (1989). Acute toxicity of aluminum to fish eliminated in silicon-rich acid waters. *Nature, 338*(6211), 146.
Birge, W. J. (1978). Aquatic toxicology of trace elements of coal and fly ash. In J. P. Thorp & J. W. Gibbons (Eds.), *Energy and Environmental Stress in Aquatic Systems: Selected Papers from a Symposium held at Augusta, Georgia, November 2-4, 1977* (pp. 219-240). Athens, GA: University of Georgia, Institute of Ecology, Savannah River Ecology Laboratory,.
Birge, W. J., Hudson, J. E., Black, J. A., & Westerman, A. G. (1978). Embryo-larval bioassays on inorganic coal elements and in situ biomonitoring of coal-waste effluents. In D. E. Samuel, J. R. Stauffer, C. H. Hocutt, & W. T. Mason (Eds.), *Surface Mining and Fish/Wildlife Needs in Eastern U.S. (FWS/OBS-78/81)* (pp. 97-104). Washington, DC: U.S. Department of the Interior, Fish and Wildlife Service, Office of Biological Services.
Birge, W. J., Westerman, A. G., & Spromberg, J. A. (2000). Chapter 14 A, Comparative toxicology and risk assessment of amphibians. In D. W. Sparling, G. Linder, & C. A. Bishop (Eds.), *Ecotoxicology of Amphibians and Reptiles* (pp. 727-791). Pensacola, FL: SETAC Press.
Bjornn, T. C., Craddock, D. R., & Corley, D. R. (1968). Migration and Survival of Redfish Lake, Idaho, Sockeye Salmon, Oncorhynchus nerka. *Transactions of the American Fisheries Society, 97*(4), 360-373. doi:10.1577/1548-8659(1968)97[360:MASORL]2.0.CO;2
Bjornn, T. C., & Reiser, D. W. (1991). Habitat requirements of salmonids in streams. In W. R. Meehan (Ed.), *Influences of Forest and Rangeland Management on Salmonid Fishes and their Habitats* (Vol. 19, pp. 83-138). Bethesda, MD: American Fisheries Society Special Publication.
BLM. (2003). *Lakeview Resource Management Plan and Record of Decision*. Retrieved from Lakeview, OR:

BLM. (2005). *Andrews/Steens Resource Management Plan: Andrews Management Unit Record of Decision and Resource Management Plan*. Retrieved from Hines, OR:

Boag, T. D. (1987). Food habits of bull char, Salvelinus confluentus, and rainbow trout, Salmo gairdneri, coexisting in a foothills stream in northern Alberta. *Canadian Field-Naturalist, 101*(1), 56-62.

Bond, C. E. (1974). *Endangered Plants and Animals of Oregon, I. Fishes* (Special Report 205). Retrieved from Corvallis, OR:

Bond, C. E. (1992). Notes on the nomenclature and distribution of the bull trout and the effects of human activity on the species, . In P. J. Howell & D. V. Buchanan (Eds.), *Proceedings of the Gearhart Mountain Bull Trout Workshop* (pp. 1-4). Corvallis, OR: Oregon Chapter of the American Fisheries Society.

Bottom, D. L., Howell, P. J., & Rodgers, J. D. (1985). *The Effects of Stream Alterations on Salmon and Trout Habitat in Oregon*. Retrieved from Portland, OR:

Bottom, D. L., Simenstad, C. A., Burke, J., Baptista, A. M., Jay, D. A., Jones, K. K., . . . Schiewe, M. H. (2005). *Salmon at River's End: The Role of the Estuary in the Decline and Recovery of Columbia River Salmon* (NOAA Technical Memorandum NMFS-NWFSC-68). Retrieved from Springfield, VA:

Brewin, P. A., & Brewin, M. K. (1997). Distribution maps for bull trout in Alberta In W. C. Mackay, M. K. Brewin, & M. Monita (Eds.), *Friends of the Bull Trout Conference Proceedings* (pp. 206-216). Calgary, AB: Bull Trout Task Force (Alberta) c/o Trout Unlimited Canada.

Brown, D. J. A. (1983). Effect of calcium and aluminum concentrations on the survival of brown trout (Salmo trutta) at low pH. *Bulletin of Environmental Contamination and Toxicology, 30*(5), 582-587. doi:10.1007/bf01610178

Brown, L. G. (1994). *On the Zoogeography and Life History of Washington's Native Charr: Dolly Varden, Salvelinus malma (Walbaum) and Bull Trout, Salvelinus confluentus (Suckley)* (Report no. 94-04). Retrieved from Olympia, WA:

BRT. (1997). *Status Review Update for West Coast Steelhead from Washington, Idaho, Oregon, and California.* Retrieved from Springfield, VA:

Buchwalter, D. B., Sweeney, B. W., & Funk, D. (2008). *Development of a mayfly model (Centroptilum triangulifer) for ecotoxicology and toxicogenomic studies.* Paper presented at the SETAC 29th Annual Meeting, Tampa, FL.

Buckler, D. R., Cleveland, L., Little, E. E., & Brumbaugh, W. G. (1995). Survival, sublethal responses, and tissue residues of Atlantic salmon exposed to acidic pH and aluminum. *Aquatic Toxicology, 31*(3), 203-216. doi:10.1016/0166-445X(94)00066-Y

Buettner, M. (2005). *Contribution of the Lost River to the Recovery of Federally Listed Lost River and Shortnose Suckers* (Draft White Paper). Retrieved from Klamath Falls, OR:

Buettner, M., & Scoppettone, C. (1990). *Life History and Status of Catostomids in Upper Klamath Lake, Oregon* (Completion report). Retrieved from Reno, NV:

Buhl, K. J. (2002). *The relative toxicity of waterborne inorganic contaminants to the Rio Grande silvery minnow (Hybognathus amarus) and fathead minnow (Pimephales promelas) in a water quality simulating that in the Rio Grande, New Mexico* (Final Rep. to U.S. Fish and Wildl. Serv., Study No. 2F33-9620003). Retrieved from Yankton, SD:

Burdick, S. M., Hendrixson, H. A., & VanderKooi, S. P. (2008). Age-0 Lost River sucker and shortnose sucker nearshore habitat use in Upper Klamath Lake, Oregon: a patch occupancy approach. *Transactions of the American Fisheries Society, 137*, 417-430.

Burdick, S. M., Hewitt, D. A., Rasmussen, J. E., Hayes, B. S., Janney, E. C., & Harris, A. C. (2015). Effects of lake surface elevation on shoreline-spawning Lost River suckers. *North American Journal of Fisheries Management, 35*, 478-490.

Burgner, R. L. (1991). Life history of sockeye slamon (*Oncorhynchus nerka*). In C. Groot & L. Margolis (Eds.), *Pacific Salmon Life Histories* (pp. 1-118). Vancouver, BC: UBC Press.

Call, D. J., Brooke, L. T., Lindberg, C. A., Markee, T. P., McCauley, D. J., & Poirier, S. H. (1984). *Toxicity of Aluminum to Freshwater Organisms in Water of pH 6.5-8.5* (Technical Report Project No. 549-238-RT-WRD). Retrieved from Superior, WI:

Cardwell, A. S., Adams, W. J., Gensemer, R. W., Nordheim, E., Santore, R. C., Ryan, A. C., & Stubblefield, W. A. (2018). Chronic toxicity of aluminum, at a pH of 6, to freshwater organisms: Empirical data for the development of international regulatory standards/criteria. *Environmental Toxicology and Chemistry, 37*(1), 36-48. doi:10.1002/etc.3901

Carmichael, R. W. (2006). *Recovery Plan for Oregon's Middle Columbia River Steelhead* (Draft Progress Report). Retrieved from La Grande, OR:

Cavender, T. M. (1978). Taxonomy and distribution of the bull trout, Salvelinus confluentus (Suckley) from the American northwest. *California Fish and Game, 64*(3), 139-174.

CBFWA. (1990). *Snake River Subbasin (Mainstem from Mouth to Hells Canyon Dam) Salmon and Steelhead Production Plan* Retrieved from Portland, OR:

CCME. (2003). *Canadian Water Quality Guidelines for the Protection of Aquatic Life: Aluminum*. Retrieved from Winnipeg, MB:

CECM. (2014). *Studies on the Effect of Aluminum in the Survival and Reproduction of Ceriodaphnia dubia at Different pHs and Hardness* (Final Report (Data summarized in Gensemer et al. 2018)). Retrieved from Santiago, Chile:

Collie, N., & Lathrop, E. W. (1976). Chemical characteristics of the standing water of a vernal pool on the Santa Rosa Plateau, Riverside County, California. In S. Jain (Ed.), *Vernal Pools, their Ecology and Conservation* (pp. 27-32). Davis, California: University of California, Davis, Institute of Ecology Publications.

Columbia River DART. Data Access in Real Time. Retrieved from http://www.cbr.washington.edu/dart

Colway, C., & Stevenson, D. E. (2007). Confirmed records of two green sturgeon from the Bering Sea and Gulf of Alaska. *Northwestern Naturalist, 88*(3), 188-193.

Connor, W. P., & Burge, H. L. (2003). Growth of wild subyearling fall Chinook salmon in the Snake River. *North American Journal of Fisheries Management, 23*(2), 594-599.

Connor, W. P., Burge, H. L., & Waitt, R. (2002). Juvenile life history of wild fall Chinook salmon in the Snake and Clearwater Rivers. *North American Journal of Fisheries Management, 22*, 703-712.

Connor, W. P., Sneva, J. G., Tiffan, K. F., Steinhorst, R. K., & Ross, D. (2005). Two alternative juvenile life history types for fall Chinook salmon in the Snake River Basin. *Transactions of the American Fisheries Society, 134*(2), 291-304.

Cooperman, M., & Markle, D. F. (2003). Rapid out-migration of Lost River and shortnose sucker larvae from in-river spawning beds to in-lake rearing grounds. *Transactions of the American Fisheries Society, 132*, 1138-1153.

Cooperman, M. S., & Markle, D. F. (2004). Abundance, size, and feeding success of larval shortnose suckers and Lost River suckers from different habitats of the littoral zone of Upper Klamath Lake. *Environmental Biology of Fishes, 71*, 365-377.

COSEWIC. (2011a). *Assessment and Status Report on the Eulachon, Nass/Skeena Rivers Population, Central Pacific Coast Population, and the Fraser River Population, Thaleichthys pacificus*. Retrieved from Ottowa, ON: http://publications.gc.ca/collections/collection_2012/ec/CW69-14-638-2011-eng.pdf

COSEWIC. (2011b). *Assessment and Status Report on the Oregon Spotted Frog Rana pretiosa in Canada*. Retrieved from Ottawa, ON:

Counihan, T. D., Hardiman, J. M., & Waste, S. (2013). *Status and trends monitoring of the mainstem Columbia River - sample frame development and review of programs relevant to the development of an integrated approach to monitoring*. Retrieved from Cook, WA:

Coutant, C. C., & Whitney, R. R. (2006). Chapter 6: Hydroelectric systems and migration. In R. N. Williams (Ed.), *Return to the River: Restoring Salmon Back to the Columbia River* (First ed., pp. 249-323). San Diego, CA: Elsevier Academic Press.

Cushman, K. A., & Pearl, C. A. (2007). *A Conservation Assessment for the Oregon Spotted Frog (Rana pretiosa)*. Retrieved from OR and WA:

CWR. (2016, December 31, 2016). Southern resident killer whale population. Retrieved from http://www.whaleresearch.com/orca-population

Dauble, D. D., & Geist, D. R. (2000). Comparison of mainstem spawning habitats for two populations of fall Chinook salmon in the Columbia River Basin. *Regulated Rivers: Research & Management, 16*(4), 345-361.

Dauble, D. D., Johnson, R. L., & Garcia, A. P. (1999). Fall chinook salmon spawning in the trailraces of Lower Snake River hydroelectric projects. *Transactions of the American Fisheries Society, 128*(4), 672-679.

Dauble, D. D., Johnson, R. L., Mueller, R. P., Abernethy, C. S., Evans, B. J., & Geist, D. R. (1994). *Identification of Fall Chinook Salmon Spawning Sites Near Lower Snake River Hydroelectric projects*. Retrieved from Walla Walla, WA:

Dauble, D. D., Johnson, R. L., Mueller, R. P., & Geist, D. R. (1995). *Identification of fall Chinook Salmon Spawning Sites Near Lower Snake River Hydroelectric Projects, 1994-1995 Season*. Retrieved from Walla Walla, WA:

DeForest, D. K., Brix, K. V., Tear, L. M., & Adams, W. J. (2018). Multiple linear regression models for predicting chronic aluminum toxicity to freshwater aquatic organisms and developing water quality guidelines. *Environmental Toxicology and Chemistry, 37*(1), 80-90. doi:10.1002/etc.3922

Delonay, A. J. (1991). *The effects of low pH and elevated aluminum on survival, growth, ionic composition, and behavior of early life stages of golden trout.* (Master of Science), University of Missouri, Columbia, MO.

Delonay, A. J., Little, E. E., Woodward, D. F., Brumbaugh, W. G., Farag, A. M., & Rabeni, C. F. (1993). Sensitivity of early-life-stage golden trout to low pH and elevated aluminum. *Environmental Toxicology and Chemistry, 12*(7), 1223-1232. doi:10.1002/etc.5620120711

DFO. (2008). Shrimp Survey Bulletin 08-02. Retrieved from http://www-ops2.pac.dfo-mpo.gc.ca/xnet/content/shellfish/shrimp/Surveys/0802.pdf

DOI. (2011). *Biological Assessment of Potential Effects to Middle Columbia River Summer Steelhead from the Taneum Canal Company Fish Bypass Repair, Kittitas County, Washington*. Retrieved from Yakima, WA:

Donald, D. B. (1983). Erratic occurrence of anostracans in a temporary pond: colonization and extinction or adaptation to variations in annual weather? *Canadian Journal of Zoology, 61*(7), 1492-1498.

Donald, D. B., & Alger, D. J. (1993). Geographic distribution, species displacement, and niche overlap for lake trout and bull trout in mountain lakes. *Canadian Journal of Zoology, 71*(2), 238-247. doi:10.1139/z93-034

Duarte, C. M. (2002). The future of seagrass meadows. *Environmental Conservation, 29*(2), 192-206.

Dumbauld, B. R., Holden, D. L., & Langness, O. P. (2008a). Do sturgeon limit burrowing shrimp populations in Pacific Northwest estuaries? *Environ Biol Fish, 83*, 283-296.

Dumbauld, B. R., Holden, D. L., & Langness, O. P. (2008b). Do sturgeon limit burrowing shrimp populations in Pacific Northwest estuaries? *Environmental Biology of Fishes, 83*(3), 283-296.

Ellsworth, C. M., Tyler, T. J., VanderKooi, S. P., & Markle, D. F. (2009). *Patterns of Larval Sucker Emigration from the Sprague and Lower Williamson Rivers of the Upper Klamath Basin, Oregon, Prior to the Removal of Chiloquin Dam - 2006 Annual Report*. Retrieved from Reston, VA:

Endangered Species Act of 1973, As Amended, U.S. Fish and Wildlife Service, Department of the Interior and National Marine Fisheries Services, National Oceanic and Atmospheric Administration, Department of Commerce, 50 C.F.R. § 402 Stat.

Eng, L. L., Belk, D., & Eriksen, C. H. (1990). Californian Anostraca: Distribution, Habitat, and Status. *Journal of Crustacean Biology, 10*(2), 247-277. doi:10.1163/193724090x00078

ENSR. (1992a). *Acute Toxicity of Aluminum to Ceriodaphnia dubia Under Static Renewal Test Conditions at Four Levels of Water Hardness*. Retrieved from Golden, CO:

ENSR. (1992b). *Chronic Toxicity of Aluminum to Ceriodaphnia dubia Under Static Renewal Test Conditions at Four Levels of Water Hardness* (Doc. No. 8505-092-047). Retrieved from Golden, CO:

Erickson, A. W. (1978). *Population Studies of Killer Whales (Orcinus orca) in the Pacific Northwest: A Radio-Marking and Tracking Study of Killer Whales*. Retrieved from Washington, DC:

ESRI. (2019). EPA_Level_III_Ecoregions (FeatureServer). Retrieved from https://services2.arcgis.com/FiaPA4ga0iQKduv3/arcgis/rest/services/EPA_Level_III_Ecoregions/FeatureServer

European A1 Association. (2009). *Systematic Characterization of the Relationship Between BLM Parameters and Aluminum Toxicity in Daphnia magna, Ceriodaphnia dubia and Pseudokirchneriella subcapitata* (Draft of the Final Report). Retrieved from Santiago, Chile:

European A1 Association. (2010). *Effect of Different Test Media Composition in A1 Acute and Chronic Toxicity* (Draft Report). Retrieved from Santiago, Chile:

Everest, F. H., & Chapman, D. W. (1972). Habitat selection and spatial interaction by juvenile Chinook salmon and steelhead trout in two Idaho streams. *Journal of the Fisheries Research Board of Canada, 29*(1), 91-100.

Ford, J., & M. Ellis, G. (2006). Selective foraging by fish-eating killer whales Orcinus orca in British Columbia. *Marine Ecology Progress Series, 316*, 185-199. doi:10.3354/meps316185

Ford, J., Wright, B., Ellis, G. M., & Candy, J. R. (2010). *Chinook Salmon Predation by Resident Killer Whales: Seasonal and Regional Selectivity, Stock Identity of Prey, and Consumption Rates* (Canadian Science Advisory Research Document 2009/101). Retrieved from Nanaimo, BC:

Ford, J. K. B. (2006). *An Assessment of Critical Habitats of Resident Killer Whales in Waters off the Pacific Coast of Canada*. Retrieved from Nanaimo, BC:

Ford, J. K. B., Ellis, G. M., & Balcomb, K. C. (2000). *Killer Whales: The Natural History and Genealogy of Orcinus orca in British Columbia and Washington* (2nd ed.). Vancouver, BC: UBC Press and University of Washington Press.

Ford, J. K. B., Ellis, G. M., Barrett-Lennard, L. G., Morton, A. B., Palm, R. S., & Balcomb, K. C., III. (1998). Dietary specialization in two sympatric populations of killer whales (*Orcinus orca*) in coastal British Columbia and adjacent waters. *Canadian Journal of Zoology, 76*, 1456-1471.

Ford, M. J., Albaugh, A., Barnas, K., Cooney, T., Cowen, J., Hard, J. J., . . . Weitkamp, L. A. (2011a). *Status Review Update for Pacific Salmon and Steelhead Listed Under the Endangered Species Act: Pacific Northwest* (NOAA Technical Memorandum NMFS-NWFSC-113). Retrieved from Seattle, WA:

Ford, M. J., Albaugh, A., Barnas, K., Cooney, T., Cowen, J., Hard, J. J., . . . Weitkamp, L. A. (2011b). *Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Pacific Northwest. Technical Memorandum NMFS-NWFSC-113*. Retrieved from Seattle, WA:

Ford, M. J., Hanson, M. B., Hempelmann, J. A., Ayres, K. L., Emmons, C. K., Schorr, G. S., . . . Balcomb-Bartok, K. (2011). Inferred Paternity and Male Reproductive Success in a Killer Whale (Orcinus orca) Population. *Journal of Heredity, 102*(5), 537-553. doi:10.1093/jhered/esr067

Fraley, J. J., & Shepard, B. B. (1989). Life history, ecology and population status of migratory bull trout (Salvelinus confluentus) in the Flathead Lake and River System, Montana. *Northwest Science, 63*(4), 133-143.

Friedman, S. (2019, September 18, 2019). [Review Comments on EPA's Draft Oregon aluminum Biological Evaluation, Status of the Vernal Pool Fairy Shrimp].

Fullerton, A. H., Lindley, S. T., Pess, G. R., Feist, B. E., Steel, E. A., & McElhany, P. (2011). Human influence on the spatial structure of threatened Pacific salmon metapopulations. *Conservation Biology, 25*(5), 932-944.

Fulton, L. A. (1968). *Spawning areas and abundance of Chinook salmon, Oncorhynchus tshawytscha, in the Columbia River basin – past and present*. Retrieved from

Ganssle, D. (1966). Ecological studies of the Sacramento-San Joaquin estuary; Part 1,: Zooplankton, zoobenthos, and fishes of San Pablo and Suisun Bays, zooplankton and zoobenthos of the Delta. *Fish Bulletin, 133*, 64-94.

Garland, T., Bennett, A. F., & Rezende, E. L. (2005). Phylogenetic approaches in comparative physiology. *Journal of experimental Biology, 208*(16), 3015-3035.

Geer, K., & Foulk, P. (2000). Endangered: Vernal pools and fairy shrimp. *Waterfowl, 12*(3), 28.

Gensemer, R. W., Gondek, J. C., Rodriquez, P. H., Arbildua, J. J., Stubblefield, W. A., Cardwell, A. S., . . . Nordheim, E. (2018). Evaluating the effects of pH, hardness, and dissolved organic carbon on the toxicity of aluminum to freshwater aquatic organisms under circumneutral conditions. *Environmental Toxicology and Chemistry, 37*(1), 49-60. doi:10.1002/etc.3920

Gensemer, R. W., & Playle, R. C. (1999). The bioavailability and toxicity of aluminum in aquatic environments. *Critical Reviews in Environmental Science and Technology, 29*(4), 315-450. doi:10.1080/10643389991259245

Gerstung, E. R. (1986). *Fishery Management Plan for Lahontan Cutthroat Trout (Salmo clarki henshawi) in California and Western Nevada Waters*. Retrieved from Sacramento, CA:

Goetz, F. (1989). *Biology of the Bull Trout, Salvelinus confluentus: A Literature Review*. Eugene, OR: U.S. Forest Service, Willamette National Forest.

Goetz, F., Jeanes, E., & Beamer, E. (2004). *Bull Trout in the Nearshore* (Preliminary draft). Retrieved from Seattle, WA:

Good, T. P., Waples, R. S., Adams, P., Bjorkstedt, E. P., Boughton, D., Cooney, T., . . . Weitkamp, L. A. (2005). *Updated Status of Federally Listed ESUs of West Coast Salmon and Steelhead* (NOAA Technical Memorandum NMFS-NWFSC-66). Retrieved from Springfield, VA: https://swfsc.noaa.gov/publications/fed/00749.pdf

Gundersen, D. T., Bustaman, S., Seim, W. K., & Curtis, L. R. (1994). pH, hardness, and humic acid influence aluminum toxicity to rainbow trout (Oncorhynchus mykiss) in weakly alkaline waters. *Canadian Journal of Fisheries and Aquatic Sciences, 51*(6), 1345-1355. doi:10.1139/f94-134

Gustafson, R. G., Ford, M. J., Teel, D., & Drake, J. S. (2010). *Status Review of Eulachon (Thaleichthys pacificus) in Washington, Oregon, and California* (NOAA Technical Memorandum NMFS-NWFSC-105). Retrieved from Springfield, VA: https://www.nwfsc.noaa.gov/assets/25/1093_06162010_142619_EulachonTM105WebFinal.pdf

Gustafson, R. G., Wainwright, T. C., Winans, G. A., Waknitz, F. W., Parker, L. T., & Waples, R. S. (1997). *Status Review of Sockeye Salmon from Washington and Oregon* (NOAA Technical Memorandum NMFS-NWFSC-33). Retrieved from Seattle, WA:

Hallock, L. A. (2013). *Draft. State of Washington Oregon spotted frog recovery plan*. Retrieved from Olympia, WA:

Hannah, R. W., & Jones, S. A. (2007). Effectiveness of bycatch reduction devices (BRDs) in the ocean shrimp (*Pandalus jordani*) trawl fishery. *Fisheries Research, 85*, 217-225.

Hanson, B., Emmons, C., Sears, M., & Ayres, K. (2010). *Prey Selection by Southern Resident Killer Whales in Inland Waters of Washington During the Fall and Early Winter* (Unpublished report, draft). Retrieved from

Hanson, M., Baird, R., Ford, J., Hempelmann, J., Van Doornik, D., R. Candy, J., . . . Balcomb, K. (2010). Species and stock identification of prey eaten by endangered southern resident killer whales in their summer range. *Endangered Species Research, 11*, 69-82. doi:10.3354/esr00263

Hanson, M. B., & Emmons, C. K. (2010). *Annual Residency Patterns of Southern Resident Killer Whales in the Inland Waters of Washington and British Columbia* (Revised draft, Workshop 1). Retrieved from

Hanson, M. B., Emmons, C. K., Ward, E. J., Nystuen, J. A., & Lammers, M. O. (2013). Assessing the coastal occurrence of endangered killer whales using autonomous passive acoustic recorders. *The Journal of the Acoustical Society of America, 134*(5), 3486-3495.

Hauser, D. D. W., Logsdon, M. G., Holmes, E. E., VanBlaricom, G. R., & Osborne, R. W. (2007). Summer distribution patterns of southern resident killer whales Orcinus orca: core areas and spatial segregation of social groups. *Marine Ecology Progress Series, 351*, 301-310.

Hay, D., & McCarter, P. B. (2000). *Status of the Eulachon Thaleichthys pacificus in Canada*. Retrieved from Nanaimo, BC:

Hayes, M. P. (1994). *The Spotted Frog (Rana pretiosa) in Western Oregon, Part II - Current Status* (Technical Report 94-1-01). Retrieved from Salem, OR:

Hayes, M. P., Engler, J. D., Van Leuven, S., Friesz, D. C., Quinn, T., & Pierce, D. J. (2001). *Overwintering of the Oregon Spotted Frog (Rana pretiosa) at Conboy Lake National Wildlife Refuge, Klickitat County, Washington, 2000-2001. Final report to: Washington Department of Transportation*. Retrieved from Olympia, WA:

Healey, M. C. (1991). Life history of chinook salmon (*Oncorhynchus tshawytscha*). In C. Groot & L. Margolis (Eds.), *Pacific Salmon Life Histories* (pp. 311-394). Vancouver, BC: UBC Press.

Hedgecock, D. (1994). Does variance in reproductive success limit effective population sizes of marine organisms. *Genetics and evolution of aquatic organisms, 122*, 122-134.

Hegg, J. C., Kennedy, B. P., Chittaro, P. M., & Zabel, R. W. (2013). Spatial structuring of an evolving life-history strategy under altered environmental conditions. *Oecologia, 172*, 1017-1029.

Hendrixson, H. A., Janney, E. C., & Shively, R. S. (2003). *Monitoring of Lost River and Shortnose Suckers at Upper Klamath Lake Non-Spawning Locations*. Retrieved from Klamath Falls, OR: http://pubs.er.usgs.gov/publication/70179837

Hickie, B. E., Ross, P. S., MacDonald, R. W., & Ford, J. K. B. (2007). Killer whales (*Orcinus orca*) face protracted health risks associated with lifetime exposure to PCBs. *Environmental Science & Technology, 41*, 6613-6619.

Higgs, D. A., Macdonald, J. S., Levings, C. D., & Dosanjh, B. S. (2010). Nutrition and feeding habits in relation to life history stage. In C. Groot, L. Margolius, & W. C. Clarke (Eds.), *Physical Ecology of Pacific Salmon* (pp. 161-261). University of British Columbia, Vancouver, BC: UBC Press.

Hobson, W. A., & Dahlgren, R. A. (1996). *Soil forming processes in vernal pools of northern California, Chico area.* . Paper presented at the California Native Plant Society: Ecology, Conservation, and Management of Vernal Pool Ecosystems. , Sacramento, California.

Hoelscher, B., & Bjornn, T. C. (1989). *Habitat, Density, and Potential Production of Trout and Char in Pend Oreille Lake Tributaries* (Project F-710R-10, Subproject III, Job No. 8.). Retrieved from Boise, ID:

Holland, R. F., & Jain, S. (1988). Vernal pools. In M. E. Barbour & J. Major (Eds.), *Terrestrial Vegetation of California, New Expanded Edition* (Vol. Publication Number 9, pp. 515-533). Sacremento, CA: California Native Plant Society.

Holtze, K. E. (1983). *Effects of pH and Ionic Strength on Aluminum Toxicity to Early Developmental Stages of Rainbow trout (Salmo gairdneri Richardson)*. Retrieved from Rexdale, ON:

Howell, P. J., & Buchanan, D. V., (Eds). (1992). *Proceedings of the Gearhart Mountain Bull Trout Workshop*. Corvallis, OR: Oregon Chapter of the American Fisheries Society.

ICTRT. (2005). *Memorandum: Updated population delineation in the Interior Columbia Basin*. Retrieved from http://www.nwfsc.noaa.gov/trt/col_docs/updated_population_delineation.pdf

IDEQ. (2001). *Middle Salmon River-Panther Creek subbasin assessment and TMDL*. Retrieved from Boise, ID:

IDEQ. (2011). *Idaho's 2010 Integrated Report* (Final). Retrieved from Boise, ID:

IDEQ, & EPA. (2003). *South Fork Clearwater River Subbasin Assessment and Total Maximum Daily Loads*. Retrieved from Boise, ID:

Ingersoll, C. G., Mount, D. R., Gulley, D. D., Point, T. W. L., & Bergman, H. L. (1990). Effects of pH, aluminum, and calcium on survival and growth of eggs and fry of brook trout (Salvelinus fontinalis). *Canadian Journal of Fisheries and Aquatic Sciences, 47*(8), 1580-1592. doi:10.1139/f90-180

Interior Columbia Technical Recovery Team. (2003). *Independent populations of Chinook, steelhead, and sockeye for listed evolutionarily significant units within the Interior Columbia River Domain*. Retrieved from

ISAB. (2007). *Climate Change Impacts on Columbia River Basin Fish and Wildlife* (ISAB 2007-2). Retrieved from Portland, OR:

Israel, J. A., & May, B. (2007). *Mixed Stock Analysis of Green Sturgeon from Washington State Coastal Aggregations*. Retrieved from Davis, CA:

Janney, E. C., Shively, R. S., Hayes, B. S., & Barry, P. M. (2008). Demographic analysis of Lost River sucker and shortnose sucker populations in Upper Klamath Lake, Oregon. *Transactions of the American Fisheries Society, 137*, 1812-1825.

Jennings, S. R., D.R. Neuman, & Blicker., P. S. (2008). *Acid Mine Drainage and Effects on Fish Health and Ecology: A Review*. Retrieved from Bozeman, MT:

Kimball, G. (1978). *The Effects of Lesser Known Metals and One Organic to Fathead Minnows (Pimephales promelas) and Daphnia magna*. Retrieved from Minneapolis, MN:

Kime, D. E. (1995). The effects of pollution on reproduction in fish. *Reviews in Fish Biology and Fisheries, 5*(1), 52-95.

Krahn, M. M., Ford, M. J., Perrin, W. F., Wade, P. R., Angliss, R. P., Hanson, M. B., . . . Waples, R. S. (2004). *2004 Status Review of Southern Resident Killer Whales (Orcinus orca) Under the Endangered Species Act* (NOAA Technical Memorandum NMFS-NWFSC-62). Retrieved from Seattle, WA: http://www.nmfs.noaa.gov/pr/species/

Krahn, M. M., Hanson, M. B., Baird, R. W., Boyer, R. H., Burrows, D. G., Emmons, C. K., . . . Collier, T. K. (2007). Persistent organic pollutants and stable isotopes in biopsy samples (2004/2006) from southern resident killer whales. *Marine Pollution Bulletin, 54*, 1903-1911.

Krahn, M. M., Hanson, M. B., Schorr, G. S., Emmons, C. K., Burrows, D. G., Bolton, J. L., . . . Ylitalo, G. M. (2009). Effects of age, sex and reproductive status on persistent organic pollutant concentrations in "southern resident" killer whales. *Marine Pollution Bulletin, 58*, 1522-1529.

Krahn, M. M., Wade, P. R., Kalinowski, S. T., Dahlheim, M. E., Taylor, B. L., Hanson, M. B., . . . Waples, R. S. (2002). *Status Review of Southern Resident Killer Whales (Orcinus orca) Under the Endangered Species Act* (NOAA Technical Memorandum NMFS-NWFSC-54). Retrieved from Springfield, VA:

Krapu, G. L. (1974). Foods of breeding pintails in North Dakota. *The Journal of Wildlife Management, 38*(3), 408-417. doi:10.2307/3800870

Lachmuth, C. L., Alava, J. J., Hickie, B. E., Johannessen, S. C., Macdonald, R. W., Ford, J. K. B., . . . Ross, P. S. (2010). *Ocean Disposal in Resident Killer Whale (Orcinus orca) Critical Habitat: Science in Support of Risk Management* (Research Document 2010/116). Retrieved from Sidney, BC:

LaLone, C. A., Villeneuve, D. L., Lyons, D., Helgen, H. W., Robinson, S. L., Swintek, J. A., . . . Ankley, G. T. (2016). Sequence Alignment to Predict Across Species Susceptibility

(SeqAPASS): A Web-Based Tool for Addressing the Challenges of Cross-Species Extrapolation of Chemical Toxicity. *Toxicological Sciences, 153*(2), 228-245. doi:10.1093/toxsci/kfw119

Langer, O. E., Shepherd, B. G., & Vroom, P. R. (1977). *Biology of the Nass River eulachon (Thaleichthys pacificus)* (Technical report servies PAC/T-77-10). Retrieved from

LCFRB. (2010). *Washington Lower Columbia Salmon Recovery and Fish and Wildlife Subbasin Plan* (Final report). Retrieved from Olympia, WA:

Leary, R. F., & Allendorf, F. W. (1997). Notes: Genetic Confirmation of Sympatric Bull Trout and Dolly Varden in Western Washington. *Transactions of the American Fisheries Society, 126*(4), 715-720. doi:10.1577/1548-8659(1997)126<0715:NGCOSB>2.3.CO;2

Lewis, A. F. J., McGurk, M. D., & Galesloot, M. G. (2002). *Alcan's Kemano River Eulachon (Thaleichthys pacificus) Monitoring Program 1988-1998*. Retrieved from Kitimat, BC:

Licht, L. E. (1974). Survival of embryos, tadpoles, and adults of the frogs Rana aurora aurora and Rana pretiosa pretiosa sympatric in southwestern British Columbia. *Canadian Journal of Zoology, 52*(5), 613-627. doi:10.1139/z74-079

Lindley, S. T., Erickson, D. L., Moser, M. L., Langness, O. P., McCovey Jr, B. W., Belchick, M., . . . Klimley, A. P. (2011). Electronic tagging of green sturgeon reveals population structure and movement among estuaries. *Transactions of the American Fisheries Society, 140*(1), 108-122.

Lindley, S. T., Moser, M. L., Erickson, D. L., Belchick, M., Welch, D. W., Rechisky, E. L., . . . Klimley, A. P. (2008). Marine migration of North American green sturgeon. *Transactions of the American Fisheries Society, 137*(1), 182-194.

Loring, S. J., MacKay, W. P., & Whitford, W. G. (1988). Ecology of small desert playas. In J. L. Thame & C. D. Ziebel (Eds.), *Small Water Impoundments in Semi-Arid Regions* (pp. 89-113). Albuquerque, NM: University of New Mexico Press.

Lowery, E. D., & Beauchamp, D. A. (2015). Trophic ontogeny of fluvial bull trout and seasonal predation on Pacific salmon in a riverine food web. *Transactions of the American Fisheries Society, 144*, 724-741.

Magnuson-Stevens Act Provisions, National Marine Fisheries Services, National Oceanic and Atmospheric Administration, Department of Commerce, 50 C.F.R. § 600.810 Stat. (1976a).

Magnuson-Stevens Act Provisions, National Marine Fisheries Services, National Oceanic and Atmospheric Administration, Department of Commerce, 50 C.F.R. § 600.10 Stat. (1976b).

Malea, P., & Haritonidis, S. (1996). Toxicity and uptake of aluminium by the seagrass Halophila stipulacea (forsk.) aschers and in response to aluminium exposure. *Fresenius Environmental Bulletin, 5*, 345-350.

Mantua, N., Tohver, I., & Hamlet, A. (2010). Climate change impacts on streamflow extremes and summertime stream temperature and their possible consequences for freshwater salmon habitat in Washington State. *Climatic Change, 102*, 187-223.

Markle, D. F., & Dunsmoor, L. K. (2007). Effects of habitat volume and fathead minnow introduction on larval survival of two endangered sucker species in Upper Klamath Lake, Oregon. *Transactions of the American Fisheries Society, 136*, 567-579.

Martin, B. A., & Saiki, M. K. (1999). Effects of ambient water quality on the endangered Lost River sucker in Upper Klamath Lake, Oregon. *Transactions of the American Fisheries Society, 128*, 953-961.

Matthews, J. R., & Moseley, C. J., (Eds). (1990). *The Official World Wildlife Fund Guide to Endangered Species of North America* (Vol. 1: Plants, Mammals, 2: Birds, Reptiles, Amphibians, Fishes, Mussels, Crustaceans, Snails, Insects, and Srachnids). Washington, DC: Beacham Publications.

McAllister, K. R., & Leonard, W. P. (1997). *Washington State status Report for the Oregon Spotted Frog*. Retrieved from Olympia, WA:

McAllister, K. R., & White, H. Q. (2001). *Oviposition Ecology of the Oregon Spotted Frog at Beaver Creek, Washington*. Retrieved from Olympia, WA:

McCauley, D. J., Brooke, L. T., Call, D. J., & Lindberg, C. (1986). *Acute and Chronic Toxicity of Aluminum to Ceriodaphnia dubia at Various pH's*. Retrieved from Superior, WI:

McFarlane, G. A., King, J. R., & Beamish, R. J. (2000). Have there been recent changes in climate? Ask the fish. *Progress in Oceanography, 47*, 147-169.

McPhail, J. D., & Baxter, J. S. (1996). *A Review of Bull Trout (Salvelinus confluentus) Life-history and Habitat Use in Relation to Compensation and Improvement Opportunities* (Fisheries Management Report #104). Retrieved from Vancouver, BC:

Mebane, C. A., & Arthaud, D. L. (2010). Extrapolating growth reductions in fish to changes in population extinction risks: copper and Chinook salmon. *Human and Ecological Risk Assessment, 16*(5), 1026-1065.

Metro. (2014). *2014 Urban Growth Report: Investing in our Communities 2015-2035*. Retrieved from Portland, OR:

Minakawa, N., & Kraft, G. F. (1999). Fall and Winter Diets of Juvenile Coho Salmon in a Small Stream and an Adjacent Pond in Washington State. *Journal of Freshwater Ecology, 14*(2), 249-254. doi:10.1080/02705060.1999.9663676

Morace, J. L. (2012). *Reconnaissance of Contaminants in Selected Wastewater-Treatment-Plant Effluent and Stormwater Runoff Entering the Columbia River, Columbia River Basin, Washington and Oregon, 2008-10* (Scientific investigations report 2012-5068). Retrieved from Reston, VA:

Morrison, J., Quick, M. C., & Foreman, M. G. G. (2002). Climate change in the Fraser River watershed: flow and temperature projections. *Journal of Hydrology, 263*, 230-244.

Moser, M., & Lindley, S. T. (2007). Use of Washington estuaries by subadult and adult green sturgeon. *Environmental Biology of Fishes, 79*, 243-253.

Mote, P. W., & Salathé, E. P. (2010). Future climate in the Pacific Northwest. *Climatic Change, 102*(1-2), 29-50.

Moyle, P. B. (2002). *Inland Fishes of California: Revised and Expanded*. Berkeley, CA: University of California Press.

Moyle, P. B., van Dyck, C. M., & Tomelleri, J. (2002). Inland fishes of California. *The Quarterly Review of Biology, 77*(4), 470-496.

Mueller, R. P. (2009). *Survey of Fall Chinook Salmon Spawning Downstream of Lower Snake River Hydroelectric Projects* (PNWD-4056). Retrieved from Walla Walla, WA:

Mullan, J. W., Rockhold, A., & Chrisman, C. R. (1992). Communications: Life histories and precocity of Chinook salmon in the mid-Columbia River. *Progressive Fish-Culturist, 54*(1), 25-28.

Nickelson, T. E., Rogers, J. D., Johnson, S. L., & Solazzi, M. F. (1992). Seasonal changes in habitat use by juvenile coho salmon (*Oncorhynchus kisutch*) in Oregon coastal streams. *Canadian Journal of Fisheries and Aquatic Sciences, 49*, 783-789.

Nilsen, E., & Morace, J. (2014). Foodweb transfer, sediment transport, and biological impacts of emerging and legacy organic contaminants in the lower Columbia River, Oregon and Washington, USA: USGS Contaminants and Habitat (ConHab) Project. *Science of the Total Environment, 484*, 319-321.

Nilsen, E., Zaugg, S., Alvarez, D., Morace, J., Waite, I., Counihan, T., . . . Grove, R. (2014). Contaminants of legacy and emerging concern in largescale suckers (*Catostomus macrocheilus*) and the foodweb in the lower Columbia River, Oregon and Washington, USA. *Science of the Total Environment, 484*, 344-352.

NMFS. (1992a). Threatened status for Snake River spring/summer Chinook salmon, threatened status for Snake River fall Chinook salmon. *Federal Register, 57*, 14653-14663.

NMFS. (1992b). Threatened status for Snake River spring/summer Chinook salmon, threatened status for Snake River fall Chinook salmon (Correction). *Federal Register, 57*, 23458.

NMFS. (1993). Designated critical habitat; Snake River Sockeye salmon, spring/summer Chinook salmon, and Snake River fall Chinook salmon. *Federal Register, 58*, 68543.

NMFS. (1997). 62 FR 43937 - Endangered and threatened species: Listing of several Evolutionary Significant Units (ESUs) of west coast steelhead. *Federal Register, 62*(159), 43937-43954.

NMFS. (1999a). Designated critical habitat: Revision of critical habitat for Snake River spring/summer Chinook salmon. *Federal Register, 64*, 57399-57403.

NMFS. (1999b). Designated critical habitat; Central California coast and southern Oregon/northern California coasts Coho salmon. *Federal Register, 64*, 24049-24062.

NMFS. (1999c). Endangered and threatened species: Threatened status for two ESUs of steelhead in Washington and Oregon. *Federal Register, 64*(57), 14517-14528.

NMFS. (1999d). Endangered and threatened species; Threatened status for three Chinook salmon Evolutionarily Significant Units (ESUs) in Washington and Oregon, and endangered status for one Chinook salmon ESU in Washington. *Federal Register, 64*(56), 14308-14328.

NMFS. (1999e). Endangered and threatened wildlife and plants; Listing of nine Evolutionarily Significant Units of Chinook salmon, Chum salmon, Sockeye salmon, and steelhead. *Federal Register, 64*, 41835-41839.

NMFS. (2000). *Biological Opinion. Reinitiation of Consultation on Operation of the Federal Columbia River Power System, Including the Juvenile Fish Transportation Program, and 19 Bureau of Reclamation Projects in the Columbia Basin*. Retrieved from Seattle, WA:

NMFS. (2005a). *Appendix F.5. Essential Fish Habitat Assessment Report for the Salmon Fisheries in the EEZ off the Coast of Alaska*. Retrieved from Juneau, AK:

NMFS. (2005b). Endangered and threatened species: Final listing determinations for 16 ESUs of west coast salmon, and final 4(d) protective regulations for threatened salmonid ESUs. *Federal Register, 70*, 37160-37204.

NMFS. (2005c). Endangered and threatened species; Designation of critical habitat for 12 Evolutionarily Significant Units of west coast salmon and steelhead in Washington, Oregon, and Idaho. *Federal Register, 70*, 52630-52858.

NMFS. (2006a). Endangered and threatened species: Final listing determinations for 10 distinct population segments of west coast steelhead. *Federal Register, 71*, 834-862.

NMFS. (2006b). Endangered and threatened species; Designation of critical habitat for southern resident killer whale. *Federal Register, 71*, 69054-69070.

NMFS. (2006c). Endangered and threatened wildlife and plants: Threatened status for Southern Distinct population segment of North American green sturgeon. *Federal Register, 71*, 17757-17766.
NMFS. (2007). *2007 Report to Congress, Pacific Coastal Salmon Recovery Fund FY 2000–2006*. Retrieved from Seattle, WA:
NMFS. (2008a). Endangered and threatened species: Final threatened listing determination, final protective regulations, and final designation of critical habitat for the Oregon coast Evolutionarily Significant Unit of Coho salmon. *Federal Register, 73*, 7816-7873.
NMFS. (2008b). *Recovery Plan for Southern Resident Killer Whales (Orcinus orca)*. Retrieved from Seattle, WA:
NMFS. (2008c). *Recovery Plan Module: Mainstem Columbia River Hydropower Projects*. Retrieved from
NMFS. (2009a). *Biological and Conference Opinion on the Long-Term Operations of the Central Valley Project and State Water Project*. Retrieved from St. Petersburg, FL:
NMFS. (2009b). *Middle Columbia River Steelhead Distinct Population Segment ESA Recovery Plan*. Retrieved from
NMFS. (2010a). Endangered and threatened wildlife and plants: Threatened status for southern distinct population segment of eulachon. *Federal Register, 75*, 13012-13024.
NMFS. (2010b). Endangered and threatened wildlife and plants: Threatened status for southern distinct population segment of eulachon. *Federal Register, 75*, 30714-30730.
NMFS. (2011a). *5-year Review: Summary and Evaluation of Snake River Sockeye, Snake River Spring-Summer Chinook, Snake River Fall-Run Chinook, Snake River Basin Steelhead*. Retrieved from Portland, OR:
NMFS. (2011b). *Biennial Report to Congress on the Recovery Program for Threatened and Endangered Species October 1, 2008 – September 30, 2010*. Retrieved from Washington, DC:
NMFS. (2011c). Endangered and threatened species; 5-Year reviews for 17 Evolutionarily Significant Units and distinct population segments of pacific salmon and steelhead. *Federal Register, 76*, 50448-50449.
NMFS. (2011d). Endangered and threatened species; Designation of critical habitat for the southern distinct population segment of eulachon. *Federal Register, 76*, 65324-65352.
NMFS. (2011e). *Southern Resident Killer Whales (Orcinus orca) 5-year Review: Summary and Evaluation*. Retrieved from Seattle, WA:
NMFS. (2012a). *Endangered Species Act (ESA) Section 7(a)(2) Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act essential fish habitat (EFH) consultation. Effects of the Pacific Coast Salmon Plan Fisheries on the Lower Columbia River Chinook Evolutionarily Significant Unit* (NMFS Consultation Number: F/NWR/2011/06415). Retrieved from
NMFS. (2012b). *Revised Draft Jeopardy and Adverse Modification of Critical Habitat Biological Opinion for the Environmental Protection Agency's Approval of Certain Oregon Administrative Rules Related to Revised Water Quality Criteria for Toxic Pollutants*. Retrieved from Seattle, WA:
NMFS. (2013). *ESA Recovery Plan for Lower Columbia River Coho Salmon, Lower Columbia River Chinook Salmon, Columbia River Chum Salmon, and Lower Columbia River Steelhead*. Retrieved from

NMFS. (2014). *Final recovery plan for the Southern Oregon/Northern California Coast evolutionarily significant unit of coho salmon (Oncorhynchus kisutch)*. Retrieved from Arcata, CA:

NMFS. (2015a, January 21, 2015). Chum salmon (*Oncorhynchus keta*). Retrieved from http://www.fisheries.noaa.gov/pr/species/fish/chum-salmon.html

NMFS. (2015b). *Consultation on the Issuance of Five ESA Section 10(a)(1)(A) Scientific Research Permits Affecting Salmon, Steelhead, Eulachon, and Sturgeon in the Pacific Northwest* (NMFS Consultation Number: WCR-2014-1852). Retrieved from

NMFS. (2015c). *ESA Recovery Plan for Snake River Sockeye Salmon (Oncorhynchus nerka)*. Retrieved from Portland, OR:

NMFS. (2015d). Listing endangered or threatened species; 12-month finding on a petition to revise the critical habitat designation for the southern resident killer whale distinct population segment. *Federal Register, 80*, 9682-9687.

NMFS. (2015e). *Southern Distinct Population Segment of the North American Green Sturgeon (Acipenser medirostris), 5-year Review: Summary and Evaluation*. Retrieved from Long Beach, CA:

NMFS. (2016a). *2016 5-year Review: Summary & Evaluation of Lower Columbia River Chinook Salmon, Columbia River Chum Salmon, Lower Columbia River Coho Salmon, Lower Columbia River Steelhead*. Retrieved from Portland, OR:

NMFS. (2016b). *2016 5-Year Review: Summary & Evaluation of Snake River Sockeye Snake River Spring-Summer Chinook Snake River Fall-Run Chinook Snake River Basin Steelhead*. Retrieved from Portland, OR: https://repository.library.noaa.gov/view/noaa/17050

NMFS. (2016c, May 26, 2016). Chinook salmon (*Oncorhynchus tshawytscha*). Retrieved from http://www.nmfs.noaa.gov/pr/species/fish/chinook-salmon.html

NMFS. (2016d). Endangered and threatened species; 5-year reviews for 28 listed species of pacific salmon, steelhead, and eulachon. *Federal Register, 81*, 33468-33469.

NMFS. (2016e). Endangered and threatened species; Designation of critical habitat for Lower Columbia River Coho salmon and Puget Sound steelhead. *Federal Register, 81*, 9252-9325.

NMFS. (2016f). Steelhead trout (*Oncorhynchus mykiss*). Retrieved from http://www.fisheries.noaa.gov/pr/species/fish/steelhead-trout.html

NMFS. (2018a). *FAQs Southern Resident Killer Whales & Columbia/Snake River Chinook Salmon Stocks*. Retrieved from

NMFS. (2018b). *Recovery Plan for the Southern Distinct Population Segment of North American Green Sturgeon (Acipenser medirostris)*. Retrieved from Sacramento, CA:

NMFS. (2018c). *Southern Resident Killer Whales and West Coast Chinook Salmon*. Retrieved from https://www.westcoast.fisheries.noaa.gov/publications/protected_species/marine_mammals/killer_whales/srkw-salmon-sources-factsheet.pdf

NOAA. (2006). *Endangered Species Act Status of West Coast Salmon and Steelhead*. Retrieved from Seattle, WA:

NOAA. (2011). Salmon population summary. from SPS database https://www.webapps.nwfsc.noaa.gov/appim/appthemes/spspub1.0/help/SPS2.htm

NOAA Fisheries. (2005). *Final Assessment of NOAA Fisheries' Critical Habitat Analytical Review Teams for 12 Evolutionarily Significant Units of West Coast Salmon and*

Steelhead. Appendix B. CHART Assessment for the Lower Columbia River Chinook Salmon ESU. Retrieved from Portland, OR:

NOAA Fisheries. (2008). Endangered and threatened species: Final threatened listing determination, final protective regulations, and final designation of critical habitat for the Oregon coast evolutionarily significant unit of Coho salmon. *Federal Register, 73*, 7815-7873.

NOAA Fisheries. (2009). *Final Critical Habitat Maps for the Southern Distinct Population Segment of Green Sturgeon*. Retrieved from

NOAA Fisheries. (2012). *Jeopardy and Destruction or Adverse Modification of Critical Habitat Endangered Species Act Biological Opinion for Environmental Protection Agency's Proposed Approval of Certain Oregon Administrative Rules Related to Revised Water Quality Criteria for Toxic Pollutants* (2008/00148). Retrieved from Seattle, WA:

NOAA Fisheries. (2014). Endangered and threatened wildlife; final rule to revise the code of federal regulations for species under the jurisdiction of the National Marine Fisheries Service. *Federal Register, 79*, 20802-20817.

NOAA Fisheries. (2015a). *Proposed ESA Recovery Plan for Snake River Fall Chinook Salmon (Oncorhynchus tshawytscha)*. Retrieved from Portland, OR:

NOAA Fisheries. (2015b). *Status Review Update for Pacific Salmon and Steelhead Listed Under the Endangered Species Act: Pacific Northwest*. Retrieved from https://www.nwfsc.noaa.gov/assets/11/8623_03072016_124156_Ford-NWSalmonBioStatusReviewUpdate-Dec%2021-2015%20v2.pdf

NOAA Fisheries. (2016a). *California Current Integrated Ecosystem Assessment (CCIEA): State of the California Current Report, 2016* (NMFS Report 1). Retrieved from Long Beach, CA:

NOAA Fisheries (Cartographer). (2016b). Critical habitat Columbia River chum salmon. Retrieved from http://www.westcoast.fisheries.noaa.gov/publications/gis_maps/maps/salmon_steelhead/critical_habitat/chum/chum_col.pdf

NOAA Fisheries (Cartographer). (2016c). Critical habitat Lower Columbia River Chinook salmon. Retrieved from http://www.westcoast.fisheries.noaa.gov/publications/gis_maps/maps/salmon_steelhead/critical_habitat/chin/chinook_lcr.pdf

NOAA Fisheries (Cartographer). (2016d). Critical habitat Lower Columbia River steelhead. Retrieved from http://www.westcoast.fisheries.noaa.gov/publications/gis_maps/maps/salmon_steelhead/critical_habitat/steelhead/steelhead_lcr_ch.pdf

NOAA Fisheries (Cartographer). (2016e). Critical habitat Middle Columbia River steelhead. Retrieved from http://www.westcoast.fisheries.noaa.gov/publications/gis_maps/maps/salmon_steelhead/critical_habitat/steelhead/steelhead_mcr_ch.pdf

NOAA Fisheries (Cartographer). (2016f). Critical habitat Oregon coast coho salmon. Retrieved from http://www.westcoast.fisheries.noaa.gov/publications/gis_maps/maps/salmon_steelhead/critical_habitat/coho/coho_orc_update.pdf

NOAA Fisheries (Cartographer). (2016g). Critical habitat Snake River Basin steelhead. Retrieved from

http://www.westcoast.fisheries.noaa.gov/publications/gis_maps/maps/salmon_steelhead/critical_habitat/steelhead/steelhead_srb_ch.pdf
NOAA Fisheries (Cartographer). (2016h). Critical habitat Upper Columbia River spring-run Chinook salmon. Retrieved from http://www.westcoast.fisheries.noaa.gov/publications/gis_maps/maps/salmon_steelhead/critical_habitat/chin/chinook_ucr.pdf
NOAA Fisheries (Cartographer). (2016i). Critical habitat Upper Columbia River steelhead. Retrieved from http://www.westcoast.fisheries.noaa.gov/publications/gis_maps/maps/salmon_steelhead/critical_habitat/steelhead/steelhead_ucr_ch.pdf
NOAA Fisheries (Cartographer). (2016j). Critical habitat Upper Willamette River Chinook salmon. Retrieved from http://www.westcoast.fisheries.noaa.gov/publications/gis_maps/maps/salmon_steelhead/critical_habitat/chin/chinook_uwr.pdf
NOAA Fisheries (Cartographer). (2016k). Critical habitat Upper Willamette River steelhead. Retrieved from http://www.westcoast.fisheries.noaa.gov/publications/gis_maps/maps/salmon_steelhead/critical_habitat/steelhead/steelhead_uwr_ch.pdf
NOAA Fisheries. (2016l). Green sturgeon biology. Retrieved from http://www.westcoast.fisheries.noaa.gov/protected_species/green_sturgeon/green_sturgeon_biology.html
NOAA Fisheries. (2016m). Green sturgeon life history. Retrieved from http://www.westcoast.fisheries.noaa.gov/protected_species/green_sturgeon/green_sturgeon_life_history.html
NOAA Fisheries. (2016n). Green sturgeon threats. Retrieved from http://www.westcoast.fisheries.noaa.gov/protected_species/green_sturgeon/green_sturgeon_threats.html
NOAA Fisheries. (2016o). *Proposed ESA Recovery Plan for Snake River Spring/Summer Chinook Salmon (Oncorhynchus tshawytscha) & Snake River Steelhead (Oncorhynchus mykiss)*. Retrieved from Portland, OR:
NOAA Fisheries. (2017a, January 2017). Killer whale (*Orcinus orca*). Retrieved from http://www.nmfs.noaa.gov/pr/species/mammals/whales/killer-whale.html
NOAA Fisheries. (2017b). Killer whales identification tips and vocalizations. Retrieved from http://www.westcoast.fisheries.noaa.gov/protected_species/marine_mammals/killer_whale/id_vocalizations.html
Noren, D. P., & Hauser, D. D. W. (2016). Surface-based observations can be used to assess behavior and fine-scale habitat use by an endangered killer whale (*Orcinus orca*) population. *Aquatic Mammals, 42*(2), 168-183.
NPFMC. (2012). *Fishery Management Plan for the Salmon Fisheries in the EEz Off Alaska. Management Plan*. Retrieved from Anchorage, AK:
ODA. (2016). *A Landowner's Guide to Oregon's Agricultural Water Quality Management Program*. Retrieved from Salem, OR: http://www.oregon.gov/ODA/shared/Documents/Publications/NaturalResources/WaterQualityandAgriculture.pdf
ODEQ. (2014). 2012 303(d) List. Retrieved from https://www.deq.state.or.us/wq/assessment/rpt02/view303dlist.asp

ODEQ. (2017a). Municipal Stormwater Permits. Retrieved from https://www.oregon.gov/deq/wq/wqpermits/Pages/MS4-Permits.aspx
ODEQ. (2017b). MS4 Phase II General Permit Development. Retrieved from https://www.oregon.gov/deq/wq/wqpermits/Pages/MS4-PhaseII-General-Permit.aspx
ODEQ. (2017c). Industrial Stormwater Permits. Retrieved from https://www.oregon.gov/deq/wq/wqpermits/Pages/Stormwater-Industrial.aspx
ODEQ. (2018a). *Table Toxic Limits in Minor or NPDES Permits*. Retrieved from
ODEQ. (2018b). Water Quality Index. Retrieved from https://www.oregon.gov/deq/wq/Pages/WQI.aspx
ODFW. (1998). *Briefing Paper – Lower Columbia River Chinook ESU*. Retrieved from Portland, OR:
ODFW. (2010). *Final Lower Columbia River Conservation & Recovery Plan for Oregon Populations of Salmon & Steelhead*. Retrieved from
ODFW. (2016). *Oregon Conservation Strategy*. Retrieved from Salem, OR:
ODFW, & NMFS. (2011). *Upper Willamette River Conservation and Recovery Plan for Chinook Salmon and Steelhead*. Retrieved from https://www.westcoast.fisheries.noaa.gov/publications/recovery_planning/salmon_steelhead/domains/willamette_lowercol/willamette/will-final-plan.pdf
ODOF. (2017). *Oregon Forest Practices Act*. Retrieved from Salem, OR: http://www.oregon.gov/ODF/Working/Pages/FPA.aspx
OFRI. (2011). *Oregon's Forest Protection Laws: An Illustrated Manual*. Retrieved from Portland, OR: https://oregonforests.org/sites/default/files/2018-02/OFRI_IllusManual_full.pdf
Olesiuk, P. F., Bigg, M. A., & Ellis, G. M. (1990). Life history and population dynamics of resident killer whales (*Orcinus orca*) in the coastal waters of British Columbia and Washington State. In P. S. Hammond, S. A. Mizroch, & D. G. P (Eds.), *Report of the International Whaling Commission. Special Issue 12: Individual Recognition Of Cetaceans: Use Of Photo-Identification And Other Techniques to Estimate Population Parameters (SC/A88/ID33)* (pp. 209-243). La Jolla, CA: International Whaling Commission.
Omernik, J. M. (1987). Ecoregions of the conterminous United States. *Ann. Ass. Am. Geogr., 77*, 118:125.
Ortiz, R. M. (2001). Osmoregulation in marine mammals. *Journal of Experimental Biology, 204*(11), 1831-1844.
Osborne, R. W. (1999). *A historical ecology of Salish Sea "resident" killer whales (Orcinus orca): with implications for management.* (Ph.D.), University of Victoria, Victoria, BC.
OSU. (2012). *Early Life-stage Toxicity of Aluminum, at pH 6, to the Fathead Minnow, Pimephales promelas, Under Flow-through Conditions*. Retrieved from Albany, OR:
OSU. (2013). *Early Life-stage Toxicity of Aluminum, at pH 6, to the Zebrafish, Danio rerio, Under Flow-through Conditions*. Retrieved from Albany, OR:
Parkhurst, B. R., Bergman, H., Fernandez, J., Gulley, D. D., Russell Hockett, J., & Sanchez, D. A. (2011). Inorganic Monomeric Aluminum and pH as Predictors of Acidic Water Toxicity to Brook Trout (Salvelinus fontinalis). *Canadian Journal of Fisheries and Aquatic Sciences, 47*, 1631-1640. doi:10.1139/f90-185

Pearl, C., Clayton, D., & Turner, L. (2010). *Surveys for Presence of Oregon Spotted Frog (Rana pretiosa): Background Information and Field Methods*. Retrieved from https://pubs.er.usgs.gov/publication/70004673

Pearl, C. A., & Hayes, M. P. (2004). *Habitat Associations of the Oregon Spotted Frog (Rama pretiosa): A Literature Review* (Final Report). Retrieved from Olympia, WA:

Peck, B. J. (2000). *Radio Telemetry Studies of Adult Shortnose and Lost River Suckers in Upper Klamath Lake and Tributaries, Oregon 1993-1999* (Unpublished draft report). Retrieved from Klamath Falls, OR:

Peles, J. D. (2013). Effects of chronic aluminum and copper exposure on growth and development of wood frog (Rana sylvatica) larvae. *Aquatic Toxicology, 140-141*, 242-248. doi:10.1016/j.aquatox.2013.06.009

Pennak, R. W. (1989). *Fresh-water Invertebrates of the United States: Protozoa and Mollusca* (3rd ed.). New York, NY: Wiley.

Perkins, D. L., Kann, J., & Scoppettone, G. G. (2000). *The Role of Poor Water Quality and Fish Kills in the Decline of Endangered Lost River and Shortnose Suckers in Upper Klamath Lake* (Final Report). Retrieved from Reno, NV:

Perkins, D. L., Mace, C. E., Scoppettone, G. G., & Rissler, P. H. (1996). *Identification of Spawning Habitats Used by Endangered Borax Lake Chub (Gila boraxobius)* (Final Report to The Nature Conservancy). Retrieved from Reno, NV:

Peterson, N. P. (1982). Immigration of juvenile coho salmon (*Oncorhynchus kisutch*) into riverine ponds. *Canadian Journal of Fisheries and Aquatic Sciences, 39*, 1308-1310.

Peterson, W. T., Fisher, J. L., Morgan, C. A., Zeman, S. M., Burke, B. J., & Jacobson, K. C. (2018). *Ocean Ecosystem Indicators of Salmon Marine Survival in the Northern California Current*. Retrieved from Newport, OR:

Peven, C. M., Whitney, R. R., & Williams, K. R. (1994). Age and length of steelhead smolts from the Mid-Columbia River Basin, Washington. *North American Journal of Fisheries Management, 14*, 77-86.

PFMC. (2000). *Review of 1999 Ocean Salmon Fisheries*. Retrieved from Portland, OR:

Phillips, A. J., Ralston, S., Brodeur, R. D., Auth, T. D., Emmett, R. L., Johnson, C., & Wespestad, V. (2007). Recent pre-recruit Pacific hake (*Merluccius productus*) occurrences in the northern California current suggest a northward expansion of their spawning area. *California Cooperative Oceanic Fisheries Investigations, Progress Report, 48*, 215-229.

Pillsbury, R. W., & Kingston, J. C. (1990). The pH-independent effect of aluminum on cultures of phytoplankton from an acidic Wisconsin lake. *Hydrobiologia, 194*(3), 225-233. doi:10.1007/bf00007122

Playle, R. C., & Wood, C. M. (1989). Water pH and aluminum chemistry in the gill micro-environment of rainbow trout during acid and aluminum exposures. *Journal of Comparative Physiology B, 159*(5), 539-550. doi:10.1007/BF00694378

Poytress, W. R., Gruber, J. J., & Van Eenennaam, J. (2012). *2011 Upper Sacramento River Green Sturgeon Spawning Habitat and Larval Migration Surveys*. Retrieved from Red Bluff, CA:

Pratt, K. L. (1992). A review of bull trout life history. In P. J. Howell & D. V. Buchanan (Eds.), *Proceedings of the Gearhart Mountain Bull Trout Workshop* (pp. 5-9). Corvallis, OR: Oregon Chapter of the American Fisheries Society.

Quinn, T. P. (2005). *The Behavior and Ecology of Pacific Salmon and Trout*. Seattle, WA: American Fisheries Society in association with University of Washington Press.

Radtke, L. D. (1966). Distribution of smelt, juvenile sturgeon, and starry flounder in the Sacramento-San Joaquin Delta with observations on food of sturgeon. In J. L. Turner & D. W. Kelley (Eds.), *Ecological Studies of The Sacramento-San Joaquin Delta, Part II: Fishes of The Delta* (pp. 115-119). Sacramento, CA: California Department of Fish and Game.

Raimondo, S., Jackson, C. R., & Barron, M. G. (2010). Influence of taxonomic relatedness and chemical mode of action in acute interspecies estimation models for aquatic species. *Environmental Science & Technology, 44*(19), 7711-7716. doi:10.1021/es101630b

Rasmussen, J. E. (2011). Status of Lost River sucker and shortnose sucker. *Western North American Naturalist, 71*(4), 442-455.

Ratliff, D. E., & Howell, P. J. (1992). The status of bull trout populations in Oregon. In P. J. Howell & D. V. Buchanan (Eds.), *Proceedings of the Gearhart Mountain Bull Trout Workshop* (pp. 10-17). Corvallis, OR: Oregon Chapter of the American Fisheries Society.

Rexstad, E. A., & Pikitch, E. K. (1986). Stomach contents and food consumption estimates of Pacific hake, *Merluccius productus*. *Fishery Bulletin, 84*(4), 947-956.

Rich, C. F., Jr. (1996). *Influence of abiotic and biotic factors on occurrence of resident bull trout in fragmented habitats, western Montana.* (Masters of Science), Montana State University, Bozeman, MT.

Ricker, W. E. (1959). *Evidence for environmental and genetic influence on certain characters which distinguish stocks of the Pacific salmons and steelhead trout*. Retrieved from Nanaimo, B.C:

Ridgely, R. S., Allnutt, T. F., Brooks, T., McNicol, D. K., Mehlman, D. W., Young, B. E., & Zook, J. R. (Cartographer). (2003). Digital distribution maps of the birds of the Western Hemisphere, version 1.0

Rieman, B., Isaak, D., Adams, S., Horan, D., David, N., Luce, C., & Myers, D. (2007). Anticipated climate warming effects on bull trout and their habitats across the Interior Columbia River Basin. *Transactions of the American Fisheries Society, 136*, 1552-1565. doi:10.1577/T07-028.1

Rieman, B. E., Lee, D. C., & Thurow, R. F. (1997). Distribution, Status, and Likely Future Trends of Bull Trout within the Columbia River and Klamath River Basins. *North American Journal of Fisheries Management, 17*(4), 1111-1125. doi:10.1577/1548-8675(1997)017<1111:Dsalft>2.3.Co;2

Rieman, B. E., & McIntyre, J. D. (1993). *Demographic and Habitat Requirements of Bull Trout Salvelinus confluentus* (General Technical Report INT-GTR- 302.). Retrieved from Ogden, UT:

Rieman, B. E., & McIntyre, J. D. (1995). Occurrence of Bull Trout in Naturally Fragmented Habitat Patches of Varied Size. *Transactions of the American Fisheries Society, 124*(3), 285-296. doi:10.1577/1548-8659(1995)124<0285:oobtin>2.3.co;2

Risser, P. G. (2000). *Oregon State of the Environment Report*. Retrieved from

Roemmich, D., & McGowan, J. (1995). Climatic warming and the decline of zooplankton in the California current. *Science, 267*(5202), 1324-1325.

Rosales-Casian, J. A., & Almeda-Jauregui, C. A. (2009). Unusual occurrence of a green sturgeon, Acipenser medirostris, at El Socorro, Baja California, Mexico. *California*

Cooperative Oceanic Fisheries Investigations, Progress Report, 50(December 2009), 169-171.

Roy, R. L., & Campbell, P. G. C. (1997). Decreased toxicity of Al to Juvenile Atlantic salmon (Salmo salar) in acidic soft water containing natural organic matter: A test of the free-ion model. *Environmental Toxicology and Chemistry, 16*(9), 1962-1969. doi:10.1002/etc.5620160928

Saiki, M. K., Monda, D. P., & Bellerud, B. L. (1999). Lethal levels of selected water quality variables to larval and juvenile Lost River and shortnose suckers. *Environmental Pollution, 105*, 37-44.

Sandercock, F. K. (1991). Life history of coho salmon (*Oncorhynchus kisutch*). In C. Groot & L. Margolis (Eds.), *Pacific Salmon Life Histories* (pp. 395-346). Vancouver, BC: UBC Press.

Scheerer, P. D., & Jacobs, S. E. (2005). *Hutton Spring Tui Chub and Foskett Spring Speckled Dace Investigations*. Retrieved from Salem, OR:

Schneider, T. R., & McFarland, W. D. (1995). *Hydrologic Data and Description of a Hydrologic Monitoring Plan for the Borax Lake Area, Oregon* (USGS Open-File Report 95-367). Retrieved from Portland, OR: https://pubs.usgs.gov/of/1995/0367/report.pdf

Scoppettone, G. G., Shea, S., & Buettner, M. E. (1995). *Information on population dynamics and life history of shortnose suckers (Chasmistes brevirostris) and Lost River suckers (Deltistes luxatus) in Tule and Clear Lakes.* Retrieved from Reno, NV:

Scoppettone, G. G., & Vinyard, G. (1991). Life history and management of four endangered lacustrine suckers. In *Battle against extinction: native fish management in the American West* (pp. 359-377). Tucson, AZ: University of Arizona Press.

Scott, W. B., & Crossman, E. J. (1973). *Freshwater Fishes of Canada* (Bulletin 184). Retrieved from Ottawa, ON:

Sedell, J. R., & Everest, F. H. (1991). *Historic Changes in Pool Habitat for Columbia River Basin Salmon Under Study for TES Listing* (Draft). Retrieved from Corvallis, OR:

Sexauer, H. M., & James, P. W. (1997). Microhabitat use by juvenile trout in four streams located in the eastern Cascades, Washington. In W. C. Mackay, M. K. Brewin, & M. Monita (Eds.), *Friends of the Bull Trout Conference Proceedings* (pp. 361-370). Calgary, AB: Bull Trout Task Force (Alberta) c/o Trout Unlimited Canada.

Short, F. T., & Neckles, H. A. (1999). Review. The effects of global climate change on seagrasses. *Aquatic Botany, 63*(3-4), 169-196. doi:10.1016/S0304-3770(98)00117-X

Sigler, W. F., & Sigler, J. W. (1987). *Fishes of the Great Basin: A Natural History*. Reno, NV: University of Nevada Press.

Simovich, M. A., & Fugate, M. (1992). Branchiopod diversity in San Diego County, California, USA. *Transactions of the Western Section of the Wildlife Society, 28*, 6-14.

Slooff, W., Van Oers, J. A. M., & De Zwart, D. (1986). Margins of uncertainty in ecotoxicological hazard assessment. *Environmental Toxicology and Chemistry, 5*(9), 841-852. doi:10.1002/etc.5620050909

Smith, W. E., & Saalfeld, R. W. (1955). Studies on Columbia River smelt *Thaleichthys pacificus* (Richardson). *WDF Fisheries Research Papers, 1*(3), 3-26.

Spence, B. C., Lomnicky, G. A., Hughes, R. M., & Novitzki, R. P. (1996). *An Ecosystem Approach to Salmonid Conservation* (TR-4501-96-6057). Retrieved from

Stanley, R. A. (1974). Toxicity of heavy metals and salts to eurasian watermilfoil (Myriophyllum spicatum L.). *Archives of Environmental Contamination and Toxicology, 2*(4), 331-341. doi:10.1007/bf02047098

State of California. (2004). *Matrix of Life History and Habitat Requirements for Feather River fish species SP-F3.2 TASK 2 Coho Salmon* (FERC Project No. 2100). Retrieved from Sacramento, CA:

Stephen, C. E., Mount, D. I., Hansen, D. J., Gentile, J. H., Chapman, G. A., & Brungs, W. A. (1985). *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* (National Technical Information Service No. PB85-227049). Retrieved from Corvallis, OR: https://www.epa.gov/sites/production/files/2016-02/documents/guidelines-water-quality-criteria.pdf

Stewart, D. B., Mochnacz, N. J., Sawatzky, C. D., Carmichael, T. J., & Reist, J. D. (2007). *Fish Diets and Food Webs in the Northwest Territories: Bull trout (Salvelinus confluentus)* (Canadian manuscript report of fisheries and aquatic sciences 1488-5387, No. 2800). Retrieved from Winnipeg, MB:

Stout, H. A., Lawson, P. W., Bottom, D. L., Cooney, T. D., Ford, M. J., Jordan, C. E., . . . Williams, T. H. (2012). *Scientific Conclusions of the Status Review for Oregon Coast Coho Salmon (Oncorhynchus kisutch)* (NOAA Technical Memorandum NMFS-NWFSC-118). Retrieved from Springfield, VA: https://www.nwfsc.noaa.gov/assets/25/1916_08132012_121939_SROregonCohoTM118WebFinal.pdf

Sugnet, P. (1993). *Preliminary Compilation of Documented Distribution, Fairy Shrimp and Tadpole Shrimp Proposed for Listing*. Retrieved from Sacramento, CA:

Suter, G. W., Barnthouse, L. W., Breck, L. E., Gardner, R. H., & O'Neill, R. V. (1985). Extrapolating from laboratory to the field: How uncertain are you? . In R. D. Cardwell, R. Purdy, & R. C. Bahner (Eds.), *Aquatic Toxicology and Hazard Assessment: 7th Symposium, ASTM STP 854* (pp. 400-413). Philadelphia, PA: American Society for Testing and Materials.

Sutherland, D. (2005). *Final Forest Practices Habitat Conservation Plan*. Retrieved from Olympia, WA:

Swanson, G. A., Krapu, G. L., Bartonek, J. C., Serie, J. R., & Johnson, D. H. (1974). Advantages in mathematically weighting waterfowl food habits data. *The Journal of Wildlife Management, 38*(2), 302-307.

Tandjung, S. D. (1982). *The acute toxicity and histopathology of brook trout (Salvelinus fontinalis, Mitchill) exposed to aluminum in acid water.* (Ph.D.), Fordham University, New York, NY.

Tattersall, G. J., & Ultsch, G. R. (2008). Physiological ecology of aquatic overwintering in ranid frogs. *Biological Reviews, 83*(2), 119-140. doi:10.1111/j.1469-185X.2008.00035.x

Terwilliger, M. R., Reece, T., & Markle, D. F. (2010). Historic and recent age structure and growth of endangered Lost River and shortnose suckers in Upper Klamath Lake, Oregon. *Environmental Biology of Fishes, 89*, 239-252.

Thomas, G. (1992). *Status of Bull Trout in Montana*. Retrieved from Helena, MT:

Tiffan, K. F., & Connor, W. P. (2012). *Seasonal Use of Shallow Water Habitat in the Lower Snake River Reservoirs by Juvenile Fall Chinook Salmon* (2010-2011 Final Report of

Research). Retrieved from Walla Walla, WA: https://pubs.er.usgs.gov/publication/70044111

Tranah, G. J., & May, B. (2006). Patterns of intra- and interspecies genetic diversity in Klamath River Basin suckers. *Transactions of the American Fisheries Society, 135*, 306-316.

TRBT. (2003). *Short-term Action Plan for Lahontan Cutthroat Trout (Oncorhynchus clarki henshawi) in the Truckee River Basin. Action Plan*. Retrieved from Reno, NV:

Trites, A. W., & Donnelly, C. P. (2003). The decline of Steller sea lions Eumetopias jubatus in Alaska: a review of the nutritional stress hypothesis. *Mammal Review, 33*(1), 3-28. doi:10.1046/j.1365-2907.2003.00009.x

Tschaplinski, P. J. (1988). The use of estuaries as rearing habitats by juvenile coho salmon. In T. W. Chamberlin (Ed.), *Proceedings of the Workshop: Applying 15 Years of Carnation Creek Results*. Nanaimo, BC: Department of Fisheries and Oceans Canada, Pacific Biological Station.

USDI. (2016). *Biological Assessment: Western Oregon Proposed Resource Management Plan*. Retrieved from Portland, OR: https://www.blm.gov/or/plans/rmpswesternoregon/files/aquaba.pdf

USEPA. (1978). Water Quality Criteria. *Federal Register, 43*, 21506-2151848.

USEPA. (1998). *R.E.D. Facts: Aluminum and Magnesium Phosphide* (EPA-738-F-98-015). Retrieved from Washington, DC https://archive.epa.gov/pesticides/reregistration/web/pdf/0025fact.pdf

USEPA. (2002). *Columbia River Basin Fish Contaminant Survey 1996-1998* (EPA 901-R-02-006). Retrieved from Seattle, WA:

USEPA. (2008). *Superfund Fact Sheet: Formosa Mine Superfund Site, Douglas County Oregon*. Retrieved from Seattle, WA:

USEPA. (2011). *Drinking Water Treatment Plant Residuals Management Technical Document*. Retrieved from Washington, DC: https://www.epa.gov/sites/production/files/2015-11/documents/dw-treatment-residuals-mgmt-tech-report-sept-2011.pdf.

USEPA (Cartographer). (2013). Level III and IV Ecoregions of the Continental United States. Retrieved from https://www.epa.gov/eco-research/level-iii-and-iv-ecoregions-continental-united-states

USEPA. (2016a). Biological Evaluation Chapters for Chlorpyrifos ESA Assessment. Retrieved from https://www.epa.gov/endangered-species/biological-evaluation-chapters-chlorpyrifos-esa-assessment#executivesummary

USEPA. (2016b). Biological Evaluation Chapters for Diazinon ESA Assessment. Retrieved from https://www.epa.gov/endangered-species/biological-evaluation-chapters-diazinon-esa-assessment

USEPA. (2017). *ICIS-NPDES*. Retrieved from https://echo.epa.gov/tools/data-downloads/icis-npdes-download-summary

USEPA. (2018). *Final Aquatic Life Ambient Water Quality Criteria for Aluminum - 2018* (EPA-822-R-18-001). Retrieved from Washington, DC:

USEPA. (2019a). Aluminum Aquatic Life Criteria Applicable to Oregon Fresh Waters - Proposed Rule. Retrieved from https://www.epa.gov/wqs-tech/aluminum-aquatic-life-criteria-applicable-oregon-fresh-waters-proposed-rule

USEPA. (2019b). *Aquatic Life Criteria for Aluminum in Oregon* (EPA-HQ-OW-2016-0694). Retrieved from Washington, DC:

https://www.federalregister.gov/documents/2019/05/01/2019-08464/aquatic-life-criteria-for-aluminum-in-oregon

USEPA. (2019c). *Economic Analysis for the Proposed Rule: Aquatic Life Criteria for Aluminum in Oregon* (EPA-HQ-OW-0694-0125). Retrieved from Washington, DC:

USFWS. (1970). Appendix D - United States list of endangered native fish and wildlilife. *Federal Register, 35*, 16045-16048.

USFWS. (1975). Threaten status for three species of trout. *Federal Register, 40*, 29863-29865.

USFWS. (1982). Endangered and threatened wildlife and plants; Endangered status and critical habitat for borax lake chub (Gila boraxobius). *Federal Register, 47*, 43957-43964.

USFWS. (1985a). Determination of threatened status for Hutton tui chub and Foskett speckled dace. *Federal Register, 50*, 12302-12306.

USFWS. (1985b). Endangered and threatened wildlife and plants; Determination that the Warner Sucker is a threatened species and designation of critical habitat. *Federal Register, 50*, 39117-39123.

USFWS. (1988). Endangered and threatened wildlife and plants; determination of endangered status for the shortnose sucker and lost river sucker. *Federal Register, 53*, 27130-27134.

USFWS. (1994a). Endangered and Threatened Wildlife and Plants; Determination of endangered status for the conservancy fairy shrimp, longhorn fairy shrimp, and the vernal pool tadpole shrimp; and Threatened Status for the vernal pool fairy shrimp. *Federal Register, 59*(180), Uknown.

USFWS. (1994b). Final rule: determination of endangered status for the conservancy fairy shrimp, longhorn fairy shrimp, and the vernal pool tadpole shrimp; and threatened status for the vernal pool fairy shrimp. *Federal Register, 59*, 48136-48153.

USFWS. (1998). *Recovery Plan for the Native Fishes of the Warner Basin and Alkali Subbasin*. Retrieved from Portland, OR:

USFWS. (1999). Endangered and threatened wildlife and plants; Determination of threatened status for bull trout in the coterminous United States. *Federal Register, 64*, 58910-58933.

USFWS. (2003a). Final rule: final designation of critical habitat for four vernal pool crustaceans and eleven vernal pool plants in California and southern Oregon *Federal Register, 68*, 46684-46761.

USFWS. (2003b). *Short Term Action Plan for Lahontan cutthroat trout (Oncorhynchus clarkii henshawi)*. Retrieved from Reno, NV:

USFWS. (2005a). Final rule: final designation of critical habitat for four vernal pool crustaceans and eleven vernal pool plants in California and southern Oregon; evaluation of economic exclusions from August 2003 final designation. *Federal Register, 70*, 46924-46999.

USFWS. (2005b). *Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon* Retrieved from Portland, OR: https://www.fws.gov/sacramento/es/Recovery-Planning/Vernal-Pool/

USFWS. (2006). *Draft Recovery Plan for the Listed Species of the Rogue Valley Vernal Pool and Illinois Valley Wet Meadow Ecosystems*. Retrieved from Portland, OR:

USFWS. (2010). Endangered and threatened wildlife and plants; Revised designation of critical habitat for bull trout in the coterminous United States. *Federal Register, 75*, 2269-2431.

USFWS. (2012a). *Borax Lake Chub (Gila boraxobius) 5 Year Review: Summary and Evaluation* (OFWO File Number: 8183.0069). Retrieved from Portland, OR: https://www.fws.gov/pacific/ecoservices/endangered/recovery/documents/SignedBoraxLakeChub5-yrReview08-23-2012.pdf

USFWS. (2012b). Endangered and threatened wildlife and plants; Designation of critical habitat for lost river sucker and shortnose sucker. *Federal Register, 77*, 73739-73768.
USFWS. (2012c). *Revised Recovery Plan for the Lost River Sucker (Deltistes luxatus) and Shortnose Sucker (Chasmistes brevirostris). Recovery Plan* (First Revision. Original Version: March 1993). Retrieved from Sacramento, CA:
USFWS. (2013a). Endangered and threatened wildlife and plants; threatened status for Oregon spotted frog (Proposed Rule). *Federal Register, 78*(168), 53582-53623
USFWS. (2013b). *Lost River Sucker (Deltistes luxatus) 5-year Review: Summary and Evaluation*. Retrieved from Klamath Falls, OR:
USFWS. (2014). Endangered and Threatened Wildlife and Plants; Threatened Status for Oregon Spotted Frog. *Federal Register, 79*(168), 51657-51710.
USFWS. (2016). *Biological Opinion on the Bureau of Land Management's Approval of the Proposed Resource Management Plan for Western Oregon* (FWS Reference Number 01EOFW00-2015-F-0279). Retrieved from Portland, OR:
USFWS. (2019a). ECOS, Environmental Conservation Online System: Warner sucker (Catostomus warnerensis). Retrieved from https://ecos.fws.gov/ecp0/profile/speciesProfile?spcode=E057
USFWS. (2019b). Oregon spotted frog. Retrieved from https://www.fws.gov/wafwo/articles.cfm?id=149489589
USFWS, & OFWO. (2017). *Status review of Foskett Spring Speckled Dace (Rhinichthys osculus ssp.)*. Retrieved from Portlan, OR: https://www.dfw.state.or.us/agency/commission/minutes/17/04_april/Exhibit%20D_Attachment%206_FSD%20Review.pdf
USGCRP. (2009). Northwest. Global climate change impacts in the United States 2009 report legacy site. Retrieved from https://nca2009.globalchange.gov/northwest/index.html
USGCRP. (2018). *Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment, Volume II: Report-in-Brief*. Retrieved from Washington, DC:
USGS. (2011). Mineral Resources Data System (MRDS). Retrieved from https://mrdata.usgs.gov/mrds/.
USGS. (2015). Estimated Annual Agricultural Use for Aluminum Phosphide, 1996 (EPest-High). Retrieved from https://water.usgs.gov/nawqa/pnsp/usage/maps/show_map.php?year=1996&map=ALUMINUMPHOSPHIDE&hilo=H.
Wang, N., Ivey, C., Brunson, E., Cleveland, D., Brumbaugh, W., & Ingersoll, C. (2016). *Columbia Environmental Research Center (CERC) preliminary data summary for acute and chronic aluminum toxicity tests with freshwater mussels and amphipods* (Memorandum to Diana Eignor. Dated January 14th). Retrieved from Columbia, MO:
Wang, N., Ivey, C. D., Brunson, E. L., Cleveland, D., Ingersoll, C. G., Stubblefield, W. A., & Cardwell, A. S. (2018). Acute and chronic toxicity of aluminum to a unionid mussel (Lampsilis siliquoidea) and an amphipod (Hyalella azteca) in water-only exposures. *Environmental Toxicology and Chemistry, 37*(1), 61-69. doi:10.1002/etc.3850
Ward, E., Hanson, B., Ford, M., & Emmons, C. (2019). *NWFSC Science to Inform SRKW Distribution and Diet. Unpublished Presentation*.
Watson, G., & Hillman, T. W. (1997). Factors Affecting the Distribution and Abundance of Bull Trout: An Investigation at Hierarchical Scales. *North American Journal of Fisheries Management, 17*(2), 237-252. doi:10.1577/1548-8675(1997)017<0237:fatdaa>2.3.co;2

Watson, J. W., McAllister, K. R., & Pierce, D. J. (2003). Home ranges, movements, and habitat selection of Oregon spotted frogs (*Rana pretiosa*). *Journal of Herpetology, 2*, 292-300.

WDFW. (1997). *Washington State Salmonid Stock Inventory - Bull Trout/Dolly Varden*. Retrieved from Olympia, WA:

WDFW, & ODFW. (2001). *Washington and Oregon Eulachon Management Plan*. Retrieved from Olympia, WA:

WDFW, & ODFW. (2002). *Status Report: Columbia River Fish Runs and fisheries, 1938–1999*. Retrieved from Olympia, WA and Salem, OR:

WDFW, & ODFW. (2018). *Joint Staff Report: Stock Status And Fisheries For Fall Chinook Salmon, Coho Salmon, Chum Salmon, Summer Steelhead, And White Sturgeon*. Retrieved from https://wdfw.wa.gov/publications/02005

WDoE. (2012). *Preparing for a Changing Climate Washington State's Integrated Climate Response Strategy* (Publication No. 12-01-004). Retrieved from Olympia, WA:

Weitkamp, L. A., Wainwright, T. C., Bryant, G. J., Milner, G. B., Teel, D. J., Kope, R. G., & Waples, R. S. (1995). *Status review of coho salmon from Washington, Oregon, and California*. Retrieved from Seattle, WA:

Wilkinson, K., Campbell, P., & Couture, P. (2011). Effect of flouride complexation on aluminum toxicity towards juvenile Atlantic salmon (Salmo salar). *Canadian Journal of Fisheries and Aquatic Sciences, 47*, 1446-1452. doi:10.1139/f90-163

Williams, C. D., & Williams, A. E. (2002). *Review of the Status of Southern Oregon/Northern California Coho with Thoughts on Recovery Planning targets*. Retrieved from Medford, OR:

Williams, J. E., & Bond, C. E. (1980). Gila boraxobius, a new species of cyprinid fish from southeastern Oregon with a comparison to G. alvordensis Hubbs and Miller. *Proceeding of the Biological Society of Washington, 93*, 291-298.

Williams, J. E., & Bond, C. E. (1983). Status and life history notes on the native fishes of the Alvord Basin, Oregon and Nevada. *The Great Basin Naturalist, 43*(3), 409-420.

Williams, J. E., Stern, M. A., Munhall, A. V., & Anderson, G. A. (1990). Conservation status of threatened fishes in Warner Basin, Oregon. *The Great Basin Naturalist, 50*(3), 243-248.

Williams, J. E., & Williams, C. D. (1980). Feeding ecology of Gila boraxobius (Osteichthyes: Cyprindae) endemic to a thermal lake in southeastern Oregon. *The Great Basin Naturalist, 40*, 101-114.

Willming, M. M., Lilavois, C. R., Barron, M. G., & Raimondo, S. (2016). Acute toxicity prediction to threatened and endangered species using Interspecies Correlation Estimation (ICE) models. *Environmental Science & Technology, 50*(19), 10700-10707.

Willson, M. F., Armstrong, R. H., Hermans, M. C., & Koski, K. (2006). *Eulachon: A Review of Biology and an Annotated Bibliography* (AFSC Processed Report 2006-12). Retrieved from Juneau, AK: https://www.afsc.noaa.gov/Publications/ProcRpt/PR%202006-12.pdf

Wissmar, R., Smith, J., McIntosh, B., Li, H., Reeves, G., & Sedell, J. (1994). A history of resource use and disturbance in riverine basins of eastern Oregon and Washington (early 1800s-1990s). *Northwest Science, 68*, 1-35.

Wood, C. M., & McDonald, D. G. (1987). The physiology of acid/aluminium stress in trout. *Royal Zoological Society of Belgium, 117*(supplement 1), 399-410.

Wood, C. M., McDonald, D. G., Ingersoll, C. G., Mount, D. R., Johannsson, O. E., Landsberger, S., & Bergman, H. L. (1990). Whole body ions of brook trout (Salvelinus fontinalis) Alevins: Responses of yolk-sac and swim-up stages to water acidity, calcium, and

aluminum, and recovery effects. *Canadian Journal of Fisheries and Aquatic Sciences, 47*(8), 1604-1615. doi:10.1139/f90-182

Woodward, D. F., Farag, A. M., Mueller, M. E., Little, E. E., & Vertucci, F. A. (1989). Sensitivity of endemic Snake River cutthroat trout to acidity and elevated aluminum. *Transactions of the American Fisheries Society, 118*(6), 630-643. doi:10.1577/1548-8659(1989)118<0630:SOESRC>2.3.CO;2

Wydoski, R. S., & Whitney, R. R. (2003). *Inland Fishes of Washington* (2nd ed.). Seattle, WA: University of Washington Press.

Zabel, R. W., Scheuerell, M. D., McClure, M. M., & Williams, J. G. (2006). The interplay between climate variability and density dependence in the population viability of Chinook salmon. *Conservation Biology, 20*(1), 190-200.

Zamon, J. E., & Welch, D. W. (2005). Rapid shift in zooplankton community composition on the northeast Pacific shelf during the 1998-1999 El Nino - La Nina event. *Canadian Journal of Fisheries and Aquatic Sciences, 62*, 133-144.