U.S. ENVIRONMENTAL PROTECTION AGENCY

Biological Evaluation of EPA's Proposed Approval Action on the Swinomish Tribe's Water Quality Standards

Aquatic Life Criteria

U.S. Environmental Protection Agency, Region 10

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Table of Contents

Acronyms and Abbreviations xv
1 Consultation Summary 1-1
1.1 Consultation History 1-1
1.2 EPA's Proposed Action and Consultation Request 1-1
1.3 Location of the Swinomish Reservation and Action Area 1-2
1.4 Threatened and Endangered Species in the Action Area 1-2
1.5 Overview of the Effects of the Action 1-3
1.6 Summary of Effects Determinations 1-4
1.6.1 Criteria for toxic chemicals 1-4
1.6.2 Criteria for temperature, dissolved oxygen, and pH 1-12
1.6.3 Narrative criteria 1-12
1.6.4 Critical habitat 1-12
1.7 Swinomish Tribal WQS not part of Consultation 1-13
2 Proposed Action 2-1
2.1 Description of Action Area 2-1
2.2 Overview of Water Quality Standards 2-5
2.2.1 Water quality certification (Clean Water Act section 401) 2-6
2.3 Overview of the Derivation of EPA Aquatic Life Criteria 2-7
2.3.1 Criteria developed after 1985 2-7
2.3.2 Criteria developed before 1985 2-9
2.4 The Swinomish Tribe Water Quality Standards that EPA Proposes to Approve 2-10
2.4.1 Numeric toxics criteria 2-10
2.4.2 Numeric temperature criteria 2-15
2.4.3 Numeric dissolved oxygen criteria 2-15
2.4.4 Numeric pH criteria 2-15
2.4.5 Narrative criteria 2-16
3 Status of the Species in the Action Area 3-1
3.1 Potential Effects from Global Climate Change 3-3
3.2 Chinook Salmon (*Oncorhynchus tshawytscha*), Puget Sound ESU 3-4
3.2.1 Distribution within the action area 3-5
3.2.2 Critical habitat within the action area 3-7
3.2.3 Life history 3-8

3.2.4 Current stressors and threats 3-9
3.3 Chum Salmon (*Oncorhynchus keta*), Hood Canal Summer-Run ESU 3-9
3.3.1 Distribution 3-9
3.3.2 Critical habitat within the action area 3-10
3.3.3 Life history 3-12
3.3.4 Current stressors and threats 3-12
3.4 Steelhead (*Oncorhynchus mykiss*), Puget Sound DPS 3-13
3.4.1 Distribution within the action area 3-13
3.4.2 Critical habitat within the action area 3-13
3.4.3 Life history 3-15
3.4.4 Current stressors and threats 3-16
3.5 Bocaccio (*Sebastes paucispinis*) 3-17
3.5.1 Distribution in the action area 3-17
3.5.2 Critical habitat 3-17
3.5.3 Life history 3-19
3.5.4 Current stressors and threats 3-20
3.6 Yelloweye Rockfish (*Sebastis ruberrimus*) 3-21
3.6.1 Distribution within the action area 3-21
3.6.2 Critical habitat 3-21
3.6.3 Life history 3-21
3.6.4 Current stressors and threats 3-23
3.7 Humpback Whale (*Megaptera novaeangliae*), Mexico DPS and Central America DPS 3-23
3.7.1 Distribution 3-23
3.7.2 Presence in the action area 3-24
3.7.3 Critical habitat 3-24
3.7.4 Life history 3-24
3.7.5 Current stressors and threats 3-25
3.8 Southern Resident Killer Whale (*Orcinus orca*) DPS 3-25
3.8.1 Distribution in the action area 3-26
3.8.2 Critical habitat within the action area 3-27
3.8.3 Life history 3-28
3.8.4 Current stressors and threats 3-28
3.9 Bull Trout (*Salvelinus confluentus*) 3-29

3.9.1 Distribution in the action area 3-29
3.9.2 Critical habitat in the action area 3-31
3.9.3 Life history 3-33
3.9.4 Current stressors and threats 3-34
3.10 Marbled Murrelet (*Brachyramphus marmoratus*) 3-35
3.10.1 Distribution within the action area 3-35
3.10.2 Critical habitat 3-35
3.10.3 Life history 3-36
3.10.4 Current stressors and threats 3-37
4 Environmental Baseline 4-1
4.1 Status of the Environment 4-1
4.1.1 Introduction 4-1
4.1.2 Data and information sources 4-4
4.1.3 Padilla Bay 4-7
4.1.4 Swinomish Channel 4-13
4.1.5 Skagit Bay 4-15
4.1.6 Similk Bay 4-16
4.1.7 Freshwater habitat 4-18
4.2 Existing Facilities and Discharges 4-24
4.2.1 Regulated cleanup sites 4-24
4.2.2 Point source discharges of pollutants (NPDES program) 4-26
4.2.3 Baseline information regarding mixing zones 4-29
4.3 Ambient Water Quality In and Near the Action Area 4-30
4.3.1 Water quality assessment and TMDLs 4-31
4.3.2 Baseline concentrations and sources of metal and metalloid pollutants in and near the action area 4-32
4.3.3 Baseline concentrations and sources of organic pollutants in and near the action area 4-49
4.3.4 Baseline concentrations and sources of nonmetallic inorganic chemicals in Swinomish tribal waters 4-56
4.3.5 Baseline physical water quality 4-65
4.4 Summary of the Environmental Baseline 4-70
5 Effects of the Action 5-1
5.1 Determinations for SRKW, Humpback Whale, and Chum Salmon Based on Considerations Specific to the Action Area 5-1

5.1.1 Not likely to adversely affect determinations for SRKW and chum salmon and no effect determination for humpback whale for all fresh water criteria 5-2
5.1.2 Not likely to adversely affect determinations for SRKW and humpback whales for all marine toxics criteria 5-3
5.2 Effects Determinations by Reference for Category 1 and 2 Toxics Criteria 5-4
5.2.1 Cadmium (fresh water) 5-10
5.2.2 Chromium III (fresh water) 5-12
5.2.3 Chromium VI (fresh water) 5-12
5.2.4 Pentachlorophenol (fresh water) 5-13
5.2.5 Cadmium (marine water) 5-14
5.2.6 Copper (marine water) 5-15
5.2.7 Selenium (marine water) 5-15
5.2.8 Silver (marine water) 5-15
5.2.9 Ammonia (fresh water) 5-16
5.2.10 Copper (fresh water) 5-16
5.2.11 Selenium (fresh water) 5-18
5.2.12 Effects determinations for marbled murrelet for category 1 and 2 freshwater criteria with a detailed assessment for selenium 5-20
5.2.13 Effects Determinations for bocaccio and yelloweye rockfish, and bull trout for category 1 marine criteria 5-23
5.3 Effects Determinations for Category 3 Criteria 5-26
5.4 Effects Assessment Methodology for Category 4 Toxics Criteria 5-28
5.4.1 Considerations regarding aquatic life criteria 5-29
5.4.2 Approach to assessments 5-30
5.4.3 Tier 2 methodologies – primary baseline effects assessment 5-32
5.4.4 Tier 3 methodologies – secondary baseline effects assessment 5-43
5.4.5 Methodology for evaluating effects to prey of ESA listed species 5-48
5.5 Freshwater Criteria Effects Assessments 5-50
5.5.1 Toxic substances 5-50
5.5.2 Dissolved oxygen, pH, and temperature 5-120
5.6 Saltwater Criteria Assessments 5-124
5.6.1 Toxic substances 5-124
5.6.2 Dissolved oxygen, pH, and temperature 5-181
5.7 Multiple Routes of Exposure for Fish Species 5-199

5.7.1 Arsenic 5-200
5.7.2 Carbaryl 5-200
5.7.3 Chlorpyrifos 5-201
5.7.4 Chromium VI 5-201
5.7.5 Diazinon 5-201
5.7.6 Lead 5-201
5.7.7 Malathion 5-202
5.7.8 Nickel 5-202
5.7.9 Nonylphenol 5-202
5.7.10 Pentachlorophenol 5-203
5.7.11 Zinc 5-203
5.8 Effects to Prey of Listed Species 5-204
5.8.1 Indirect effects analysis for acrolein 5-204
5.8.2 Indirect effects analysis for ammonia 5-205
5.8.3 Indirect effects analysis for arsenic 5-205
5.8.4 Indirect effects analysis for carbaryl 5-208
5.8.5 Indirect effects analysis for chlorine 5-210
5.8.6 Indirect effects analysis for chlorpyrifos 5-213
5.8.7 Indirect effects analysis for chromium VI 5-216
5.8.8 Indirect effects analysis for cyanide 5-219
5.8.9 Indirect effects analysis for diazinon 5-220
5.8.10 Indirect effects analysis for hydrogen sulfide 5-224
5.8.11 Indirect effects analysis for iron 5-225
5.8.12 Indirect effects analysis for lead 5-226
5.8.13 Indirect effects analysis for malathion 5-229
5.8.14 Indirect effects analysis for nickel 5-230
5.8.15 Indirect effects analysis for nonylphenol 5-232
5.8.16 Indirect effects analysis for pentachlorophenol 5-235
5.8.17 Indirect effects analysis for elemental phosphorus 5-238
5.8.18 Indirect effects analysis for silver 5-239
5.8.19 Indirect effects analysis for zinc 5-241
5.8.20 Summary of indirect effects assessment conclusions 5-244
5.9 Narrative Criteria 5-245

5.10 Critical Habitat Assessments 5-246
5.10.1 Analysis of effects of criteria for toxic pollutants on critical habitat for Chinook salmon, bull trout, and killer whale 5-247
5.11 Cumulative Effects 5-249
5.11.1 Land and marine environment use change 5-250
5.11.2 Permitted discharges 5-250
5.11.3 Commercial, recreational, and subsistence fishing 5-251
5.11.4 Climate change 5-251
5.12 Effects from Interrelated/Interdependent actions 5-253
6 Essential Fish Habitat Assessment 6-1
6.1 Introduction 6-1
6.2 Essential Fish Habitat Designations 6-3
6.2.1 Coastal pelagic species 6-3
6.2.2 Pacific coast groundfish 6-4
6.2.3 Pacific coast salmon 6-4
6.3 Habitat Areas of Particular Concern 6-5
6.4 EFH near the Swinomish Reservation 6-5
7 References 7-1
Appendix A: Swinomish Tribe Water Quality Standards Code 1
Appendix B: Literature Search Process for Biological Evaluations 1
Appendix C: Effects Assessment Data for Multiple Routes of Exposure 1
Appendix D: Oregon Acute Cadmium Analysis 1
Appendix E: Freshwater Criteria Effects Assessment for Ammonia 1
Appendix F: Estimation of Acute and Chronic Toxicity Values for Rockfish for Criteria in Assessment Category 1 1
Appendix G: Freshwater Toxicity Data for Silver 1

List of Tables

Table 1. Listed Species and Critical Habitat Evaluated Relative to the Swinomish Tribe's Fresh Water and Marine Water Criteria 1-2
Table 2. EPA Effects Determinations for EPA's Proposed Approval of the Swinomish Tribe's Toxics Criteria, Fresh Water 1-6
Table 3. EPA Effects Determinations for EPA's Proposed Approval of the Swinomish Tribe's Toxics Criteria, Marine Water 1-9
Table 4. Effects Determinations for EPA's Proposed Approval of the Swinomish Tribe's Temperature, Dissolved Oxygen, and pH Criteria 1-12
Table 5. Facilities with NPDES Permits and Authorized Mixing Zones Near the Action Area 2-5
Table 6. Numeric Aquatic Life Criteria for Toxics. All values are in µg/L unless otherwise noted 2-10
Table 7. Federally Protected Species 3-1
Table 8. Puget Sound Chinook salmon preliminary population structure, abundances, and hatchery contributions (T P Good et al., 2005) 3-5
Table 9. Physical and Biological Features of Designated Steelhead Critical Habitat 3-15
Table 10. Summary of estuarine and nearshore conditions in WRIA 3 associated with the action area from Smith (2005) 4-6
Table 11. Summary of watershed and channel characteristics for streams in the action area (Beamer et al. 2013) 4-22
Table 12. Juvenile salmonid presence in action area streams as determined by electrofishing (Beamer et al. 2013) 4-22
Table 13. Cleanup Sites Within or Near the Action Area 4-24
Table 14. Chemical Concentration of March Point Discharge and Comparison to Swinomish Tribe Aquatic Life Criteria 4-25
Table 15. NPDES Permits in the Action Area 4-26
Table 16. Summary of Effluent Data for Swinomish Reservation Sewer District Shelter Bay Community WWTP 4-27
Table 17. Summary of Effluent Ammonia Data for the Swinomish Indian Tribal Community North End WWTP 4-27
Table 18. Summary of Effluent Data for the City of La Conner 4-28
Table 19. Acute WET Test Results for City of La Conner 4-29
Table 20. Summary of Effluent Data for La Conner Maritime Services 4-29
Table 21. NPDES Discharges in the Action Area with Mixing Zones 4-30
Table 22. 303(d) Listings in the Action Area 4-32
Table 23. Tissue Screening Concentrations for Assessing Ecological Risk of Bioaccumulated Chemicals to Aquatic Biota 4-32
Table 24. Hardness, Calcium and Magnesium in Surface Freshwater in the Action Area 4-34
Table 25. Summary Statistics for Arsenic in Marine Sediment and Freshwater 4-35
Table 26. Results for Arsenic in Animal Tissue 4-35
Table 27. Summary of Results for Arsenic in Eelgrass Tissue 4-36
Table 28. Summary of Results for Cadmium in Marine Sediment 4-36
Table 29. Results for Cadmium in Animal Tissue 4-36
Table 30. Summary of Results for Cadmium in Eelgrass Tissue 4-37
Table 31. Summary of Results for Chromium in Marine Sediment and Freshwater 4-38

Table 32. Summary of Results for Chromium in Animal Tissue 4-38
Table 33. Summary of Results for Chromium in Eelgrass Tissue 4-39
Table 34. Summary of Results for Copper in Marine Sediment and Freshwater 4-39
Table 35. Summary of Results for Copper in Animal Tissue 4-40
Table 36. Summary of Results for Copper in Eelgrass Tissue....4-40
Table 37. Summary of Results for Iron in Marine Sediment and Freshwater....4-41
Table 38. Summary of Results for Iron in Eelgrass Tissue....4-41
Table 39. Summary of Results for Lead in Marine Sediment and Freshwater 4-42
Table 40. Summary of Results for Lead in Animal Tissue 4-42
Table 41. Summary of Results for Lead in Eelgrass Tissue....4-43
Table 42. Summary of Results for Mercury in Marine Sediment and Freshwater 4-44
Table 43. Summary of Results for Mercury in Animal Tissue 4-44
Table 44. Summary of Results for Nickel in Marine Sediment and Freshwater 4-45
Table 45. Summary of Results for Nickel in Animal Tissue 4-45
Table 46. Summary of Results for Nickel in Eelgrass Tissue 4-45
Table 47. Summary of Selenium Results in Marine Sediment....4-46
Table 48. Results for Selenium in Animal Tissue....4-46
Table 49. Summary of Results for Silver in Marine Sediment and Freshwater 4-47
Table 50. Results for Silver in Animal Tissue....4-48
Table 51. Summary of Results for Zinc in Marine Sediment and Freshwater....4-48
Table 52. Results for Zinc in Animal Tissue 4-49
Table 53. Summary of Results for Zinc in Eelgrass Tissue....4-49
Table 54. Quantifiable Results for Banned Chemicals in Marine Sediment (Dry Weight Basis)....4-50
Table 55. Quantifiable Results for Banned Chemicals in Animal Tissue (wet basis)....4-51
Table 56. Quantifiable Results for Banned Chemicals in Freshwater 4-53
Table 57. Banned Chemicals Analyzed for but Not Detected....4-54
Table 58. Current Use Organic Chemicals that were Analyzed but Not Detected 4-55
Table 59. Results for Carbaryl in Water 4-56
Table 60. Summary of Results for Malathion in Water (Indian Slough and Sullivan Slough) 4-56
Table 61. Summary of Results for Ammonia (Total as Nitrogen) in Surface Water 4-56
Table 62. Ammonia Criteria at Maximum pH and Temperature at Ammonia Monitoring Stations 4-57
Table 63. Summary of Results for Ammonia in Marine Sediment....4-58
Table 64. Summary of Results for Chloride in Freshwater 4-58
Table 65. Summary of Results for Sulfide in Marine Sediment 4-59
Table 66. DO Data from USGS NWIS....4-62
Table 67. Summary of Results for pH in Water....4-64
Table 68. Temperatures in Fornsby Creek....4-67
Table 69. Temperatures in Snee-Oosh Creek 4-68
Table 70. Temperatures in Munks Creek....4-68
Table 71. Temperatures in Lone Tree Creek....4-69
Table 72. Temperature in Lone Tree Creek, Spring 2013 4-69
Table 73. Temperature Data from USGS NWIS....4-69
Table 74. Species Included in Swinomish and Referenced ESA Consultations for Category 1 and 2 Criteria 5-5

Table 75. Referenced ESA Consultations for Category 1 and 2 Criteria.....5-5
Table 76. Example hardness-based criteria values for cadmium in Oregon, the Swinomish Tribe and Idaho, and EPA's current cadmium criteria recommendation 5-11
Table 77. Marine cadmium criteria adopted by Oregon and the Swinomish Tribe 5-14
Table 78. Comparison between the taxonomic surrogate geomean toxicity values for rockfish (Sabastes spp.) and criteria concentrations for category 1 chemicals.....5-23
Table 79. Assessment Category 3 Criteria.....5-26
Table 80. Assessment Category 4 Criteria.....5-28
Table 81. Hierarchical Biological Evaluation Methodology for Tier 2 Assessment of Acute Water Quality Criteria 5-32
Table 82. Hierarchical Biological Evaluation Methodology for Tier 2 Assessment of Chronic Water Quality Criteria.....5-35
Table 83. Bioaccumulation potential for chemicals in this BE 5-41
Table 84. 96 hour LC50 studies with arsenite (As+3) of acceptable data quality published since 1985 5-54
Table 85. Output from Interspecies Correlation Estimation models to estimate acute arsenic toxicity to Chinook salmon, chum salmon and bull trout. 5-55
Table 86. Chemicals for which Aquatic Toxicity Data Searches were Performed in ECOTOX. 5-65
Table 87. Output from Interspecies Correlation Estimation (ICE) models to estimate acute chlorine toxicity to Chinook salmon and bull trout. 5-66
Table 88. Chlorine acute effects assessment concentration estimates for Chinook salmon and bull trout derived from ICE model estimates of 96 hour LC_{50}s. 5-66
Table 89. Predicted chlorpyrifos chronic no effect concentrations (NOECs) derived from Interspecies Correlation Estimation (ICE) modeled acute toxicity data for listed species.....5-73
Table 90. Summary of the acute toxicity data cited in EPA (1976f) for acute toxicity of H2S to fish. When multiple 96 hour LC50s are reported for a species within a single study, only the lowest 96 hour LC50 is shown.....5-80
Table 91. Lowest chronic MATCs for hydrogen sulfide for six fish species.....5-83
Table 92. The 96 hour LC50 of 2 µg/L for lake whitefish 5-84
Table 93. Output from Interspecies Correlation Estimation models to estimate acute lead toxicity to Chinook salmon, chum salmon and bull trout.....5-89
Table 94. Calculated LC50s of the Five Fish Species Tested*.....5-94
Table 95. Output from Interspecies Correlation Estimation models to estimate acute Nickel toxicity to Chinook salmon, chum salmon and bull trout.....5-95
Table 96. Interspecies correlation estimation result for acute zinc toxicity to chum salmon.....5-116
Table 97. Acute 96 hour LC50 data for fish species in the class Actinopterygii used as surrogates to assess chlorpyrifos toxicity to ESA listed species in the marine portions of the action area 5-134
Table 98. Chronic toxicity data for fish species in the class Actinopterygii used as surrogates to assess chlorpyrifos toxicity to ESA listed species in the marine portions of the action area 5-136
Table 99. Comparative Toxicity of Hexavalent Chromium to Early Life-Stages of Juveniles of Marine Fish 5-139
Table 100. Chronic Toxicity of Hexavalent Chromium to Marine Fish and Polychaetes 5-140
Table 101. Marine fish species with reported 96 hour LC_{50} values in the EPA (1985) aquatic life criteria document.....5-143

Table 102. Marine fish species with reported 96 hour LC_{50} values published after the EPA (1985) cyanide aquatic life criteria document was issued 5-143
Table 103. Summary of the acute toxicity data cited in EPA (USEPA, 1976e) for acute toxicity of H2S to freshwater fish. When multiple 96 hour LC50s are reported for a species within a single study, only the lowest 96 hour LC50 is shown 5-150
Table 104. Lowest chronic MATCs, for hydrogen sulfide for six fish species.....5-153
Table 105. Further evidence of lack of protectiveness for members of the Salmonidae evaluated by dividing the lowest measured 96 hour LC50 for each salmonid species for which empirical acute toxicity data are available by the assumed ACR for hydrogen sulfide of 8.2 5-155
Table 106. Species mean acute values for nonylphenol exposure of marine species.....5-164
Table 107. Species sensitivity distribution for the acute toxicity of zinc to marine fish.....5-179
Table 108. Estimated zinc chronic NOECs for marine fish with empirical 96 hour LC50 values but without empirical chronic NOECs.....5-180
Table 109. Species sensitivity distribution for the chronic toxicity of zinc to marine fish.....5-180
Table 110. Dissolved oxygen criteria recommendations as presented in EPA 1986 (USEPA, 1986a) ..5-184
Table 111. Acute freshwater toxicity data (µg/L) for acrolein. The proposed criterion is 3.0 µg/L. 5-204
Table 112. Chronic freshwater toxicity data (µg/L) for acrolein. The proposed criterion is 3.0 µg/L. .5-205
Table 113. Acute freshwater toxicity data (µg/L) for arsenic. The proposed criterion is 340 µg/L.....5-206
Table 114. Range of adverse arsenic acute and chronic effect assessment concentrations for categories of prey species of the ESA listed marine species within the action area (number of species in each category in parentheses). 5-207
Table 115. Acute freshwater toxicity data (µg/L) for carbaryl. The proposed criterion is 2.1 µg/L. 5-208
Table 116. Chronic freshwater toxicity data (µg/L) for carbaryl. The proposed criterion is 2.1 µg/L. .5-209
Table 117. Acute marine toxicity data (µg/L) for carbaryl. The proposed criterion is 1.6 µg/L.....5-209
Table 118. Toxicity of Chlorine to Food Items of ESA Listed Fish Species 5-211
Table 119. Range of adverse chlorine acute and chronic effect assessment concentrations for categories of prey species of the ESA listed fish species within the action area (number of species in each category in parentheses). 5-212
Table 120. Acute freshwater toxicity data (µg/L) for a range of listed species prey items. The proposed chlorpyrifos freshwater chronic criterion is 0.083 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine acute values (PCMAV) are geometric means of all species acute toxicity values. 5-213
Table 121. Chronic fresh water toxicity data (µg/L) for a range of listed species prey items. The proposed chlorpyrifos freshwater chronic criterion is 0.041 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine chronic values (PCMCV) are geometric means of all species acute toxicity values. 5-214
Table 122. Acute marine toxicity data (µg/L) for a range of listed species prey items. The proposed chlorpyrifos freshwater chronic criterion is 0.010 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine acute values (PCMAV) are geometric means of all species acute toxicity values. 5-214
Table 123. Chronic marine toxicity data (µg/L) for a range of listed species prey items. The proposed chlorpyrifos freshwater chronic criterion is 0.006 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine chronic values (PCMCV) are geometric means of all species acute toxicity values. 5-215

Table 124. Acute freshwater toxicity data (µg/L) for CrVI. The proposed criterion is 16 µg/L.....5-216
Table 125. Chronic freshwater toxicity data (µg/L) for CrVI. The proposed criterion is 11 µg/L.....5-217
Table 126. Acute marine toxicity data (µg/L) for CrVI. The proposed criterion is 1100 µg/L.....5-217
Table 127. Chronic marine (freshwater) toxicity data (µg/L) for CrVI. The proposed criterion is 50 µg/L. Because there were not marine chronic data available, chronic freshwater were used. 5-218
Table 128. Acute marine toxicity data (µg/L) for cyanide. The proposed acute criterion is 1 µg/L. 5-219
Table 129. Chronic marine toxicity data (µg/L) for cyanide. The proposed acute criterion is 1 µg/L. .5-220
Table 130. Acute freshwater toxicity data (µg/L) for a range of listed species prey items. The proposed diazinon freshwater chronic criterion is 0.17 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine acute values (PCMAV) are geometric means of all species acute toxicity values. 5-221
Table 131. Chronic fresh water toxicity data (µg/L) for a range of listed species prey items. The proposed diazinon freshwater chronic criterion is 0.17 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine chronic values (PCMCV) are geometric means of all species acute toxicity values. 5-221
Table 132. Acute marine toxicity data (µg/L) for a range of listed species prey items. The proposed diazinon freshwater chronic criterion is 0.82 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine acute values (PCMAV) are geometric means of all species acute toxicity values. 5-222
Table 133. Chronic marine toxicity data (µg/L) for a range of listed species prey items. The proposed diazinon freshwater chronic criterion is 0.82 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine chronic values (PCMCV) are geometric means of all species acute toxicity values. 5-223
Table 134. Freshwater chronic toxicity data (µg/L) for a range of listed species prey items are summarized below. The proposed hydrogen sulfide freshwater chronic criterion is 2 µg/L .Prey categories are sorted from highest PCLTV to lowest. Prey category mean acute values (PCMAV) are geometric means of all species chronic toxicity values. 5-224
Table 135. Marine toxicity data (µg/L) for a range of listed species prey items. The proposed chronic criterion is 2 µg/L. Prey category mean acute values (PCMAV) are geometric means of all species acute toxicity values. 5-224
Table 136. Range of adverse chronic effect assessment concentrations for categories of prey species of the ESA listed fish species within the action area. 1000 µg/L is the proposed criterion.....5-226
Table 137. Range of adverse lead acute and chronic effect assessment concentrations for categories of prey species of the ESA listed fish species within the action area (number of species in each category in parentheses). 5-227
Table 138. Range of adverse lead acute and chronic effect assessment concentrations for categories of prey species of the ESA listed fish species within the action area (number of species in each category in parentheses). 5-227
Table 139. Chronic freshwater toxicity data (µg/L) for malathion. The proposed criterion is 0.1 µg/L. 5-229
Table 140. Chronic marine toxicity data (µg/L) for malathion. The proposed criterion is 0.1 µg/L. 5-229
Table 141. Range of adverse nickel acute and chronic effect assessment concentrations for categories of prey species of the ESA listed fish species within the action area (number of species in each category in parentheses). 5-231

Table 142. Range of adverse nickel acute and chronic effect assessment concentrations for categories of prey species of the ESA listed fish species within the action area (number of species in each category in parentheses). 5-231
Table 143. Acute freshwater toxicity data (ug/L) for nonylphenol. The proposed criterion is 28 ug/L. 5-233
Table 144. Chronic freshwater toxicity data (ug/L) for nonylphenol. The proposed criterion is 6.6 ug/L. 5-233
Table 145. Acute marine toxicity data (ug/L) for nonylphenol. The proposed criterion is 7.0 ug/L.....5-234
Table 146. Chronic marine toxicity data (ug/L) for nonylphenol. The proposed criterion is 1.7 ug/L. 5-234
Table 147. Acute freshwater toxicity data (µg/L) for a range of listed species prey items. The proposed acute freshwater criterion is 19 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category mean acute values (PCMAV) are geometric means of all species acute toxicity values. 5-235
Table 148. Chronic freshwater toxicity data (µg/L) for a range of listed species prey items. The proposed chronic freshwater criterion is 15 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category mean chronic values (PCMCV) are geometric means of all species acute toxicity values. 5-236
Table 149. Acute marine toxicity data (µg/L) for a range of listed species prey items. The proposed acute criterion is 13 µg/L. Prey category mean acute values (PCMAV) are geometric means of all species acute toxicity values. 5-237
Table 150. Chronic toxicity data (µg/L) for a range of listed species prey items. The proposed chronic criterion is 7.9 µg/L. Prey category mean chronic values (PCMCV) are geometric means of all species acute toxicity values.....5-237
Table 151. Chronic marine toxicity data (µg/L) for a range of listed species prey items. The proposed EP marine chronic criterion is 0.10 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine acute values (PCMAV) are geometric means of all species acute toxicity values.....5-238
Table 152. Acute Freshwater toxicity data (µg/L) for a range of listed species prey items. The proposed silver freshwater acute criterion is based on the hardness adjusted toxicity value to prey species. The CMC for silver at a hardness of 100 mg CaC1003/L is 3.22 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category mean acute values (PCMAV) are geometric means of all species acute toxicity values. 5-239
Table 153. Marine toxicity data (µg/L) for a range of listed species prey items. The proposed acute criterion is 1.9 µg/L. Prey category mean acute values (PCMAV) are geometric means of all species acute toxicity values. 5-240
Table 154. Range of adverse freshwater zinc acute and chronic effect assessment concentrations for categories of prey species of the ESA listed fish species within the action area (number of species in each category in parentheses). 5-241
Table 155. Range of adverse marine zinc acute and chronic effect assessment concentrations for categories of prey species of the ESA listed fish species within the action area (number of species in each category in parentheses). 5-243
Table 156. Summary of Indirect Effects Assessments 5-244

List of Figures

Figure 1: Swinomish regulatory boundary for purposes of treatment in a similar manner as a state for administering the water quality standards program 2-2
Figure 2: Action area with bathymetry 2-3
Figure 3: Derivation of aquatic life criteria 2-9
Figure 4: Upper Skagit Chinook escapement plotted by year (WDFW Dataset) 3-6
Figure 5: Lower Skagit Chinook escapement plotted by year (WDFW Dataset) 3-6
Figure 6: Puget Sound Chinook critical habitat 3-8
Figure 7: Critical habitat for Hood Canal summer-run chum salmon. Source: NOAA Fisheries 2016b (NMFS, 2016f) 3-11
Figure 8: Critical habitat for Puget Sound steelhead trout 3-14
Figure 9: Final critical habitat for the bocaccio and yelloweye rockfish DPSs 3-18
Figure 10: Southern resident killer whale critical habitat (Data source: NOAA 2006) 3-27
Figure 11: Location of current and historic pocket estuaries and nearshore habitats along the western Fidalgo Island shoreline of Skagit Bay and within the Swinomish Reservation (Beamer et al., 2004) 3-31
Figure 12: Critical habitat for bull trout within and surrounding the action area 3-33
Figure 13: NPDES permitted discharges, cleanup sites, and water quality listings near the action area. 4-2
Figure 14: Monitoring stations used to characterize the environmental baseline 4-3
Figure 15: Bathymetry near the action area 4-5
Figure 16: Land uses near the action area 4-8
Figure 17: Eelgrass beds near the action area 4-10
Figure 18: Shoreline modifications near the action area 4-12
Figure 19: Map of Swinomish Reservation water resources from Drost (1979) 4-19
Figure 20: Location of freshwater streams and their watershed boundaries within the Swinomish Reservation (Beamer et al., 2004) 4-20
Figure 21: Wetlands near the action area 4-21
Figure 22: Salmon streams and culverts from SalmonScape 4-23
Figure 23: Marine DO data collected by the Swinomish Tribe 4-60
Figure 24: Freshwater DO data collected by the Swinomish Tribe 4-61
Figure 25: Dissolved oxygen at Skagit County boat ramp (Swinomish Channel) collected by Skagit County 4-62
Figure 26: Marine water pH data collected by the Swinomish Tribe 4-63
Figure 27: Freshwater pH data collected by the Swinomish Tribe 4-64
Figure 28: Marine temperature data collected by the Swinomish Tribe 4-66
Figure 29: Temperature at Skagit County boat ramp (Swinomish Channel) 4-67
Figure 30: Magnified detail of the Swinomish Reservation and surrounding area from Figure 3, map C of NRC (2016). Rockfish abundance was reported as being low. 5-24
Figure 31: Figure 9 from NRC (2016), “Hot spots” of yelloweye rockfish occurrence 5-24
Figure 32: Figure 11 from NRC (2016), “hot spots” of bocaccio occurrence 5-25
Figure 33: Example time-to-event curves. Curve A (left) represents acute copper toxicity (hardness = 249 mg/L) to a freshwater amphipod (Gammarus pulex) over various exposure durations, up to 96 hours(Stephenson, 1983). Curve B (right) represents chronic copper toxicity (hardness = 250 mg/L; test ID1) to Daphnia magna over various exposure durations, up to 21-days (Dave, 1984). 5-44
Figure 34: Species sensitivity distribution for the acute toxicity of nonylphenol to marine fish 5-165

Figure 35: pH ranges for outer BC continental shelf aquatic organisms as summarized in Haigh et al. (2015). Excerpted Figure 2 5-188
Figure 36: Synthesis of thresholds of ocean acidification impacts on calcifying pteropods excerpted from Bednaršek et al. (2019) 5-189
Figure 37: Echinoderm effects thresholds at a range of pH values. Sublethal larval effects at pH = 7.75 at a duration of 7 days or more (Bednaršek et al., 2021). 5-190
Figure 38: Data for two years (morning grabs) at the Skagit Boat Ramp demonstrate the lower-end pH interannual variability. Although it is likely that pH fell below the lowest values overnight through dawn, these data demonstrate that starting from 8-10AM and for the rest of the day, pH met or exceeded the lower bound of the range of the Swinomish Tribe's criteria over the two-year period. This information on daily variability demonstrates that aquatic life in the action area are not exposed to constant pH at the criteria values. 5-191
Figure 39: The HAPCs generated by the EFH Mapper Tool (seagrass and estuaries). Both are listed in the Pacific Coast Groundfish Fishery Management Plan (FMP). 6-3

Acronyms and Abbreviations

ACR	Acute-to-Chronic Ratio
Ag	Silver
As	Arsenic
ASTM	American Society for Testing and Materials
ATPase	A class of enzymes that catalyze the decomposition of adenosine triphosphate and is essential for metabolism in all known forms of life
AWQC	Ambient Water Quality Criteria
BA	Biological Assessment
BAF	Bioaccumulation Factor
BCF	Bioconcentration Factor
BiOp	Biological Opinion
BLM	Biotic Ligand Model
BMF	Biomagnification Factor
BW	Body Weight
Ca	Calcium
CCC	Criterion Continuous Concentration
Cd	Cadmium
CF	Conversion Factor
CFS	cubic feet per second
Chronic Value	A synonym for MATC in the context of water quality criteria
CMC	Criterion Maximum Concentration
CN	Cyanide
Cr	Chromium
CrIII	Trivalent Chromium
CrVI	Hexavalent Chromium
Cu	Copper
CWA	Clean Water Act
DDD	Dichlorodiphenyldichloroethane
DDE	Dichlorodiphenylethylene
DDT	Dichlorodiphenyltrichloroethane
DPS	Distinct Population Segment
dw	dry weight
EC	Effects Concentration
EC_{50}	Concentration that caused effects to 50% of the test population
ECp	Effect Concentration Percentile
EFH	Essential Fish Habitat
EPA	U.S. Environmental Protection Agency
ESA	Endangered Species Act
ESU	Evolutionary Significant Unit
FAV	Final Acute Value
FDA	Food and Drug Administration
FIR	Food Ingestion Rate of a Wildlife Species
GMAV	Genus Mean Acute Value
Hg	Mercury
ICE	Interspecies Correlation Estimates
LAA	Likely to Adversely Affect

LC_{50}	Lethal Concentration of 50%
LOAEL	Lowest Observed Adverse Effect Level
LOEC	Lowest Observed Effects Concentration
ln(X)	Natural logarithim of the number "X"
MATC	Maximum Acceptable Toxicant Concentration
Mg	Magnesium
mg/L	milligram per liter, or parts per million
Mixing Zone	An area where an effluent discharge undergoes initial dilution and is extended to cover the secondary mixing in the ambient water body
MSA	Magnuson-Stevens Fishery Conservation and Management Act
ng/L	nanogram per liter, or parts per trillion
Ni	Nickel
NLAA	Not Likely to Adversely Affect
NMFS	National Marine Fisheries Service
NOAEL	No Observed Adverse Effect Level
NOEC	No-Observed Effects Concentration
NPDES	National Pollution Discharge Elimination System
Pb	Lead
PBF	Physical and biological features
PCB	Polychlorinated Biphenyl
PCE	Primary Constituent Elements
PCP	Pentachlorophenol
RPA	Reasonable and Prudent Alternative
RPM	Reasonable and Prudent Measures
Se	Selenium
SMAV	Species Mean Acute Value
SMCV	Species Mean Chronic Value
SSD	Species Sensitivity Distribution
TMDL	Total Maximum Daily Loads
TRV	Toxicity Reference Value
TSC	Tissue screening concentration
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
WET	Whole Effluent Toxicity
WIR	Water Ingestion Rate of a Wildlife Species
WTEC	Wildlife Threshold Effect Concentration
WQS	Water Quality Standards
ww	wet weight
Zn	Zinc
µg/L	microgram per liter or parts-per-billion
7Q10	The lowest 7-day running average of streamflows occurring in a 10-yearperiod

1 Consultation Summary

1.1 Consultation History

The Swinomish Indian Tribal Community (Tribe) initially submitted water quality standards (WQS) to the U.S. Environmental Protection Agency (EPA) for review and action under Clean Water Act (CWA) section 303(c) on February 8, 2017. The Tribe revised its WQS and the Swinomish Senate adopted the revised WQS into law on April 8, 2019, and the Tribe submitted the revised WQS to EPA on April 30, 2019. The WQS incorporate all of the EPA's CWA section 304(a) recommended water quality criteria as of April 2019 for the protection of aquatic life, with the exception of aluminum.[1]

This Biological Evaluation (BE) addresses the proposed action in compliance with section 7(c) of the Endangered Species Act (ESA) of 1973, as amended. The federal action that is the subject of this BE is EPA's proposed approval of the Swinomish Tribe's WQS that are intended to protect aquatic life and aquatic-dependent wildlife. The Tribe's WQS are set forth in Title 19, Chapter 6 (Chapter 19-06) of the Swinomish Tribal code and are included in Appendix A of this BE.

EPA held conference calls with the U.S. Fish and Wildlife Service (USFWS) and National Marine Fisheries Service (NMFS) (collectively, the Services) to discuss the consultation on June 21, July 19, and December 14, 2018, and April 4, 2019. On March 9, 2018, EPA sent letters to the Services requesting confirmation of the species lists. The USFWS responded to EPA's request on March 22, 2018, and NMFS responded on March 27, 2018. An additional email follow up from EPA to NMFS occurred on April 2, 2018. EPA held an additional conference call with the Services on June 8, 2022 to provide summary information and conclusions in the BE prior to its transmittal to initiate ESA consultation.

To ensure the lists of threatened and endangered species evaluated in the BE are still accurate, consistent with 40 CFR 402.12(e), on March 7, 2022 and March 9, 2022, EPA staff accessed the NOAA Fisheries Protected Resources App managed by the West Coast Region and USFWS's Information for Planning and Consultation (IPaC), respectively. EPA's reviews of the current species lists and their designated critical habitat areas are documented in memoranda to the administrative record.

1.2 EPA's Proposed Action and Consultation Request

EPA proposes to take an approval action on the Swinomish Tribe's numeric aquatic life criteria for toxic chemicals; numeric criteria for temperature, dissolved oxygen (DO), and pH; and the narrative criteria that support the aquatic life use. Please see section 2.2 for a general regulatory overview of the WQS program, section 2.4 for a list of the Swinomish Tribe's criteria related to protection of aquatic life, and Appendix A for the Tribe's WQS code.

Based on the analyses addressed in this BE, EPA's effects determinations for the toxics criteria are summarized in Table 2 for fresh water criteria and Table 3 for marine criteria. EPA requests concurrences on the Agency's not likely to adversely affect (NLAA) determinations from the Services for the species and critical habitat under their respective jurisdictions.

For the species and critical habitat that are likely to be adversely affected (LAA) by the Agency's proposed action, EPA requests initiation of formal consultation in accordance with 50 CFR 402.14.

[1] Available at https://www.epa.gov/wqc/national-recommended-water-quality-criteria-tables

EPA requests simultaneous informal and formal consultations. This would allow the Agency to proceed towards a CWA decision on the criteria that the Services concur with a NLAA determination, while concurrently engage in formal consultations with the Services for the LAA determinations.

1.3 Location of the Swinomish Reservation and Action Area

The Swinomish Reservation, located in northwestern Washington near La Conner, includes approximately 7,450 acres of uplands and approximately 3,000 acres are tidelands within Skagit Bay and Padilla Bay (Figure 1, section 2.1). The eastern Reservation boundary extends to the historical midpoint of the former Swinomish Slough (now the Swinomish Channel). The Tribe has identified four creeks within the Reservation boundary, which originate within the Reservation and drain to Skagit Bay (Snee-Oosh Creek and Lone Tree Creek) and Swinomish Channel (Monks Creek and Fornsby Creek). Although surface freshwater is scarce and unlikely to support salmonid spawning, the small streams within the Reservation provide rearing habitat for juvenile salmonids that enter the creeks from marine waters at their mouth (Eric M Beamer et al., 2013) and foraging for adult salmonids during migration to Skagit River (Todd Mitchell, pers. com.).

Section 2.1 includes a complete description of the action area, which includes the Swinomish Reservation and 400 feet of surrounding marine waters, as well as the full width of the Swinomish Channel.

1.4 Threatened and Endangered Species in the Action Area

The BE assesses effects of the action (i.e., effects of exposure at the criteria concentrations) for six ESA-listed species for both fresh water criteria and salt water criteria, and for three additional species for salt water criteria only, as shown in Table 1.

Table 1. Listed Species and Critical Habitat Evaluated Relative to the Swinomish Tribe's Fresh Water and Marine Water Criteria

Species	Criteria May Affect Species[1]	
	Fresh Water	Marine Water
National Marine Fisheries Service		
Chinook salmon and critical habitat[2]	X	X
Steelhead	X	X
Chum salmon	X	X
Bocaccio and critical habitat		X
Yelloweye rockfish and critical habitat		X
Humpback whale		X
SRKW and critical habitat[2]	X	X
Fish and Wildlife Service		
Bull trout and critical habitat[2]	X	X
Marbled murrelet	X	X

[1] See section 2.4 for the Swinomish Tribe's aquatic life criteria.

[2] Critical habitat is present in the marine portions of the action area.

No critical habitat has been designated within the Swinomish Reservation boundary, and no critical habitat is present for fresh waters within the action area. Critical habitat is present in marine areas located within the action area but outside of the Reservation boundary.

1.5 Overview of the Effects of the Action

As defined in the ESA regulations at 50 CFR 402.02, effects of the action are all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action. A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur. Effects of the action may occur later in time and may include consequences occuring outside the immediate area involved in the action.

EPA's action to approve the Swinomish Tribe's criteria related to protection of aquatic life does not have any immediate direct effects to listed species or critical habitat, but rather has the potential to affect listed species or critical habitat in *subsequent actions* if/when the WQS are applied. Under the CWA, subsequent actions that apply the WQS include: CWA section 303(d) listing of impaired waters that exceed applicable WQS; issuance of total maximum daily loads (TMDLs) that define the sources contributing to the exceedances of WQS and the amount that must be reduced to attain the WQS; issuance of National Pollutant Discharge Elimination System (NPDES) permits to point sources to regulate the discharge of pollutants; CWA section 401 certifications of federally licensed projects to ensure compliance with WQS; and establishment of nonpoint source pollution management plans designed to reduce pollutants and meet WQS. Under the Comprehensive Environmental Response, Comprehension, and Liability Act (CERCLA or "Superfund"), WQS may be used as baseline standards when cleanup plans are developed to address contaminated sites.

The subsequent actions listed above generally apply the aquatic life criteria to *reduce* pollutant levels to attain the criteria. Therefore, the effects to listed species and critical habitat are beneficial by reducing pollutant exposure levels when the aquatic life criteria are applied in these subsequent actions. It is possible for issuance of a NPDES permit for a new or expanded discharge to increase pollutant levels above baseline conditions up to the criteria levels. However, such actions in Swinomish Reservation waters are rarely expected to occur and antidegradation requirements to protect existing water quality and conservative permitting assumptions would serve to highly restrict such discharges. Since EPA is the permitting authority, any such actions would undergo a separate ESA consultation.

Although subsequent actions applying aquatic life criteria mostly result in beneficial effects to listed species and critical habitat by improving baseline water quality, there can be residual adverse effects to listed species and critical habitat at the criteria concentration level if the criteria are not sufficiently low to avoid these effects. Much of the focus of this BE is to examine the potential residual effects to listed species from exposure to chemicals at the aquatic life criteria levels. This examination has relevance for re-issuance of NDPES permits for existing sources because the criteria are used to limit the pollutant levels in the discharge. Thus, to the extent that exposure to pollutants at criteria levels results in adverse effects to listed species, re-issuance of the NPDES permit could potentially result in adverse effects. However, as discussed above, the broader context is that the consequences of EPA's approval are solely through application of the criteria in subsequent actions that are mostly beneficial to listed species and critical habitat, and this context should frame the ESA review.

Finally, it is also important to recognize that EPA's approval of the Swinomish Tribe's aquatic life criteria does not mean reservation waters will increase in pollutant concentration up to the criteria concentrations for the reasons discussed above.

There are currently no CWA-effective WQS for protection of aquatic life for the Swinomish Reservation waters. EPA must approve the Tribe's WQS before they become applicable under the CWA. EPA has the authority and responsibility to implement CWA programs for the surface waters within the Swinomish Reservation, such as issuance of NPDES permits. In the absence of tribal WQS, EPA generally uses the state's WQS as a basis for developing NPDES permits limits. Once approved, EPA will rely on the Swinomish Tribe's WQS (as applicable) to derive permit limits for point source discharges to the Tribe's waters. In addition, the Tribe will be able to take subsequent actions to apply the EPA approved aquatic life criteria, including issuing 401 certifications for federal permits (including EPA-issued NPDES permits) and licenses.

In general, the Swinomish Tribe's aquatic life criteria are as protective or more protective than the state of Washington's criteria because the Tribe adopted EPA's updated CWA section 304(a) national recommended criteria, several of which were updated based on the outcomes of ESA consultations in Oregon and Idaho (e.g., freshwater copper, ammonia, and selenium criteria). EPA's approval of the Swinomish Tribe's aquatic life criteria will result in beneficial effects due to the application of more protective criteria to Reservation waters and the ability for the Swinomish Tribe and EPA to carry out subsequent actions to apply the WQS that reduce pollutant levels that benefit listed species and critical habitat.

As summarized in the next section, EPA has determined, based on a conservative assumption of exposure at the criterion level, that most of the Swinomish Tribe's aquatic life criteria are not likely to adversely affect listed species or critical habitat and a few criteria are likely to adversely affect listed species. For those criteria that are likely to adversely affect listed species if exposed at the criterion level, EPA does not expect the adverse effects to be significant due to limited actual exposure potential, limited number of individuals exposed in the action area, and other factors discussed in more detail in the BE.

1.6 Summary of Effects Determinations

The Swinomish WQS include several categories of criteria that support aquatic life, including aquatic life criteria for toxic chemicals; water quality criteria for temperature, dissolved oxygen (DO), and pH; and narrative criteria that specify general water quality conditions that must be met. In addition to the beneficial effects as summarized above, the following are effect determinations for each criterion assuming waters are at the criterion level and species are exposed to this concentration.

1.6.1 Criteria for toxic chemicals

The Swinomish Tribe's criteria for toxic chemicals include freshwater criteria for 41 chemicals and marine water criteria for 38 chemicals. Toxicity assessments completed for previous ESA consultations in EPA Region 10 were used for this BE when available. EPA defines four assessment categories and approaches for evaluation of toxics criteria:

1. EPA incorporates toxicity assessments by reference for criteria with NLAA concurrences and/or with no jeopardy/adverse modification determinations by the Services in recent ESA consultations on Oregon and Idaho toxic WQS.
2. EPA incorporates toxicity assessments and no jeopardy/adverse modification ESA conclusions by reference for criteria adopted by the Swinomish Tribe that are consistent

with the reasonable and prudent alternatives (RPAs) for recent ESA consultations on WQS adopted by Oregon and Idaho for toxic chemicals.

3. EPA relies on exposure assessments for criteria with current use bans in place, similar to the approach taken for recent ESA consultations on Oregon and Idaho WQS for toxic chemicals.
4. EPA provides toxicity assessments in this BE for criteria with no or unknown EPA Region 10 WQS consultation history and for selected criteria that EPA chose to update the toxicity assessments from previous consultations.

EPA effects determinations for the toxics criteria, addressed in this BE are summarized in Table 2 for fresh water criteria and Table 3 for marine criteria.

Table 2. EPA Effects Determinations for EPA's Proposed Approval of the Swinomish Tribe's Toxics Criteria, Fresh Water

Chemical	Acute/ Chronic	Chinook Salmon	Steelhead	Chum Salmon [a]	Bull Trout	SRKW [b]	Marbled Murrelet [c]
Assessment Category 1 [d]							
Cadmium	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chromium (III)	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chromium (VI)	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Pentachlorophenol	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Assessment Category 2 [e]							
Ammonia	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Copper	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Selenium	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Assessment Category 3 [f]							
Aldrin	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chlordane	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Demeton	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Dieldrin	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Endosulfan-Alpha+Beta	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Endrin	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
gamma-Hexa-chlorocyclohexane (HCH) (Lindane)	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Guthion	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Heptachlor	CMC	NLAA	NLAA	NLAA	NLAA	ONLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Heptachlor Epoxide	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

Chemical	Acute/ Chronic	Chinook Salmon	Steelhead	Chum Salmon [a]	Bull Trout	SRKW [b]	Marbled Murrelet [c]
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Methoxychlor	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Mirex	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
p,p'-Dichlorodiphenyl-trichloroethane (4,4'-DDT)	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Parathion	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Polychlorinated biphenyls	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Toxaphene	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Tributyltin (TBT)	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Assessment Category 4 [g]							
Acrolein	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Arsenic [h]	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Carbaryl	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chlorine	CMC	NLAA	NLAA	NLAA	NLAA [h]	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA [h]	NLAA	NLAA
Chlorpyrifos	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chromium VI [h]	CCC	**LAA[2]**	**LAA[2]**	NLAA	**LAA[2]**	NLAA	**LAA[2]**
Diazinon	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Hydrogen Sulfide	CCC	**LAA[1]**	**LAA[1]**	NLAA	**LAA[1]**	NLAA	NLAA
Iron	CCC	**LAA[2]**	**LAA[2]**	NLAA	**LAA[2]**	NLAA	**LAA[2]**
Lead [h]	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Malathion	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Nickel [h]	CMC	**LAA[2]**	**LAA[2]**	NLAA	**LAA[2]**	NLAA	**LAA[2]**
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

Chemical	Acute/ Chronic	Chinook Salmon	Steelhead	Chum Salmon [a]	Bull Trout	SRKW [b]	Marbled Murrelet [c]
Nonylphenol	CMC	**LAA[2]**	**LAA[2]**	NLAA	**LAA[2]**	NLAA	**LAA[2]**
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Silver [h]	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Zinc [h]	CMC	NLAA	**LAA[1]**	NLAA	**LAA[1]**	NLAA	NLAA
	CCC	NLAA	**LAA[1]**	NLAA	**LAA[1]**	NLAA	NLAA

[a] NLAA determinations for chum salmon are based on the low likelihood that the listed Hood Canal summer-run chum would enter fresh waters of the action area, as described in section 5.1.1.
[b] NLAA determinations for fresh water criteria for SRKW are based on no direct exposure to freshwater and minimal potential effects to the quantity or quality of their food source from prey that may be affected in fresh waters of the action area (see section 5.1.1).
[c] NLAA determinations for category 1 and 2 fresh water criteria for marbled murrelet are based on minimal potential effects to the quantity or quality of their food source from prey that may be affected in freshwaters of the action area and a dietary exposure assessment for selenium (section 5.2.12).
[d] EPA determinations for category 1 fresh water criteria for Chinook salmon, steelhead, and bull trout are based on the toxicity assessments in previous consultations (section 5.2).
[e] EPA determinations for category 2 criteria for Chinook salmon, steelhead, and bull trout are based on previous ESA consultation documents that include implementing RPAs in Oregon and Idaho (section 5.2).
[f] NLAA determinations for fresh water criteria for category 3 banned chemicals are based on minimal exposure potential (section 5.3).
[g] Determinations for fresh water criteria for category 4 chemicals are based on effects assessments provided in sections 5.5, 5.7, and 5.8 for Chinook salmon, steelhead, bull trout, and marbled murrelet (effects summaries for individual criteria are included in section 5.5).
[h] Consultation on these criteria was previously completed in Oregon and/or Idaho, but updated toxicity assessments are included in this BE.

LAA determinations for category 4 criteria are based on the following effects, as indicated in Table 2 (see section 5.5):
[1] Direct water-column exposure effects assuming exposure at the criterion level
[2] Effects to listed species' prey assuming prey exposure at the criterion level (referred to as indirect effects in the BE)

Table 3. EPA Effects Determinations for EPA's Proposed Approval of the Swinomish Tribe's Toxics Criteria, Marine Water

Chemical	Acute/ Chronic	Chinook Salmon	Steelhead	Chum Salmon	Bull Trout	Bocaccio	Yelloweye Rockfish	Humpback Whale [a]	SRKW [a]	Marbled Murrelet
Assessment Category 1 [b]										
Cadmium	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Copper	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Selenium	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	**LAA**	**LAA**	**LAA**	**LAA**	NLAA	NLAA	NLAA	NLAA	**LAA**
Silver	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Assessment Category 3 [c]										
Aldrin	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chlordane	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Demeton	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Dieldrin	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Endosulfan-Alpha+Beta	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Endrin	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
gamma-Hexachlorocyclo-hexane (HCH) (Lindane)	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Guthion	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Heptachlor	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Heptachlor Epoxide	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Methoxychlor	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Mirex	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

Chemical	Acute/ Chronic	Chinook Salmon	Steelhead	Chum Salmon	Bull Trout	Bocaccio	Yelloweye Rockfish	Humpback Whale [a]	SRKW [a]	Marbled Murrelet
p,p'-Dichlorodiphenyl-trichloroethane (4,4'-DDT)	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Polychlorinated biphenyls	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Toxaphene	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Tributyltin (TBT)	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Assessment Category 4 [d]										
Ammonia	CMC	Addendum pending						NLAA	NLAA	Pending
	CCC	Addendum pending						NLAA	NLAA	Pending
Arsenic [e]	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	**LAA[3]**	**LAA[3]**	**LAA[3]**	**LAA[3]**	**LAA[3]**	**LAA[3]**	NLAA	NLAA	NLAA
Carbaryl	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chlorine	CMC	**LAA[1]**	**LAA[1]**	**LAA[1]**	**LAA[1]**	**LAA[1]**	**LAA[1]**	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chlorpyrifos	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Chromium VI [e]	CMC	**LAA[2]**	**LAA[2]**	**LAA[2]**	**LAA[2]**	**LAA[2]**	**LAA[2]**	NLAA	NLAA	**LAA[2]**
	CCC	**LAA[2]**	**LAA[2]**	**LAA[2]**	**LAA[2]**	**LAA[2]**	**LAA[2]**	NLAA	NLAA	**LAA[2]**
Cyanide	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Diazinon	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Hydrogen Sulfide	CCC	**LAA[1]**	**LAA[1]**	**LAA[1]**	**LAA[1]**	**LAA[1]**	**LAA[1]**	NLAA	NLAA	NLAA
Lead [e]	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Malathion	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Nickel [e]	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Nonylphenol	CMC	**LAA[1]**	**LAA[1]**	**LAA[1]**	**LAA[1]**	**LAA[1]**	**LAA[1]**	NLAA	NLAA	NLAA
	CCC	**LAA[1]**	**LAA[1]**	**LAA[1]**	**LAA[1]**	**LAA[1]**	**LAA[1]**	NLAA	NLAA	NLAA
Pentachlorophenol [e]	CMC	**LAA[1]**	**LAA[1]**	**LAA[1]**	**LAA[1]**	**LAA[1]**	**LAA[1]**	NLAA	NLAA	NLAA

Chemical	Acute/ Chronic	Chinook Salmon	Steelhead	Chum Salmon	Bull Trout	Bocaccio	Yelloweye Rockfish	Humpback Whale [a]	SRKW [a]	Marbled Murrelet
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Phosphorus (elemental)	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
Zinc [e]	CMC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA
	CCC	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA	NLAA

[a] NLAA determinations for marine criteria for humpback whale and SRKW are based on minimal or no direct exposure to action area waters and minimal potential effects to the quantity or quality of their food source from prey that may be affected in marine waters of the action area (see section 5.1.2).
[b] EPA determinations for category 1 marine water criteria for Chinook salmon, steelhead, chum salmon, and marbled murrelet are based on the toxicity assessments in previous consultations (section 5.2) and extrapolated to assess bull trout, bocaccio, and yelloweye rockfish (section 5.2.13).
[c] NLAA determinations for marine water criteria for category 3 banned chemicals are based on minimal exposure potential (section 5.3).
[d] Determinations for marine water criteria for category 4 chemicals are based on effects assessments provided in sections 5.6, 5.7, and 5.8 (effects summaries for individual criteria are included in section 5.6).
[e] Consultation on these criteria was previously completed in Oregon, but updated toxicity assessments are included in this BE.

LAA determinations for category 4 criteria are based on the following effects, as indicated in Table 3 (see section 5.6):
[1] Direct water-column exposure effects assuming exposure at the criterion level
[2] Effects to listed species prey assuming prey exposure at the criterion level (referred to as indirect effects in BE)
[3] Dietary Exposure

1.6.2 Criteria for temperature, dissolved oxygen, and pH

The Swinomish Tribe adopted temperature, DO, and pH criteria for fresh and marine waters for protection of aquatic life, including salmonids. For fresh waters, the criteria protect juvenile salmonid rearing and bull trout foraging and migration (Swinomish WQS Code 19-06.120(B)(1)), which is identified as the most sensitive life stage use and is present on a year-round basis.

EPA evaluated the protectiveness of the Tribe's criteria for temperature, DO, and pH, relative to listed species, as described in section 5.5.2 for fresh water criteria, and section 5.6.2 for marine criteria. EPA's effects determinations are provided in Table 4. The Tribe's criteria are provided in sections 2.4.2, 2.4.3, and 2.4.4.

Table 4. Effects Determinations for EPA's Proposed Approval of the Swinomish Tribe's Temperature, Dissolved Oxygen, and pH Criteria

Parameter	Chinook Salmon, Chum Salmon, Steelhead	Bull Trout	Bocaccio, Yelloweye Rockfish	Humpback Whale, SRKW	Marbled Murrelet
Fresh water criteria[1]					
Temperature	NLAA	NLAA	n/a[2]	NE[3]	NLAA
Dissolved oxygen	NLAA	NLAA	n/a[2]	NE[3]	NLAA
pH	NLAA	NLAA	n/a[2]	NE[3]	NLAA
Marine water criteria[1]					
Temperature	NLAA	NLAA	NLAA	NLAA	NLAA
Dissolved oxygen	NLAA	NLAA	NLAA	NLAA	NLAA
pH	LAA	LAA	LAA	NLAA	LAA

[1] Criteria are included in sections 2.4.2 (temperature), 2.4.3 (dissolved oxygen), and 2.4.4 (pH)
[2] Fresh water criteria are not applicable to bocaccio and yelloweye rockfish.
[3] No effect, see section 5.1.

1.6.3 Narrative criteria

Federal regulations concerning water quality standards state that "States [and authorized tribes] should…establish narrative criteria or criteria based upon biomonitoring methods where numerical criteria cannot be established or to supplement numerical criteria" (40 CFR 131.11(b)(2)). The Swinomish Tribe has adopted narrative criteria in sections 19-06.100, 19-06.140 and 19-06.150 of the Tribal code. See section 2.4.5 for the full text of these narrative criteria.

EPA has determined that its proposed approval of the narrative criteria is **not likely to adversely affect** Chinook salmon, chum salmon, steelhead, bull trout, bocaccio, yelloweye rockfish, Southern Resident Killer Whales (SRKW), marbled murrelet and their relevant critical habitat based on the plain language of the narrative criteria, as described in section 5.9.

1.6.4 Critical habitat

The marine portions of the action area outside of the Swinomish Reservation boundary include critical habitat for Chinook, bocaccio, yelloweye rockfish, SRKW, and bull trout. The only critical habitat features that are relevant to EPA's proposed action are water characteristics and prey quality and availability.

EPA evaluated these water characteristics (effects of exposure at the criteria levels) and effects to prey quality and availability as part of the effects assessments for the species. EPA found that exposure at criteria levels in some cases was LAA listed species (*Table 3*), and because the same conditions that affect a species also affect its critical habitat, the same effects assessment applies to both the species and its critical habitat. On this basis, the effects determinations in *Table 3* for Chinook salmon, bull trout, bocaccio, yelloweye rockfish, and SRKW also apply to these species' critical habitat in the action area, as follows:

- Chinook salmon critical habitat: LAA for marine criteria for chronic arsenic, acute chlorine, acute and chronic hexavalent chromium, chronic hydrogen sulfide, acute and chronic nonylphenol, acute pentachlorophenol, and selenium; NLAA for the remaining marine criteria.
- Bull trout critical habitat: LAA for marine criteria for chronic arsenic, acute chlorine, acute and chronic hexavalent chromium, chronic hydrogen sulfide, acute and chronic nonylphenol, acute pentachlorophenol, and selenium; NLAA for the remaining marine criteria.
- Bocaccio and yelloweye rockfish critical habitat: LAA for marine criteria for chronic arsenic, acute chlorine, acute and chronic hexavalent chromium, chronic hydrogen sulfide, acute and chronic nonylphenol, and acute pentachlorophenol; NLAA for the remaining marine criteria.
- SRKW critical habitat: NLAA for all marine criteria.

Additional site-specific considerations are described in section 5.10 for the toxics criteria.

Critical habitat was also evaluated relative to the temperature, dissolved oxygen, and pH criteria in section 5.6.2, and for the narrative criteria in section 5.9. EPA determined that its proposed approval of these criteria is not likely to adversely affect critical habitat for Chinook, bocaccio, yelloweye rockfish, SRKW, and bull trout.

1.7 Swinomish Tribal WQS not part of Consultation

EPA is not taking CWA action, and therefore is not consulting, on the following Swinomish Tribe aquatic life criteria as part of this consultation:

- The chronic arsenic criterion for fresh waters
- The acute and chronic chloride criteria for fresh waters
- The acute and chronic cyanide criteria for fresh waters
- The acute and chronic mercury criteria for fresh and marine waters.

The above freshwater criteria for arsenic, cyanide, and mercury for the State of Idaho, which are the same as the Swinomish criteria, were found by NMFS and/or USFWS to jeopardize ESA-listed salmonid species. EPA has elected not to consult on these criteria at this time.

On May 20, 2019, EPA approved the Swinomish Tribe WQS that are not subject to ESA consultation. EPA's approval action included the mixing zone, compliance schedule, and antidegradation provisions. Because the mixing zone and compliance schedule provisions are not self-implementing (i.e. they would only be applied in NPDES permits), and since EPA is the NPDES permitting authority for tribal waters, after conferring with the Services, EPA has deferred ESA consultation to when permits are issued. EPA does not consult on WQS antidegradation provisions due to lack of agency discretion.

2 Proposed Action

2.1 Description of Action Area

EPA's regulatory authority is to take action under section 303(c) of the CWA, on the Swinomish Tribe's WQS applicable to the waters within the reservation boundary. The Swinomish Reservation, located in the State of Washington, north of Seattle, consists of approximately 10,450 acres of land, of which 7,450 acres are uplands and approximately 3,000 acres are tidelands. The Reservation boundary extends to the historical midpoint of the Swinomish Slough (now the Swinomish Channel), to the extreme low water tide mark of the southern and western waters surrounding the Reservation, and to a line that trends east from the extreme low water mark of Turner's Bay then north to the extreme low water mark of Padilla Bay (Figure 1). The actual Reservation boundaries may differ from those depicted in Figure 1 because the extreme low water mark is not permanently fixed.

The action area, shown in Figure 2, includes the entire width of the Swinomish Channel (on the east side of the reservation) and a 400-foot buffer around the rest of the Swinomish Tribe's waters. The ESA regulations at 50 CFR 402.02 define the action area as "all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action." Therefore, the action area, for purposes of the effects analysis in the BE, is larger than the "project area" and extends out to a point where there are no measurable effects[2] from the proposed action.

[2] The Section 7 Consultation Handbook (USFWS, 1998) states that a not likely to adversely affect determination is "the appropriate conclusion when effects on listed species are expected to be discountable, insignificant, or completely beneficial." Discountable effects are extremely unlikely to occur. "Based on best judgment, a person would not: (1) be able to meaningfully measure, detect, or evaluate insignificant effects…"

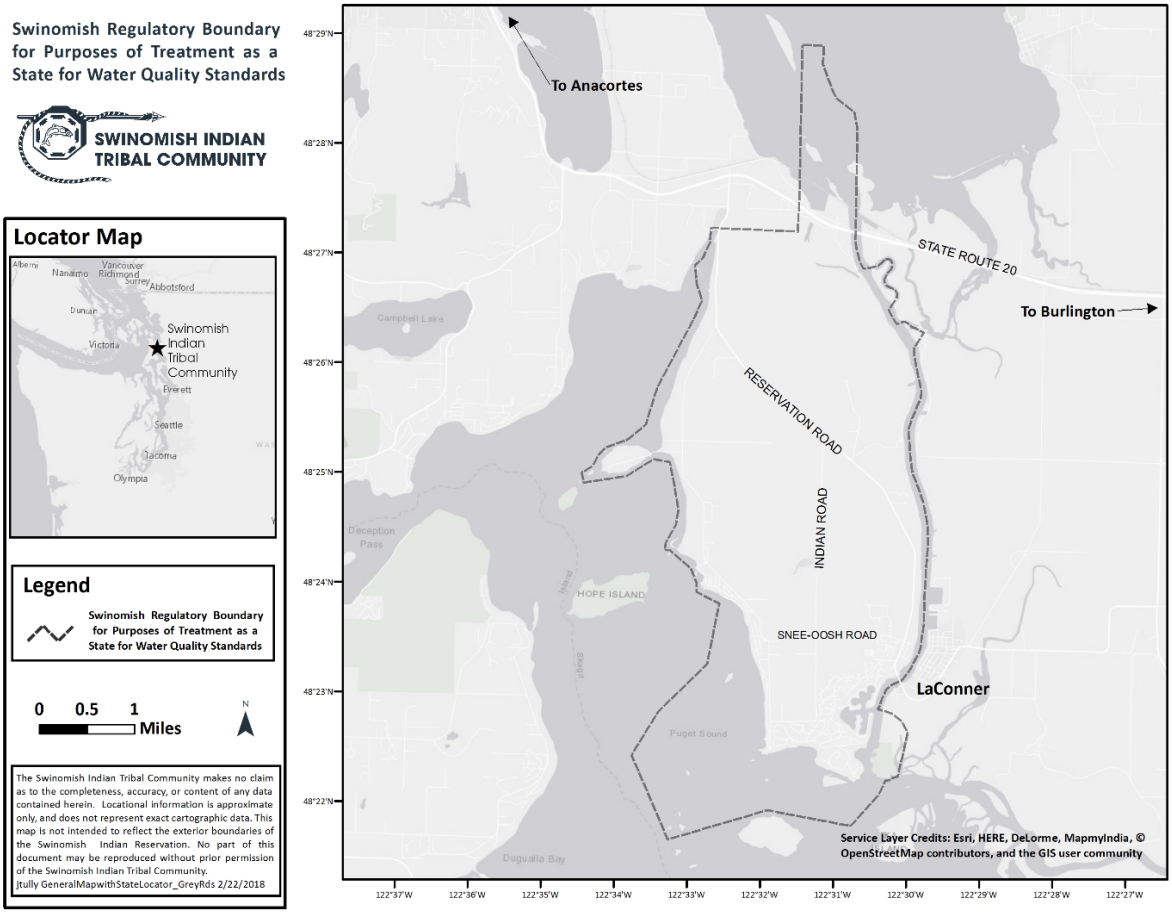


Figure 1: Swinomish regulatory boundary for purposes of treatment in a similar manner as a state for administering the water quality standards program

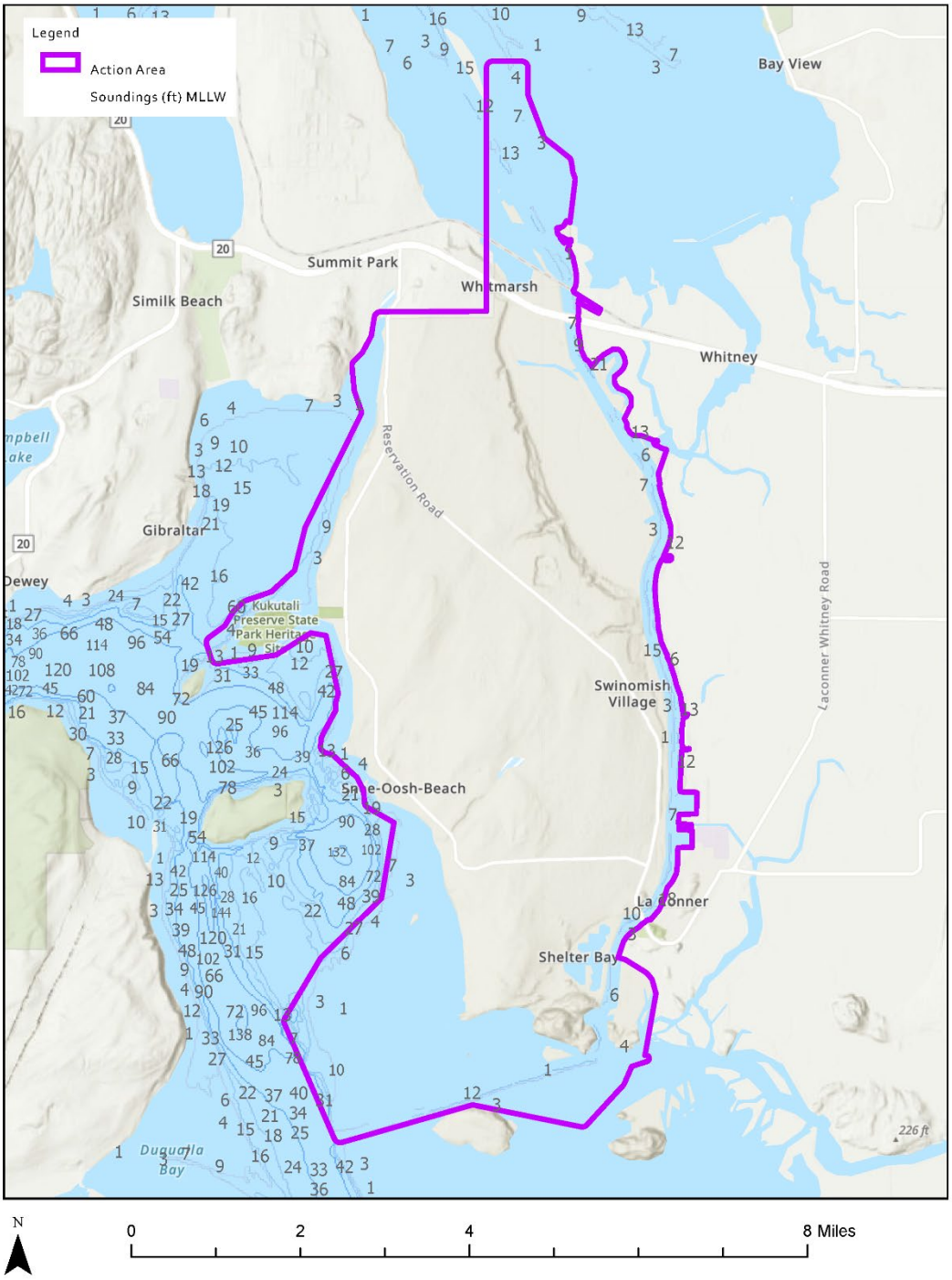


Figure 2: Action area with bathymetry

The Swinomish Reservation includes all lands and waters within the exterior boundaries of the Reservation. The Tribe has applied its water quality standards to all named and unnamed marine and fresh surface waters of the Reservation (See the definition of Regulated Surface Waters at 19-06.080(A)(40)). Surface water resources of the Swinomish Reservation include marine tidelands, an artificial marine channel, estuarine wetlands, small streams, and freshwater wetlands. The following

waters, which lie wholly or partially within the Reservation boundaries, are specifically identified in 19--06.080(A)(40) Tables 1 and 2:

Marine waters

Skagit Bay (i.e., tidelands to the extreme low water mark)
Padilla Bay (i.e., tidelands to the extreme low water mark)
Swinomish Channel (to the historic midpoint of the channel)

Fresh waters

Lone Tree Creek
Snee-Oosh Creek
Munks Creek
Fornsby Creek

The Skagit River North Fork delta also falls partly within the Reservation boundary. For estuarine waters, the more stringent of the fresh water or salt water aquatic life criteria will apply (19-06.100(E)(3)).

As noted above, EPA's CWA action applies only to the surface waters under the jurisdiction of the Swinomish Tribe. The Tribe's WQS apply to pollutant levels that may occur in these waters. Outside of these waters, Washington state standards apply (WAC 173-201A). Waters entering the reservation must meet the Tribe's WQS at the reservation boundary, and waters leaving the reservation must meet Washington's WQS at the reservation boundary. For purposes of ESA consultation on water quality standards actions, EPA has in the past defined the action area as the regulated waters of a tribe or state that are the subject of a CWA section 303(c) action by EPA.

For purposes of the effects analysis in this BE, and as requested by NMFS during early consultation coordination, EPA relies on the dilution potential of the receiving waters to account for indirect effects in delineating the action area. The indirect analysis includes three NPDES permitted discharges that are either within the Tribal boundary or discharging to the Swinomish Channel (Table 5) contained within the action area. When a permit is issued, the magnitude of dilution of the discharged pollutants is predicted using a model. This measure of reasonable dilution was used to estimate the distance from the reservation boundary where the indirect effects on the action area could be considered discountable. To be clear, this exercise was completed strictly for the analysis of indirect effects of any particular pollutant that may exist in tribal waters. Any NPDES permits or authorization of discharges by EPA to tribal waters must undergo a separate ESA consultation process.

As shown in Table 5, the dilution factor at approximately 20 and 200 feet from the source was determined for each of the three known discharges as part of the development of a mixing zone. The discharge flow rates range from 0.05 to 0.52 million gallons per day (mgd), and the distance to the edges of the chronic mixing zone range from 202 to 208 ft from the point of discharge, with dilution factors of 12 to 433 respectively. The largest discharge, from the City of La Conner, has the smallest dilution factor (12:1). As the data indicate, the level of dilution from a hypothetical point discharge at the boundary of the Tribal waters into Washington marine waters would be substantial.

Table 5. Facilities with NPDES Permits and Authorized Mixing Zones Near the Action Area

Permit	Permit Number	Flow Rate (mgd)	Acute MZ Distance	Acute Dilution Factor	Chronic MZ Distance	Chronic Dilution Factor
Swinomish Indian Tribal Community North End WWTP	WA0025062	0.05	20.25 ft	23.9	202.5 ft	433
Swinomish Shelter Bay WWTP	WA0024422	0.227	21.4 ft	11	214.0 ft	53
City of La Conner	WA0022446	0.52	21.0 ft	2.9	208.5 ft	12.3

Fir Island separates the North and South Fork Skagit River and serves to isolate the south fork Skagit River delta from the north fork delta. The currents in Skagit Bay carry water from south to north and out through Deception Pass. Therefore, the regulated waters in the south end of the Reservation do not influence areas of Skagit Bay south of the north fork Skagit River delta.

Based on the information presented above and to provide an additional level of protection, the ESA action area used in this BE was extended 400 ft seaward from the boundary of the Swinomish Tribe's Regulated Surface Waters (i.e. the "buffer zone"), doubling the 200 ft distance that the evaluation of the point source mixing zones indicates would result in more than a 10-fold dilution of pollutants in Tribal waters, and includes the entire Swinomish channel from bank to bank (Figure 2). The surface water area of the action area that is outside the exterior boundary of the Tribe's Regulated Surface Waters is 515 acres (= 208 hectares). Given the significant dilution of pollutants regulated by the WQS, any effects outside of the ESA action area are considered negligible.

2.2 Overview of Water Quality Standards

A water quality standard defines the water quality goals for a waterbody by designating the use or uses of the water, setting criteria necessary to protect the uses, and by preventing or limiting degradation of water quality through antidegradation provisions. The CWA provides the statutory basis for the water quality standards program and defines broad water quality goals. For example, section 101(a) requires, in part, that wherever attainable, waters 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 ("fishable/swimmable").

EPA's CWA action on state or tribal WQS does not authorize the existence of pollutants in any waterbody. Approved WQS are implemented through various CWA programs including NPDES permits (CWA section 402), state or tribal certification of federal licenses and permits (CWA section 401), wetland permits (CWA section 404), and water quality assessment and TMDLs (CWA section 303(d)). States and authorized tribes may adopt additional implementing programs, such as nonpoint source management programs to control nonpoint source pollution.

In addition to adopting WQS, states and authorized tribes are required to review and revise standards, as necessary, every 3 years. This public process, commonly referred to as the triennial review, allows for new technical and scientific data to be considered in the standards. The regulatory requirements governing water quality standards are established at 40 CFR Part 131.

CWA section 518(e) authorizes EPA to treat an Indian tribe in a similar manner as a state (TAS) to manage and protect water resources "within the borders of an Indian reservation," provided certain

requirements are satisfied. The Swinomish Tribe received TAS approval from EPA to administer the water quality standards and water quality certification programs, CWA sections 303(c) and 401, on April 18, 2008.

The minimum requirements that must be included in state or tribal WQS are designated uses, criteria to protect the uses, and an antidegradation policy to protect existing uses and high-quality waters. In addition to these elements, the regulations allow for states and tribes to adopt discretionary policies as part of their water quality standards, such as allowances for mixing zones for NPDES discharges and water quality standard variances. These policies are also subject to EPA review and action.

Section 303(c)(2)(B) of the CWA requires states and tribes to adopt numeric criteria for all toxic pollutants for which criteria have been published under section 304(a). To provide scientific guidance to states and authorized tribes, EPA publishes, and from time to time revises, criteria for water quality under section 304(a) that reflect the latest scientific knowledge. States and tribes consider these criteria documents, along with the most recent scientific information, when adopting regulatory criteria.

The CWA requires states and authorized tribes to adopt new or revised WQS and submit them to EPA for review and approval or disapproval action. If EPA determines that the new or revised WQS are consistent with the CWA and 40 CFR Part 131, EPA approves the WQS. However, EPA disapproves the WQS if they are not consistent with the CWA. EPA may approve some provisions and not others within the same WQS submittal. In case of a disapproval, the EPA is required to identify the changes needed to meet the requirements of the CWA. The state or tribe is then given an opportunity to make appropriate changes. If the state or tribe does not adopt changes sufficient to meet the CWA and regulation requirements, EPA must promulgate federal regulations, as necessary, to replace those disapproved portions.

2.2.1 Water quality certification (Clean Water Act section 401)

Under section 401 of the CWA, a federal agency cannot issue a permit or license for an activity that may result in a discharge to waters of the U.S. until the state or tribe with jurisdiction over the location of the discharge has granted or waived section 401 water quality certification. CWA section 401 water quality certification provides states and authorized tribes with an effective tool to help protect water quality under their jurisdiction, by providing an opportunity to address the aquatic resource impacts of federally-issued permits and licenses. Any conditions required by the state or tribe must be incorporated into the final license or permit, thus ensuring federal agencies comply with state and/or tribal WQS.

Examples of federal licenses and permits subject to section 401 certification include NPDES permits in locations where EPA administers the permitting program, CWA section 404 permits for discharge of dredged or fill material issued by the Army Corps of Engineers, Federal Energy Regulatory Commission (FERC) hydropower licenses, and Rivers and Harbors Act sections 9 and 10 permits for activities that have a potential discharge in navigable waters issued by the Corps of Engineers. Many states and tribes rely on CWA section 401 certification to ensure that discharges of dredge or fill material into a water of the U.S. do not cause unacceptable environmental impacts. More generally, certification is regarded as the primary regulatory tool for protecting wetlands and other aquatic resources.

2.3 Overview of the Derivation of EPA Aquatic Life Criteria

All of the Tribe's adopted criteria are the same as EPA's current 304(a) recommendations. These recommendations were derived as described below.

2.3.1 Criteria developed after 1985

National ambient water quality criteria (AWQC) are established by EPA under the CWA. EPA reviews and from time to time revises 304(a) AWQC as necessary to ensure the criteria are consistent with the latest scientific information. Section 304(a) aquatic life criteria serve as recommendations to states and tribes in defining ambient water concentrations that will protect against adverse ecological effects to aquatic life resulting from exposure to a pollutant found in water from direct contact or ingestion of contaminated water and/or food. Aquatic life criteria address the CWA goals of providing for the protection and propagation of fish and shellfish. When adopted into state or tribal standards, these criteria can become a basis for establishing permit limits and TMDLs, and they can be used for other implementing programs such as assessment of the condition of the waters.

For development of 304(a) criteria, EPA follows its 1985 *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* (the Guidelines)(C.E. Stephan et al., 1985). EPA assembles all available toxicity test data and considers which data are relevant that also meet data quality acceptance standards for all genera. Where data allow, acute and chronic criterion values are developed. The criteria are based on a sensitivity distribution (SD) comprised of ranked genus mean acute values (GMAVs), calculated from combined species mean acute values (SMAVs within each genus) for acceptable data. SMAVs are calculated using the geometric mean for all acceptable measures of effect based on the results of toxicity tests within a given species (e.g., all EC50s from acceptable acute tests for Daphnia magna). GMAVs are then calculated using the geometric means of all SMAVs within a given genus (e.g., all SMAVs for genus Daphnia, such as Daphnia pulex, Daphnia magna). If only one SMAV is available for a genus, then the GMAV is represented by that value. GMAVs are then rank-ordered by sensitivity from most sensitive to least sensitive. The final acute value (FAV) is determined by regression analysis using a log-triangular fit based on the four most sensitive genera (reflected as GMAVs) in the data set to interpolate or extrapolate (as appropriate) to the 5th percentile of the distribution represented by the tested genera. If there are 59 or more GMAVs, as is the case with ammonia for example, the four GMAVs closest to the 5th percentile of the distribution are used to calculate the FAV. The acute criterion magnitude is the FAV divided by two, in order to provide an acute criterion magnitude protective of nearly all individuals in 95% of all genera, since the effect endpoint is a 50th percentile effect (e.g., LC50 or EC50) (see 1985 Guidelines, Section XI. Criterion, B.)(C.E. Stephan et al., 1985).

Although the aquatic life criteria derivation process relies on selected toxicity endpoints from the sensitive species tested, it does not necessarily mean that the selected toxicity endpoints reflect the sensitivity of the most sensitive species existing in a given environment. The intent of the eight minimum data requirements is to serve as a sample representative of the aquatic community. These minimum data requirements represent different ecological, trophic, taxonomic, and functional differences observed in the natural aquatic ecosystem. The use of the four most sensitive genera to determine the final criterion value is a censored statistical approach than improves estimation of the lower tail (most sensitive) of the distribution when the shape of the overall distribution, particularly in the less sensitive part of the distribution, is uncertain.

The chronic criterion may be determined by one of two methods. If all eight minimum data requirements are met with acceptable chronic test data, then the chronic criterion is derived using the same method used for the acute criterion. Genus Mean Chronic Values (GMCVs) are derived from available Species Mean Chronic Values (SMCVs) and are then rank-ordered from least to most sensitive, and the Final Chronic Value (FCV) is calculated based on regression analysis of a censored distribution using the four most sensitive GMCVs, similar to calculation of the FAV. Unlike the FAV, however, the FCV directly serves as the basis for the chronic criterion without further adjustment because the endpoint measured represents a low level [e.g., EC20 or no observed effect concentration (NOEC)] of effect (see 1985 Guidelines)(C.E. Stephan et al., 1985).

The overall process for deriving aquatic life water quality criteria using the 1985 Guidelines is shown in Figure 3.

Derivation of Numerical National Water Quality Crtieria for the Protection of Aquatic Organisms and Their Uses

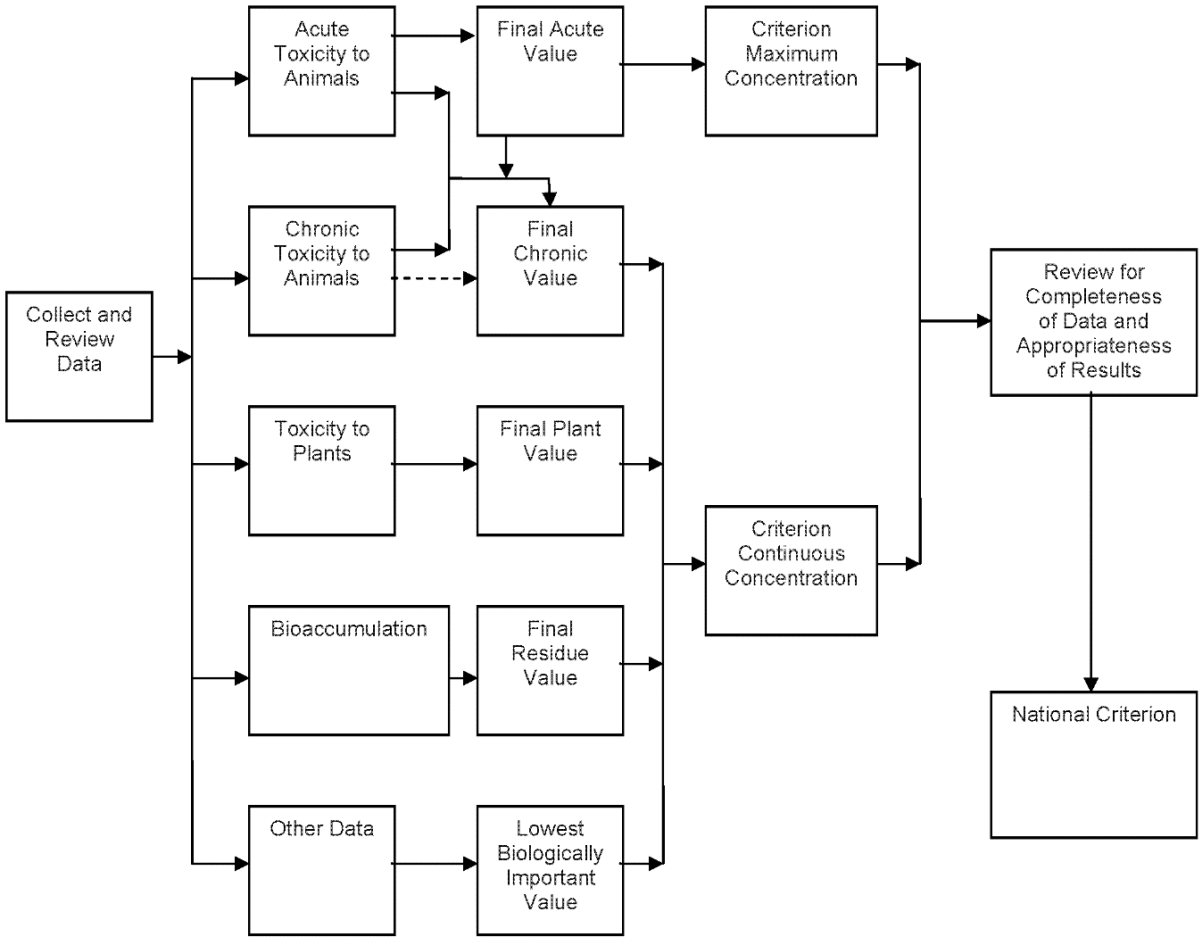


Figure 3: Derivation of aquatic life criteria

A more detailed description of procedures that EPA uses to derive 304(a) recommendations for aquatic life criteria is provided in the 1985 Guidelines (C.E. Stephan et al., 1985).

2.3.2 Criteria developed before 1985

Aquatic life criteria for aldrin, alpha-Endosulfan, beta-Endosulfan, chlordane, dieldrin (freshwater CCC), gamma-BHC/Lindane (saltwater CCC), heptachlor, heptachlor epoxide, silver, and 4,4'-DDT were developed based on guidelines published in 1980, at 45 FR 79341. The 1980 Guidelines had different minimum data requirements and derivation procedures.

Changes in the 1985 Guidelines relative to the 1980 Guidelines are explained in the 1985 Guidelines at Pages 8-9. One of the more significant differences is that a CMC based on the 1980 Guidelines is equal to the FAV, whereas a CMC based on the 1985 Guidelines is equal to the FAV divided by 2.

While most CCCs developed using the 1985 Guidelines are expressed as 4-day average concentrations, CCCs developed using the 1980 Guidelines are expressed as 24-hour average concentrations. CMCs developed using the 1985 Guidelines are expressed as 1-hour average concentrations, whereas CMCs developed using the 1980 Guidelines are expressed as instantaneous maximum concentrations that are not to be exceeded.

2.4 The Swinomish Tribe Water Quality Standards that EPA Proposes to Approve

The Tribe submitted its WQS to EPA for approval or disapproval action under the CWA on April 30, 2019.[3] These WQS are the first that the Tribe has adopted since acquiring TAS status, and as such they constitute new protections for Swinomish waters.

The Tribe has designated uses to protect aquatic life for all of its fresh and marine waters. The Tribe's aquatic life toxics criteria and the temperature, dissolved oxygen, pH, and narrative criteria described below are intended to protect aquatic life in fresh and marine waters as specified in Tribal Code sections 19-06.120(A), (B), and (C)(1), and 19-06.130(A), (B), and (C)(1), respectively. ESA-listed species that are found in the Tribe's waters are included in the aquatic life uses designated by the Tribe. EPA approved the Tribe's aquatic life use designations on May 20, 2019.

2.4.1 Numeric toxics criteria

The Swinomish Tribe's water quality criteria for toxics appear in section 19-06.140 of the Tribal Code, and the numeric aquatic life criteria are listed in Table 15 of the WQS and shown in Table 6, below. Footnotes are included as they apply to the specifics of this biological evaluation. Complete water quality standards are included in Appendix A.

Table 6. Numeric Aquatic Life Criteria for Toxics. All values are in µg/L unless otherwise noted.

CAS	Compound Name (or Class)	Fresh Water				Marine Water			
		Acute[a] Criteria		Chronic[b] Criteria		Acute[a] Criteria		Chronic[b] Criteria	
7664417	Ammonia, total (mg/L)	equation-based	f	equation-based	g	equation-based	k	equation-based	k
	Ammonia, unionized (mg/L)					equation-based	k	equation-based	k
16887006	Chloride	860,000		230,000					
7782505	Chlorine	19		11		13		7.5	
57-12-5	Cyanide	22		5.2		1		1	
7723140	Phosphorus Elemental							0.1	
7783064	Sulfide - Hydrogen Sulfide			2				2	
7440382	Arsenic	340	h,l	150	h,l	69	h,l	36	h,l
7440439	Cadmium	hardness-based	j,l	hardness-based	j,l	33	l	7.9	l
18540299	chromium (vi)	16	l	11	l	1100	l	50	l

[3] On May 20, 2019, EPA approved the Tribe's WQS that are not related to protection of aquatic life and wildlife. The approval documents can be found on EPA's website at https://www.epa.gov/wqs-tech/water-quality-standards-regulations-swinomish-indian-tribal-community.

CAS	Compound Name (or Class)	Fresh Water				Marine Water			
		Acute[a] Criteria		Chronic[b] Criteria		Acute[a] Criteria		Chronic[b] Criteria	
16065831	chromium (iii)	hardness-based	j,l	hardness-based	j,l				
7440508	Copper	BLM-based	o	BLM-based	o	4.8	l	3.1	l
7439896	Iron			1,000[p]					
7439921	Lead	hardness-based	j,l	hardness-based	j,l	210	l	8.1	l
7439976	Mercury [m]	2.4	p	0.012	p	2.1	p	0.025	p
7440020	Nickel	hardness-based	j,l	hardness-based	j,l	74	l	8.2	l
7782492	Selenium		p,u		p	290	p,u	71	p
7440224	Silver	hardness-based	j,l			1.9	l		
	Tributyltin (TBT)	0.46		0.072		0.42		0.0074	
7440666	Zinc	hardness-based	j,l	hardness-based	j,l	90	l	81	l
107-02-8	Acrolein	3		3					
63252	Carbaryl	2.1		2.1		1.6			
84852153	Nonylphenol	28		6.6		7		1.7	
87-86-5	Pentachlorophenol	19	n	15	n	13		7.9	
309-00-2	Aldrin	3	e			1.3	e		
58-89-9	gamma- Hexachlorocyclohexane (HCH) (Lindane)	0.95				0.16	e		
57-74-9	Chlordane	2.4	e	0.0043	e	0.09	e	0.004	e
2921882	Chlorpyrifos	0.083		0.041		0.01		0.0056	
333415	Diazinon	0.17		0.17		0.82		0.82	
50-29-3	p,p'- Dichlorodiphenyl-trichloroethane (4,4'-DDT)	1.1	e,q	0.001	e,q	0.13	e,q	0.001	e,q
8065483	Demeton			0.1				0.1	
60-57-1	Dieldrin	0.24	e	0.056	e	0.71	e	0.0019	e
	Endosulfan-Alpha+Beta	0.22	e	0.056	e	0.03	e	0.0087	e
72-20-8	Endrin	0.086		0.036		0.04	e	0.0023	e
86500	Guthion			0.01				0.01	
76-44-8	Heptachlor	0.52	e	0.0038	e	0.05	e	0.0036	e
1024-57-3	Heptachlor Epoxide	0.52	e	0.0038	e	0.05	e	0.0036	e
121755	Malathion			0.1				0.1	
72-43-5	Methoxychlor			0.03				0.03	
2385855	Mirex			0.001				0.001	
56382	Parathion	0.065		0.013					
	Polychlorinated biphenyls			0.014	r			0.03	r

CAS	Compound Name (or Class)	Fresh Water				Marine Water			
		Acute[a] Criteria		Chronic[b] Criteria		Acute[a] Criteria		Chronic[b] Criteria	
8001-35-2	Toxaphene	0.73		0.0002		0.21		0.0002	

Footnotes:

a. Acute criteria: The Criteria Maximum Concentration (CMC) is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed briefly without resulting in an unacceptable effect. With the exception of those pollutants which cite to footnote e, the CMC concentration is expressed as a one hour average not to be exceeded more than once in a three year period.

b. Chronic criteria: The Criterion Continuous Concentration (CCC) is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect. With the exception of those pollutants which cite to footnotes e and g, the CCC concentration is expressed as a four day average not to be exceeded more than once in a three year period.

e. The acute values shown are expressed as maximum values not to be exceeded. The chronic values are expressed as a 24-hour average not to be exceeded. These criteria are based on the 1980 criteria which used different Minimum Data Requirements and derivation procedures from the 1985 Guidelines. If evaluation is to be done using an averaging period, the acute criteria values given should be divided by 2 to obtain a value that is more comparable to a CMC derived using the 1985 Guidelines.

f. Ammonia Aquatic Life Criteria for Fresh Waters, Acute Criterion: The one-hour average concentration of total ammonia nitrogen (in mg N/L) does not exceed, more than once every three years on the average, the CMC (acute criterion) calculated using the following equations.

Where salmonid fish are present:

$$CMC = MIN\left(\left(\frac{0.275}{1+10^{7.204-pH}}+\frac{39.0}{1+10^{pH-7.204}}\right),\left(0.7249\times\left(\frac{0.0114}{1+10^{7.204-pH}}+\frac{1.6181}{1+10^{pH-7.204}}\right)\times\left(23.12\times 10^{0.036\times(20-T)}\right)\right)\right)$$

Or where salmonid fish are not present:

$$CMC = 0.7249 \times \left(\frac{0.0114}{1+10^{7.204-pH}}+\frac{1.6181}{1+10^{pH-7.204}}\right)\times MIN\left(51.93, 23.12\times 10^{0.036\times(20-T)}\right)$$

Tables 5a (salmonids present) and 5b (salmonids absent) of the EPA ammonia criteria document, *Aquatic Life Ambient Water Quality Criteria for Ammonia – Freshwater, 2013* (USEPA, 2013a) may be used to determine CMC values. When using these tables, measured temperature and pH values that fall between available table values should be rounded up to yield protective criteria.

g. Ammonia Aquatic Life Criteria for Fresh Waters, Chronic Criterion: The 30-day average concentration of total ammonia nitrogen (in mg N/L) does not exceed, more than once every three years on the average, the CCC (chronic criterion) calculated using the following equation.

$$CCC = 0.8876 \times \left(\frac{0.0278}{1+10^{7.688-pH}}+\frac{1.1994}{1+10^{pH-7.688}}\right)\times\left(2.126\times 10^{0.028\times(20-MAX(T,7))}\right)$$

In addition, the highest four-day average within the 30-day period should not exceed 2.5 times the CCC more than once in three years on average.

Table 6 of the EPA ammonia criteria document, *Aquatic Life Ambient Water Quality Criteria for Ammonia – Freshwater, 2013* (USEPA, 2013a) may be used to determine CCC values. When using this table, measured temperature and pH values that fall between available table values should be rounded up to yield protective criteria.

CAS	Compound Name (or Class)	Fresh Water		Marine Water	
		Acute[a] Criteria	Chronic[b] Criteria	Acute[a] Criteria	Chronic[b] Criteria

h. The aquatic life criteria were derived from data for arsenic (III), but are applied as total arsenic. The human health criteria refer to the inorganic form only.

j. Fresh water aquatic life criteria for these metals are expressed as a function of total hardness (expressed as mg/L CaCO3) according to the following equations. The factors for the equations are provided in the following matrix.

Acute criterion (dissolved) = exp[mA[ln(hardness)]+bA] x conversion factor

Chronic criterion (dissolved) = exp[mC[ln(hardness)]+bC] x conversion factor

Note to the table: The term "exp" represents the base e exponential function.

* The conversion factors (CF) for cadmium and lead are hardness dependent. Conversion factors can be calculated for any hardness using the following equations:

Cadmium Acute CF = 1.136672–[(ln hardness) (0.041838)]

Cadmium Chronic CF = 1.101672–[(ln hardness) (0.041838)]

Lead (Acute and Chronic) CF = 1.46203–[(ln hardness) (0.145712)]

Factors for Calculating Metals Criteria

Metal	m_A	b_A	m_C	b_C	Conversion Factors	
					Acute	Chronic
Cadmium	0.8403	-3.572	0.6247	-3.384	*	*
Chromium (III)	0.8190	3.7256	0.8190	0.6848	0.316	0.860
Lead	1.273	-1.460	1.273	-4.705	*	*
Nickel	0.8460	2.255	0.8460	0.0584	0.998	0.997
Silver	1.72	-6.59	N/A	N/A	0.85	N/A
Zinc	0.8473	0.884	0.8473	0.884	0.978	0.986

Note to the table: The term "exp" represents the base e exponential function.

* The conversion factors (CF) for cadmium and lead are hardness dependent. Conversion factors can be calculated for any hardness using the following equations:

Cadmium Acute CF = 1.136672–[(ln hardness) (0.041838)]

Cadmium Chronic CF = 1.101672–[(ln hardness) (0.041838)]

Lead (Acute and Chronic) CF = 1.46203–[(ln hardness) (0.145712)]

k. To develop total ammonia criteria use Ambient Water Quality Criteria for Ammonia (Saltwater) – 1989 (EPA 440/5-88-04)(USEPA, 1989). Ammonia criteria for saltwater are for unionized ammonia. Unionized ammonia is the most toxic form of ammonia to aquatic life. In saltwater, the fraction of the total ammonia that is unionized depends mainly on the pH, temperature and salinity of the water. For implementation purposes, the unionized ammonia criteria are generally converted to total ammonia to be consistent with standard water quality monitoring methods.

l. These freshwater and marine criteria for metals are expressed in terms of the dissolved metal in the water column.

m. If the ambient concentration of total mercury exceeds 0.012 µg/L more than once in a 3-year period in the ambient water, the edible portion of aquatic species of concern must be analyzed to determine whether the concentration of methyl mercury exceeds the FDA action level (1.0 mg/kg). If the FDA action level is exceeded, the Tribe must notify the EPA Region 10 Regional Administrator, initiate a site-specific criterion or a revision of its mercury criterion so as to protect designated uses, and take other appropriate action, such as issuance of a fish consumption advisory for the affected area.

n. Fresh water aquatic life criteria for pentachlorophenol are expressed as a function of pH, and are calculated as follows (values in the table correspond to a pH of 7.8):

CAS	Compound Name (or Class)	Fresh Water		Marine Water	
		Acute[a] Criteria	Chronic[b] Criteria	Acute[a] Criteria	Chronic[b] Criteria

Acute criterion = exp(1.005(pH)–4.869)

Chronic criterion = exp(1.005(pH)–5.134)

o. The available toxicity data, when evaluated using the procedures described in the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" (C.E. Stephan et al., 1985) indicate that freshwater aquatic life should be protected if the 1-hour average and four-day average concentrations do not respectively exceed the acute and chronic criteria concentrations calculated by the Biotic Ligand Model.

A return interval of 3 years between exceedances of the criterion continues to be EPA's general recommendation. However, the resilience of ecosystems and their ability to recover differ greatly. Therefore, scientific derivation of alternative frequencies for exceeding criteria may be appropriate. See Aquatic Life Ambient Freshwater Quality Criteria – Copper, 2007 Revision (USEPA, 2007a) .

Freshwater copper criteria shall be developed using EPA's current Biotic Ligand Model [BLM current criteria document: EPA-822-R-07-001 (2007a)]. When criteria are developed such criteria must be protective of aquatic life for all expected water quality conditions. A minimum number of 24X samples over two years, reflecting intra-annual or seasonal flow and spatial variability related water quality variability must be collected. If inter-annual or spatial variability in water quality occurs regularly, the monitoring plan shall reflect these characteristics. In the absence of sufficient ambient data for any of the BLM input parameters, default values corresponding to the 10th percentile of the applicable ecoregional dataset for the relevant stream order for each missing parameter shall be used. Default values shall be found in EPA's Missing Parameters Technical Support Document [EPA 820-R-15-106, (2016c)], hereby incorporated by reference. All BLM criteria shall be made available on the Tribe's website.

p. This criterion is expressed in terms of the total recoverable metal in the water column.

q. This criterion applies to DDT and its metabolites (i.e., the total concentration of DDT and its metabolites should not exceed this value).

r. This criterion applies to total PCBs (e.g., the sum of all congeners, isomers, homologs or Aroclors).

u. Selenium Aquatic Life Criteria for Fresh Waters:

Criterion Element	**Magnitude**	**Duration**	**Frequency**
Fish Tissue[a] (Egg-Ovary)[b]	15.1 mg/kg dw	Instantaneous measurement[c]	Not to be exceeded
Fish Tissue[a] (Whole Body or Muscle)[d]	8.5 mg/kg dw or 11.3 mg/kg dw muscle (skinless, boneless filet)	Instantaneous measurement[c]	Not to be exceeded
Water Column[e] (Monthly Average Exposure)	1.5 µg/L in lentic aquatic systems 3.1 µg/L in lotic aquatic systems	30 days	Not more than once in three years on average
Water Column[e] (Intermittent Exposure)[f]	$WQC_{int} = \frac{WQC_{30\text{-}day} - C_{bkgrnd}(1 - f_{int})}{f_{int}}$	Number of days/month with an elevated concentration	Not more than once in three years on average

[a] Fish tissue elements are expressed as steady-state.
[b] Egg/ovary supersedes any whole-body, muscle, or water column element when fish egg/ovary concentrations are measured.
[c] Fish tissue data provide point measurements that reflect integrative accumulation of selenium over time and space in fish population(s) at a given site.
[d] Fish whole-body or muscle tissue supersedes water column element when both fish tissue and water concentrations are measured.
[e] Water column values are based on dissolved total selenium in water and are derived from fish tissue values via bioaccumulation modeling. Water column values are the applicable criterion element in the absence of steady-state condition fish tissue data.

CAS	Compound Name (or Class)	Fresh Water		Marine Water	
		Acute[a] Criteria	Chronic[b] Criteria	Acute[a] Criteria	Chronic[b] Criteria
	[f] Where $WQC_{30\text{-}day}$ is the water column monthly element, for either a lentic or lotic waters; C_{bkgrnd} is the average background selenium concentration, and f_{int} is the fraction of any 30-day period during which elevated selenium concentrations occur, with f_{int} assigned a value ≥0.033 (corresponding to 1 day).				

The following criteria are not addressed in this BE and are not part of this ESA consultation (section 1.7):

- The chronic arsenic criterion for fresh waters
- The chloride criteria for fresh waters
- The cyanide criteria for fresh waters
- The mercury criteria for fresh and marine waters.

The entries for these criteria are identified with gray highlight in the above table.

2.4.2 Numeric temperature criteria

2.4.2.1 Freshwater

The aquatic life criterion for salmon and trout in fresh water is a 7-day average of daily maximum temperatures (7-DADMax) of 16 °C (19-06.120(C)(2)(b), Table 4).

2.4.2.2 Marine water

The aquatic life criterion for fish use in marine water is a 1-day maximum temperature (1-DADMax) of 16 °C (19-06.130(C)(2)(b), Table 10).

2.4.3 Numeric dissolved oxygen criteria

2.4.3.1 Freshwater

The aquatic life criterion for dissolved oxygen for salmon and trout use in fresh water is a 1-day minimum of 9.5 mg/L (19-06.120(C)(2)(c), Table 5).

2.4.3.2 Marine water

The aquatic life criterion for dissolved oxygen for fish use in marine water is a 1-day minimum of 6.0 mg/L (19-06.130(C)(2)(c), Table 11).

2.4.4 Numeric pH criteria

2.4.4.1 Fresh water

The aquatic life criterion for pH for salmon and trout fish use in fresh water is, “may not be less than 6.5 or greater than 8.5, with a human-caused variation within the above range of less than 0.2 units” (19-06.120(C)(2)(d), Table 6).

2.4.4.2 Marine water

The aquatic life criterion for pH for fish use in marine water is, “may not be less than 7.0 or greater than 8.5, with a human-caused variation within above range of 0.2 units” (19-06.130(C)(2)(d), Table 12).

2.4.5 Narrative criteria

Federal regulations concerning water quality standards state that "States [and authorized tribes] should…establish narrative criteria or criteria based upon biomonitoring methods where numerical criteria cannot be established or to supplement numerical criteria" (40 CFR 131.11(b)(2)).

The Tribe's narrative water quality criteria appear in sections 19-06.100, 19-06.140 and 19-06.150 of the Tribal code. The following narrative criteria protect the aquatic life and aquatic-dependent wildlife uses:

19-06.100(A): "All Regulated Surface Waters shall be free from pollutants in concentrations or combinations that would preclude attainment of the most sensitive use of the water body, except as provided for within this Chapter."

19-06.140(B): "Toxic pollutants shall not be introduced into Regulated Surface Waters in concentrations which have the potential either singularly or cumulatively to adversely affect existing and designated water uses, cause acute or chronic toxicity to the most sensitive biota dependent upon those waters, or adversely affect public health, as determined by the Director based on best available science, except as provided for in this Chapter."

19.06.150(A)(1): "Floating Solids, Oil and Grease: Regulated Surface Waters, stream banks, and shorelines shall be free from visible oils, scum, foam, grease, and other floating materials and suspended substances of a persistent nature resulting from other than natural causes."

19.05.150(A)(2): "Color: True color-producing materials resulting from anthropogenic causes shall not create an aesthetically undesirable condition, nor should color inhibit photosynthesis."

19.05.150(A)(4): "Nuisance Conditions: Pollutants resulting from anthropogenic causes shall not be present in Regulated Surface Waters in concentrations which will produce objectionable algal densities or nuisance species which may impair ecological integrity."

19-06.150(A)(5)(a): "Turbidity resulting from anthropogenic causes in Regulated Surface Waters shall not be at a level that may have a deleterious effect on an aquatic life community present in those waters."

19-06.150(A)(6): "Bottom Deposits: Regulated Surface Waters shall be free from pollutants resulting from anthropogenic causes…that may settle and have a deleterious effect on an aquatic life community present in those waters"

19-06.150(B)(1): "Regulated Surface Waters shall be of sufficient quality to preserve, protect, and enhance all life stages of resident and/or anadromous finfish and any aquatic biotas or aquatic life communities present in those waters."

19-06.150(B)(3): "Regulated Surface Waters shall be free from substances, whether attributable to point source discharges, nonpoint sources, or instream activities, in concentrations or combinations which may have a deleterious effect on an aquatic life community present in those waters and the ecological integrity of those waters."

19-06.150(C): "Water quality in wetlands within the exterior boundaries of the Reservation, including created wetlands which are not constructed wetlands, shall be protected by maintaining the hydrological conditions, hydrophilic vegetation, and substrate characteristics to support existing and

designated uses. Narrative and numeric water quality criteria described in Section 19-06.100 and Sections 19-06.140 through 19-06.150 of this Chapter, shall apply to wetlands."

19-06.150(C)(3): "All wetlands, as defined by the HGM classification scheme, shall maintain biological, physical, and hydrological conditions - as determined by reference wetlands including, but not limited to: floristic quality; pH of wetland waters; water levels or elevations; and water temperature variations."

19-06.150(D): "Regulated Surface Waters shall be of sufficient quality to preserve, protect, and enhance all life stages of resident and/or migratory wildlife species which live in, on, or near those waters."

19-06.150(E): "Instream flow necessary to maintain the physical, chemical, and biological integrity of Regulated Surface Waters shall not be modified if such modification will or may impair water quality or existing or designated uses."

3 Status of the Species in the Action Area

Lists of endangered and threatened species that may be affected by the proposed action were obtained from the NMFS and USFWS endangered species websites and additional information was gathered from species experts, including a review of available literature. Under the ESA of 1973, species of plants and animals are listed as either threatened or endangered. Some species have been divided into Evolutionarily Significant Units (ESU), which describe distinctive groups or populations of animals; any subspecies of fish or wildlife or plants; and any distinct population segment (DPS) of any species of vertebrate fish or wildlife which interbreeds when mature. For the purposes of this BE, each ESU/DPS was considered a separate species under the ESA. Of the 21 aquatic or aquatic dependent species that occur within or adjacent to the action area, 11 species were identified as being present in the defined action area (Table 7), and are discussed in detail in sections below. The critical habitat for those species that it is designated is noted in Table 7 below, as well as its overlap with the buffer area outside of the Tribal waters. Critical habitat is further discussed in the species subsections below.

The species profiles in this section are modified from the Biological Assessment for the Northwest Area Contingency Plan for the Response to Spills of Oil and Hazardous Substances (Windward, 2018 (USEPA, 2018c)). Information specific to the Swinomish action area was included here where available.

Table 7. Federally Protected Species

Protected Species	Scientific Name	ESA Status	Species Found in Action Area	Critical Habitat	
				Designation	Overlap with Action Area Buffer Zone
Responsible Agency - NMFS					
Fish					
Puget Sound ESU Spring Chinook salmon	*Oncorhynchus tshawytscha*	T 70 FR 37160	X	D 70 FR 52630	X
Puget Sound ESU Summer Chinook salmon	*O. tshawytscha*	T 70 FR 37160	X	D 70 FR 52630	X
Puget Sound ESU Fall Chinook salmon	*O. tshawytscha*	T 70 FR 37160	X	D 70 FR 52630	X
Hood Canal ESU chum salmon	*O. keta*	T 70 FR 37160		D 70 FR 52630	
Puget Sound DPS steelhead trout	*O. mykiss*	T 72 FR 26722	X	D 81 FR 9252	Freshwater Only Outside of Action Area
Puget Sound/Georgia Basin Bocaccio rockfish	*Sebastes paucispinis*	E 75 FR 22276	X	D 79 FR 68042	X

Protected Species	Scientific Name	ESA Status	Species Found in Action Area	Critical Habitat	
				Designation	Overlap with Action Area Buffer Zone
Puget Sound/Georgia Basin yelloweye rockfish	*S. ruberrimus*	T 75 FR 22276	X	D 79 FR 68042	X
Sea Turtles[4]					
Leatherback sea turtle	*Dermochelys coriacea*	E 35 FR 8491		D 77 FR 4169	
Green sea turtle	*Chelonia mydas*	T 81 FR 20057		D 63 FR 46693	
Loggerhead sea turtle, North Pacific Ocean DPS	*Caretta Caretta*	E 76 FR 58868		D 76 FR 39855	
Marine Mammals[5]					
Blue whale	*Balaenoptera musculus*	E 35 FR 18319			
Fin whale	*B. physalus*	E 35 FR 18319			
Humpback whale, Central America DPS	*Megaptera novaeangiliae*	E 35 FR 18319			
Humpback whale, Mexico DPS	*M. novaeangiliae*	T 35 FR 18319			
North Pacific right whale	*Eubalena japonica*	E 73 FR 12024		D 73 FR	

[4] Green, Loggerhead and Leatherback turtles are proposed to be removed from the list of assessed species due to their lack of presence in the action area.
Green and Loggerhead report, WDFW: https://wdfw.wa.gov/publications/01870/draft_wdfw01870.pdf, had no reports of Green or Loggerhead sea turtles in the Swinomish action area.
Leatherback sea turtle report, WDFW: https://wdfw.wa.gov/publications/01871/wdfw01871.pdf, had no reports of Leatherback sea turtles in the Swinomish action area.

[5] The action area is included in the area identified as biologically important for cetaceans and gray whales. https://response.restoration.noaa.gov/maps-and-spatial-data/environmental-response-management-application-erma/pacific-northwest-erma.html However, the action area is predominantly a depth of less than twenty feet, see Figure 1, and therefore unlikely to be visited by whales.

Communication with John Calambokidis, Research Biologist, Cascadia Research, 2/9/2019 indicates, of the whale species, Humpback would be the most likely to be in the vicinity of the Swinomish action area.

Review of the 2016-Feburary 2019 sighting of whales reported on the Orca Network, http://www.orcanetwork.org/Main/index.php?categories_file=Sightings, found gray and humpback whales in Skagit Bay South of the action area but no sightings within the action area and one sighting of orcas within the action area in Swinomish Channel.

Protected Species	Scientific Name	ESA Status	Species Found in Action Area	Critical Habitat	
				Designation	Overlap with Action Area Buffer Zone
Sei whale	*B. borealis*	E 35 FR 12222			
Killer whale, Southern Resident DPS	*Orcinus orca*	E 70 FR 69903	X	D 71 FR 690	X
Sperm whale	*Physeter macrocephalus*	E 35 FR 18319			
Gray whale, Western North Pacific	*Eschrichtius robustus*	E(F) 35 FR 18319			
Responsible Agency- USFW					
Fish					
Bull Trout	*Salvelinus confluentus*	T 64 FR 58910	X	D 75 FR 63897	X
Marine Bird					
Marbled murrelet	*Brachyrampus marmoratus*	T 57 FR 45328	X	D 76 FR 61599	
Amphibian					
Oregon Spotted Frog	*Rana prestiosa*	T 79 FR 51658	Not found per communication with USFWS, 5/14/19	78 FR 53538	
NOTES: T = Threatened, E = Endangered, D = Designated; P = Proposed; DPS = distinct population segment					

3.1 Potential Effects from Global Climate Change

Climate change has an influence on baseline conditions related to both habitat and species, resulting in effects on ESA-listed and proposed species and their designated and proposed critical habitat that cannot be ignored but are difficult to assess.

The increasing rate of global climate change is supported by a preponderance of scientific evidence (Ecology, 2012). Observed and predicted impacts to Pacific Northwest 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), which may substantially affect the ESA-listed species and their habitats considered in this BE.

The changing climate is expected to affect species and habitats in many ways. For example, the distribution of shoreline-dependent species will shift because of rising sea levels. The California Current Large Marine Ecosystem will see northern shifting isotherms, which will shift species distribution as well as increase ocean stratification, impeding nutrient transport and plankton production (NMFS, 2016e).

Migration patterns of salmon are also expected to be affected by changing water temperatures (Ecology, 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 Lower Columbia River (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) (NMFS, 2015c). Climate-driven impairment of invertebrate and fish populations may cause a cascading effect on species at higher tropic levels; for example, loss of prey due to climate change is a key threat to fin whales (*Balaenoptera physalus*) (NMFS, 2010). Ocean acidification is predicted to increase mortality in shell-forming species (e.g., shellfish) (NMFS, 2016e), which are important as prey for ESA-listed species (e.g., loggerhead sea turtle, *Caretta caretta*).

Effects will be seen outside of the ocean as well. Changes in the lengths of typical seasons have been observed to shift the timing of bird and other animals' migration (Ecology, 2012). The spread of invasive species and disease is predicted to increase. For example, forests in Washington have become more susceptible to pine beetle (*Dendroctonus ponderosae*) infestation as a result of drought-stress (Ecology, 2012). Climate change-driven disruptions to ecosystems are predicted to result in further declines of species populations and biodiversity.

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. According to the (USGCRP, 2018) projections for elevated instream temperature predict a 22% reduction in salmonid habitat in Washington State by late century under a high emissions future scenario. Average regional temperatures are likely to increase over the next century (USGCRP, 2009, 2018);(Mantua, Tohver, & Hamlet, 2010; Mote & Salathé, 2010; Zabel, Scheuerell, McClure, & Williams, 2006).

3.2 Chinook Salmon (*Oncorhynchus tshawytscha*), Puget Sound ESU

Chinook salmon, also called king salmon, are the largest and least abundant species of Pacific salmon (Good, Waples, & Adams, 2005). 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

then die. Compared to other Pacific salmon species, Chinook prefer larger and deeper stream habitat (Thomas P. Good et al., 2005).

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, 2016d).

NMFS recognizes three ESA-listed ESUs of Chinook salmon that spawn in Washington, the two Columbia River ESUs and a single Puget Sound ESU listed in August 1999 (64 FR 41835). In 2005, NMFS 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 (Thomas P. Good et al., 2005). The five-year status review completed in 2010 (76 FR 50448) concluded that all Chinook salmon ESUs should remain listed. Each ESU is treated as a separate species under the ESA (76 FR 50448). ESUs may include both naturally spawned and artificially propagated (hatchery stock) fish.

3.2.1 Distribution within the action area

The Skagit River watershed supports salmon spawning, but the streams within the action area do not have spawning habitat. Some fraction of the spring, summer and fall Chinook that migrate up the Skagit River are likely pass through the southern part of the action area. According to the Washington Department of Fish and Wildlife's SalmonScape (https://apps.wdfw.wa.gov/salmonscape/) these fish do not migrate through the Swinomish channel, but it is possible that Chinook could be in any of the marine waters within the action area.

Historical estimates of spawner capacity are higher than spawner abundances observed throughout the Puget Sound ESU (T P Good et al., 2005). The contribution of hatchery reared fish to the abundance of Skagit River Chinook salmon appears to be low. Population abundance information for Skagit River Chinook salmon is summarized in Table 8.

Table 8. Puget Sound Chinook salmon preliminary population structure, abundances, and hatchery contributions (T P Good et al., 2005)

Independent Populations	**Historical Abundance** (model estimated)	**Mean Number of Spawners** (geometric mean 1998-2002)	**Hatchery Abundance Contributions**
Lower Skagit Watershed	22,000	2,527	0.2%
Upper Skagit Watershed	35,000	9,489	2%

Annual variability in the upper and lower Skagit River Chinook populations is shown in Figure 4 and Figure 5 (https://data.wa.gov/Natural-Resources-Environment/WDFW-Salmonid-Stock-Inventory-Population-Escapemen/fgyz-n3uk).

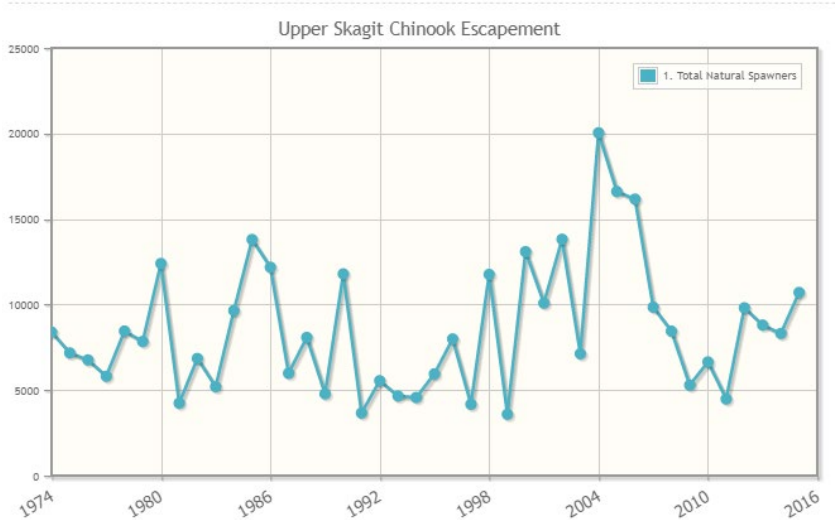


Figure 4: Upper Skagit Chinook escapement plotted by year (WDFW Dataset)

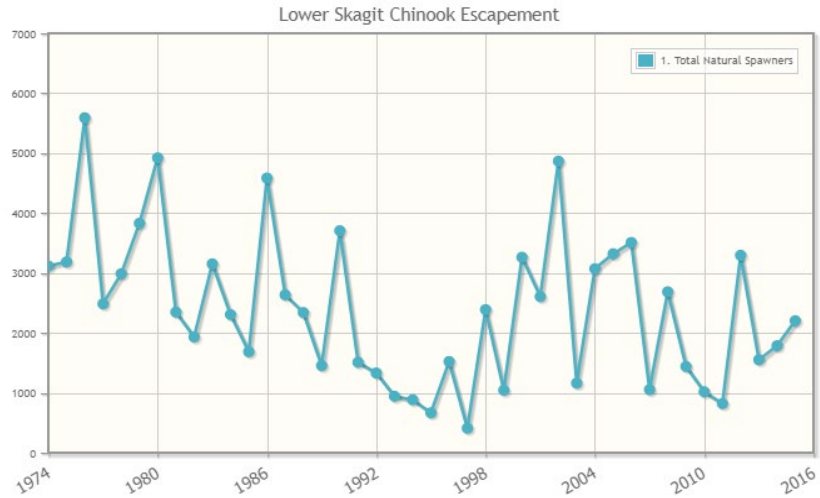


Figure 5: Lower Skagit Chinook escapement plotted by year (WDFW Dataset)

Chinook salmon juveniles out-migrate from natal rivers and streams into marine waters as sub-yearlings or yearlings, and return to spawn as adults, generally after 3 to 5 years. Most Puget Sound Chinook head to coastal waters, but some remain in Puget Sound for a portion or all of their marine residence ((Pressey, 1953), as cited in Chamberlin, Kagley, Fresh, and Quinn (2011)). Smaller outmigrants tend to migrate in schools along nearshore areas (Nightingale & Simenstad, 2001). Larger outmigrants are not associated with the nearshore. In nearshore areas of Puget Sound, juvenile Chinook salmon outmigration peaks in June and July, and then slowly decreases through the fall (Fresh, 2006).

Juvenile Chinook use pocket estuaries near the mouths of the main stem and North Fork Skagit River, including within the action area, primarily in the late winter through the spring months. Greater numbers were found at estuaries closest to the Skagit River relative to the action area (Griffith, Tribe, Zackey, & Tribes, 2006). Juvenile outmigrants are present around the entire Swinomish reservation close to shore and use lagoons and areas near small streams (E. Beamer, Swinomish Tribe. Pers. comm. with

A. LaTier, EPA R10, 12/20/2019). Adult Chinook, including those that migrate through the action area to the Skagit River, do not feed on their way to the spawning areas.

3.2.2 Critical habitat within the action area

Critical habitat for this ESU was designated in 2005 (70 FR 52630) (Figure 6). Critical habitat for the species includes all waters that drain into Puget Sound in which Chinook salmon rear and naturally spawn or are planted (by hatcheries). Specifically, the designated critical habitat includes 2709 km of streams, 106 km^2 of lakes and 3721 km of nearshore marine habitat.

The designation for this ESU includes sites necessary to support one or more life stages. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The primary constituent elements (PCEs) of Puget Sound Chinook salmon critical habitat are:

- Freshwater spawning sites with water quantity and quality conditions and substrate that support spawning, incubation, and larval development;
- Freshwater rearing sites with (1) water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility, (2) water quality and forage that support juvenile development, and (3) natural cover such as shade, submerged and overhanging large wood, logjams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks;
- Freshwater migration corridors free of obstruction and excessive predation with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks that support juvenile and adult mobility and survival;
- Estuarine areas free of obstruction and excessive predation with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation;
- Nearshore marine areas free of obstruction and excessive predation with water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and
- Offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.

Source: (NMFS, 2016g)

Since critical habitat was not designated for Swinomish Tribal waters, the area within the 400-foot buffer waterward of the Swinomish Reservation waters boundary and the non-Tribal portion of the Swinomish Channel is the portion of critical habitat that will be evaluated in this consultation. This area of critical habitat is 515 acres (= 208 hectares)(Figure 2).

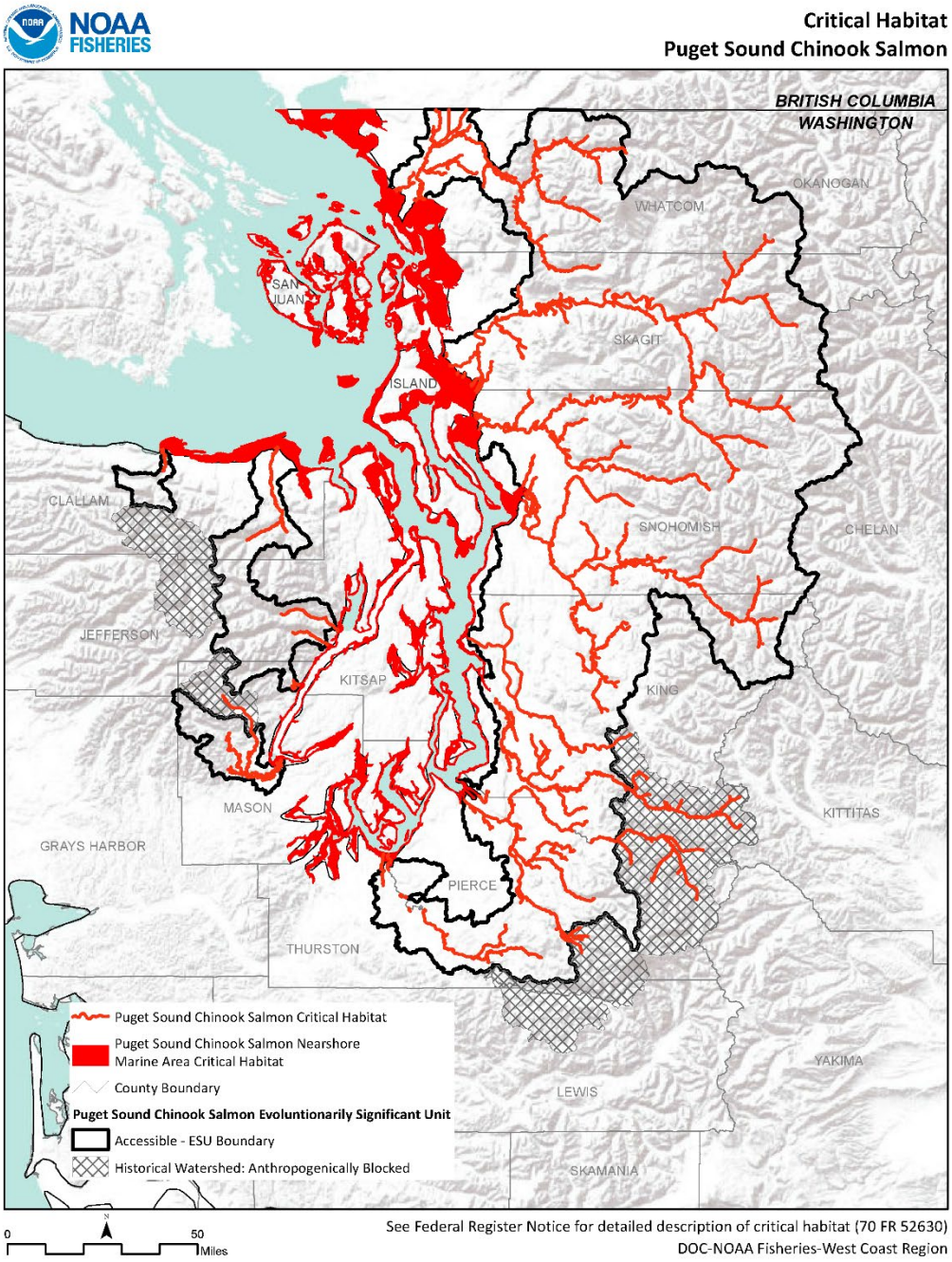


Figure 6: Puget Sound Chinook critical habitat

3.2.3 Life history

Chinook salmon exhibit consistent seasonal immigration patterns, with groups "running" (i.e., returning to freshwater streams) during spring, early summer, or early fall. Typically, spring-run Puget Sound Chinook salmon return to freshwater between April and May and spawn between August and September (Orrell (1976) as cited in (Myers et al., 1998); (Washington Department of Fisheries,

Washington Department of Wildlife, & Western Washington Treaty Indian Tribes, 1993)). Pre-spawn adults migrate upstream into pools, where they physiologically mature in preparation for spawning. Summer-run fish return to freshwater streams between June and July and spawn in September. Fall-run Chinook salmon return in August and spawn between late September and January (Washington Department of Fisheries et al., 1993). Within a single river, spawning times of different seasonal runs may overlap, but geographic separation maintains some amount of reproductive isolation between runs (Myers et al., 1998). There are three runs of Chinook in the Lower Skagit River, including spring, summer, and fall. Most Puget Sound Chinook salmon migrate to the sea as young-of-year juveniles, where they often reside in estuaries that provide important nursery habitat for smolting juveniles; these habitat types are present within the action area.

3.2.4 Current stressors and threats

Limiting factors for Puget Sound Chinook salmon include degraded floodplain and in-river channel structure, degraded estuarine conditions and loss of estuarine habitat, degradation of riparian habitat and loss of in-river large woody debris, excessive fine-grained sediment in spawning gravels, degraded water quality and temperature (i.e., increased temperatures), degraded nearshore conditions, impaired passage for migrating fish, and severely altered flow regimes. Other stressors include commercial and recreational harvest or bycatch (accidental) and predation (e.g., by killer whale) (Windward 2018 (USEPA, 2018c)).

[1] “Young of year” is a term used to describe individuals of age less than one year.

[2] Smolting is a physiological change that allows anadromous salmon to move from freshwater to saltwater environments.

3.3 Chum Salmon (*Oncorhynchus keta*), Hood Canal Summer-Run ESU

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 to 1.1 m (3.6 ft) 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 they migrate as far north as Alaska, as far south as California, and as far west as Russia and Japan (R J Beamish & Bouillon, 1993). On June 28, 2005, the NMFS listed Hood Canal summer-run (HCS) chum salmon as a threatened species (70 FR 37160). This ESU includes all non-hatchery and some hatchery-raised individuals within the ESU designated distribution.

3.3.1 Distribution

The HCS chum salmon ESU comprises all naturally spawned populations of summer-run chum salmon in Hood Canal and its tributaries, as well as populations in Olympic Peninsula rivers between Hood Canal

and Dungeness Bay, Washington. Four artificial propagation programs were listed as part of the ESU (79 FR 20802): the Hamma Fish Hatchery Program, Lilliwaup Creek Fish Hatchery Program, Tahuya River Program, and Jimmycomelately Creek Fish Hatchery Program.

The HCS chum salmon ESU has two populations (Strait of Juan de Fuca and Hood Canal populations), each containing multiple stocks or spawning aggregations. In the Strait of Juan de Fuca population, there are small but persistent natural spawning aggregations in three streams: Salmon, Snow, and Jimmycomelately Creeks. In the Dungeness River, spawning occurs but the aggregation of spawners is not known. In Chimacum Creek, HCS chum salmon were extirpated in the mid-1980s. Spawning aggregations have persisted in most of the major rivers draining from the Olympic Mountains into the western edge of Hood Canal, including Big Quilcene, Little Quilcene, Dosewallips, Duckabush, and Hamma Rivers and Lilliwaup Creek. On the eastern side of Hood Canal, persistent spawning is restricted to the Union River (Sands et al., 2009). Based on river size and historical Tribal fishing records, a major spawning aggregation once occurred in the Skokomish River before the construction of Cushman Dam in the 1920s. State and Tribal biologists also identified recent extinctions in Big Beef Creek, Anderson Creek, Dewatto River, Tahuya River, and Finch Creek. Historically, streams including but not limited to Seabeck, Stavis, Big Mission, and Little Mission Creek probably supported summer-run chum salmon.

No information about the presence of Hood Canal summer-run chum salmon in the Swinomish action area was identified and the streams within the action area do not have spawning habitat.

3.3.2 Critical habitat within the action area

Critical habitat for HCS chum salmon was designated for Washington in 2005 and includes freshwater, estuarine, and marine areas shown in Figure 7 (70 FR 52630). There are approximately 127 km (79 miles) of stream habitats and 607 km (377 miles) of nearshore marine habitats designated as critical habitat. The essential physical and biological features (PBFs) for HCS chum salmon are the same as those described above for Puget Sound Chinook salmon.

Critical habitat for the HCS summer-run chum salmon does not occur in the Swinomish action area.

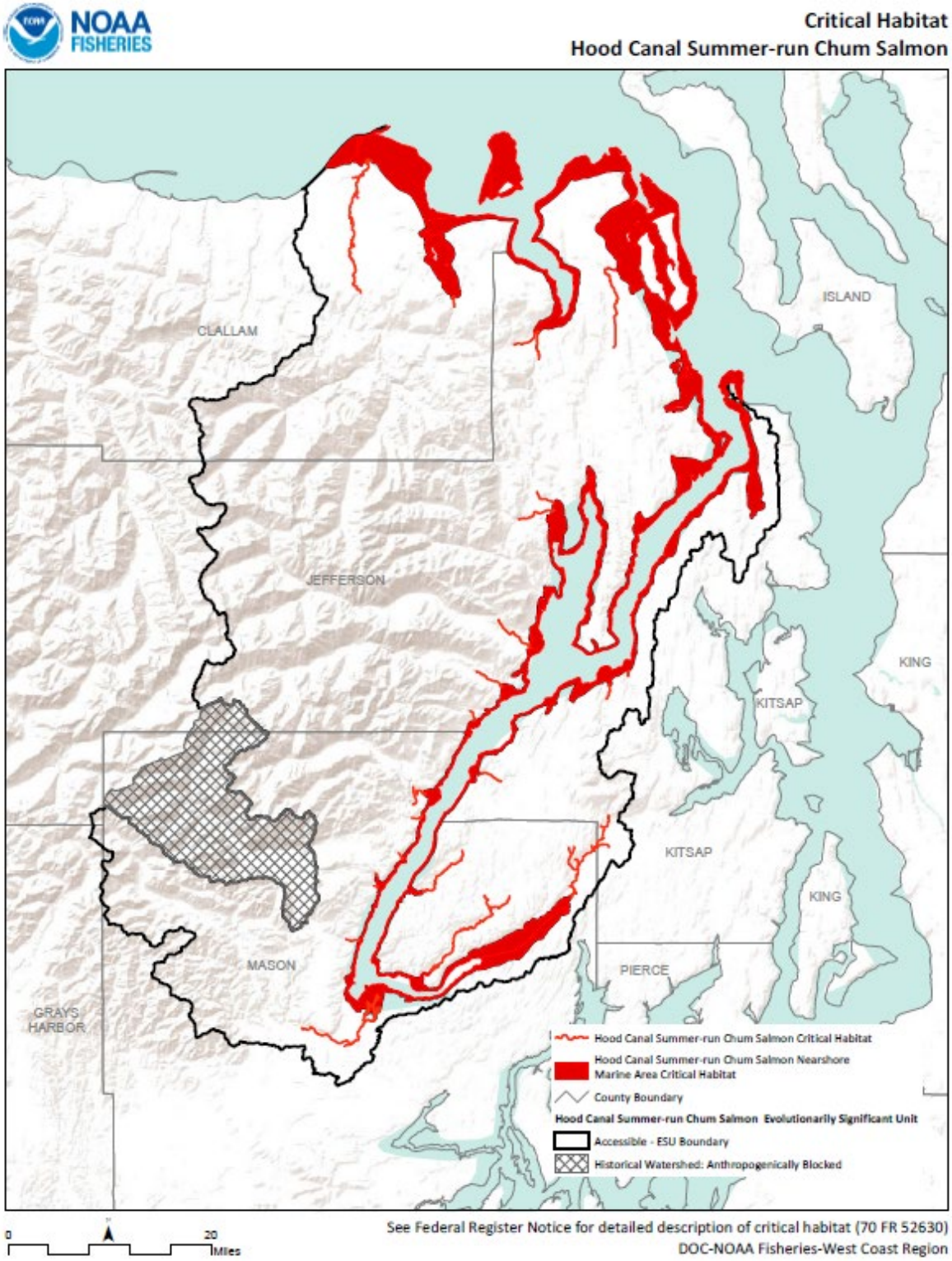


Figure 7: Critical habitat for Hood Canal summer-run chum salmon. Source: NOAA Fisheries 2016b (NMFS, 2016f)

3.3.3 Life history

HCS chum salmon return to freshwater during summer. Juveniles, typically as fry, emerge from the gravel and emigrate almost immediately to seawater, indicative of an ocean-type life strategy. Upon reaching saltwater, HCS chum salmon spend several weeks in the top 2 to 3 (0.79 to 1.18 inches) of estuarine surface waters very close to the shoreline (WDFW & PNPTT, 2000). Subadults and adults forage in coastal and offshore waters of the North Pacific Ocean before returning to spawn in their natal streams at age three to five. HCS chum salmon spawn from mid-September to mid-October, typically in river mainstems and lower river basins. HCS chum salmon ESU fecundity estimates average 2,500 eggs per female, and the proportion of female spawners is approximately 45% of escapement in most populations (WDFW & PNPTT, 2000).

3.3.4 Current stressors and threats

Limiting factors for the HCS chum salmon ESU include degraded habitat, barriers to migration, and changes to the salmon prey base. More specifically, limitations on habitat are caused by degraded water quality, loss of floodplain connectivity and function, loss of channel structure and complexity, loss of riparian habitat, reduced large woody debris recruitment, altered stream substrate (e.g., embeddedness by fine sediments), and altered stream flow (NMFS, 2016f).

Human activities that affect the PBFs of critical habitat for HCS chum salmon include forestry; agriculture; road building/maintenance; channel modifications/diking; urbanization; sand and gravel mining; dams; river, estuary, and ocean traffic; and the removal of beavers. In addition to these, the harvest of salmonid prey species (e.g., herring, anchovy, and sardines) affects nearshore marine PBFs. Stream channels and estuaries are, with few exceptions, moderately to highly degraded throughout the HCS chum salmon ESU. During the past 150 years, logging, road building, rural development, agriculture, water withdrawal, and channel manipulations (e.g., dredging and channelization) were common and widespread, especially within low gradient stream reaches utilized by summer-run chum salmon. Three quarters of the HCS chum salmon ESU's watersheds contain simplified, degraded channels either completely lacking a forested riparian zone or surrounded by small diameter, deciduous-dominated forests. Most streams have degraded or reduced pool densities and insufficient large woody debris. Development has occurred in nearly all estuaries within Hood Canal and the eastern Strait of Juan de Fuca. Dikes, roads or causeways, ditches, and fill are the primary causes of estuarine habitat degradation. In estuarine and nearshore areas, bulkheads, revetments, and impaired riparian corridors have reduced the amount of rearing habitat. Altered river and tidal dynamics have likely reduced estuarine food web productivity and, thus, the carrying capacity for chum salmon and other salmonids in the estuarine environment (NMFS, 2016c).

Based on the most recent 5-year status review, NMFS (NWFSC, 2015) found that the overall trend in spawning abundance, productivity, and spatial structure and viability parameters has increased since ESA-listing for the Hood Canal population (all natural spawners and natural-origin only spawners) and Strait of Juan de Fuca population (all natural spawners). However, the ESU did not meet all of the recovery criteria for population viability (NWFSC, 2015).

3.4 Steelhead (*Oncorhynchus mykiss*), Puget Sound DPS

The Puget Sound steelhead distinct population segment (DPS) was federally listed as threatened in 2007 (72 FR 26722) and includes all naturally spawned, anadromous steelhead populations in river basins draining to the Strait of Juan de Fuca, Puget Sound, and Hood Canal, Washington.

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 that (NMFS, 2016i). 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, 2016i).

3.4.1 Distribution within the action area

The Puget Sound steelhead DPS area is bounded to the west by the Elwha River (inclusive) and to the north by the Nooksack River and Dakota Creek (inclusive). The Puget Sound steelhead DPS also includes six hatchery stocks that are considered relatively similar (genetically) to their associated non-hatchery counterparts (79 FR 20802). Non-anadromous "resident" O. mykiss occur within the range of Puget Sound steelhead DPS but are not part of the DPS due to marked differences in physical, physiological, ecological, and behavioral characteristics (Hard, 2007). The action area does not include steelhead spawning streams.

3.4.2 Critical habitat within the action area

Critical habitat was designated for the Puget Sound steelhead DPS on February 24, 2016 (81 FR 9252). There are approximately 3,269 km (2,031 miles) of freshwater and estuarine habitat designated as critical habitat for Puget Sound steelhead (Figure 8). There are 18 sub-basins containing 66 watersheds within the range of the Puget Sound steelhead DPS. The PBFs for West Coast steelhead critical habitat are provided in Table 9. The action area does not include critical habitat for steelhead; the nearest critical habitat is south of the Swinomish action area in the Skagit river.

Figure 8: Critical habitat for Puget Sound steelhead trout

Table 9. Physical and Biological Features of Designated Steelhead Critical Habitat

Critical 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	Juvenile development
	Natural cover	Juvenile mobility and survival
Freshwater migration	Free of artificial obstructions, water quality and quantity, and natural cover	Juvenile and adult mobility and survival
Estuarine	Water quality, water quantity, and salinity	Juvenile and adult physiological transitions
	Natural cover and forage	Juvenile growth and maturation and adult conservation
Nearshore marine	Water quality, water quantity, forage, natural cover	Juvenile growth and maturation
Offshore marine	Water quality and forage	Juvenile growth and maturation

3.4.3 Life history

Steelhead in the NW mature in one of two distinct modes, either stream-maturing or ocean-maturing (NMFS, 2016i). 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 re-enter 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, 2016i). Winter-run salmon are the predominant group in the Puget Sound DPS, which means that most steelhead mature in the ocean and return to streams to spawn in winter or early spring (Myers et al., 2015). In lowland, rain-dominated streams, steelhead tend to return earlier than in higher elevation, snowmelt dominated streams.

Spawning occurs over coarse substrates (gravel) in cold, fast-flowing streams with highly oxygenated waters, and spawning may occur more than once (NMFS, 2016i). After hatching (three to four weeks after spawning), steelhead may reside in freshwater streams for up to seven years before migrating into estuaries to smolt, and they may reside in marine environments for three years (NMFS, 2016i). 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, 2016i).

3.4.4 Current stressors and threats

Limiting factors for Puget Sound steelhead trout include widespread declines in adult abundance (total run size) despite significant reductions in harvest, reduced diversity resulting from two hatchery steelhead stocks (i.e., Chambers Creek and Skamania), uncertain but weak status of summer-runs, and reduced spatial structure. Reduced habitat quality and quantity are also key limitations. The major categories of human activities with the potential to impact Puget Sound steelhead PBFs include forestry, grazing, agriculture, road building/maintenance, channel modifications/diking, urbanization, sand and gravel mining, mineral mining, dams, irrigation impoundments and withdrawals, vessel traffic, wetland loss/removal, beaver removal, and exotic/invasive species introductions. In addition to these, the harvest of salmonid prey (e.g., herring, anchovy, and sardines) adversely influences nearshore marine PBFs.

Dams have dramatically affected steelhead habitat use in a number of Puget Sound sub-basins. In addition to eliminating accessible habitat, dams affect habitat quality by changing river hydrology, temperature, downstream gravel recruitment, and large woody debris movement. Dams have impeded upstream access to historical steelhead habitat in the Middle Fork Nooksack, Baker, Cedar, Green, White, Nisqually, and North Fork Skokomish Rivers. “Trap-and-haul” programs (capture of live spawners and transport above impassible dams) have made passage of Puget Sound steelhead above the dams on the Baker and White Rivers possible. A smolt collection facility has similarly made downstream passage of juveniles possible on the Baker River. On the White River, downstream migrants can pass directly through the dams.

Urban development has dramatically altered many of the lower reaches of rivers and their tributaries in Puget Sound. Urbanization has destroyed historical land cover (e.g., forests) and exchanged it for large areas of impervious surfaces (e.g., roads). Wetland and riparian habitat loss has dramatically changed urban stream hydrology by increasing flood frequency and peak flows during storm events while decreasing groundwater-driven summer flows. Conversion to agricultural land has impacted river morphology, since much of this type of development occurs in river floodplains. Dike construction, bank hardening, and channelization have reduced river braiding and sinuosity. Constricting a river, especially during high flow events, increases the likelihood of gravel scour and dislocation of rearing juveniles.

Habitat blockage and/or degradation occur throughout the Puget Sound steelhead DPS range. In general, upper tributaries have been adversely affected by forest practices, whereas lower tributaries and mainstem rivers have been degraded by agriculture and/or urbanization. Diking for flood control, draining and filling freshwater and estuarine wetlands, and sedimentation from timber harvests and urban development are cited as problems throughout the Puget Sound steelhead DPS (Washington Department of Fisheries et al., 1993). Bishop and Morgan (Bishop & Morgan, 1996; Washington Department of Fisheries et al., 1993) identified a variety of stream habitat limitations in the range of this species including flow regime changes, sedimentation, high temperatures (in the Dungeness, Elwha, Green/Duwamish, Skagit, Snohomish, and Stillaguamish Rivers), streambed instability, estuarine loss, large woody debris loss (in the Elwha, Snohomish, and White Rivers), pool habitat loss (in the Nooksack,

Snohomish, and Stillaguamish Rivers), and blockage or passage problems associated with dams or other structures (in the Cedar, Green/Duwamish, Snohomish, and White Rivers).

3.5 Bocaccio (*Sebastes paucispinis*)

On April 27, 2010, the NMFS listed the Puget Sound/Georgia Basin (PS/GB) DPS of bocaccio as endangered (75 FR 22276). The listing includes populations occupying all waters of Puget Sound, the Strait of Juan de Fuca east of Victoria Sill, and south of the Northern Strait of Georgia. A recent 5-year status review concluded that the status of the PS/GB bocaccio DPS had not improved significantly since they were listed in 2010 (NMFS, 2016a). A conservation plan to aid in recovery of Puget Sound rockfish was developed by Washington Department of Fish and Wildlife (WDFW) and approved by NMFS in 2011 (Bargmann, Palsson, Burley, Friedel, & Tsou, 2011).

3.5.1 Distribution in the action area

The geographic range of the listed PS/GB rockfish DPS (including bocaccio) includes Puget Sound, Georgia Basin, the Strait of Georgia, and the Strait of Juan de Fuca east of Victoria Sill, therefore juvenile bocaccio rockfish are potentially found in all marine waters within the action area. Bocaccio are found from Stepovac Bay on the Alaska Peninsula to Punta Blanca in central Baja California (NMFS, 2014a). Information on habitat requirements for most rockfishes is limited despite years of research, and even less is known about bocaccio in Puget Sound (Drake et al., 2010; Palsson et al., 2009).

Palsson et al. (2009) reviewed Puget Sound rockfish species distributions and the relative number of occurrences. This review relied heavily on Miller and Borton (1980) data, but also included the review of historical literature, fish collections, unpublished log records, and other sources. Palsson et al. (2009) noted bocaccio were only recorded 110 times in their review of historical studies, with most records associated with sport catches from the 1970s in Tacoma Narrows and Appletree Cove (near Kingston). Bocaccio have been documented in Hood Canal, Possession Sound, and the west side of Bainbridge Island (NMFS, 2014a; USEPA, 2007b). In northern Puget Sound, bocaccio comprised less than 0.2 percent of recreational rockfish catch between 1980 and 2007 and comprised this same percentage in southern Puget Sound during the 1980s but prior to 1996. Bocaccio was not observed after 1996 in southern Puget Sound (Palsson et al., 2009). A recent survey by WDFW detected one bocaccio in the main basin of Puget Sound ((Frierson et al., 2015), EPA-WDFW comm.).

The West side of the action area has nearshore habitat with sand, rock, and/or cobble suitable for use by young of the year boccacio.

3.5.2 Critical habitat

Critical habitat was designated for both PS/GB DPS of bocaccio and yelloweye rockfish on November 13, 2014 (79 FR 68042 31F). The critical habitat in the US includes five interconnected, biogeographic basins: The San Juan/Strait of Juan de Fuca, Main, Whidbey, South Puget Sound, and Hood Canal Basins; detailed maps of these area are provided in the Federal Register (79 FR 68042), see Figure 9. Nearly 1,500 sq km (600 square miles) of nearshore habitat in Puget Sound, Washington, is PS/GB bocaccio critical habitat. Nearshore critical habitat consists of underwater substrates such as sand, rock, and/or cobble compositions from extreme high water to 30 m (98 ft) deep (the limit of the photic zone in Puget Sound). This critical habitat supports kelp, enables forage opportunities and refuge from predators, and enables behavioral and physiological changes needed for juveniles to eventually occupy deeper adult

habitats. The physical and biological features (PBFs) of nearshore habitats include sufficient quantity, quality, and availability of prey species and suitable water quality and dissolved oxygen to support growth, survival, reproduction, and feeding opportunities.

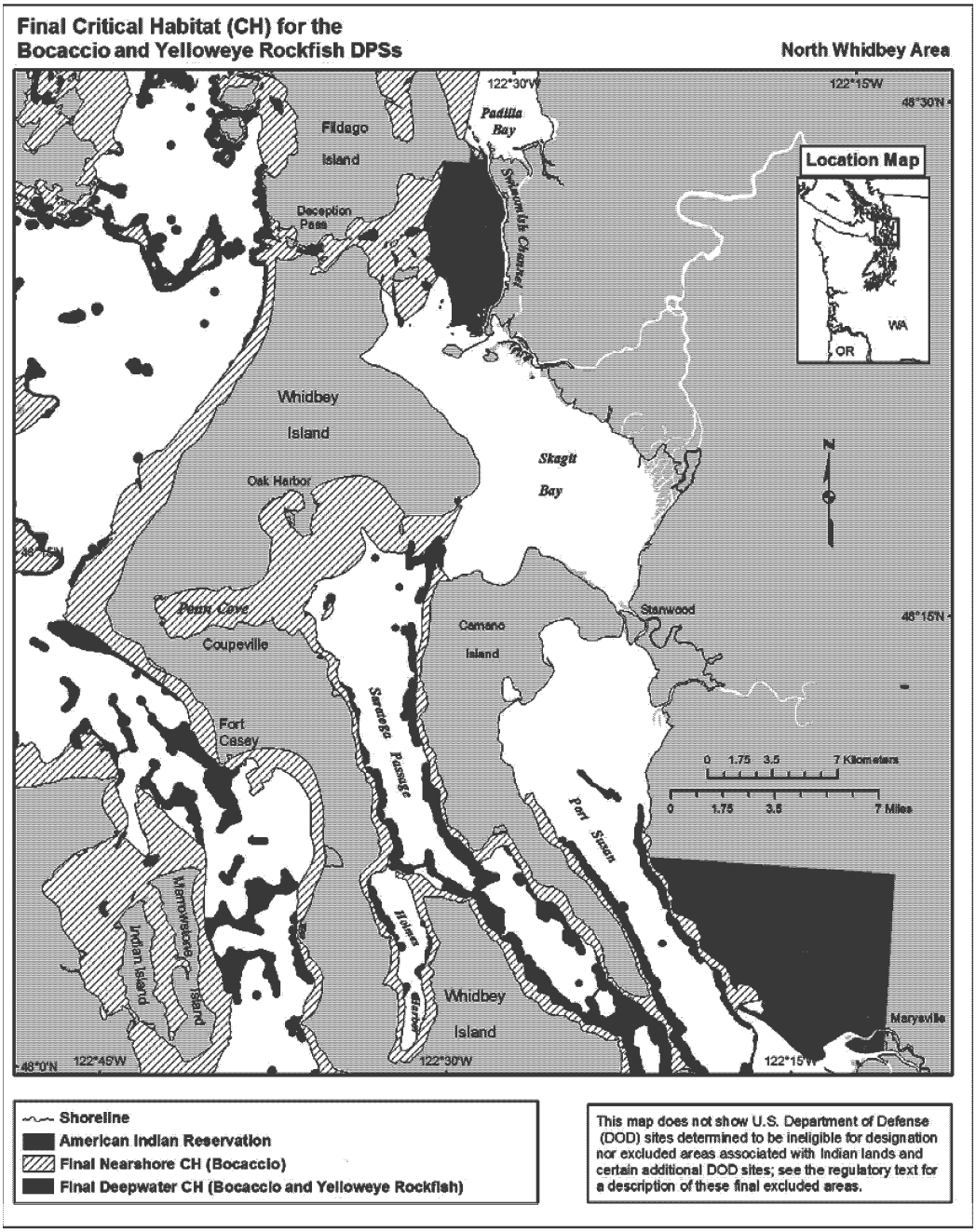


Figure 9: Final critical habitat for the bocaccio and yelloweye rockfish DPSs

Deepwater critical habitat consists of sites deeper than 30 m (98 ft) that possess or are adjacent to areas of complex bathymetry consisting of rock or other highly rough surfaces. This habitat is essential to conservation because these features provide structure for rockfish to avoid predation, seek food, and

persist for decades. The PBFs for deep water habitats include sufficient quantity, quality, and availability of prey species; sufficient water quality and dissolved oxygen; and appropriate types and amounts of structure and roughness that supports feeding opportunities and predator avoidance.

The 400-foot buffer area between the Swinomish Reservation boundary and the action area boundary on the west side includes bocaccio rockfish critical habitat.

3.5.3 Life history

Bocaccio life history includes a larval/pelagic juvenile stage followed by a nearshore juvenile stage, and sub-adult and adult stages. In contrast to the majority of bony fishes, rockfish fertilize their eggs internally, and the young are extruded as larvae. Individual female bocaccio produce from 20,000 to 2,298,000 eggs, and as females grow and age, the number of young produced per individual increases (Love, Yoklavich, & Thorsteinson, 2002). Larval release timing varies throughout the geographic range. Along the Washington State coast, female bocaccio release larvae between January and April. Larvae are observed under free-floating algae, seagrass, and detached kelp (Love et al., 2002); (Shaffer, Doty, Buckley, & West, 1995) but are also distributed throughout the water column (L. J. Weis, 2004). Larvae can make small local movements to pursue food immediately after birth (Tagal et al., 2002), but dispersal is driven by passive movement on prevailing currents. In Puget Sound, sills regulate water exchange from one basin to the next, likely resulting in most larvae staying within the basin where they are released rather than being broadly dispersed (Drake et al., 2010). On average, larval bocaccio occur at approximately 25 to 30 m (Lenarz, Larson, & Ralston, 1991).

Most bocaccio remain pelagic for 3.5 months prior to settling in shallow areas, though some individuals remain pelagic as long as 5.5 months. Several weeks after settlement, fish tend to move toward deeper nearshore waters over rocky or cobble substrates with or without kelp ((Love, Carr, & Haldorson, 1991);(Love et al., 2002). These habitat features offer a beneficial mix of warmer temperatures, food, and refuge from predators (Love et al., 1991). Young-of-the-year are often found in shallow, nearshore waters over rocky bottoms associated with algae, within and near kelp canopies, and in 18 to 30 m (59 to 98 ft) deep waters associated with rocky reefs and high relief areas ((Carr, 1983; Feder, Turner, & Limbaugh, 1974; D. W. Johnson, 2006; Love & Yoklavich, 2008; Sakuma & Ralston, 1995)) Pelagic juveniles opportunistically feed on fish larvae, copepods, krill, and other prey. Larger juveniles and adults are primarily piscivores, eating other rockfishes, hake, sablefish, anchovies, lanternfishes, and squid. Chinook salmon, terns, and harbor seals are predators of smaller bocaccio (Love et al., 2002).

Bocaccio typically mature between ages three and eight, at lengths from 32 to 61 cm (13 to 24 inches) (Love et al., 2002; Wyllie-Echeverria, 1987), and bocaccio may begin to mature at earlier ages in declining populations (MacCall, 2002). Sub-adult and adult bocaccio typically utilize habitats with moderate to extreme steepness, complex bathymetry, and rocky substrates (Love et al., 2002), although they may also be associated with unconsolidated sediments (e.g., silt, sand, or clay) (Borton & Miller, 1980; Washington, 1977). Bocaccio have large home ranges, move long distances, and spend time swimming in the water column (Love et al., 2002). Adult bocaccio inhabit waters from 12 to 478 m, but they are most common at depths between 50 and 250 m (Feder et al., 1974; Love et al., 2002; Orr, Brown, & Baker, 2000b). Some adults are semi-pelagic and form schools above rocky areas, while some are non-schooling, solitary, benthic individuals (Yoklavich et al., 2000). Solitary bocaccio are associated with large sea anemones, rock ledges, and crevices of isolated rock outcrops (Yoklavich et al., 2000).

Though difficult to age, adults may live as long as 54 years (Drake et al., 2010). Their natural annual mortality is approximately 8% (Palsson et al., 2009). Bocaccio typically mature between ages three and eight, at lengths from 32 to 61 cm (13 to 24 inches) (Love et al., 2002; Wyllie-Echeverria, 1987), and bocaccio may begin to mature at earlier ages in declining populations (MacCall, 2002). Sub-adult and adult bocaccio typically utilize habitats with moderate to extreme steepness, complex bathymetry, and rocky substrates (Love et al., 2002), although they may also be associated with unconsolidated sediments (e.g., silt, sand, or clay) (Borton & Miller, 1980; Washington, 1977). Bocaccio have large home ranges, move long distances, and spend time swimming in the water column (Love et al., 2002). Adult bocaccio inhabit waters from 12 to 478 m, but they are most common at depths between 50 and 250 m (Feder et al., 1974; Love et al., 2002; Orr, Brown, & Baker, 2000a). Some adults are semi-pelagic and form schools above rocky areas, while some are non-schooling, solitary, benthic individuals (Yoklavich et al., 2000). Solitary bocaccio are associated with large sea anemones, rock ledges, and crevices of isolated rock outcrops (Yoklavich et al., 2000). Though difficult to age, adults may live as long as 54 years (Drake et al., 2010). Their natural annual mortality is approximately 8% (Palsson et al., 2009).

Bocaccio are long-lived, mature slowly, and have sporadic episodes of successful reproduction; (Drake et al., 2010; Tolimieri & Levin, 2005), which suggests that they have inherently low productivity. PS/GB bocaccio have a very low intrinsic rate of population growth, even in the absence of a targeted fishery (Tolimieri & Levin, 2005). Productivity in bocaccio is driven by high fecundity and episodic recruitment events, largely correlated with rare climatic and oceanographic conditions. Demographically, this species demonstrates some of the highest recruitment variability among rockfish species, with many years of failed recruitment being the norm (Tolimieri & Levin, 2005).

3.5.4 Current stressors and threats

Benthic habitat degradation within PS/GB waters is a threat to listed rockfish, including the presence of derelict fishing gear, loss of eelgrass and kelp habitat, introduction of non-native species, and degradation of water quality (75 FR 22276). Derelict fishing gear is considered a threat to the PS/GB bocaccio DPS (75 FR 22276); up to 117,000 derelict nets and pots are estimated to lie beneath the waters of Puget Sound (WDFW, unpublished data). Most derelict gillnets were recovered from high-relief habitats featuring rocky ledges and boulders (Drake et al., 2010), which are habitats frequently used by subadult and adult rockfish (Love et al., 2002). Gilardi et al. (Drake et al., 2010; Gilardi et al., 2010) estimated from their Puget Sound study of derelict gillnets that a fish becomes entangled in each derelict net every 3.6 days.

Non-native species are an emerging threat to the native Puget Sound biotic habitat. For example, Sargassum muiticum, an introduced brown algae now common throughout much of Puget Sound, is a competitor of kelp and eelgrass (Mumford, 2007). Invasive tunicates may also impact Puget Sound rockfish or their habitats based on observations from other regions (Levin, Coyer, Petrik, & Good, 2002) that suggest the potential for widespread impacts on rocky-reef fish populations.

Habitat degradation for kelp and eelgrass is another potential threat to PS/GB bocaccio. Kelp coverage is highly variable and has shown long-term declines in some regions, though some kelp beds have increased in areas where artificial substrate provides sufficient habitat structure (Palsson et al., 2009). Kelp and eelgrass are stressed by light availability (which is reduced by turbidity), nutrient levels (e.g., eutrophication), pollutants (e.g., oils, metals, or sulfides), and physical disturbance (e.g., propeller wash

and scour) (Mumford, 2007). Kelp and eelgrass habitats are important for larval and young juvenile rockfish (Love et al., 2002). Low dissolved oxygen levels are also of increasing concern in the PS/GB region (Palsson et al., 2009).

Boccacio are believed to have commonly occurred along steep walls in most of Puget Sound prior to fishery exploitations, although they are currently very rare in these habitats (Love et al., 2002). Although rockfish are typically long-lived, recruitment is generally poor as larval survival and settlement are dependent on a variety of factors including marine currents, adult abundance, habitat availability, and predator abundance (Drake et al., 2010; Love, 2002; Love et al., 2002; Palsson et al., 2009). The combination of these factors, and the threats described above, has contributed to declines in the species within Georgia Basin and Puget Sound in the last few decades (75 FR 22276). A recent 5-year status review concluded that the statuses of the PS/GB yelloweye and bocaccio DPSs had not improved significantly since they were listed in 2010 (NMFS, 2016a). A conservation plan to aid in recovery of Puget Sound rockfish was developed by WDFW and approved by NMFS in 2011 (Bargmann et al., 2011).

3.6 Yelloweye Rockfish (*Sebastis ruberrimus*)

On April 27, 2010, the NMFS listed the PS/GB DPS of yelloweye rockfish as threatened (75 FR 22276). The population of interest to the action area is the Puget Sound/Georgia Basin DPS. On April 27, 2010, the NMFS listed the Puget Sound/Georgia Basin (PS/GB) DPS of yelloweye as threatened (75 FR 22276). The listing includes populations occupying all waters of Puget Sound, the Strait of Juan de Fuca east of Victoria Sill, and south of the Northern Strait of Georgia.

3.6.1 Distribution within the action area

The geographic range of the listed Puget Sound/Georgia Basin (PS/GB) rockfish DPS includes Puget Sound, Georgia Basin, the Strait of Georgia, and the Strait of Juan de Fuca east of Victoria Sill.

Prior to contemporary fishery harvests, each major basin in the PS/GB yelloweye rockfish DPS likely hosted relatively large yelloweye rockfish populations (Moulton & Miller, 1987; Washington, 1977; Washington, Gowan, & Ito, 1978). Currently, yelloweye rockfish are probably most abundant within the San Juan Islands Basin, but the likelihood of juvenile recruitment from this basin to adjacent basins is low due to limited circulation of currents from that basin to adjacent basins.

Within the action area, juvenile rockfish may use the limited habitat on the west side of the action area that is below 15m as described in the Life History section below.

3.6.2 Critical habitat

The critical habitat for yelloweye rockfish is the same as that for the PS/GB bocaccio rockfish DPS described above. The action area includes critical habitat for yelloweye rockfish.

3.6.3 Life history

The yelloweye rockfish life-history includes a larval/pelagic juvenile stage followed by a nearshore juvenile stage, and sub-adult and adult stages. Yelloweye rockfish are among the largest rockfish, weighing up to 11 kg (25 pounds), and they are easily recognizable by their bright yellow eyes and red-orange color (Love et al., 2002). Adult female yelloweye rockfish may store sperm internally for several months until fertilization occurs, commonly between September and April, though fertilized individuals

may be found year-round, depending on location (Wyllie-Echeverria, 1987). In Puget Sound, yelloweye rockfish are believed to fertilize eggs during the winter to summer months and give birth in early spring to late summer (Washington et al., 1978). Fecundity ranges from 1.2 to 2.7 million eggs, considerably more than many other rockfish species (Love et al., 2002). Although yelloweye rockfish are generally thought to spawn once a year (MacGregor, 1970), a Puget Sound study offered evidence of at least two spawning periods per year (Washington et al., 1978).

Larval yelloweye rockfish can make small local movements to pursue food immediately after birth (Tagal et al., 2002), but dispersal is likely driven by passive dispersion on prevailing currents. In the Puget Sound, sills regulate water exchange from one basin to the next, likely resulting in limited dispersal between sub-basins (Drake et al., 2010). Larvae are observed under free-floating algae, seagrass, and detached kelp (Love et al., 2002; Shaffer et al., 1995), but they are also distributed throughout the water column (L. J. Weis, 2004). Larval yelloweye rockfish remain pelagic for up to three months.

When yelloweye rockfish reach sizes of 2.5 to 10 cm (1 to 4 inches), they settle primarily in shallow, high relief zones, caves, crevices, and areas with sponge gardens (Love et al., 1991; Richards, Cass, Selsby, van der Broek, & Loynachan, 1985). Juvenile yelloweye rockfish reside in waters as shallow as 15 m (49 ft) but generally move deeper as they get older (Love et al., 2002). Juvenile yelloweye rockfish eventually settle in 30 to 40 m (98 to 131 ft) of water near the upper depth range of adults (Yamanaka & Lacko, 2001).

Sub-adult and adult yelloweye rockfish typically utilize habitats with moderate to extreme steepness, complex bathymetry, and rocky substrates (Love et al., 2002) that include caves, crevices, rocky pinnacles, and boulder fields (H. R. Carlson & Straty, 1981; Love et al., 1991; Yoklavich et al., 2000). Within Puget Sound, yelloweye rockfish are associated with areas of high relief rocky and non-rocky substrates such as sand, mud, and other unconsolidated sediments (B. S. Miller & Borton, 1980; Washington, 1977). In waters less than 90 m (295 ft) deep, adult yelloweye rockfish are observed at a mean depth of 45.8 m (150 ft) (S. W. Johnson, Murphy, & Csepp, 2003). Overall, yelloweye rockfish adults are most commonly found from 40 to 250 m (131 to 820) (Love et al., 2002; Orr et al., 2000a, 2000b).

Yelloweye rockfish are generally solitary, demersal residents with small home ranges, though they are infrequently found in aggregations (Coombs, 1979; DeMott, 1983; Love et al., 2002). They are opportunistic feeders, targeting different food sources during different phases of their life history; early life stage rockfish diets include sand lance, gadids, flatfishes, shrimps, crabs, and gastropods (Love et al., 2002; Yamanaka et al., 2006). Due to their large sizes, subadult and adult yelloweye rockfish are able to handle large prey (e.g., smaller rockfish), and they are preyed upon less frequently than earlier life stages (Rosenthal et al., 1982). Yelloweye rockfish predators include salmon and killer whale (J. K. B. Ford et al., 1998; Love et al., 2002). Yelloweye rockfish are among the longest lived rockfish species, living up to at least 118 years (Love, 1996; Love et al., 2002) with natural mortality rates estimated from 2 to 4.6% (Wallace, 2007; Yamanaka & Kronlund, 1997).

Because yelloweye rockfish are long-lived, mature slowly, and have sporadic episodes of successful reproduction, it is likely that the species has inherently low productivity; (Drake et al., 2010; Tolimieri & Levin, 2005). Adult yelloweye rockfish typically occupy relatively small ranges (Love et al., 2002) and may not move to find suitable mates. As the density of mature fish decreases, productivity may also be impacted by density-dependent Allee effects. Moreover, commercial and recreational fishing may have

forced the yelloweye rockfish population below the threshold at which optimal productivity is attainable (Drake et al., 2010).

3.6.4 Current stressors and threats

Current stressors and threats to the PS/GB yelloweye rockfish DPS are the same as those discussed above for the PS/GB bocaccio rockfish DPS.

3.7 Humpback Whale (*Megaptera novaeangliae*), Mexico DPS and Central America DPS

The humpback whale is a large mysticete, reaching lengths of 18 m (60 ft) at maturity ((Winn & Reichley, 1985); cited in (NMFS, 1991)). In the North Pacific, humpback whales feed in coastal waters from California to Alaska and west to Russia.

Humpback whales are listed as endangered under the ESA (35 FR 18319) and protected and depleted under the Marine Mammal Protection Act. The stock structure of humpback whales was defined by the NMFS based on feeding areas because of the species' fidelity to feeding grounds (James V. Carretta et al., 2014). The California, Oregon, and Washington stock occurs within Puget Sound. In 2016, the NMFS finalized a rule recognizing 14 DPSs of humpback whales, listing some DPSs as endangered and some as threatened (81 FR 62259). Two DPSs, the threatened Mexico DPS and the endangered Central America DPS, occur in the action area. The majority of humpback whales that feed off the coast of Oregon and Washington are from the Mexico DPS, whereas only a few individuals from the Central America DPS have been observed in feeding areas in Washington State. Individuals from the Central American DPS tend to feed in areas farther south, offshore of Oregon and California (81 FR 62259).

Humpback whales, like many marine mammals, use acoustic signals to communicate, navigate, locate prey, and sense their environment (Navy, 2008, 2011; NMFS, 1991, 2011a). Humpback whales are known to form small groups that occasionally aggregate for long periods of time in areas of concentrated food (NMFS, 2011a). Worldwide, there are thought to be about 13 stocks that winter in sub-tropical waters in lower latitudes (NMFS, 1991). Observations of humpback whales indicate that they have diurnal dive patterns, with deeper dives occurring during the daytime and shallower dives occurring at night. Humpbacks are generally easy to detect due to their level of surface activity (e.g., feeding with head above water, breaching, fin slapping).

The current population of humpback whales in the North Pacific is approximately 22,000 individuals, of which approximately 1,900 are part of the Washington/Oregon/California stock (J V Carretta et al., 2016). Average abundance estimates for subpopulations of the North Pacific DPS ranges from approximately 200 individuals for southern British Columbia/northern Washington and 1,700 for California/Oregon.

3.7.1 Distribution

Humpback whales occur in all major oceans of the world, but those in the northern hemisphere spend the summer between 40° and 75° latitude, where food productivity is greater than in southern latitudes ((J. H. Johnson & Wolman, 1984; Nemoto, 1957; NMFS, 1991; Tomilin, 1967) all cited in (NMFS, 1991)). Humpback whales are present in waters over continental shelves and along their edges, and around oceanic islands ((Balcomb & Nichols, 1978; NMFS, 1991; Whitehead, 1987) both cited in (NMFS, 1991)).

The California/Oregon/Washington stock winters in coastal waters of Mexico and Central America and migrates to feeding areas from California to southern British Columbia in summer and fall (James V. Carretta et al., 2013). Humpback whale densities tend to be greatest in foraging areas and less so during migration. Feeding grounds tend to be shallow banks or ledges with high sea floor relief ((Hamazaki, 2002; Payne et al., 1990) cited in (Navy, 2011)). The majority of humpback whale feeding occurs during the summer in northern latitudes (NMFS, 1991). Their summer habitats tend to be closer to shore and include major coastal embayments and channels ((Brueggeman, Green, Grotefendt, & Chapman, 1987, 1988; NMFS, 1991) cited in (NMFS, 1991)). The winter distribution also reflects areas of greater prey abundance, which are related to oceanographic factors such as upwelling, converging currents, and other factors characteristic of fjords, channels, continental shelves and their edges, and offshore banks (NMFS, 1991).

In recent years, humpback whales have been sighted with increasing frequency in the inside waters of Washington, including Puget Sound, primarily during the fall and spring. Important feeding areas in the NW include the mouth of the Strait of Juan de Fuca and Heceta Bank in Oregon (Calambokidis et al., 2015). As humpback whale populations along the US West Coast increase, their distribution has expanded into the Strait of Juan de Fuca and Salish Sea (Cascadia Research, 2017).

3.7.2 Presence in the action area

Communication with John Calambokidis, Research Biologist, Cascadia Research on February 9, 2019 indicates that the Humpback would be the most likely whale species to be in the vicinity of the Swinomish action area.

In Washington inland waters, most humpback whale sightings occur in the Strait of Juan de Fuca and in the San Juan Island area. Sightings of humpback whales in Puget Sound vary by location, but historically were infrequent. A small number of humpback whales (based on concurrent sightings of one to four individuals, including a cow/calf pair) were present in Puget Sound from September 2015 to July 2016 (Network, 2016). While, humpback whales may occur in Puget Sound in any season, most of the sightings reported to Orca Network since 2003 were in the main basin of Puget Sound with numerous sightings in the waters between Point No Point and Whidbey Island, Possession Sound, and southern Puget Sound in Colvos Passage. In addition, review of the 2016-Feburary 2019 sighting of whales reported on the Orca Network, (http://www.orcanetwork.org/Main/index.php?categories_file=Sightings), found humpback whales in Skagit bay South of the action area but no sightings within the action area.

3.7.3 Critical habitat

NMFS designated critical habitat for the humpback whale in 2021 (86 FR 21082). The action area does not overlap with critical habitat for the humpback whale.

3.7.4 Life history

Humpback whales follow their prey and are known to have the most diverse feeding behaviors of all the baleen whales, which include bubble netting, herding prey by maneuvering, using the water surface as a barrier, feeding in formation, synchronized feeding lunges, and short- and long-term cooperation between individuals ((C. S. Baker, 1985; C. S. Baker & Herman, 1985; D'Vincent, Nilson, & Hanna, 1985; Hain, Carter, Kraus, Mayo, & Winn, 1982; Hays, Winn, & Petrecig, 1985; Ingebrigtsen, 1929; Jurasz &

Jurasz, 1979; Watkins & Schevill, 1979; Weinrich, 1983; Winn & Reichley, 1985) all cited in (NMFS, 1991)).

Major prey species for humpback whales include small schooling fish and large zooplankton, primarily krill (Klumov, 1963; Krieger & Wing, 1984, 1986; Nemoto, 1957, 1959, 1970; Tomilin, 1967); all cited in (NMFS, 1991)). Fish prey species consist of Pacific herring (*Clupea pallasii*), juvenile walleye (*Sander vitreus*), pollock, capelin, and sand lance ((C. S. Baker et al., 1985; P. J. Bryant, Nichols, Bryant, & Miller, 1981; Dolphin, 1987; Krieger & Wing, 1984, 1986; Perry, Baker, & Herman, 1985); all cited in (NMFS, 1991, 2006)).

Diving is a key aspect of whale behavior and highlights the importance of the deep oceanic environment for humpback whales. North Pacific humpback whale dive times are typically less than 5 minutes but occasionally last up to 10 minutes (Navy, 2011). Most of their prey base is located within 300 m (984 ft) of the surface, and humpback whales spend most of their dive time between 92 and 120 m (302 and 394 ft) deep (NMFS, 2011b), although they are known to dive as deep as 500 m (1,640 ft) (Navy, 2011).

3.7.5 Current stressors and threats

As reported by NMFS (1991), major threats to the humpback whale population include entanglement in fishing gear, ship strikes, and noise disturbance. Changes in oceanic conditions (e.g., food web) related to climate change will also affect the humpback whale. Entanglement in fishing gear is the most frequent human-related cause of injury and death among humpback whales (NMFS, 1991). Netting can be easily broken by a swimming humpback, but lead and anchor ropes are stronger and can cause serious injury. From 2010 to 2014, there were 27 incidents along the US West Coast involving humpback whales and pot/trap fisheries and eight incidents involving gillnet (or other) fisheries, resulting in an annual average (for that time period) of about five whale deaths per year)(J V Carretta et al., 2016).

Ship strikes are an increasing threat to humpback whales, as well as to many other whale species (NMFS, 1991). Humpback feeding grounds are located within major shipping lanes off the US West Coast. Calves and juveniles are more vulnerable to ship strikes because they are smaller, more difficult to see, and spend more time at the surface(Herman, Baker, Forestell, & Antinoja, 1980; Mobley, Bauer, & Herman, 1999) cited in (Navy, 2011). Noise disturbance from ships, aircraft, coastal development, industrial activities, and research can affect humpback whale behaviors such as resting, feeding, nursing, mating, calving, and migrating (NMFS, 1991). Humpback whales generally avoid busy or noisy areas, but some will approach or circle boats, especially fishing and whale-watching boats (NMFS, 1991).

3.8 Southern Resident Killer Whale (*Orcinus orca*) DPS

Killer whales are listed as endangered under the ESA, 70 FR 69903 and critical habitat is described in 71 FR 690. It is 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) (NMFS, 2017b). The DPS are social and are found in familial pods of 20 to 40 individuals led by a dominant matriarch (NMFS, 2017b, 2017c). 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 (NMFS, 2017b). 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). Transient killer whales and offshore killer whales also occur in the area. It is nearly impossible to

distinguish the three types of killer whales (i.e., resident, transient, and offshore killer whales) visually; however, their behaviors are substantially different. 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. Killer whales use several types of calls, whistles, and clicks to communicate or to navigate and hunt (NMFS, 2017c).

Observations of Southern Resident Killer Whale (SRKW) behavior indicate 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. K. B. Ford, 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.

The historical abundance of SRKWs was between 140 and 400 whales (Krahn et al., 2004; Olesiuk, Bigg, & Ellis, 1990). 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 pod, which has 24 members, and then K pod, which only has 19 members (CWR, 2016).

3.8.1 Distribution in the action area

SRKWs are present in the Salish Sea (Puget Sound, Strait of Juan de Fuca, and the Strait of Georgia) from spring to fall each year (NMFS, 2017b). 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) (NWFSC, 2015). 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 (NMFS, 2017b). 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). No reports of Killer whales in the action area were found, probably do to the shallow waters (<20 feet). One exception is a suspected stray juvenile seen in the Swinomish channel January 2 and 3, 2002 (http://www.whaleresearch.com/).

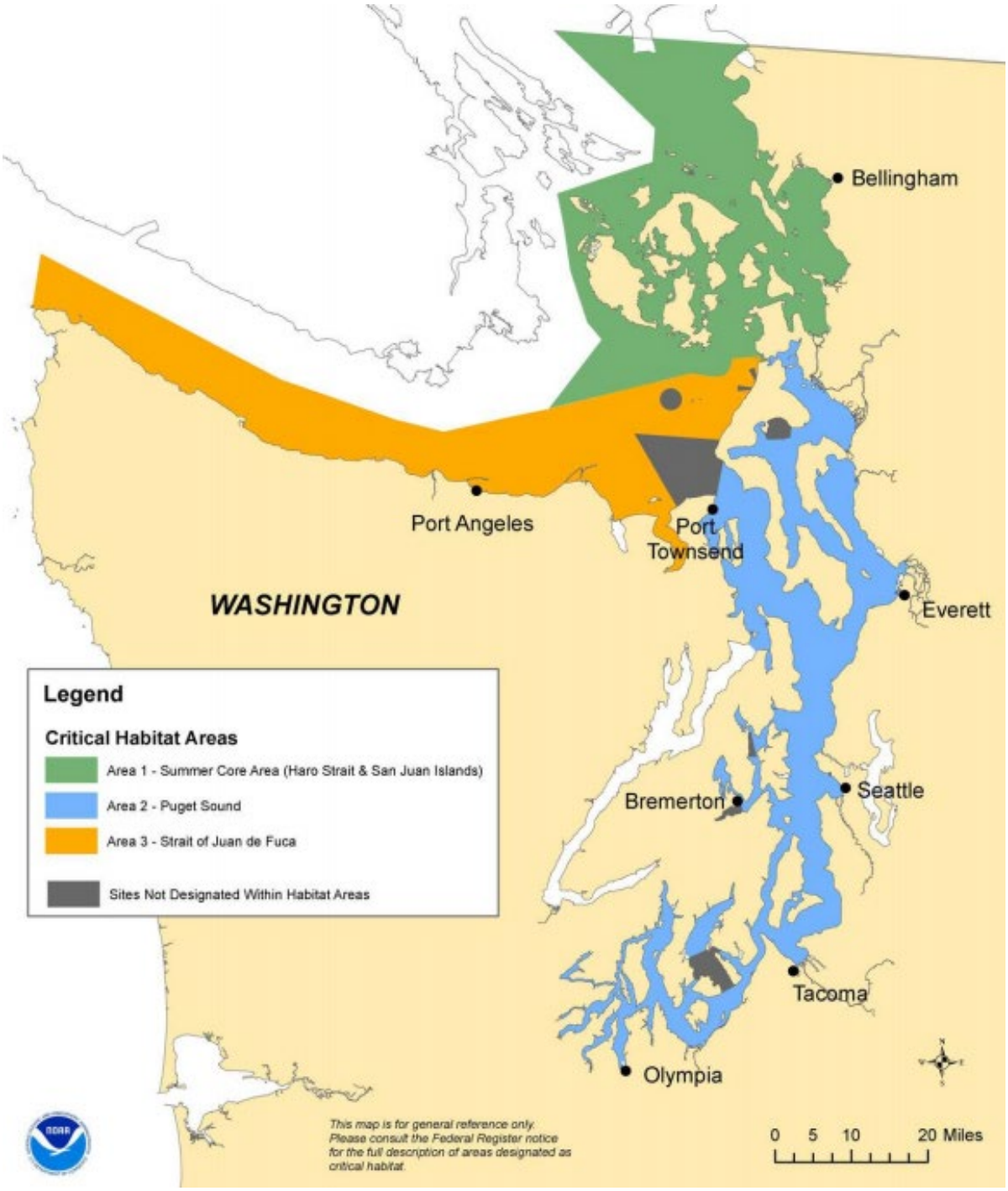


Figure 10: Southern resident killer whale critical habitat (Data source: NOAA 2006)[6]

3.8.2 Critical habitat within the action area

Approximately 6,630 sq km (2,560 square miles) of critical habitat were designated for the SRKW at the end of 2006 (71 FR 69054) (Figure 10). Critical habitat includes all US waters within the Salish Sea,

[6] https://www.westcoast.fisheries.noaa.gov/publications/protected_species/marine_mammals/killer_whales/SRKW-CH-Map.jpg

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 Strait of Juan de Fuca, Haro Strait (between San Juan Island and Vancouver Island), and Georgia Strait (in Canada) are narrow areas that concentrate salmon as they return to inland Washington and British Columbia, Canada waters from the Pacific Ocean.

PBFs for this critical habitat are stated in 71 FR 69054 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.

In 2014, a petition was submitted requesting revisions to SRKW critical habitat to include (in addition to those areas just noted) "Pacific Ocean marine waters along the West Coast of the US that constitute essential foraging and wintering areas…" (80 FR 9682). The petition also requests that the NMFS expand the PBFs for killer whales to include "protective in-water sound levels," which was initially considered as a PBF in 2006 but ultimately was not included (71 FR 69054). On August 2, 2021, NMFS revised the critical habitat designation for SRKW. The final rule maintains the previously designated critical habitat in inland waters of Washington and expands it to include certain coastal waters off Washington, Oregon, and California. The revision adds to critical habitat approximately 15,910 square miles of marine waters between the 6.1-meter and 200-meter depth contours from the U.S.-Canada border to Point Sur, California.

For SRKW, critical habitat includes waters that are at least 20 feet deep at extreme high tide. The 400-foot buffer between the Swinomish Reservation boundary and the action area boundary includes critical habitat for SRKW. Washington State's water quality standards are in effect outside the Swinomish Reservation boundary, including within the buffer area.

3.8.3 Life history

The SRKW is a long-lived species with a late onset of sexual maturity (NMFS, 2008). Mothers and offspring maintain highly stable social bonds throughout their lives, which is the basis for the matrilineal social structure in the SRKW population.

SRKWs live in inland and coastal marine waters, generally 49 to 55 m (160 to 180 ft) deep (Noren & Hauser, 2016). 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 prey are the most important habitat element for SRKWs. SRKWs preferentially consume Chinook salmon, but their diets also include squid and several other fish species (i.e., other salmonids and bottomfish) (J. K. B. Ford et al., 1998). Chinook salmon are the preferred salmonid prey item (J. K. B. Ford et al., 1998).

3.8.4 Current stressors and threats

Key stressors and threats to the SRKW population include human factors such as fishing, boating, water (and prey) quality, and noise pollution (e.g., caused by military activities) (NMFS, 2015b). Water quality in Puget Sound is degraded (L. Johnson et al., 2010; NMFS, 2015b). For example, elevated

concentrations of pollutants in the Salish Sea and elsewhere have been linked to elevated concentrations in salmon and in killer whales (B E Hickie, S, MacDonald, & Ford, 2007; M M Krahn et al., 2007; Krahn et al., 2009; Lachmuth et al., 2010). Once in the environment, many pollutants 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 pollutants 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 (M M Krahn et al., 2007; Krahn et al., 2009).

SRKWs are subject to intensive whale watching and have been since the 1980s. In 2011, the NMFS promulgated regulations to limit effects from vessel traffic around the whales. The rule prohibits vessels from approaching any killer whale closer than 183 m (200 yards) and forbid vessels from intercepting a whale or positioning a vessel in the path of the whales (76 FR 20870). There are exemptions for safety, including vessels actively fishing, cargo vessels traveling in shipping lanes, and government and research vessels, but spill response vessel activities (either during spill response training or otherwise) are not exempt under this provision of the regulation.

Vessel activity has been shown to affect SRKWs and has been identified as one of the key threats to this population. Whale watching and other boating activities have the potential to disturb whales from foraging or resting, although this appears to be less of a stressor than decreased prey resources (Ayres et al., 2012). As noted above, the NMFS promulgated a regulation that restricts vessel approaches to killer whales in the inland waters of Washington. This is in addition to an active campaign at the federal, state, and local levels to educate mariners about SRKWs and encourage responsible boating. The operation of large boats and the use of subsea acoustic equipment (e.g., military sonar) interferes with whale communication and navigation and is likely to interrupt feeding as well, which is facilitated by echolocation (NMFS, 2015b). Human activities can interfere with movements of the whales and impact their passage. In particular, vessels and acoustic disturbance may present obstacles to whale passage, causing the whales to swim further and change direction more often. This increases energy expenditure and impacts foraging behavior (NMFS, 2011b).

3.9 Bull Trout (*Salvelinus confluentus*)

The coterminous US population of bull trout was listed as threatened on November 1, 1999 (64 FR 58910). 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 & F.W., 1997).

3.9.1 Distribution in the action area

Bull trout use estuarine and shoreline habitats near the mouth of the main stem and North Fork Skagit River, including within the action area. This area supports 19 local populations and is considered one of four bull trout strongholds. According to USFWS (2015c) the status of bull trout in this core area is considered strong based upon information available when the species was listed (USFWS, 1998) and still remains strong, possibly with an increasing trend.

The action area is in the vicinity of this core area and within the Puget Sound Coastal Recovery Unit 2. This recovery unit includes marine, estuarine and freshwater habitats outside of natal core areas. The habitat type found within this action area is foraging, migration and overwintering (FMO) habitat, which is important to the anadromous bull trout life forms. The fresh waters located within the action area do not provide suitable habitat for bull trout spawning.

Beamer, Henderson, and Wolf (2004) describe the population numbers and structure of bull trout within the Swinomish Reservation waters, as well as habitat types used by this species. Population size has increased (four-fold since 1995) and the age structure has become more complex. Since 2003 the population consists of sub-adult, first year spawner and older spawner size fish. Bull trout that migrate to estuarine and nearshore habitat are often greater than 150 mm in length. Fish are caught in nearshore water deeper than 1.5 meters and are most abundant in May and June but could be present year round.

The authors also describe four habitat types found within the reservation, and the importance of these habitat types and their use (direct and indirect) by bull trout. These habitat types include: 1) freshwater streams, 2) lagoon or saltmarsh dominated pocket estuaries, 3) shoreline areas in Skagit Bay, and 4) historic deltas within the Swinomish Channel.

There was no evidence of bull trout use of the freshwater streams. Nor are the streams suitable for bull trout spawning. However, the freshwater streams serve an important function in nearshore habitat by supporting bull trout prey. Using the Washington Department of Ecology Coastal Atlas (https://fortress.wa.gov/ecy/coastalatlas/tools/PublicAccess.aspx), it is apparent that these areas are prevalent in the north side of the action area where numerous small stream feed into Padilla Bay from the reservation. A few streams also exist within the Swinomish Channel as evidenced by aquatic bed and emergent wetlands.

Pocket estuaries within the action area are not used by bull trout (Beamer et al., 2004), specifically bull trout are not present in these habitats for rearing, foraging or seeking refuge. However, bull trout utilize spit beaches with lagoon-type pocket estuaries (Figure 11). One area of note within the action area is along Kiket Island, an elongated island adjacent to a fringed salt marsh area. Hayes et al. (2011) documented bull trout use at this location.

The Southern Swinomish Channel is isolated from the Skagit River by a rock jetty and dredged for navigation, creating an “artificial pocket estuary” habitat. Sampling has shown that bull trout use this Swinomish Channel habitat (Beamer et al., 2004).

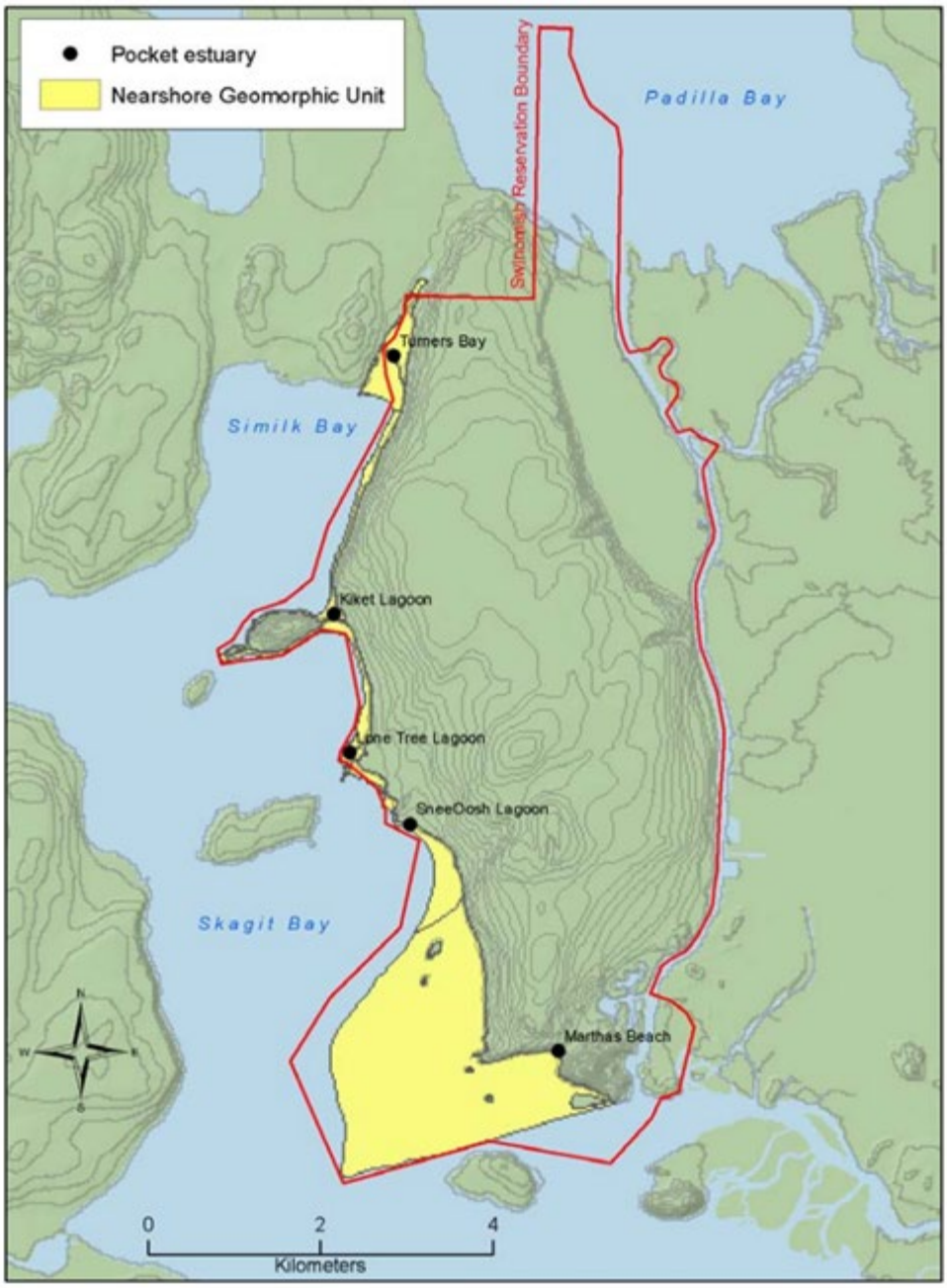


Figure 11: Location of current and historic pocket estuaries and nearshore habitats along the western Fidalgo Island shoreline of Skagit Bay and within the Swinomish Reservation (Beamer et al., 2004)

3.9.2 Critical habitat in the action area

A final ruling on critical habitat for bull trout in the coterminous US was made on October 18, 2010 (effective November 17, 2010) (75 FR 63898). 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 (488,001 acres) of lacustrine habitat.

Critical habitat within the action area is located within the 400-foot buffer between the Tribal waters and the boundary of action area (Figure 12).[7]

The USFWS did not designate waters within the Swinomish Reservation as critical habitat for bull trout under ESA section 4(b)(2), as the Swinomish Tribe has a management plan that addresses surface water resources of the Swinomish Reservation, which include marine tidelands, an artificial marine channel, estuarine wetlands, small streams, and freshwater wetlands (75 FR 63966). In total, 45.2 km (28.1 miles) of stream and shoreline are excluded based on Tribal ownership.

The 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
- Source: 75 FR 63898

[7] https://ecos.fws.gov/ecp0/profile/speciesProfile?spcode=E065

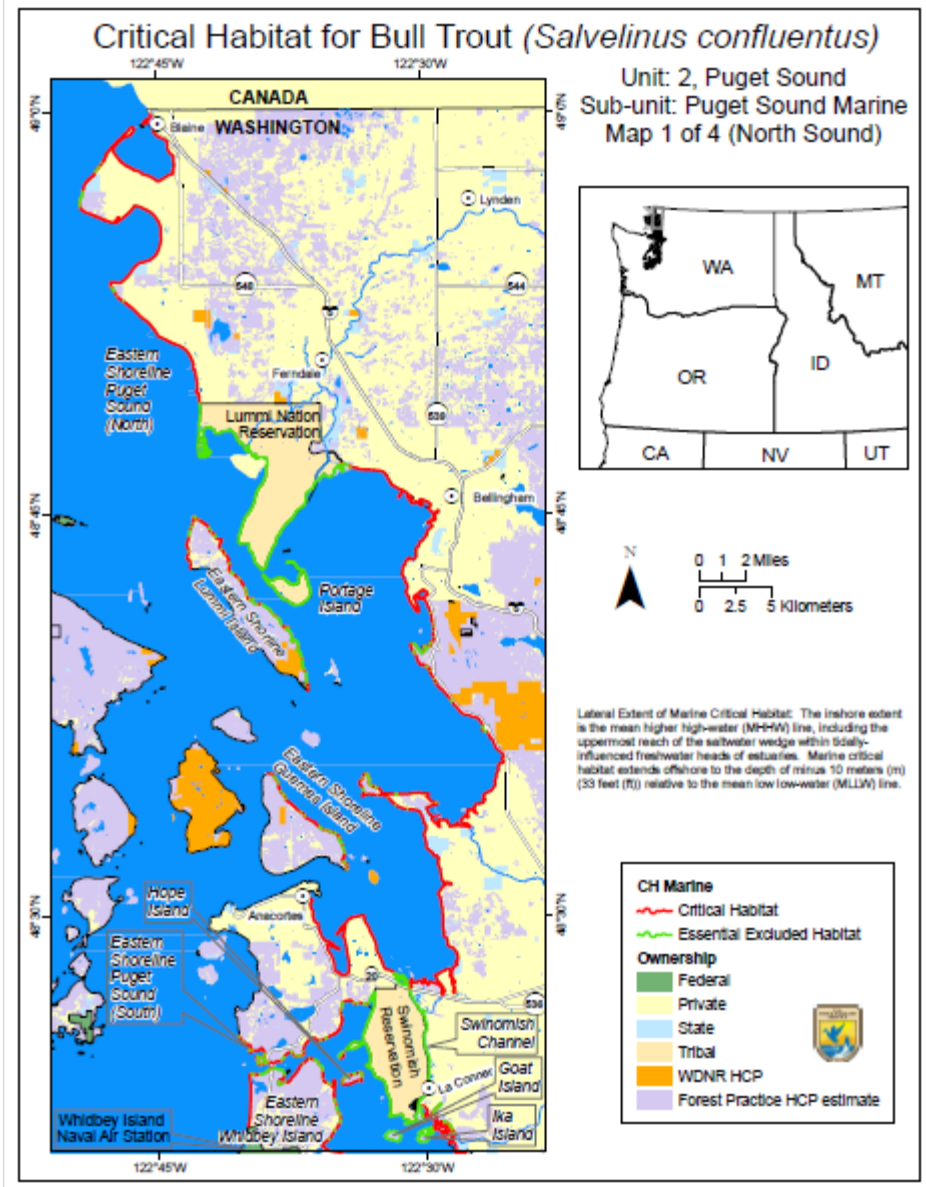


Figure 12: Critical habitat for bull trout within and surrounding the action area[8]

3.9.3 Life history

As described previously bull trout in the action area are anadromous subadults, first year spawners, and older individuals. Bull trout use of the habitat within the action area is for foraging, migration and overwintering. Therefore, this section will focus on the natural history characteristics of bull trout pertinent to its residence in the action area.

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

[8] https://www.fws.gov/pacific/bulltrout/finalcrithab/washington/Unit_02_Puget_Sound_Marine_Map_01_of_04_North.pdf

behavior (Bruce 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; F 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.

Bull trout have more specific habitat requirements than most other salmonids (Bruce 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 (Fraley & Shepard, 1989; F Goetz, 1989; P. Howell & Buchanan, 1992; Pratt, 1992; Rich, 1996; Bruce E. Rieman & McIntyre, 1993, 1995). 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; F Goetz, 1989; Hoelscher & Bjornn, 1989; Pratt, 1992; Rich, 1996; Sexauer & James, 1997; Thomas, 1992).

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; F Goetz, 1989). Subadult and adult migratory bull trout feed on various fish species (L. G. Brown, 1994; Donald & Alger, 1993; Fraley & Shepard, 1989; Leathe & Graham, 1982). In marine nearshore areas of western Washington, bull trout feed on Pacific herring, Pacific sand lance, and surf smelt (F. 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).

3.9.4 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 (64 FR 58910). 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; Bruce E 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.

Of the current threats listed above the proposed action may indirectly affect water quality in the action area, which, while not specifically listed relates to habitat quality. The purpose of approving WQS for the

Swinomish Tribal waters is to protect the beneficial uses of the waters to the benefit of ESA-listed species. There are no other stressors generated by the action related to the threats identified by the Service in the listing document (64 FR 58910). Therefore, the environmental baseline for the action area will focus on water quality resulting from the proposed action.

3.10 Marbled Murrelet (*Brachyramphus marmoratus*)

The marbled murrelet was listed as a threatened species on October 1, 1992, in Washington, Oregon, and northern California (57 FR 45328). The marbled murrelet is a small diving seabird that nests mainly in coniferous forests and forages in nearshore marine habitats. Males and females have sooty-brown upper parts with dark bars and underparts that are light, mottled brown. Winter adults have brownish-gray upper parts and white scapulars (shoulders). The plumage of fledged young is similar to that of adults in winter. Chicks are downy and tan colored with dark speckling (USFWS, 2015b). During a 2013 population abundance survey conducted in Washington, Oregon, and California, the population was estimated to be between 15,400 and 23,900 birds (USFWS, 2016). The 2013 estimate signifies a downward trend in the murrelet population; abundance during the early 1990s in Washington, Oregon, and California was estimated at 18,550 to 32,000 birds (Ralph, Hunt, Raphael, & Piatt, 1995).

3.10.1 Distribution within the action area

Historically, the breeding range of the marbled murrelet extended from Alaska through British Columbia, Washington, Oregon, to northern Monterey Bay in central California. This species winters throughout its breeding range and also occurs in small numbers off southern California (USFWS, 2015b). They can occur year-round in Puget Sound, although their flock size, density, and distribution vary by season (Falxa et al., 2008; Nysewander, Evenson, Murphie, & Cyra, 2005).

At the time of listing, the distribution of active nests in nesting habitat was described as noncontinuous (USFWS, 1997). The at-sea extent of the species currently encompasses an area similar in size to the species' historic distribution.

The Swinomish action area is within the historical range of the marbled murrelet. The Seattle Audubon seabird survey (http://seattleaudubon.org/seabirdsurvey/default.aspx) has not recorded them in the action area, but has in the areas around the action area. It is very likely the birds occasionally visit the action area.

3.10.2 Critical habitat

On May 24, 1996, the USFWS designated critical habitat for the marbled murrelet encompassing approximately 1.6 million ha (4.0 million acres) across Washington, Oregon and California (17 FR 26256). The final rule revising critical habitat for the marbled murrelet was published on October 5, 2011 (76 FR 61599). This rule reduced critical habitat by 76,751 ha (189,656 acres) in Northern California and Oregon based on new data indicating that the areas removed did not meet the definition of critical habitat.

The USFWS revisited critical habitat designation for the marbled murrelet on August 4, 2016 (81 FR 51348) and concluded that the current (2006 and 2011) designations for critical habitat met satisfactory requirements for the species. Currently, there are approximately 1.5 million ha (3.7 million acres) of designated critical habitat in Washington, Oregon and California.

The PBFs considered essential to the conservation of the marbled murrelet are those features critical for supporting suitable nesting habitat for successful reproduction, including:

- Individual trees with potential nesting platforms
- Forested areas within 0.8 km (0.5 miles) of individual trees with potential nesting platforms, and with a canopy height of at least one-half the site-potential tree height.

The action area does not contain critical habitat for marbled murrelet (61 FR 26256, 81 FR 51348).

3.10.3 Life history

Marbled murrelets spend most of their lives in the marine environment where they forage in near-shore areas and consume a diversity of prey species, including small fish and invertebrates. In their terrestrial environment, the presence of platforms (large branches or deformities) used for nesting is the most important characteristic of their nesting habitat. Murrelet habitat use during the breeding season is positively associated with the presence and abundance of mature and old growth forests, large core areas of old growth, low amounts of edge habitat, reduced habitat fragmentation, proximity to the marine environment, and forests that are increasing in stand age and height (USFWS, 2015b).

Nest stands are typically composed of low elevation conifer species. In California, nest sites have been located in stands containing old growth redwood (Sequoia sempervirens) and Douglas-fir, while nests in Oregon and Washington have been located in stands dominated by Douglas-fir, western hemlock, and Sitka spruce (USFWS, 2015b).

Nelson (1997) identified that there may be a general opportunistic shift from exposed outer coasts into more protected waters during the winter; for example, many marbled murrelets breeding on the exposed outer coast of Vancouver Island appear to congregate in the more sheltered waters within the Puget Sound and the Strait of Georgia in fall and winter (Burger, 1995).

Murrelets are usually found within 8 km (5 miles) of shore, and in water less than 60 m (197 ft) deep (Ainley, Allen, & Spear, 1995; Burger, 1995; S. K. Nelson, 1997). In general, this species occurs closer to shore in exposed coastal areas and farther from shorelines in protected coastal areas (S. K. Nelson, 1997). Courtship, foraging, loafing, molting, and preening occur in marine waters.

Marbled murrelets are wing-propelled pursuit divers that forage both during the day and at night (H. R. Carter & Sealy, 1986; Kuletz, 2005). This species can make substantial changes in foraging sites within the breeding season, but many individuals routinely forage in the same general areas at productive foraging sites, as evidenced by repeated use over a period of time throughout the breeding season (H. R. Carter & Sealy, 1986; Hull et al., 2001; Mason, Burger, & Hansen, 2002). Murrelets are also known to forage in freshwater lakes (S. K. Nelson, 1997). Activity patterns and foraging locations are influenced by biological and physical processes that concentrate prey, such as weather, climate, time of day, season, light intensity, upwellings, tidal rips, narrow passages, hallow banks, and kelp beds (Ainley et al., 1995; Burger, 1995; S. K. Nelson, 1997).

Throughout their range, marbled murrelets are opportunistic feeders and utilize prey of diverse sizes and species. They feed primarily on fish and invertebrates in marine waters, although they have also been detected on rivers and inland lakes (50 CFR 17) (H. R. Carter & Sealy, 1986). In general, small schooling fish and large pelagic crustaceans are the species' main prey items. Pacific sand lance

(Ammodytes hexapterus), northern anchovy (Engraulis mordax), immature Pacific herring (Clupea harengus), capelin (Mallotus villosus), Pacific sardine (Sardinops sagax), juvenile rockfishes (*Sebastas* spp.), and surf smelt (*Osmeridae*) are the most common fish species taken. Squid (*Loligo* spp.), euphausiids, mysid shrimp, and large pelagic amphipods are the main invertebrate prey. Marbled murrelets are able to shift their diet in response to prey availability (Becker, Peery, & Beissinger, 2007). Long-term adjustment to less energy-rich prey resources appears to be partly responsible for poor reproduction in California (USFWS, 2015b). Prey species in Washington coastal and inland waters have not been well documented, but include sand lance, anchovy, immature Pacific herring, shiner perch, and small crustaceans (especially euphausiids) (Burkett, 1995). Invertebrates are a primary prey source in the non-breeding season, whereas fish are a source year-round.

Breeding adults exercise more specific foraging strategies when feeding chicks, usually carrying a single, energy-rich fish to their chicks (Burkett, 1995; S. K. Nelson, 1997). Freshwater prey is important to some individuals during several weeks in summer and may facilitate more frequent chick feedings (Hobson, 1990). Nesting marbled murrelets that are returning to their nest at least once per day must balance the energetic costs of foraging trips; this may result in their preferring to forage in marine areas in close proximity to their nesting habitat. However, if adequate or appropriate foraging resources are unavailable in close proximity to their nesting areas, the species may be forced to forage at greater distances or abandon their nests (Huff, Raphael, Miller, Nelson, & Baldwin, 2006). As a result, the distribution and abundance of prey suitable for feeding chicks may greatly influence the overall foraging behavior and location during the nesting season and may affect reproductive success (Becker et al., 2007). It may also significantly affect the energy demand on adults by influencing both the foraging time and number of trips required (Kuletz, 2005). In the Puget Sound region, their nesting season is asynchronous between April 1 and September 23.

3.10.4 Current stressors and threats

Several anthropogenic threats were identified as having caused the dramatic decline in the species when the marbled murrelet was listed under the ESA (57 FR 45328) and in the Recovery Plan (USFWS, 1997). These threats include habitat destruction and modification in the terrestrial environment from timber harvest and human development, which caused a severe reduction in the amount of nesting habitat, unnaturally high levels of predation resulting from forest edge effects, the existing regulatory mechanisms, inadequate regulatory mechanisms, and human-caused factors such as mortality from oil spills and entanglement in fishing nets used in gill-net fisheries.

Human factors that affect the continued existence of the species include derelict fishing gear leading to mortality from entanglement, energy development projects (wave, tidal, and on-shore wind energy projects) leading to mortality, and disturbance in the marine environment (from exposures to lethal and sublethal levels of high underwater sound pressures caused by pile-driving, underwater detonations, and potential disturbance from high vessel traffic) (USFWS, 2009).

There have been changes in the levels of these threats since the 1992 listing (USFWS, 2004, 2009). The regulatory mechanisms implemented since 1992 that affect land management in Washington, Oregon, and California, and new gill-netting regulations in northern California and Washington, have reduced some threats to the marbled murrelet (USFWS, 2004). The levels for the other threats identified in the 1992 listing (57 FR 45328), including the loss of nesting habitat, predation rates, and mortality risks from

oil spills and gill net fisheries, have remained unchanged. However, new threats have been identified (USFWS, 2009). These new stressors are due to several environmental factors affecting marbled murrelets in the marine environment, including habitat destruction and modification, or curtailment of marine environmental conditions necessary to support the species due to elevated levels of PCBs in prey species; changes in prey abundance and availability; changes in prey quality; harmful algal blooms that produce biotoxins leading to mortality; and climate change in the Pacific Northwest.

Climate change is expected to further exacerbate some existing threats such as the projected potential for increased habitat loss from drought-related fire, mortality, insects and disease, and increases in extreme flooding, landslides and windthrow events in the short term (10 to 30 years) (USFWS, 2009).

4 Environmental Baseline

4.1 Status of the Environment

4.1.1 Introduction

The regulations implementing section 7(a)(2) of the ESA define the environmental baseline as "the past and present impacts of all Federal, State or private actions and other human activities in an action area, the anticipated impacts of all proposed Federal projects in an action area that have already undergone formal or early section 7 consultation and the impact of State or private actions that are contemporaneous with the consultation in process" (50 CFR §402.02).

Federal actions which have undergone section 7(a)(2) consultation in the last 10 years include formal and informal consultations for boat ramp installations and repair, bank stabilization, levee repair, culvert removal and replacements and dredging. Not all these actions were conducted in the action area, but nonetheless may have affected the environmental baseline. These actions have both improved (culvert removal and replacement) and continued to reduce habitat quality (levee repair and bank stabilization) in the action area. The EPA action subject to this consultation is the proposal to approve the Swinomish Tribe's water quality standards for protection of aquatic life, an action that is intended to improve and protect water quality within the Tribal Waters.

Various federal and state organizations have and continue to fund restoration projects within the action area. To date, there are 2 active and 4 completed restoration projects in the action area. The organization overseeing these projects is the Skagit Watershed Council (SWC). Those projects contemporaneous with the consultation in process includes the McGlinn Island Causeway, and the Skagit Basin Riparian Restoration 2a. We also discuss past (since 2009) restoration projects as they have influenced the environmental baseline by offsetting some of the impacts from land use. These projects will be discussed in the sections describing the specific waterbodies in which they were implemented in the action area (i.e. Padilla Bay, Swinomish Channel, Skagit Bay and Similk Bay).

EPA or the state of Washington permit the five active NPDES discharges that are located within or near the action area (Figure 13 and Table 15). Additionally, there are eight formerly or currently contaminated sites. EPA manages three of these sites and Ecology manages the other five. Two of these sites are in the action area, the Swinomish Tribal Economic Zone 1 and March Point Landfill. These sites are managed by EPA and Ecology, respectively. The discharges that are contemporaneous with the action are discussed in addition to sites that have historically contributed to the condition of the environmental baseline. These sites and discharges are discussed in more detail in section 4.2.

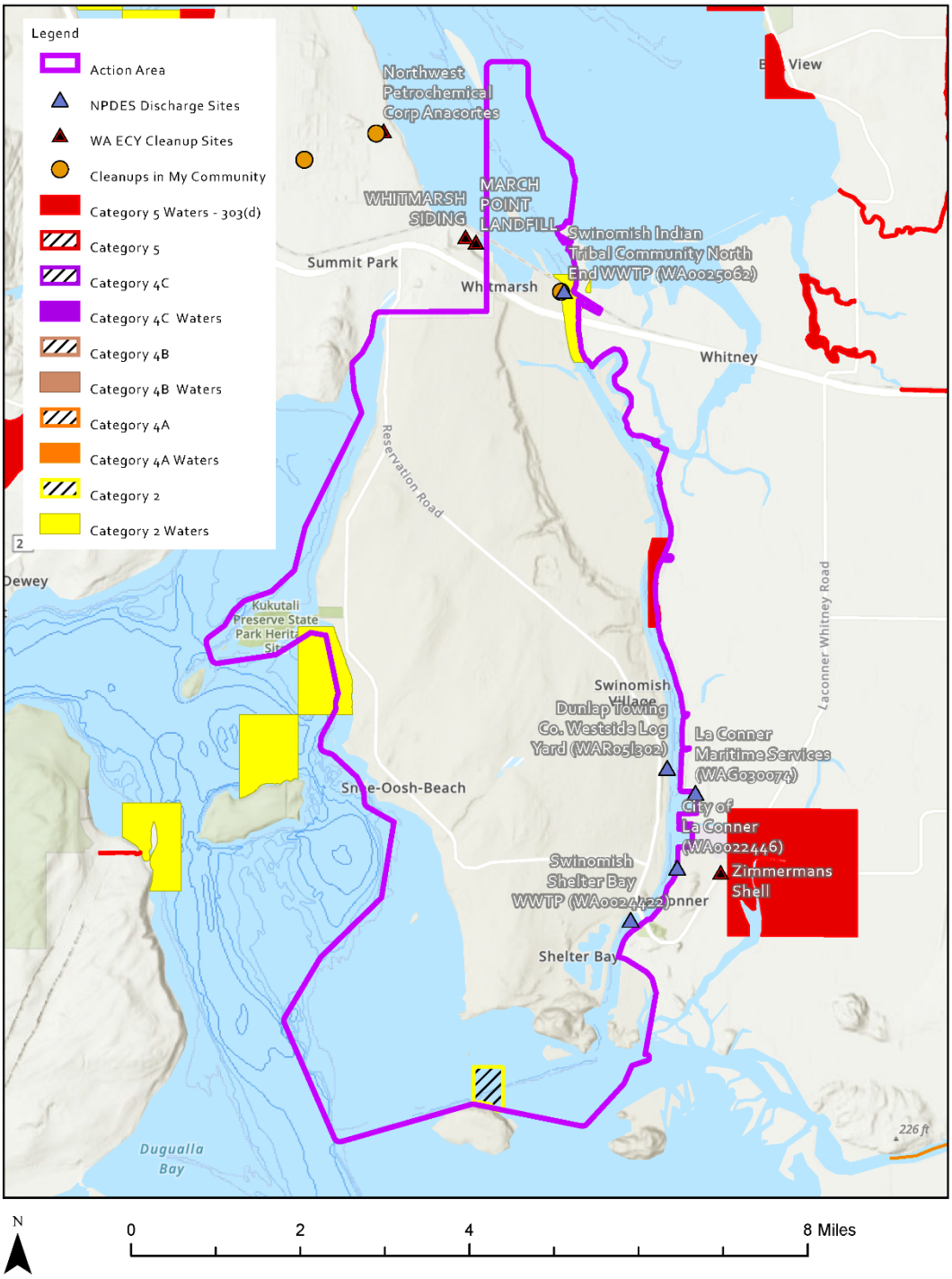


Figure 13: NPDES permitted discharges, cleanup sites, and water quality listings near the action area

Additionally, since the action under consultation is related to water quality criteria, we have included a comprehensive compilation of water, sediment and tissue data collected in the action area for the chemicals with criteria that are being considered for approval in this consultation (section 4.3).

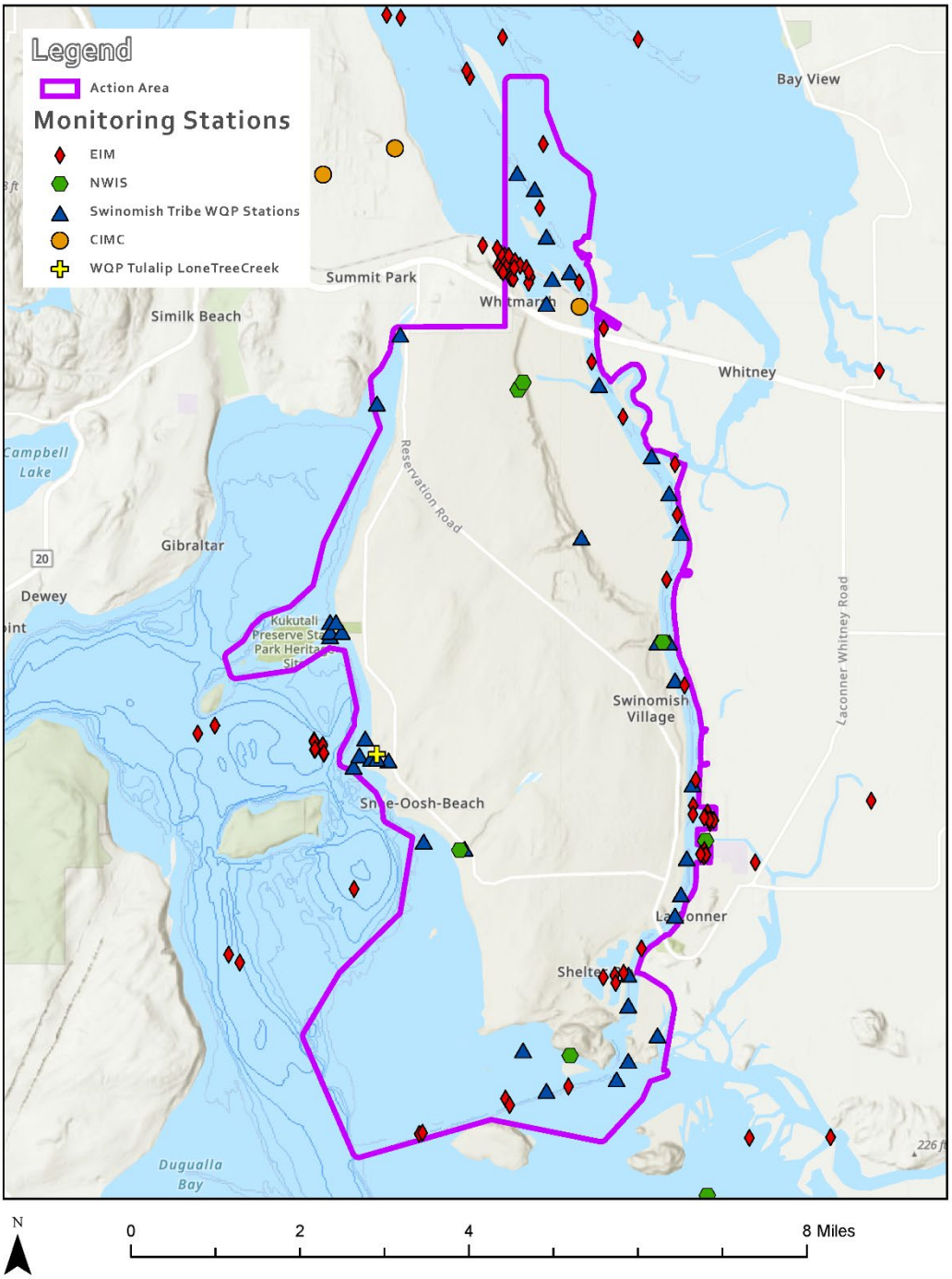


Figure 14: Monitoring stations used to characterize the environmental baseline

Smith (2005) and Odden (2014) reviewed two different aspects of the habitat. The purpose of the Conservation Value Analysis (CVA) was to identify areas of high conservation value and potential for conservation (based on land use and ownership), while the report by Smith (2005) was intended to rate various categories of habitat conditions based upon a set of standards. Both studies provide useful information for considering the condition of the environmental baseline to determine the extent to

which environmental stressors affect salmonids excluding the proposed action. This characterization of the baseline is necessary to determine the overall effects of the action on ESA-listed fish according to 50 CFR §402.02.

The action area encompasses portions of the Padilla Bay, Swinomish Channel, Skagit Delta fed by the North Fork Skagit River, and east Similk Bay. Section 4.1 is organized according to these areas loosely followed as the conditions identified by Smith (2005) that affect the function and value of the habitat for the listed species under consultation. The discussion of the environmental baseline is organized by waterbody and discussed: 1) modification to the hydrologic conditions, 2) habitat condition, and 3) shoreline modification.

4.1.2 Data and information sources

The action area for this consultation consists of the reservation waters of the Swinomish Tribe with a 400 ft buffer waterward of this Tribal boundary (see section 2.1). The marine portion of the action area primarily consists of shallow intertidal zones where depths range from 0 to 10 ft with some localized or channelized areas of deeper water, primarily in the dredge channel for Swinomish Bay and Similk Bay (Figure 15).

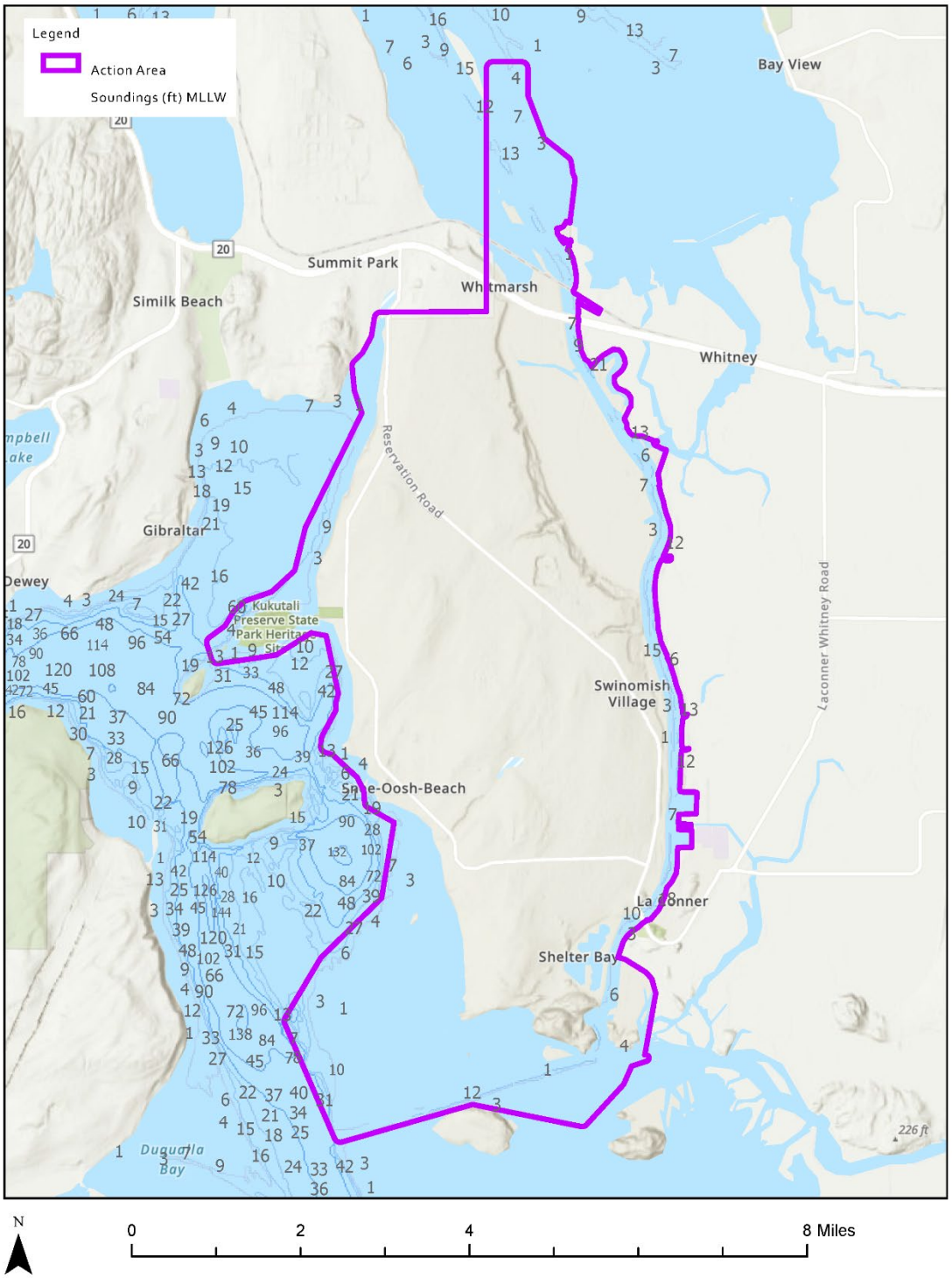


Figure 15: Bathymetry near the action area

The action area is within Ecology's Water Resource Inventory Area (WRIA) 3. EPA used the following as primary sources of information to characterize the environmental baseline in the action area:

- NOAA Environmental Response Management Application (https://response.restoration.noaa.gov/pacific-northwest-erma);
- Ecology's Coastal Atlas (https://fortress.wa.gov/ecy/coastalatlas/tools/map.aspx);
- EPA's EnviroAtlas (https://www.epa.gov/enviroatlas), and
- Salmon and Steelhead Habitat Limiting Factors Water Resource Inventory Areas 3 and 4, the Skagit and Samish Basins (C. J. Smith, 2005).
- Marine Shoreline Protection Assessment for Skagit County (Odden, 2014).

These documents and others, along with mapping applications were used to obtain information on development, land use and environmental conditions (i.e. shoreline modification) in the action area. In addition, we compiled surface water, sediment and tissue data to characterize the level of contamination in these areas (section 4.3). The mapping applications are useful in that they provide recent data layers that can be overlaid on the action area to depict the "current" state of the biotic and abiotic conditions germane to the environmental baseline.

EPA used Smith (2005) to describe the condition of the environmental baseline in the action area as it relates to salmon conservation and recovery. The Skagit Delta is principally used for farming and Smith (2005) states that many of the estuarine areas are near lands used for agriculture. With respect to the action area these agricultural areas are concentrated in the south Padilla Bay deltas, the Skagit delta, and along parts of the Swinomish Channel (Skagit County Planning and Permits 1997; cited in Smith (2005). The limiting factors report is somewhat dated, but nonetheless provides a comprehensive synopsis of the conditions important to supporting functional salmon populations in WRIA 3 and 4. Table 10 lists the factors affecting the quality of aquatic habitat and the resulting conditions of each Region included in the action area. Additional details about specific waterbodies within the action area are discussed in the pertinent sections below.

Table 10. Summary of estuarine and nearshore conditions in WRIA 3 associated with the action area from Smith (2005)

Region	Hydro-modifications	Water Quality/Sediment Contamination	Wetland/ Habitat Loss	Boat Ramps, Slips, Piers	Riparian and Instream Habitat
ESTUARIES					
Skagit Delta/Estuary	Poor	Poor in SF Skagit	Poor	Mostly Good (some in NF)	Poor
Skagit Bay Sloughs	Poor	Poor	Poor	NA	Poor
Padilla Bay Sloughs	Poor	Poor	Likely Poor	NA	Poor
NEARSHORE AREAS					
Skagit Bay East	Poor	Good	Poor	Good	NA
Skagit Bay West	Good	Poor	Good	Good	NA
Swinomish Channel	Poor	Poor (DG)	Poor	Poor	NA

DG= Data Gap; When a DG accompanies a rating of good, fair, or poor, it means that the rating is provisional and additional assessments are needed.

The Odden (2014) marine shoreline protection assessment was used to help characterize the condition of the habitat in the action area. This assessment used a CVA based on two primary conservation targets developed considering the 10 shoreline process attributes and 15 nearshore habitat and species attributes. The assessment was developed using a spatial analysis of Skagit County marine shorelines conducted to identify marine shoreline reaches with both high conservation value and land protection potential.

This spatial analysis for the CVA was conducted by the Skagit Land Trust (SLT) and the Skagit County Marine Resources Committee. It was intended to identify reaches that are central to maintaining or improving marine shoreline ecosystem processes. Important shoreline values, particularly those that enhance fish and wildlife habitat, include eroding bluffs or sediment source beaches, eelgrass beds, forage fish spawning beaches, and estuaries. The two primary conservation targets for protection of marine shorelines are "shorelines critical for ecosystem processes (Process) and important nearshore habitats and species (Habitat)." (Odden, 2014) used data sources from the State of Washington including the Department of Natural Resources, Department of Fish and Wildlife, Department of Ecology and the Puget Sound Nearshore Ecosystem Restoration Project to develop this assessment.

The Skagit Land Trust added a 400 ft buffer on the shorelines to capture the conservation values within the nearshore. This buffer includes only a portion of the action area as EPA added a 400 ft buffer at the edge of the Swinomish Tribal waters, and we are uncertain as to the extent of the overlap of these two buffers (shoreline vs Tribal waters) within the action area. However, the nearshore and associated habitats contain critical values for listed salmonids and for prey species for salmonids and other listed species considered in this consultation, and the conservation values attributed to them helps to inform the condition of the environmental baseline in the action area.

Odden (2014) developed the CVA for all 114 reaches in Skagit County to determine which reaches had the most conservation value and whether protection was feasible, so that it could be made a priority. Only a portion of these 114 reaches are in the action area and those will be discussed in the pertinent sections below.

According to Odden (2014) the conservation target ranking was quantified in the following manner:

Each conservation target ranking, both Process and Habitat, was classified based on 4 quantiles, or groups. Those reaches that rank in the highest 25% are in the top tier for that target. Those that rank in between 50 and 75% are in the second tier. In order for a reach to be considered highly ranked in the CVA, it needed to rank in the top tier for either one or both of the targets or be in the second tier for both targets.

Of the 114 reaches evaluated, 53 ranked as high in the CVA based on the condition of the two conservation targets (Process and Habitat) considered. We discuss the reaches present in Padilla Bay, Swinomish Channel, Samish Bay and Similk Bay that are within the action area and where they ranked in the CVA in the following sections.

4.1.3 Padilla Bay

Land use in the Padilla Bay watershed is primarily agriculture. Agricultural lands are bordered by diked sloughs that deliver freshwater to the bay (Figure 16). Smith (2005) evaluated the condition of the lower

Skagit Sub-Basin and includes two small islands and a portion of the dredged channel, mud flats, and nearshore marine waters. And while the sloughs themselves are not within the action area, they exhibit a significant influence on water quality in southern Padilla Bay, including within the action area, and access to freshwater refugia for salmonids.

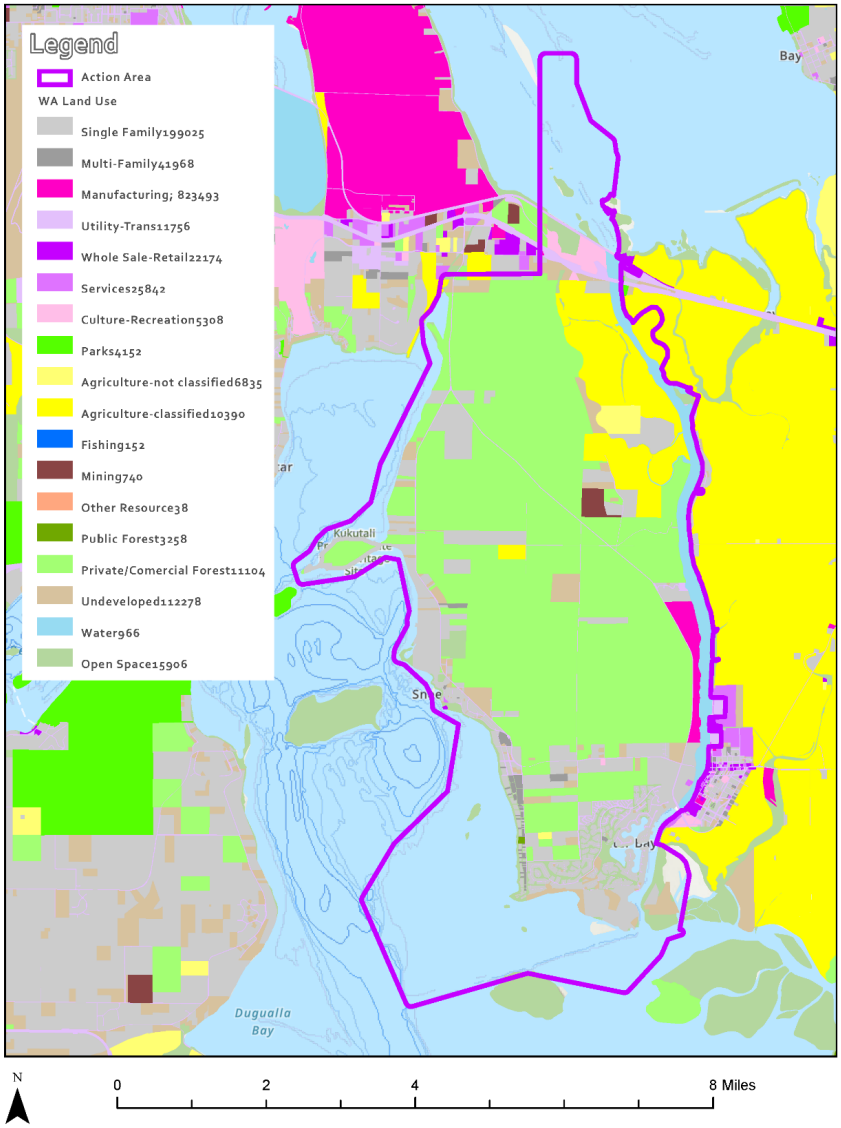


Figure 16: Land uses near the action area

4.1.3.1 Hydrologic modifications

Padilla Bay was isolated from the Skagit River in the early 1800's and 1900's when diking and draining was initiated for the development of farmland. Currently, most of the water flowing to Padilla Bay is through four major sloughs that have been dammed, diked and fitted with tide gates to support agriculture (Bulthuis, 2010). These sloughs include Joe Leary slough which drains approximately 4,700 ha (11,600 ac) as far as 30 miles east of the Bay; Big Indian Slough which drains 2,025 ha (5,000 ac) of light industrial and residential areas before flowing through agricultural lands to reach the Bay; No Name

Slough which drains 990 ha (2,450 ac) through low intensity agricultural and rural housing; and finally Little Indian Slough which drains 220 ha (550 ac) through a small industrial area and Bay View Ridge before flowing into the Bay. The water flow for all these sloughs is controlled by tide gates which eliminates or reduces access to important salmon spawning and rearing habitat, and refugia during high flow events.

4.1.3.2 Habitat conditions

Historically, the intertidal marsh area of Padilla Bay was much more extensive until reduced by diking and drainage of marshes over the last 150 years (B. Collins, 2000; B. Collins & Sheikh, 2005; Thom & Hallum, 1990)); all as cited in Bulthuis (2010)). What remains of these marsh areas are narrow bands located seaward of dikes and along the fringes of the lagoons and the dredge spoil islands along the Swinomish Channel (Bulthuis, 2010). According to the rating in Smith (2005), Padilla Bay is classified as poor for wetland loss, this means 30 percent or greater loss of this habitat. Smith (2005) asserts that of the 454 wetlands identified in the Padilla Bay watershed, most no longer have contact with streams that "either provide or directly connect to salmonid habitat." Connection to wetland habitat is part of the floodplain features and provides rearing and refuge habitat, as well as contributes to water storage and recharge, macroinvertebrate production (food), and sediment storage.

Acknowledging the historic and continued loss of wetland and marsh habitat, Odden (2014) identified reaches and islands within Padilla Bay (and the action area) of high conservation value. These areas ranked high in the CVA for both habitat and process, the two characteristics considered in the analysis. Most are outside of the action area except for two small islands, one just north and east of the dredge channel leading to the Swinomish Channel and the other to the north and west of this island. These areas were ranked high with one criterion in the top tier and the other criterion in the second tier. Although, these islands are in Padilla Bay they are included in the Islands unit of the CVA for reporting purposes in Odden (2014).

Padilla Bay has extensive intertidal Eelgrass beds, including within the action area, which may have benefited from the diversion of freshwater from the bay (Figure 17). The Eelgrass beds in Padilla Bay include *Z. japonica* and *Z. marina* these species are found at depths between 5.25 and 16.7 ft MLLW. The long-term trend has indicated an increase in plant cover in this location (WDNR 2019).[9]

[9] https://www.dnr.wa.gov/programs-and-services/aquatics/aquatic-science/puget-sound-eelgrass-monitoring-data-viewer

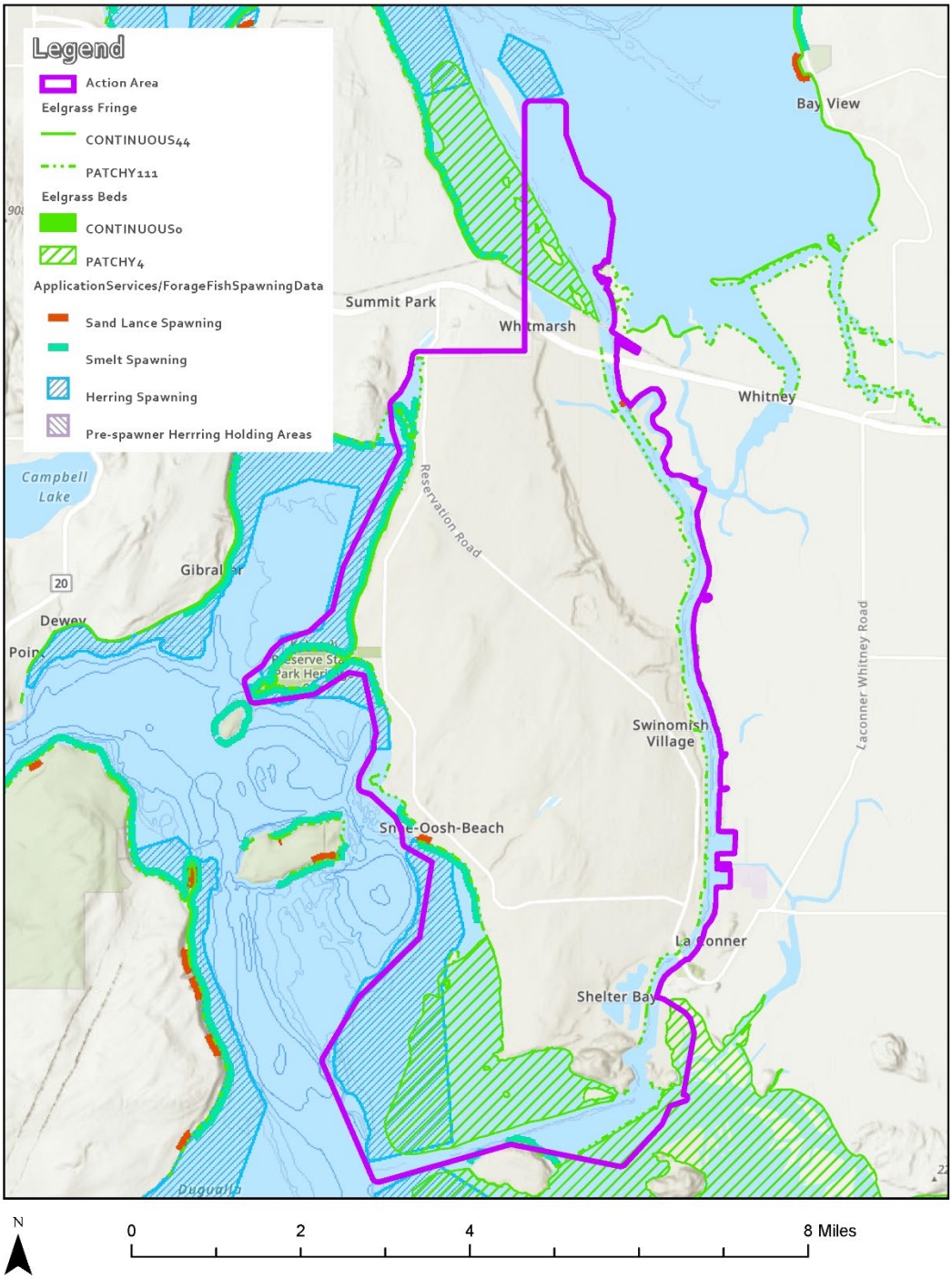


Figure 17: Eelgrass beds near the action area

Water quality in Padilla Bay is rated tentatively good, even with elevated temperature (up to 23 ^{o}C) and low dissolved oxygen (< 6 mg/L) in summer. It is assumed that the elevated water temperature and low DO result from natural conditions due to the shallow nature of the bay (Bulthuis 1993 cited in (C. J. Smith, 2005)).

Two water quality impacts that are not considered natural include sediment toxicity and the potential for eutrophication. Contaminated sediments are rated as poor due to elevated phenols and phthalates associated with locations which failed toxicity tests (C. J. Smith, 2005). Agricultural activities in the basin that lead to excess nutrients, which are then carried through the sloughs contribute to or cause eutrophication.

4.1.3.3 Shoreline modifications

Shoreline modification has a significant influence on the estuarine system, it interrupts the hydrology of riverine and tidal processes that create and sustain delta and nearshore habitats; the impacts include, elimination or reduction of beach nourishment required for forage fish spawning habitat; disruption of longshore sediment recruitment and deposition; and, changes in the beach substrate over time. These impacts affect the macroinvertebrate communities and the upper trophic level species that depend on them. These structures also affect water depth and their presence increases susceptibility of juvenile salmonids to predation, as juveniles require shallow water for avoidance of predators and for feeding and migrating. Finally, the presence of these structures affects the intertidal eelgrass and macroalgae critical to supporting a prey base for a complex aquatic community (Figure 17).

According to the Washington state coastal atlas[10] between 81 and 100 percent of the shoreline in Padilla Bay portion of the action area has been modified (Figure 18). The shorelines on the small dredge spoil islands at the mouth of the Swinomish Channel range from 0 percent to 10 percent modified. According to Ecology, “the percentage of human-caused shoreline modification” include bulkheads in place as of the year 2000. The action area also includes the Swinomish Channel Boat Launch and the shoreline in this area is 100 percent modified according to the Washington state coastal atlas. There are no other piers or ramps in the action area portion of Padilla Bay.

[10] https://fortress.wa.gov/ecy/coastalatlas/tools/map.aspx

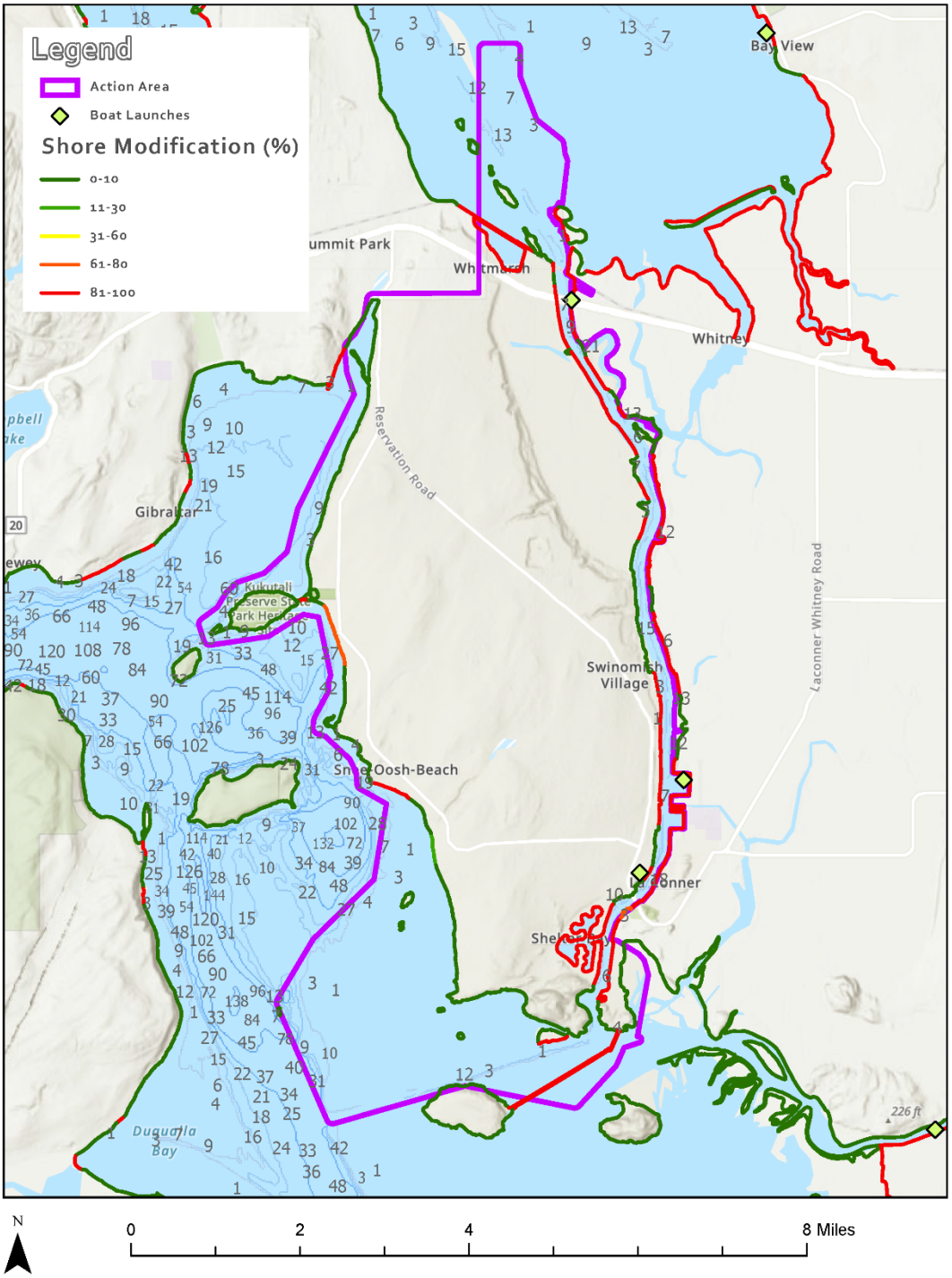


Figure 18: Shoreline modifications near the action area

4.1.3.4 Summary

The hydrodynamics of Padilla Bay have been substantially modified since the last century and most of the water flowing to the bay is through four major sloughs that have been dammed, diked and fitted

with tide gates to support agriculture (Bulthuis, 2010). The expanse of wetlands and marshes has been significantly reduced eliminating off channel and refugia habitat for fish. Nonetheless, eelgrass beds are extensive with an increasing trend providing foraging and rearing habitat, and a source of prey for listed species and others critical to the aquatic community. Much of the shoreline has been modified reducing the quality of the nearshore habitat. Overall, the quality of the environmental baseline has been reduced due to human actions including light industrial and residential development and agriculture.

4.1.4 Swinomish Channel

The Swinomish Channel is an 11-mile channel bordered by agricultural lands and low-density residential development in addition to the Swinomish Tribal Reservation and the city of La Conner. It is an active navigation channel, which is regularly dredged by the Army Corps of Engineers. The most recent dredging operation was in October and November of 2018 when 154,000 cubic yards of material was dredged from the channel and disposed of at a regulated dredge disposal location.

Two restoration projects have been completed under the oversight of the SWC. These include the Swinomish Channel Fill Removal and Restoration[11] and the Smokehouse Floodplain Restoration Phase 1.[12]

The Skagit River System Cooperative sponsored and partnered with the Swinomish Tribal Community to conduct the Swinomish Channel Fill Removal and Restoration project, which was completed in 2012. The description of the project is as follows:

The Swinomish Channel is a salt-water channel which connects Skagit Bay, to the south, and Padilla Bay, to the north, separating Fidalgo Island from mainland Skagit County. The Swinomish Channel Fill Removal and Restoration Project is strategically located along a salmonid migratory corridor (the Swinomish Channel) connecting the natal Skagit River with extensive juvenile salmon rearing habitat of Padilla Bay eelgrass. Rearing and refuge habitat along the Swinomish Channel, which would facilitate juvenile Chinook and other salmon migration between Skagit and Padilla Bays, is severely reduced compared to historical conditions, primarily because of dredging and straightening of the channel for navigation purposes that began in the early 1900's. The type of habitat restored has been identified as a critical limiting factor in the Skagit Chinook Recovery Plan (SRSC & WDFW, 2005).

This successful project resulted in creating of 4 ha (10 ac) of new estuary while removing and controlling invasive plants and removal of 0.07 miles of shoreline armoring.

The Smokehouse Floodplain Restoration Phase I project was sponsored by the Swinomish Indian Tribe, conducted on Washington Department of Fish and Wildlife Property and a cooperative effort between State and private partners. The project was completed in 2009 and is described below:

> The site was once an expansive estuarine emergent marsh over 900 acres in size (B. D. Collins & Sheikh, 2002). Hydraulic modifications including installation of flap-style tide gates converted this emergent marsh to arable uplands. The modern site still contains a significant network of remnant slough channels, albeit simplified by decades of agriculture. These remnant channels

[11] https://srp.rco.wa.gov/project/280/11869; https://secure.rco.wa.gov/prism/search/projectsnapshot.aspx?ProjectNumber=07-1827. Accessed 5/17/2022.
[12] https://srp.rco.wa.gov/project/280/29757. Accessed 5/17/2022.

are presently influenced by small freshwater tributary streams and seeps but isolated from tidal influence.

This project also replaced existing impassible tide gates with self-regulating tidegates (SRTs). This project was funded through the Salmon Recovery Funding Board (SRFB) 04-1226 and 02-1563. Tide gate replacement should restore tidal influence to the channels, enable fish passage, and increase the amount of available blind channel, distributary, and tributary habitat for all salmonid species. Allowing a wide range of tidal influence to interact with the remnant channels' freshwater flows on the floodplain will create estuary-type freshwater and salt water mixing zones. These mixing zones are critical rearing habitat for juvenile salmonids.

The intent of the project was to restore natural functions and processes within the area including restoration of tidal marsh, blind sloughs and distributary channels to provide critical rearing habitat for Chinook and other salmonids. Implementation of this restoration project resulted in the creation of 25.3 ha (62.4 ac) of estuary. Additionally, fish passage was improved through the removing 10 blockages impeding salmonid migration; 6 ha (15 ac) of riparian area and 9.6 km (6 mi) of streambank were improved.

4.1.4.1 Hydrologic modification

The Swinomish Channel connects Padilla Bay and Skagit Bay, it did so historically as a slough, and now in its current configuration as dredged navigation channel. Prior to dredging, it was a collection of shallow tidal sloughs and marshes that provided feeding and rearing habitat for salmonids. According to Bulthuis (2010) a robust salmon fishery flourished in the slough until 1937 when a rock jetty was built and diverted fresh water flows from the North Fork of the Skagit River and Padilla Bay. The jetty and other landscape alterations diverted the flow and likely juvenile salmon away from the channel and Padilla Bay.

4.1.4.2 Habitat conditions

Habitat conditions in the channel were rated "poor" by Smith (2005), primarily because of the extensive shoreline modification resulting in a loss of riparian habitat (Figure 18). A majority of the Swinomish Channel has less than 10 percent overhanging riparian vegetation affecting the functions and values of the habitat for salmonids, including overhead cover, a source of invertebrate prey and influence on water temperature.

4.1.4.3 Shoreline modification

According to the Washington Coastal Atlas, the Swinomish channel along the east side of the action area is highly developed, with between 81 percent and 100 percent of the shoreline modified (Figure 18). Coincident with shoreline development is a loss of overhanging and riparian vegetation and increase in overwater structures. According to the 2016 State of our Watershed Report (Swinomish Indian Tribal Community, 2016) restoration projects including a series of small marsh sites along the Swinomish Channel were created by the removal of dredge spoils, and tidal inundation at WDFW's Milltown Island in the South Fork was expanded. These restoration projects are contributing to the success of the 50-year Chinook recovery goals outlined in the species recovery plan.

The (Swinomish Indian Tribal Community, 2016) state that based on the current Skagit County Shoreline Management Plan and citing multiple state agencies:

> The soft shorelines of the nearshore could be 83% armored under a full build-out scenario. At present, approximately 55% of soft Skagit County shorelines (excluding bedrock areas that have no need for armoring) have bulkheads or levees. In addition to shorelines already armored or modified, current Skagit County zoning would allow 28% of soft shoreline to be bulkheaded under the residential exemption in the Shoreline regulations. This would mean a total of over 83% of all of the soft shoreline in Skagit County's jurisdiction could be armored behind bulkheads and/or levees if the county is fully developed. To add further concern, nearly 1 mile of shoreline has been armored in Skagit County since 2005.

A significant proportion of this armored area is on the east bank of the Swinomish, which borders the Swinomish Tribal Reservation. In addition, there are two marinas, a city float, and boat launches all within the action area in the lower Swinomish Channel. These shorelines are 100 percent modified to accommodate these facilities.

Smith (2005) rated the Swinomish Channel as poor for shoreline modification due to extensive (greater than 30 percent by miles) of hardscaping (rip rap, bulkheads) and diking. There are a significant number of overwater structures which reduce riparian vegetation, cause shading and reduce cover for juvenile fish.

4.1.4.4 Summary

The environmental baseline in the Swinomish Channel is degraded due to the extensive shoreline armoring, reduced access to off channel habitat by migrating salmon, substantial number of overwater structures and loss of riparian and overhanging vegetation.

4.1.5 Skagit Bay

The action area includes only a portion of Skagit Bay, specifically where the North fork Skagit River drains. Historically, this Skagit River delta contained numerous channels that branched from the mainstem of the river to the estuary. Before development to support agriculture and human occupation, tidal habitat and wetlands were extensive in the tens of thousands of acres in the Skagit Basin.

One ongoing restoration project in the action area, conducted under the SWC and by the Skagit River System Cooperative, is to increase fish passage between the North Fork Skagit River and the Swinomish Channel. This restoration project began in 2005 and is scheduled to end in 2055. Two of the five goals set for the project have been met, and a specific description of the project follows:

> The objective of this project is to improve the hydraulic connection between the North Fork of the Skagit River and the Swinomish Channel north of McGlinn Island. This action is expected to improve access by juvenile Chinook to estuarine rearing habitat in Padilla Bay. The current access, through a small opening in the rock jetty (known as the "Fish Hole") is limited because river flow is directed away from Swinomish Channel, and the opening is inaccessible at low tides. Channel dredging began in the 1890s so local farmers could transport goods to market. Dredge spoils were used to build a causeway between La Connor and McGlinn Island, to block freshwater flow/sediment transport from the river to the channel. In 1938 a jetty was built to

further restrict river-channel connectivity. The causeway/jetty have also greatly restricted juvenile salmonid access to extensive rearing habitat in Padilla Bay. In addition to physical obstruction of migratory pathways for juvenile (and returning adult) salmon, reduced freshwater input to the channel has greatly increased channel salinity and created a physiological barrier for juvenile chinook, which are very sensitive to high salinity. Restoration of river-channel connectivity is necessary to allow juvenile salmon to access rearing habitat in Padilla Bay. It is also a necessary precursor to further salmon habitat restoration along the channel. To date, two phases of feasibility studies and design have been completed.

4.1.5.1 Hydrologic modifications

Land uses, primarily agricultural has resulted in the loss of estuarine habitat in Skagit Bay [(C. J. Smith, 2005);Figure 16]. Extensive modifications due to diking have isolated former estuarine habitat. These habitats modifications have affected the most important areas for salmonids because of the proximity to the Skagit River (C. J. Smith, 2005). Virtually all the eastern shoreline of Skagit Bay from the southern end of the Swinomish Channel to West Pass is diked, as is Fir Island, which provided productive delta habitat.

4.1.5.2 Habitat conditions

Smith (2005) rated the Lower Skagit floodplain habitat "poor" for conditions resulting from extensive diking and loss of wetland habitat. This diking results in a loss of critical off channel habitat for salmonids. Smith (2005) states that more than 90 percent of floodplain loss and delta habitat are due to diking, draining and ditching. Currently, numerous former channels have been transformed into more than 161 km (100 mi) ditches to drain agricultural lands and thus have closed off access to flood refuge habitat, and important wetland habitat (C. J. Smith, 2005).

Odden (2014) cites a WDFW habitat study that concludes "habitat quality varies along the delta, with lower ranked habitat along the North Fork," which is within the vicinity of the action area. The Skagit Delta Management Unit is part of the CVA conducted by Odden (2014). This unit includes the action area at the extreme northern boundary. The portion of the action area within the management unit ranked one criterion (Process or Habitat) in the first tier in the CVA analysis. Therefore, only one of these criteria in this unit was ranked high enough to be considered further, but this reach was not ranked high enough for protection potential.

Eelgrass is prevalent in the action area portion of Skagit Bay at a depth of 3.31 to - 8.53 ft MLLW, while salt marsh is more limited and located at the mouth of the Swinomish Channel (Figure 17). Generally correlative wih eelgrass, herring spawning occurs in the action area as well, although in the deeper regions approximately 1 km from shore past Goat Island.

4.1.6 Similk Bay

The Similk Bay portion of the action area appears to be the least impacted by human development. Smith (2005) rates most of the shoreline as good with less than 10 percent of the miles of shoreline modified. The shoreline consists of rock bluffs and importantly, some feeder bluffs. The primary land use around the bay is residential development (primarily small parcels), although there are several large parcels and substantial tideland owned by the Swinomish Tribe (Figure 16). Because of this divided shoreline, Odden (2014) concluded that there was low potential for permanent protection.

A recent restoration project was completed in the north end of Similk Bay. The Turners Bay Lagoon restoration project was completed in 2012.[13] The project was overseen by the SWC and implemented through a partnership between the SRSC (Skagit River System Cooperative) and a private landowner. This restoration project was described as follows:

> Turner's Bay pocket estuary site is a nearly 60-acre salt marsh/lagoon complex located at the head of Similk Bay, on the north end of the Skagit Bay in Puget Sound. This project enhanced fish access and restored tidal inundation to 7.3 acres of the 59.4-acre tidal channel lagoon and marsh complex. Formerly, the upper portions of the salt marsh received muted tidal volumes and fish access was severely limited due to a road crossing and a non-functioning tide gate. An extensive feasibility study was completed in 2007 that identified removal of Similk Bay Road as the restoration action with the greatest habitat benefit (McBride, 2007). The study also documented portions of the site that had been ditched and/or filled and identified creosote-treated debris on the site from a previous log storage operation.

This project corrected the impaired processes that maintained this historic lagoon, which increased habitat area and quality for migrating salmonids and forage fish. Once completed the project restored 24 ha (59.4 ac) of nearshore estuarine habitat, created 2.9 ha (7.2 ac) of estuary, removed 14.5 km (9 mi) of shoreline armoring and removed one fish passage barrier.

4.1.6.1 Hydrologic modification

There are no hydromodifications in the form of diking, dredging or installation of tide gates within the action area to Similk Bay.

4.1.6.2 Habitat conditions

Eelgrass is prevalent in northern Similk bay with both *Z. marina* and *Z. japonica* represented. Eelgrass occurs between 2.82 and -14.83 ft MLLW (Figure 15 and Figure 17). The recent trend shows eelgrass increasing from 2012 to 2015 (last data point). Beaches provide habitat for surf smelt (*Hypomesus pretiosus*) and limited sand lance (*Ammodytes americanus*) spawning, while herring spawning occurs in the eelgrass beds that line the shorelines (Figure 17).

4.1.6.3 Shoreline modifications

The shoreline along Similk Bay includes rocky bluffs, and some feeder bluffs. Low density development is located along the southern third of the west side of the reservation near the shoreline. According to the Washington State Coastal Atlas the number of piers and docks along this shoreline is low from 0 to 1 (latest data from 2000), and there were no large boat facilities or motorboat launches. The amount of shoreline modification was low as well (data from 1994 to 2000) with 0 to 10 percent over most of the west side of the island and a few areas where 60 to 80 percent (0.36 km) and 81 to 100 percent (0.48 km) of the shoreline were modified.

4.1.6.4 Summary

Similk Bay is likely the least impacted portion of the action acrea. This due to the low-density residential land use (Figure 16). There is limited shoreline modification and only in areas of high population density.

[13] https://srp.rco.wa.gov/project/280/11865. Accessed 5/17/2022.

The area does not lend itself to the same activities (diking ditching or dredging) that impact salmonid habitat in other parts of the action area. There is eelgrass present which provide herring spawning and beaches where sand lance spawning has been documented. Therefore, the environmental baseline in Similk Bay appears to be in good condition.

4.1.7 Freshwater habitat

Although surface freshwater in the action area is scarce and not suitable for salmonid spawning, the small streams within the action area nonetheless provide salmon rearing habitat for juveniles that enter the creeks from Skagit Bay (Snee-Oosh Creek and Lone Tree Creek) or Swinomish Channel (Monks Creek and Fornsby Creek) (Eric M Beamer et al., 2013).

Drost (1979) identified three perennial streams in the action area: Snee-Oosh Creek, Munks Creek and "unnamed creek no.1", which is identified as Fornsby Creek in Gendaszek and Opatz (2013), as well as a marshy area and shallow pond in section 26, range 2 east, township 34 north (Figure 19). The aquatic life use is applied to these three streams, along with Lone Tree Creek, in the Swinomish WQS (Figure 20). The streams have cut deep ravines through the glacial sediments (Gendaszek & Opatz, 2013). During the summer dry season, creeks in the action area are primarily fed by groundwater seeps and springs (Drost, 1979; Gendaszek & Opatz, 2013). Snee-Oosh Creek originates from a large wetland complex (Gendaszek & Opatz, 2013). Groundwater cools streams and reduces diurnal temperature fluctuations (Gendaszek & Opatz, 2013).

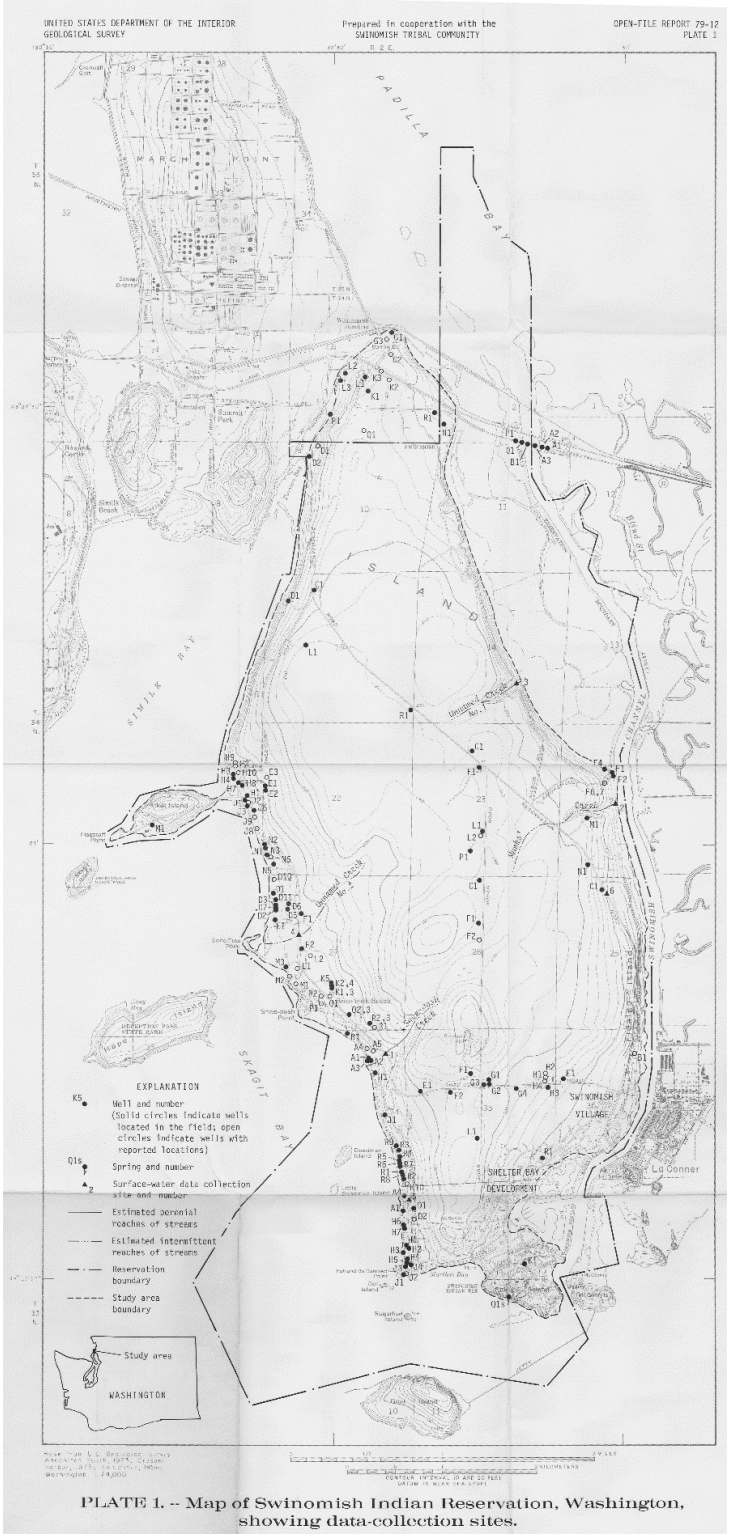


PLATE 1. -- Map of Swinomish Indian Reservation, Washington, showing data-collection sites.

Figure 19: Map of Swinomish Reservation water resources from Drost (1979)

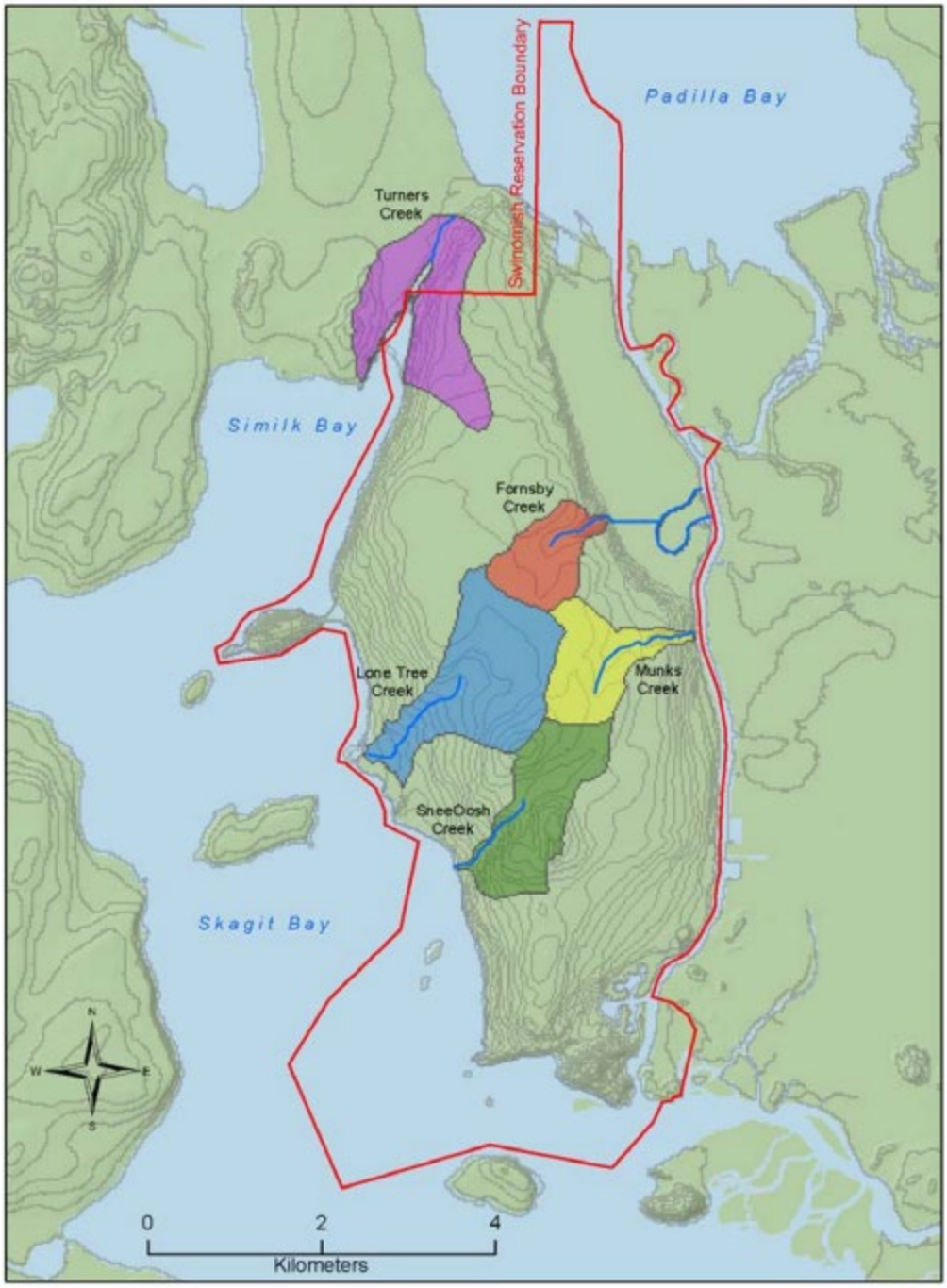


Figure 20: Location of freshwater streams and their watershed boundaries within the Swinomish Reservation (Beamer et al., 2004)

The USGS measured the discharges of Snee-Oosh and Munks Creeks between October 1975 and December 1976; flows never exceeded 0.9 CFS (Drost, 1979). Flows in Munks Creek and Snee-Oosh Creek were measured again in 1996 and the flows were similar to those measured in 1976 (USGS, 1998).

The USFWS National Wetlands Inventory (https://www.fws.gov/wetlands/) shows several wetlands within the action area (Figure 21).

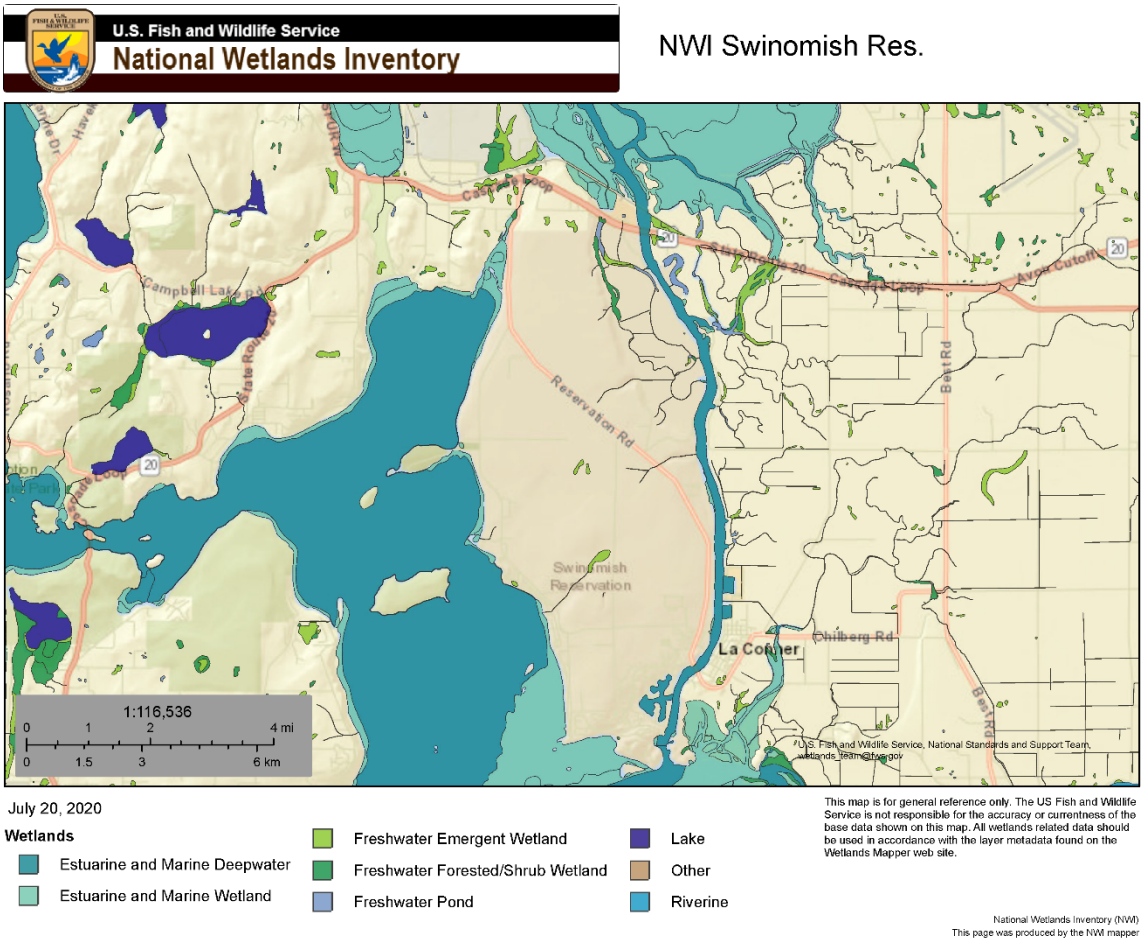


Figure 21: Wetlands near the action area

Six streams in or near the action area were electrofished to determine the presence of juvenile Chinook salmon (Eric M Beamer et al., 2013). Watershed and channel characteristics of these streams are listed in Table 11. Eric M Beamer et al. (2013) found that the presence of a culvert at the stream's mouth as well as whether the culvert is backwatered at high tide (see Table 11) influences the presence rate of juvenile Chinook salmon. Streams that have longshore sediment deposition at their mouth (see Table 11) have higher juvenile Chinook salmon presence rates than streams without longshore deposition. The presence of juvenile salmonids as determined by electrofishing is shown in Table 12.

Upland culverts can also impede fish passage. In 2006, the Swinomish Indian Tribal Community and partners removed a culvert on Lone Tree Creek, replacing it with a bridge (Beamer et al., 2009). Following replacement of the culvert, juvenile Chinook and coho were more abundant and their distribution within the creek was extended (Beamer et al., 2009). Culvert locations are available from the Washington Department of Fish and Wildlife's SalmonScape mapper (https://apps.wdfw.wa.gov/salmonscape/) and are shown in Figure 21.

Table 11. Summary of watershed and channel characteristics for streams in the action area (Beamer et al. 2013)

Stream name	Landscape characteristics		Channel characteristics					Stream mouth characteristics		
	Nearest river mouth & distance (km)	Watershed area (ha)	Slope %	Avg width (m)	Avg wetted width (m)	Avg max pool depth (m)	Survey length (m)	Longshore sediment deposition present	Enters estuary *	Culvert(s) present
Turners Cr	NF Skagit 13.12	47	1.0%	1.6	1.3	0.18	120	no	yes	yes, not tidally backwatered
Turners Spit Cr	NF Skagit 12.48	10	2.0%	0.8	0.5	0.04	28	no	yes	no
Fornsby Cr	NF Skagit 8.97	107	1.0%	3.4	1.1	0.18	185	no	yes	yes, tidally backwatered
Munks Cr	NF Skagit 6.22	123	2.5%	3.5	0.7	0.15	60	no	no	no
Lone Tree Cr	NF Skagit 6.76	253	2.4%	2.6	1.2	0.26	180 (60)	no	yes	no
Snee-Oosh Cr	NF Skagit 4.62	170	2.2%	3.1	1.7	0.21	175	no	no	yes, tidally backwatered

Table 12. Juvenile salmonid presence in action area streams as determined by electrofishing (Beamer et al. 2013)

Stream name	Chinook	Steelhead	Coho	Cutthroat	Chum
Turners Cr	yes	none found	none found	none found	none found
Turners Spit Cr	none found	none found	none found	none found	none found
Fornsby Cr	yes	none found	yes	none found	none found
Monks Cr	yes	yes	yes	none found	none found
Lone Tree Cr	yes	none found	yes	none found	yes
Snee-Oosh Cr	yes	none found	yes	yes	yes

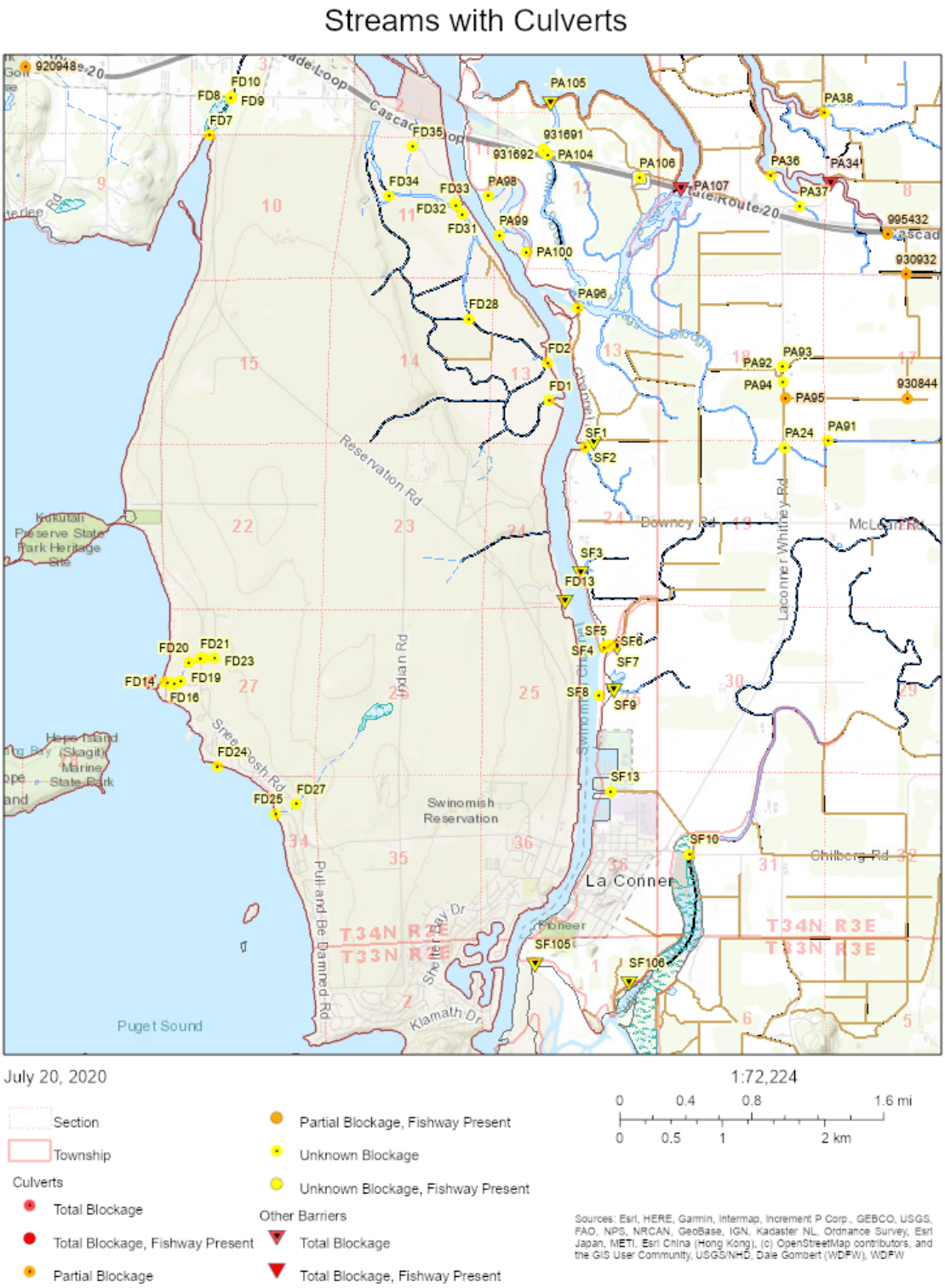


Figure 22: Salmon streams and culverts from SalmonScape

4.2 Existing Facilities and Discharges

This section presents (a) information about sites managed by EPA and Ecology, which have been identified for cleanup and (b) facilities with permitted discharges. This information along with the known habitat condition (described above) support the determination of a degraded baseline.

4.2.1 Regulated cleanup sites

This section identifies and describes hazardous cleanup sites that are known sources of pollutants subject to the Swinomish Tribe's aquatic life criteria for the purposes of describing the environmental baseline. EPA relied on the following data sources to obtain information about hazardous waste sites in the action area:

- EPA's Cleanups in My Community database (https://www.epa.gov/cleanups/cleanups-my-community)
- Washington Department of Ecology database (https://ecology.wa.gov/Spills-Cleanup/Contamination-cleanup/Cleanup-sites/Locate-contaminated-sites)

The EPA database provided information on eight sites near the action area, three of which may have the potential to contribute to water quality impacts in the action area (Table 13). The Washington Department of Ecology database identified 23 sites within a four-mile radius of the center of the action area. Of these, 18 have been designated 'No Further Action.' Therefore, five of the identified sites may affect surface waters (Table 13).

Of the eight sites with the potential to affect action area waters, only the Swinomish Tribal Economic Zone 1 is within the action area, and the March Point Landfill site is adjacent to the Swinomish Reservation, with fresh waters discharging from the site that may enter Padilla Bay within the action area (Figure 13, section 4.1.1). The March Point Landfill had reported surface water chemical analysis results (Table 13). The maximum, minimum and median concentrations are compared to the Tribe's marine water quality criteria for dissolved arsenic, copper, nickel, silver, and mercury. Exceedances of acute and chronic criteria are shaded in Table 14 and the number of individual exceedances per total is reported. For copper, nickel and silver the water quality criteria are expressed as dissolved metals concentrations, thus, we compared the observed dissolved concentrations of those metals to the dissolved criteria. Copper concentrations exceeded the acute criteria in two of nine samples and silver exceeded the acute criteria in the only sample taken. Nickel exceeded the chronic criteria in two out of two samples and mercury exceeded the chronic criteria. There was no exceedance of arsenic.

Figure 13 depicts the location of the sites found in both EPA's and Ecology's databases in relation to the action area. Table 13 describes the pollutant(s) that were released (or suspected to be released) to surface water in the action area and the site's cleanup status.

Table 13. Cleanup Sites Within or Near the Action Area

Sites Managed by EPA/Ecology	Surface Water Impacted	Cleanup Status	Within Action Area	Surface Water Pollutant
Shell Oil Products US Puget Sound Refinery (EPA)	unknown	Groundwater control in place	No	unknown

Sites Managed by EPA/Ecology	Surface Water Impacted	Cleanup Status	Within Action Area	Surface Water Pollutant
Swinomish Tribal Economic Zone 1 (EPA)	Yes	Unknown	Yes	Lead, PCBs, other metals and contaminants
Tecnal Corp (EPA)	Unknown	Unknown	No	Unknown
March Point Landfill (Ecology)	Suspected	Cleanup Started	Yes/Padilla Bay	Metals, solvents, pesticides, PAHs, PCBs
Northwest Petrochemical Corp Anacortes (Ecology)	Suspected	Cleanup Started	No	Phenolic compounds
Belisle Bros Crop Dusting Inc (Ecology)	Historical	CC-O&M/Monitoring	No	Inorganic conventional contaminants
Zimmerman's Shell (Ecology)	Suspected	Tanks decommissioned	No	Non-halogenated solvents, petroleum products
Whitmarsh Siding (Ecology)	Suspected	Awaiting Cleanup	No	Corrosive waste, petroleum products, phenolic compounds, PAHs

Table 14. Chemical Concentration of March Point Discharge and Comparison to Swinomish Tribe Aquatic Life Criteria.

March Point (Whitmarsh) Surface Water Samples (µg/L)					
	Dissolved Arsenic	**Dissolved Copper**	**Dissolved Nickel**	**Dissolved Silver**	**Total Mercury**
Number of sample results exceeding acute criteria	0/32	2/9	0/2	1/1	0/4
Number of sample results exceeding chronic criteria	0/32	3/9	2/2	na	3/4
Maximum concentration	5.1	6	11	8	0.071
Minimum concentration	0.5	3	9	8	0.022
Median concentration	2.2	3	10	8	0.047
Swinomish Marine AWQC					
Acute	69	4.8	74	1.9	2.1
Chronic	36	3.1	8.2	na	0.025
na=none available					

4.2.2 Point source discharges of pollutants (NPDES program)

Point sources are sources of pollutants which are discrete conveyances, such as discharge pipes or ditches (40 CFR 122.2). The NPDES permitting program is established under section 402 of the CWA and its implementing regulations for point source discharges of pollutants. These regulations include the requirement that permits be established to achieve compliance with the water quality standards.

The standards that EPA proposes to approve will be the basis for establishing water quality-based effluent limits to meet these standards in future NPDES permits. Previously, EPA had used the State of Washington's water quality standards to establish permit limits for point source discharges to waters within the Swinomish Reservation. For existing NPDES permits where the Tribe's water quality standards are more stringent than the State of Washington's water quality standards, EPA's approval of the Tribal standards will be beneficial relative to the environmental baseline because when the permits are reissued, the effluent limits will become more stringent. This may also be the case for permits issued by the Washington State Department of Ecology, for point sources discharging to waters of the State of Washington near Tribal waters, because EPA regulations require that permits be conditioned to ensure compliance with the applicable water quality requirements of all affected states (40 CFR 122.4(d)). Similarly, per the requirements at 40 CFR 131.10(b), states/authorized tribes must "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."

4.2.2.1 NPDES permits in the action area

The existing NPDES permits in the action area are listed in Table 15.

Table 15. NPDES Permits in the Action Area

Permit Name and Number	Receiving Water	Discharge Location	Issuing Agency	Permit Type	Mixing Zones?
Swinomish Indian Tribal Community North End WWTP (WA0025062)	Swinomish Channel (300 feet from Padilla Bay)	48.458056 °N, 122.515833 °W	EPA	Individual	Yes
Swinomish Shelter Bay WWTP (WA0024422)	Swinomish Channel	48.386667 °N, 122.504444 °W	EPA	Individual	Yes
City of La Conner (WA0022446)	Swinomish Channel	48.39258193 °N, 122.4964726 °W	Ecology	Individual	Yes
La Conner Maritime Services (WAG030074)	Swinomish Channel	48.401081560 °N, -122.493367500 °W	Ecology	General	No
Dunlap Towing Co. Westside Log Yard (WAR05I302)	Swinomish Channel	48.40392 °N, -122.498202 °W	EPA	General	No

Within the Swinomish Reservation, EPA has the authority to issue and enforce NPDES permits. Outside of the reservation boundaries, NPDES permits are issued and enforced by the Washington State Department of Ecology. EPA has oversight authority of the State's implementation of the NPDES permits program.

These permits set limits on the amount of pollutants that may be discharged to ambient waters. Limitations are established in permits wherever: a) EPA or the state has established minimum technology-based controls for a wastewater pollutant for the type of activity being regulated, or b) a reasonable potential exists for the wastewater discharge to exceed an applicable water quality standard.

Permit conditions generally include effluent limits, periodic monitoring to ensure that the effluent limits are being met, compliance conditions requiring improvements in operation or special studies, special operating conditions, and other administrative requirements such as prompt reporting of any spills. New water quality standards are considered and incorporated, if applicable, when permits are reissued every five years. The development of limitations, monitoring requirements, and other conditions in NPDES permits is discussed in the U.S. EPA NPDES Permit Writers' Manual (USEPA, 2010). Determining the need for water quality-based effluent limitations and the derivation of such limits from water quality standards is discussed in more detail in the Technical Support Document for Water quality-based Toxics Control (USEPA, 1991).

Effluent data for the facilities permitted by Ecology were obtained from Ecology's Permitting and Reporting Information System (PARIS) database (https://apps.ecology.wa.gov/paris), and effluent data for facilities permitted by EPA were obtained from EPA's Integrated Compliance Information System (ICIS) database. Although ICIS is an internal EPA database, effluent data from ICIS are publicly available from EPA's Enforcement and Compliance History Online database (https://echo.epa.gov). Effluent data for parameters regulated by the Tribe's aquatic life water quality criteria for each facility are summarized below.

4.2.2.2 Swinomish Reservation Sewer District Shelter Bay Community WWTP

Effluent data from ICIS for the Swinomish Reservation Sewer District Shelter Bay Community WWTP are summarized in Table 16.

Table 16. Summary of Effluent Data for Swinomish Reservation Sewer District Shelter Bay Community WWTP

	Chlorine, total (mg/L)		Ammonia, total as N, mg/L	Temperature, °C
	Monthly Average	Daily Max	Quarterly Maximum	Monthly Maximum
Minimum	0.01	0.02	0.09	13
Average	0.025	0.047	9.36	17.5
Maximum	0.12	0.12	29.5	25
Count	147	219	28	82
Date Range	2003 - 2018	2003 - 2018	2012 - 2019	2012 - 2019

4.2.2.3 Swinomish North End WWTP

According to the permit application received by EPA in December 2016, although this facility is authorized to discharge to surface water, it discharges to infiltration ponds. Effluent ammonia concentrations for the infiltration pond discharges, as provided in the application, are summarized in Table 17.

Table 17. Summary of Effluent Ammonia Data for the Swinomish Indian Tribal Community North End WWTP

	Ammonia, total as N, mg/L
Minimum	0.02
25th Percentile	0.03
Median	0.04

	Ammonia, total as N, mg/L
Average	2.37
75th Percentile	0.06
Maximum	52
Count	150
Date Range	10/2013 – 9/2016

4.2.2.4 City of La Conner

Effluent data from PARIS for the City of La Conner's wastewater treatment plant are summarized in Table 18. Effluent data for chlorine from 2008 and earlier were available in PARIS, however, these data are no longer relevant because the facility now has ultraviolet disinfection and is therefore no longer a source of residual chlorine. For phosphorus, an outlying value of 117 mg/L was not included in the summary in Table 18.

Table 18. Summary of Effluent Data for the City of La Conner

Parameter and Units	Reported Statistic	Minimum	Average	Maximum	Count	Date Range
Copper, total, µg/L	Average	4	8.89	22	23	2009-2013
	Daily max.	4	7.16	12	16	2009-2013
	Single sample	4.1	4.1	4.1	1	2014
Ammonia, total, mg/L	Average	0.06	70.7	28	16	2014-2018
	Single sample	0	7.48	28	13	2014-2018
Nitrate + Nitrite, total, mg/L	Average	0.12	21.8	42.6	16	2014-2018
	Single sample	0.12	20.96	36.6	13	2014-2018
Total Kjeldahl nitrogen, mg/L	Average	1.29	9.85	28.7	15	2014-2018
	Single sample	1.29	8.07	26.6	14	2014-2018
Phosphorus, total, mg/L	Average	1.76	5.99	13.4	11	2014-2018
	Single sample	1.76	5.48	9.68	13	2014-2017
Temperature, °C	Average	12	16.2	20.4	2	2015-2019
	Daily max.	15	18.5	22	2	2015-2019
	Single sample	10	16.2	22	43	2015-2019

In addition to the chemical-specific effluent data summarized in Table 18, reports of the results of acute whole effluent toxicity (WET) testing were also available in PARIS for this facility. Results are listed in Table 19.

Table 19. Acute WET Test Results for City of La Conner

Date	Species	Endpoint	Duration (hr)	NOEC (% effluent)	LOEC (% effluent)	LC50 (% effluent)
March 2015	Ceriodaphnia	Survival	48	100	>100	>100
March 2015	Fathead minnow	Survival	96	100	>100	>100
May 2015	Ceriodaphnia	Survival	48	100	>100	>100
May 2015	Fathead minnow	Survival	96	100	>100	>100
August 2015	Ceriodaphnia	Survival	48	100	>100	>100
August 2015	Fathead minnow	Survival	96	100	>100	>100
October 2015	Ceriodaphnia	Survival	48	100	>100	>100
October 2015	Fathead minnow	Survival	96	34	50	74.3
January 2019	Ceriodaphnia	Survival	48	100	>100	>100
January 2019	Fathead minnow	Survival	96	100	>100	>100

4.2.2.5 La Conner Maritime Services

La Conner Maritime Services is authorized to discharge pollutants under the State of Washington's general NPDES permit for boatyards (Figure 13). Effluent data from PARIS for this facility are summarized in Table 20.

Table 20. Summary of Effluent Data for La Conner Maritime Services

	Copper, total (µg/L)			Lead, total (µg/L)	Zinc, total (µg/L)	Nitrate + Nitrite (mg/L)
	Seasonal Avg.	Single Sample	Maximum	Single Sample	Single Sample	Single Sample
Minimum	37.72	12	12.0	0.2	5	0.29
Average	108.6	145.4	51.7	3.82	126	0.29
Maximum	549	1371	133	24	915	0.29
Count	17	75	36	38	74	1
Date Range	2011-2018	2011-2018	2011-2018	2011-2016	2011-2018	2012

4.2.2.6 Dunlap Towing Company Westside Log Yard

No effluent data for pollutants regulated by the Tribe's aquatic life water quality criteria were available in ICIS.

4.2.3 Baseline information regarding mixing zones

A mixing zone is a limited area or volume of water where initial dilution of a discharge takes place and where certain numeric water quality criteria may be exceeded (USEPA, 2014). The Swinomish Tribe's mixing zone policy appears in section 19-06.240 of the Tribal Code.

As stated in the Tribe's mixing zone policy, mixing zones may be authorized in conjunction with CWA section 402 (NPDES) or 404 permits. Since EPA is the permitting authority for discharges to Tribal waters, these permits, including any mixing zones that they incorporate, are federal actions that would be subject to future ESA section 7 consultations. Since the Tribe's mixing zone policy is not self-

implementing (i.e. it is only authorized in conjunction with NPDES permits), it is not subject to this ESA consultation and not analyzed in this BE.

To date, NPDES permits issued by EPA for discharges to the Tribe's waters have relied on the State of Washington's water quality standards, including the state's mixing zone policy at WAC 173-201A-400. On May 30, 2019, EPA approved certain provisions of the Swinomish Tribe's WQS not subject to the ESA section 7 consultation requirements and will have no effect on listed species or their critical habitat under the ESA, including the mixing zone policy. Any future mixing zones for NPDES permitted discharges to the Tribe's waters will be authorized by the Tribe's Department of Environmental Protection in accordance with the Tribe's mixing zone policy, and will undergo a separate ESA consultation process. Mixing zones for nearby discharges to waters of the State of Washington, such as the City of La Conner's sewage treatment plant, which are permitted by the Department of Ecology, will continue to be subject to the State of Washington's mixing zone policy.

Characteristics of the mixing zones that are currently authorized for NPDES permitted discharges within the action area are listed in Table 21. A summary of data for these discharges is provided in section 4.2.2.

Table 21. NPDES Discharges in the Action Area with Mixing Zones

Permit	Receiving Water	Discharge Location	Constituents	Flow Rate (mgd)	Acute MZ Distance (feet)	Acute Dilution Factor	Chronic MZ Distance (feet)	Chronic Dilution Factor
Swinomish Indian Tribal Community North End WWTP (WA0025062)	Swinomish Channel (300 feet from Padilla Bay)	48.458056 °N, 122.515833 °W	Ammonia, dissolved oxygen, pH, temperature	0.05	20.25	23.9	202.5	433
Swinomish Shelter Bay WWTP WA0024422	Swinomish Channel	48.386667 °N, 122.504444 °W	Ammonia, chlorine, dissolved oxygen, fecal coliform, pH, temperature	0.227	21.4	11	214.0	53
City of La Conner WA0022446	Swinomish Channel	48.39258193 °N, 122.4964726 °W	Ammonia, copper, Dissolved oxygen, fecal coliform, temperature, turbidity	0.52	21.0	2.9	208.5	12.3

The Tribe's mixing zone policy is more restrictive than the State's in that it does not allow mixing zones in freshwater or for bioaccumulative pollutants (section 19-06.240(7)). The Tribe's mixing zone policy also states that "mixing zones shall be designed to avoid or minimize adverse effects to species and critical habitat listed under the Endangered Species Act" (section 19-06.240(2)).

4.3 Ambient Water Quality In and Near the Action Area

EPA has obtained water quality data in and near the action area from the following sources:

- The State of Washington's Environmental Information Management (EIM) database.
- A public records request from the Washington Department of Fish and Wildlife (WDFW).
- The Swinomish Tribe
- The Water Quality Portal
- The USGS National Water Information System (NWIS)

The data used to characterize ambient water quality in and near the action area include sampling locations outside of the Swinomish Reservation, where Washington's WQS (WAC 173-201A) apply. However, all of the data are compared to the Tribe's aquatic life criteria for consistency.

In general, for summary statistics, "non-detect" values were assumed to be equal to the reporting limit. An exception was made when an unusually high reporting limit resulted in a "non-detect" result being the maximum value. Thus, when the data set includes non-detect values (i.e., if the detection frequency is less than 100%) the average values will be biased high, and the minimum values will also be biased high if the minimum result was a non-detect. Any result flagged as an estimated value was assumed to be equal to the reported estimate. Results from EIM with a data qualifier of "REJ," meaning that the data are unusable for all purposes and the presence or absence of the analyte cannot be verified, were excluded from the calculation of summary statistics.

For animal tissue, results for gills, gut contents, and liver or hepatopancreas tissue were excluded. Results for whole organisms, muscle and fillets were included. To determine if the level of pollutants in tissues represented a hazard to the aquatic community, we compared the tissues concentration data to tissue screening benchmarks. These tissue benchmarks are equally applicable to fresh and marine waters, so we developed one table to represent both fresh and marine biota.

4.3.1 Water quality assessment and TMDLs

Section 303 of the CWA establishes the WQS and TMDL programs. Under section 303(c), WQS are set by states, territories, and authorized tribes and these jurisdictions are required to develop lists of impaired waters (waters that are not attaining WQS). Tribes are not required to develop 303(d) lists and TMDLs unless they also have TAS to administer the section 303(d) program. No tribes currently have TAS authority for section 303(d). Furthermore, EPA does not develop 303(d) lists for tribal waters, but sometimes develops TMDLs for tribal waters. The CWA requires that those authorized jurisdictions set priority rankings for the impaired waters listed and then develop TMDLs for these waters, in order to bring them back into compliance with WQS. EPA must approve or disapprove the 303(d) lists and any TMDLs developed to restore the impaired waters.

Although the Swinomish Tribe does not have TAS authority to implement section 303(d), the State of Washington has a 303(d) listing program, and the action area includes waters listed by the State of Washington, as shown in Table 22. Note that both listings are based on animal tissue concentrations (specifically oysters) and not water column concentrations.

Table 22. 303(d) Listings in the Action Area

Listing ID	Waterbody Name	Parameter Name	CAS Number	Medium Name	Grid Cell Latitude	Grid Cell Longitude
12367	Swinomish Channel	Benzo(a)anthracene	56-55-3	Tissue	48.425 °N	122.495 °W
12371	Swinomish Channel	Chrysene	218-01-9	Tissue	48.425 °N	122.495 °W

TMDLs are developed to address both point source and non-point sources of pollution to waters of the United States. Basically, a TMDL is a "pollution budget" applicable to "impaired" water bodies. It is developed by assessing the water quality problems in a particular water body, identifying the pollutant sources that contribute to the identified problems, calculating the maximum amount of a pollutant that a water body can receive without violating the applicable WQS, and allocating an amount of pollution to each of the pollutant's sources contributing to that water body.

A TMDL is the sum of the allowable loads of a single pollutant from all contributing point and non-point sources. The calculation must include a margin of safety to allow for any uncertainties in the scientific methods used to derive the TMDL such as water quality modeling assumptions and must also account for seasonal changes in water quality. Calculations to establish TMDLs are subject to public review. Regarding point sources, all NPDES permits must comply with the applicable load allocations developed in a TMDL.

4.3.2 Baseline concentrations and sources of metal and metalloid pollutants in and near the action area

4.3.2.1 Tissue screening concentrations

Many of the chemicals regulated by the Tribe's numeric water quality criteria have been measured in animal tissue. Tissue screening concentrations, calculated as the product of the lower of the chronic freshwater or marine chronic criterion and the chemical specific bioconcentration factor (BCF) (Dyer, White-Hull, & Shephard, 2000) are listed in Table 23.

Table 23. Tissue Screening Concentrations for Assessing Ecological Risk of Bioaccumulated Chemicals to Aquatic Biota

Pollutant Class	Pollutant	Chemical Abstract Services (CAS) Number	Tissue Screening Concentrations (mg/kg) wet weight
Metal/Metalloid	Arsenic	7440-38-2	1.6
	Cadmium	7440-43-9	0.20
	Chromium $^{+3}$	16065-83-1	4.8
	Chromium $^{+6}$	18540-29-9	0.25
	Copper	7440-50-8	0.66[e]
	Iron	7439-89-6	Not available
	Lead	7439-92-1	0.24
	Mercury	7439-97-6	0.37

Pollutant Class	Pollutant	Chemical Abstract Services (CAS) Number	Tissue Screening Concentrations (mg/kg) wet weight
	Nickel	7440-02-0	0.39
	Selenium	7782-49-2	0.57
	Silver	7440-22-4	0.15
	Zinc	7440-66-6	35.6
Organic Chemicals			
	Chlorpyrifos	2921-88-2	0.0073
	4,4'-DDD	72-54-8	0.054
	4,4'-DDE	72-55-9	0.054
	4,4'-DDT	50-29-3	0.054
	Demeton	8065-48-3	0.00018
	Diazinon	333-41-5	0.019
	Dieldrin	60-57-1	0.0089
	Endosulfan I	959-98-8	0.0023
	Endosulfan II	33213-65-9	0.0023
	Endosulfan sulfate	1031-07-8	0.0023
	Heptachlor	76-44-8	0.057
	Heptachlor epoxide	1024-57-3	0.052
	Methoxychlor	72-43-5	0.047
	Mirex	2385-85-5	0.018
	4-Nitrophenol	100-02-7	0.50
	Parathion	56-38-2	0.0018
	Pentachlorophenol	87-86-5	3.1
	PCBs	1336-36-3	0.44
	Toxaphene	8001-35-2	0.0026
	Tributyltin	1461-22-9	0.0051

4.3.2.2 Hardness, calcium and magnesium

The freshwater aquatic life criteria for cadmium, chromium III, lead, nickel, silver, and zinc are based on the hardness of the receiving water. The freshwater aquatic life criteria for copper are based on the biotic ligand model (BLM), for which hardness (i.e., calcium and magnesium ions) is one of the model inputs.[14] Hardness data for surface freshwater in the action area are available from USGS NWIS and are reproduced in Table 24. Table 24 also lists the calcium and magnesium concentrations for each hardness value.

Metals with hardness-based or biotic ligand model (BLM) criteria become less toxic (and criteria become less stringent) as hardness increases. Therefore, when comparing ambient metal data in freshwater to the water quality criteria, EPA will make the conservative assumption that the water hardness is equal

[14] The necessary water quality input parameters for BLM calculations are temperature, pH, dissolved organic carbon, major geochemical cations (calcium, magnesium, sodium, and potassium), dissolved inorganic carbon (DIC, the sum of dissolved carbon dioxide, carbonic acid, bicarbonate, and carbonate), and other major geochemical anions (chloride, sulfate). USEPA, 2007 available at https://www.epa.gov/sites/production/files/2019-02/documents/al-freshwater-copper-2007-revision.pdf.

to the minimum hardness observed in surface freshwater in the action area, which is 30.9 mg/L as $CaCO_3$.

Table 24. Hardness, Calcium and Magnesium in Surface Freshwater in the Action Area

Station Number	Station Name	Sample Date	Hardness (mg/L as $CaCO_3$)	Calcium, filtered (mg/L)	Magnesium, filtered (mg/L)
12200704	Unnamed Tributary No. 2 Skagit Bay near La Conner, WA	3/2/1976	57	9.8	7.9
		5/21/1976	100	17	14
		8/26/1976	130	23	18
		9/24/1976	130	21	19
		12/16/1976	140	22	20
12200706	Unnamed Tributary to Swinomish Channel near La Conner, WA	3/2/1976	58	11	7.4
		5/21/1976	120	22	15
		8/26/1976	120	24	15
		9/24/1976	120	23	15
		12/16/1976	130	23	17
482655122313201	Seep Site 1	5/21/1991	30.9	6.6	3.5
482658122312901	Seep Site 2	5/21/1991	31.8	6.8	3.6
		Minimum	30.9	6.6	3.5
		Average	97.3	17.4	13.0
		Maximum	140	24.0	20.0
		Standard Deviation	41.0	6.9	5.8
		Count	12	12	12

4.3.2.3 Arsenic

Arsenic has been measured at quantifiable concentrations in water, animal and plant tissue, and marine sediment in or near the action area.

Summary statistics for arsenic in marine sediment and freshwater are listed in Table 25.

All arsenic results for water in Table 25 were obtained from EIM and were collected as part of a contaminated site investigation at the March Point Landfill, and thus may not be representative of arsenic concentrations elsewhere in the action area. The March Point Landfill is located adjacent to action area marine waters, northeast of the Swinomish Reservation boundary, but most of the associated sampling locations are within the action area. Nonetheless, none of the dissolved arsenic results for freshwater in Table 25 exceed the Tribe's freshwater aquatic life water quality criteria for dissolved arsenic.

None of the results for arsenic in sediment exceed the criterion for arsenic in marine sediment in the State of Washington's Sediment Management Standards (SMS), which is 57 mg/kg.

Table 25. Summary Statistics for Arsenic in Marine Sediment and Freshwater

Medium	Earliest Field Collection Start Date	Latest Field Collection Start Date	Minimum Result Value	Average Result Value	Maximum Result Value	Standard Deviation of Result Values	Count of Result Values	Detection Frequency (%)
Marine Sediment	7/11/1988	9/12/2017	2.4 mg/kg	8.9 mg/kg	52 mg/kg	7.0 mg/kg	71	87
Freshwater (total)	10/14/2008	10/6/2010	0.8 µg/L	4.2 µg/L	21.3 µg/L	4.6 µg/L	37	97
Freshwater (dissolved)	10/14/2008	10/6/2010	0.5 µg/L	2.5 µg/L	5.1 µg/L	1.1 µg/L	37	86

Results for arsenic in animal tissue are listed in Table 26. The tissue screening concentration (TSC) for arsenic is 1.6 mg/kg (wet basis). The maximum wet basis results for arsenic in dungeness crab (muscle) and pacific littleneck clam exceed this concentration. Estimating a wet basis tissue concentration by dividing the dry basis tissue concentrations by a factor of 5 (i.e., assuming wet tissue is 80% water), the dry basis arsenic tissue concentrations for dungeness crab (muscle and whole organism) and bent-nosed clam also exceed the TSC.

Table 26. Results for Arsenic in Animal Tissue

Sample Taxon Common Name	Tissue Type	Basis	Field Collection Start Date	Result Value (mg/kg)
Dungeness crab	Muscle	Dry	6/23/2008	50.2
Dungeness crab	Muscle	Wet	5/26/1999	7.35
Dungeness Crab	Muscle	Wet	6/1/2012	6.41
Dungeness Crab	Whole organism, not exoskeleton or shell	Dry	6/23/2008	36.6
Foolish Mussel	Whole organism, not exoskeleton or shell	Wet	11/13/2012	0.816
Starry Flounder	Fillet, skin off	Dry	6/18/2008	6.7
Pacific oyster	Whole organism, not exoskeleton or shell	Wet	6/1/1999	1.36
Bent-nosed clam	Whole organism, not exoskeleton or shell	Dry	6/17/2008	15.2
Pacific littleneck clam	Whole organism, not exoskeleton or shell	Wet	6/1/1999	2.04

Results for arsenic in eelgrass tissue are summarized in Table 27. The concentration in eelgrass tissues ranged from 0.593 mg/kg to 1.65 mg/kg dw) in leaves (which are greater than the concentrations in roots), converting these to wet weight to compare to the tissue screening benchmark results in a range of 0.119 mg/kg to 0.33 mg/kg; thus, there were no exceedance of the tissue benchmark (1.6 mg/kg).

Table 27. Summary of Results for Arsenic in Eelgrass Tissue

Tissue Type	Basis	Field Collection Start Date	Minimum Result Value (µg/g)	Average Result Value (µg/g)	Maximum Result Value (µg/g)	Standard Deviation of Result Values (µg/g)	Count of Result Values	Detection Frequency (%)
Leaves	Dry	1/8/2013	0.593	1.002	1.65	0.568	3	100
Roots, whole	Dry	1/8/2013	0.32	0.34	0.376	0.031	3	100

4.3.2.4 Cadmium

Cadmium has been measured at quantifiable concentrations in animal and plant tissue and marine sediment in or near the action area. There were 22 results each for dissolved and total cadmium in freshwater in EIM, with cadmium levels below the reporting limit in all cases. The reporting limits for dissolved cadmium ranged from 2-4 µg/L. At the minimum observed hardness of 30.9 mg/L as CaCO3, the acute and chronic water quality criteria for dissolved cadmium in freshwater are 0.64 µg/L and 0.11 µg/L, respectively. Because the reporting limits are higher than the criteria (evaluated at the minimum observed hardness), it is not clear whether the non-detect cadmium results would comply with water quality criteria.

Cadmium concentrations in marine sediment are summarized in Table 28. Only one of the 71 results (5.2 mg/kg) exceeded criterion for cadmium in marine sediment in the Washington SMS (5.1 mg/kg), and this result was one of the older results in the data set, measured in 1992. The next-highest concentration was 3.7 mg/kg.

Table 28. Summary of Results for Cadmium in Marine Sediment

Earliest Field Collection Start Date	Latest Field Collection Start Date	Minimum Result Value (mg/kg)	Average Result Value (mg/kg)	Maximum Result Value (mg/kg)	Standard Deviation of Result Values (mg/kg)	Count of Result Values	Detection Frequency (%)
7/11/1988	9/12/2017	0.07	0.63	5.2	0.82	71	77

Results for cadmium in animal tissue are listed in Table 29. The TSC for cadmium is 0.2 mg/kg (wet basis). The wet basis results for cadmium in foolish mussel, pacific oyster, and pacific littleneck clam exceed this concentration. Estimating a wet basis tissue concentration by dividing the dry basis tissue concentrations by a factor of 5 (i.e., assuming wet tissue is 80% water), the dry basis whole body cadmium tissue concentration for dungeness crab also exceeds the TSC.

Table 29. Results for Cadmium in Animal Tissue

Sample Taxon Common Name	Tissue Type	Basis	Field Collection Start Date	Result Value (mg/kg)
Dungeness crab	Muscle	Dry	6/23/2008	0.49
Dungeness crab	Muscle	Wet	5/26/1999	0.025 U
Dungeness Crab	Muscle	Wet	6/1/2012	0.002 U

Sample Taxon Common Name	Tissue Type	Basis	Field Collection Start Date	Result Value (mg/kg)
Dungeness Crab	Whole organism, not exoskeleton or shell	Dry	6/23/2008	1.55
Foolish Mussel	Whole organism, not exoskeleton or shell	Wet	11/13/2012	0.297
Starry Flounder	Fillet, skin off	Dry	6/18/2008	0.42 J
Pacific oyster	Whole organism, not exoskeleton or shell	Wet	6/1/1999	1.44
Bent-nosed clam	Whole organism, not exoskeleton or shell	Dry	6/17/2008	0.65 J
Pacific littleneck clam	Whole organism, not exoskeleton or shell	Wet	6/1/1999	0.317
Data Qualifier Definitions: U: Analyte was not detected at or above the reported result. J: Analyte was positively identified. The reported result is an estimate.				

Results for cadmium in eelgrass tissue are summarized in Table 30. The concentrations of cadmium in eelgrass leaves (which are higher than in roots) ranges from 1.91 mg/kg to 4.13 mg/kg (dry weight) converted to wet weight this range is 0.382 mg/kg to 0.826 mg/kg; these concentrations exceed the TSC of 0.2 mg/kg wet weight. The cadmium concentrations in eelgrass root do not exceed the TSC.

Table 30. Summary of Results for Cadmium in Eelgrass Tissue

Tissue Type	Basis	Field Collection Start Date	Minimum Result Value (µg/g)	Average Result Value (µg/g)	Maximum Result Value (µg/g)	Standard Deviation of Result Values (µg/g)	Count of Result Values	Detection Frequency (%)
Leaves	Dry	1/8/2013	1.91	2.69	4.13	1.25	3	100
Roots, whole	Dry	1/8/2013	0.719	0.773	0.801	0.047	3	100

4.3.2.5 Chromium

The Swinomish WQS establish criteria for chromium III (CAS #16065-83-1) and chromium VI (CAS #18540-29-9).

However, nearly all available results for chromium in or near the action area are for total chromium (CAS #7440-47-3), which includes all of chromium's oxidation states. There were no results for chromium III. There was a single "non-detect" result for chromium VI in marine sediment measured in 1992.

Results for total chromium (all oxidation states) in marine sediment and freshwater are summarized in Table 31. None of the quantifiable results for chromium in marine sediment exceeded the marine sediment chromium criterion in the Washington SMS (260 mg/kg), although one non-detect result had a reporting limit equal to the sediment criterion.

The Tribe's water quality criteria for chromium III and chromium VI are expressed in terms of dissolved metal. All results for dissolved chromium (all oxidation states) in water were non-detect. All chromium results for water were collected as part of a contaminated site investigation at the March Point Landfill, and thus may not be representative of chromium concentrations elsewhere in the action area.

Table 31. Summary of Results for Chromium in Marine Sediment and Freshwater

Medium	Earliest Field Collection Start Date	Latest Field Collection Start Date	Minimum Result Value	Average Result Value	Maximum Result Value	Standard Deviation of Result Values	Count of Result Values	Detection Frequency (%)
Marine Sediment	7/11/1988	9/12/2017	8.89 mg/kg	39.2 mg/kg	88 mg/kg[1]	22.4 mg/kg	54	98
Freshwater (total)	10/14/2008	7/23/2009	5 µg/L U	7 µg/L	31 µg/L	5.7 µg/L	22	18

Notes:

1. There was one non-detect result in the data set, and the reporting limit for this result was 260 mg/kg. Since this reporting level is higher than the highest quantifiable result (88 mg/kg), and since the actual concentration is less than 260 mg/kg, it was not listed as the maximum result in the table.

Data Qualifier Definitions:
U: Analyte was not detected at or above the reported result.

Results for chromium in animal tissue are listed in Table 32. The TSCs for chromium III and chromium VI are 4.8 and 0.25 mg/kg, respectively (wet weight). Estimating a wet basis tissue concentration by dividing the dry basis tissue concentrations by a factor of 5 (i.e., assuming wet tissue is 80% water), the tissue concentration of chromium in bent-nosed clam exceeds the TSCs for both chromium III and chromium VI. The whole-body tissue concentration of chromium in dungeness crab exceeds the TSC only for chromium VI. The tissue concentration of chromium in starry flounder and dungeness crab muscle are below the TSCs for both chromium III and chromium VI.

Table 32. Summary of Results for Chromium in Animal Tissue

Sample Taxon Common Name	Tissue Type	Basis	Field Collection Start Date	Result Value (mg/kg)
Dungeness crab	Muscle	Dry	6/23/2008	0.41
Dungeness crab	Whole organism, not exoskeleton or shell	Dry	6/23/2008	5.66
Starry flounder	Fillet, skin off	Dry	6/18/2008	0.63
Bent-nosed clam	Whole organism, not exoskeleton or shell	Dry	6/17/2008	30.5

Results for chromium in eelgrass tissue are summarized in Table 33. Concentrations of chromium (total) in eelgrass are estimated and range from 0.046 mg/kg to 0.081 mg/kg (dw; which ae high than the concentration in roots). Conversion to wet weight results in a range of 0.092mg/kg to 0.0162 mg/kg; thus, the concentrations of chromium in eelgrass are below the tissue the TSCs for chromium III and chromium VI, which are 4.8 and 0.25 mg/kg, respectively (wet weight).

Table 33. Summary of Results for Chromium in Eelgrass Tissue

Tissue Type	Basis	Field Collection Start Date	Minimum Result Value (µg/g)	Average Result Value (µg/g)	Maximum Result Value (µg/g)	Standard Deviation of Result Values (µg/g)	Count of Result Values	Detection Frequency (%)
Leaves	Dry	1/8/2013	0.046 J	0.060 J	0.081 J	0.019	3	100
Roots, whole	Dry	1/8/2013	0.040 J	0.062 J	0.074 J	0.019	3	100
Data Qualifier Definitions: J: Analyte was positively identified. The reported result is an estimate.								

4.3.2.6 Copper

Copper has been measured at quantifiable concentrations in water, animal and plant tissue, and marine sediment in or near the action area.

Summary statistics for copper in marine sediment and freshwater are listed in Table 34. None of the quantifiable results for copper in marine sediment exceeded the marine sediment copper criterion in the Washington SMS (390 mg/kg).

All copper results for water were collected as part of a contaminated site investigation at the March Point Landfill, and thus may not be representative of copper concentrations elsewhere in the action area. Copper results for water are not compared to the Tribe's criteria because insufficient data were available for input values to use the copper BLM to generate criteria values.

Table 34. Summary of Results for Copper in Marine Sediment and Freshwater

Medium	Earliest Field Collection Start Date	Latest Field Collection Start Date	Minimum Result Value	Average Result Value	Maximum Result Value	Standard Deviation of Result Values	Count of Result Values	Detection Frequency (%)
Marine Sediment	7/11/1988	9/12/2017	5.46 mg/kg	31.4 mg/kg	235 mg/kg[1]	27.3 mg/kg	131	99
Freshwater (total)	10/31/2008	7/29/2009	2 µg/L U	6 µg/L	38 µg/L	7.6 µg/L	22	77
Freshwater (dissolved)	11/4/2008	7/28/2009	2 µg/L U	2.8 µg/L	6 µg/L	1.1 µg/L	22	50
Notes: 1. There was one non-detect result in the data set, and the reporting limit for this result was 380 mg/kg. Since this reporting level is higher than the highest quantifiable result (235 mg/kg), it was not listed as the maximum result in the table. Data Qualifier Definitions: U: Analyte was not detected at or above the reported result.								

Results for copper in animal tissue are listed in Table 35. The TSC for copper is 0.66 mg/kg (wet basis). The wet basis results for copper in dungeness crab (muscle) and foolish mussel exceed this concentration. Estimating a wet basis tissue concentration by dividing the dry basis tissue concentrations by a factor of 5 (i.e., assuming wet tissue is 80% water), the dry basis copper tissue concentrations for dungeness crab (muscle and whole body), starry flounder, and bent-nosed clam also exceed the TSC.

Table 35. Summary of Results for Copper in Animal Tissue

Sample Taxon Common Name	Tissue Type	Basis	Field Collection Start Date	Result Value (mg/kg)
Dungeness crab	Muscle	Dry	6/23/2008	42
Dungeness crab	Muscle	Wet	6/1/2012	9.31
Dungeness crab	Whole organism, not exoskeleton or shell	Dry	6/23/2008	180
Foolish mussel	Whole organism, not exoskeleton or shell	Wet	11/13/2012	0.815
Starry flounder	Fillet, skin off	Dry	6/18/2008	13.1 J
Bent-nosed clam	Whole organism, not exoskeleton or shell	Dry	6/17/2008	22.9
Data Qualifier Definitions: J: Analyte was positively identified. The reported result is an estimate.				

Results for copper in eelgrass tissue are summarized in Table 36. The concentration in eelgrass tissues ranged from 93.2 mg/kg to 323 mg/kg (dw), converting these to wet weight to compare to the tissue screening benchmark results in a range of 18.6 mg/kg to 64.6 mg/kg; thus, the concentrations in eelgrass exceed the tissue benchmark (0.66 mg/kg) by up to two orders of magnitude.

Table 36. Summary of Results for Copper in Eelgrass Tissue

Tissue Type	Basis	Field Collection Start Date	Minimum Result Value (µg/g)	Average Result Value (µg/g)	Maximum Result Value (µg/g)	Standard Deviation of Result Values (µg/g)	Count of Result Values	Detection Frequency (%)
Leaves	Dry	1/8/2013	93.2	177.1	323	127	3	100
Roots, whole	Dry	1/8/2013	96.6	112	129	16.3	3	100

4.3.2.7 Iron

Iron has been measured at quantifiable concentrations in freshwater, plant tissue, and in marine sediment in or near the action area. Summary statistics for iron in marine sediment and freshwater are listed in Table 37.

All iron results for freshwater were collected as part of a contaminated site investigation at the March Point Landfill, and thus may not be representative of iron concentrations elsewhere in the action area. Of the 22 results for total iron in freshwater, 16 exceeded the Tribe's water quality criterion of 1,000 µg/L.

There was a single non-detect result for dissolved iron in marine water collected by the USGS at station # 482425122331601 (Lone Tree Pocket Estuary) on September 21, 2006.

The Washington SMS do not establish criteria for iron. EPA Region 3 has not established a marine sediment screening benchmark for iron but has established a freshwater sediment screening benchmark of 20,000 mg/kg. Four of the five marine sediment results exceed this benchmark.

Table 37. Summary of Results for Iron in Marine Sediment and Freshwater

Medium	Earliest Field Collection Start Date	Latest Field Collection Start Date	Minimum Result Value	Average Result Value	Maximum Result Value	Standard Deviation of Result Values	Count of Result Values	Detection Frequency (%)
Marine Sediment	6/4/1997	8/3/2006	13,300 mg/kg	31,032 mg/kg	41,700 mg/kg	11,343 mg/kg	5	100
Freshwater (total)	10/14/2008	7/23/2009	490 µg/L	4,061 µg/L	18,000 µg/L	5,248 µg/L	22	100
Freshwater (dissolved)	10/14/2008	7/23/2009	50 µg/L U	158 µg/L	530 µg/L	140 µg/L	22	50
Data Qualifier Definitions: U: Analyte was not detected at or above the reported result.								

Results for iron in eelgrass tissue are summarized in Table 38. However, there is no TSC to compare to the iron concentrations in these tissues.

Table 38. Summary of Results for Iron in Eelgrass Tissue

Tissue Type	Basis	Field Collection Start Date	Minimum Result Value (µg/g)	Average Result Value (µg/g)	Maximum Result Value (µg/g)	Standard Deviation of Result Values (µg/g)	Count of Result Values	Detection Frequency (%)
Leaves	Dry	1/8/2013	93.2	177.1	323	127	3	100
Roots, whole	Dry	1/8/2013	96.6	112	129	16.3	3	100

4.3.2.8 Lead

Lead has been measured at quantifiable concentrations in animal and plant tissue and marine sediment in or near the action area. The Tribe's aquatic life water quality criteria for lead are expressed in terms of dissolved metal. There were 37 results for dissolved lead in freshwater in EIM, but these were all non-detect, with reporting limits ranging from 1 -5 µg/L. At the minimum observed hardness of 30.9 mg/L as CaCO3, the acute and chronic water quality criteria for dissolved lead in freshwater are 17.6 µg/L and 0.69 µg/L, respectively. Because the reporting limits are higher than the chronic criterion (evaluated at the minimum observed hardness), it is not clear whether the non-detect lead results would comply with the chronic water quality criterion, although they do demonstrate compliance with the acute criterion. Total lead has been measured at quantifiable concentrations in surface water, however.

Summary statistics for lead in marine sediment and freshwater are listed in Table 39. None of the quantifiable results for lead in marine sediment exceeded the marine sediment lead criterion in the Washington SMS (450 mg/kg).

Table 39. Summary of Results for Lead in Marine Sediment and Freshwater

Medium	Earliest Field Collection Start Date	Latest Field Collection Start Date	Minimum Result Value	Average Result Value	Maximum Result Value	Standard Deviation of Result Values	Count of Result Values	Detection Frequency (%)
Marine Sediment	7/11/1988	9/12/2017	0.59 mg/kg	7.56 mg/kg	37 mg/kg[1]	5.62 mg/kg	71	89
Freshwater (total)	10/14/2008	10/6/2010	1 µg/L U	3.2 µg/L	24 µg/L	4.4 µg/L	37	24

Notes:

1. One of the non-detect results in the data set had a reporting limit of 440 mg/kg, which was more than 10 times the maximum quantifiable result (37 mg/kg). It was therefore excluded from the summary statistics.

Data Qualifier Definitions:
U: Analyte was not detected at or above the reported result.

Results for lead in animal tissue are listed in Table 40. The TSC for lead is 0.24 mg/kg (wet basis). None of the measured wet basis tissue concentrations exceed this concentration. Estimating a wet basis tissue concentration by dividing the dry basis tissue concentrations by a factor of 5 (i.e., assuming wet tissue is 80% water), the dry basis lead tissue concentrations for dungeness crab muscle and whole organism, starry flounder, and bent-nosed clam exceed the TSC.

Table 40. Summary of Results for Lead in Animal Tissue

Sample Taxon Common Name	Tissue Type	Basis	Field Collection Start Date	Result Value (mg/kg)
Dungeness crab	Muscle	Dry	6/23/2008	5.79
Dungeness crab	Muscle	Wet	5/26/1999	0.020 J
Dungeness crab	Muscle	Wet	6/1/2012	0.0041 U
Dungeness crab	Whole organism, not exoskeleton or shell	Dry	6/23/2008	8.19
Foolish mussel	Whole organism, not exoskeleton or shell	Wet	11/13/2012	0.0341
Starry flounder	Fillet, skin off	Dry	6/18/2008	5.82
Pacific oyster	Whole organism, not exoskeleton or shell	Wet	6/1/1999	0.043
Bent-nosed clam	Whole organism, not exoskeleton or shell	Dry	6/17/2008	8.31
Pacific littleneck clam	Whole organism, not exoskeleton or shell	Wet	6/1/1999	0.128

Data Qualifier Definitions:
J: Analyte was positively identified. The reported result is an estimate.
U: Analyte was not detected at or above the reported result.

Results for lead in eelgrass tissue are summarized in Table 41. The concentrations of lead in eelgrass leaves ranges from 0.095 mg/kg to 0.303 mg/kg (dw), converting this to wet weight results in a range of 0.019 mg/kg to 0.060 mg/kg; thus, the concentrations of lead in eelgrass tissues does not exceed the TSC of 0.24 mg/kg.

Table 41. Summary of Results for Lead in Eelgrass Tissue

Tissue Type	Basis	Field Collection Start Date	Minimum Result Value (µg/g)	Average Result Value (µg/g)	Maximum Result Value (µg/g)	Standard Deviation of Result Values (µg/g)	Count of Result Values	Detection Frequency (%)
Leaves	Dry	1/8/2013	0.095	0.165	0.303	0.119	3	100
Roots, whole	Dry	1/8/2013	0.025 J	0.045 J	0.068 J	0.022	3	100
Data Qualifier Definitions: J: Analyte was positively identified. The reported result is an estimate.								

4.3.2.9 Mercury

Mercury has been measured at quantifiable concentrations in animal tissue and marine sediment in or near the action area. There were 37 results for dissolved mercury in freshwater in EIM, all of which were non-detect, with a reporting limit of 20 ng/L. Total mercury has been measured at detectable concentrations in surface water, however. Mercury was analyzed in eelgrass tissue, but all results were rejected and the presence of absence of mercury could not be verified.

All mercury results for water were collected as part of a contaminated site investigation at the March Point Landfill, and thus may not be representative of mercury concentrations elsewhere in the action area. The reporting limit for the total mercury results in water was 20 ng/L, which is higher than the chronic criterion for total mercury in freshwater (12 ng/L), thus, all four of the quantifiable results for mercury in freshwater exceed the chronic mercury criterion.

Summary statistics for mercury in marine sediment and freshwater are listed in Table 42. There were three non-detect results in marine sediment that had reporting limits that were higher than the maximum quantifiable result. Thus, including the non-detect results in the summary statistics would have artificially increased the average and maximum values, and therefore only the quantifiable results for marine sediment are included in the summary. There were an additional 3 results for marine sediment in EIM that had a data qualifier of “REJ,” which means that the data are unusable for all purposes and the presence or absence of the analyte cannot be verified; these were not included in the summary statistics. None of the quantifiable results for mercury in marine sediment exceed the criterion in the Washington SMS (0.41 mg/kg).

Table 42. Summary of Results for Mercury in Marine Sediment and Freshwater

Medium	Earliest Field Collection Start Date	Latest Field Collection Start Date	Minimum Result Value	Average Result Value	Maximum Result Value	Standard Deviation of Result Values	Count of Result Values	Detection Frequency (%)
Marine Sediment (quantifiable results only)	7/11/1988	9/12/2017	0.005 mg/L	0.068 mg/L	0.20 mg/L	0.041 mg/L	85	68
Freshwater (total)	10/14/2008	10/6/2010	20 ng/L U	23 ng/L	71 ng/L	11 ng/L	37	11
Data Qualifier Definitions: U: Analyte was not detected at or above the reported result.								

Results for mercury in animal tissue are listed in Table 43. The TSC for mercury is 0.37 mg/kg or 370 µg/kg (wet basis). None of the mercury tissue concentrations that were reported on a wet basis exceed this concentration. Estimating a wet basis tissue concentration by dividing the dry basis tissue concentrations by a factor of 5 (i.e., assuming wet tissue is 80% water), none of the mercury tissue concentrations reported on a dry basis exceed the TSC.

Table 43. Summary of Results for Mercury in Animal Tissue

Sample Taxon Common Name	Tissue Type	Basis	Field Collection Start Date	Result Value (mg/kg)
Dungeness crab	Muscle	Dry	6/23/2008	0.28
Dungeness crab	Muscle	Wet	5/26/1999	0.0561
Dungeness crab	Muscle	Wet	6/1/2012	0.0563
Dungeness crab	Whole organism, not exoskeleton or shell	Dry	6/23/2008	0.16
Foolish mussel	Whole organism, not exoskeleton or shell	Wet	11/13/2012	0.00544
Starry flounder	Fillet, skin off	Dry	6/18/2008	0.35
Pacific oyster	Whole organism, not exoskeleton or shell	Wet	6/1/1999	0.022
Bent-nosed clam	Whole organism, not exoskeleton or shell	Dry	6/17/2008	0.06
Pacific littleneck clam	Whole organism, not exoskeleton or shell	Wet	6/1/1999	0.015

4.3.2.10 Nickel

Nickel has been measured at quantifiable concentrations in water, animal and plant tissue, and marine sediment in or near the action area.

Summary statistics for nickel in marine sediment and freshwater are listed in Table 44.

At the minimum observed hardness of 30.9 mg/L as $CaCO_3$, the acute and chronic water quality criteria for dissolved nickel in freshwater are 173 µg/L and 19.3 µg/L, respectively. All results for dissolved nickel in freshwater are below the water quality criteria.

The Washington SMS do not establish a criterion for nickel in marine sediment but do establish a freshwater sediment cleanup objective for nickel of 26 mg/kg. Thirty six of the 50 results for nickel in marine sediment exceed the freshwater sediment cleanup objective from the SMS.

Table 44. Summary of Results for Nickel in Marine Sediment and Freshwater

Medium	Earliest Field Collection Start Date	Latest Field Collection Start Date	Minimum Result Value	Average Result Value	Maximum Result Value	Standard Deviation of Result Values	Count of Result Values	Detection Frequency (%)
Marine Sediment	7/11/1988	6/19/2014	6.2 mg/kg	38.7 mg/kg	82 mg/kg	18.6	50	100
Freshwater (total)	10/30/2008	7/31/2009	2 µg/L	10.1 µg/L	72.2 µg/L	14.3 µg/L	22	100
Freshwater (dissolved)	10/30/2008	7/31/2009	1.7 µg/L	5.2 µg/L	11.0 µg/L	2.2 µg/L	22	100

Results for nickel in animal tissue are listed in Table 45. The TSC for nickel is 0.39 mg/kg or 390 µg/kg (wet basis). Estimating a wet basis tissue concentration by dividing the dry basis tissue concentrations by a factor of 5 (i.e., assuming wet tissue is 80% water), the nickel tissue concentrations in dungeness crab (whole organism) and bent-nosed clam exceed the TSC.

Table 45. Summary of Results for Nickel in Animal Tissue

Sample Taxon Common Name	Tissue Type	Basis	Field Collection Start Date	Result Value (mg/kg)
Dungeness crab	Muscle	Dry	6/23/2008	0.84
Dungeness crab	Whole organism, not exoskeleton or shell	Dry	6/23/2008	3.64
Starry flounder	Fillet, skin off	Dry	6/18/2008	1.88
Bent-nosed clam	Whole organism, not exoskeleton or shell	Dry	6/17/2008	16.6

Results for nickel in eelgrass tissue are summarized in Table 46. The concentrations of nickel in eelgrass leaves range from 0.513 mg/kg to 1.12 mg/kg (dw). Converting this to wet weight the range is 0.102 mg/kg to 0.224 mg/kg, below the TSC of 0.39 mg/kg (ww) in all cases.

Table 46. Summary of Results for Nickel in Eelgrass Tissue

Tissue Type	Basis	Field Collection Start Date	Minimum Result Value (µg/g)	Average Result Value (µg/g)	Maximum Result Value (µg/g)	Standard Deviation of Result Values (µg/g)	Count of Result Values	Detection Frequency (%)
Leaves	Dry	1/8/2013	0.513	0.727	1.12	0.341	3	100
Roots, whole	Dry	1/8/2013	0.176	0.186	0.205	0.016	3	100

4.3.2.11 Selenium

Selenium has been measured in marine sediment and animal tissue in or near the action area. There were 22 results each for dissolved and total selenium in freshwater in EIM, all non-detect. The water column portion of the freshwater selenium criterion is expressed in terms of total recoverable selenium. The reporting limits for the freshwater selenium results in EIM ranged from 50 – 100 µg/L, which are higher than the water column freshwater criteria (1.5 µg/L in lentic systems and 3.1 µg/L in lotic systems). Thus, it is not clear whether the concentrations of total selenium in freshwater would comply with the water column criteria.

Selenium concentrations in marine sediment are summarized in Table 47. The Washington SMS do not establish a criterion for selenium in marine sediment but do establish a freshwater sediment cleanup objective for selenium of 11 mg/kg. None of the results for selenium in marine sediment exceed the cleanup objective.

Table 47. Summary of Selenium Results in Marine Sediment

Earliest Field Collection Start Date	Latest Field Collection Start Date	Minimum Result Value (mg/kg)	Average Result Value (mg/kg)	Maximum Result Value (mg/kg)	Standard Deviation of Result Values (mg/kg)	Count of Result Values	Detection Frequency (%)
5/1/1992	9/12/2017	0.019 mg/kg U	0.66 mg/kg	6.0 mg/kg	1.2 mg/kg	23	35
Data Qualifier Definitions: U: Analyte was not detected at or above the reported result.							

Results for selenium in animal tissue are listed in Table 48. The Tribe's water quality criteria for selenium in freshwater are expressed, in part, in terms of fish tissue. Although the tissue results in Table 48 are organisms collected from marine water, EPA will compare the Tribe's freshwater tissue criteria to these results. The criteria are 8.5 mg/kg dry weight in a whole organism or 11.3 mg/kg dry weight in muscle. Estimating dry weight concentrations by multiplying the wet tissue concentrations by a factor of 5 (i.e., assuming wet tissue is 80% water), none of the tissue results in Table 48 would exceed the freshwater tissue criteria.

Table 48. Results for Selenium in Animal Tissue

Sample Taxon Common Name	Tissue Type	Basis	Field Collection Start Date	Result Value
Dungeness crab	Muscle	Wet	5/26/1999	496 µg/kg
Pacific oyster	Whole organism, not exoskeleton or shell	Wet	6/1/1999	470 µg/kg
Pacific littleneck clam	Whole organism, not exoskeleton or shell	Wet	6/1/1999	535 µg/kg

4.3.2.12 Silver

Silver has been measured at quantifiable concentrations in water, animal tissue, and marine sediment in or near the action area.

Summary statistics for silver in marine sediment and freshwater are listed in Table 49. For sediment, both unqualified and J-flagged (estimated) data were included in the summary statistics (except for the overall count and detection frequency) but U and UJ flagged data were not included. Half of the results for silver in marine sediment were non-detect, and many of the non-detect results had reporting limits that were higher than the maximum quantifiable marine sediment result in Table 49. Thus, including the non-detect results in the summary statistics would have artificially increased the average and maximum values.

At the minimum observed hardness of 30.9 mg/L as CaCO3, the acute water quality criterion for dissolved silver in freshwater is 0.43 µg/L. Only one of the 22 results for dissolved silver in freshwater was quantifiable, although the reporting limits were relatively high, at 3 – 6 µg/L. However, the maximum concentration of silver of 8 µg/L was quantifiable and exceeds the acute water quality criterion for silver. All silver results for water were collected as part of a contaminated site investigation at the March Point Landfill, and thus may not be representative of silver concentrations elsewhere in the action area.

The marine sediment criterion for silver in the Washington SMS is 6.1 mg/kg. None of the quantifiable results for silver in marine sediment exceed this criterion.

Table 49. Summary of Results for Silver in Marine Sediment and Freshwater

Medium	Earliest Field Collection Start Date	Latest Field Collection Start Date	Minimum Result Value	Average Result Value	Maximum Result Value	Standard Deviation of Result Values	Count of Result Values	Detection Frequency (%)
Marine Sediment (quantifiable results only)	7/11/1988	8/3/2008	0.056 mg/kg	0.15 mg/kg	0.43 mg/kg	0.11	70	50
Freshwater (total)	10/14/2008	7/23/2009	3 µg/L U	3.5 µg/L	7 µg/L	1.2 µg/L	22	9.1
Freshwater (dissolved)	10/14/2008	7/23/2009	3 µg/L U	3.5 µg/L	8 µg/L	1.3 µg/L	22	4.5
Data Qualifier Definitions: U: Analyte was not detected at or above the reported result.								

Results for silver in animal tissue are listed in Table 50. The TSC for silver is 0.15 mg/kg (wet basis). Estimating a wet basis tissue concentration by dividing the dry basis tissue concentrations by a factor of 5 (i.e., assuming wet tissue is 80% water), the silver tissue concentrations in dungeness crab (muscle and whole organism) exceed the TSC.

Table 50. Results for Silver in Animal Tissue

Sample Taxon Common Name	Tissue Type	Basis	Field Collection Start Date	Result Value (mg/kg)
Dungeness crab	Whole organism, not exoskeleton or shell	Dry	6/23/2008	1.4 J
Dungeness crab	Muscle	Dry	6/23/2008	0.67 J
Starry flounder	Fillet, skin off	Dry	6/18/2008	0.02 U
Bent-nosed clam	Whole organism, not exoskeleton or shell	Dry	6/17/2008	0.3
Data Qualifier Definitions: J: Analyte was positively identified. The reported result is an estimate. U: Analyte was not detected at or above the reported result.				

4.3.2.13 Zinc

Zinc has been measured at quantifiable concentrations in water, animal and plant tissue, and marine sediment in or near the action area.

Summary statistics for zinc in marine sediment and freshwater are listed in Table 51. All zinc results for water were collected as part of a contaminated site investigation at the March Point Landfill, and thus may not be representative of zinc concentrations elsewhere in the action area.

At the minimum observed hardness of 30.9 mg/L as CaCO3, the acute and chronic water quality criteria for dissolved zinc in freshwater are 43.3 µg/L and 43.7 µg/L, respectively. All results for dissolved zinc in freshwater are below the water quality criteria, and only one of the 22 results was quantifiable.

None of the results for zinc in marine sediment exceed the marine sediment criterion from the Washington SMS, which is 410 mg/kg.

Table 51. Summary of Results for Zinc in Marine Sediment and Freshwater

Medium	Earliest Field Collection Start Date	Latest Field Collection Start Date	Minimum Result Value	Average Result Value	Maximum Result Value	Standard Deviation of Result Values	Count of Result Values	Detection Frequency (%)
Marine Sediment	7/11/1988	9/12/2017	20.1 mg/kg	64.1 mg/kg	199 mg/kg[1]	28.2 mg/kg	131	99
Freshwater (total)	10/14/2008	7/23/2009	10 µg/L U	21 µg/L	150 µg/L	30 µg/L	22	36
Freshwater (dissolved)	10/14/2008	7/23/2009	10 µg/L U	12 µg/L	20 µg/L	3.9 µg/L	22	4.5
Notes: 1. There was one non-detect result in the data set, and the reporting limit for this result was 400 mg/kg. Since this reporting level is higher than the highest quantifiable result (199 mg/kg), it was not listed as the maximum result in the table. Data Qualifier Definitions: U: Analyte was not detected at or above the reported result.								

Results for zinc in animal tissue are listed in Table 52. The concentration of zinc in dungeness crab muscle measured on June 1, 2012 exceeds the TSC of 35.6 mg/kg (wet weight). Estimating a wet basis

tissue concentration by dividing the dry basis tissue concentrations by a factor of 5 (i.e., assuming wet tissue is 80% water), the dry weight concentration of zinc in dungeness crab muscle tissue measured on June 23, 2008 also exceeds the TSCs. Otherwise, tissue concentrations are below the TSC.

Table 52. Results for Zinc in Animal Tissue

Sample Taxon Common Name	Tissue Type	Basis	Field Collection Start Date	Result Value (mg/kg)
Dungeness crab	Muscle	Dry	6/23/2008	201 J
Dungeness crab	Muscle	Wet	6/1/2012	49.6
Dungeness crab	Whole organism, not exoskeleton or shell	Dry	6/23/2008	120 J
Foolish mussel	Whole organism, not exoskeleton or shell	Wet	11/13/2012	13.9
Starry flounder	Fillet, skin off	Dry	6/18/2008	48.9 J
Bent-nosed clam	Whole organism, not exoskeleton or shell	Dry	6/17/2008	72.4 J
Data Qualifier Definitions: J: Analyte was positively identified. The reported result is an estimate.				

Results for zinc in eelgrass tissue are summarized in Table 53. The zinc concentration in eelgrass tissues ranges from 56 mg/kg to 97 m/kg in leaves. Converting this range to wet weigh to compare to the 35.6 mg/kg (ww) TSC results in a range of 11.2 mg/kg to 19.4 mg/kg, both are below the TSC.

Table 53. Summary of Results for Zinc in Eelgrass Tissue

Tissue Type	Basis	Field Collection Start Date	Minimum Result Value (µg/g)	Average Result Value (µg/g)	Maximum Result Value (µg/g)	Standard Deviation of Result Values (µg/g)	Count of Result Values	Detection Frequency (%)
Leaves	Dry	1/8/2013	56	71	97	23	3	100
Roots, whole	Dry	1/8/2013	33.9	36.6	38.9	2.5	3	100

4.3.3 Baseline concentrations and sources of organic pollutants in and near the action area

4.3.3.1 Organic chemicals with partial or total use bans

Section 5.3 lists the chemicals for which the Swinomish Tribe has adopted aquatic life water quality criteria that have partial or total use bans in place.

Ambient data for these chemicals are of interest because the effects analysis for these chemicals assumes that exposure of ESA-listed species to these chemicals is negligible, based on available information, and will decline over time (see discussion in section 5.3).

Among the organic chemicals with partial or total use bans, the following have been measured at quantifiable concentrations in or near the action area (see detection frequency Table 54 for context).

- 4,4'-DDD (n = 5, 3 in tissue, 1 in sediment, 1 in water)
- 4,4'-DDE (n = 15, 14 in tissue, 1 in sediment)
- 4,4'-DDT (n = 3, all in tissue)

- Aldrin (n = 1, in sediment)
- Dieldrin (n = 2, 1 each in sediment and tissue)
- Lindane (n = 2, in water)
- Oxychlordane (n = 2, in tissue)
- PCBs (n = 19, 13 in tissue and 6 in sediment)
- Total Chlordane (n = 1, in sediment)
- Trans-Chlordane (n = 1, in sediment)
- Tributyltin Chloride (n = 1, in tissue)
- Tributyltin Ion (n = 1, in sediment)

The quantifiable results for banned organic chemicals in marine sediment are listed in Table 54.

Table 54. Quantifiable Results for Banned Chemicals in Marine Sediment (Dry Weight Basis)

Parameter	Overall Detection Frequency (%)	Individual Results					
		Field Collection Start Date	Result Value (µg/kg)	Result Data Qualifier	Sediment Criterion or Screening Benchmark (µg/kg)	Latitude (NAD83)	Longitude (NAD83)
4,4'-DDD	2.6	6/17/2014	1.3	J	310	48.400488	-122.494438
4,4'-DDE	2.6	2/23/1993	1.1		21	48.400014	-122.494864
Aldrin	2.3	2/23/1993	0.91		2	48.400014	-122.494864
Dieldrin	2.3	6/2/1994	1		4.9	48.36938	-122.51823
PCBs, ∑ Aroclors	19	6/16/2014	9.6		133[1]	48.399307	-122.494022
PCBs, ∑ Aroclors		9/12/2017	23		19[1]	48.36411592	-122.5431932
PCBs, ∑ Aroclors		9/12/2017	23		7.4[1]	48.40127463	-122.496896
PCBs, ∑ Aroclors		9/12/2017	23		20[1]	48.45160144	-122.5142542
PCBs, ∑ Aroclors		9/12/2017	24		29[1]	48.48390126	-122.5351189
PCBs, ∑ congeners	100	8/3/2008	0.103	J	122	48.38345396	-122.5744492
Total Chlordane	3.7	6/19/2014	2.5	J	2.26	48.381771	-122.512269
trans-Chlordane	6.3	6/19/2014	2.5		2.26	48.381771	-122.512269
Tributyltin Ion	17	8/30/2007	0.48	T	47	48.5006	-122.562

Data qualifier definitions:
J: Analyte was positively identified. The reported result is an estimate.
T: Reported result below associated quantitation limit but above method detection limit (MDL).

Notes:
1. The marine sediment criterion for PCBs is 12 mg/kg, expressed as the sum of aroclors and normalized to the organic carbon content of the sediment.

Among the banned chemicals detected in sediment in or near the action area, the State of Washington's SMS establish sediment criteria for PCBs. The SMS also include cleanup objectives for total DDDs, total DDEs, dieldrin, and tributyltin in freshwater sediment[15], but do not establish criteria for these chemicals in marine sediment. The marine sediment criteria and freshwater sediment cleanup objectives represent a no adverse effects level to the benthic community (Ecology, 2013). The Washington SMS do not establish criteria for aldrin or chlordane. For these chemicals, the measured data was compared to EPA Region 3's sediment screening benchmarks.

Measured concentrations are less than the sediment criteria or screening benchmarks for DDD, DDE, aldrin, dieldrin, and tributyltin. The maximum concentrations of total chlordane and trans-chlordane exceed the chlordane screening benchmark by 10%. However, detection frequencies were low (4% - 6%). PCBs (as the sum of aroclors) were quantifiable in five samples, three of which exceed the sediment criterion, and the overall detection frequency was 19% (5 of 27).

The quantifiable results for banned organic chemicals in animal tissue are listed in Table 55.

Table 55. Quantifiable Results for Banned Chemicals in Animal Tissue (wet basis)

Parameter	Field Collection Start Date	Taxon Common/scientific Name	Result Value (µg/kg)	Tissue Screening Conc. (µg/kg)	Detection Frequency (%)	Latitude (NAD83)	Longitude (NAD83)
4,4' DDD	6/1/1999	Pacific oyster *Crassostrea gigas*	0.27	54	20	48.42690221	-122.5014381
4,4' DDD	6/1/2012	Dungeness crab *Metacarcinus magister*	0.25			48.41035	-122.57871
4,4' DDD	6/19/2013	Chinook salmon *Oncorhynchus tshawytscha*	0.17			48.40944	-122.58164
4,4' DDE	5/26/1999	Dungeness crab *M. magister*	0.22	54	93	48.50512235	-122.5554928
4,4' DDE	6/1/1999	Pacific oyster *C. gigas*	0.65			48.42690221	-122.5014381
4,4' DDE	6/1/2012	Dungeness crab *M. magister*	0.27			48.41035	-122.57871
4,4' DDE	11/13/2012	foolish mussel *Mytilus trossulus*	0.37			48.49955	-122.56746
4,4' DDE	5/30/2013	Chinook salmon *O. tshawytscha*	1.3			48.36354	-122.48728

[15] The total dichlorodiphenyldichloroethane (DDDs) criterion represents the sum of o,p'-DDD and p,p'-DDD, and the total dichlorodiphenyldichloroethylene (DDEs) criterion in Table VI of the Washington Sediment Management Standards (WAC 173-204-563) represents the sum of o,p'-DDE and p,p'-DDE. Only p,p' DDD and p,p' DDE have been measured at quantifiable concentrations in marine sediment in or near the action area.

Parameter	Field Collection Start Date	Taxon Common/scientific Name	Result Value (µg/kg)	Tissue Screening Conc. (µg/kg)	Detection Frequency (%)	Latitude (NAD83)	Longitude (NAD83)
4,4' DDE	5/30/2013	Chinook salmon, *O. tshawytscha*	1.9			48.36354	-122.48728
4,4' DDE	5/30/2013	Chinook salmon, *O. tshawytscha*	2	54	93	48.36354	-122.48728
4,4' DDE	5/30/2013	Chinook salmon, *O. tshawytscha*	2.1			48.36354	-122.48728
4,4' DDE	6/19/2013	Chinook salmon, king salmon	1.3			48.40944	-122.58164
4,4' DDE	6/19/2013	Chinook salmon, *O. tshawytscha*	1.4			48.40944	-122.58164
4,4' DDE	6/19/2013	Chinook salmon, *O. tshawytscha*	1.1			48.40944	-122.58164
4,4' DDE	6/19/2013	Chinook salmon, *O. tshawytscha*	1.2			48.40944	-122.58164
4,4' DDE	6/19/2013	Chinook salmon, *O. tshawytscha*	1			48.40944	-122.58164
4,4' DDT	6/1/1999	Pacific oyster *C. gigas*	0.2	54	20	48.42690221	-122.5014381
4,4' DDT	5/30/2013	Chinook salmon, *O. tshawytscha*	0.16			48.36354	-122.48728
4,4' DDT	5/30/2013	Chinook salmon, *O. tshawytscha*	0.2			48.36354	-122.48728
Dieldrin	6/19/2013	Chinook salmon, *O. tshawytscha*	0.15	8.9	6.7	48.40944	-122.58164
Oxychlordane	6/19/2013	Chinook salmon, *O. tshawytscha*	0.24	56[1]	4.4[2]	48.40944	-122.58164
Oxychlordane	6/1/2012	Dungeness crab *M. magister*	0.56			48.41035	-122.57871
PCBs, ∑ congeners	11/13/2012	Foolish mussel *M. trossulus*	0.59	440	100	48.49955	-122.56746
PCBs, ∑ congeners	6/19/2013	Chinook salmon, *O. tshawytscha*	5.73	440	100	48.40944	-122.58164
PCBs, ∑ congeners	6/19/2013	Chinook salmon, *O. tshawytscha*	7.67	440	100	48.40944	-122.58164
PCBs, ∑ congeners	6/19/2013	Chinook salmon, *O. tshawytscha*	7.22	440	100	48.40944	-122.58164
PCBs, ∑ congeners	6/19/2013	Chinook salmon, *O. tshawytscha*	3.59	440	100	48.40944	-122.58164
PCBs, ∑ congeners	6/19/2013	Chinook salmon, *O. tshawytscha*	6.22	440	100	48.40944	-122.58164
PCBs, ∑ congeners	5/30/2013	Chinook salmon, *O. tshawytscha*	6.47	440	100	48.36354	-122.48728
PCBs, ∑ congeners	5/30/2013	Chinook salmon, *O. tshawytscha*	6.515	440	100	48.36354	-122.48728
PCBs, ∑ congeners	5/30/2013	Chinook salmon, *O. tshawytscha*	5.82	440	100	48.36354	-122.48728
PCBs, ∑ congeners	5/30/2013	Chinook salmon, *O. tshawytscha*	6.46	440	100	48.36354	-122.48728

Parameter	Field Collection Start Date	Taxon Common/scientific Name	Result Value (µg/kg)	Tissue Screening Conc. (µg/kg)	Detection Frequency (%)	Latitude (NAD83)	Longitude (NAD83)
PCBs, ∑ congeners	6/1/2012	Dungeness crab *M. magister*	0.58	440	100	48.41035	-122.57871
PCBs, ∑ congeners	6/1/1999	Pacific Littleneck Clam *Protothaca staminea*	2.3	440	29	48.496944	-122.554444
PCBs, ∑ congeners	6/1/1999	Pacific oyster *C. gigas*	3	440	29	48.42690221	-122.5014381
Tributyltin Chloride	6/1/1999	Pacific oyster *C. gigas*	7.9	5.1	33	48.42690221	-122.50144

Notes:
1. The is the TSC for Chlordane
2. The detection frequency for the oxychlordane results is based on the number of measurements for all forms of chlordane.

As shown in Table 55, the quantifiable concentrations of all banned chemicals are below the TSCs, except for tributyltin chloride.

The only banned chemical measured in quantifiable concentrations in water in or near the action area was lindane. The maximum concentration is less than 0.3% of the applicable chronic water quality criterion. Both quantifiable results were collected as part of a contaminated site investigation at the March Point Landfill and thus may not be representative of the rest of the action area. Results are listed in Table 56.

Table 56. Quantifiable Results for Banned Chemicals in Freshwater

Parameter	Field Collection Start Date	Result Value (µg/L)	Water Quality Criterion (µg/L)	Detection Frequency (%)	Latitude (NAD83)	Longitude (NAD83)
Lindane	4/13/2010	0.0017	0.95[1]	5.4	48.46175123	-122.5294172
	4/13/2010	0.0026			48.46109702	-122.5280528

Notes:
1. The listed results for lindane were measured in freshwater, thus, the freshwater acute criterion is listed here. However, the measured results are less than all of the Tribe's water quality criteria for lindane (including human health criteria).

Banned chemicals which were sampled for in or near the action area but never detected are listed in Table 57. Most of the data in Table 57 were obtained from EIM, except for one non-detect result in marine sediment in NWIS, collected on September 18, 2006, for each of the following chemicals: alpha-endosulfan, heptachlor, heptachlor epoxide, methoxychlor, mirex, parathion, and toxaphene.

Table 57. Banned Chemicals Analyzed for but Not Detected

Parameter	Medium	Number of Measure-ments	Maximum Reporting Limit	Maximum Reporting Limit Units	Earliest Field Collection Date	Latest Field Collection Date
Alpha-endosulfan	Animal tissue	15	0.27	µg/kg	5/26/1999	6/19/2013
	Plant Tissue	6	0.055		1/8/2013	1/8/2013
	Marine sediment	7	5		6/4/1997	8/3/2008
	Freshwater	37	0.042	µg/L	10/14/2008	10/6/2010
Beta-endosulfan	Animal tissue	3	0.26	µg/kg	5/26/1999	6/1/1999
	Marine sediment	6	5		6/4/1997	8/3/2008
	Freshwater	37	0.083	µg/L	10/14/2008	10/6/2010
Demeton-O	Freshwater	1	0.05	µg/L	6/14/1992	6/14/1992
Demeton-S	Freshwater	1	0.05	µg/L	6/14/1992	6/14/1992
Endosulfan	Marine sediment	2	0.052	µg/kg	7/15/2002	7/16/2002
Endosulfan sulfate	Animal tissue	3	0.26	µg/kg	5/26/1999	6/1/1999
	Marine sediment	4	5	µg/kg	6/4/1997	8/3/2008
	Freshwater	37	0.083	µg/L	10/14/2008	10/6/2010
Heptachlor	Animal tissue	15	0.27	µg/kg	5/26/1999	6/19/2013
	Plant tissue	6	0.055		1/8/2013	1/8/2013
	Marine sediment	44	5	µg/kg	7/11/1988	9/12/2017
	Freshwater	37	0.042	µg/L	10/14/2008	10/6/2010
Heptachlor epoxide	Animal tissue	15	0.27	µg/kg	5/26/1999	6/19/2013
	Plant tissue	6	0.055	µg/kg	1/8/2013	1/8/2013
	Marine sediment	7	5	µg/kg	6/4/1997	8/3/2008
	Freshwater	37	0.16	µg/L	10/14/2008	10/6/2010
Methoxychlor	Animal tissue	3	0.26	µg/kg	5/26/1999	6/1/1999
	Marine sediment	4	5	µg/kg	6/4/1997	8/3/2008
	Freshwater	37	0.42	µg/L	10/14/2008	10/6/2010
Methyl parathion	Animal tissue	3	1.3	µg/kg	5/26/1999	6/1/1999
Mirex	Animal tissue	15	0.32	µg/kg	5/26/1999	6/19/2013
	Marine sediment	7	5	µg/kg	6/4/1997	8/3/2008
Parathion	Animal tissue	3	1.3	µg/kg	5/26/1999	6/1/1999
Toxaphene	Animal tissue	3	13	µg/kg	5/26/1999	6/1/1999
	Marine sediment	6	200	µg/kg	6/4/1997	8/3/2008
	Freshwater	37	4.2	µg/kg	10/14/2008	10/6/2010
Tributyltin	Marine porewater	7	0.02	µg/L	3/21/2001	6/18/2014

4.3.3.2 Current use organic chemicals

In general, the organic chemicals that are regulated by the Tribe's aquatic life water quality criteria and which are not subject to use bans were either not measured in or near the action area or were measured but not detected.

EPA could not locate any data for acrolein in water, sediment, or aquatic animal or plant tissue in or near the action area. Data for malathion were not available for the action area, however, monitoring stations on Indian Slough and Sullivan Slough, somewhat to the west of the action area, had data for malathion in sediment and tissue (Figure 14). Malathion data for these stations are included here because these streams could contribute pollutants to the action area.

Current use organic chemicals regulated by the Tribe's aquatic life water quality criteria which were analyzed in or near the action area but not detected are listed in Table 58.

Table 58. Current Use Organic Chemicals that were Analyzed but Not Detected

Parameter	Medium	Number of Measure-ments	Maximum Reporting Limit	Maximum Reporting Limit Units	Earliest Field Collection Date	Latest Field Collection Date
Chlorpyrifos	Animal tissue	3	0.64	µg/kg	5/26/1999	6/1/1999
	Marine sediment	1	0.66		6/4/1997	6/4/1997
Diazinon	Animal tissue	3	64	µg/kg	5/26/1999	6/1/1999
4-nonylphenol	Marine sediment	2	28	µg/kg	6/4/1997	6/5/2007
Pentachlorophenol	Freshwater	22	5	µg/L	10/14/2008	7/23/2009
	Marine sediment	63	34,000	µg/kg	7/11/1988	9/12/2017

Carbaryl and malathion are the only current-use organic chemicals regulated by the Tribe's water quality criteria that were detected in or near the action area at a quantifiable concentration.

Both quantifiable results for carbaryl were collected as part of a contaminated site investigation at the March Point Landfill and thus may not be representative of the rest of the action area. Both results are below the applicable water quality criteria. Results are listed in Table 59.

For malathion, only one result out of 279 measurements was quantifiable. However, the single quantifiable result, which was measured in Indian Slough on March 25, 2009, does exceed the Tribe's water quality criterion of 0.1 µg/L. Results are summarized in Table 60.

There was a single non-detect result for malathion in freshwater sediment in Sullivan Slough collected on October 1, 1992.

Table 59. Results for Carbaryl in Water

Field Collection Start Date	Result Value (µg/L)	Water Quality Criterion (µg/L)	Detection Frequency (%)	Latitude (NAD83)	Longitude (NAD83)
12/17/2008	1.8	2.1	9.1	48.46446038	-122.5304103
4/28/2009	1.2				
Notes: 1. The listed results for carbaryl were measured in freshwater, thus, the freshwater chronic criterion is listed here. The marine chronic criterion is 1.6 µg/L.					

Table 60. Summary of Results for Malathion in Water (Indian Slough and Sullivan Slough)

Earliest Field Collection Start Date	Latest Field Collection Start Date	Min. Result Value (µg/L)	Avg. Result Value (µg/L)	Max. Result Value (µg/L)	Std. Dev. of Result Value (µg/L)	Count of Result Value	Detection Frequency (%)
3/2/2006	9/9/2014	0.007	0.034	0.9	0.053	279	0.36

4.3.4 Baseline concentrations and sources of nonmetallic inorganic chemicals in Swinomish tribal waters

4.3.4.1 Ammonia

Ammonia has been measured at quantifiable concentrations in water and sediment in or near the action area.

A summary of the available data for ammonia in water is provided in Table 61.

Table 61. Summary of Results for Ammonia (Total as Nitrogen) in Surface Water

Monitoring Location Number or Code	Monitoring Location Name	Earliest Field Collection Start Date	Latest Field Collection Start Date	Min. Result Value (mg/L)	Avg. Result Value (mg/L)	Max. Result Value (mg/L)	Standard Deviation (mg/L)	Count
SCCBR (Skagit County)	Swinomish Channel at Skagit Boat Ramp	3/23/2009	9/23/2010	0.05	0.06	0.07	0.008	7
SSLW (Skagit County)	Sullivan Slough at La Conner Whitney Road	3/23/2009	9/23/2010	0.05	0.66	1.61	0.71	7
12200704 (USGS)	Unnamed Tributary No. 2 to Skagit Bay near La Conner, WA	11/20/1975	12/16/1976	0.06	0.13	0.40	0.14	6

Monitoring Location Number or Code	Monitoring Location Name	Earliest Field Collection Start Date	Latest Field Collection Start Date	Min. Result Value (mg/L)	Avg. Result Value (mg/L)	Max. Result Value (mg/L)	Standard Deviation (mg/L)	Count
12200704 (USGS)	Unnamed Tributary to Swinomish Channel near La Conner, WA	11/20/1975	12/16/1976	0.02	0.05	0.12	0.04	6
SWINOMISH-FOR2 (Swinomish Tribe)	Fornsby Creek	2013	10/13/2016	0	0.03	0.12	0.03	17
SWINOMISH-LON10 (Swinomish Tribe)	Lone Tree Creek	2013	10/13/2016	0	0.05	0.13	0.05	11
SWINOMISH-MUN2 (Swinomish Tribe)	Munks Creek	2013	10/13/2016	0	0.02	0.09	0.03	17
SWINOMISH-SKR1 (Swinomish Tribe)	Skagit River	2013	10/13/2016	0	0.04	0.11	0.03	16
SWINOMISH-SNE2 (Swinomish Tribe)	Snee-Oosh Creek	2013	10/13/2016	0	0.07	0.18	0.05	17

The ammonia criteria for freshwater vary based on the pH and temperature of the receiving water, with the criteria becoming more stringent with increasing pH and temperature. The acute and chronic criteria for ammonia, at the maximum pH and temperature observed at the monitoring stations listed in Table 61, are listed in Table 62 (Figure 14).

Table 62. Ammonia Criteria at Maximum pH and Temperature at Ammonia Monitoring Stations

Monitoring Location Number or Code	Monitoring Location Name	Maximum Temp. (°C)	Maximum pH (s.u.)	Acute Criterion (mg/L)	Chronic Criterion (mg/L)
SCCBR	Swinomish Channel at Skagit Boat Ramp (marine water)	16	8.02	6.69	1.00
SSLW	Sullivan Slough at La Conner Whitney Road	20.8	8.89	0.69	0.17
12200704	Unnamed Tributary No. 2 to Skagit Bay near La Conner, WA	11.5	7.6	11.37	2.20
12200704	Unnamed Tributary to Swinomish Channel near La Conner, WA	10.8	8	5.62	1.41
SWINOMISH-FOR2	Fornsby Creek	14.26	8.5	2.14	0.50

Monitoring Location Number or Code	Monitoring Location Name	Maximum Temp. (°C)	Maximum pH (s.u.)	Acute Criterion (mg/L)	Chronic Criterion (mg/L)
SWINOMISH-LON10	Lone Tree Creek	16.78	7.99	5.20	0.97
SWINOMISH-MUN2	Munks Creek	17.92	8.62	1.41	0.32
SWINOMISH-SKR1	Skagit River	20.41	8.54	1.33	0.32
SWINOMISH-SNE2	Snee-Oosh Creek	13.7	8.45	2.35	0.57

The maximum ammonia concentrations observed at the stations listed in Table 61 and Table 62 are less than the water quality criteria (evaluated at the worst-case observed pH and temperature), except in Sullivan Slough. Although this location is outside of the action area, it is included here because Sullivan Slough is a tributary to the Swinomish Channel and water quality in Sullivan Slough could thereby affect water quality within the action area.

Results for ammonia in marine sediment are summarized in Table 63. None of the results for ammonia in marine sediment exceed the freshwater sediment cleanup objective in the Washington SMS, which is 230 mg/kg (no criterion is established for marine sediment).

Table 63. Summary of Results for Ammonia in Marine Sediment

Earliest Field Collection Start Date	Latest Field Collection Start Date	Minimum Result Value (mg/kg)	Average Result Value (mg/kg)	Maximum Result Value (mg/kg)	Standard Deviation of Result Values (mg/kg)	Count of Result Values	Detection Frequency (%)
7/11/1988	9/12/2017	0.05	34.1	124	32.6	79	100

4.3.4.2 Chloride

The only available data for chloride in surface freshwater in the action area were collected by the USGS in 1976. Results are summarized in Table 64. All results are well below the acute and chronic water quality criteria (860 mg/L and 230 mg/L, respectively).

Table 64. Summary of Results for Chloride in Freshwater

Station Number	Earliest Sample Date	Latest Sample Date	Minimum Result Value (mg/L)	Average Result Value (mg/L)	Maximum Result Value (mg/L)	Standard Deviation of Result Values (mg/L)	Count of Result Values
12200704	3/2/1976	12/16/1976	14	18.2	21	2.6	5
12200706	3/2/1976	12/16/1976	11	13.6	15	1.7	5

4.3.4.3 Cyanide

The only available data for cyanide in or near the action area were four results for marine sediment collected near March Point in 1992. All results were non-detect, and reporting limits ranged from 0.17 – 0.18 ppm. The State of Washington has not established any sediment criteria or cleanup objectives for cyanide, however EPA Region 3 has established a freshwater sediment screening benchmark of 0.1 ppm. Because the reporting limits are higher than the screening benchmark, it is not clear of the cyanide concentrations are above or below the screening benchmark.

4.3.4.4 Elemental phosphorus

There are no available data for elemental phosphorus (not to be confused with phosphate phosphorus) in or near the action area.

4.3.4.5 Hydrogen sulfide

EPA could not locate any data for hydrogen sulfide in water in or near the action area.

Data are available for sulfide in marine sediment. Results are summarized in Table 65. There were an additional 16 results in EIM that had a data qualifier of "REJ," which means that the data are unusable for all purposes and the presence or absence of the analyte cannot be verified. These results are not included in the summary statistics in Table 65. Thirty six of the 68 (53%) results for sulfide in marine sediment exceed the freshwater sediment cleanup objective in the Washington SMS, which is 39 mg/kg (no criterion is established for marine sediment).

Table 65. Summary of Results for Sulfide in Marine Sediment

Earliest Field Collection Start Date	Latest Field Collection Start Date	Minimum Result Value (mg/kg)	Average Result Value (mg/kg)	Maximum Result Value (mg/kg)	Standard Deviation of Result Values (mg/kg)	Count of Result Values	Detection Frequency (%)
7/11/1988	9/12/2017	0.2	641	4080	947	68	78

4.3.4.6 Dissolved oxygen

Most of the available dissolved oxygen (DO) data were collected by the Swinomish Tribe. There were a total of 857 results for DO in marine water and 313 results for DO in freshwater available in the Water Quality Portal. Results are summarized in Figure 23 and Figure 24.

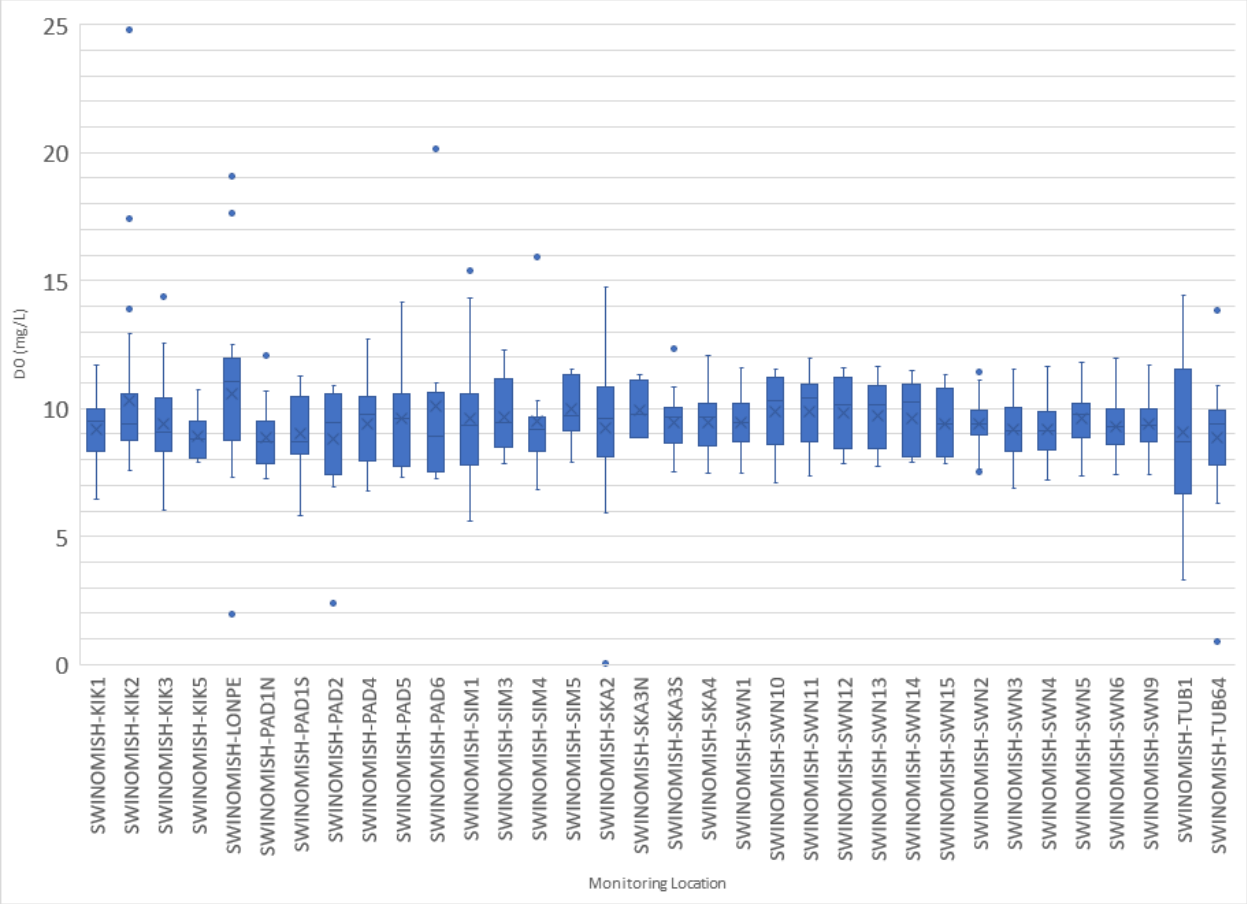


Figure 23: Marine DO data collected by the Swinomish Tribe

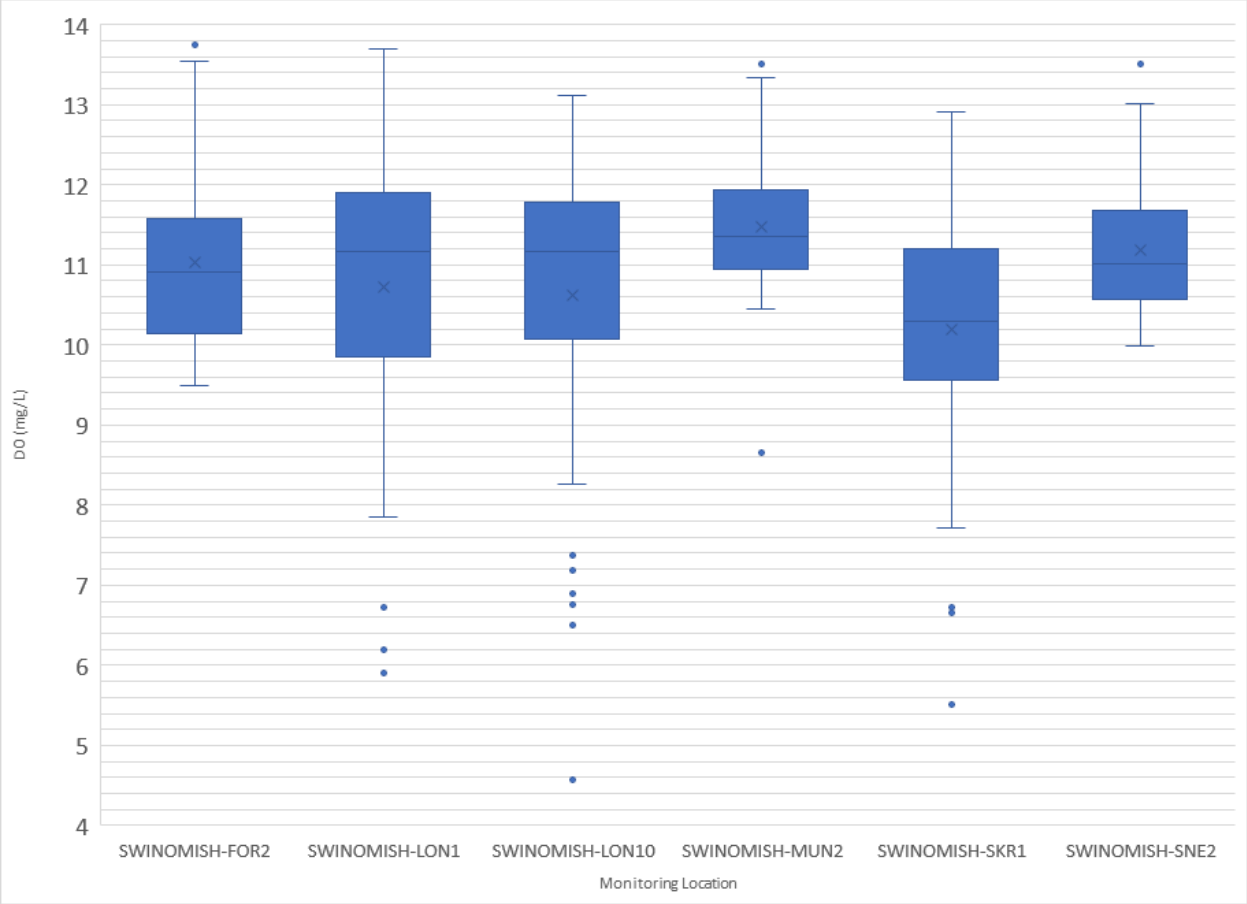


Figure 24: Freshwater DO data collected by the Swinomish Tribe

The lower quartile of DO concentrations is above the criterion of 6 mg/L at all marine stations, but a few stations have a minimum DO concentration below 6 mg/L (Figure 22). The lower quartile of DO concentrations is above the criterion of 9.5 mg/L at all freshwater stations, but four of the six stations have a minimum DO concentration below 9.5 mg/L (Figure 23).

Skagit County has established a monitoring station near the north end of the Swinomish Channel at the County's boat launch (Figure 14). The latitude and longitude are 48.4554° N and 122.5122 W, and 66 DO measurements are available from EIM and the County's website. Results are summarized as a box-and-whisker plot in Figure 24. DO at this location is consistently above the criterion of 6 mg/L.

Figure 25: Dissolved oxygen at Skagit County boat ramp (Swinomish Channel) collected by Skagit County

DO data are available from USGS NWIS, although most results are from the 1970s and early 1990s. Results are listed in Table 66. No exceedances of water quality criteria were observed.

Table 66. DO Data from USGS NWIS

Station Number	Station Name	Sample Date	DO (mg/L)
12200704	Unnamed Tributary No. 2 Skagit Bay near La Conner, WA	11/20/1975	12.0
		3/2/1976	12.2
		5/21/1976	11.9
		8/26/1976	10.4
		9/24/1976	10.2
		12/16/1976	10.8
12200706	Unnamed Tributary to Swinomish Channel near La Conner, WA	11/20/1975	12.5
		3/2/1976	12.6
		5/21/1976	11.8
		8/26/1976	11.1
		9/24/1976	10.5
		12/16/1976	11.1
482655122313201	Seep Site 1	5/21/1991	10.7
482658122312901	Seep Site 2	5/21/1991	10.6
482125122293501	Skagit Delta (old dist)	4/8/2008	12.3

4.3.4.7 pH

Most of the available pH data were collected by the Swinomish Tribe. There are a total of 903 results for pH in marine water and 318 results for pH in freshwater available in the Water Quality Portal. Results are summarized in Figure 26 and Figure 27.

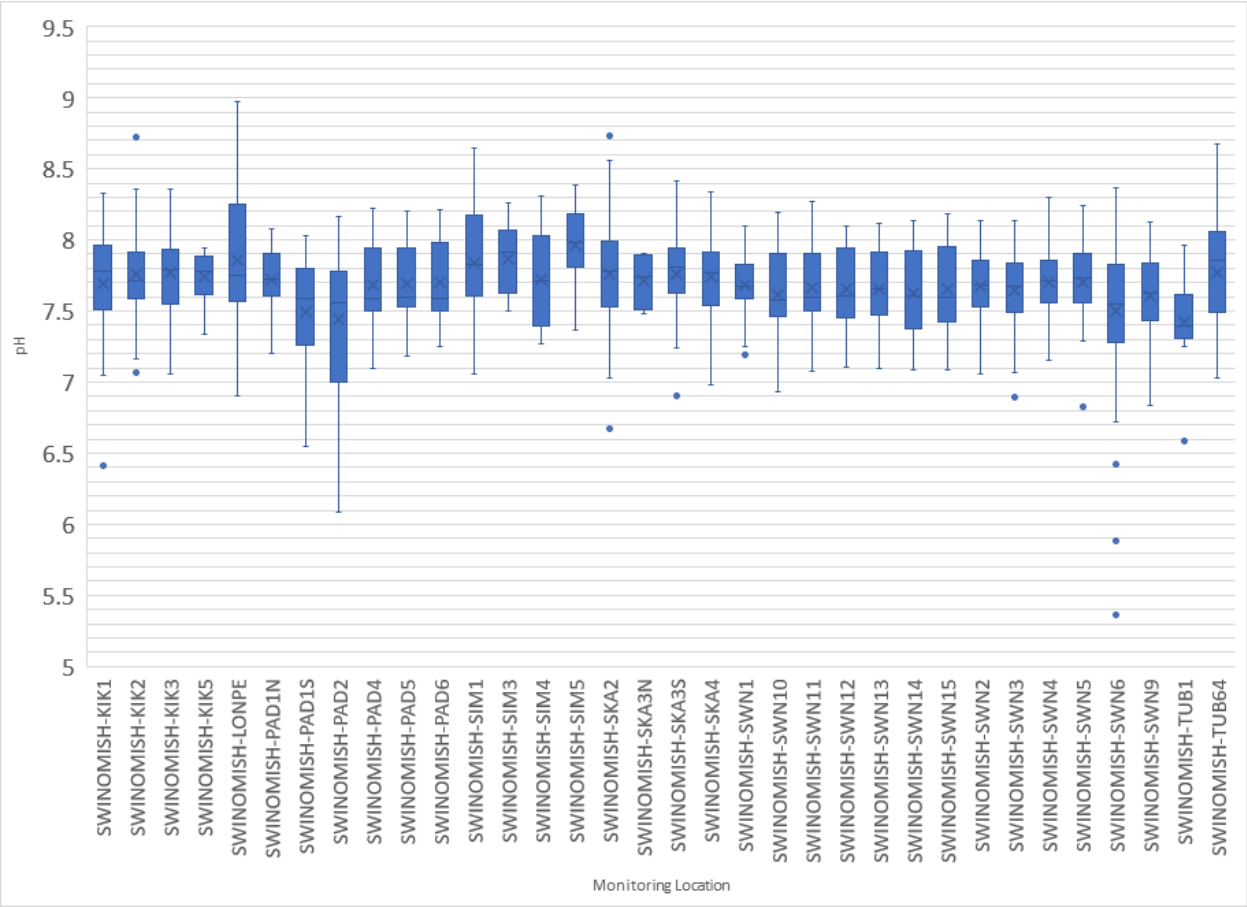


Figure 26: Marine water pH data collected by the Swinomish Tribe

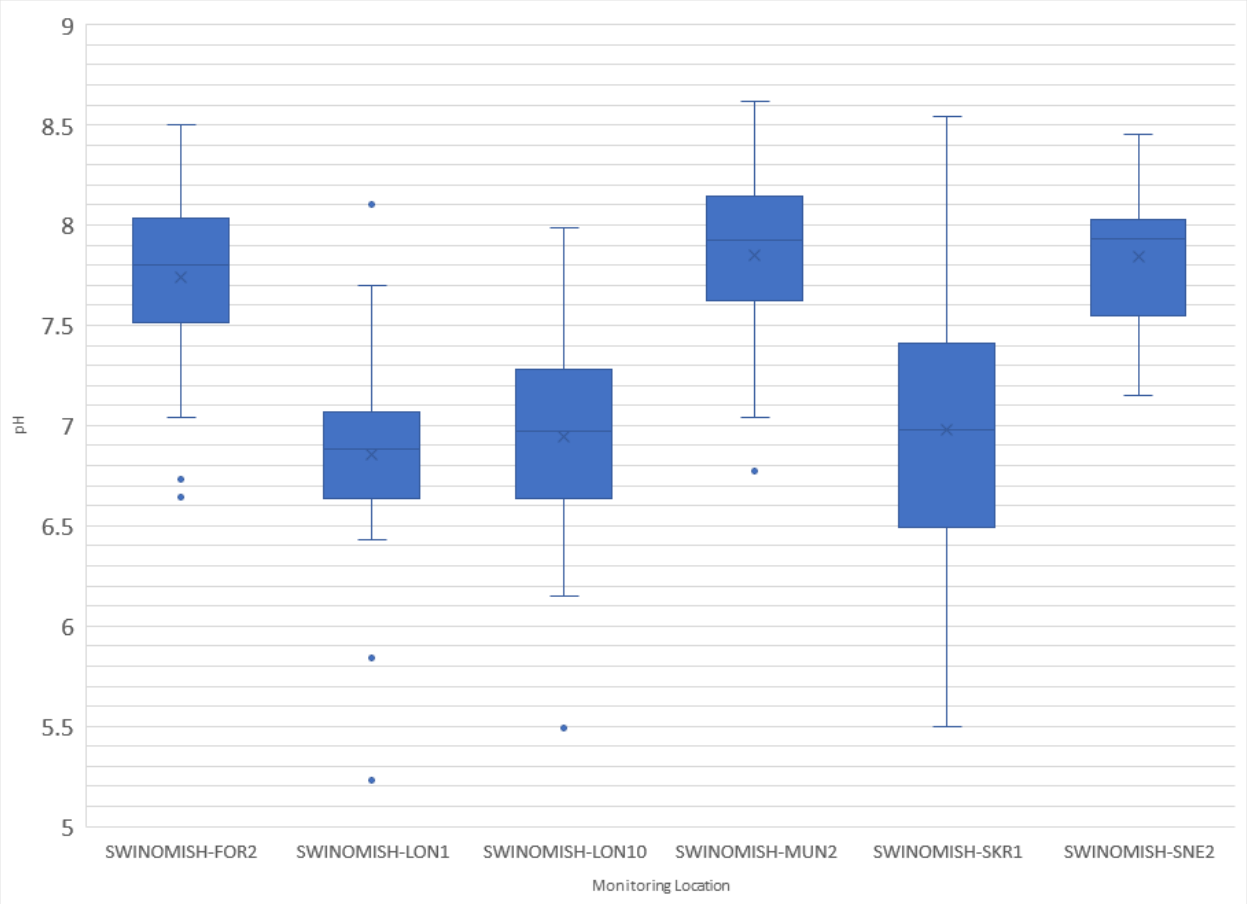


Figure 27: Freshwater pH data collected by the Swinomish Tribe

The interquartile range of the pH is within the water quality criterion (7 – 8.5) at all marine stations, but a few stations have excursions, generally for low pH (Figure 26). The interquartile range of the pH is within the water quality criterion (6.5 – 8.5) at all freshwater stations except for SKR1, where the 25th percentile pH is 6.49. Four of the six freshwater stations have at least one pH excursion (Figure 27).

Results for pH in water from EIM and USGS NWIS are summarized in Table 67.

Table 67. Summary of Results for pH in Water

Monitoring Location Number or Code	Monitoring Location Name	Earliest Field Collection Start Date	Latest Field Collection Start Date	Min. Result Value	Avg. Result Value	Max. Result Value	Standard Deviation	Count
SCCBR (Skagit County)	Swinomish Channel at Skagit Boat Ramp	12/16/2008	9/20/2011	6.95	7.65	8.02	0.24	63
WHITMARSH-2662-SW-01	—	10/14/2008	7/23/2009	6.83	6.91	6.97	0.07	3
WHITMARSH-2662-SW-03	—	10/15/2008	7/23/2009	6.66	7.22	7.62	0.44	4

Monitoring Location Number or Code	Monitoring Location Name	Earliest Field Collection Start Date	Latest Field Collection Start Date	Min. Result Value	Avg. Result Value	Max. Result Value	Standard Deviation	Count
WHITMARSH-2662-SW-04	—	10/15/2008	7/23/2009	6.76	7.16	7.60	0.39	4
WHITMARSH-2662-SW-05	—	10/15/2008	7/23/2009	6.71	7.56	8.83	0.93	4
WHITMARSH-2662-SW-06	—	10/15/2008	7/23/2009	6.76	7.37	8.22	0.65	4
WHITMARSH-2662-SW-07	—	12/17/2008	4/28/2009	6.75	6.76	6.76	0.01	2
12200704 (USGS)	Unnamed Tributary No. 2 to Skagit Bay near La Conner, WA	11/20/1975	5/21/1976	6.2	7.1	7.6	0.8	3
12200706 (USGS)	Unnamed Tributary to Swinomish Channel near La Conner, WA	11/20/1975	5/21/1976	7.5	7.8	8.0	0.3	3

4.3.5 Baseline physical water quality

4.3.5.1 Marine temperature

There were a total of 903 results for temperature in marine water available in the Water Quality Portal, with most provided by the Swinomish Tribe for reservation waters. Results are summarized in Figure 28.

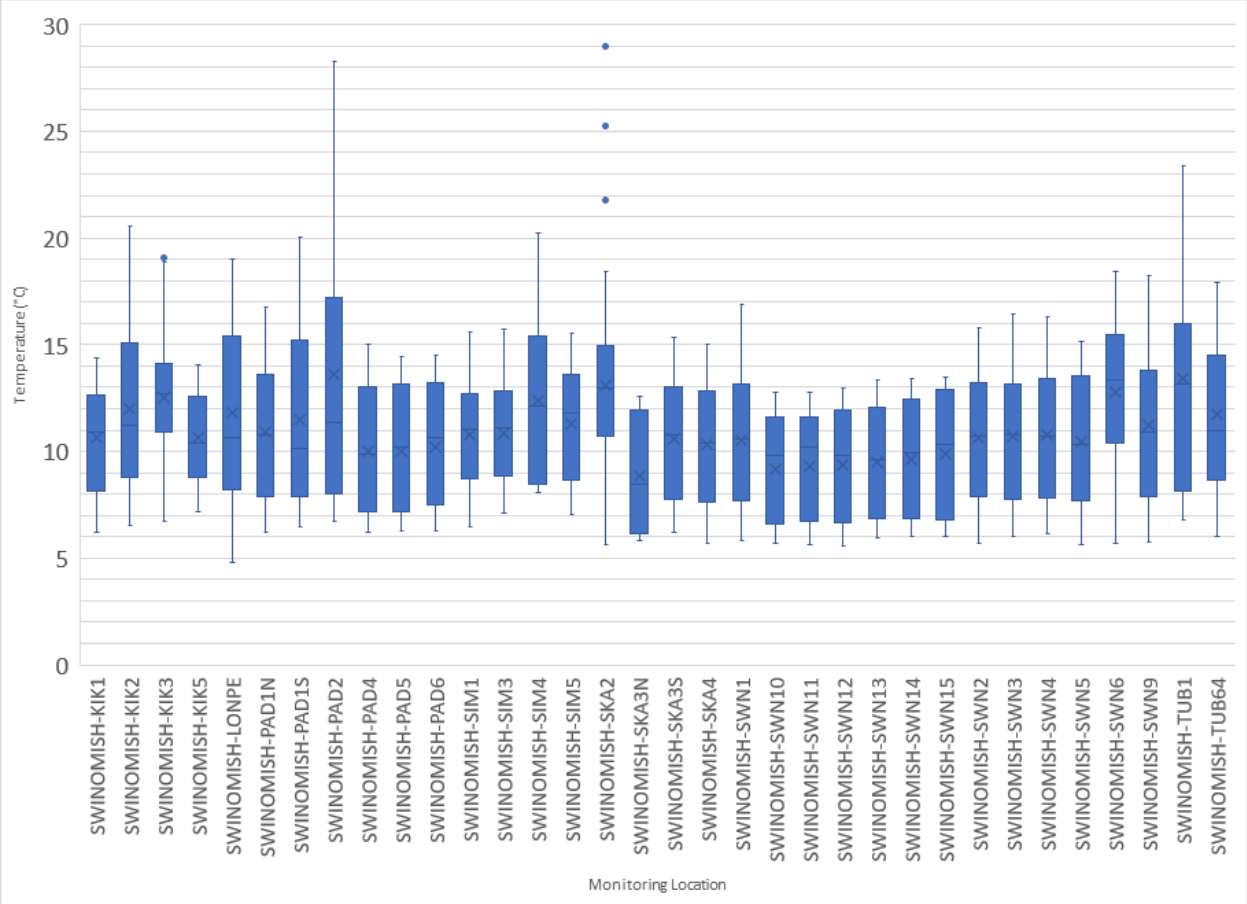


Figure 28: Marine temperature data collected by the Swinomish Tribe

The upper quartile of temperature is below the criterion of 16 °C at most marine stations (Figure 28).

A total of 67 temperature measurements are available from EIM and the County's website for Skagit County's monitoring station at the County's boat launch on the Swinomish Channel. Results are summarized as a box-and-whisker plot in Figure 28. Temperatures at this location are consistently at or below the criterion of 16 °C.

Figure 29: Temperature at Skagit County boat ramp (Swinomish Channel)

4.3.5.2 Freshwater temperature

Hourly temperature data for Fornsby, Lone Tree, Munks and Snee-Oosh Creeks were obtained from the Swinomish Tribe. Results are summarized in Table 68, Table 69, Table 70, and Table 71. The "difference" columns summarize the differences between the 7-DADMax temperatures and the 7-day average temperatures. Exceedances of the 16 °C 7-DADMax temperature criterion were observed only in Lone Tree Creek, which is an intermittent stream.

Table 68. Temperatures in Fornsby Creek

Month	Avg 7-DADMax (°C)	Max 7-DADMax (°C)	Avg 7-day Avg (°C)	Max 7-day Avg (°C)	Min Difference (°C)	Avg Difference (°C)	Max Difference (°C)	Std. Dev. of Difference (°C)	Count
Jan	5.607	7.995	5.222	7.616	0.200	0.385	0.653	0.092	118
Feb	5.798	9.432	5.398	9.263	0.169	0.399	0.736	0.114	113
Mar	6.816	10.894	6.306	10.131	0.313	0.510	0.853	0.113	165
Apr	9.362	11.844	8.748	11.176	0.278	0.614	1.082	0.195	159
May	11.347	12.613	10.783	12.004	0.357	0.563	0.912	0.126	186
Jun	12.549	14.825	12.026	14.116	0.288	0.523	0.812	0.131	180

Month	Avg 7-DADMax (°C)	Max 7-DADMax (°C)	Avg 7-day Avg (°C)	Max 7-day Avg (°C)	Min Difference (°C)	Avg Difference (°C)	Max Difference (°C)	Std. Dev. of Difference (°C)	Count
Jul	13.901	15.559	13.383	14.844	0.265	0.518	0.815	0.128	186
Aug	14.381	15.161	13.882	14.558	0.293	0.499	0.818	0.122	186
Sep	13.243	14.966	12.877	14.578	0.152	0.366	0.652	0.109	158
Oct	11.041	12.847	10.716	12.592	0.175	0.325	0.512	0.082	180
Nov	9.050	11.722	8.718	11.442	0.137	0.332	0.757	0.107	150
Dec	6.409	8.800	6.067	8.525	0.143	0.342	0.885	0.130	133

Table 69. Temperatures in Snee-Oosh Creek

Month	Avg 7-DADMax (°C)	Max 7-DADMax (°C)	Avg 7-day Avg (°C)	Max 7-day Avg (°C)	Min Difference (°C)	Avg Difference (°C)	Max Difference (°C)	Std. Dev. of Difference (°C)	Count
Jan	6.449	8.165	6.242	7.979	0.067	0.207	0.413	0.069	149
Feb	6.427	9.167	6.218	9.067	0.085	0.208	0.425	0.067	113
Mar	7.148	10.019	6.913	9.699	0.103	0.235	0.461	0.073	161
Apr	8.982	10.723	8.756	10.605	0.058	0.226	0.450	0.093	180
May	10.494	11.300	10.302	11.146	0.043	0.193	0.395	0.072	186
Jun	11.377	12.507	11.212	12.309	0.054	0.165	0.602	0.103	180
Jul	12.359	13.116	12.209	12.910	0.050	0.151	0.427	0.073	164
Aug	12.788	13.257	12.645	13.039	0.059	0.143	0.316	0.056	171
Sep	12.327	13.083	12.218	12.972	0.044	0.109	0.224	0.040	107
Oct	10.892	12.194	10.739	12.019	0.049	0.152	0.352	0.063	173
Nov	9.247	11.230	9.025	11.119	0.083	0.223	0.462	0.084	142
Dec	7.264	9.265	7.057	9.167	0.085	0.207	0.479	0.071	153

Table 70. Temperatures in Munks Creek

Month	Avg 7-DADMax (°C)	Max 7-DADMax (°C)	Avg 7-day Avg (°C)	Max 7-day Avg (°C)	Min Difference (°C)	Avg Difference (°C)	Max Difference (°C)	Std. Dev. of Difference (°C)	Count
Jan	6.335	8.261	6.050	7.982	0.105	0.285	0.415	0.063	160
Feb	6.338	9.194	6.030	9.047	0.134	0.307	0.695	0.114	105
Mar	7.091	9.999	6.763	9.669	0.189	0.328	0.506	0.066	164
Apr	9.043	11.227	8.730	10.923	0.146	0.314	0.599	0.084	173
May	10.712	11.543	10.419	11.307	0.132	0.293	0.622	0.103	186
Jun	11.682	13.274	11.423	12.995	0.129	0.259	0.418	0.067	180
Jul	12.742	13.904	12.506	13.671	0.145	0.236	0.498	0.065	186
Aug	13.156	13.757	12.919	13.508	0.165	0.237	0.358	0.041	170
Sep	12.545	13.986	12.329	13.420	0.120	0.216	0.565	0.099	109
Oct	10.882	12.098	10.667	11.951	0.107	0.216	0.440	0.066	155
Nov	8.990	11.112	8.712	10.984	0.119	0.277	0.506	0.088	154
Dec	6.917	9.100	6.618	8.855	0.129	0.299	0.768	0.116	152

Table 71. Temperatures in Lone Tree Creek

Month	Avg 7-DADMax (°C)	Max 7-DADMax (°C)	Avg 7-day Avg (°C)	Max 7-day Avg (°C)	Min Difference (°C)	Avg Difference (°C)	Max Difference (°C)	Std. Dev. of Difference (°C)	Count
Jan	5.579	7.850	5.250	7.510	0.143	0.329	0.542	0.073	159
Feb	5.689	9.080	5.213	8.778	0.218	0.476	0.912	0.165	124
Mar	7.683	12.472	6.540	10.363	0.329	1.143	3.473	0.713	172
Apr	11.347	17.241	9.658	12.348	0.521	1.688	5.780	1.189	180
May	13.618	16.920	12.062	13.509	0.680	1.556	3.814	0.595	138
Jun	15.071	17.130	13.566	15.345	0.941	1.505	2.545	0.389	70
Jul	16.146	17.423	14.582	15.744	1.276	1.564	1.894	0.144	31
Aug	17.399	17.737	15.835	16.178	1.386	1.564	1.719	0.105	12
Sep	No data								0
Oct	12.761	15.916	12.042	14.198	0.270	0.718	1.880	0.355	63
Nov	10.060	12.729	9.659	12.383	0.125	0.401	1.978	0.314	118
Dec	6.727	8.964	6.423	8.838	0.121	0.304	0.959	0.169	121

The Tulalip Tribes of Washington collected temperature data in Lone Tree Creek, on the Swinomish Reservation, on four days during the spring of 2013. The coordinates of the monitoring station are 48.4071° N and 122.551° W. These data were obtained from the Water Quality Portal, and comments on the data indicate that the data were collected during electrofishing. Multiple temperature measurements were reported for each day, however, the times that the temperatures were measured were not reported. Thus, only the maximum temperature observed during each day is reported in Table 72. No exceedances of temperature criteria were observed.

Table 72. Temperature in Lone Tree Creek, Spring 2013

Date	Maximum Temperature (°C)	Number of temperature measurements
3/21/2013	6.1	12
4/2/2013	9.2	10
4/18/2013	8.5	10
5/2/2013	11.7	10

Historical temperature data (mostly from the 1970s) for surface freshwater on the Swinomish Reservation is available from USGS NWIS. Results are listed in Table 73. No exceedances of water quality criteria were observed in the USGS NWIS data.

Table 73. Temperature Data from USGS NWIS

Station Number	Station Name	Sample Date	Temperature (°C)
12200704	Unnamed Tributary No. 2 Skagit Bay near La Conner, WA	3/2/1976	3.4
		5/21/1976	8.6
		8/26/1976	11.4
		9/24/1976	11.5

Station Number	Station Name	Sample Date	Temperature (°C)
		12/16/1976	8.8
12200706	Unnamed Tributary to Swinomish Channel near La Conner, WA	3/2/1976	3.5
		5/21/1976	9
		8/26/1976	10.6
		9/24/1976	10.8
		12/16/1976	9
482655122313201	Seep Site 1	5/21/1991	9.3
482658122312901	Seep Site 2	5/21/1991	9.7

4.4 Summary of the Environmental Baseline

The condition of salmonid habitat throughout the entire action area, Similk Bay notwithstanding, has been degraded due to human activities. The primary impacts are due to historic diking, dredging, shoreline modification and the resulting loss of wetland and saltmarsh habitat. These impacts have reduced the amount and condition of off-channel habitat, riparian and nearshore habitat that would otherwise provide refugia, feeding and rearing functions necessary to support healthy salmonid populations. Similk Bay, due primarily due to its location has not been impacted by agricultural activities, and although present, residential development is for the most part low-density. Therefore, shoreline modification is minimal and nearshore habitat appears to be intact. Past and ongoing restoration projects are improving habitat quality and creating and opening access to off-channel habitat, which improves the condition of the environmental baseline.

Except for the Swinomish Channel, eelgrass is present in the action area and although considered patchy, is nonetheless widespread. Eelgrass provides the substrate for herring spawning as does the shoreline of Similk Bay for sand lance and to a lesser degree surf smelt spawning. Forage fish are an important component of the marine food web and support salmonids.

There are five facilities permitted to discharge in the action area and all are within the Swinomish Channel. Because of the treatment technologies employed by some of these facilities, many of the pollutants are eliminated from the discharge. In addition to the permitted facilities, there are State and Federal cleanup sites near the action area. Remediation of these sites will continue to reduce the contribution of contamination to the action area.

The Swinomish Tribe's WQS for protection of aquatic life cover metals and metalloids, organic chemicals with total or partial use bans, and current use organic chemicals. Metals and metalloids have been measured in surface water, sediment and tissues in the action area during various monitoring programs. In most cases, these metals and metalloids were not detected in surface water; when they were, it was from the March Point Landfill, which is likely not representative of conditions in the entire action area.

Concentrations of the various metals and metalloids were compared to the Swinomish Tribal water quality criteria, Washington SMS and TSC, to determine if the concentrations represented a potential hazard to the aquatic community and informed the condition of the environmental baseline. Comparison to these standards and criteria demonstrated that most exceedances were in crab, oyster and clam tissue (arsenic, cadmium, chromium, copper, nickel, silver zinc), followed by sediment (cadmium, copper and nickel) and finally water (iron, mercury and silver).

A total of 12 organic chemicals with partial or total use bans have been detected in media in or near the action area; these include DDT and metabolites, pesticides, PCBs and tributyltin. The detections were primarily in tissues, followed by sediment and water (Lindane only). Five of these organic chemicals were detected below sediment standards or screening benchmarks, two were close to the benchmark with low detection frequencies. PCBs were present in some samples as well, and a few (three) exceeded sediment standards. None of the organic chemicals exceeded TSC except for tributyltin chloride which was outside the action area.

In general, the organic chemicals that are regulated by the Tribe's aquatic life water quality criteria and which are not subject to use bans were either not measured in or near the action area, or were measured, but not detected. Only carbaryl and malathion were detected in or near the action area, but at concentrations below the Swinomish Tribal Water Quality Criteria.

Finally, data for nonmetallic inorganic and conventional pollutants (e.g. dissolved oxygen, pH) were reviewed and compared to standards and benchmarks. Only hydrogen sulfide exceeded the Washington SMS.[16] In general, DO concentrations in marine and freshwater stations are acceptable however, there are a limited number of DO measurements below criteria. As with DO, pH data collected at both marine and freshwater stations generally meet criteria with a limited number of exceptions. Finally, water temperature also meets criteria at most marine stations and all freshwater stations, with some samples exceeding the marine temperature criteria.

Permitted discharges are limited in number and spatial extent in the action area to primarily the Swinomish Channel; the highest density of human development is in this area as well. Habitat degradation is more widespread than chemical contamination, with the latter primarily from metals and metalloids in tissues. Sources of pollution are from point and non-point sources including municipal discharges, agricultural runoff and septic and stormwater from municipal and industrial sources. Although chemicals have been detected in environmental media, exceedance of the various applicable standards is not widespread. The environmental baseline in a portion of the action area is considered degraded, primarily due to physical impacts. There is a chemical component to this degradation as well, which is not unexpected due to the ubiquitous nature of pollution in urban environments. However, based on the available data, the chemicals while present, appear to have less influence on the condition of the baseline than physical impacts.

[16] Washington Sediment Management Standards. Chapter 173-204 WAC. Available at https://www.epa.gov/sites/default/files/2014-12/documents/wa-chapter173-204.pdf

5 Effects of the Action

EPA's action includes the proposed approval the Swinomish Tribe's aquatic life criteria that include toxics criteria, conventional criteria (i.e., temperature, disolved oxygen, and pH), and narrative criteria. As discussed in section 1.5, approval of the Swinomish Tribe's WQS would have beneficial effects to listed species and critical habitat through application of the WQS in subsequent actions by the Tribe and EPA that are intended to reduce pollutant levels in tribal waters. This section includes effect determinations for each criterion based on a conservative assumption that the waters are at the criterion level and species are exposed to this concentration.

Effect determinations based on considerations specific to the action area for several species for all freshwater and/or marine criteria are covered in section 5.1. Effect determinations for the toxics criteria are covered in sections 5.2 through 5.8. The approach for the effects assessment for the toxics criteria is divided into 4 categories as described below:

Category 1 (section 5.2) – For these criteria, EPA incorporates toxicity assessments by reference for criteria with NLAA concurrences and/or with no jeopardy/adverse modification determinations by the Services in recent ESA consultations on Oregon and Idaho WQS for toxic chemicals.

Category 2 (section 5.2) – For these criteria, EPA incorporates toxicity assessments and Services' letters with no jeopardy/adverse modification ESA conclusions by reference for criteria adopted to meet RPAs in recent ESA consultations on Oregon and Idaho WQS for toxic chemicals.

Category 3 (section 5.3) – For these criteria, EPA relies on exposure assessments with current use bans in place to make NLAA determinations, similar to the approach taken for recent ESA consultations on Oregon and Idaho WQS for toxic chemicals.

Category 4 (sections 5.4 – 5.8) – For these criteria, EPA provides toxicity assessments in this BE with no or unknown EPA Region 10 WQS consultation history and for selected criteria that EPA chose to update the toxicity assessments from previous consultations.

Effects associated with the conventional criteria are covered in section 5.5 for freshwater criteria and in section 5.6 for marine criteria. Section 5.9 covers effects associated with the narrative criteria. Effects to critical habitat are covered in section 5.10. Cumulative effects are covered in section 5.11.

5.1 Determinations for SRKW, Humpback Whale, and Chum Salmon Based on Considerations Specific to the Action Area

Based on the location of the action area and the nature and extent of its fresh and marine waters, EPA is making NLAA determinations for its proposed approval of fresh water and marine toxics criteria for SRKW, marine toxics criteria for humpback whale, and freshwater criteria for chum. EPA also determined that its proposed approval of the Tribe's freshwater criteria will have no effect on humpback whales.

5.1.1 Not likely to adversely affect determinations for SRKW and chum salmon and no effect determination for humpback whale for all fresh water criteria

SRKW: As discussed in section 3.8.3, SRKW primarily consume Chinook salmon (>85% of their diet), but this proportion varies seasonally depending on the relative availability of Chinook compared to chum and coho (M. J. Ford et al., 2016; M Bradley Hanson et al., 2021). As Chinook availability seasonally declines, coho or chum are taken despite their lower (~1/3) caloric content than Chinook. Despite the shift to salmonids other than Chinook, Chinook abundance is essential and is known to be strongly correlated with SRKW vital rates (PFMC, 2020). Therefore, this assessment focuses on Chinook as a conservative approach. However, given similar sensitivity to chemicals across all salmonids (section 5.5) that may use Swinomish freshwaters, the conclusions also apply to other salmonids.

The Skagit, Snohomish, and Stillaguamish river systems, which discharge into Puget Sound in the vicinity of the action area, are among the river systems that provide spawning habitat for the salmonids that constitute prey species for SRKW. The Swinomish Tribe's fresh waters do not themselves support salmonid spawning. The Tribe identified four small streams that originate within the Swinomish Reservation and flow toward the Swinomish Channel or Skagit Bay, as described above in section 2.1. Although these streams may be used by migrant Chinook salmon fry that originate in the nearby rivers (E.M. Beamer et al., 2013), the extent of the streams is small, and the streams are unlikely to host a large population of Chinook salmon or other species that would ultimately serve as prey for SRKW.

In support of this perspective, recent information suggests that north Puget Sound Chinook stocks account for approximately 5% of the fall/early winter and mid-winter/early spring diets of SRKW (M Bradley Hanson et al., 2021). This information is a result of an analysis of SRKW scat collected in SRKW range and is certainly not conclusive or static but does provide a data-based idea of the small proportion (much less than 5%) of prey likely provided by Swinomish freshwaters for SRKW.

Due to the small extent of fresh waters in the Swinomish Reservation and the absence of salmonid spawning habitat, the anadromous fish that may use the fresh water streams within the reservation would not constitute a significant fraction of the prey that any individual SRKW would consume; therefore, EPA approval of the Tribe's fresh water criteria is **not likely to adversely affect** SRKWs both with respect to prey quality (direct effect) and prey availability (indirect effect).

Chum: As described in section 3.3, ESA listed Hood Canal summer-run chum salmon spawn in Hood Canal and its tributaries and in rivers between Hood Canal and Dungeness Bay and critical habitat is designated in these areas. Most of these juvenile chum salmon likely enter marine waters proximate to these areas and migrate west to the Pacific Ocean. Although not described by NMFS, it could be possible that some of these juvenile chum salmon swim east to the Swinomish Tribal waters (action area) prior to migrating to the Pacific Ocean. However, it would be highly unlikely these juvenile chum salmon would re-enter the four fresh water streams of the action area. Therefore, any effects to these chum salmon from EPA's approval of Swinomish Tribe's freshwater water quality standards would be discountable, and **not likely to adversely affect** Hood Canal summer-run Chum salmon.

Humpback Whale: As described in section 3.7, humpback whales have been sighted in Puget Sound, primarily during the fall and spring. Most of the sightings reported to Orca Network since 2003 were in the main basin of Puget Sound, with sightings in the waters between Point No Point and Whidbey Island, Possession Sound, and southern Puget Sound in Colvos Passage. In addition, the Orca Network

(http://www.orcanetwork.org/Main/index.php?categories_file=Sightings) reported humpback whales in Skagit Bay south of the action area, but no sightings were reported within the action area. The action area only includes fairly shallow waters (see Figure 2).

Major prey species for humpback whales include small schooling fish and large zooplankton, primarily krill (NMFS, 1991). Fish prey species consist of Pacific herring (*Clupea pallasii*), juvenile walleye (*Sander vitreus*), pollock, capelin, and sand lance (NMFS, 1991, 2006). The above prey species for humpback whales are saltwater species and are not present in freshwater. Based on the absence of sightings of humpback in the action area and on their diet, which does not include salmonids (or would include salmonids only incidentally (NMFS, 2012)), EPA has determined that its approval of the Swinomish Tribe's aquatic life criteria for fresh waters will have **no effect** on humpback whales.

5.1.2 Not likely to adversely affect determinations for SRKW and humpback whales for all marine toxics criteria

Chemicals subject to the Tribe's marine water quality standards may affect SRKW and humpback whale via exposure to marine waters in the action area through the ingestion of these chemicals in their prey, or by reductions in the availability of their prey exposed to marine waters in the action area. Humpback whale have not been recorded in the action area, as indicated above. Additionally, SRKW are unlikely to spend much time in the action area, given the shallow nature of the waters (predominantly intertidal waters, apart from the shallow Swinomish Channel). As a result, dermal exposure to any chemicals in the action area that are subject to the Tribe's standards is unlikely to be significant for both SRKW and humpback whale.

EPA's proposed approval of the Tribe's water quality standards is also unlikely to affect availability and quality of prey to a significant extent. Regarding prey quality, among criteria in assessment categories 1, 2, and 4, selenium is the only bioaccumulative chemical for which the Tribe has adopted criteria (see section 5.4.3.4.1). Although selenium is bioaccumulative, it is most toxic to oviparous species such as fish and birds, and is of lower concern for mammals (Janz et al., 2010). Assessment category 3, section 5.3, includes additional bioaccumulative chemicals, but these are all banned for use in the U.S., and the NLAA determinations for these category 3 criteria are based on exposure considerations. NMFS (2012, p.539) identified 3 bioaccumulative chemicals that are of particular concern for SRKW because they are found in higher levels in SRKW: the legacy organochlorine compounds PCBs and DDTs, and PBDEs. The Tribe has adopted criteria for DDT, DDD, DDE, and PCBs, which are addressed in section 5.3. However, the exposure considerations described below for prey and for its ingestion by SRKW would also apply to these criteria.

As indicated above, north Puget Sound Chinook stocks account for approximately 5% of the fall/early winter and mid-winter/early spring diets of SRKW (Hanson et al., 2021), and Chinook that enter the action area would represent a fraction of this small percentage. In turn, as adults these Chinook would generally be expected to spend little time in action area waters as they migrate to spawning areas in the Skagit River basin; the body burden of toxic chemicals would be expected to accumulate predominantly outside of the action area for these Chinook, and for other salmon species that are prey for SRKW and may enter action area waters. Since north Puget Sound Chinook that have entered action area waters represent a small fraction (much less than 5%) of SRKW prey, and extended exposure of adult Chinook and other prey species to action area waters is unlikely, exposure of SRKW prey to chemicals at the

criteria levels in the action area is unlikely to have adverse effects on the quality of SRKW prey overall. Availability of prey is also not likely to be reduced significantly, since individuals that may rear for some time in the action area and could theoretically be exposed to criteria levels of toxic chemicals constitute a very small fraction of prey available to the SRKW. Based on the above considerations, EPA has determined that its approval of the Swinomish Tribe's aquatic life criteria for marine waters is **not likely to adversely affect** SRKW.

Similar considerations regarding effects of the proposed action (EPA's approval of the Tribe's criteria) on prey availability and prey quality apply to the humpback whale. As indicated above for freshwater criteria, prey species for humpback whales include small schooling fish and large zooplankton, primarily krill (NMFS, 1991). These may be present in the action area, and if chemicals reach criteria levels or toxic levels below the criteria levels, adverse effects to prey may occur in the action area. However, the action area constitutes a small fraction of habitat available for prey of humpback whale, and any adverse effects to potential prey species in the action area would not be expected to have a significant effect on the prey base of humpback whale.

Based on the absence of sightings of humpback in the action area and on the small fraction of potential prey that would be affected by any chemicals in the action area that reach levels that may result in adverse effects to the prey, EPA has determined that its approval of the Swinomish Tribe's aquatic life criteria for marine waters is **not likely to adversely affect** humpback whales.

5.2 Effects Determinations by Reference for Category 1 and 2 Toxics Criteria

Category 1 criteria are those for which a recent ESA consultation has been completed for the same criterion in another jurisdiction for species addressed in this consultation, and the Services have determined that any effects of exposure at the criterion level would not likely to adversely affect the species, or would not result in jeopardy to the species or adverse modification of critical habitat.

Category 2 criteria are those for which EPA recently issued new criteria recommendations under CWA section 304(a) to address a RPA that resulted from jeopardy determinations in Oregon and/or Idaho on older 304(a) criteria recommendations. As such, the category 2 chemicals were also the subject of recent ESA consultations, and the new 304(a) criteria recommendations, which the Tribe adopted, represent RPAs provided by NMFS for EPA approval of Oregon's criteria.

Because previous reviews and evaluations of the category 1 and 2 chemicals are available, EPA incorporates the detailed toxicity assessments and associated consultation documents as the primary basis for the effects determinations for these criteria in this BE. Category 1 and 2 criteria and the applicable referenced ESA documents are shown in Table 75.

Species were included in the referenced ESA consultation as shown in Table 74:

Table 74. Species Included in Swinomish and Referenced ESA Consultations for Category 1 and 2 Criteria

Referenced ESA Consultation[a]	Bull trout		Marbled Murrelet		Chinook		Chum	Steelhead		Hump-back	SRKW	
	FW	SW	FW	SW	FW	SW	SW	FW	SW	SW	FW	SW
Idaho BiOp, NMFS 2014					x			x			n/a[b]	
Idaho BiOp, USFWS 2015	x											
Oregon BiOp, NMFS 2012					x	x	x	x	x	n/a[b]	n/a[b]	n/a[b]
Oregon BiOp, USFWS 2012	x			x								
Idaho Cadmium Consultation, USEPA 2010	x				x			x				
Oregon BE, USEPA 2008	x		x	x	x	x	x	x	x	n/a[b]		

Notes:
FW – Fresh water criteria
SW – Salt water criteria (marine)
[a] Bocaccio and yelloweye rockfish are not addressed in any of the ESA consultations in EPA Region 10. Effects to these species are addressed in section 5.2.13.
[b] Not applicable. SRKW was addressed for fresh water criteria and SRKW and humpback whale were addressed for salt water criteria in the referenced consultation; however, we are not relying on the Idaho or Oregon ESA consultation for this BE. See section 5.1.

Table 75. Referenced ESA Consultations for Category 1 and 2 Criteria

Criterion	CMC (µg/L)	CCC (µg/L)	Species	Referenced ESA Documents
Category 1, Fresh Water Criteria				
Cadmium (hardness-based criteria) [a]	1.3 100 mg/L hardness	0.6 100 mg/L hardness	Steelhead, Chinook	• NMFS 2011 NLAA concurrence Letter on Idaho's cadmium criteria
			Bull trout	• USFWS 2011 NLAA concurrence letter on Idaho's cadmium criteria

Criterion	CMC (µg/L)	CCC (µg/L)	Species	Referenced ESA Documents
Chromium (III) [a,b]	570	74	Steelhead, Chinook	• EPA 2008 Oregon BE (NLAA pgs. 5-170, 7-2 and 7-6) • NMFS 2014 Idaho BiOp (not likely to result in adverse effects) p. 278) • NMFS 2012 Oregon BiOp (acute and chronic toxic effects likely but no RPA to avoid Jeopardy/adverse modification p. 280 and 547)
			Bull trout	• EPA 2008 Oregon BE (acute NLAA; chronic may affect, but due to minimal exposure potential NLAA pgs. 5-170, 7-2 and 7-6) • USFWS 2015 Idaho BiOp (NLAA p. 212 and 252) • USFWS 2012 Oregon BiOp (effects for acute exposure likely to be adverse but not likely to cause injury or death; effects for chronic exposure likely to be insignificant p. 197)
Chromium (VI), CMC [a]	16	-- [c]	Steelhead, Chinook	• EPA 2008 Oregon BE (NLAA pgs. 5-196 and 7-2) • NMFS 2014 Idaho BiOp (not likely to result in adverse effects p. 278) • NMFS 2012 Oregon BiOp (acute toxic effects likely but no RPA to avoid jeopardy/adverse modification p. 280 and 547)
			Bull trout	• EPA 2008 Oregon BE (NLAA pgs. 5-196 and 7-2) • USFWS 2015 Idaho BiOp (NLAA pgs. 212 and 253) • USFWS 2012 Oregon BiOp (effects of acute exposure insignificant p. 197);
Pentachloro-phenol [d]	19	15	Steelhead, Chinook	• EPA 2008 Oregon BE (NLAA, except chronic criterion may affect steelhead p. 5-397) • NMFS 2014 Idaho BiOp (minor effects associated with acute and chronic criteria, but no jeopardy/adverse modification pgs. 209 and 278) • NMFS 2012 Oregon BiOp (acute and chronic toxic effects likely but no RPA to avoid Jeopardy/adverse modification pgs. 225 and 547)
			Bull trout	• EPA 2008 Oregon BE (NLAA p. 5-397) • USFWS 2015 Idaho BiOp (NLAA pgs. 251 and 254) • USFWS 2012 Oregon BiOp (NLAA/discountable effects p. 210)
Category 1, Marine Water Criteria				
Cadmium	33	7.9	Steelhead, Chinook, Chum	• EPA 2008 Oregon BE (NLAA p. 5-536) • NMFS 2012 Oregon BiOp (acute or chronic toxic effects but no jeopardy/adverse modification p. 367)
			Marbled murrelet	• EPA 2008 Oregon BE (NLAA p. 5-536) • USFWS 2012 Oregon BiOp (NLAA p. 5)

Criterion	CMC (µg/L)	CCC (µg/L)	Species	Referenced ESA Documents
Copper [a]	4.8	3.1	Steelhead, Chinook, Chum	• EPA 2008 Oregon BE (NLAA p. 5-569) • NMFS 2012 Oregon BiOp (acute or chronic toxic effects but no jeopardy/adverse modification p. 372)
			Marbled murrelet	• EPA 2008 Oregon BE (NLAA p. 5-569) • USFWS 2012 Oregon BiOp (NLAA p. 5)
Selenium [a]	290	71	Steelhead, Chinook, Chum	• EPA 2008 Oregon BE (acute NLAA, chronic may affect, but NLAA based on low exposure p. 5-654) • NMFS 2012 Oregon BiOp (acute or chronic toxic effects but no jeopardy/adverse modification p. 387)
			Marbled murrelet	• EPA 2008 Oregon BE (acute NLAA, chronic may affect but NLAA based on low exposure p. 5-654) • USFWS 2012 Oregon BiOp (chronic LAA, no take) (pp. 6, 324-329)
Silver [a]	1.9	-- [e]	Steelhead, Chinook, Chum	• EPA 2008 Oregon BE (NLAA p. 5-661) • NMFS 2012 Oregon BiOp (chronic toxic effects but no jeopardy/adverse modification p. 388)
			Marbled murrelet	• EPA 2008 Oregon BE (NLAA p. 5-661) • USFWS 2012 Oregon BiOp (NLAA p. 6)
Category 2, Fresh Water Criteria				
Ammonia	Temp., pH depen-dent	Temp., pH depen-dent	Steelhead, Chinook	ESA documents including analysis of the effects of the EPA 2013 ammonia criteria associated with implementing the NMFS BiOp RPA in Oregon (see section 5.2.9)
			Bull trout	ESA documents including analysis of the effects of the EPA 2013 ammonia criteria associated with implementing the NMFS BiOp RPA in Oregon (see section 5.2.9)
Copper	Copper BLM	Copper BLM	Steelhead, Chinook	Documents associated with implementing the NMFS BiOp RPA in Idaho (see section 5.2.10)
			Bull trout	Documents associated with implementing the USFWS BiOp RPA in Idaho (see section 5.2.10)
Selenium	Tissue-based	Tissue-based	Steelhead, Chinook	ESA documents including analysis of the effects of the EPA 2016 Selenium criteria associated with implementing the NMFS BiOp RPA in Idaho (see section 5.2.11)
			Bull trout	ESA documents including analysis of the effects of the EPA 2016 Selenium criteria associated with implementing the USFWS BiOp RPA in Idaho (see section 5.2.11)

Criterion	CMC (µg/L)	CCC (µg/L)	Species	Referenced ESA Documents
[a] ESA consultation was completed on the same criteria values for Idaho and/or Oregon (as relevant). [b] The criteria are hardness-dependent equations; the values shown are the criteria for 100 mg/L hardness as $CaCO_3$. [c] An assessment for the hexavalent chromium CCC is included in this BE; see section 5. [d] For pentachlorophenol, ESA consultation was completed on the same criteria values for Oregon. Idaho's CMC is slightly higher at 20 ug/L, and the CCC is slightly lower at 13 ug/L. [e] The Tribe only adopted an acute criterion for silver.				

The following ESA consultation documents are referenced for category 1 and category 2 criteria:

1. ESA consultation on EPA's approval of Oregon's toxics criteria

Biological evaluation, EPA (2008):

USEPA. 2008. Biological Evaluation of Oregon's Water Quality Criteria for Toxics. USEPA Region 10, Seattle, WA. January 2008.

Biological Opinion, USFWS (2012):

USFWS. 2012. Formal section 7 consultation on USEPA's proposed approval of Oregon water quality criteria for toxics. Oregon Fish and Wildlife Office, Portland, Oregon. TAILS no. 13420-2009-F-0011. July 30, 2012.

Biological Opinion, NMFS (2012):

NMFS. 2012. Jeopardy and Destruction or Adverse Modification of Critical Habitat Endangered Species Act Biological Opinion. National Marine Fisheries Service, Northwest Region. Seattle, WA. August 14, 2012. NMFS No.: 2008/00148

Communications related to the RPA for the copper BLM and Oregon's adoption of the copper BLM for fresh waters:

Letter dated October 8, 2015 to Kim Kratz, Assistant Regional Administrator Oregon and Washington Office, NMFS, from Christine Psyk, Associate Director, Office of Water and Watersheds. Re: Confirm USEPA's understanding of RPAs for the copper aquatic life criteria specified in NMFS 2012 biological opinion on Oregon's toxic WQS.

Letter dated December 3, 2015 to Christine Psyk, Associate Director, Office of Water and Watersheds, from Kim Kratz, Assistant Regional Administrator Oregon and Washington Office, NMFS. Re: Implementation of RPA for Acute and Chronic Copper in 2012 Biological Opinion on Oregon Toxics Water Quality Criteria.

Letter dated January 19, 2016 to Christine Psyk, Associate Director, Office of Water and Watersheds, from Kim Kratz, Assistant Regional Administrator Oregon and Washington Office, NMFS. Re: Corrections to December 3, 2015 Letter on Implementation of RPA for Acute and Chronic Copper in 2012 Biological Opinion on Oregon Toxics Water Quality Criteria.

Letter dated January 6, 2017 to Paul Henson, State Supervisor, Oregon Fish and Wildlife Office, USFWS, from Angela Chung, Manager, WQS Program EPA R10. Re: The EPA's upcoming Action on Oregon's freshwater aquatic life criteria for copper for waters under the jurisdiction of the State of Oregon.

RPA assessment for Oregon's acute cadmium criterion, EPA (2016):

USEPA. 2016. EPA's Review of Revised 2016 Freshwater Acute Cadmium Criterion for Oregon in Accordance with the Reasonable and Prudent Alternatives in the National Marine Fisheries Service 2012 Biological Opinion on Oregon's Toxics Water Quality Standards. December 8, 2016.

Available at https://www.regulations.gov/document/EPA-HQ-OW-2016-0012-0181 and in Appendix D of this BE.

2. ESA consultation on EPA's approval of Idaho's toxics criteria

Biological Opinion, USFWS (2015):

USFWS. 2015. Biological Opinion for the Idaho Water Quality Standards for Numeric Water Quality Criteria for Toxic Pollutants. Idaho Fish and Wildlife Office. USFWS No.: 01E1FW00-2014-F-0233. Boise, Idaho. June 25, 2015.

Biological Opinion, NMFS (2014):

NMFS. 2014. Endangered Species Act section 7(a)(2) Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat (EFH) Consultation. National Marine Fisheries Service, West Coast Region. Seattle, WA. May 7, 2014. NMFS No.: 2000-1484

Communications confirming protectiveness of Idaho's adoption of the copper BLM for fresh waters:

Letter dated February 28, 2019 to William M. Lind, Southern Snake Branch Chief, NMFS, from Hanh Shaw, WQS Unit Manager, USEPA R10. Re: RPA for the Copper Aquatic Life Criteria in the 2014 Biological Opinion on Idaho's WQS for Toxic Substances.

Letter dated February 28, 2019 to Gregory M. Hughes, State Supervisor, USFWS, Idaho Fish and Wildlife Office, from Hanh Shaw, WQS Unit Manager, USEPA R10. Re: RPA for the Copper Aquatic Life Criteria in the 2015 Biological Opinion on Idaho's WQS for Toxic Pollutants.

Letter dated March 12, 2019 to Hanh Shaw, WQS Unit Manager, USEPA R10, from Michael P. Tehan, Assistant Regional Administrator, NMFS. Re: Implementation of RPA for the Copper Aquatic Life Criteria in the 2014 Biological Opinion on Idaho's WQS for Toxic Substances.

Letter dated March 19, 2019 to Hanh Shaw, WQS Unit Manager, USEPA R10, from Gregory M. Hughes, State Supervisor, USFWS, Idaho Fish and Wildlife Office. Re:

Implementation of RPA for the Copper Aquatic Life Criteria in the 2015 Biological Opinion on Idaho's WQS for Toxic Pollutants (01E1FW00-2014-F-0233).

Communications confirming protectiveness of Idaho's adoption of revised selenium criteria for fresh waters:

Letter dated October 25, 2018 to Bill Lind, Southern Snake Branch Chief, NMFS, from Angela Chung, Associate Director Office of Water and Watersheds EPA R10. Re: USEPA's Proposed Approval of Idaho's new and revised Aquatic Life Selenium Criterion.

Letter dated November 2, 2018 to Greg Hughes, State Supervisor, USFWS, from Angela Chung, Associate Director Office of Water and Watersheds EPA R10. Re: USEPA's Proposed Approval of Idaho's new and revised Aquatic Life Selenium Criterion.

Letter dated October 26, 2018 to Angela Chung, Associate Director Office of Water and Watersheds EPA R10, from William M. Lind, Southern Snake Branch Chief, NMFS. Re: Implementation of RPA for the Chronic Selenium Aquatic Life Criteria in the 2014 Biological Opinion on Idaho's WQS for Toxic Substances. NMFS No.: WCR-2000-1484.

Letter to Angela Chung, Associate Director Office of Water and Watersheds EPA R10, from Gregory M. Hughes, Idaho State supervisor. Re: Implementation of RPA for the Chronic Selenium Aquatic Life Criteria in the 2015 Biological Opinion on Idaho's WQS for Toxic Pollutants.

3. ESA consultation on EPA's approval of Idaho's cadmium criteria

USEPA. 2010. Final Biological Evaluation of the Idaho Water Quality Criteria for Cadmium with Revised hardness cap for the USFS and the NMFS. Seattle, WA. September 1, 2010.

NMFS. 2011. Endangered Species Act section 7 Informal Consultation and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for the proposed approval of Idaho's Water Quality Criteria for Cadmium (One project). January 31, 2011. NMFS No.: 2010/04306.

Letter dated March 7, 2011 to Jannine Jennings, WQS Unit Manager, EPA R10, from Brian T. Kelly, State Supervisor, Idaho Fish and Wildlife Office, USFWS. Re: Idaho Cadmium Standards – Idaho Statewide – Informal Consultation CONS-100a 14420-2011-I-0071.

5.2.1 Cadmium (fresh water)

ESA consultation on fresh water cadmium criteria has been completed in Oregon and Idaho. The Swinomish Tribe has adopted essentially the same fresh water cadmium criteria as Idaho (C.A. Mebane, 2006, 2010). Cadmium criteria are hardness-based. Example criteria are shown in Table 76 for waters of various hardness for comparison.

Table 76. Example hardness-based criteria values for cadmium in Oregon, the Swinomish Tribe and Idaho, and EPA's current cadmium criteria recommendation

Hardness[a]	Acute or chronic criterion[b]	Swinomish Tribe (Mebane 2010)[c]	Idaho ESA consultation (Mebane 2006)	Oregon ESA consultation (2001 EPA criteria recommendation)	Current EPA criteria recommendation (EPA 2016)
10	CMC	0.20	0.20	0.21	0.21
	CCC	0.14	0.15	0.049	0.13
25	CMC	0.42	0.42	0.52	0.49
	CCC	0.24	0.25	0.094	0.25
50	CMC	0.73	0.75	1.0	0.94
	CCC	0.37	0.38	0.15	0.43
100	CMC	1.3	1.3	2.0	1.8
	CCC	0.55	0.57	0.25	0.72
200	CMC	2.2	2.4	3.9	3.4
	CCC	0.82	0.85	0.40	1.2

[a] Units are mg/L as $CaCO_3$. Hardness measurements with a range of 31 to 140 mg/L have been reported for the Swinomish Tribe's fresh waters (section 4.3.2.2).
[b] Units are µg/L for the cadmium criteria values.
[c] The Swinomish Tribe adopted a conversion factor for the acute criterion to calculate the criterion on a dissolved basis, which was not included in Mebane (2010). As a result, the Swinomish Tribe's acute cadmium criterion is slightly lower (more protective) than Mebane (2010).

Idaho adopted new cadmium criteria in 2010, which EPA approved in 2011 following ESA consultation. The Swinomish Tribe has adopted essentially the same criteria as Idaho. Idaho's cadmium criteria were developed in 2006 (C.A. Mebane, 2006) and revised slightly in 2010 (C.A. Mebane, 2010). The revised 2010 chronic criterion is slightly lower than the initial 2006 criterion, while the acute criterion is essentially the same with only very small increase. The Swinomish Tribe adopted the Idaho 2010 criteria C.A. Mebane (2010), and additionally adopted a conversion factor that is applied to the acute criterion, which slightly lowers the acute criterion below the Idaho 2006 (C.A. Mebane, 2006) values (see Table 76).

EPA and the Services completed ESA consultation on EPA's approval of Idaho's cadmium criteria (EPA 2010, USFWS 2011, NMFS 2011). USFWS concurred with EPA's determination that approval of Idaho's cadmium criteria is not likely to adversely affect bull trout, and that any effects of exposure at the criteria level would be insignificant (USFWS, 2011, p. 7). NMFS concurred that EPA's determination that approval of Idaho's criteria is not likely to adversely affect Chinook salmon and steelhead in Idaho. NMFS (2011 p. 3) found that EPA's approval of Idaho's cadmium criteria would have only insignificant effects on Chinook and steelhead, provided that conservation measures were followed as EPA described in its BE.

NMFS (2011, p. 29) found that exposure at the level of Idaho's acute cadmium criterion is unlikely to adversely affect Chinook salmon, but such exposure may result in adverse effects to steelhead. In its BE, EPA described conservative measures that are used when EPA prepared NPDES permits, and NMFS determined that with implementation of these conservative measures, a permit limit written to the criterion would result in cadmium concentrations of half the criterion level or less at least 90 percent of

the time. NMFS concluded that with implementation of these conservative measures, effects to steelhead of EPA approval of Idaho's acute cadmium criterion would be insignificant (NMFS 2012, p. 29). Unlike Idaho, the Swinomish Tribe has prohibited the use of mixing zones for NPDES permits in fresh waters in WQS section 19-06.240(B)(7). There are currently no permitted point source discharges to the Swinomish Tribe's fresh waters (section 4.2.2), and cadmium was not detected for all 22 results available for the action area (section 4.3.2.4). If a NPDES permit were to be written for a fresh water point source in the future, the cadmium criteria would be met at "end of pipe," (i.e. no mixing zone would be authorized), and EPA would implement the same conservative measures in a permit as used in Idaho.

EPA's NLAA determinations for approving the Swinomish Tribe's acute and chronic cadmium criteria relative to bull trout, Chinook salmon, and steelhead are based on the toxicity evaluation in the BE for approval of Idaho's cadmium criteria (USEPA, 2010), the evaluation and concurrence letters provided by the Services (NMFS 2011, USFWS 2011), and the additional considerations regarding ambient cadmium levels and the prohibition of mixing zones in the Swinomish Tribe's WQS.

The Swinomish Tribe has not adopted a low end hardness cap, and the hardness-dependent cadmium criteria formulas would apply at all hardness levels. As a result, considerations related to the decline of protectiveness at low hardness values resulting from Idaho's hardness cap of 10 mg/L as $CaCO_3$, made by both Services (NMFS 2012, USFWS 2012), are not relevant to the Swinomish Tribe's WQS.

Based on the above, EPA concludes that exposure at the concentration of the Swinomish Tribe's fresh water acute and chronic cadmium criteria is **not likely to adversely affect** Chinook salmon, steelhead, and bull trout.

5.2.2 Chromium III (fresh water)

The Swinomish fresh water acute and chronic chromium III criteria are the same criteria EPA consulted on in Oregon and Idaho. In Idaho, NMFS (NMFS, 2014b) determined exposure of at the criteria levels was NLAA Chinook and Steelhead and USFWS (USFWS, 2015a) determined NLAA for bull trout. EPA acknowledges that in the Oregon toxic consultation, NMFS determined that some adverse effects from the Chromium III criteria were possible, but EPA defers to the more recent assessments in the Idaho consultation. Further, Chinook, steelhead, and bull trout exposure to the chromium III at the criterion levels in fresh waters of action area is unlikely due to the lack of current and anticipated sources of chromium III.

Based on the above, EPA concludes that exposure at the concentration of the Swinomish Tribe's fresh water acute and chronic chromium III criteria is **not likely to adversely affect** Chinook salmon, steelhead, and bull trout.

5.2.3 Chromium VI (fresh water)

The Swinomish fresh water acute criterion is the same criterion EPA consulted on in Oregon and Idaho. In Idaho, NMFS (NMFS, 2014b) determined exposure of at the criterion level was NLAA Chinook and steelhead and USFWS (USFWS, 2015a) determined NLAA for bull trout. EPA acknowledges that in the Oregon toxic consultation, NMFS determined some adverse effects from the acute chromium VI criteria were possible, but EPA defers to the more recent assessments in the Idaho consultation. Further, Chinook, steelhead, and bull trout exposure to the chromium VI at the criterion level in fresh waters of action area is unlikely due to the lack of current and anticipated sources of chromium VI.

Based on the above, EPA concludes that exposure at the concentration of the Swinomish Tribe's fresh water acute chromium VI criterion is **not likely to adversely affect** Chinook salmon, steelhead, and bull trout.

5.2.4 Pentachlorophenol (fresh water)

The Swinomish Tribe's fresh water pentachlorophenol criteria are pH-dependent equations (see Table 6, footnote n), which are the same as EPA's current CWA section 304(a) recommendations. The acute (19 µg/L at pH 7.8) and chronic (15 µg/L at pH 7.8) PCP criteria are the same criteria EPA consulted on in Oregon and similar to those consulted on in Idaho. Idaho's acute criterion is slightly higher (20 µg/L at pH 7.8) and the chronic criterion is slightly lower (13 µg/L at pH 7.8). The differences are small and the effects conclusions for Idaho are also relevant to EPA's action on the Swinomish Tribe's aquatic life criteria.

In both the Oregon and Idaho consultations, NMFS (NMFS, 2014b and 2012) determined that exposure at the criteria levels would likely cause adverse toxic effects for Chinook and steelhead. For example, based on a review of available toxicity data, NMFS (2014b, p. 222) concluded:

> "In summary, a number of toxicity studies reported concentrations that are less than the acute and chronic criteria concentrations for pentachlorophenol, which implies that listed species exposed to waters equal to criteria concentrations will suffer acute or chronic toxic effects. Conversely, a number of toxicity studies reported concentrations that are greater than the acute and chronic criteria concentrations for pentachlorophenol, which implies that listed species exposed to waters equal to criteria concentrations may not suffer acute or chronic toxic effects. When the available information is equivocal, NMFS gives the benefit of the doubt in its analysis to the listed species. Based on this principle and the considerations of the shortcomings and implications of laboratory-derived toxicity tests, the relative percent mortality analysis, and the ecological consequences for field-exposed fishes, listed species exposed to waters equal to the acute or chronic criteria concentrations will suffer acute and chronic toxic effects."

However, NMFS found that EPA approval of Oregon's PCP criteria would not cause jeopardy to salmonid species or adverse modification of critical habitat based on exposure considerations (NMFS 2014b, p. 278):

> "Discharges of PCP in Idaho are only expected from the use of treated wood in construction in or around surface water. Therefore any effects from the proposed approval of the PCP chronic or acute criteria are expected to be short-term events and have only minor effects on listed species and designated critical habitat. Therefore, NMFS concludes that the proposed criteria for PCP are not likely to jeopardize Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead, or result in the adverse modification of designated critical habitat."

In both the Oregon and Idaho consultations, USFWS (USFWS, 2015a and 2012) similarly determined that EPA's approval action was not likely to adversely affect bull trout, based largely on exposure considerations. The USFWS determination for bull trout in Oregon is based wholly on exposure considerations and concludes (USFWS 2012, p. 210):

> "As discussed in the "Exposure" section of this document, it is highly unlikely that bull trout could be exposed to pentachlorophenol. … For those reasons, we conclude that the effects of pentachlorophenol on the bull trout are discountable (i.e., extremely unlikely to occur)."

For Idaho, USFWS found that exposure to PCP at acute criteria levels may result in toxic effects to bull trout (USFWS 2015a, p. 249), but determined that EPA's action is not likely to adversely affect bull trout based on exposure considerations, and that any effects would be insignificant or discountable (USFWS 2014a, p. 251).

It is similarly unlikely that Chinook, steelhead, and bull trout in the action area will be exposed to PCP at toxic levels in fresh waters due to the lack of current and anticipated sources of PCP. No NPDES-permitted point sources discharge to action area fresh waters, and no other sources of PCP are likely to be present, as PCP is a restricted use product that is only used for pressure or thermal treatment of wood. PCP was analyzed for but not detected in fresh water sediments in or near the action area (see Table 58).

Based on the above, consistent with the approach taken by NMFS and USFWS in Oregon and Idaho, EPA concludes that exposure at the concentration of the Swinomish Tribe's fresh water acute and chronic pentachlorophenol criteria is **not likely to adversely affect** Chinook salmon, steelhead, and bull trout.

5.2.5 Cadmium (marine water)

The Swinomish Tribe has adopted EPA's current (2016) 304(a) criteria recommendations for marine cadmium. EPA and the Services have completed ESA consultation on Oregon's marine cadmium criteria for Chinook, steelhead, chum, and marbled murrelet. EPA determined that its approval of Oregon's cadmium criteria are NLAA Chinook and steelhead (USEPA, 2008a), while NMFS determined that "the available evidence for cadmium indicates that listed species exposed to waters equal to the acute and chronic criteria concentrations will suffer acute or chronic toxic effects including mortality (moderate intensity) and sublethal effects (moderate intensity)" (NMFS, 2012, p. 367), but a jeopardy determination was not made.

EPA published new cadmium aquatic life criteria in 2016 (USEPA, 2016a), including marine criteria. These new criteria are more stringent than the earlier criteria that Oregon had adopted, and the Swinomish Tribe has adopted the current EPA recommended criteria (Table 77).

Table 77. Marine cadmium criteria adopted by Oregon and the Swinomish Tribe

Marine Cadmium Criterion (dissolved)	Oregon ESA consultation	Swinomish Tribe and EPA (2016)
CMC (µg/L)	40	33
CCC (µg/L)	8.8	7.9

EPA (2008a) concluded both the Oregon acute and chronic criteria would NLAA Chinook, steelhead, chum, and marbled murrelet. NMFS (2012) surmised that toxic effects would be associated the Oregon criteria based on inadequacies in EPA's criteria derivation approach but did not identify studies indicating such effects. USFWS (2014b) concluded both Oregon criteria would be NLAA marble murrelet. The Tribe's criteria are more stringent than Oregon's, as indicated above.

Based on the above, EPA concludes that exposure at the concentration of the Swinomish Tribe's marine acute and chronic cadmium criteria is **not likely to adversely affect** Chinook salmon, steelhead, chum, and marble murrelet.

5.2.6 Copper (marine water)

The Swinomish Tribe's marine water acute and chronic copper criteria are the same criteria EPA consulted on in Oregon. EPA (2008a) concluded both the acute and chronic criteria would NLAA Chinook, steelhead, chum, humpback whale, and marble murrelet. NMFS (2012) surmised that toxic effects would be associated the criteria based on inadequacies in EPA's criteria derivation approach but did not identify studies indicating such effects. USFWS (2014b) concluded both criteria would be NLAA marble murrelet.

Based on the above, EPA concludes that exposure at the concentration of the Swinomish Tribe's marine acute and chronic copper criteria is Not Likely to Adversely Affect ESA Listed Chinook salmon, steelhead, chum, and marble murrelet.

5.2.7 Selenium (marine water)

The Swinomish marine water acute and chronic selenium criteria are the same criteria EPA consulted on in Oregon. EPA (2008a) concluded the acute criteria would NLAA Chinook, steelhead, chum, humpback whale, and marble murrelet. NMFS (2012) surmised that toxic effects would be associated with both acute and chronic criteria based on inadequacies in EPA's criteria derivation approach but did not identify studies indicating such effects. USFWS (2014b) concluded NLAA for the acute criteria and LAA for chronic criteria for marble murrelet. EPA (2008) concluded the chronic criteria would adversely affect salmon, steelhead, humpback whale, and marble murrelet due to bioaccumulation effects with waters at the criterion level. However, due to low potential for exposure from the lack of selenium discharges, EPA concluded NLAA for these species in Oregon. It is well understood that selenium toxicity is attributable to chronic rather than acute exposures; therefore, EPA (USEPA, 2016b) only recommended chronic selenium freshwater criteria. Additionally, because selenium toxicity is a result of internal exposures (tissue residues), salinity in exposure water does not affect how duration of internal selenium exposure affects toxicity. Exposure to selenium at the chronic criterion level in Swinomish Tribal waters is uncertain.

Based on the above, EPA concludes that exposure at the concentration of the Swinomish Tribe's marine acute selenium criterion is **not likely to adversely affect** Chinook salmon, steelhead, chum, and marbled murrelet.

EPA further concludes that exposure at the concentration of the Swinomish Tribe's marine chronic selenium criterion is **likely to adversely affect** Chinook salmon, steelhead, chum, and marbled murrelet.

5.2.8 Silver (marine water)

The Swinomish Tribe's marine water acute silver criterion is the same criterion EPA consulted on in Oregon. EPA (2008a) concluded both the acute criteria would NLAA Chinook, steelhead, chum, humpback whale, and marble murrelet. NMFS (2012) surmised that toxic effects would be associated the criterion based on inadequacies in EPA's criteria derivation approach but did not identify studies indicating such effects. USFWS (2014b) concluded the criterion would NLAA marble murrelet.

Based on the above, EPA concludes that exposure at the concentration of the Swinomish Tribe's marine acute silver criterion is **not likely to adversely affect** ESA Listed Chinook salmon, steelhead, chum, marble murrelet.

5.2.9 Ammonia (fresh water)

ESA consultation on fresh water ammonia criteria has been completed in Oregon for species included in this ESA consultation. Oregon had adopted EPA's earlier (USEPA, 1999a) recommended ammonia criteria, and NMFS found that exposure at the levels of these criteria would jeopardize listed species in Oregon, including Chinook, coho, and steelhead. As an RPA, NMFS indicated that EPA should develop new ammonia criteria recommendations and that these should be adopted in Oregon (NMFS, 2012, p. 548). USFWS found that bull trout exposure to ammonia at the Oregon acute and chronic criteria levels would result in "Adverse effects that do not rise to the level of significantly disrupting breeding, feeding, or sheltering behavior or cause injury or death of affected individuals." (USFWS, 2012, p. 220).

Consistent with the RPA received from NMFS, EPA issued updated criteria recommendations for ammonia in 2013 (USEPA, 2013a), which Oregon adopted in 2015. NMFS indicated that these actions satisfied the conditions of the RPA for the acute and chronic ammonia criteria (letter from Kim Kratz, NMFS, to Christine Psyk, EPA, August 4, 2015), and EPA subsequently approved Oregon's revised ammonia criteria. The Swinomish Tribe has adopted EPA's 2013 recommended ammonia criteria, in effect implementing the RPA for the Oregon jeopardy determination.

Additional analysis of the effects of the proposed ammonia criteria on ESA species was completed to evaluate effects of exposure at criteria levels using low effects concentrations at pH and temperature inputs measured in tribal waters. This analysis is provided in Appendix E. Adjustment factors were developed by analyzing ammonia acute or chronic concentration-response data from taxonomically related species and the appropriate adjustment factor was applied to the available toxicity data to calculate a low effects concentration for each species. Local pH and temperature data were used to adjust both the criteria (acute or chronic) and the species-specific low effects concentrations (acute or chronic) to evaluate effects to the species and their prey specific to Tribal waters.

Based on this analysis, EPA determined that exposure to ammonia at criteria levels **is not likely to adversely affect** Chinook salmon, steelhead, and bull trout in the Swinomish Tribe's fresh waters (see Appendix E, section 3).

5.2.10 Copper (fresh water)

The Swinomish Tribe has adopted EPA's current recommended aquatic life criteria for copper, which are equation-based criteria that are calculated for ambient conditions at a site using a biotic ligand model (BLM) (USEPA, 2007a). The BLM uses inputs for 10 water quality parameters that affect copper bioavailability – temperature, pH, dissolved organic carbon, calcium, magnesium, sodium, potassium, sulfate, chloride, and alkalinity – to calculate the acute and chronic criteria for the input conditions. Since ambient conditions vary over various time scales, sites must be well characterized and/or conservative assumptions must be made during derivation of criteria values and permit limits in order to ensure that the criteria values or permit limits protect against effects of copper exposure under all conditions found at the site, including the most bioavailable conditions.

Both Oregon and Idaho initially adopted EPA's earlier copper criteria recommendation (USEPA, 1984), which were hardness-based equations for derivation of acute and chronic criteria values, and ESA consultation was completed on these hardness-based criteria for both states. In summary, the USFWS and NMFS biological opinions for Idaho and Oregon determined that EPA approval of Idaho's and Oregon's hardness-based copper criteria was likely to adversely affect salmonids, including bull trout, Chinook, chum, and steelhead. USFWS determined that in Idaho, exposure at the criteria level throughout the range of bull trout would jeopardize the continued existence of bull trout in Idaho, and NMFS determined jeopardy for listed salmonids in both Idaho and Oregon. The RPA for these jeopardy determinations was for the states to adopt (or EPA to promulgate) copper criteria that are at least as stringent as EPA's 2007 copper criteria recommendation, the copper BLM. The Swinomish Tribe's adoption of EPA's 2007 copper criteria recommendation is consistent with this RPA for Oregon and Idaho's hardness-based criteria.

EPA's **not likely to adversely affect** determination for the proposed approval of the Tribe's copper criteria relies on the ESA consultations regarding Oregon's and Idaho's copper criteria and the analysis of the RPA for the Idaho consultations (NMFS, 2014b, Appendix C). The toxicity analyses completed in NMFS's and USFWS's Idaho biological opinions and the analysis of the copper BLM in Appendix D of the NMFS biological opinion are incorporated here by reference (NMFS, 2014b) (NMFS No: 2000-1484).

The Idaho and Oregon biological opinions on the states' hardness-based copper criteria and the RPAs as relevant to the Swinomish copper criteria are summarized in the remainder of this section.

USFWS (2015a, p. 161) determined that Idaho's hardness-based copper criteria were "likely to cause significant adverse effects to bull trout" in Idaho based on an evaluation of copper toxicity data for salmonids from peer-reviewed sources (USFWS, 2015a, pp. 160-161), and assuming exposure of bull trout to copper at the criteria levels throughout its range. The USFWS biological opinion provided a jeopardy determination on Idaho's hardness-based copper criteria (USFWS, 2015a, p. 262). The RPA for Idaho included adoption of the copper BLM (USFWSa, 2015, p. 274), along with the statement, "The Service does not anticipate that additional consultation will be required if the 2007 national recommended aquatic life criteria or other alternative criteria which would be as protective for copper are adopted by EPA."

Finally, USFWS (2015a, p. 275) indicated, as part of its RPA analysis of Idaho's adoption of the hardness-based copper criteria, that "Some adverse effects would still be expected if ambient concentrations were at the 2007 chronic aquatic life criterion, but these would be minimized by further limiting mixing zone fractions to ¼ (25 percent) of the receiving water discharge in flowing waters[, which] is effectively similar to reducing the criteria by about 0.25X (NMFS, 2014b). Few if any adverse effects to listed species or habitats would be expected at about 0.25X the concentration resulting from the 2007 version of EPA's copper criteria." The Swinomish Tribe's WQS include a provision that prohibits mixing zones for fresh waters, which is more protective than limiting the mixing zone to 25 percent of the receiving water flow.

Idaho adopted the BLM as its copper criteria in 2019, fulfilling the RPA requirements for adopting copper criteria that are protective of bull trout and its critical habitat. In a letter to EPA dated March 19, 2019, USFWS agreed with EPA's assessment that Idaho's 2019 copper criteria met the RPA in the USFWS biological opinion (USFWS letter to USEPA, 2019).

Similar to the USFWS jeopardy determination for bull trout in Idaho, and based on assumptions of exposure at criteria concentrations throughout the species' range in Idaho, NMFS determined that EPA approval of Idaho's hardness-based copper criteria is likely to jeopardize listed salmon and steelhead in Idaho (NMFS, 2014b, pp. 275-276). NMFS (2014b, p. 282-283) provided the same RPAs for salmon species and steelhead as USFWS provided for bull trout. The primary RPA was for the state to adopt (or EPA to promulgate) copper criteria that are at least as stringent as EPA's 2007 copper criteria recommendation, the copper BLM, and as an interim measure, to ensure that any mixing zone is not greater than 25% of stream flow or width to ensure an adequate zone of passage. As indicated above, the Swinomish Tribe's BLM-based copper criteria comply with the RPA, and the Tribe's mixing zone prohibition is more protective than the interim measure that was prescribed for Idaho.

Appendix C of NMFS (2014b) provides an RPA analysis for the copper BLM. The NMFS (2014b) opinion concludes (page C-60): "As is, the 2007 criteria represent a huge improvement over the NTR [National Toxics Rule, a reference to the previously recommended hardness-based] copper criteria and generally represent a major advance in the science of water quality criteria. Its application appears to be protective of listed salmon, steelhead, and their ecosystems."

Similar to USFWS (above), NMFS provided a letter to EPA dated March 12, 2019 that indicated agreement that Idaho's adoption of the copper BLM fulfilled the RPA for the copper criteria and would protect listed species, including salmonids.

Based on the above, EPA concludes that exposure at the concentration of the Swinomish Tribe's fresh water acute and chronic copper criteria is **not likely to adversely affect** Chinook salmon, steelhead, and bull trout.

5.2.11 Selenium (fresh water)

ESA consultation on fresh water selenium criteria has been completed in Oregon and Idaho for listed species included in this ESA consultation. Oregon and Idaho had adopted EPA's earlier (USEPA, 1999b) recommended selenium criteria. These were 5 µg/L for the chronic criterion, and a weighting formula that accounted for the proportion of selenate and selenite for the acute criterion, with underlying chronic criteria of 12.8 and 186 µg/L for the two chemicals, respectively.

In Oregon, NMFS found that although direct and dietary exposure to fresh waters at the levels of the selenium criteria would have adverse effects, including mortality, sub-lethal effects, and effects resulting from dietary exposure (NMFS, 2012, p. 338), NMFS did not identify the selenium criteria among criteria that would be "likely to reduce appreciably the likelihood of both the survival and recovery of the listed species" (NMFS, 2012, p. 534). Similarly, USFWS found that bull trout exposure to selenium at criteria levels would be likely to adversely affect bull trout and result in take (USFWS, 2012, p. 337), but EPA approval of the criteria would not result in jeopardy to the species based on exposure considerations for bull trout populations in Oregon (USFWS, 2012, section 6.1).

In Idaho, ESA consultation was completed on Idaho's selenium criteria of 20 µg/L (acute) and 5 µg/L (chronic). NMFS (2014b) and USFWS (2015a) focused their assessments on the chronic criterion because bioaccumulation is the most important exposure pathway (NMFS, 2014b, p. 169). NMFS estimated that whole body fish tissue selenium levels up to 7.6 mg/kg dw or 2 µg/L in the water column approximate a low-effect level. NMFS estimated that continuous exposure at the selenium criterion level of 5 µg/L would result in tissue concentrations above 19.5 mg/kg dw in juvenile salmonids, and would "result in

growth reductions and increased mortality in juvenile anadromous salmonids" (NMFS, 2014b, p. 181). NMFS (2014b) determined that EPA approval of Idaho's chronic selenium criterion would result in jeopardy for anadromous salmonids in Idaho, including Chinook and steelhead among others, and USFWS (2015a, p. 258) similarly found that bull trout would be jeopardized.

The RPA provided by NMFS and USFWS for the salmonids included the requirement for EPA to promulgate federal criteria or approve a new chronic selenium criterion in Idaho, including ESA consultation on the new criterion. Both Services included the interim measures to require permitted dischargers to monitor fish tissue levels where aqueous selenium levels exceeded 2 µg/L (as an annual geometric mean) (or exceeded natural levels, if higher), and to include provisions in permits to reduce selenium loading in order to reduce impairment of aquatic life uses where fish tissue levels were >7.6 mg/kg dw, unless biomonitoring demonstrated that appreciable adverse effects were not present.

EPA published updated selenium criteria recommendations in 2016 (USEPA, 2016b), which Idaho adopted in 2018, implementing the RPA provided by the Services in their 2014 and 2015 biological opinions. EPA's selenium criteria recommendations and Idaho's statewide criteria[17] include a whole body fish tissue criterion of 8.5 mg/kg dw (derived from the egg/ovary criterion of 15.1 mg/kg dw), and water column criteria of 1.5 µg/L in lentic aquatic systems and 3.1 µg/L in lotic systems (both 30 day averages).

EPA provided letters to NMFS (October 25, 2018)[18] and USFWS (November 2, 2018)[19] that addressed two remaining inconsistencies between the Idaho biological opinions and EPA's 2018 selenium criteria recommendation, i.e., the difference between the whole body tissue criteria (the Services' 7.3 mg/kg dw and EPA's 8.5 mg/kg dw), and the differences in the water column criteria (the Services' 2 µg/L as an annual geometric mean and EPA's 3.1 µg/L as a monthly arithmetic average for lotic waters).

Attachment 2 of EPA's letter to NMFS, prepared by Christopher Mebane (USGS), addressed the first question by evaluating the uncertainty associated with the data and models used to determine the criteria, and concluded, "As the 2016 EPA ALC whole body value of 8.5 mg/kg, falls between the equally plausible Chinook salmon growth EC_{10} values of 7.3 to 8.9 mg/kg, the ALC is thus statistically indistinguishable from the Opinion EC_{10} value."

Attachment 2 of EPA's letter to USFWS evaluated the whole body tissue criteria against an effects study for brook trout, a surrogate species for bull trout, and concluded, "the criteria concentrations are lower than effects value surrogate estimates for bull trout, which indicates that the criteria would be protective of bull trout."

Attachment 1 to both letters provides a comparison of the protectiveness of EPA's 2016 criterion for lotic waters and the benchmark developed by the Services in their Idaho biological opinions, based on an analysis of the frequency at which criteria were exceeded in Idaho given the two averaging periods. EPA used existing monitoring data in Idaho to determine if a once in three-year exceedance of a monthly average 3.1 µg/L was similarly or more stringent than an annual geometric mean of 2.0 µg/L in that

[17] Idaho also adopted less stringent selenium criteria on a site-specific basis, which are not considered here.
[18] October 25, 2018. Letter from Angela Chung, Associate Director, Office of Water and Watersheds, U.S. EPA Region 10 to Bill Lind, Southern Snake Branch Chief, NOAA Fisheries, Boise Idaho. With attachments.
[19] November 2, 2018. Letter from Angela Chung, Associate Director, Office of Water and Watersheds, U.S. EPA Region 10 to Greg Hughes, State Supervisor, U.S. FWS, Boise Idaho. With attachments.

dataset. Based on this analysis, EPA concluded that the Idaho standard, which is based on EPA's recommended criterion for lotic waters, is consistent with USFWS and NMFS biological opinions.

NMFS (October 26, 2018)[20] and USFWS (November 29, 2018)[21] concurred with EPA's analysis and agreed that Idaho's selenium criteria satisfy the requirements of the selenium RPA, and EPA took approval action on Idaho's selenium criteria in 2019.

EPA is relying on the above analyses for its NLAA determination for approval of the Tribe's selenium criteria, and additionally notes that there are currently no NPDES-permitted discharges to the Tribe's fresh waters and that current and future exposure to selenium at the criteria levels in the Tribe's freshwaters is unlikely.

Based on the above, EPA concludes that exposure at the concentration of the Swinomish Tribe's fresh water selenium criteria is **not likely to adversely affect** Chinook salmon, steelhead, and bull trout.

5.2.12 Effects determinations for marbled murrelet for category 1 and 2 freshwater criteria with a detailed assessment for selenium

Although EPA provided assessments for the freshwater criteria in the BE (USEPA, 2008), USFWS did not consider the fresh water criteria to apply to marbled murrelet in Oregon. USFWS (2012, p. 128) states, "Marbled murrelets are opportunistic feeders utilizing prey of diverse sizes and species. They feed primarily on fish and invertebrates in marine waters although they have also been detected on rivers and inland lakes (Carter and Sealy 1986, USFWS 1992a). In general, small schooling fish and large pelagic crustaceans are the main prey items. Marbled murrelets are able to shift their diet throughout the year and over years in response to prey availability (Becker et al., 2007). Freshwater prey appears to be important to some individuals during several weeks in summer. Because marbled murrelets spend the majority of time in marine environments, the saltwater criterion is applicable. Given their life history patterns, only bioaccumulative chemicals in this consultation have the potential to have any adverse effects on the marbled murrelets."

Because marbled murrelets use freshwater habitat for part of the year, EPA assessed effects to marbled murrelet from prey ingestion. An assessment is provided below for selenium, which is the only bioaccumulative chemical among the fresh water criteria in assessment categories 1 and 2 (see Table 83). As the remaining freshwater criteria in categories 1 and 2 are not bioaccumulative, and marbled murrelets are able to shift their diet in response to prey availability, EPA concludes that its approval of the Swinomish Tribe's fresh water criteria for cadmium, chromium (III), chromium (VI), pentachlorophenol, ammonia, and copper is **not likely to adversely affect** marbled murrelet.

[20] October 26, 2018. Letter from William M. Lind, Southern Snake Branch Chief, NOAA Fisheries, Boise Idaho to Angela Chung, Associate Director, Office of Water and Watersheds, U.S. EPA Region 10. Re: Implementation of Reasonable and Prudent Alternative for the Chronic Selenium Aquatic Life Criteria in the 2014 Biological Opinion on Idaho's Water Quality Standards for Toxic Substances.

[21] November 29, 2018. Letter from Gregory M. Hughes, State Supervisor, U.S. FWS, Boise Idaho to Angela Chung, Associate Director, Office of Water and Watersheds, U.S. EPA Region 10. Re: Implementation of Reasonable and Prudent Alternative for the Chronic Selenium Aquatic Life Criteria in the 2015 Biological Opinion on Idaho's Water Quality Standards for Toxic Pollutants.

5.2.12.1 Analysis of the effects of the freshwater selenium water quality standard on marbled murrelet

5.2.12.1.1 Approach to evaluate the protectiveness of the water column criteria for marbled murrelet.

Birds feeding on aquatic species in the action area could potentially be exposed to selenium (Se). EPA is using an approach based on the Presser and Luoma (2010) selenium food web model to evaluate potential effects of Se in the water column at the level of the tribe's criteria on birds. The model uses the egg-ovary (EO) selenium effect concentration, a composite trophic transfer factor (TTF) and an enrichment factor (EF); the EF represents the partitioning of selenium between the particulate and dissolved phases to estimate water column concentrations that are protective of marbled murrelet (MAMU). Egg-Ovary EC_{10}'s were converted to EC_5's using a concentration-response curve estimated using the 'drc' code in R (R_Core_Team, 2020). The EFs were taken from USEPA (2016b), as this information was not available for Swinomish waters. For lotic waters (the type of waters most likely used by breeding MAMU in the action area), the EF is 0.40. The TTF composites are species-specific based on dietary information from USFWS sources, the NatureServe website, Birds of the World, and other scientific literature. TTFs for fish and invertebrates were derived from Tables 3.10 and 3.11 in USEPA (2016b); (USEPA, 2021).

The following description of the marbled murrelet's diet is taken from Birds of the World (S. Nelson, 2020):

> *Considered an opportunistic feeder rather than a specialist (Sanger 1987, Burkett 1995). Seems to prefer euphausiids in spring and fish in summer, however. Consumption of forage fish coincides with nestling and fledgling periods (Sealy 1975, Carter 1984).*
>
> *Pacific sand lance are the most important prey species in summer, followed by northern anchovy, Pacific herring, osmerids (capelin and surf smelt), and seaperch. Also feeds on Pacific sardine, walleye pollock, rockfish, and squid during breeding season. Euphausiids are key prey in spring (Sealy 1975), and during breeding season in some years (L. Krasnow and G. Sanger unpubl. data). Euphausiids, mysids, gammarids (amphipods), osmerids, and herring are dominant prey in winter (Munro and Clemens 1931, Sanger 1987, Vermeer 1992). Also feeds on rockfish, squid, and shrimp during winter. Feeds on salmon (sockeye and Kokanee) in freshwater lakes, primarily in summer (Carter and Sealy 1986).*

Based on the above description, MAMUs may use freshwater in the action area during their nesting season. During this stage, MAMU could be exposed to Se in forage fish tissues found in four small streams in the action area.

5.2.12.1.2 Derivation of an Egg Chronic Minimum Effect Threshold Representative of Marbled Murrelet

The tribe does not have a freshwater criterion specifically intended to protect birds, although the WQS apply the freshwater aquatic life criteria to protect the wildlife designated use (Swinomish WQS code, 19-06 Table 3). USEPA (2018a) published egg-ovary criteria based on mallard studies (Heinz, Hoffman, & Gold, 1989; Heinz, Hoffman, Krynitsky, & Weller, 1987; Stanley Jr, Smith, Hoffman, Heinz, & Rosscoe, 1996). Data from the three studies were combined into a single concentration-response relationship for hatchability, the most sensitive endpoint, versus Se concentration in eggs. The dose-response curve

(drc) package in R (version 3.4.3) was used to calculate an egg EC_{10} of 11.2 mg Se/kg dw and an egg EC_5 (i.e., minimum effect threshold) of 7.78 mg Se/kg dw.

5.2.12.1.3 Water-Column Criterion Assessment

EPA is using an adaptation of the Presser and Luoma (2010) selenium food web model to evaluate the protectiveness of the tribe's water column criteria. A water column based chronic minimum effect threshold protective for MAMU was calculated by solving Equation 1, using mallard EC_5 data as a surrogate for the MAMU's egg-based chronic minimum effect threshold as the numerator, and the product of the $TTF^{composite}$ and *EF* as the denominator.

$$C_{water} = C_{bird\ egg/fish\ egg} \ / \ (TTF^{composite} \times EF)$$

Calculation of composite trophic transfer factor ($TTF^{composite}$)

A *TTF* is an estimate of the selenium transfer to MAMU from its food. MAMU's diet during nesting is dominated by forage fish. EPA used MAMU dietary information to construct model diets to derive a $TTF^{composite}$ for this species. The dietary information suggests this species is largely dependent on fish. As the empirically measured fish TTFs were not available for all of the fish species that may be consumed by breeding MAMU, a mean composite fish TTF of 2.34 was used (Appendix Table B-12 in USEPA (2018)). Additionally, because empirical bird TTFs were not available for MAMU, a composite TTF for this species was calculated using a surrogate species (pied-billed grebe) with a similar diet for which there is 187 an empirically-derived bird to fish TTFs. Therefore, a pied-billed grebe bird to fish TTF of 0.78 was applied, and composite TTF calculations are provided below.

$$TTF^{composite} = 0.78_{\text{bird}} \text{ x } 2.34_{\text{fish}} = 1.83$$

In tribal freshwaters, MAMU is expected to be most associated with lotic habitats. Following Equation 5-1 of USEPA (2018), the MAMU surrogate egg-based chronic minimum effect threshold was divided by the product of the median $TTF^{composite}$ and geometric mean lotic *EF* to calculate the murrelet's water column-based chronic minimum effect thresholds of 12.14 µg/L for lotic habitats.

$$Water_{Lotic} = 7.78 / (1.83 \text{ x } 0.40) = 10.63 \text{ ug/l}$$

This value represents aqueous selenium concentrations at which reproductive effects in the marbled murrelet would be likely. These values are higher than the tribe's lotic water column criteria for selenium (3.1 ug/L). Therefore, EPA has determined that approval of the tribe's freshwater criteria for selenium **is not likely to adversely affect** the marbled murrelet.

5.2.13 Effects Determinations for bocaccio and yelloweye rockfish, and bull trout for category 1 marine criteria

Rockfish - Bocaccio and yelloweye rockfish, which reside in marine waters, were not found in the action areas for the Oregon consultation and were therefore not addressed for that consultation. Therefore, EPA conducted a rapid screening analysis to determine if available surrogate species toxicity data indicates a potential adverse effect to rockfish. No toxicity data were available for *Sabastes* spp. for all four chemicals. In lieu of specific toxicity data, EPA calculated geometric mean acute and chronic toxicity values at the class or kingdom level. See Table 78 for results and Appendix F for more description and the data used for the evaluation. Although this level of extrapolation is not typical, there were no data available for a more specific assessment. Given that the surrogate toxicity values were all significantly greater than the corresponding criteria concentrations, EPA concludes these category 1 marine criteria (cadmium, copper, selenium, silver) are **not likely to adversely affect** bocaccio and yelloweye rockfish.

Table 78. Comparison between the taxonomic surrogate geomean toxicity values for rockfish (Sabastes spp.) and criteria concentrations for category 1 chemicals

Chemical	Taxonomic level match for acute data	Geomean Acute Toxicity Value (µg/L)	Taxonomic level match for chronic data	Geomean Chronic Toxicity Value (µg/L)	Swinomish Criteria (µg/L)	
					CMC	CCC
Copper	Class	1077.3	Kingdom	31.1	4.8	3.1
Selenium	Class	12050.0	No data	1205.0[a]	290	71
Silver	Class	62.0	NA	NA	1.9	None adopted
Cadmium	Class	2810.3	Class	1533.8	33	7.9

a. This value was obtained by dividing by 10 (S. Raimondo, B. J. Montague, & M. G. Barron, 2007) to estimate chronic effects from acute data, as no empirical chronic were available.

In addition to the toxicity assessment above for rockfish, another factor supporting a NLAA determination is the minimal presence of rockfish is the action acrea, especially in the shallow areas near the point source discharges. Natural Resource Consultants, Inc., under contract to NMFS, completed an analysis of rockfish habitat and historical areas of occurrence (NRC, 2016). The report assessed the spatial distribution of rockfish and indicates that rockfish are not abundant in action area waters relative to other areas in Puget Sound. NRC's figure 3 maps the relative abundance of rockfish in Puget Sound, and identifies an area of relatively low rockfish abundance based on infrequent sightings near the west side of the Swinomish Reservation in Skagit Bay (see Figure 29). Abundance "hot spots" (rockfish hot spot areas, or RHAs) were identified west of the action area, in the area of western Deception Pass and west of Fidalgo and Whidbey Islands.

Figure 30: Magnified detail of the Swinomish Reservation and surrounding area from Figure 3, map C of NRC (2016). Rockfish abundance was reported as being low.

Figure 9 of the NRC report (see Figure 31 below) shows RHAs for yelloweye rockfish, with the nearest to the action area located west of Deception Pass. Figure 11 identifies RHAs for bocaccio only south of Whidbey and Camano Islands (see Figure 32 below).

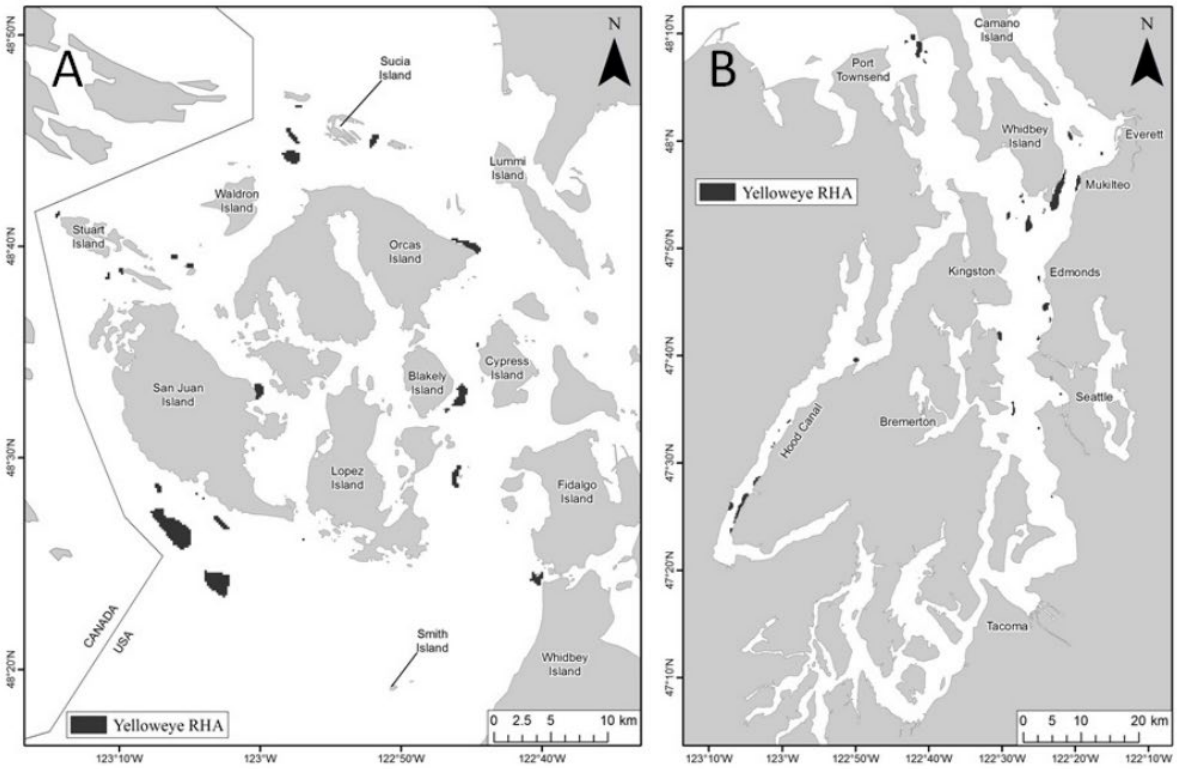


Figure 9. Spatial distribution of Yelloweye RHA's in (A) San Juan/Strait of Juan de Fuca Basin, and (B) Whidbey, Puget Sound Main, Hood Canal and South Puget Sound Basins.

Figure 31: Figure 9 from NRC (2016), "Hot spots" of yelloweye rockfish occurrence

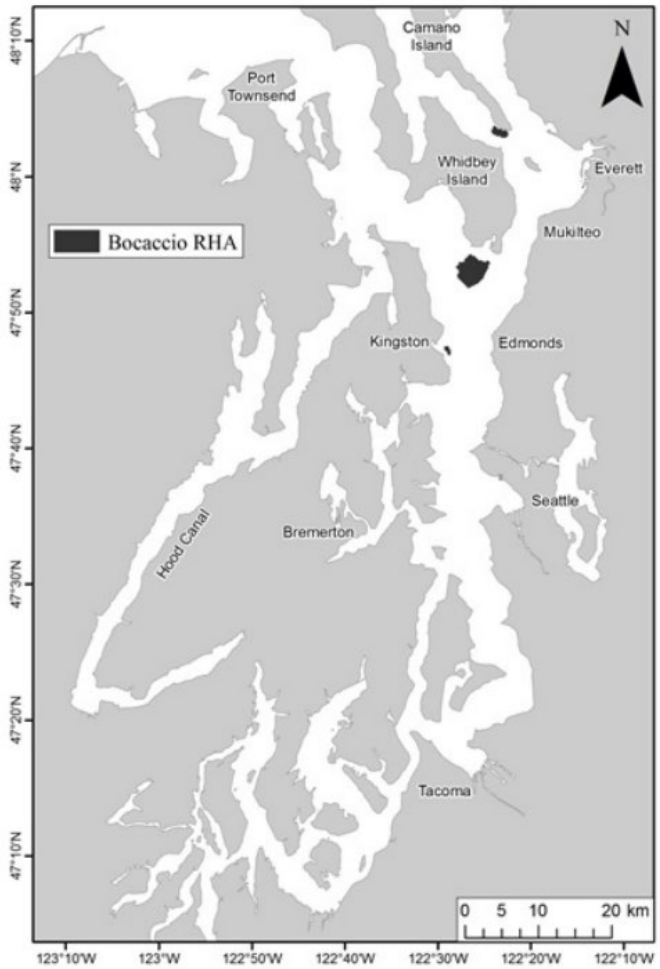


Figure 11. Spatial distribution of Bocaccio RHA's in Whidbey and Puget Sound Main Basins

Figure 32: Figure 11 from NRC (2016), "hot spots" of bocaccio occurrence

This analysis of rockfish occurrence indicates that because of the relative low abundance of yelloweye rockfish and bocaccio in the action area, few individual rockfish are expected to be exposed to water at the marine criterion levels.

Bull Trout – The ESA consultation on Oregon's toxics criteria did not explicitly address bull trout; EPA's BE (USEPA, 2008a) did not address salt water criteria for bull trout, but the USFWS BiOp included NLAA determinations in Table 1 (USFWS, 2012, p. 5) without a specific rationale. As a result, there is no previous consultation to reference for bull trout toxicity assessments and effect determinations for category 1 marine criteria. However, since bull trout are salmonid species, the toxicity assessments in EPA's Oregon Toxics WQS BE for other salmonid species can be extrapolated to bull trout. In the EPA Oregon Toxics WQS BE, toxicity tests are summarized for fish and invertebrate species for exposure to cadmium, copper, selenium, and silver in salt water. Either one or two toxicity tests were available for each of these compounds for salmonids (coho, steelhead, Chinook), but no tests were available for bull trout. These salmonid tests were used to estimate the acute effect concentration (EC) and the chronic NOECs for the salmonid ESA listed species. In all cases, the marine acute and chronic criteria were an

order of magnitude lower the EC and NOEC values, which led to NLAA determinations for the ESA salmonid species. The one exception was multiple routes of exposure and bioaccumulation effects for exposure to the marine chronic selenium criteria, which resulted in LAA determination for chronic selenium criterion.

EPA has concluded that in the absence of bull trout toxicity tests, the information in the Oregon Toxics WQS BE for the salmonid species can be extrapolated to bull trout, and the other salmonid toxicity tests are the best available data to estimate toxicity to bull trout. Thus, EPA has concluded that it is reasonable to have the same effect determinations for bull trout as for the other salmonid species for the marine category 1 criteria. For the reasons discussed above, EPA has determined that exposure at the marine criteria level for cadmium, copper, and silver is **not likely to adversely affect** bull trout and exposure at the marine selenium criteria is **likely to adversely affect** bull trout.

5.3 Effects Determinations for Category 3 Criteria

Category 3 criteria include criteria for chemicals with current use bans in place in the U.S. (Table 79).

Table 79. Assessment Category 3 Criteria

Chemical	Acute, chronic criteria	Fresh Water Criteria	Marine Water Criteria
Aldrin	CMC	x	x
Chlordane	CMC, CCC	x	x
Demeton	CCC	x	x
Dieldrin	CMC, CCC	x	x
Endosulfan-Alpha+Beta	CMC, CCC	x	x
Endrin	CMC, CCC	x	x
gamma-Hexa-chlorocyclohexane (HCH) (Lindane)	CMC	x	x
Guthion	CCC	x	x
Heptachlor	CMC, CCC	x	x
Heptachlor Epoxide	CMC, CCC	x	x
Methoxychlor	CCC	x	x
Mirex	CCC	x	x
p,p'- Dichlorodiphenyl-trichloroethane (4,4'-DDT)	CMC, CCC	x	x
Parathion	CMC, CCC	x	
Polychlorinated biphenyls	CCC	x	x
Toxaphene	CMC, CCC	x	x
Tributyltin (TBT)	CMC, CCC	x	x

EPA's NLAA determinations for these criteria are based on exposure considerations. The approach assumes that where current use bans are in place, there is low potential for co-occurrence of ESA-listed species with the chemicals in the action area, and if present, the concentration of these banned chemicals is expected to decrease over time.

The Services have previously considered no or *de minimus* exposure of ESA-listed species to banned chemicals to support NLAA determinations. The following examples of this approach are found in the biological opinions on EPA approval of Idaho's water quality standards. Relative to Idaho's DDT criteria, the NMFS BiOp concludes (NMFS (2014b), p. 279):

> Exposure of Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon, Snake River sockeye salmon and Snake River Basin steelhead and their designated critical habitat to DDT is unlikely. This is based on the inability to use DDT as a pesticide and the lack of other new discharges. An additional safety factor is provided by the applicability of a lower recreation criterion based on fish consumption. The more meaningful exposure scenario for DDT is chronic exposure to low level concentrations that have persisted in sediments of rivers, reservoirs, and lakes. On the whole, the available information indicates that the risk of adverse effects from exposure to DDT by listed Snake River salmon and Snake River Basin steelhead at concentrations at or below the chronic criterion is very low and because no new discharges will occur and any potential exposure from existing contamination will be reduced over time.

Similarly, the USFWS BiOp considers effects of EPA approval of Idaho's DDT criteria to be discountable based on exposure considerations (USFWS (2015a), p. 239):

> Also, given that DDT has not been allowed for agricultural use for 42 years, that the 1972 ban precludes any lawful releases into the environment, that the soil half-life would minimize the potential for any post-use inputs into aquatic habitats, and potential exposure via inwater disturbance/disposal of DDT contaminated sediments during USCOE actions (e.g., dredging) are minimized by their sediment screening guideline, the risks to listed species and critical habitat from the proposed DDT criteria are discountable.

For this action, exposure-based assumptions are supported by environmental baseline data in section 4.3.3.1. While many of the category 3 chemicals are persistent in the environment, and lindane has been quantified in the action area, existing monitoring data indicates that concentrations were lower than the Swinomish water quality criteria. The maximum concentration of lindane was less than 0.3% of the applicable chronic water quality criterion (see Table 56).

Several of the category 3 chemicals were sampled but never observed in quantifiable concentrations in water, sediment, or tissue in or near the action area, specifically demeton-o, demeton-s, endosulfan (alpha, beta, and sulfate), heptachlor, heptachlor epoxide, methoxychlor, mirex, parathion (including methyl parathion), toxaphene, and tributyltin (CASRN 688-73-3); see Table 57.

For those chemicals observed in quantifiable concentrations in sediment and/or animal tissue, the measured concentrations are generally lower than sediment quality criteria and ecological sediment and tissue screening benchmarks. For sediment, the exceptions were total chlordane, trans-chlordane, and PCBs. For total chlordane and trans-chlordane, the maximum sediment concentrations exceeded the chlordane sediment screening benchmark by 10%. However, the overall detection frequencies were low (4% - 6%). PCBs (as the sum of aroclors) were quantifiable in five samples (out of a total of 27), three of which exceeded the State of Washington's sediment criterion (see Table 54). The quantifiable concentrations of all of the banned chemicals in animal tissue are below the tissue screening concentrations (TSCs), except for tributyltin chloride (CASRN 1461-22-9); see Table 55. These infrequent

excursions above sediment criteria or screening benchmarks or TSCs do not demonstrate that these chemicals are present in the water column at concentrations close to water quality criteria.

Based on the bans and the available data, exposure within the action area to the chemicals in Table 79 is negligible and will continue to decline over time. Thus, the potential for adverse effects on ESA-listed species is minimal and EPA approval of the criteria is **not likely to adversely affect** the listed species or their critical habitat.

5.4 Effects Assessment Methodology for Category 4 Toxics Criteria

Assessment category 4 includes the criteria for which toxicity assessments are included in this BE (Table 80).

Table 80. Assessment Category 4 Criteria

Criterion		Fresh Water	Marine Water
Acrolein	CMC	x	
	CCC	x	
Ammonia	CMC		x
	CCC		x
Arsenic	CMC	x	x
	CCC		x
Carbaryl	CMC	x	x
	CCC	x	
Chlorine	CMC	x	x
	CCC	x	x
Chlorpyrifos	CMC	x	x
	CCC	x	x
Chromium VI	CMC		x
	CCC	x	x
Cyanide	CMC		x
	CCC		x
Diazinon	CMC	x	x
	CCC	x	x
Hydrogen Sulfide	CCC	x	x
Iron	CCC	x	
Lead	CMC	x	x
	CCC	x	x
Malathion	CCC	x	x
Nickel	CMC	x	x
	CCC	x	x
Nonylphenol	CMC	x	x
	CCC	x	x
Pentachlorophenol	CMC		x
	CCC		x
Phosphorus (elemental)	CCC		x
Silver	CMC	x	

Criterion		Fresh Water	Marine Water
Zinc	CMC	x	x
	CCC	x	x

The effects assessments and determinations described in section 5 were completed using the methods described in this section.

5.4.1 Considerations regarding aquatic life criteria

Both the acute and chronic numeric aquatic life criteria consist of three components.

1. Magnitude (the CMC for acute criteria and CCC for chronic criteria; i.e., the maximum allowed concentration of the chemical in water)

2. Duration (the averaging period associated with the criterion magnitude, generally one hour for the CMC and four days for the CCC for criteria developed after 1985)

3. Frequency (maximum allowable frequency of exceedance of the CMC and CCC, generally once in three years for criteria developed after 1985).

For example, EPA's aquatic life criteria recommendation for chlorine is that the 1-hour average concentration of chlorine may not exceed 19 ug/L more than once every 3 years on average, and the 4-day average concentration of total residual chlorine may not exceed 11 ug/L more than once every 3 years on average (USEPA, 1986e). The Tribe's adopted aquatic life criteria are consistent with EPA's recommendations under CWA section 304(a), which include all three components for most chemicals.

The approval of a water quality standard has no immediate effect on the environment or listed species. However, the implementation of the approved standard, for example in a NPDES permit or TMDL, may result in a chemical concentration in surface water that may affect listed species. Since the plain language of the standard allows a chemical to be at criteria levels for a specific timeframe in surface waters in the action area, a toxicity evaluation generally includes an evaluation of whether the presence of a particular chemical at the criteria level is likely to adversely affect any pertinent Federally listed species or their critical habitat.

To date, EPA Region 10 biological evaluations of criteria and standards have evaluated the magnitude component of the criteria, generally making the conservative assumption that listed species are constantly exposed to the criteria concentrations for the duration of the toxicity tests. In addition to being an unrealistic representation of chemical exposure in the environment, this assumption ignores an evaluation of the exposure duration and frequency of exceedance components of a criterion.

Advances in the field of aquatic toxicology now permit EPA to quantitatively evaluate the protectiveness of all three components of aquatic life criteria, not just the magnitude component, to more fully evaluate whether a criterion in its entirety is likely to adversely affect a listed species. The effects assessment methodology described in this section addresses all three components of aquatic life criteria and provides a more complete and detailed evaluation of whether EPA approval of criteria is likely to adversely affect any listed species.

Toxicity assessments in this biological evaluation are performed using a standard USEPA (1998) ecological risk assessment paradigm. The ecological risk assessment paradigm describes a framework for evaluating the potential for adverse organism-level effects to occur to aquatic life and aquatic-dependent wildlife as a result of hypothetical exposure to chemicals at criteria levels (including magnitude, duration, and frequency).

A review of toxicity literature was completed for chemicals when required to assess criteria in the four assessment categories. Appendix B describes the literature search methodology, which was completed using Appendix D of the USEPA (2006b) Draft Framework for Conducting Biological Evaluations.

5.4.2 Approach to assessments

Ecological risk assessments, and the effects assessments in this BE, are based on two primary elements: characterization of exposure, and characterization of population-relevant toxicological effects. These two elements are evaluated within the three phases of an ecological risk assessment: problem formulation, analysis, and risk characterization (USEPA, 1998).

Ecological effects assessment describes the potential for and types of effects that could be expected from exposure to criteria chemicals at criteria levels. Ecological effects assessments have historically been called toxicity assessments by risk assessors. For consistency with the historically used terminology, we will use the term toxicity assessment interchangeably with effects assessment throughout this biological evaluation.

For this BE, we consider toxicological endpoints that can be related to adverse effects on either individual fitness (an organism's ability to perpetuate itself as measured by its reproductive success (Pianka, 1983)), or adverse effects at population or higher levels of biological organization. Toxicological endpoints meeting this ecologically relevant effect guideline are population level effects, and effects at the organism level of biological organization (survival, reproduction, growth and some behaviors such as ability to avoid predation). Methods used to identify literature studies with the information needed to quantify adverse effects on listed species, and with acceptable levels of data quality are given in Appendix B.

The toxicity evaluations followed a tiered approach, similar to the approach used for an ecological risk assessment, to determine whether exposure to each criterion chemical at the criterion level would be likely to adversely affect an action area listed species. Each tier represents a successively more focused assessment that addresses the three components of the criteria, magnitude, duration, and frequency. If a NLAA determination is made at a lower tier, then analyses at the next tier is not necessary at it will not provide additional useful information.

A summary of the assessments performed in Tiers 1-4 is provided in the next several paragraphs.

5.4.2.1 Screening tier assessment (tier 1)

Tier 1 Assessment – Akin to a screening level ecological risk assessment, this Tier 1 analysis defines the action area, identifies listed species range and critical habitat locations within the action area, characterizes the current status, habitat, and environmental baseline of listed species and critical habitats in the action area. Where the species range and critical habitat do not overlap the action area, EPA made a No Effect determination for those species and they are not evaluated further in the BE.

Where the range and/or critical habitat overlaps the action area, species received a May Affect determination and were advanced to a more comprehensive baseline effects assessment in Tier 2 of the methodology.

5.4.2.2 Description of assessments conducted in tiers 2 through 4

During the analysis phase of the effects assessments (Tiers 2, 3 and 4), comparable to a baseline ecological risk assessment, two primary tasks are performed in each tier: ecological effects assessment and ecological exposure assessment. Within the ecological effects assessment, an assumption is made in Tier 2 that ESA listed species are constantly exposed to acute criteria concentrations of chemicals for the 96 hour exposure duration of an acute toxicity test. The conservative Tier 2 assumption for chronic exposures is that organisms are exposed to chronic criteria concentrations for durations longer (e.g., 21 days) than the 4-day averaging period relevant to most chronic criteria. Within the exposure assessment, data are evaluated to determine how likely listed species are to be exposed to the chemicals associated with the water quality criteria adopted by the Swinomish Tribe. If it is determined that the exposure concentration does not adversely affect a listed species and it is unlikely that the chemical concentration will increase in the future, the species is not at risk from those chemicals.

Tier 2 Assessment – Tier 2, the primary effects assessment, evaluates the magnitude (concentration) component of the Swinomish criteria while assuming a listed species is constantly exposed to the criteria concentrations. If the adverse effect concentration as determined in this tier of the effects assessment under the constant exposure assumption for a listed species is higher than the concentration value of the criterion, the criterion is considered to be protective of the listed species, and EPA will make a NLAA determination for the chemical/listed species pair (for direct effects). If the adverse effect concentration is lower than or equal to the criterion, the chemical/listed species pair will be carried forward into Tier 3 of the effects assessment without a Tier 2 determination of the adverse effect of the EPA approval of the standard.

Tier 2 also includes assessment of the following indirect effects of the EPA action: effects assessment on prey species of ESA listed species, assessment of effects on listed species from multiple routes of exposure (e.g. water column exposure, dietary exposure) to a chemical, and effect assessments on aquatic-dependent ESA listed species (i.e. mammals and birds that prey partially or completely feed on fully aquatic species such as fish).

Tier 3 Assessment – Tier 3, the secondary effects assessment, addresses the duration component of the criteria (1-hour exposure for acute criteria, 4-day exposure for chronic criteria) in combination with the criteria magnitude assessment in Tier 2. If the adverse effect concentration as determined for a 1-hour (acute criterion) or 4-day (chronic criterion) exposure in this tier of the effects assessment for a listed species is higher than the criterion, the criterion is considered to be protective of the listed species, and EPA will make a NLAA determination for the chemical/listed species pair. If the adverse effect concentration for a 1-hour or 4-day exposure is lower than or equal to the criterion, the chemical/listed species pair will be carried forward into Tier 4 of the effects assessment.

5.4.3 Tier 2 methodologies – primary baseline effects assessment

5.4.3.1 Tier 2 acute toxicity effects assessment methods

There are three general approaches to identifying the acute toxicity 96-hour LC_{50} concentrations that are the starting point from which acute adverse effect concentrations can be calculated or estimated.

1. Empirically measured toxicity data
2. Interspecies correlation estimation modeling
3. Species sensitivity distribution (SSD) analyses

The goal of the above procedures is to identify minimum effect threshold concentrations for each chemical/listed species pair. The above procedures yield acute LC_{50} concentrations, which are adjusted downwards to establish minimum effect thresholds as described in this section.

Using empirically measured toxicity data for listed species is the preferred approach for assessing adverse effects of chemicals to listed species. Unfortunately, empirical data are unavailable for most listed species/chemical pairs. This data gap is the reason that empirical toxicity data from taxonomically closely related species to listed species is used to estimate the toxicity of most chemicals to the listed species. Table 81 below presents the hierarchy of methods used in this BE to calculate or estimate the acute toxicity of chemicals to listed species. When empirical data are not available, interspecies correlation estimation (ICE) methods are preferred to SSDs because of the increased statistical power and closer taxonomic relationships to listed species available in ICE models relative to SSDs. In this BE, SSD use is generally limited to providing toxicity data for listed species without empirical toxicity data at the family level or lower. Detailed descriptions of the methods listed in Table 81 immediately follow the table.

Table 81. Hierarchical Biological Evaluation Methodology for Tier 2 Assessment of Acute Water Quality Criteria

Tier	Acute LC_{50} Determination Method
1	Empirical Acute LC_{50} Data for the Species (SMAV)
2	Empirical Acute LC_{50} Data for Genus-Level Surrogate Species (GMAV)
3	Species-level ICE-Estimated LC_{50} Data for the Species (ICE-SMAV)
4	Genus-level ICE-Estimated LC_{50} Data for the Species (ICE-GMAV)
5	Family-level ICE-Estimated LC_{50} Data for the Species (ICE-FMAV)
6	SSD Acute LC_{50} Data for Order-Level Surrogate Species (OMAV)
7	SSD Acute LC_{50} Data for Class-Level Surrogate Species (CMAV)

5.4.3.1.1 Conversion of empirical acute toxicity data to protective concentration data

An LC_{50} concentration where 50% of test organisms are killed is obviously not a protective concentration for the test species. Acute criteria derived directly from LC_{50} data would not provide criteria protective of aquatic life. Both acute water quality criteria derived by EPA and the effects assessment of this BE therefore convert the empirical acute toxicity data to concentrations with low (LC_{LOW}) or no discernable effects on survival of aquatic species. LC_{LOW} values are intended to be within the concentration range where 0-10% mortality occurs in an acute toxicity test. This is the allowable control mortality range of standard aquatic toxicity testing protocols, and is the range where adverse effects of the test chemical

cannot be separated from any adverse effects of the test procedure itself or the health of test organisms at the initiation of the toxicity test.

The first step to establishing a low effect concentration using empirical data for a chemical-species combination is to obtain LC_{50} data for each species where toxicological data are available. Once the LC_{50} values are obtained, each LC_{50} or the lower 95% confidence interval of the LC_{50}, if available is divided by 2.27 to calculate the acute minimum effect threshold value for each species. The ratio of 2.27 is used to convert the LC_{50} concentration to an "LC_{LOW}" value that is expected to result in little or no toxicity to the test species.

In accordance with the Guidelines (C.E. Stephan et al., 1985), the FAV is divided by 2.0 to calculate the acute criteria concentration, the CMC. The basis for this adjustment factor is an analysis of data from 219 acute toxicity tests showing that the mean concentration lethal to 0-10% of the test population was 0.44 times the LC_{50}, or the LC_{50} divided by 2.27. The data and analysis on which the 2.27 value is based is described in the Federal Register on May 18, 1978 (43 FR 21506-21518). Briefly, the analysis consisted of calculating the geometric mean of the ratios of the highest concentration (HC) affecting or killing 0-10% of organisms divided by the LC_{50} or EC_{50} for the same organisms in the same acute test (i.e. the geometric mean of 219 HC/LC_{50} ratios from toxicity tests with a variety of chemicals). Best professional judgment was used to round the FAV "adjustment factor" of 2.27 to 2 in revisions of the Guidelines that were published (C.E. Stephan et al., 1985) subsequent to the 1978 Federal Register notice. We have used the 2.27 adjustment factor in this BE to add an additional degree of conservatism in assessing effects on ESA listed species. The use of the 2.27 versus the 2.0 adjustment factor during calculation of acute minimum effect threshold values results in acute minimum effect thresholds 11.9% lower than is calculated by dividing an FAV by two.

If sufficient information is available in the literature, data from individual studies can be used to calculate an LC_5 concentration directly, rather than inferring a minimum effect threshold from the LC_{50}. The LC_5 concentrations can then be used as minimum effect thresholds for comparison to criteria concentrations without further adjustment.

5.4.3.1.2 Interspecies correlation estimation (ICE) methodology

ICE models are statistical regressions that permit estimations of LC_{50}s to be made for a species or higher taxon (genus, family) that has no measured acute toxicity data using a related species for which five or more LC_{50}s have been measured. The detailed description of how ICE models were developed and their use to estimate LC_{50}s for taxa for which no toxicity information is available is given in Asfaw et al. Additional information including a bibliography and access to the model is available at EPA's WebICE webpage at https://www3.epa.gov/webice/index.html.

5.4.3.1.3 Species sensitivity distribution (SSD) methodology

SSDs are cumulative distribution functions of toxicity values for multiple species and are used to estimate a chemical concentration that is protective of most test species (e.g., 95%) by estimating the concentration at a corresponding percentile (e.g., 5th) of the distribution (De Zwart, 2002). SSDs are based on the assumption that available toxicity data for a group of species are representative of the responses to chemical exposures to other species for which toxicity data are not available, and that a statistical description of the toxicity data can be generated to calculate the proportion of all species that would be adversely affected by exposure to a chemical at a given concentration. As long as the

toxicological endpoint for each species is the same (e.g. an LC_{50}, EC_{20}, NOEC, etc.), an SSD can be generated for any toxicological endpoint and mix of species. This feature of SSDs is one of the great advantages of SSDs relative to other endpoint estimation methodologies, in that they explicitly consider the results of all available toxicological information for a given assemblage of species.

5.4.3.1.4 Tier 2 acute toxicity effects determination

The acute minimum effect threshold for each ESA listed species, regardless of how it is derived (i.e. direct measure, ICE, SSD) is compared to the acute water quality criterion and the following determinations are made:

If the Acute Minimum Effect Threshold Concentration > the Acute Criterion Magnitude, then:

1. Consideration of criterion duration and frequency in Tiers 3 and 4 is not required.
2. Consider adverse effects on prey species (indirect effects) and the effects of multiple routes of exposure (another direct effect assessment) prior to making a final effects determination.
3. Criterion is determined to be NLAA if both direct and indirect effects of the criterion are determined as NLAA.
4. Criterion is determined to be LAA if exposure at the criterion concentration is determined to adversely affect prey species and/or adversely affects the listed species due to chemical exposure via multiple exposure routes (i.e. dietary ingestion in addition to water column exposure).

If the Acute Minimum Effect Threshold Concentration ≤ the Acute Criterion Magnitude, then:

1. Perform a Tier 3 assessment of criterion duration for those listed species with acute effect threshold concentrations lower than criterion magnitude.
2. Consider adverse effects on prey of the listed species, and effects of multiple routes of chemical exposure to the listed species prior to making a final effects determination.

5.4.3.2 Tier 2 chronic toxicity effects assessment methods

As was the case when performing the Tier 2 acute toxicity assessment of listed species, we use a hierarchical system of methods to evaluate chronic toxicity, with the specific method used for each chemical/listed species pair dependent on the type of toxicity data available.

The initial step of a chronic effect assessment involves compiling the available chronic toxicity data (EC_{20} or NOEC) for all listed species in the action area. Empirical chronic toxicity data are less common than acute toxicity data, and likely will not be available for all listed species. This means that to a greater extent than is done for acute toxicity data, chronic toxicity data for listed species will have to be derived either from surrogate species toxicity data or modeled chronic toxicity data.

This limited availability of empirical chronic toxicity data is the reason that using empirical toxicity data from taxonomically closely related species to listed species is the primary approach used to estimate the toxicity of most chemicals to listed species. Table 82 below presents the hierarchy of methods used in this BE to calculate or estimate the chronic toxicity of chemicals to listed species. When empirical data are not available, interspecies correlation estimation (ICE) methods are preferred to SSDs because of the

increased statistical power and closer taxonomic relationships to listed species available in ICE models relative to SSDs. Modeled acute LC50 data from ICE for listed species is divided by the acute-chronic ratio for the chemical to estimate the chronic NOEC for each listed species. SSD use is generally limited to estimating toxicity data for listed species without empirical toxicity data at the family, order or class level. Detailed descriptions of the methods listed in Table 82 follow the table.

Where empirical chronic toxicity data are unavailable, EPA's approach is to estimate chronic toxicity data for listed species from the available chronic toxicity data from the most taxonomically related species to the listed species. The hierarchy for chronic effects assessment is given in Table 82.

Table 82. Hierarchical Biological Evaluation Methodology for Tier 2 Assessment of Chronic Water Quality Criteria

Tier	Chronic NOEC Determination Method
1	Empirical Chronic NOEC Data for the Listed Species
2	Empirical Chronic Data ≤ EC_{20} for the Listed Species
3	Empirical Acute LC_{50} Data for the Listed Species Divided by the Acute-Chronic Ratio (ACR) for the Chemical
4	Empirical Chronic NOEC Data for Genus-Level Surrogate Species for the Listed Species
5	Empirical Chronic Data ≤ EC_{20} for Genus-Level Surrogate Species for the Listed Species
6	Species-level ICE-Estimated Acute LC_{50} Data for the Listed Species Divided by the ACR for the Chemical
7	Genus-level ICE-Estimated Acute LC_{50} Data for the Listed Species Divided by the ACR for the Chemical
8	Empirical Chronic NOEC Data for Family-Level Surrogate Species for the Listed Species
9	Empirical Chronic Data ≤ EC_{20} for Family-Level Surrogate Species for the Listed Species
10	Family-level ICE-Estimated Acute LC_{50} Data for the Listed Species Divided by the ACR for the Chemical
11	SSD 5th Percentile NOEC Data for Order-Level Surrogate Species for the Listed Species
12	SSD 5th Percentile ≤ EC_{20} Data for Order-Level Surrogate Species for the Listed Species
13	SSD 5th Percentile NOEC Data for Class-Level Surrogate Species for the Listed Species
14	SSD 5th Percentile ≤ EC_{20} Data for Class-Level Surrogate Species for the Listed Species
15	SSD 5th Percentile Acute LC_{50} Data for Order-Level Surrogate Species for the Listed Species Divided by the ACR for the Chemical
16	SSD 5th Percentile Acute LC_{50} Data for Class-Level Surrogate Species for the Listed Species Divided by the ACR for the Chemical

The goal of the above procedures is to identify a chronic no effect threshold concentration for each chemical/listed species pair. ICE models and SSD generation follows the basic procedures described in the acute effects assessment section of this methodology. The differences are that ICE modeled acute LC_{50} data are converted to chronic NOEC values by dividing the acute LC_{50} by the ACR for the chemical before the calculated chronic NOEC for a listed species is compared to the chronic water quality criterion, and that chronic NOEC or EC_{20} or lower data are used to derive the 5th percentile of the SSD, not acute LC_{50}/2.27 data as is done for the acute threshold effect concentration calculations used in the acute effects assessments..

In ecological risk assessments, effects on species where empirical toxicity data are lacking are usually not quantified. Instead, they are presented as uncertainties in the risk assessment. Chronic NOECs based on toxicity information from order or class data are considered to be highly uncertain in this BE.

5.4.3.2.1 Tier 2 chronic aquatic toxicity effects determination

The chronic minimum effect threshold for each ESA listed species, regardless of how it is derived (i.e. direct measure, ICE, SSD) is compared to the chronic water quality criterion and the following determinations are made:

If the Chronic Minimum Effect Threshold Concentration > the Chronic Criterion Magnitude

1. Consideration of criterion duration and frequency in Tiers 3 and 4 is not required.

2. Consider adverse effect on prey of the listed species (indirect effects), and direct effects of multiple routes of chemical exposure to the listed species prior to making a final effects determination.

3. Criterion is determined to be NLAA if both direct and indirect effects of the criterion are determined as NLAA.

4. Criterion is determined to be LAA if the criterion is determined to adversely affect prey species of the listed species and/or adversely affects the listed species due to chemical exposure via multiple exposure routes (i.e. dietary ingestion in addition to water column exposure)

If the Chronic Minimum Effect Threshold Concentration ≤ the Chronic Criterion Magnitude

1. Perform a Tier 3 assessment of criterion duration for those listed species with chronic effect threshold concentrations lower than criterion magnitude.

2. Consider indirect effects on adverse effect on prey of the listed species, and effects of multiple routes of chemical exposure to the listed species prior to making a final effects determination.

5.4.3.3 Tier 2 methodology for evaluation of effects from multiple routes of chemical exposure

EPA's derivation of ambient aquatic life criteria includes only studies that expose aquatic species to pollutants in water, as opposed to studies that expose aquatic species to chemicals in sediment or their diet in the natural environment. Simultaneous exposure of aquatic species to multiple sources of a pollutant may result in greater toxic effects than would be predicted from water column toxicity tests alone. This section of the methodology describes an approach that evaluates the amount of risk that might occur on a chronic basis through other routes of exposure not considered during criteria development, and which may not be accounted for in the analyses described in the previous sections. Investigations of the effects of other routes of exposure such as pollutant ingestion from diet are much less common than water only toxicity tests, and do not have well standardized methods. As such, definitive published guidance on how adverse effects from these other routes of exposure should be analyzed is unavailable.

Meyer (2005), after an extensive review of the toxicity of diet borne metals to aquatic species, concluded that the state of the science is currently not advanced enough to explicitly incorporate dietary metal toxicity into aquatic life criteria derivation under current regulatory frameworks. In part, this is due to the known differences in internal organ distributions of chemicals in aquatic species depending on whether the metal exposure occurs from a waterborne or dietary exposure (Clearwater, Farag, & Meyer, 2002). Differences in the organs in which pollutants are accumulated can result in pollutants having different sites, modes and mechanisms of toxic action depending on the exposure route by which aquatic species obtain their internal pollutant dose. This is one explanation for the multiple modes of action of metals, including the variation in mode of action during short term (damage to respiratory surfaces) and long term (systemic injuries to organs and biochemical processes) exposures. Although the processes controlling waterborne and dietary pollutant uptake in aquatic species are generally independent of each other, whole body pollutant concentrations are integrators of tissue residues resulting from waterborne and dietary pollutant uptake.

Numerous investigators have presented evidence that whole body residues are a reasonable surrogate for the chemical concentration at the site of toxic action (Connell, Chaisuksant, & Yu, 1999; Cook, Carlson, & Lee II, 1992; Escher & Hermens, 2004; Peter F Landrum, Eadie, & Faust, 1991; L. S. McCarty & Mackay, 1993; van Wezel & Opperhuizen, 1995). This is particularly true for organic compounds (Meyer, 2005), and has been considered a suitable assumption for making screening level predictions with metals (U. Borgmann, Norwood, Reynoldson, & Rosa, 2001; P. M. Chapman, Wang, Janssen, Goulet, & Kamunde, 2003; Meyer, 2005; B. K. Shephard, 1998). If the whole body tissue residue of a bioaccumulated chemical can be related to the concentration of that chemical in water, the level of protection water quality criteria provide from the adverse effects of chemicals obtained via multiple exposure pathways can be evaluated by comparing the amount of chemical bioaccumulated from exposure at criteria concentrations to residue levels associated with adverse effects in aquatic species.

The approach used to evaluate toxicity to aquatic species from multiple concurrent routes of exposure in this BE is based on a fundamental principle of toxicology. The dose-response relationship describes the proportionality of the chemical concentration in tissue at the site of toxic action to the toxic response (D. L. Eaton & Klaassen, 1995). The chemical concentrations in exposure media such as water or diet that are used as surrogates for the actual dose of toxic chemical in tissue have limitations when used to assess ecological risks to aquatic biota, some of which are listed below (modified from Landrum et al. [(Peter F. Landrum, Lydy, & Lee, 1992)]).

- The bioavailable and toxicologically active fraction of the total exposure media chemical concentration may not be known
- It does not consider multiple uptake routes of chemicals
- Intermittent, pulsed or variable exposures cannot be readily assessed
- Chemical mixture toxicity cannot be easily assessed
- Exposure duration (i.e. bioaccumulation kinetics) effects on toxicity may not be well defined
- Metabolic transformations which reduce or enhance parent compound toxicity are not considered
- Animal behavior such as seasonal migration or toxicant avoidance is not considered
- Acclimation to toxicants can yield differential sensitivity to exposure media concentrations under different exposure regimes

By associating the toxic response of aquatic biota with the body burden of the chemical causing the effect, the above complicating factors can largely be eliminated. Toxic effects can then be directly expressed as a function of tissue residues.

In this BE, evaluation of tissue residues bioaccumulated from multiple exposure routes are first described as residues bioconcentrated from water only exposure to chronic criteria concentrations, then as residues bioaccumulated concurrently from waterborne and dietary exposure to chronic criteria concentrations. The tissue residues in the prey species ingested by aquatic biota are assumed to be at steady state with respect to their exposure to chronic criteria concentrations. Finally, the bioaccumulated tissue residues in aquatic species exposed to criteria concentrations via concurrent waterborne and dietary exposure are compared to available literature associating whole body tissue residues with adverse effects on reproduction and growth. Appendix C contains the effects assessment data for multiple routes of exposure.

Calculation of whole-body tissue residues bioconcentrated from water were derived from the one compartment first order kinetic (1CFOK) toxicological model shown in Equation 5.2, assuming the organism tissue contains no chemical at the beginning of its exposure period to the chemical.

Equation 5.2:

$$\frac{dC_b}{dt} = (k_u \times C_w) - (k_e \times C_b)$$

where:
C_b = chemical concentration in biota (mg/kg)
t = time (hours)
k_u = chemical uptake rate constant (L/kg/hr)
C_w = chemical concentration in water (mg/L)
k_e = chemical elimination rate constant $(\text{hour})^{-1}$

Xu and Pascoe (1993) present a modified version of Equation 5.2 that allows the assumption that an organism starts with no residue of the pollutant to be relaxed. This allows the 1CFOK model to be used with nutritionally essential metals where the organism has to contain some level of the metal to survive. The modification which accounts for the body burden of nutritionally essential metal results in the BCF = $(k_u/k_e + C_e/C_w)$ where C_e is the concentration of the essential metal in tissue (mg/kg).

If the chemical concentration in water is assumed to be constant, Equation 5.2 may be exactly integrated to yield Equation 5.3.

Equation 5.3:

$$C_b = C_w \times \frac{k_u}{k_e} \times \left(1 - e^{-k_e t}\right) + \left(C_{b(t=0)} e^{-k_e t}\right)$$

If it is further assumed that an organism starts with no tissue residue of the chemical of interest and the tissue residue is at steady state with respect to the water concentration, Equation 5.3 reduces to Equation 5.4.

Equation 5.4:

$$C_b = C_w \times \frac{k_u}{k_e}$$

By redefining the terms in Equation 5.4, C_w becomes a chronic water quality criterion (CWQC), k_u/k_e is a BCF, and Equation 5.4 with redefined terms can be used to derive the chemical concentration bioconcentrated in the whole body tissue of an aquatic species exposed to a chemical concentration equal to a chronic water quality criterion, as shown in Equation 5.5.

Equation 5.5:

$$C_b = CWQC \times BCF$$

All tissue residues in this BE, unless specifically noted otherwise in the text are presented as mg/kg whole body wet weight (WBWW).

Equation 5.2 can be expanded to incorporate pollutant uptake from both water and ingestion of contaminated prey, as shown in Equation 5.6.

Equation 5.6:

$$\frac{dC_b}{dt} = (k_u \times C_w) + \left(AE \times IR \times C_{prey}\right) - ([k_e + g] \times C_b)$$

where: AE = pollutant assimilation efficiency from ingested prey (expressed as a proportion)
IR = food ingestion rate (mg/kg/day)
C_{prey} = chemical concentration in prey (mg/kg)
g = exponential growth rate of the organism (day^{-1})

and all other terms are defined as per Equation 5.2.

If it is assumed that the organism is at steady state (i.e. chemical uptake is equal to the sum of chemical elimination plus growth dilution, or $dC_b/d_t = 0$, a constant tissue residue in the organism is reached (Equation 5.8).

Equation 5.8:

$$C_b = \frac{k_u \times C_w}{k_e + g} + \frac{AE \times IR \times C_{prey}}{k_e + g} = \frac{(k_u \times C_w) + \left(AE \times IR \times C_{prey}\right)}{(k_e + g)}$$

Equation 5.8 calculates the tissue residue in an aquatic species resulting from its exposure to multiple sources of a chemical, the definition of a bioaccumulation factor (BAF). Equation 5.8 is equivalent to the DYMBAM (dynamic multi bioaccumulation model) proposed by Luoma and Rainbow (Luoma & Rainbow, 2005) as a unifying concept to explain why trace metal bioaccumulation differs among metals, species and environmental conditions. As was done with Equation 5.4, redefinition of the terms in Equation 5.8 results in C_w becoming a chronic water quality criterion (CWQC), while the remaining terms describe a BAF, and Equation 5.8 with redefined terms can be used to derive the chemical concentration bioaccumulated by an aquatic species exposed to a chemical concentration equal to a chronic water quality criterion, as shown in Equation 5.9.

Equation 5.9:

$$C_b = CWQC \times BAF$$

Although simple in form, Equations 5.5 and 5.9 are derived from 1CFOK models of bioaccumulation in aquatic species. The following assumptions were made to simplify the more complex kinetic forms of the bioaccumulation models (Equations 5.3 and 5.8) to forms that required only water quality criteria values and BAFs (or BCFs) to calculate tissue residues that protect aquatic life from the adverse effect of pollutant exposure via multiple routes of exposure.

Chronic water quality criteria for all chemicals undergoing consultation in this BE except ammonia, chlorine, elemental phosphorus and sulfide were multiplied by chemical specific BAFs to define tissue residues in aquatic species associated with a low likelihood of eliciting adverse effects. The BCF values used in this multiple routes of exposure analysis are presented in Appendix C (Table C1).

The use of "nationally representative" BAFs has been proposed by EPA (USEPA, 2005c) to translate aquatic life tissue criteria into media concentrations in situations where site-specific BAFs are lacking. An EPA (USEPA, 2006c) Science Advisory Board consultation on the EPA (USEPA, 2005c) *Proposed Revisions to Aquatic Life Guidelines, Tissue-Based Criteria for Bioaccumulative Chemicals* was complimentary of the approach, and expressed support for incorporation of kinetic modeling and tissue based approaches into the future development of water quality criteria. The freshwater selenium criteria adopted by the Swinomish Tribe is an example of tissue-based criteria. The equation for the kinetic modeling approach presented by EPA to the Science Advisory Board during their consultation on revisions to EPA's national water quality criteria derivation methodology was the integral form of Equation 5.3 of this methodology.

Equation 5.3 as used in this BE is the basis for assessing risks to ESA listed aquatic species from multiple routes of exposure to criteria concentrations. Central tendency BCF values used in conjunction with water quality criteria have successfully derived ecological tissue guidelines that are protective of aquatic species (Dyer et al., 2000; B. K. Shephard, 1998). The supportive Science Advisory Board review of the use of nationally representative BAFs during the translation of tissue criteria to water column concentrations further supports the use of central tendency BCF and BAF values for aquatic species where they are required for the evaluation of the effects of water quality criteria on all aquatic species.

In theory, exposure of aquatic species to chemical concentrations in water equal to EPA chronic water quality criteria, which are designed to be protective of 95% of aquatic genera, should result in tissue residues lower than 95% of the residues associated with adverse effects on reproduction and growth. The product of a chronic water quality criterion and a BAF, identified as a tissue screening concentration (Dyer et al., 2000; B. K. Shephard, 1998), is compared to measured whole body residues associated with adverse effects on reproduction and growth in aquatic species.

This comparison will permit an evaluation of potential hazard to aquatic species from multiple routes of exposure (specifically, waterborne exposure and bioaccumulation into prey species consumed by aquatic biota) to chronic criteria concentrations. The approach is not new, having been previously proposed by multiple investigators (ACOE et al., 2005; E. J. Calabrese & Baldwin, 1993; Cook et al., 1992; Dyer et al., 2000; L. McCarty et al., 2011; Nendza, Herbst, Kussatz, & Gies, 1997; B. K. Shephard, 1998). This approach has been used frequently during ecological risk assessments for aquatic species of both metals (EPA 2006b) and organics (Johnston, Halkola, & George, 2003), and in both freshwater (Dyer et al., 2000) and marine (Micheletti, Critto, Carlon, & Marcomini, 2004) systems, including during evaluations of highly bioaccumulative chemicals such as mercury (Vidal & Horne, 2003) and DDT (Riedel, Schlenk, Frank, & Costa-Pierce, 2002).

Whole body tissue residues associated with adverse effects on reproduction and growth were compiled from literature available in Jarvinen and Ankley (Alfred William Jarvinen & Ankley, 1999), the Environmental Residue Effects Database (ERED, available at http://el.erdc.usace.army.mil/ered/) and other sources identified in the primary literature. The lowest observed effect residue (LOER) on reproduction and/or growth for each species for which information is available is compiled, rank ordered from the lowest to the highest LOER value, then compared to the tissue screening concentration (TSC) to evaluate the potential hazard from multiple routes of exposure to criteria chemicals.

5.4.3.4 Tier 2 methodology for aquatic-dependent wildlife evaluations

Aquatic dependent wildlife species are mammals, birds and reptiles whose diet contains an appreciable proportion of aquatic organisms. The three aquatic dependent wildlife species evaluated in this BE are SRKW, humpback whale and marbled murrelet. Only compounds with known bioaccumulative properties were evaluated for their effects on aquatic-dependent wildlife, as described in section 5.4.3.4.1.

5.4.3.4.1 Bioaccumulation potential of toxic substances

For marbled murrelet, humpback whale, and SRKW, ingestion of prey items that were exposed to waters at the criteria level is unlikely to result in adverse effects for all criteria chemicals that do not have bioaccumulation potential. Chemicals with a modeled or measured $\log_{10}$ bioconcentration factor (BCF), bioaccumulation factor (BAF), or (for organic chemicals), an octanol-water partition coefficient (K_{ow}) > 3.0 were considered to have a potential to bioaccumulate from water into prey consumed by ESA-listed aquatic dependent species (see Table 83 for a list of chemicals with these properties shown). For metals, EPA considered other information (e.g., the scientific literature, information within 304(a) documents, etc) and determined that selenium is bioaccumulative (Presser & Luoma, 2010) and its effects should be assessed in aquatic-dependent wildlife. The Tribe's selenium criteria are consistent with EPA's current recommendations under CWA section 304(a) and fall under assessment category 2 (see section 5.2).

Table 83. Bioaccumulation potential for chemicals in this BE

The CompTox Chemicals Dashboard (https://comptox.epa.gov/dashboard/) was queried on May 26, 2022 for this information. Log_{10} bioconcentration factor (BCF) and the octanol-water partition coefficients (K_{ow}) are the average of available data at the date of the query. Blank cells are those with negligible (i.e., less than $\log_{10}$ 1.0) BCF or Kow values. High bioaccumulation potential is defined as a $\log_{10}$ BCF and a $\log K_{ow}$ being >3.0.

Chemical	CASRN	Assessment Category		log10BCF	log10Kow	High bio-accumulation potential?
		FW	SW			
Assessment category 1 and 2 criteria						
Cadmium	7440-43-9	1	1			
Chromium (III)	16065-83-1	1	n/a			
Copper	7440-50-8	2	1			
Selenium	7782-49-2	2	1			Yes
Assessment category 3 criteria						
Aldrin	309-00-2	3	3	3.68	6.5	Yes
Chlordane	57-74-9	3	3	4.12	6.12	Yes

Chemical	CASRN	Assessment Category		log10BCF	log10Kow	High bio-accumulation potential?
		FW	SW			
DDT	50-29-3	3	3	4.21	6.91	Yes
Demeton	8065-48-3	3	3			
Dieldrin	60-57-1	3	3	3.65	5.12	Yes
Endosulfan	115-29-7	3	3	0.59	3.83	No
Endrin	72-20-8	3	3	3.65	5.12	Yes
Guthion	86-50-0	3	3	0.99	2.75	No
Heptachlor	76-44-8	3	3	3.82	6.1	Yes
Heptachlor epoxide B	1024-57-3	3	3	3.71	4.98	Yes
Lindane	58-89-9	3	3	2.8	3.72	No
Methoxychlor	72-43-5	3	3	2.67	5.08	No
Mirex	2385-85-5	3	3	4.31	6.89	Yes
Parathion	56-38-2	3	n/a	1.92	3.83	No
Polychlorinated biphenyls	1336-36-3	3	3	4.34	7.75	Yes
Toxaphene	8001-35-2	3	3	3.48	5.78	Yes
Tributyltin	688-73-3	3	3	2.4	4.1	No
Assessment category 4 criteria						
Acrolein	107-02-8	4	n/a			
Ammonia	7664-41-7	2	4			
Arsenic	7440-38-2	4	4			
Carbaryl	63-25-2	4	4	0.83	2.36	No
Chlorine	7782-50-5	4	4			
Chlorpyrifos	2921-88-2	4	4	2.98	4.96	No
Chromium (VI) ion	18540-29-9	1, 4	4			
Cyanide	57-12-5	4	4			
Diazinon	333-41-5	4	4	1.92	3.81	No
Hydrogen sulfide	7783-06-4	4	4			
Iron	7439-89-6	4	n/a			
Lead	7439-92-1	4	4			
Malathion	121-75-5	4	4	1.33	2.36	No
Nickel	7440-02-0	4	4			
Nonylphenol	84852-15-3	4	4	2.82	5.77	No
Pentachlorophenol	87-86-5	1	4	2.85	5.12	No
Phosphorus	7723-14-0	4	4			
Silver	7440-22-4	4	1			
Zinc	7440-66-6	4	4			

5.4.3.4.2 Tier 2 chronic wildlife toxicity effects determination

The chronic wildlife threshold effect concentration for each ESA listed wildlife species is based on toxicity reference values derived from toxicity studies where both water ingestion and prey ingestion rates (or administered doses) were monitored during the wildlife toxicity study. As such, the wildlife TRVs already incorporate pollutant ingested doses from multiple routes of exposure of the wildlife

species to the pollutant. Therefore, a separate evaluation of chemical effects from multiple routes of exposure is not needed to assess criteria effects on aquatic-dependent wildlife. This assumes that dermal uptake of chemicals is insignificant for birds and mammals, as feathers, hair and skin prevent some chemicals from entering the body. The effects assessment for aquatic-dependent wildlife bases determinations only on direct effects to the wildlife species, and indirect effects on their prey.

The chronic wildlife threshold effects concentration is compared to the chronic water quality criterion and the following determinations are made:

If the Chronic Wildlife Threshold Effects Concentration > the Chronic Criterion Magnitude:

1. Consider adverse effects on fully aquatic prey of the listed wildlife species.

2. Criterion is determined to be NLAA if both direct effects on the listed wildlife species and indirect effects of the criterion via reduction of prey are determined as NLAA.

If the Chronic Wildlife Threshold Effects Concentration ≤ the Chronic Criterion Magnitude:

1. Criterion is determined to be LAA if adverse effects to the listed wildlife species would be likely as a result of chemical exposure via multiple exposure routes (i.e. dietary ingestion and water ingestion) and/or exposure at the criteria level is determined to adversely affect prey species of the listed wildlife species.

2. No further evaluation of aquatic-dependent wildlife is required.

5.4.4 Tier 3 methodologies – secondary baseline effects assessment

5.4.4.1 Tier 3 acute toxicity effects assessment methods

Tier 3 of the effects assessment evaluates the exposure duration component of a water quality criterion. The exposure duration component of an acute water quality criterion is a one-hour exposure to the criterion magnitude (concentration) component of a complete acute criterion definition.

The exposure duration component of both acute and chronic criteria is evaluated using time-to-event calculations. Acute toxicity data for fish are normally presented in the literature as a statistically reduced 96-hour LC_{50} (it is rare in an empirical toxicity test to observe exactly 50% mortality 96 hours after test initiation). Many empirical toxicity studies present effect concentrations at other exposure durations (e.g. 24-hour LC_{50}, 48-hour LC_{50}, etc.), or less commonly, present the raw toxicity data from which mortality at a given exposure concentration and duration can be calculated. Ideally, if an empirical study presents a 1-hour LC_{50} for a species, the 1-hour LC_{50} can be compared directly to the acute criterion magnitude for a 1-hour exposure duration.

Time to event toxicity data most commonly fits an exponential statistical distribution. If the 1-hour LC_{50} is not provided in a study with a listed species, but other time to event concentrations and exposure durations are presented, these data can be entered into a statistical program that fits data distributions, and the fitted distribution can be used to estimate the 1-hour LC_{50} for the listed species. Examples of this (Figure 33) are shown for two copper toxicity studies, one an acute toxicity study, and the second for a chronic copper toxicity study.

Similarly, if multiple time-to-event adverse effect concentrations are available in a study for a taxonomically closely related species to a listed species (genus or family level relationship), the study with the lowest of the effect concentrations can be used to calculate the time-to-event curve, then estimate the 1-hour LC_{50} for comparison to the criterion magnitude concentration at a 1-hour exposure duration. In this case, the taxonomically closely related species is used as a surrogate for the listed species without empirically derived 1-hour LC_{50} information.

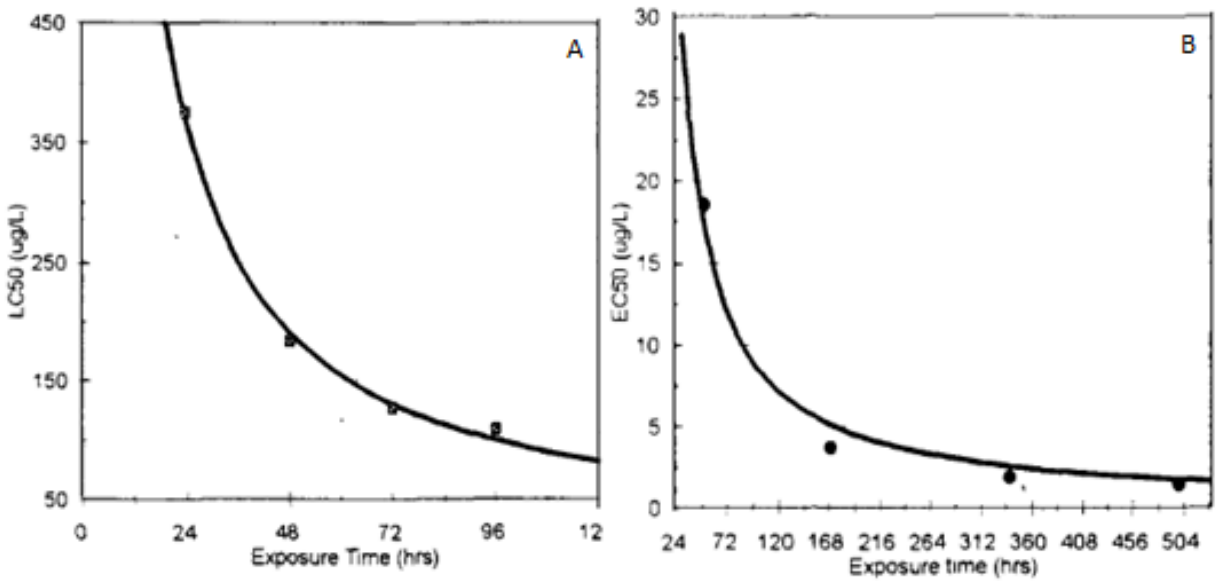


Figure 33: Example time-to-event curves. Curve A (left) represents acute copper toxicity (hardness = 249 mg/L) to a freshwater amphipod (Gammarus pulex) over various exposure durations, up to 96 hours(Stephenson, 1983). Curve B (right) represents chronic copper toxicity (hardness = 250 mg/L; test ID1) to Daphnia magna over various exposure durations, up to 21-days (Dave, 1984).

If empirical 1-hour LC_{50} data are not available for the listed species, but other exposure duration effect concentrations are available, use statistical curve fitting software to identify the best fitting curve to the empirical time-to-event data, then calculate the 1-hour acute LC_{50} for the toxicity data set.

5.4.4.1.1 Tier 3 acute toxicity effects determination

The acute 1-hour LC_{50} for each listed species is compared to the acute water quality criterion at a 1-hour exposure duration and the following determinations are made:

If the 1-hour acute LC_{50} > the Acute Criterion Magnitude for a 1-hour exposure duration,

1. Consideration of criterion frequency (average recurrence interval between exceedances of the magnitude component of the criteria) in Tier 4 is not required.

2. Consider indirect effects on adverse effect on prey of the listed species, and effects of multiple routes of chemical exposure to the listed species prior to making a final effects determination.

3. Criterion is determined to be NLAA if both the Tier 3 exposure duration effects assessment and Tier 2 indirect effects of the criterion are determined as NLAA.

If the 1-hour acute LC_{50} ≤ the Acute Criterion Magnitude for a 1-hour exposure duration,

1. Perform a Tier 4 assessment of criterion recurrence frequency for those listed species with acute effect threshold concentrations lower than criterion magnitude and 1-hour exposure duration.

2. Consider indirect effects on adverse effect on prey of the listed species, and effects of multiple routes of chemical exposure to the listed species prior to making a final effects determination.

If 1-hour acute LC_{50} data are unavailable for a chemical/listed species pair,

1. Consideration of criterion frequency (average recurrence interval between exceedances of the magnitude component of the criterion) in Tier 4 is not required.

1. Criterion is determined to be LAA for the listed species Note that the same acute criterion may be determined to be NLAA for some listed species, and LAA for other listed species, depending on the availability of the data needed to develop time-to-event relationships for a species.

If time-to-event data for a listed species, or close taxonomic surrogate (e.g. genus or family level surrogacy) suggest adverse effects do not occur from exposures to an acute criterion magnitude consistent with the corresponding acute criterion 1-hour exposure duration, then that species will not be expected to be affected by the pollutant at concentrations and exposure durations consistent with the criterion. In this case, an NLAA determination is made for the chemical/listed species pair, and the effects assessment does not need to proceed to Tier 4 of the effects assessment in this BE. If the 1-hour LC_{50} is lower than the criterion concentration, the chemical/listed species pair is forwarded to Tier 4 of the effects assessment for additional evaluation before a final NLAA or LAA determination is made.

5.4.4.2 Tier 3 chronic aquatic toxicity effects assessment methods

This portion of the Tier 3 effects assessment evaluates the exposure duration component of a chronic water quality criterion. The exposure duration component of a chronic water quality criterion is a 4-day exposure to the criterion magnitude (concentration) component of a complete chronic criterion definition. The desired length of a chronic toxicity test with fish is 30 days, although historically some chronic fish toxicity tests have extended past 30 days.

The exposure duration component of a chronic criterion is also evaluated using time-to-event methods. The approach is the same as that used to assess acute criteria exposure durations, with the exception that the goal is to determine chronic 4-day NOEC values for each chemical/listed species pair, not a 1-hour LC_{50} for each chemical/listed species pair.

Chronic toxicity data is increasingly statistically reduced and reported as EC_x data (effect concentration to x% of the test population) in the literature as a replacement for NOEC values. NOEC vary depending on the exposure concentration spacing in an individual toxicity test, making NOECs a more variable test endpoint than statistically reduced EC_x values.

The value of x to use in EC_x data has engendered a great deal of discussion in aquatic toxicology. The x values most commonly used in lieu of a chronic NOEC are EC_{20}, EC_{10} and EC_5 values. If EC_x are to co-exist with and eventually replace NOEC values, the question then becomes which EC_x is the best replacement for a NOEC. An EC_x needs to represent a low concentration that protects sensitive species, but not so

low that the EC_x is outside of the range of data in a study that permits reliable EC_x calculation (Beasley, Belanger, Brill, & Otter, 2015). EC_{20}s have been used as concentrations that cannot be shown to have adverse effects on field populations due to natural variation in the environment. EC_{20}s are also commonly used *in lieu* of a NOEC because many chronic toxicity test methods define control performance as acceptable if < 20% of test organisms experience an adverse effect. This implies that a 20% effect level in a chronic toxicity test is functionally equivalent to a no statistically significant adverse effect for the test, even though in some contexts a 20% effect could be considered biologically significant. Some consider no effect to be a statistical misnomer (Beasley et al., 2015).

In this biological evaluation EPA considers a chronic EC_{20} or lower as the functional equivalent of a chronic NOEC value, and use them interchangeably in the Tier 3 chronic effects assessment. The exact endpoints used depend on the endpoints available in the literature. Many empirical chronic toxicity studies present effect concentrations at multiple exposure durations, including calculated or statistically reduced EC_x or NOEC values. Occasionally raw toxicity data are available from which EC_xs or NOECs at a given exposure concentration and duration can be calculated. The most preferred data from an empirical study presents a 4-day chronic NOEC or EC_x for a listed species, which can be compared directly to the chronic criterion magnitude for a 4-day exposure duration.

Time to event chronic toxicity data most commonly fits an exponential statistical distribution. If the 4-day NOEC or EC_x is not provided in a study with a listed species, but other time to event concentrations and exposure durations are presented, these data can be entered into a statistical program that fits data distributions, and the fitted distribution can be used to estimate the 4-day NOEC or EC_x for the listed species.

Chronic NOEC or EC_x data for a taxonomically closely related surrogate species (genus or family level) to a listed species will also be used in this Tier 3 exposure duration effects assessment. The data from related species can be either directly measured and reported (preferred) or can be estimated by fitting a time-to-event curve for data for the surrogate species, then calculating the 4-day NOEC or EC_x from the statistically fitted data.

5.4.4.3 Tier 3 chronic wildlife toxicity effects assessment methods

The Tier 2 aquatic-dependent wildlife toxicity effects assessment made the assumptions that aquatic-dependent species obtained 100% of its prey from the action area, and reside in the action area for their entire lifetime. These exposure assumptions are not realistic for any of the aquatic-dependent species evaluated in this BE.

In Tier 3 of the effects assessment, EPA incorporates area use factors (AUF) and seasonal use factors (SUF) to calculate a more realistic wildlife threshold effects concentration of chemicals in water for aquatic-dependent wildlife than was done in Tier 2. Equation 5.25 shows the calculation used to estimate the wildlife threshold effects concentration for aquatic-dependent wildlife more realistically than was done in Tier 2 of the wildlife effects assessment.

Equation 5.25:

$$WTEC = \frac{TRV \times BW}{WIR + \sum_{i=1}^{4}\left(FIR_{TL_i} \times BAF_{TL_i}\right)} \times AUF \times SUF$$

where: AUF = area use factor, the proportion of the foraging range of the listed species comprised by the action area (dimensionless)

SUF = seasonal use factor, the proportion of a calendar year the listed species forages in the action area (dimensionless)

and all other terms of Equation 5.25 are identical to those of Equation 5.19.

5.4.4.4 Tier 3 chronic aquatic toxicity effects determination

The chronic 4-day NOEC (which can be an EC_x value where $x \leq 20$) for each listed species is compared to the chronic water quality criterion at a 4-day exposure duration and the following determinations are made:

If the chronic 4-day NOEC > the Chronic Criterion Magnitude for a 4-day exposure duration

1. Consideration of criterion frequency (average recurrence interval between exceedances of the magnitude component of the criteria) in Tier 4 is not required.

2. Consider adverse effect on prey of the listed species (i.e., prey availability), and effects of multiple routes of chemical exposure to the listed species prior to making a final effects determination.

3. Criterion is determined to be NLAA if both the Tier 3 exposure duration effects assessment and Tier 2 indirect effects of the criterion are determined as NLAA.

If the Chronic 4-day NOEC ≤ the Chronic Criterion Magnitude for a 4-day exposure duration

1. Perform a Tier 4 assessment of criterion recurrence frequency for those listed species with chronic NOEC concentrations lower than criterion magnitude and 4-day exposure duration.

2. Consider indirect effects on adverse effect on prey of the listed species, and effects of multiple routes of chemical exposure to the listed species prior to making a final effects determination.

If Chronic 4-day NOEC data are unavailable for a chemical/listed species pair,

1. Consideration of criterion frequency (average recurrence interval between exceedances of the magnitude component of the criterion) in Tier 4 is not required.

2. Criterion is determined to be LAA for the listed species.

3. Note that the same chronic criterion may be determined to be NLAA for some listed species, and LAA for other listed species, depending on the availability of the data needed to develop time-to-event relationships for a species.

5.4.4.5 Tier 3 chronic wildlife toxicity effects determination

If the Wildlife Threshold Effect Concentration > the Chronic Criterion Magnitude

1. No further quantitative direct effects assessment needs to be made on the aquatic-dependent species.

2. Consider indirect criterion adverse effects on prey of the listed species prior to making a final effects determination.

3. Criterion is determined as NLAA if both direct and indirect effects of the criterion are determined as NLAA.

If the Wildlife Threshold Effect Concentration ≤ the Chronic Criterion Magnitude

1. Criterion is determined as LAA for the listed aquatic-dependent wildlife species.

2. No further quantitative effects assessment will be made on the aquatic-dependent species.

5.4.5 Methodology for evaluating effects to prey of ESA listed species

EPA conducted an evaluation of the potential effects of exposure to chemicals at criteria levels to the prey base of listed aquatic and aquatic-dependent wildlife species; i.e., the indirect effects of the action. The methodology for this evaluation is described below.

The toxicity data were compiled, reviewed, sorted, and incorporated into the analysis as follows.

Tier 1 approach

- Toxicity data were acquired for each chemical in this order:
    - Data used in recent (~last five years) ESA consultations. These data were deemed acceptable by other EPA programs and so not reviewed for this BE.
    - If the above was unavailable, ECOTOX was queried and solely relied upon for this BE unless substantial newer data were available. “Substantial newer data” was defined as 2-3 relevant studies in the open literature, but not available in ECOTOX.
    - When substantial newer data were available, ECOTOX data were supplemented with published data found via a Web of Science literature search. Available literature was initially reviewed for relevance for use in this BE. Toxicity values were extracted from relevant papers.
- Chronic toxicity (NOEC) and acute toxicity values (LC_{Low} (LD50/2.27)) were sorted from low to high by each of six prey categories (fish, crustaceans, aquatic invertebrates, aquatic insects, molluscs, or amphibians).

- The study providing the acute or chronic prey category lowest toxicity value (PCLTV) was reviewed for reliability ("QA'd") according to procedures described in Appendix B [22,23].
- All available PCLTVs for each of up to four assessments (i.e., acute freshwater, chronic freshwater, acute marine, or chronic marine) from qualified studies were compared to the proposed criteria value.
- If all available PCLTVs were greater than the criteria, the criteria were assumed to be protective of prey species for listed species and thus NLAA a listed species. END OF TIER 1 and evaluation was complete. If, however, one or more PCLTVs were less than the criteria, EPA proceeded to a Tier 2 assessment as described below.

Tier 2 approach

- When one or more PCLTVs were less than the criteria, the criteria was deemed potentially unprotective and all available species mean acute or chronic values (SMAVs or SMCVs, respectively) were compared to the acute or chronic criteria, respectively.
    - In such a case, all studies providing SMAVs or SMCVs were QA'd as described in Appendix B (see the two footnotes in the Tier 1 section above).
- If <20% SMAVs or SMCVs were less than the criteria value, the proposed criteria was NLAA a listed species. If >20% SMAVs or SMCVs were less than the criteria value, the proposed criteria was LAA a listed species (USEPA, 1999c) (Suter II, Efroymson, Sample, & Jones, 2000).

The Tier 2 screening level indirect effects assessment of the prey of ESA listed species evaluates whether criteria concentrations elicit toxicity to a "meaningful portion" of the listed species diet (Suter II et al., 2000). All potential prey or surrogate prey species of a listed species (i.e. the community of prey species of an ESA listed species) were assessed. This section of the BE evaluates whether the proposed water quality criterion may adversely affect the assemblage of potential prey items through the "lens" of predator-prey relationships between ESA listed species and their prey. The ESA listed fish species, salmonids and rockfish, evaluated in this BE are secondary or tertiary consumers in aquatic and marine food webs, whose health may be adversely affected by reductions in the number of prey species available to them. This situation is most likely to occur when prey species are as a group more sensitive to a chemical than are the listed fish species, as is the case with many insecticides. An uncertainty in the methodology described here is the case when listed fish as a group are more sensitive to a chemical than are invertebrate prey species. In this situation, indirect effects on a listed species may be more impactful to a species than direct [toxic] effects of a chemical on the listed species.

[22] Only studies providing the PCLTV were QA'd because these studies formed the basis for the effects determination. Only studies providing the SMAVs or SMCVs < the criteria value were QA'd because these studies formed the basis for the effects determination.

[23] EPA did not fully evaluate (QA) studies that provided an indication that a species sensitivity to a chemical was low (i.e., toxicity value > criteria value) because the results of the QA process would not change the effects determination, as the determination is based on lowest toxicity values. That is, there is no enhanced risk to a species if a toxicity value indicating an insensitive species (i.e., toxicity value < criteria value) from a lower quality study is maintained in the dataset.

Effects to a "meaningful portion" of prey has not been previously defined for the purposes of biological evaluations by EPA (USEPA, 2006b, 2017b). Therefore, a definition of "meaningful portion" is described herein. Informed by recent biodiversity research on how changes in species richness may affect ecological communities, it is apparent that a decline of more than 20% in species richness has detrimental effects to communities (Hooper et al., 2012; Vaughn, 2010). Therefore, a "meaningful portion" for purposes of the analysis in this BE is defined as an adverse effect to ≥20% of prey species potentially consumed by a listed species. A 20% change in species richness is consistent with other lines of evidence in water quality criteria derivation and ecological risk assessment, where a 20% change in a parameter is used as a threshold for adverse effects (Suter II et al., 2000).

Reduction in the availability of prey for fish may result in reduced fish growth, fitness and density (number of individuals in a population per unit of area, such as within the action area) when and where fish are food-limited (Grunblatt, Meyer, & Wipfli, 2019). Although other community components can also indirectly affect an ESA listed species, species richness effects on communities is by far the most studied community structure metric used to evaluate biodiversity effects on ecosystem structure and function (Daam, Teixeira, Lillebø, & Nogueira, 2019; van der Plas, 2019). Reduction in prey species richness have been directly linked to changes in fish biomass, production and yield(Brooks, Holland, Darwall, Eigenbrod, & Tittensor, 2016; Smokorowski & Kelso, 2002), allowing prey species richness to serve as a surrogate measure for predator species abundance.

The conclusions presented in section 5.8 based on a tier 1 or tier 2 analysis are used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

5.5 Freshwater Criteria Effects Assessments

5.5.1 Toxic substances

Each chemical section below, for the assessment category 4 criteria, contains the acute and/or chronic effects assessments, depending on whether each chemical has an acute criterion, a chronic criterion, or both (section 2.4.1, Table 6). Table 2 and Table 3 summarize the results of all assessments.

5.5.1.1 Acrolein – Freshwater

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = 3 µg/L

Chronic (CCC) criterion = 3 µg/L

CAS ID 107-02-8

Acrolein (2-propenal) has one available empirical acute toxicity study on action area listed species that was used to derive the EPA (2009a) freshwater acrolein water quality criteria. Holcombe et al. (G. W. Holcombe, G. L. Phipps, A. H. Sulaiman, & A. D. Hoffman, 1987) measured a 4-day LC_{50} for rainbow trout (*Oncorhynchus mykiss*, mean weight 2.5 g, age at test initiation not given) of 16 µg/L (95% confidence limits of 14 – 19 µg/L). A second available salmonid study is that of Lorz et al. (Lorz et al., 1979), who

determined a 4-day LC_{50} of 68 µg/L for yearling coho salmon (*Oncorhynchus kisutc*h). Although useful as a second study on the sensitivity of salmonids to acrolein, coho salmon are not an ESA listed species within the action area.

Holcombe et al. (1987) also determined a 4-day LC_{50} for African clawed frog tadpoles (*Xenopus laevis*) of 7 µg/L (95% confidence interval of 6 – 8 µg/L). The *Xenopus* LC_{50} is the lowest SMAV of all species used in the derivation of the national water quality criterion for acrolein. The *Xenopus* LC_{50} is also the only amphibian species used in the development of the acrolein criteria.

5.5.1.1.1 Additional acute toxicity data for acrolein published since 2009

A search of the ECOTOX database found no acute toxicity studies of acrolein on any fish or amphibian species published between 2008 and present day. The evaluation of acrolein effects on acute toxicity to listed species is therefore based on the toxicity information available during the derivation of the EPA (2009a) acrolein criteria.

5.5.1.1.2 Effects assessment of acute acrolein criterion on listed species

The lowest known salmonid 4-day LC_{50} to acrolein is the 16 µg/L effect concentration to rainbow trout observed by Holcombe et al. (1987). Division of the lower 95% confidence level of 14 µg/L around this LC_{50} by 2.27 yielded a lowest LC_{LOW} or minimum acute effect concentration of 6.2 µg/L to two significant figures. This value is higher than the acrolein acute criterion of 3 µg/L. Assuming that the minimum acute effect concentration for the remaining listed salmonid species in the freshwater portions of the action area is greater than or equal to 6.2 µg/L, exposure at the level of the acrolein acute criterion of 3 µg/L is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, and bull trout.

5.5.1.1.3 Water-column exposure toxicity data - chronic criterion for acrolein in freshwater

None of the action area listed species in freshwater had empirical acrolein chronic toxicity data available at the time of publication of the EPA acrolein criteria.

5.5.1.1.4 Additional chronic toxicity data for acrolein published since 2009

A search of the ECOTOX database found no chronic toxicity studies of acrolein on population, growth, reproductive or behavioral endpoints for any fish or amphibian species published between 2008 and present day. The evaluation of acrolein effects on its chronic toxicity to listed species is therefore based on the toxicity information available during the derivation of the EPA acrolein criteria.

5.5.1.1.5 Acute-chronic ratio for acrolein

Because no empirical chronic toxicity data for any listed species exists, the chronic minimum effect threshold concentration for listed species is calculated by dividing the acute minimum effect threshold concentration by the ACR for acrolein. Four measured ACR values are reported in the EPA (2009a) acrolein criteria document, ranging between and having a geometric mean of 1.906. But according to the Guidelines, if the geometric mean of all measured ACRs is less than two, the value of the ACR used to estimate chronic no effect concentrations from acute toxicity data is set to 2.0. Thus, the acrolein ACR used in this BE is 2.0.

5.5.1.1.6 Effects assessment of chronic acrolein criterion on listed species

The lowest acute minimum effect threshold concentration for any salmonid species is 6.2 µg/L, derived from a rainbow trout 4-day LC_{50} of 16 µg/L. Dividing the acute minimum effect threshold for salmonids

of 6.2 µg/L by the ACR of 2.0 yields a chronic minimum effect threshold concentration for acrolein of 3.1 µg/L. This value is higher than the acrolein chronic criterion of 3 µg/L. Assuming the minimum chronic effect concentration for the remaining listed salmonid species in the freshwater portions of the action area is greater than or equal to 3.1 µg/L, exposure at the level of the acrolein chronic criterion of 3 µg/L is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon and bull trout.

5.5.1.1.7 Summary of freshwater effect determinations for acrolein

EPA has made the following determinations on the acute and chronic acrolein criteria for the listed species in the freshwater portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.5.1.2 Arsenic – Freshwater

The Tribe submitted the following criteria to EPA for CWA approval (the chronic criterion for arsenic is not addressed in this BE):

Acute (CMC) criterion = 340 µg/L

CAS ID 7440-38-2

5.5.1.2.1 Introduction to Arsenic

Arsenic in freshwater is primarily found in one of four chemical forms. Two forms are inorganic, trivalent arsenic (As^{+3}, arsenite) and pentavalent arsenic (As^{+5}, arsenate). The other two forms are organic, methylarsonic acid (MAA) and dimethylarsinic acid (DMAA). The two inorganic forms of arsenic are more toxic to aquatic life than are the two organic forms of arsenic. Arsenite is more toxic to freshwater

aquatic life than is arsenate. In aerobic waters at circumneutral pH, As^{+3} is slowly oxidized to the less toxic As^{+5}: this reaction can be reversed in anaerobic waters.

According to EPA (2019) and the Swinomish Tribe's WQS, the arsenic freshwater quality criteria were derived from toxicity data for the more toxic trivalent arsenic but are applied to the total arsenic concentration in freshwaters. Both the acute and chronic freshwater arsenic criteria are expressed in terms of the dissolved metal in the water column.

Although known as a poison for over 2000 years, arsenic has recently been determined to be a nutritionally essential element for some mammals and birds (Uthus, 1992) at low dietary levels near 25 ng/g. Arsenic has also been shown to be an essential element for some aquatic bacteria (Wolfe-Simon et al., 2011). It is unknown at this time whether arsenic is an essential element for fish species.

5.5.1.2.2 Water-column exposure toxicity data - acute criterion for arsenic in freshwater

The numeric values for both the acute and chronic freshwater arsenic criteria have not been changed since the publication of EPA (USEPA, 1985a). The EPA (1996a) update of a number of criteria, including freshwater arsenic did not result in a change to the freshwater arsenic criteria concentrations, although it did identify and incorporate several arsenic toxicity studies of acceptable data quality published subsequent to the publication of the EPA (1985a) arsenic criteria into the criteria development database.

The combined toxicity data presented in the EPA (1985a) and EPA (1996a) arsenic criteria documents allows for the calculation of 14 genus mean acute values (GMAVs). Seven of the GMAVs are for fish, the remaining seven GMAVs are for invertebrates. The four most sensitive GMAVs were all for invertebrate species, ranging between 874 – 2444 µg/L. The most sensitive GMAV for a fish species was 13,340 µg/L for rainbow trout (W. W. Johnson & Finley, 1980), one of the ESA listed species within the action area. There is also a GMAV of 14,960 µg/L for brook trout (Cardwell, Foreman, Payne, & Wilbur, 1976) , the same genus as the ESA listed bull trout within the action area.

5.5.1.2.3 Additional freshwater acute toxicity data for arsenic published since 1985

Kumari et al. (2017) published a literature review of freshwater toxicity studies with arsenic containing 25 96 hour LC_{50} studies published since the publication of the USEPA (1985a) update of its freshwater arsenic criteria. Several of the 96 hour LC_{50}s in Kumari et al. (2017) were published between 1985 and 1996 but are not discussed in the EPA (1996a) update of the arsenic criteria.

Between the studies identified in Kumari et al. (2017) and other studies identified through review of literature sources such as ECOTOX, Google Scholar and Web of Science, we have identified the seven additional arsenite 96 hour LC_{50} values published since 1985 and listed in Table 84 as being of acceptable data quality for use in deriving EPA aquatic life criteria. The studies listed in Table 84 are used in conjunction with the GMAVs listed in USEPA (1985a) to perform the toxicity assessment of the freshwater arsenic criterion. Although several studies of arsenate toxicity published since 1985 were also identified, they are not included in Table 84, because arsenate is of lower toxicity than arsenite.

Table 84. 96 hour LC50 studies with arsenite (As+3) of acceptable data quality published since 1985

Species	**96 hour LC_{50}, µg/L**	**Literature citation**
Zebrafish (*Danio rerio*)	149,800	(Li et al., 2009)
Mozambique tilapia (*Oreochromis mossambicus*)	28,220	(Ahmed, Habibullah-Al-Mamun, Parvin, Akter, & Khan, 2013)
Coho salmon (*Oncorhynchus kisutch*)	18,500	(Buhl & Hamilton, 1991)
Bluegill (*Lepomis macrochirus*)	17,300	(F. L. Mayer & M. R. Ellersieck, 1986b)
Rainbow trout (*Oncorhynchus mykiss*)	16,000	(Buhl and Hamilton, 1991)
Arctic grayling (*Thymallus arcticus*)	13,700	(Buhl and Hamilton, 1991)
Golden shiner (*Notemigonus crysoleucas*)	12,500	(Hartwell, Jin, Cherry, & Cairns Jr, 1989)

5.5.1.2.4 Effects assessment of acute freshwater arsenic criterion on listed species

The only ESA listed species within the action area with empirical acute toxicity data for arsenic is rainbow trout (steelhead). The lowest empirical 96 hour LC_{50} for rainbow trout is 13,360 µg/L from Johnson and Finley (1980). Dividing this 96 hour LC_{50} by 2.27 to estimate an LC_{low} results in an acute effects assessment concentration of 5885 µg/L. The rainbow trout acute effects concentration is 16~~x~~ times higher than the acute arsenic criterion of 360 µg/L. This indicates that exposure at the level of the freshwater acute arsenic criterion is not likely to adversely affect steelhead.

Without empirical toxicity data for the remaining ESA listed species, the acute effects assessment concentrations for Chinook salmon, chum salmon and bull trout will be calculated from Interspecies Correlation Estimation (ICE) models. The rainbow trout 96 hour LC_{50} of 13,360 µg/L from Johnson and Finley (1980) will be used to calculate effects assessment concentrations for Chinook and chum salmon, while the brook trout 96 hour LC_{50} of 14,960 µg/L from Cardwell et al. (1976) will be used to calculate the effects assessment concentration for bull trout.

The ICE estimated 96 hour LC_{50}s for Chinook and chum salmon and bull trout are shown in Table 85. Dividing the estimated 96 hour LC_{50}s by 2.27 yields the acute effects assessment concentrations for the ESA listed species without empirical acute arsenic toxicity data as follows:

Chinook salmon – 6432 µg/L

Chum salmon – 3267 µg/L

Bull trout – 1596 µg/L

All of the calculated acute effects assessment concentrations for the ESA listed fish species without empirical toxicity data are well in excess of the freshwater acute arsenic criterion of 360 µg/L. This indicates that exposure at the level of the freshwater acute arsenic criterion is not likely to adversely affect Chinook salmon, chum salmon or bull trout.

Table 85. Output from Interspecies Correlation Estimation models to estimate acute arsenic toxicity to Chinook salmon, chum salmon and bull trout.

Predicted Taxa	Chinook salmon (*Oncorhynchus tshawytscha*)	Oncorhynchus (used for chum salmon)	Salvelinus (used for bull trout)
Model Level	species	genus	Genus
Surrogate	Rainbow trout (*Oncorhynchus mykiss*)	Rainbow trout (*Oncorhynchus mykiss*)	Brook trout (*Salvelinus fontinalis*)
Estimated Toxicity (µg/L)	14600.18	7415.35	3624.43
95% Confidence Intervals	9780.27 - 21795.43	5259.12 – 10455.62	1383.55 – 9494.75
Df (N-2)	16	60	7
R^2	0.96	0.92	0.88
p-value	0	0	0.0001
Mean Square Error (MSE)	0.07	0.15	0.1
Cross-validation success (%)	94	89	89
Tax. Dist.	1	1	1
Slope	0.94	0.93	0.8
Intercept	0.27	0	0.21

5.5.1.2.5 Summary of freshwater effect determinations for arsenic

EPA has made the following determinations on the acute arsenic criterion for the listed species in the freshwater portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above, and the indirect effects analysis for prey availability provided in section 5.8 (the multiple routes of exposure analysis was not relevant for a CMC as indicated in section 5.7). The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.5.1.3 Carbaryl – Freshwater

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = 2.1 µg/L

Chronic (CCC) criterion = 2.1 µg/L

CAS ID 63-25-2

5.5.1.3.1 Water-column acute exposure toxicity data

EPA (2012) listed 47 genera of freshwater species with acute toxicity data meeting all EPA quality assurance requirements for use in deriving water quality criteria. In general, invertebrate species are more sensitive to carbaryl than are fish. Of the 47 genera with data of acceptable quality for criteria derivation, the 15 most sensitive genera are all invertebrates. The toxicity of carbaryl is consistent with its registered use in the United States for controlling more than 160 insect species on over 115 agricultural and non-crop use applications, including home and garden uses.

Nine species of salmonid freshwater acute toxicity tests with carbaryl were used in the derivation of the EPA (USEPA, 2012) acute carbaryl criterion. These nine species belong to one of three genera of salmonids, described below along with the relative sensitivity of the three salmonid genera within the 47 genera used to derive the freshwater acute carbaryl criterion. Many of the individual toxicity tests with salmonid species used to derive the carbaryl acute freshwater criterion were performed at the Columbia (Missouri) National Fisheries Research Laboratory of the U.S. Fish and Wildlife Service, currently the Columbia Environmental Research Center of the U.S. Geological Survey (USGS)(W. W. Johnson & Finley, 1980; F. L. Mayer & M. R. Ellersieck, 1986b; Sanders, Finley, & Hunn, 1983).

Salmo (16th most sensitive genus, most sensitive fish genus, genus mean acute value = 889 µg/L)

- Brown trout (*Salmo trutta*) – SMAV = 700 µg/L
- Atlantic salmon (*Salmo salar*) – SMAV = 1129 µg/L

Salvelinus (18th most sensitive genus, GMAV = 1269 µg/L)

- Lake trout (*Salvelinus namaycush*) – SMAV = 988 µg/L
- Brook trout (*Salvelinus fontinalis*) – SMAV = 1629 µg/L

Oncorhynchus (22nd most sensitive genus, GMAV = 1810 µg/L)

- Rainbow trout (*Oncorhynchus mykiss*) – SMAV = 860 µg/L)
- Apache trout (*Oncorhynchus apache*) – SMAV = 1540 µg/L)
- Coho salmon (*Oncorhynchus kisutch*) – SMAV = 1654 µg/L)
- Chinook salmon (*Oncorhynchus tshawytscha*) – SMAV = 2690 µg/L)
- Cutthroat trout (*Oncorhynchus clarkii*) – SMAV = 3300 µg/L)

By comparison, the four most sensitive genera to carbaryl were all stoneflies, with the most sensitive stonefly species (*Isogenus* sp.) having a species and GMAV of 3.175 µg/L.

5.5.1.3.2 Additional freshwater acute toxicity data for carbaryl published since 2012

EPA has identified one additional freshwater acute toxicity test result with a salmonid that met data quality requirements for use in this BE. The study by Boran et al. (Boran, Altinok, & Capkin, 2010) was not identified in the carbaryl aquatic life criteria (USEPA, 2012). Boran et al. (2010) determined a 96 hour LC_{50} of 1390 µg/L for rainbow trout. This is well within the rainbow trout 96 hour LC_{50} range of <320 to 5400 µg/L of the 37 rainbow trout 96 hour LC_{50} values identified as being of acceptable data quality for use in the derivation of the freshwater acute carbaryl criterion (USEPA, 2012). As such, the results from Boran et al. (2010) are unlikely to appreciably change the SMAV of 860 µg/L for rainbow trout used in EPA (2012).

5.5.1.3.3 Effects assessment of acute freshwater carbaryl criterion on listed species

Two of the ESA listed salmonid species within the freshwater portions of the action area, rainbow trout (steelhead) and Chinook salmon have empirical toxicity data, allowing a direct assessment of carbaryl acute toxicity to these two species. The rainbow trout SMAV of 860 µg/L is the lowest SMAV of the five *Oncorhynchus* spp. with SMAVs in the carbaryl criteria document (USEPA, 2012). It is the geometric mean of 37 96 hour LC_{50} values from 11 different literature citations. The lowest individual 96 hour LC_{50} for rainbow trout in the carbaryl criteria document (USEPA, 2012) is a <320 µg/L value from Mayer and Ellersieck (F. L. Mayer & M. R. Ellersieck, 1986b). Dividing the rainbow trout SMAV and the lowest individual study acute value by 2.27 results in minimum acute toxicity threshold of 379 µg/L and <141 µg/L, respectively. Both of these acute toxicity threshold NOECs are higher than the freshwater acute carbaryl criterion of 2.1 µg/L. Because carbaryl is not expected to elicit any short term acutely toxic effects on rainbow trout (steelhead) at a carbaryl concentration of 141 µg/L or lower, exposure at the level of the acute carbaryl criterion of 2.1 µg/L is not likely to adversely affect rainbow trout.

The Chinook salmon 96 hour LC_{50} of 2690 µg/L used in the carbaryl freshwater acute criterion derivation is from Phipps and Holcombe (Gary L Phipps & Gary W Holcombe, 1985). Dividing this 96 hour LC_{50} by 2.27 to convert it to an LC_{low} yields a minimum acute toxicity threshold of 1185 µg/L. Because carbaryl is not expected to elicit any short term acutely toxic effects on Chinook salmon at a carbaryl concentration of 1185 µg/L or lower, exposure at the level of the acute carbaryl criterion of 2.1 µg/L is not likely to adversely affect Chinook salmon.

In the hierarchy of biological evaluation methods for assessment of acute water quality data (Table 81), the first and preferred tier of the methodology is always to compare empirical acute LC_{50} (i.e., the $LC_{50}/2.27 = LC_{Low}$) data for the listed species to the acute criterion itself. If empirical data for the listed species is not available, which is the case for chum salmon and bull trout in this carbaryl evaluation, the second tier of the methodology is to use empirical acute LC_{50} data from a genus-level surrogate species for the listed species as the LC_{50} for the listed species itself. The surrogate species LC_{50} is compared to the acute criterion, assuming the known surrogate species LC_{50} is equal to the unknown LC_{50} of the listed species.

Both rainbow trout and Chinook salmon are members of the genus *Oncorhynchus*, as is chum salmon. This taxonomic similarity permits the lowest empirical 96 hour LC_{50} from either rainbow trout or Chinook salmon to be used to estimate carbaryl acute toxicity to chum salmon). In this case we assume that the lowest rainbow trout 96 hour LC_{50} of <320 µg/L is equal to the 96 hour LC_{50} for chum salmon. Dividing this rainbow trout 96 hour LC_{50} by 2.27 results in minimum acute toxicity threshold of <141 µg/L for chum salmon. Because carbaryl is not expected to elicit any short term acutely toxic effects on chum salmon at a carbaryl concentration of 141 µg/L or lower, exposure at the level of the acute carbaryl criterion of 2.1 µg/L is not likely to adversely affect chum salmon.

The toxicity of carbaryl to the fifth ESA listed salmonid within the action area (bull trout) can be estimated from the empirical toxicity of carbaryl to either brook trout or lake trout toxicity data, as both of these species are in the same genus (*Salvelinus*) as the bull trout. Thus, brook trout and lake trout empirical toxicity data meet the genus-level surrogate species relationship described in Table 81, allowing EPA to assume that the LC_{50} for brook trout or lake trout is equivalent to the bull trout LC_{50} for carbaryl.

The lowest individual carbaryl 96 hour LC_{50}s for brook trout and lake trout are both from Mayer and Ellersieck (F. L. Mayer & M. R. Ellersieck, 1986b), and are nearly identical at 680 µg/L and 690 µg/L, respectively. The lower of the two SMAVs is that of lake trout (988 µg/L), derived from results of five individual toxicity studies compared to the 1629 µg/L SMAV for brook trout, derived from nine individual toxicity tests. Using lake trout as the surrogate species for the listed bull trout, dividing the lake trout SMAV and the lowest individual study lake trout acute value by 2.27 results in minimum acute toxicity thresholds of 435 µg/L and 304 µg/L, respectively. These acute toxicity thresholds are assumed to represent carbaryl concentrations that will not elicit toxic effects on bull trout during short-term exposures to carbaryl. Both of these acute toxicity thresholds are higher than the freshwater acute carbaryl criterion of 2.1 µg/L. Because carbaryl is not expected to elicit any short term acutely toxic effects on bull trout at a carbaryl concentration of 304 µg/L or lower, exposure at the level of the acute carbaryl criterion of 2.1 µg/L is not likely to adversely affect bull trout.

5.5.1.3.4 Acute-chronic ratio for carbaryl

Four valid ACRs are available for carbaryl using the fifth (*Ceriodaphnia dubia*), eighth (*Daphnia magna*) and thirty-sixth (fathead minnow *Pimephales promelas*) most sensitive genera of the 47 freshwater genera used to derive the acute carbaryl criterion. Two fathead minnow ACRs are available, only one of which was used in the carbaryl criteria document (USEPA, 2012). The difference between the lowest ACR (1.094 for *Ceriodaphnia dubia*) and the highest ACR (23.82 for fathead minnow) is a factor of 22, greater than the factor of 10 difference in ACRs the Guidelines identify as an area where the data should be checked before a final ACR is calculated. For carbaryl, it is unknown why the ACRs derived from fish studies are higher than the ACRs derived from zooplankton species. The Guidelines also stipulate that if the species mean ACRs seem to increase as the SMAV increases, the FACR (final ACR) should be calculated as the geometric mean of the ACRs for species whose SMAVs are close to the FAV. Using this recommendation, the FACR in EPA (USEPA, 2012) was the geometric mean of 1.094 (*C. dubia*) and 1.581 (*D. magna*), or 1.315. However, the Guidelines also stipulate that if the most appropriate species mean ACRs are less than 2.0, the FACR should be assumed to be 2.0. Low ACRs may reflect acclimation to the toxicant during the chronic test, or test organisms being fed vs. unfed which may affect bioavailability and susceptibility. Thus, the FACR for carbaryl is assumed to be 2.0 for freshwater organisms which is the ACR used in this BE.

5.5.1.3.5 Water-column exposure toxicity data - chronic criterion for carbaryl in freshwater

The carbaryl criteria document (USEPA, 2012) lists only three fish species with empirical chronic toxicity data meeting data quality requirements for use in deriving EPA aquatic life criteria: bonytail chub (*Gila elegans*), fathead minnow (*Pimephales promelas*) and Colorado pikeminnow (*Ptychochelius lucius*). All three of these species are members of the family *Cyprinidae*, the fish family with the largest number of individual fish species. EPA (2012) only identified two other species with chronic toxicity data meeting data quality requirements for use in deriving EPA aquatic life criteria: the zooplankton cladocerans *Daphnia magna* and *Ceriodaphnia dubia*. As was the case for acute toxicity data, the two invertebrates have substantially lower SMCVs, which are designed as chronic no effect concentrations than do the fish species. Whereas the lowest of the three fish chronic SMCVs is 620.8 µg/L for Colorado pikeminnow (Beyers, Carlson, & Keefe, 1994), the two invertebrate chronic SMCVs were 3.77 µg/L for *Daphnia magna* (D. Surprenant, 1985) and 10.6 µg/L for *Ceriodaphnia dubia* (Oris, Winner, & Moore, 1991).

Appendix H (Other Data on Effects of Carbaryl to Freshwater Aquatic Organisms) of USEPA (2012) contained no salmonid toxicity data that could be used to either directly evaluate chronic toxicity, or

data from possible surrogate fish species that could be used to estimate carbaryl concentrations that elicit chronic toxicity to the ESA listed salmonids within the freshwater portions of the action area. This lack of empirical chronic toxicity data on carbaryl toxicity to salmonids or other surrogate species means that unless additional suitable quality data for chronic toxicity can be identified from studies published since the carbaryl criteria document was issued in 2012, acute toxicity LC_{50} data for the listed and surrogate species used in the effects assessment portion for the freshwater acute carbaryl criterion will need to be divided by the carbaryl ACR of 2.0 to estimate chronic NOEC concentrations for listed salmonids. This process is summarized in Table 82.

5.5.1.3.6 Additional freshwater chronic toxicity data for carbaryl published since 2012

One chronic toxicity study with carbaryl on an ESA listed salmonid within the action area was identified that was published either concurrently with or shortly after the carbaryl criteria document (USEPA, 2012) was issued. Kennedy and Ross (2012) exposed juvenile sockeye salmon to carbaryl under flow-through conditions for up to 84 days while monitoring the growth of the fish. Effects were quantified every 14 days of exposure, including the quantification of effects on growth after the 28 day exposure to carbaryl. Consistent with the Guidelines, the 28 day chronic NOEC from Kennedy and Ross (2012) was used as the measure of chronic effect on sockeye salmon in this BE. Kennedy and Ross (2012) also evaluated carbaryl effects on growth of northern pikeminnow (*Ptychocheilus oregonensis*) for up to 84 days. Northern pikeminnow is a close taxonomic relative of the Colorado pikeminnow (*P. lucius*) whose chronic toxicity NOEC of 620.8 µg/L was identified in USEPA (2012) as of suitable data quality, as well as the lowest fish chronic NOEC for use in developing chronic carbaryl aquatic life criteria.

The experimental design of Kennedy and Ross (2012) for sockeye salmon concurrently tested carbaryl effects on fish growth at two water temperatures (5°C and 15°C), and with fed and unfed fish at each of the two water temperatures. Northern pikeminnow were tested only at 15°C with fed and unfed fish. Carbaryl exposure concentrations were 10 µg/L and 1000 µg/L plus a control.

Kennedy and Ross (2012), strictly speaking does not meet the EPA data quality requirements for a chronic toxicity study suitable for water quality criteria derivation. This is because only two exposure concentrations (plus the control) were tested, not the required minimum of three exposure concentrations (plus the control) called for in the Guidelines. But since Kennedy and Ross (2012) is the only known chronic toxicity test of the required duration for any salmonid species, let alone an ESA listed salmonid present within the action area, we evaluate its results in this section.

Carbaryl had no effect on the growth of either sockeye salmon at 5°C or 15°C, or on northern pikeminnow growth at 15°C at the end of a 28 day exposure to 10 µg/L, relative to growth in controls. These 28-day no effect findings were observed in both the fed and unfed exposures. No effects on growth of either species were measured at the end of the 84 day carbaryl exposure in the fed fish exposures at either 5°C or 15°C.

The only statistically significant adverse growth effects found by Kennedy and Ross (2012) were in the 1000 µg/L carbaryl exposures for both species. For sockeye salmon, the only statistically significant growth reductions relative to controls occurred after 84 days exposure to 1000 µg/L at 5°C, and after 70 and 84 days exposure to 1000 µg/L carbaryl at 15°C. For northern pikeminnow, statistically significant growth reductions occurred after 56, 70 and 84 day exposures to 1000 µg/L carbaryl at 15°C.

Kennedy and Ross (2012) is the only carbaryl chronic toxicity study identified for any fish species published since the carbaryl criteria document (USEPA, 2012) was published. Given the limited amount of chronic toxicity data for any species exposed to carbaryl, we will use the measured NOEC of 10 µg/L carbaryl in the effects assessment of the chronic carbaryl criterion on listed species.

5.5.1.3.7 Effects assessment of freshwater chronic carbaryl criterion on listed species

In the hierarchy of biological evaluation methods for assessment of chronic water quality data (Table 82), the third tier of the methodology is to take the empirical acute LC_{50} data for the listed species and divide it by the ACR for the chemical. Both rainbow trout (steelhead) and Chinook salmon have empirically measured 96 hour LC_{50} data that was used to directly evaluate the acute carbaryl criterion effects on these two species earlier in this section. The rainbow trout 96 hour LC_{50} SMAV of 860 µg/L and the Chinook salmon 96 hour LC_{50} SMAV of 2690 µg/L divided by the ACR of 2.0 result in estimated chronic NOECs of 430 µg/L and 1345 µg/L, respectively. Both of these chronic NOECs are higher than the chronic carbaryl criterion of 2.1 µg/L. Even using the highest available carbaryl ACR from any species, the 23.82 ACR value from the fathead minnow results in estimated chronic NOECs for rainbow trout and Chinook salmon of 36.1 µg/L and 112.9 µg/L, respectively. Chronic NOEC values calculated from use of the largest known value of a carbaryl ACR are also higher than the chronic carbaryl criterion. These analyses all indicate that exposure at the level of the freshwater chronic carbaryl criterion of 2.1 µg/L is not likely to adversely affect rainbow trout (steelhead) and Chinook salmon.

The fourth tier of the BE methodology for assessing chronic water quality criteria is using empirical chronic NOEC data from a genus-level surrogate species for the listed species. Using either the lowest (10 µg/L) or highest (1000 µg/L) empirical chronic NOEC for sockeye salmon (*Oncorhynchus nerka*) from Kennedy and Ross (2012) we assume the chronic NOEC for the chum salmon (*Oncorhynchus keta*) is equal to that of the sockeye salmon. Assuming the chum salmon is no more sensitive to carbaryl than is sockeye salmon, exposure at the level of the freshwater chronic carbaryl criterion of 2.1 µg/L is not likely to adversely affect chum salmon.

The last ESA listed salmonid to evaluate in this BE is the bull trout (*Salvelinus confluentus*). Bull trout are in a different genus than are the other four salmonids evaluated here. We are unaware of any empirical toxicity data for bull trout exposed to carbaryl. Bull trout and sockeye salmon are in the same fish family (*Salmonidae*). The eighth tier of the BE methodology for assessing chronic water quality criteria is using empirical chronic NOEC data from a family-level surrogate species for the listed species. We therefore use the empirical carbaryl chronic NOEC data from sockeye salmon as a surrogate to estimate carbaryl chronic toxicity to bull trout. If bull trout are no more sensitive to carbaryl than are sockeye salmon, exposure at the level of the freshwater chronic carbaryl criterion of 2.1 µg/L is not likely to adversely affect bull trout.

5.5.1.3.8 Summary of freshwater effect determinations for carbaryl

EPA has made the following determinations on the acute and chronic carbaryl criteria for the listed species in the freshwater portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.5.1.3.9 Why are the freshwater acute and chronic carbaryl criteria identical?

The freshwater acute and chronic carbaryl criteria are both 2.1 µg/L. This is because the available toxicity data for carbaryl, when entered into the calculation methodologies from the Guidelines for the two criteria, by coincidence result in the same criteria value when rounded to two significant figures.

There were 47 GMAVs used to derive the acute carbaryl criterion in EPA (2012). The FAV (the 5th percentile of the SSD for carbaryl) was 4.219 µg/L. The calculated FAV is then divided by two to account for the fact that acute toxicity tests are designed to produce LC_{50} values, a concentration resulting in the death of 50% of test organisms under the exposure conditions of the individual toxicity tests. The goal of an acute criterion is to calculate a concentration needed to protect aquatic species at levels below which effects on test organisms are indistinguishable from control organisms not exposed to the test chemical. According to the Guidelines, this is accomplished by using the FAV/2 as the CMC. The acute criterion calculated in this manner should be protective for all freshwater organisms potentially exposed to carbaryl under short-term exposure conditions, defined in the criteria as a 1-hour exposure once every 3-years on average.

Chronic criteria can be developed using the same computational procedures used to derive acute criteria if a sufficient number of acceptable chronic toxicity studies are available. EPA (2012) identified only five freshwater species with chronic toxicity data meeting the data quality requirements in the Guidelines, fewer than the minimum eight species worth of data needed to directly calculate a chronic criterion. In this case, the Guidelines call for dividing the FAV by the ACR to calculate the CCC. As noted above the carbaryl FAV is 4.219 µg/L. The carbaryl ACR was calculated in EPA (2012) as 1.315, the geometric mean of ACR values for *Ceriodaphnia dubia* (1.094) and *Daphnia magna* (1.581), the species with SMAVs closest to the FAV as per the Guidelines. However, the Guidelines also stipulate that if the most appropriate species specific ACRs are lower than 2.0, the final ACR should be assumed to be equal to 2.0. Dividing the freshwater FAV of 4.219 µg/L by the assumed ACR of 2.0 yields a freshwater final

chronic value (FCV), defined as a chronic NOEC concentration in the Guidelines, of 2.1 µg/L. This is why both the acute and chronic freshwater carbaryl criteria are identical concentrations.

5.5.1.4 Chlorine – Freshwater

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = 19 µg/L

Chronic (CCC) criterion = 11 µg/L

5.5.1.4.1 Introduction

Elemental chlorine is a gas under all conditions normally found in the environment except for extreme cold temperatures (boiling point = -34°C or -29°F). Except for minute amounts released to the atmosphere from volcanic eruptions, elemental chlorine is not found in a free state in nature due to its reactive nature.

A common source of chlorine to surface waters is wastewater treatment plant discharges to which chlorine is added as a disinfectant (see sections 4.2.2 and 4.2.3). Use of chlorine since the early 1900's as a disinfectant in both drinking water and sewage before it is discharged to surface water, with the concomitant reduction or elimination of many waterborne infectious diseases has been identified as one of the top ten advances in public health in the last 100 years. There are currently no NPDES permitted discharges to the Swinomish Tribe's fresh waters (section 4.2.2.1, Table 15).

The chemistry of chlorine in freshwater is complex, and there are multiple names in the literature for the same or similar combinations of chlorine chemical forms. The complete list of Chemical Abstracts Service identification numbers (CAS IDs) used in our literature search for toxic effects of chlorine on aquatic life are presented in Table 90, below.

The EPA aquatic life criteria recommendations for chlorine (USEPA, 1985b) describes the toxicity of total residual chlorine (TRC), which is the combined concentration of different chemical forms of chlorine able to react with other substances, or which can interconvert among each other. Within the literature, TRC is generally synonymous with reactive chlorine (RC), combined residual chlorine (CRC), and total available chlorine (TAC).

Total residual chlorine includes free available chlorine (FAC; hypochlorous acid [HOCl] and the hypochlorite ion [OCl^-]; also referred to as free residual chlorine [FRC]) and combined available chlorine (CAC; organic and inorganic chloramines [NH_2Cl or monochloramine, $NHCl_2$ or dichloramine, and NCl_3 or nitrogen trichloride]). Chloramines are also often termed N-chloramides.

Collectively, these products are referred to as "chlorine-produced oxidants," and chlorine toxicity data is therefore referred to as total residual chlorine (TRC) or chlorine-produced oxidants (CPO) (EPA, 1985).

Like elemental chlorine, chlorine dioxide is also a gas at temperatures found in the environment. Rather than hydrolyzing in water as chlorine does, chlorine dioxide forms a true solution in water under typical surface water conditions. Chlorine dioxide is volatile and is easily lost from water. Chlorine dioxide is a powerful oxidant but unlike chlorine, does not readily combine with ammonia to form chloramines. Chlorine dioxide also does not form trihalomethanes. Due to its reactive nature, chlorine dioxide is produced on-site at locations where it is used as a disinfectant.

Monochloramine can be a dominant chemical form if sufficient nitrogen, particularly in the form of ammonia/ammonium ion is present in surface water. Di- and trichloramines are only formed in water at

pH < 6 and when the Cl_2:NH_3 is at least 5:1 (Hankin, 2001). Free chlorine gas (Cl_2) becomes the dominant chemical form only in low organic content waters with a pH < 2. Chlorine can also react with naturally occurring organic matter in water to form a number of disinfection byproducts, including trihalomethane compounds such as chloroform. Free chlorine gas and the chloramines are unlikely to occur in appreciable quantities in marine and estuarine waters, where the ambient pH is approximately 8.0 – 8.1.

The initial chemical reaction when Cl_2 is added to surface water is one of hydrolysis (EPA 1976):

$$Cl_2 + H_2O \rightarrow HOCl + H^+ + Cl^-$$

Hypochlorous acid (HOCl) is a weak acid, and undergoes a pH dependent dissociation:

$$HOCl \leftrightarrow H^+ + OCl^-$$

The release of hydrogen ions from hydrolysis of Cl_2 and the dissociation of hypochlorous acid are the reasons chlorination of surface water tends to reduce the pH of the water. The ratio of HOCl to OCl^- is pH dependent, with 96% HOCl present at pH 6, 75% HOCl at pH 7, 22% HOCl at pH 8, and only 3% HOCl at pH 9. The proportion of HOCl present in water is significant, as HOCl is the chemical form most effective as a disinfectant (Shannon et al., 2008).

Analytical determination of the various chemical species within TRC is generally not performed, and is generally not feasible at the low µg/L concentrations of toxicological relevance in surface waters. This is the reason water quality standards for chlorine, as well as the EPA aquatic life criteria on which they are based are expressed in terms of TRC, not as criteria for the individual chemical forms comprising TRC.

Without continuous addition of chlorine to water, TRC concentrations in water can be quickly reduced through several chemical, physical and biological processes. In addition to the chemical reactions in the water column described above, these processes include volatilization, photodegradation, adsorption on solids, and reactions with aquatic life.

Degradation rates of chlorine species in natural waters are generally rapid, ranging between seconds and hours. The half-life of chlorine gas (Cl_2) in surface water has been reported as 0.005 second (EPA, 1994). Cooper, Jones, Whitehead, and Zika (2007) have performed a number of photodegradation half-life studies with HOCl / OCl^- mixtures under various pH values and water depths, and at several dissolved organic matter concentrations. The light intensity used was based on that at solar noon in both summer and winter at the latitude of Miami, Florida (24° N). In distilled water, the photodegradation half-life of a HOCl / OCl^- mixture ranged between 41 minutes at pH 5.0 to 17 minutes at pH 7.0 to six minutes at pH 12.0. Half-lives of a HOCl / OCl^- mixture were shortest in waters exposed to higher light intensity (i.e. summer light intensities), in waters with the lowest dissolved organic matter concentrations, and in waters of the shallowest depths. Shortest half-lives of just over nine minutes occurred under conditions of summer light intensity in surface water at 0 meters depth and with dissolved organic matter concentrations of either 0.53 or 17.6 mg C/L. The only half-lives longer than 10 hours observed by Cooper et al. (2007) occurred under conditions of water with a depth ≥ 1 meter with a dissolved organic matter concentration of 17.6 mg C/L under either summer or winter light intensity. In water containing 0.53 mg C/L dissolved organic matter and with depth ≤ 5 meters, all HOCl / OCl^- mixture half-lives were 5.85 hours or shorter under all light intensities tested.

The short persistence of chlorine in water relative to the duration of standard toxicity tests with fish and invertebrates has direct bearing on the experimental design of toxicity studies useable to evaluate chlorine toxicity. In order to maintain a consistent concentration of chlorine in laboratory toxicity tests,

flow through studies where chlorine concentrations are constantly replenished are needed. EPA's water quality criteria are designed to apply in situations of continuous exposure to a contaminant. They are not designed to be applied in situations of intermittent contaminant exposure. Much of the available aquatic toxicity data for chlorine describes information generated during either very short term studies (three hours or shorter) or from intermittent exposures. These short term and intermittent studies are not suited for EPA water quality criteria development or evaluation of effects on threatened or endangered species, as they are not representative of effects from continuous exposure to chlorine. The chlorine effects determination within this BE are therefore based only on continuous flow through exposures of acceptable duration (96 hours for acute mortality studies with fish).

The toxicity assessment focuses on surface water concentrations of chlorine that affect the respiratory surfaces of aquatic species. Dietary ingestion, bioaccumulation of and dermal contact with chlorine are insignificant to non-existent exposure routes for both fish and their prey. This being the case, no evaluation of multiple routes of chlorine exposure to aquatic species is performed in this BE. Because chlorine does not bioaccumulate in tissues of aquatic species, no evaluation of effects on aquatic dependent wildlife feeding on aquatic species is possible. This assessment of chlorine is thus limited to evaluating the direct toxicity of chlorine to T&E species and their prey.

5.5.1.4.2 Toxicity assessment

All measures of toxic effect in this toxicity assessment are laboratory toxicity tests where empirically measured chlorine concentrations in water were associated with adverse effects on survival, reproduction or growth of aquatic species. Mixture studies where chlorine was part of a mixture of contaminants to which a test species was exposed are not included in the measures of effect data, as it is generally not possible to attribute the proportion of the response due to chlorine. In order to ensure that constant chlorine concentrations during exposure of test species in toxicity tests was maintained, only flow through toxicity test results are evaluated in this BE.

The four sources of measures of effect are:

1. the acute and chronic toxicity data for aquatic species in the EPA (1985) Ambient Water Quality Criteria for Chlorine, specifically Tables 2 (empirical chronic toxicity) and 3 (empirical rank ordered genus and species mean acute toxicity data) from the chlorine criteria document,
2. the additional toxicity data identified by EPA during its 2012 literature review on chlorine toxicity (*Biological Evaluation of the Coeur d'Alene Tribe Water Quality Standards*)(USEPA, 2013b),
3. a supplemental EPA 2013 literature review that searched specifically for toxicity information on chloramines and other chlorine chemical forms not searched for during the EPA 2012 literature review, also included in EPA 2013,
4. a literature review for chlorine toxicity data published between 2012 and 2022 in ECOTOX, to identify any chlorine toxicity data published since the Coeur d'Alene BE was completed.

Because of chlorine's complex chemistry in aquatic systems, where multiple chemical forms may elicit toxicity to aquatic life, effects of the chemical forms in Table 86, which collectively can be considered total residual chlorine, were searched for in ECOTOX. The ECOTOX search included flow through studies with freshwater species where effects on survival, reproduction and growth were observed.

Table 86. Chemicals for which Aquatic Toxicity Data Searches were Performed in ECOTOX.

Chemical	Chemical Abstracts Service ID
Chlorine (same CAS ID as TRC)	7782-50-5
Chlorine dioxide (chlorine oxide)	10049-00-4
Monochloramine	10599-90-3
Dichloramine	3400-09-7
Trichloramine (nitrogen trichloride)	10025-85-1
Hypochlorous acid	7790-92-3
Hypochlorite anion	14380-61-1
Calcium hypochlorite	7778-54-3

No additional acute or chronic toxicity data meeting current EPA data quality criteria requirements for the listed fish species were found in addition to those already identified in Tables 2 (chronic) or 3 (acute) of the (USEPA, 1985b) chlorine water quality criteria document in any subsequent literature review.

5.5.1.4.3 Effects assessment of acute freshwater chlorine criterion on listed species

The only listed species in the freshwater portions of the action area with empirical acute toxicity data is rainbow trout (*Oncorhynchus mykiss*). The species mean acute value for rainbow trout given in EPA (1985) is 61.92 µg/L for the 96 hour LC_{50}, based on studies by Wolf et al. (1975), Ward et al. (1976), and Ward and DeGraeve (1978). Dividing this species mean acute value by 2.27 to convert it to an LC_{low} results in an acute effects assessment concentration of 27.3 µg/L. The rainbow trout acute effects concentration is higher than the acute chlorine criterion of 19 µg/L. This indicates that exposure at the level of the freshwater acute chlorine criterion is not likely to adversely affect rainbow trout.

No empirical acute toxicity data for chlorine exists for the other listed species of the genus Oncorhynchus present in the action area, chinook salmon (*O. tshawytscha*), or for the salmonid bull trout (*Salvelinus confluentus*). However, the existence of empirical chlorine acute toxicity data for rainbow trout and cutthroat trout (*O. clarkii*) in the genus *Oncorhynchus*, and for brook trout (*Salvelinus fontinalis*), the same genus as bull trout, permits the use of the endangered species module of WebICE to estimate chlorine acute toxicity 96 hour LC_{50} values for Chinook salmon and bull trout.

Table 87 contains the WebICE estimates of the 96 hour LC_{50}s for Chinook salmon and bull trout. These estimates, when divided by 2.27 become the acute effects assessment concentrations for Chinook salmon and bull trout. Two species to species ICE models could be calculated to estimate acute chlorine toxicity to Chinook salmon, using the empirical acute toxicity for rainbow trout and cutthroat trout. The lower of the two ICE estimates for Chinook salmon, that from the cutthroat trout ICE model, was used to estimate the Chinook acute effects assessment concentration. Results of the LC_{low} calculations used as the acute effects assessment concentrations are presented in Table 87.

Table 87. Output from Interspecies Correlation Estimation (ICE) models to estimate acute chlorine toxicity to Chinook salmon and bull trout.

Predicted Taxa	Chinook salmon (*Oncorhynchus tshawytscha*)	Chinook salmon (*Oncorhynchus tshawytscha*)	Salvelinus (predictor for Bull trout *S. confluentus*)
Model Level	species	species	genus
Surrogate	Rainbow trout (*Oncorhynchus mykiss*)	Cutthroat trout (*Oncorhynchus clarkii)*	Brook trout (*Salvelinus fontinalis*)
Estimated Toxicity (mg/L)	92.18	82.03	74.92
95% Confidence Intervals	60.74 – 139.90	26.89 – 250.23	34.34 – 163.47
Df (N-2)	16	4	7
R^2	0.96	0.95	0.88
p-value	0	0.0006	0.0001
Mean Square Error (MSE)	0.07	0.11	0.1
Cross-validation success (%)	94	83	89
Tax. Dist.	1	1	1
Slope	0.94	1.08	0.8
Intercept	0.27	-0.17	0.21

Table 88. Chlorine acute effects assessment concentration estimates for Chinook salmon and bull trout derived from ICE model estimates of 96 hour LC_{50}s.

Species	**ICE estimated 96 hour LC_{50} (µg/L)**	**Assessment effects concentration (µg/L) = LC_{50}/2.27**
Chinook salmon	82.03	36.1
Bull trout	74.92	33.0

The assessment effects concentrations for Chinook salmon and bull trout are all higher than the chlorine acute criterion of 19 µg/L. This indicates that exposure at the level of the freshwater acute chlorine criterion is not likely to adversely affect Chinook salmon or bull trout.

5.5.1.4.4 Effects assessment of chronic freshwater chlorine criterion on listed species

The chronic freshwater toxicity data in EPA (1985) includes no empirical chronic toxicity data for any of the listed fish species found within the action area. Literature reviews of chlorine toxicity data published since 1985 also show no empirical chronic toxicity data available for the listed species in the action area.

In the absence of suitable empirical chronic toxicity data for chlorine with listed species within the action area, the empirical 96 hour LC_{50} acute toxicity concentration for rainbow trout (61.92 µg/L) and the WebICE estimated 96 hour LC_{50} values for Chinook salmon and bull trout from Table 88 are divided by the chlorine acute-chronic ratio (ACR) of 3.345 (USEPA, 1985) to obtain an estimate of chronic no effect concentrations (i.e. chronic assessment effects concentrations for the ESA listed species.

The chronic assessment effects concentrations for the listed species within the freshwater portions of the action area are listed below.

Rainbow trout (steelhead) – 18.5 µg/L

Chinook salmon – 24.5 µg/L

Bull trout – 22.4 µg/L

Chlorine chronic assessment effects concentrations for all four listed species evaluated are higher than the chronic chlorine criterion of 11 µg/L. This indicates that exposure at the level of the freshwater chronic chlorine criterion is not likely to adversely affect Chinook salmon or bull trout.

5.5.1.4.5 Summary of freshwater effect determinations for chlorine

EPA has made the following determinations on the acute and chronic chlorine criterion for the ESA listed species in the freshwater portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above, and the indirect effects analysis for prey availability provided in section 5.8 (the multiple routes of exposure analysis was not relevant as indicated in section 5.7). The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.5.1.5 Chlorpyrifos – Freshwater

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = 0.083 µg/L

Chronic (CCC) criterion = 0.041 µg/L

CAS ID 2921-88-2

5.5.1.5.1 Water-column acute exposure toxicity data

Chlorpyrifos is an organophosphate insecticide historically used on a wide variety of crops including apples, oranges, strawberries, corn, wheat, citrus, and others. Its registration calls for language on product labels stating that chlorpyrifos should not be directly applied to water. Its mode of toxic action is inhibition of acetylcholinesterase (AChE) activity. AChE inhibition adversely affects the nervous system, and can lead to convulsions, muscular paralysis, bronchial constriction, depressed respiration and ultimately death.

The EPA (USEPA, 1986c) chlorpyrifos aquatic life criterion document listed 15 genera of freshwater species with acute toxicity data meeting all EPA quality assurance requirements for use in deriving water quality criteria. In general, invertebrate species are more sensitive to chlorpyrifos than are fish. Of the 15 genera with data of acceptable quality for acute criterion derivation, the seven most sensitive genera are all invertebrates.

Three species of salmonid freshwater acute toxicity tests with chlorpyrifos were used in the derivation of the EPA acute chlorpyrifos criterion. These three species belong to one of two salmonid genera, described below along with the relative sensitivity of the two salmonid genera within the 15 genera used to derive the freshwater acute chlorpyrifos criterion.

Salvelinus (11th most sensitive genus, GMAV = 98 µg/L)

Lake trout (*Salvelinus namaycush*) –SMAV = 98 µg/L

Oncorhynchus (10th most sensitive genus, GMAV = 12.36 µg/L)

Rainbow trout (*Oncorhynchus mykiss*) – SMAV = 8.485 µg/L

Cutthroat trout (*Oncorhynchus clarkii*) – SMAV = 18 µg/L

The most acutely sensitive fish GMAV is *Lepomis*, based on the SMAV of 10 µg/L chlorpyrifos to the bluegill (*Lepomis macrochirus*). Note that the rainbow trout SMAV of 8.485 µg/L is slightly lower than the 10 µg/L bluegill SMAV, even though the *Oncorhynchus* GMAV is higher than the *Lepomis* GMAV (USEPA, 2003). By comparison, the most sensitive genus of any aquatic to chlorpyrifos listed in EPA (USEPA, 1986c) is the amphipod *Gammarus* spp., based on toxicity to three different *Gammarus* species. The GMAV for *Gammarus* is 0.185 µg/L, based on three SMAVs of 0.32 µg/L, 0.11 µg/L and 0.18 µg/L for *G. fasciatus*, *G. lacustris* and *G. pseudolimnaeus*, respectively.

The cutthroat trout, lake trout and one of the rainbow trout SMAVs cited in EPA (USEPA, 1986c) all come from Johnson and Finley (W. W. Johnson & Finley, 1980). The other rainbow trout SMAVs cited in EPA (USEPA, 1986c) come from Holcombe et al. (Gary W Holcombe, Phipps, & Tanner, 1982; G.L. Phipps & G.W. Holcombe, 1985; Gary L Phipps & Gary W Holcombe, 1985), and (Macek, Lindberg, Sauter, Buxton, & Costa, 1976).

5.5.1.5.2 Additional freshwater acute toxicity data for chlorpyrifos published since 1986

EPA (USEPA, 2017a) prepared a national biological evaluation for chlorpyrifos exposure to all ESA listed species in the United States. This national BE included a review of ECOTOX data on chlorpyrifos toxicity to all aquatic species that was available as of the date of the national BE. Once identified and compiled, EPA (2017a) then performed a data quality review of the ECOTOX data, and compiled it into a table of toxicity data of acceptable data quality for use in water quality criteria development (Data Appendix 2-

2.xlsx of EPA 2017)(USEPA, 2017a). Data from the national chlorpyrifos BE was used to identify additional toxicity information not presented in EPA (USEPA, 1986c). None of the studies described in this section were cited in the references section of the EPA (USEPA, 1986c) chlorpyrifos water quality criterion document.

Mayer and Ellersieck (1986b) reported four additional 96 hour LC_{50} values for rainbow trout (*Oncorhynchus mykiss*) not reported in the EPA (USEPA, 1986c) chlorpyrifos criterion document. These 96 hour LC_{50}s were 1 µg/L, 7.1 µg/L, 15 µg/L and 51 µg/L. When these new data are combined with the existing rainbow trout acute toxicity data in EPA (USEPA, 1986c), the SMAV can be recalculated as 8.327 µg/L. This recalculated SMAV is marginally lower than the 8.485 SMAV for rainbow trout in EPA (USEPA, 1986c). What does change in the updated rainbow trout acute toxicity dataset is that the lowest 96 hour LC_{50} is now 1.0 µg/L. This lowest chlorpyrifos 96 hour LC_{50} for rainbow trout from Mayer and Ellersieck (1986b) will be used in this BE instead of the SMAV used in the chlorpyrifos criterion document (USEPA, 1986c) to evaluate the protectiveness of the Swinomish Tribe acute chlorpyrifos criterion to rainbow trout.

Although not an ESA listed species within the action area, EPA (USEPA, 2017a) identified four 96 hour LC_{50}s for cutthroat trout (*Oncorhynchus clarkii*) not described in the chlorpyrifos criterion document (USEPA, 1986c), to augment the one cutthroat trout 96 hour LC_{50} of 18 µg/L (W. W. Johnson & Finley, 1980) listed in the criterion document. Cutthroat trout are in the same genus as three of the four ESA listed freshwater salmonids (rainbow trout, Chinook salmon, and chum salmon) under assessment in this BE. The four new cutthroat trout 96 hour LC_{50} values are all from Mayer and Ellersieck (1986b), and are 13.4 µg/L, 26 µg/L, 18.4 µg/L and 5.4 µg/L. When these new 96 hour LC_{50}s are combined with the single 96 hour LC_{50} listed in EPA (USEPA, 1986c), a SMAV of 14.4 µg/L can be calculated for cutthroat trout.

Both the new rainbow trout and cutthroat trout toxicity data can be useful in the acute effects assessment of chlorpyrifos to members of the genus *Oncorhynchus* that are ESA listed species within the freshwater portions of the action area. The other ESA listed *Oncorhynchus* within the action area are Chinook salmon and chum salmon.

EPA (USEPA, 2017a) also found six new 96 hour LC_{50} values for lake trout (*Salvelinus namaycush*), in the same genus as the fifth ESA listed salmonid (bull trout, *Salvelinus confluentus*) in the freshwater portions of the action area. All of the new lake trout 96 hour LC_{50} data is from Mayer and Ellersieck (F. L. Mayer & M. R. Ellersieck, 1986b). When combined with the one lake trout 96 hour LC_{50} value of 98 µg/L (W. W. Johnson & Finley, 1980) in EPA (USEPA, 1986c), the new lake trout data can be used to calculate a new lake trout SMAV of 141.1 µg/L. The lowest individual toxicity test 96 hour LC_{50} from all of the acceptable lake trout acute toxicity data is 73 µg/L, lower than the 98 µg/L lake trout 96 hour LC_{50} from EPA (USEPA, 1986c).

Although not useable in water quality criteria derivation because only one chlorpyrifos concentration plus a control was tested, Clifford et al. (Clifford, Eder, Werner, & Hedrick, 2005) report an empirically measured 96 hour NOEC of 5.0 µg/L chlorpyrifos for Chinook salmon, one of the ESA listed species in the action area. The Clifford et al. 2005 study was designed to evaluate synergistic toxicity between chlorpyrifos and infectious hematopoietic necrosis virus (IHNV). The 5.0 µg/L chlorpyrifos exposure was the positive control for the study, no significant effect of 5.0 µg/L chlorpyrifos on Chinook salmon survival was noted at the end of the 21 day exposure duration for the complete Clifford et al. (2005)

study. Although Clifford et al. 2005 is not used in this BE to calculate the acute minimum effect threshold for Chinook salmon, it has been used to confirm the calculation of the acute minimum effect threshold of chlorpyrifos for Chinook salmon by other methods.

Finally, EPA (USEPA, 2017a) identified 96 hour LC_{50} data for 3,5,6-trichloro-2-pyridinol (CAS ID 6515-38-4) to five species of *Oncorhynchus*: rainbow trout, Chinook salmon, chum salmon, pink salmon (*Oncorhynchus gorbuscha*) and sockeye salmon (Wan, Moul, & Watts, 1987). 3,5,6-Trichloro-2-pyridinol (3,5,6-TCP) is the primary metabolic transformation product of the parent chlorpyrifos molecule in fish (Chemistry, 1999). With the exception of pink salmon the remaining four *Oncorhynchus* species are the four ESA listed *Oncorhynchus* species within the action area. The 96 hour LC50s for 3,5,6-TCP to rainbow trout, Chinook salmon, chum salmon and sockeye salmon are 1500 µg/L, 2100 µg/L, 1800 µg/L and 2500 µg/L, respectively. The toxicity of the 3,5,6-TCP metabolic transformation product of chlorpyrifos appears to be substantially less than the toxicity of the parent chlorpyrifos.

5.5.1.5.3 Effects assessment of acute freshwater chlorpyrifos criterion on listed species

Rainbow trout (steelhead) is the only ESA listed salmonid to have empirical chlorpyrifos acute toxicity data, allowing a direct assessment of chlorpyrifos acute toxicity to this species. The rainbow trout SMAV of 8.327 µg/L is the recalculated SMAV from use of both the toxicity data in the chlorpyrifos criterion document (USEPA, 1986c) and the rainbow trout toxicity data published subsequent to the issuance of EPA (USEPA, 1986c) chlorpyrifos criterion document. The lowest individual 96 hour LC_{50} for rainbow trout in the chlorpyrifos criteria document (USEPA, 1986c) is a 7.1 µg/L value from Johnson and Finley (1980). Including the new toxicity data that became available since the chlorpyrifos criteria document was issued, the lowest individual study 96 hour LC_{50} is 1.0 µg/L from Mayer and Ellersieck (F. L. Mayer & M. R. Ellersieck, 1986b). Dividing the recalculated rainbow trout SMAV and the lowest individual study acute value by 2.27 results in minimum acute toxicity thresholds of 3.13 µg/L and 0.44 µg/L, respectively. Both of these acute toxicity thresholds are higher than the freshwater acute chlorpyrifos criterion of 0.083 µg/L. Because chlorpyrifos is not expected to elicit any short term acutely toxic effects on rainbow trout (steelhead) at a chlorpyrifos concentration of 0.44 µg/L or lower, exposure at the level of the acute chlorpyrifos criterion of 0.083 µg/L is not likely to adversely affect rainbow trout.

In the hierarchy of biological evaluation methods for assessment of acute water quality data (Table 81 of the BE methodology), the first and preferred tier of the methodology is always to compare empirical acute LC_{50} data for the listed species to the acute criterion itself. If empirical data for the listed species is not available, which is the case for Chinook salmon, chum salmon, and bull trout in this chlorpyrifos evaluation, the second tier of the methodology is to use empirical acute LC_{50} data from a genus-level surrogate species for the listed species as the LC_{50} for the listed species itself. The surrogate species LC_{50} is compared to the acute criterion, assuming the known surrogate species LC_{50} is equal to the unknown LC_{50} of the listed species.

Three of the four ESA listed salmonid species within the freshwater portions of the action area are members of the genus *Oncorhynchus*. This taxonomic similarity permits the lowest empirical 96 hour LC_{50} from rainbow trout to be used to estimate chlorpyrifos acute toxicity to Chinook and chum salmon. In this case we assume that the lowest rainbow trout 96 hour LC_{50} of 1.0 µg/L is equal to the 96 hour LC_{50} for Chinook salmon and chum salmon. Dividing the lowest rainbow trout 96 hour LC_{50} by 2.27 results in a minimum acute toxicity threshold of 0.44 µg/L for Chinook salmon and chum salmon. Because chlorpyrifos is not expected to elicit any short term acutely toxic effects on Chinook or chum

salmon at a chlorpyrifos concentration of 0.083 µg/L or lower, exposure at the level of the acute chlorpyrifos criterion of 0.083 µg/L is not likely to adversely affect Chinook salmon and chum salmon.

The toxicity of chlorpyrifos to the fourth ESA listed salmonid within the freshwater portions of the action area (bull trout) can be estimated from the empirical toxicity of chlorpyrifos to lake trout, as lake trout are in the same genus (*Salvelinus*) as the bull trout. Thus, lake trout empirical toxicity data meet the genus-level surrogate species relationship described as the second tier of the acute effect assessment methods in Table 81. This assumption allows EPA to assume that the LC_{50} for lake trout is equivalent to the bull trout LC_{50} for chlorpyrifos.

The lowest individual chlorpyrifos 96 hour LC_{50} for lake trout is 73 µg/L from Mayer and Ellersieck (F. L. Mayer & M. R. Ellersieck, 1986b). Using lake trout 96 hour LC_{50} data from both EPA (USEPA, 1986c) chlorpyrifos criteria document and that published more recently than was EPA (USEPA, 1986c) results in a lake trout SMAV of 141.1 µg/L. We are unaware of any other chlorpyrifos empirical toxicity data for a member of the genus *Salvelinus*. Using lake trout as the surrogate species for the listed bull trout, dividing the lake trout SMAV and the lowest individual study lake trout acute value by 2.27 results in minimum acute toxicity threshold NOECs of 62.2 µg/L and 32.2 µg/L, respectively. These acute toxicity thresholds are assumed to represent chlorpyrifos concentrations that will not elicit toxic effects on bull trout during short-term exposures to chlorpyrifos. Both of these acute toxicity thresholds are higher than the freshwater acute chlorpyrifos criterion of 0.083 µg/L. Because chlorpyrifos is not expected to elicit any short term acutely toxic effects on bull trout at a chlorpyrifos concentration of 32.2 µg/L or lower, exposure at the level of the acute chlorpyrifos criterion of 0.083 µg/L is not likely to adversely affect bull trout.

5.5.1.5.4 Acute-chronic ratio for chlorpyrifos

The chlorpyrifos ACR in EPA (USEPA, 1986c) is 4.064. This ACR was derived from the geometric mean of five species specific ACRs ranging between 1.374 and 12.5.

5.5.1.5.5 Water-column exposure toxicity data - chronic criterion for chlorpyrifos in freshwater

The chlorpyrifos criteria document (USEPA, 1986c) lists the fathead minnow (*Pimephales promelas*) as the only species with empirical chronic toxicity data meeting data quality requirements for use in deriving an EPA freshwater chronic aquatic life criterion. Fathead minnow is a member of the family *Cyprinidae*, the fish family with the largest number of individual fish species, but not a particularly close taxonomic relative of salmonids. In a 32-day early life-stage test, a chronic value for growth effects using technical-grade chlorpyrifos was 2.26 µg/L (Alfred W Jarvinen & Tanner, 1982). In a multi-generation life-cycle test with fathead minnow (A. W. Jarvinen, Nordling, & Henry, 1983), a reproductive MATC for the number of eggs produced per spawning event occurred at 0.41 µg/L (= geometric mean of the highest NOEC of 0.27 µg/L and LOEC of 0.63 µg/L for egg production) in the first generation, and a 9% growth reduction at 0.12 µg/L in the second generation. Both of these fathead minnow chronic values are higher than the chronic criterion of 0.041 µg/L.

EPA (USEPA, 1986c) identified no other fish or invertebrate species with chronic toxicity data meeting data quality requirements for use in deriving EPA aquatic life criteria.

Table 6 (Other Data on Effects of Chlorpyrifos to Freshwater Aquatic Organisms) of EPA (USEPA, 1986c) contained no salmonid toxicity data that could be used to either directly evaluate chronic toxicity, or

data from possible surrogate fish species that could be used to estimate chlorpyrifos concentrations that elicit chronic toxicity to the ESA listed salmonids within the freshwater portions of the action area. This lack of empirical chronic toxicity data on chlorpyrifos toxicity to salmonids or other surrogate species means that unless additional suitable quality data for chronic toxicity can be identified from studies published since the chlorpyrifos criteria document was issued in 1986, acute toxicity LC_{50} data for the listed and surrogate species used in the effects assessment portion for the freshwater acute chlorpyrifos criterion will need to be divided by the chlorpyrifos ACR of 4.064 to estimate chronic NOEC concentrations for listed salmonids. This process is described in Table 82 (Hierarchical Biological Evaluation Methodology for Tier 2 Assessment of Chronic Water Quality Criteria) of the methodology section of this BE.

5.5.1.5.6 Additional freshwater chronic toxicity data for chlorpyrifos published since 1986

One chronic toxicity study with a salmonid (Eder, Clifford, Hedrick, Kohler, & Werner, 2008) published subsequent to release of the chlorpyrifos criteria document (USEPA, 1986c) was identified. It did not meet data quality requirements for use in deriving an aquatic life criterion.

Eder et al. (2008) exposed Chinook salmon to 3.7 µg/L chlorpyrifos for 96 hours, then observed cumulative mortality and growth effects of the exposure up to 60 days post exposure. No significant effect on either survival or growth was observed at the end of either the 96 hour or 60 day exposure compared to controls. Eder et al. (2008) is not useable in either acute or chronic aquatic life criteria development because only one chlorpyrifos exposure concentration was used in the study, not the required minimum of three exposure concentrations. The no observed effect of 3.7 µg/L chlorpyrifos on either survival or growth after four days exposure and 60 day post exposure may be useful as supporting information for chlorpyrifos effect concentrations derived by other approaches.

Several other chlorpyrifos chronic toxicity studies with species in fish families other than *Salmonidae* have been published since EPA (USEPA, 1986c) was released. With the exception of the Chinook salmon study described previously, all of the chlorpyrifos freshwater chronic toxicity data of acceptable data quality published after 1986 was performed with non-native fish species that have been introduced into the United States and have subsequently established breeding populations. Most of the newer chronic toxicity data are behavioral studies with zebrafish (*Danio rerio*), a member of the family *Cyprinidae* native to Asia, but which has several introduced breeding populations in the United States. Chronic toxicity studies with two species of tilapia (family *Cichlidae*) introduced into the United States are also available. These studies are with Nile tilapia (*Oreochromis niloticus*) and redbelly tilapia (*Copodon zillii*, formerly *Tilapia zillii*), both originally native to Africa. One chronic study with the guppy (*Poecilia reticulata*, family *Poeciliidae*), introduced to the United States from South America is also available. Details of these studies are presented in Data Appendix 2-2.xlsx of EPA (USEPA, 2017a).

When combined with the fathead minnow chronic toxicity data described in the chlorpyrifos aquatic life criteria document (USEPA, 1986c), a total of five fish species (fathead minnow, zebrafish, Nile tilapia, redbelly tilapia and guppy) have chronic toxicity data of acceptable data quality for use in this BE. None of these species are in the same family (*Salmonidae*) as are the five freshwater ESA listed fish species within the action area.

5.5.1.5.7 Effects assessment of freshwater chronic chlorpyrifos criterion on listed species

There are no empirical chronic toxicity data for chlorpyrifos for any of the ESA listed species under assessment. In the hierarchy of biological evaluation methods for assessment of chronic water quality data (Table 82 of the BE methodology), the first two tiers of the methodology are to compare empirical chronic NOEC data or chronic EC_{20} data for the listed species to the chronic criterion itself, which cannot be done for chlorpyrifos.

The third tier of the chronic toxicity methodology, dividing an empirical acute toxicity LC_{50} for the listed species by the ACR for the chemical is applicable to rainbow trout (steelhead) through the use of the lowest individual study 96 hour LC_{50} of 1.0 µg/L from Mayer and Ellersieck (F. L. Mayer & M. R. Ellersieck, 1986b). Dividing this 96 hour LC_{50} by the chlorpyrifos ACR of 4.064 results in a chronic NOEC of 0.246 µg/L. Because chlorpyrifos is not expected to elicit any long-term chronic toxic effects on rainbow trout (steelhead) at a chlorpyrifos concentration of 0.246 µg/L or lower, exposure at the level of the chronic chlorpyrifos criterion of 0.041 µg/L is not likely to adversely affect rainbow trout.

Chronic toxicity of chlorpyrifos to the remaining four ESA listed salmonid species is evaluated through the 6th or 7th tier of the chronic methodology, which is dividing the species-level Interspecies Correlation Estimation (ICE) estimated acute LC_{50} (6th tier) or genus-level ICE estimated acute LC_{50} (7th tier) divided by the ACR for chlorpyrifos. The use of the species-level or genus-level ICE model (Raimondo, Lilavois, & Barron, 2015) is dependent on the amount of species-level or genus-level correlation data between two different taxonomic levels in the ICE program. The endangered species module of WebICE (https://www3.epa.gov/webice/index.html) has been used for the toxicity assessments for bull trout, Chinook salmon, and chum salmon in this section.

This approach results in estimated chronic NOEC concentrations, which are then compared to the chronic chlorpyrifos criterion of 0.041 µg/L. This approach starts with the lowest measured 96 hour LC_{50} for rainbow trout of 1.0 µg/L from Mayer and Ellersieck (F. L. Mayer & M. R. Ellersieck, 1986b) for the three *Oncorhynchus spp*. ESA listed salmonids without empirical acute toxicity data (Chinook and chum salmon), or with the lowest measured 96 hour LC_{50} for lake trout of 73 µg/L from Mayer and Ellersieck (F. L. Mayer & M. R. Ellersieck, 1986b), a member of the genus *Salvelinus*. Bull trout and lake trout are both members of the genus *Salvelinus*, allowing the ICE modeling approach to be used to estimate the chronic toxicity of chlorpyrifos to bull trout.

The ICE predicted 96 hour LC_{50}s from measured salmonid 96 hour LC_{50}s, and the predicted chronic NOECs (predicted 96 hour LC_{50}s divided by the chlorpyrifos ACR) are given in Table 88.

Table 89. Predicted chlorpyrifos chronic no effect concentrations (NOECs) derived from Interspecies Correlation Estimation (ICE) modeled acute toxicity data for listed species.

Predicted species	**Surrogate species**	**Empirical 96 hour LC_{50} for surrogate species (µg/L)**	**Predicted species 96 hour LC_{50} (µg/L)**	**Predicted species chronic NOEC (µg/L)**
Bull trout (*Salvelinus confluentus*)	Lake trout (*Salvelinus namaycush*)	73	46.88	11.5
Chinook salmon (*Oncorhynchus tshawytscha*)	Rainbow trout (*Oncorhynchus mykiss*)	1.0	1.88	0.46

Predicted species	Surrogate species	Empirical 96 hour LC_{50} for surrogate species (µg/L)	Predicted species 96 hour LC_{50} (µg/L)	Predicted species chronic NOEC (µg/L)
Chum salmon (*Oncorhynchus keta*)	Rainbow trout (*Oncorhynchus mykiss*)	1.0	0.988	0.243

The modeled chlorpyrifos chronic NOEC concentrations for bull trout, Chinook salmon, and chum salmon (Table 89 above) are all higher than the chlorpyrifos chronic criterion of 0.041 µg/L. Because chlorpyrifos is not expected to elicit any long-term chronic toxic effects on bull trout, Chinook salmon, or chum salmon at a chlorpyrifos concentration of 0.243 µg/L or lower, which is the lowest of the four predicted chronic NOECs in Table 89, exposure at the level of the chronic chlorpyrifos criterion of 0.041 µg/L is not likely to adversely affect bull trout, Chinook salmon, or chum salmon.

5.5.1.5.8 Summary of freshwater effect determinations for chlorpyrifos

EPA has made the following determinations on the acute and chronic chlorpyrifos criteria for the listed species in the freshwater portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.5.1.6 Chromium, hexavalent – Freshwater

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) Criterion = 16 µg/L (addressed in category 1)

Chronic (CCC) Criterion = 11 µg/L

An assessment is provided here for the chronic chromium (VI) criterion. The acute criterion is addressed in section 5.2.

5.5.1.6.1 Water-column exposure toxicity data – chronic criterion for hexavalent chromium in fresh water

There is extensive data on the toxic effects of chromium on freshwater organisms, but more data exist for hexavalent chromium toxicity than for trivalent chromium. Available data indicate that water hardness has no influence on the toxicity of hexavalent chromium in freshwater; therefore, there is no basis to develop a criterion as a function of water quality. On the other hand, existing freshwater toxicity data show that water hardness has a significant influence on the acute toxicity of trivalent chromium (Schatzow, 1980)

Chronic toxicity of hexavalent chromium in freshwater have been determined for three fish species. Norberg-King (T. Norberg-King, 1987; Norberg - King, 1989a) conducted several subchronic and chronic studies with Fathead Minnow embryos (*Pimephales promelas*). Embryos were subjected to several tests (Tests 9 through 13). Those exposed to chromium in test #11 were less sensitive than unexposed embryos in Test #10. The NOECs were 2,940 and 2,870 µg/L and the weights of the fish were significantly reduced in both tests at 5,560 and 5,490 µg/L, respectively. These exposed embryos (Test #11) were exposed to a 12-day period, yet no greater effect was observed. Renewal Test # 12 and flow-through Test # 13 run simultaneously yielded similar results.

In a study by Cearley (Cearley, 1971), largemouth bass (*Micropterus salmoides*) and bluegill (*Lepomis macrochirus*) were exposed to 50,000 µg/L chromium for periods ranging from 30 to 182 days. These exposures caused significant changes in growth and feeding behavior. Both species were first observed to be less aggressive in their feeding behavior on the fourth day of exposure. By the second week of exposure, several of the bass and bluegill did not feed at all; the remaining fishes were very sluggish when feeding. All of the fishes eventually refused food 4 to 7 days prior to death.

5.5.1.6.2 Effects assessment of chronic freshwater criterion for hexavalent chromium on listed species.

The lowest chronic toxicity value for fish obtained from ECOTOX data pull for hexavalent chromium is from a study by Norberg-King (Norberg - King, 1989a) with Fathead Minnow embryos (*Pimephales promelas*). The study reported a 32-day NOEC of 1,280 µg/L for growth of embryo. This study met all the quality assurance criteria and the data is suitable for use in the Swinomish BE. Therefore, this value when compared to the chronic criterion of 11 µg/L proposed for the Swinomish BE, is higher. If the minimum chronic effect concentration for the remaining listed fish (including salmonids) in the freshwater portions of the action area is greater than or equal to 11 µg/L, exposure at the level of the hexavalent chromium chronic criterion of 11 µg/L is **not likely to adversely affect** rainbow trout (steelhead), Chinook salmon, and bull trout.

5.5.1.6.3 Summary of freshwater effect determinations for hexavalent chromium

EPA has made the following determinations on the chronic hexavalent chromium criterion for the listed species in freshwater portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Chronic criterion:

Rainbow trout (steelhead) – **Likely to adversely affect** (indirect effects - availability of prey)

Chinook salmon – **Likely to adversely affect** (indirect effects - availability of prey)

Bull trout – **Likely to adversely affect** (indirect effects - availability of prey)

Marbled murrelet – **Likely to adversely affect** (indirect effects - availability of prey)

5.5.1.7 Diazinon – Freshwater

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = 0.17 µg/L

Chronic (CCC) criterion = 0.17 µg/L

CAS ID 333-41-5

5.5.1.7.1 Water-column exposure toxicity data - acute criterion for diazinon in freshwater

Diazinon is a broad-spectrum insecticide belonging to the organophosphate group. It is designed to control adult and juvenile forms of flying insects, crawling insects and spiders. It has a molecular weight of 304.35 g/mole and a log octanol/water partition coefficient (log Kow) of 3.40 (Hunter, Faulkner, Culver, & Hill, 1985; WHO, 1998)).

Diazinon has been detected in freshwater and seawater (Bailey et al., 2000; Domagalski, Dubrovsky, & Kratzer, 1997; Land & Brown, 1996; Lowden, Saunders, & Edwards, 1969; McConnell, LeNoir, Datta, & Seiber, 1998; Ritter, Johnson, Lovely, & Molnau, 1974). It has also been detected in point source discharges such as wastewater treatment plant effluents (Villarosa, McCormick, Carpenter, & Marriott, 1994). In the aquatic environment, the fate of diazinon is believed to be regulated by two main processes - chemical hydrolysis and microbial degradation, which are both influenced by pH, temperature and the organic content of the water. Diazinon is stable at pH 7.0 and can persist in the environment for as long as six months (USEPA, 2005b). During chemical hydrolysis, cleavage of the phosphorous-oxygen bond is determined as the critical step (Ku, Chang, & Cheng, 1998). The half-life of Diazinon due to hydrolysis was estimated to be 43.3 days in well water at pH 7.4 to 7.7 and 16ºC (H. G. Morgan, 1976) and 171 days at pH 7.3 and 21ºC (Mansour, Feicht, Behechti, Schramm, & Kettrup, 1999).

The toxicity of diazinon in freshwater has been determined in 13 invertebrate species, 10 fish species and one amphibian species. The acute toxicity ranged from 0.25 µg/L for the cladoceran, *Ceriodaphnia dubia* to 11,640 µg/L for planaria, *Dugesia tigrina* (Phipps, 1988). The most sensitive organisms tested were invertebrates of the Crustacean class. The cladoceran, *C. dubia*, had the lowest GMAV of 0.3773 µg/L; which was calculated from 14 tests (Ankley, Dierkes, Jensen, & Peterson, 1991; T. J. Norberg-King, 1987b).

Freshwater fish species tested showed moderate sensitivity to diazinon. The SMAVs ranged from 425.8 µg/L for the rainbow trout (*Oncorhynchus mykiss)*, to 9,000 µg/L for the goldfish (*Carassius auratus)*. The LC50 values reported from five tests ranged from 90 µg/L (Cope, 1965; Corporation, 1976; W. W. Johnson & Finley, 1980; F. L. Mayer & M. R. Ellersieck, 1986b) to 3,200 µg/L (Bathe et al., 1975).

Cutthroat trout (*Oncorhynchus clarki)*, was reported to be generally less sensitive to diazinon (SMAV = 2,166 µg/L) than rainbow trout (SMAV = 425.8 µg/L). Other warmwater fish species like the flagfish *(Jordanella floridae*), fathead minnow (*Pimephales promelas*), goldfish (*Carassius auratus*), and zebrafish (*Danio rerio*; formerly *Brachydanio rerio*) are less sensitive to diazinon than the coldwater species, rainbow trout, brook trout (*Salvelinus fontinalis*), and lake trout (*Salvelinus namaycush*). The only exceptions include the warmwater bluegill, which is more sensitive to diazinon than the coldwater fish species, and the coldwater cutthroat trout, which is less sensitive than the warmwater flagfish (USEPA, 2005b).

5.5.1.7.2 Effects assessment of acute freshwater diazinon criterion on listed species

The lowest 4-day acute effect concentration for salmonids based on the ECOTOX data pull is from a study by Banaee et al. (Banaee, Sureda, Mirvaghefi, & Ahmadi, 2011) which reported an LC_{50} of 1170 µg/L for rainbow trout (*Oncorhynchus mykiss*). This study was published after the release of the diazinon criteria document; therefore, was not used in the derivation of the acute freshwater criterion. In addition, the study met all the quality assurance criteria and deemed suitable for use in the Swinomish BE. To estimate the acute NOEC for all salmonids, the study's LC_{50} is divided by 2.27, that is: 1170 µg/L /2.27 = 515.4 µg/L. This value is higher than the diazinon acute freshwater criterion of 0.17 µg/L. If the minimum acute effect concentration for the remaining listed salmonids in the freshwater portions of the action area is greater than or equal to 515.4 µg/L, exposure at the level of the diazinon freshwater acute criterion of 0.17 µg/L is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, and bull trout.

5.5.1.7.3 Water-column exposure toxicity data - chronic criterion for diazinon in freshwater

Chronic toxicity of diazinon in freshwater has been determined in 4 freshwater species, 1 invertebrate and 3 fish. T. Norberg-King (1987) conducted a 7-day life cycle study with Ceriodaphnia dubia (C. *dubia*) in diluted reconstituted mineral water, all test organisms survived in the 3 lowest concentrations of diazinon (0.063, 0.109, and 0.220 µg/L). No reproductive effects were also observed in the lowest concentration. However, no test organisms survived at diazinon concentrations greater than 0.520 µg/L. Another study conducted at the same laboratory with the same dilution water by T. J. Norberg-King (1987a) and Ankley et al. (1991), reported a 48-hr acute value of 0.3760 µg/L. Since a 7-day chronic value of 0.3382 µg/L has been established for diazinon (Norberg-King 1987)(T. J. Norberg-King, 1987b), the acute to chronic ratio (ACR) is derived by dividing the 48-hr acute value (0.3760 µg/L) by the chronic value (0.3382 µg/L), and the result is 1.112 for C. *dubia.*

Partial life cycle test with brook trout (*Salvelinus fontinalis*) with yearlings exposed for 173 days to measured concentrations ranging from 0.55 to 9.6 µg/L After 173 days, survival of parental stock was significantly reduced at 9.6 µg/L, but not at 4.8 µg/L, but some deformities were seen and the instantaneous growth rate was reduced at 4.8 µg/L diazinon (Allison & Hermanutz, 1977). In another study with brook trout larvae exposed for 122 days showed effects after 30 days. Thus, the chronic value for brook trout derived from this study was < 0.8 µg/L, which appeared as the lowest exposure concentration in which growth effects were observed.

Allison and Hermanutz (1977) calculated an acute value of 723 µg/L, from three 96-hr acute tests they conducted and derived an ACR for brook trout by dividing the acute value by the chronic value of 0.8 µg/L, which result in an ACR of 903.8 for brook trout.

Norberg-King (1989a) exposed fathead minnow embryos and the resulting larvae to diazinon for 32 days in an early-life stage test. At the end of the test, the wet weight and survival of fish exposed to the highest concentration (285 µg/L) were significantly different from the control. A chronic value for the test was 24.97 µg/L and was derived based on reduced dry weight.

In another study by Allison and Hermanutz (Allison & Hermanutz, 1977) conducted with flagfish (*Jordanella floridae)* larvae that lasted for 60 days and continued with the fish progeny for 35 days post-hatch, the only significant effect observed was a 23.3 percent reduction in the average weight of male fish after 61 days exposed to 88 µg/L diazinon. Thus, the chronic value for flagfish, is the geometric mean of 54 µg/L (NOEC) and 88 µg/L (LOEC), or 68.93 µg/L, based on Hatch success.

5.5.1.7.4 Effects assessment on freshwater chronic diazinon criterion on listed species

The lowest 32-day chronic no effect concentration for fish based on the Ecotox data pull is from a study by Norberg-King (1989a). The study reported a lowest chronic NOEC of 16.5 µg/L for fathead minnow (*Pimephales promelas*). This study was cited and used in deriving the Aquatic Life Criteria for the diazinon (Brooke & Smith, 2005). The chronic no effect concentration of 16.5 µg/L of will be used for all fish. This value is higher than the diazinon freshwater chronic criterion of 0.17 µg/L. If the minimum acute effect concentration for the remaining listed salmonids in the freshwater portions of the action area is greater than or equal to 515.4 µg/L, exposure at the level of the diazinon freshwater chronic criterion of 0.17 µg/L is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, and bull trout.

5.5.1.7.5 Summary of freshwater effect determinations for diazinon

EPA has made the following determinations on the acute and chronic diazinon criteria for the listed species in the freshwater portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.5.1.8 Hydrogen sulfide – Freshwater

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion – No criterion

Chronic (CCC) criterion – 2.0 µg/L

CAS ID – 7783-06-4 (hydrogen sulfide)
1313-82-2 (sodium sulfide)
16721-80-5 (sodium bisulfide)

5.5.1.8.1 Introduction to hydrogen sulfide

The EPA aquatic life criteria document for hydrogen sulfide was originally published in EPA (USEPA, 1976f), commonly known as the "Red Book". The most recently published hydrogen sulfide aquatic life criteria document was published in EPA (USEPA, 1986f), commonly known as the "Gold Book". The EPA (USEPA, 1986f) hydrogen sulfide criterion is a largely verbatim republication in a more legible form of EPA (USEPA, 1976f). As there are no differences in the chronic criterion concentration or the literature cited to derive both the EPA (USEPA, 1976f) and EPA (USEPA, 1986f) hydrogen sulfide criteria documents, we cite EPA (USEPA, 1976f) as the source of the hydrogen sulfide criterion in this BE.

Hydrogen sulfide (H_2S) is a colorless gas at ambient temperatures greater than -60°C. At low concentrations it has a characteristic smell of rotten eggs. Its water solubility varies with water temperature, with maximum solubilities reported as 5.3 g/L (= 5,300,000 µg/L) at 10°C, 3.98 g/L at 20°C and 3.2 g/L at 30°C (ATSDR, 2016). These maximum water solubilities are well in excess of the EPA (USEPA, 1976f) chronic water quality criterion of 2.0 µg/L. The rotten egg odor of sulfide in water is detectable at concentrations greater than 29 ng/L (ATSDR, 2016), a concentration approximately 69 times lower than the EPA chronic aquatic life criterion.

A brief description of the water chemistry of hydrogen sulfide is given in ANZECC/ARMCANZ (ANZECC & ARMCANZ, 2000). Hydrogen sulfide is a diprotic acid that dissociates in aqueous solution to form an equilibrium between unionized H_2S, bisulfide ions HS^- and sulfide ions S^2.

The following equations illustrate the equilibria:

$$H_2S \leftrightarrow HS^- + H^+ \quad K_1$$

$$HS^- \leftrightarrow S^{-2} + H^+ \quad K_2$$

At environmental conditions (pH < 10), only the first dissociation (K_1) is significant hence the concentration of S^{-2} is negligible compared to the concentrations of H_2S and HS^-. For instance at pH 9, around 99% is in the form of HS^- and at pH 5 about 99% is at H_2S. At a pH of 7.04, the concentrations of H_2S and HS^- are equal. The unionized H_2S gas is believed to be the chemical form that elicits toxicity to aquatic life. As such, it is likely that hydrogen sulfide is more toxic to aquatic life in acidic and

circumneutral waters than it is in more alkaline waters. This pH dependent chemical speciation of H_2S may be responsible in part for the lower toxicity of H_2S in marine waters, with a pH of approximately 8, in which the total sulfide concentration consists of about 9% H_2S. To date we are unaware of any effort to develop a pH dependent hydrogen sulfide aquatic life criterion as currently exists for pentachlorophenol.

The concentrations of sulfide are usually expressed either as total sulfides (the sum of concentrations of H_2S, HS^- and acid-soluble metallic sulfides present in solution) or in terms of unionized hydrogen sulfide H_2S. Either expression for concentration may take into account the amount of sulfide as H_2S.

In addition to the toxicity information on H_2S itself available in the EPA ECOTOX database, some toxicity studies have used either sodium sulfide (Na_2S) or sodium bisulfide (NaSH) as the added toxicant. Both of these compounds are water soluble solids at room temperature, quickly release H_2S in water, and as such can be easier to work with in aquatic toxicity tests than is H_2S gas.

Hydrogen sulfide is both a naturally occurring chemical and a pollutant discharged into surface waters from multiple industrial processes. It forms naturally in anaerobic aquatic systems such as those found in thermally stratified lakes. It is also naturally occurring in hot springs, volcanic gases and crude oil. In aerobic aquatic systems, its half-life ranges between one hour and several hours (Environment and Climate Change Canada and Health Canada (ECCC, 2017), depending on the amount of volatilization into air from the water, pH and temperature of the water, and concentrations of metals, suspended solids and organic matter in the water. Industrial operations that release hydrogen sulfide include oil and gas facilities, kraft pulp and paper mills, wastewater treatment systems, mining production, and intensive livestock operations.

5.5.1.8.2 Water-column exposure toxicity data – acute hydrogen sulfide toxicity in freshwater

While the Swinomish Tribe is not proposing an acute criterion for hydrogen sulfide, a discussion of acute toxicity data in the EPA (USEPA, 1976f) hydrogen sulfide criteria document is warranted. This is because the EPA (USEPA, 1976f) chronic hydrogen sulfide criterion under assessment in this BE was based largely on acute toxicity data where short-term survival was the measured endpoint. To this day, relatively little empirical hydrogen sulfide chronic toxicity data for aquatic life is available for aquatic species. Thus, the chronic toxicity evaluation presented below is largely based on empirical acute toxicity data translated into modeled chronic toxicity data.

The acute toxicity data cited in EPA (USEPA, 1976f) for acute toxicity of H_2S to fish is summarized in the table below. When multiple 96 hour LC_{50}s are reported for a species within a single study, only the lowest 96 hour LC_{50} is shown below.

Table 90. Summary of the acute toxicity data cited in EPA (1976f) for acute toxicity of H2S to fish. When multiple 96 hour LC50s are reported for a species within a single study, only the lowest 96 hour LC50 is shown

Species	**96-hour LC_{50} (µg/L)**	**Citation**
Walleye (*Sander vitreum*)	7 (Sac fry)	(L. Smith & Oseid, 1972)
White sucker (*Catostomus commersoni*)	13 (Sac fry)	(L. Smith & Oseid, 1972)

Species	96-hour LC_{50} (µg/L)	Citation
Rainbow trout (*Oncorhynchus mykiss*)	< 5 (Sac fry)	(L. Smith & Oseid, 1972)
Northern pike (*Esox lucius*)	8 (Sac fry)	(Adelman & Smith, 1970)
Fathead minnow (*Pimephales promelas*)	7 (Fry)	(L. Smith & Oseid, 1974)
Brook trout (*Salvelinus fontinalis*)	17 (Juvenile)	(L. Smith & Oseid, 1974)

5.5.1.8.3 Additional freshwater acute toxicity data for hydrogen sulfide published since 1976

There are three summaries of the literature regarding hydrogen sulfide toxicity to aquatic life published since the hydrogen sulfide criterion was issued (USEPA, 1976f). EPA (USEPA, 1976b) describes results of acute and chronic toxicity test results for seven fish and eight invertebrate species. These studies, performed at EPA's Duluth, Minnesota aquatic toxicology laboratory, were apparently not completed in time for inclusion in the EPA (USEPA, 1976f) hydrogen sulfide criteria document. EPA (USEPA, 2009c) and Environment and Climate Change Canada and Health Canada (Environment and Climate Change Canada and Health Canada (ECCC, 2017) both contain updated literature reviews of toxicity studies performed since 1976. These reviews were used to identify and obtain papers for use in this toxicity assessment of hydrogen sulfide.

Although there are multiple studies of the acute toxicity of hydrogen sulfide published since the EPA (USEPA, 1976f) H_2S aquatic life criteria was published, among the most pertinent for this BE is that of Fung and Bewick (Fung & Bewick, 1980). Fung and Bewick (Fung & Bewick, 1980) measured 96 hour LC_{50}s for multiple life stages of four freshwater fish species: lake whitefish (*Coregonus clupeaformis*), yellow perch (*Perca flavescens*), largemouth bass (*Micropterus salmoides*) and rainbow trout (*Oncorhynchus mykiss*). Rainbow trout are one of the ESA listed species within the action area, while lake whitefish is also a member of the family Salmonidae, the same family to which all ESA listed salmonids in the action area belong.

The measured 96 hour LC_{50} for juvenile rainbow trout from Fung and Bewick (1980) was 7 µg/L hydrogen sulfide, the lowest 96 hour LC_{50} of any rainbow trout lifestage tested in this study. Fung and Bewick (1980) observed a general trend of lifestage sensitivity where the youngest rainbow trout lifestages were the most tolerant of hydrogen sulfide (e.g. 96 hour LC_{50} of 47 µg/L for sac fry) to the oldest lifestages being the most sensitive to hydrogen sulfide (e.g. the previously mentioned 96 hour LC_{50} of 7 µg/L for juveniles and a 12 µg/L 96 hour LC_{50} for adult rainbow trout.

Lake whitefish are even more sensitive to hydrogen sulfide than are rainbow trout, with a sac fry 96 hour LC_{50} of 2 µg/L and a juvenile 96 hour LC_{50} of 12 µg/L. Fung and Bewick (1980) did not report a 96 hour LC_{50} for adult whitefish. Notably, the lake whitefish 96 hour LC_{50} of 2 µg/L is identical to the EPA (USEPA, 1976f) chronic hydrogen sulfide criterion of 2 µg/L. As it is likely that the chronically toxic concentration of hydrogen sulfide to lake whitefish and other members of the family Salmonidae is lower than the empirically measured acutely toxic concentration of 2 µg/L, the acute toxicity results in Fung and Bewick (1980) for Salmonidae provide evidence that the EPA (USEPA, 1976f) chronic hydrogen sulfide criterion of 2 µg/L may not be protective of ESA listed Salmonidae within the action area.

Although not a member of the Salmonidae, Fung and Bewick (1980) found yellow perch sac fry to be even more sensitive to hydrogen sulfide than lake whitefish or rainbow trout. The yellow perch 24 hour

LC_{50} was 2 µg/L, the same concentration as was the 96 hour LC_{50} of 2 µg/L for lake whitefish sac fry. The 48, 72 and 96 hour LC_{50}s for yellow perch sac fry were all <2 µg/L. Finally, Fung and Bewick (Fung & Bewick, 1980) found largemouth bass (*Micropterus salmoides*) to be the most tolerant of the four species they tested, with a 96 hour LC_{50} of 11 µg/L for sac fry. Juvenile largemouth bass had a 96 hour LC_{50} of 63 µg/L, the highest 96 hour LC_{50} of any lifestage of any species tested.

Several other acute toxicity studies with salmonids are available. Smith and Oseid (L. Smith & Oseid, 1975), in a study not reported in EPA (USEPA, 1976f) observed a 96 hour LC_{50} of 21.6 µg/L for brook trout swim-up fry. EPA (USEPA, 1976b) reported a brook trout sac fry 96 hour LC_{50} as low as 13.8 µg/L. Brook trout are in the same genus (*Salvelinus*) as bull trout, and are a taxonomically close relative to one of the ESA listed species within the action area. Reynolds and Haines (F. A. Reynolds & Haines, 1980) reported a 96 hour LC_{50} of 7 µg/L for brown trout (*Salmo trutta*) sac fry. EPA (USEPA, 1976b) reported a juvenile rainbow trout 96 hour LC_{50} of 12.5 µg/L, within a factor of two of the 7 µg/L 96 hour LC_{50} for juvenile rainbow trout reported by Fung and Bewick (1980).

Other 96 hour LC_{50} values for various fish species reported in EPA (USEPA, 1976b) include a relatively low 8.6 µg/L for bluegill (*Lepomis macrochirus*) swim-up fry, and a relatively high 25 µg/L for goldfish fry (*Carassius auratus*).

5.5.1.8.4 Water-column exposure toxicity data - chronic hydrogen sulfide toxicity in freshwater

The chronic criterion of 2.0 µg/L in EPA (USEPA, 1976f) is derived from a chronic study with bluegills reported in Smith and Oseid (L. Smith & Oseid, 1974). Egg production was significantly reduced after a 46 day exposure to 1.4 µg/L hydrogen sulfide. Juvenile white suckers showed growth reductions from exposure to 1 µg/L hydrogen sulfide according to EPA (USEPA, 1976f), although the original literature citation for the white sucker growth reduction was not provided in EPA (USEPA, 1976f).

5.5.1.8.5 Additional chronic toxicity data for hydrogen sulfide published since 1976

EPA (USEPA, 1976b) measured the chronic toxicity of hydrogen sulfide to seven species of freshwater fish: fathead minnows, goldfish, bluegill, walleye, white sucker, brook trout, and rainbow trout. Long-term effects on survival, growth, reproduction, and behavior were evaluated in studies whose durations ranged from 45 days (brook trout reproduction) to 826 days (bluegill multi-generation study on survival, growth and reproduction). Recommendations from EPA (USEPA, 1976b) included that 0.002 mg/liter H_2S be considered a safe limit for protection of all fish species and bottom-inhabiting invertebrates in areas of lakes and streams not used for spawning by nesting fish, and that in areas used for reproduction by nesting fish concentrations of 0.001 mg/liter H_2S not be exceeded.

A summary of the lowest chronic MATCs (calculated as the geometric mean of the NOEC and LOEC published in Table 91 of EPA (USEPA, 1976b)) for six fish species is presented in the table below.

Table 91. Lowest chronic MATCs for hydrogen sulfide for six fish species

Species	Lifestage	MATC (µg/L)	Endpoint
Fathead minnow	Sac fry	0.0047	Growth, reproduction
Goldfish	Adult	0.0071	Reproduction
Bluegill	Adult	<0.0007	Reproduction
Rainbow trout	Sac fry	0.0040	Growth
Brook trout	Adult	<0.0055	Reproduction
Walleye	Juvenile	0.0040	Survival

Confidence limits were not provided in EPA (USEPA, 1976b) for any of the measured endpoints, only the range of measured exposure concentrations in each study, the observed NOEC and the observed LOEC values. Thus, it is unknown whether any of the chronic MATCs in the above table, except for that of the bluegill, have lower confidence limits below the 0.002 µg/L hydrogen sulfide chronic criterion.

The lowest chronic LOEC identified in a freshwater fish since the publication of the chronic hydrogen sulfide criterion (USEPA, 1976f) is a 5.4% growth reduction in a 28 day study of juvenile Asian redtail catfish (*Hemibagrus nemurus*) exposed to 0.5 µg/L hydrogen sulfide (Hoque, Yusoff, Law, & Syed, 1998). The 0.5 µg/L H_2S concentration was the lowest of the four H_2S exposure concentrations tested, meaning a chronic NOEC cannot be calculated for this species. As the Asian redtail catfish is native to southeast Asia, and has not been introduced into North America, the toxicity test results for this species cannot be used in the derivation of EPA aquatic life criteria according to the Guidelines. But the 0.5 µg/L chronic LOEC in Hoque et al. (1998) is the lowest known chronic toxicity effect concentration for hydrogen sulfide of any freshwater fish species.

5.5.1.8.6 Acute-chronic ratio for hydrogen sulfide

EPA water quality criteria documents do not contain any estimates of or empirically determined ACRs for use with hydrogen sulfide toxicity data. EPA (USEPA, 1976b) does summarize several application factors (the inverse of the ACR) for several taxonomic groups of fish derived from the empirical toxicity test data presented in the report. For trout, EPA (USEPA, 1976b) reports ACRs between 3 - 5 (converted from the application factors originally reported in the document), and ACRs between 15 – 20 in warmwater fish.

However, the European Centre for Ecotoxicology and Toxicology of Chemicals (ECETOC, 2003) has calculated ACRs for a number of chemicals, including hydrogen sulfide using an ACR derivation approach that combines toxicity data from mixed species (i.e. where an acute toxicity point for a chemical from one or more species could be compared with a chronic value from one or more species, not necessarily including the species used to derive the acute value). This approach differs from the EPA approach to ACR determination, which requires both empirical acutely toxic concentrations and empirical chronic no effect concentrations for the same species, although not necessarily from the same study to derive the ACR.

For freshwater fish, ECETOC (2003) has estimated a hydrogen sulfide ACR of 8.2, within the range of ACRs for fish species described in EPA (USEPA, 1976b). As the ECETOC estimated hydrogen sulfide ACR is also comparable to the median ACR of 8.3 calculated from a compilation of 456 same species pairs ACRs by Raimondo et al. (S. Raimondo et al., 2007), used in this BE's methodology as a default ACR for chemicals without a published empirically determined ACRs, we employ the ECETOC (2003) hydrogen

sulfide ACR of 8.2 for converting empirical acute toxicity LC_{50}s to estimated chronic MATCs or NOECs in this BE.

5.5.1.8.7 Effects assessment of chronic hydrogen sulfide criterion on listed species

The 96 hour LC_{50} of 2 µg/L for lake whitefish, a member of the family *Salmonidae*, equivalent to the chronic criterion for all species is indicative that the chronic hydrogen sulfide criterion may not be protective of both lake whitefish and other *Salmonidae* during chronic exposures. Further evidence of this lack of protectiveness for members of the *Salmonidae* can be evaluated by dividing the lowest measured 96 hour LC_{50} for each salmonid species for which empirical acute toxicity data are available by the assumed ACR for hydrogen sulfide of 8.2. This information is shown in the table below.

Table 92. The 96 hour LC50 of 2 µg/L for lake whitefish

Species	**H_2S 96 hour LC_{50}**	**Citation**	**H_2S Chronic NOEC (96 hour LC_{50} / ACR**
Brown trout (*Salmo trutta*)	7	(F. A. Reynolds & Haines, 1980)	0.85
Rainbow trout (*Oncorhynchus mykiss*)	< 5	(L. Smith & Oseid, 1972))	<0.61
Brook trout (*Salvelinus fontinalis*)	13.8	(USEPA, 1976b)	1.7
Lake whitefish (*Coregonus clupeaformis*)	2	(Fung & Bewick, 1980)	0.24

The above table contains estimated chronic NOEC concentrations for species within four genera of *Salmonidae* (*Coregonus*, *Oncorhynchus*, *Salmo*, *Salvelinus*) that are all lower than the chronic hydrogen sulfide criterion of 2 µg/L. All ESA listed salmonids within the action area are members of either the genus *Oncorhynchus* or genus *Salvelinus*. Thus, evidence indicates that the chronic hydrogen sulfide criterion may not protect multiple salmonid species from chronic toxicity of hydrogen sulfide.

Evidence further indicates that members of the *Salmonidae* may not be the only fish species where the chronic criterion for hydrogen sulfide is not protective of fish from chronic effects of hydrogen sulfide. The 96 hour LC_{50} of < 2 µg/L determined by Fung and Bewick (1980) for yellow perch sac fry indicates that the chronic criterion may adversely affect short term survival of yellow perch (family *Percidae*), as well as resulting in longer term adverse chronic effects of hydrogen sulfide. The 96 hour LC_{50} of 7 µg/L determined for walleye (also a member of the *Percidae*) by Smith and Oseid (L. Smith & Oseid, 1972) divided by the hydrogen sulfide ACR of 8.2 results in a chronic walleye NOEC of 0.85 µg/L, lower than the chronic 2 µg/L criterion. Other taxonomic families where either the empirical or estimated chronic LOEC or NOEC of one or more members of the family is lower than the 2 µg/L chronic criterion include members of the *Catostomidae* (white sucker), *Centrarchidae* (bluegill), *Cyprinidae* (fathead minnow) and *Esocidae* (northern pike).

The estimation of chronic NOEC hydrogen sulfide concentrations for multiple fish species from multiple taxonomic families provides support for the following conclusion. Exposure at the level of the Swinomish Tribe's chronic criterion of 2 µg/L for hydrogen sulfide is likely to adversely affect multiple fish species and families.

5.5.1.8.8 Summary of freshwater effect determinations for hydrogen sulfide

EPA has made the following determinations on the chronic hydrogen sulfide criterion for the listed species in the freshwater portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above, and the indirect effects analysis for prey availability provided in section 5.8 (the multiple routes of exposure analysis was not relevant as indicated in section 5.7) . The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Chronic criterion:

Rainbow trout (steelhead) – **Likely to adversely affect** (water column exposure)

Chinook salmon – **Likely to adversely affect** (water column exposure)

Bull trout – **Likely to adversely affect** (water column exposure)

Marbled murrelet – **Not likely to adversely affect**

5.5.1.9 Iron – Freshwater

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = None

Chronic (CCC) criterion = 1000 µg/L (total recoverable)

CAS ID 7439-89-6

5.5.1.9.1 Water-column exposure toxicity data - chronic criterion for iron in freshwater

Iron can exist in two valence states in freshwater. The most studied valence state is ferrous or divalent iron (Fe^{+2}), which is water soluble at any pH and can be released from sediment to surface water under anaerobic conditions. Water solubility of ferrous iron can exceed 100,000 µg/L under circumneutral pH conditions.

Toxicity of ferric or trivalent iron (Fe^{+3}) is less well studied, due to its negligible water solubility. At pH > 3.5 ferric iron precipitates out of solution, forming a flocculant or solid material commonly known as yellowboy. At pH > 5, the water solubility of ferric iron is approximately 10 µg/L. Above 10 µg/L the ferric iron is present only in a suspension or complexed form (J. Hem & Cropper, 1962). Yellowboy can elicit toxicity in aquatic species by coating or covering gills or respiratory surfaces, causing suffocation. However, this toxicity is a physical toxicity, not a chemical toxicity, and will not be discussed further in this BE.

The original iron criterion (USEPA, 1976c), unlike most EPA metals criteria, is expressed in terms of total recoverable iron.

None of the action area listed species in freshwater had empirical iron chronic toxicity data available at the time of publication of the EPA (USEPA, 1976c) iron criteria. The EPA (USEPA, 1976c) chronic iron criterion was based on a limited data set which included several field surveys describing the presence or

absence of fish at various iron concentrations. EPA (USEPA, 1976c) did not refer to any of several chronic toxicity studies performed by Sykora and co-workers (E. J. Smith & Sykora, 1976; J. Sykora, Smith, Synak, & Shapiro, 1975; J. L. Sykora, Smith, & Synak, 1972; Updegraff & Sykora, 1976), mostly with brook trout (*Salvelinus fontinalis*), a salmonid in the same genus as bull trout (*Salvelinus confluentus*), but also several studies with coho salmon (*Oncorhynchus kisutch*).

A close reading of Sykora et al. (1972) indicates that Figure 5 of the paper combined with information in the text of the study contains sufficient information to qualitatively estimate a 35-day (5-week) NOEC for growth of brook trout. This 5-week growth NOEC is approximately 12,000 µg/L total iron, the nominal exposure concentration for one of the four exposure concentrations studied. The mean iron concentration in the nominal 12,000 µg/L exposure during this 35-week long study was 13,420 µg/L, ranging between 7700 – 19,000 µg/L. Brook trout growth in the nominal 6000 µg/L iron concentration (mean = 7800 µg/L) for the entire 35 week duration of the study was not significantly different from growth in controls

5.5.1.9.2 Additional chronic toxicity data for iron published since 1976

Brenner and Cooper (1978) studied the effects of iron hydroxide on coho salmon (*Oncorhynchus kisutch*) in a 90 day exposure, starting with fertilized eggs and continuing to the alevin stage. No effect on the hatchability, embryonic development, survival and maturation of coho salmon exposed to 3000 µg/L iron was observed at the end of the 90 day exposure. Unfortunately, Brenner and Cooper (1978) only exposed fish to the one concentration of 3000 µg/L, making the study unusable in aquatic life criteria development (a minimum of three exposure concentrations plus a control are required for a study to be considered for criteria development).

The EPA ECOTOX database provides limited information about a development study with rainbow trout performed by Amelung (Amelung, 1982). ECOTOX provides insufficient information about the Amelung (1982) study to determine if it meets quality assurance requirements for inclusion in this effects assessment, specifically with respect to the actual toxicological endpoint measured, whether the effect is a LOEC, NOEC or an ECx value, the number of exposure concentrations tested and the duration of exposure. The Amelung (1982) study is in an obscure journal (Archiv für Fischereiwissenschaft), and the study could not be obtained by EPA Region 10. However, since this appears to be the only available chronic toxicity study showing iron effects on a listed species within the action area, it is worthwhile to mention the reported effect concentration on trout development of 5700 µg/L, higher than the 1000 µg/L iron chronic criterion concentration.

Cadmus et al. (P. Cadmus, S. F. Brinkman, & M. K. May, 2018) performed 30 day chronic growth toxicity studies on two members of the family *Salmonidae*, although neither is a listed species within the action area. The 30-day growth EC_{20} concentrations were >5146 µg/L and 1318 µg/L for brown trout (*Salmo trutta*) and mountain whitefish (*Prosopium williamsoni*), respectively. As all data quality requirements for aquatic life criteria derivation appear to have been met in Cadmus et al. (2018), the 30-day growth EC_{20} concentration for mountain whitefish will be assumed to be the lowest chronic minimum effect threshold concentration for all listed salmonid species in the freshwater portions of the action area.

In recent years as discussed in the BE methodology, the use of chronic EC_{20} concentrations has increasingly replaced and is preferred over the use of NOECs, NOELs and MATCs as the reported chronic no effect concentration. This is because of both statistical and biological concerns with NOECs, NOELs

and MATCs. In particular, statistical no effect or insignificance, as implied in a NOEC or NOEL does not guarantee ecological, biological, or ecotoxicological insignificance.

5.5.1.9.3 Acute-chronic ratio for iron

Because an empirical 30-day chronic growth toxicity study exists for a fish species in the same family (*Salmonidae*) that contains all of the listed species in the action area, the minimum chronic effect threshold concentration for listed species is estimated by assuming the lowest empirically measured minimum chronic effect threshold concentration is as low or lower than any minimum chronic effect threshold concentration for any salmonid species. Using this assumption, use of an ACR to convert an iron 4-day LC_{50} to a chronic NOEC is not needed to complete the effects assessment for the chronic iron criterion.

5.5.1.9.4 Effects assessment of chronic iron criterion on listed species

The lowest chronic minimum effect threshold concentration for any salmonid species is 1318 µg/L, the mountain whitefish 30-day EC_{20} for growth. This value is higher than the iron chronic criterion of 1000 µg/L. Assuming that the minimum chronic effect concentration for the remaining listed salmonid species in the freshwater portions of the action area is greater than or equal to 1318 µg/L, exposure at the level of the iron chronic criterion of 1000 µg/L is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, and bull trout.

5.5.1.9.5 Summary of freshwater effect determinations for iron

EPA has made the following determinations on the chronic iron criterion for the listed species in the freshwater portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above, and the indirect effects analysis for prey availability provided in section 5.8 (the multiple routes of exposure analysis was not completed as indicated in section 5.7). The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Chronic criterion:

Rainbow trout (steelhead) – **Likely to adversely affect** (indirect effects - availability of prey)

Chinook salmon – **Likely to adversely affect** (indirect effects - availability of prey)

Bull trout – **Likely to adversely affect** (indirect effects - availability of prey)

Marbled murrelet – **Likely to adversely affect** (indirect effects - availability of prey)

5.5.1.10 Lead – Freshwater

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = hardness dependent (30 µg/L dissolved lead at 50 mg/L hardness)

Chronic (CCC) criterion = hardness dependent (1.2 µg/L dissolved lead at 50 mg/L hardness)

CAS ID 7439-92-1

For the purposes of this lead biological evaluation, we have normalized all freshwater effect concentrations to a water hardness of 50 mg/L $CaCO_3$, as was done in the USEPA (1985d) lead criteria document. Because nearly all freshwater lead toxicity literature studies have been run at different hardness values and the freshwater lead criteria values are hardness dependent, the hardness normalization is done so that lead toxicity results are comparable to each other, and can be compared to a single criterion concentration.

5.5.1.10.1 Water-column exposure toxicity data - acute criterion for lead in freshwater

The hardness-based formulas that constitute the EPA recommendations for acute and chronic freshwater lead criteria have not been changed since the publication of USEPA (1985d), although USEPA (1985c) recommended adoption of criteria for total lead, and EPA now recommends adoption of dissolved metals criteria (USEPA, 1993, 2022). The Swinomish Tribe adopted EPA's recommended criteria for dissolved lead in Table 15 and footnotes j and l of their WQS.

The toxicity data presented in USEPA (1985d) allow for the calculation of 10 genus mean acute values (GMAVs). Six of the GMAVs are for fish, the remaining four GMAVs are for invertebrates. The three most sensitive GMAVs were all for invertebrate species, ranging between 142.6 – 1040 µg/L. The most sensitive GMAV for a fish species was 2448 µg/L for rainbow trout (John P Goettl, Davies, & Sinley, 1972), one of the ESA listed species (i.e., steelhead) within the action area. There is also a GMAV of 4820 µg/L for brook trout (Holcombe et al., 1976), the same genus as the ESA listed bull trout within the action area.

5.5.1.10.2 Additional freshwater acute toxicity data for lead published since 1985

Vardy, Santore, Ryan, Giesy, and Hecker (2014) evaluated the sensitivity of white sturgeon (*Acipenser transmontanus*) embryos to four metals, including lead. 96 hour LC_{50} tests were initiated with embryos that were eight days post hatch old. The lead 96 hour LC_{50} was 177 µg/L at a hardness of 59 mg/L as $CaCO_3$. Normalized to a hardness of 50 mg/L, the 96 hour LC_{50} was 143 µg/L.

C.A. Mebane, Dillon, and Hennessy (2012) evaluated toxicity of three metals, including lead to multiple fish and invertebrate species present in the Coeur d'Alene River watershed. Fish species tested with lead were rainbow trout, westslope cutthroat trout (*Oncorhynchus clarki lewisi*) and shorthead sculpin (*Cottus confusus*). Multiple 96 hour EC_{50} studies were performed with different water sources of different hardnesses, with survival as the endpoint. The lowest 96 hour EC_{50} results for each species were 127 µg/L for rainbow trout at a hardness of 32 mg/L $CaCO_3$, 47 µg/L for westslope cutthroat trout at a hardness of 11 mg/L $CaCO_3$, and >855 µg/L for shorthead sculpin at a hardness of 21 mg/L $CaCO_3$. Normalized to a hardness of 50 mg/L $CaCO_3$, the 96 hour EC_{50}s are 224, 323 and 2580 µg/L for rainbow trout, westslope cutthroat trout and shorthead sculpin, respectively. The rainbow trout 96 hour EC_{50} from C.A. Mebane et al. (2012) is the lowest available rainbow trout acute toxicity value for lead, and is used in the remainder of this freshwater lead acute effects assessment.

5.5.1.10.3 Effects assessment of acute freshwater lead criterion on listed species

The only ESA listed species within the action area with empirical acute toxicity data for lead is rainbow trout (steelhead). The lowest empirical 96 hour survival EC_{50} for rainbow trout is 224 µg/L from C.A. Mebane et al. (2012). Dividing this 96 hour EC_{50} by 2.27 to estimate an LC_{low} results in an acute effects assessment concentration of 99 µg/L. The rainbow trout acute effects concentration is higher than the acute lead criterion of 30 µg/L. This indicates that exposure at the level of the freshwater acute lead criterion is not likely to adversely affect rainbow trout.

Without empirical toxicity data for the remaining ESA listed species, the acute effects assessment concentrations for Chinook salmon, chum salmon and bull trout will be calculated from Interspecies Correlation Estimation (ICE) models. The rainbow trout 96 hour EC_{50} of 224 µg/L from C.A. Mebane et al. (2012) will be used to calculate effects assessment concentrations for Chinook and chum salmon, while the brook trout 96 hour LC_{50} of 4820 µg/L from Holcombe et al. (1976) will be used to calculate the effects assessment concentration for bull trout.

The ICE estimated 96 hour LC_{50}s for Chinook and chum salmon, and bull trout are shown in Table 93. Dividing the estimated 96 hour LC_{50}s by 2.27 yields the acute effects assessment concentrations for the ESA listed species without empirical acute lead toxicity data as follows:

Chinook salmon – 137 µg/L

Chum salmon – 117 µg/L

Bull trout – 868 µg/L

All of the calculated acute effects assessment concentrations for the ESA listed fish species without empirical toxicity data are higher than the freshwater acute lead criterion of 30 µg/L. This indicates that exposure at the level of the freshwater acute lead criterion is not likely to adversely affect Chinook salmon, chum salmon or bull trout.

Table 93. Output from Interspecies Correlation Estimation models to estimate acute lead toxicity to Chinook salmon, chum salmon and bull trout

Predicted Taxa	Chinook salmon (*Oncorhynchus tshawytscha*)	Oncorhynchus (predictor for Chum salmon)	Salvelinus (predictor for Bull trout *S. confluentus*)
Model Level	species	genus	genus
Surrogate	Rainbow trout (*Oncorhynchus mykiss*)	Rainbow trout (*Oncorhynchus mykiss*)	Brook trout (*Salvelinus fontinalis*)
Estimated Toxicity (mg/L)	309.71	265.37	1970.55
95% Confidence Intervals	218.80 – 438.40	226.07 – 311.50	1153.07 – 3367.61
df (N-2)	16	60	7
R^2	0.96	0.96	0.88
p-value	0	0	0.00002
Mean Square Error (MSE)	0.074	0.077	0.054
Cross-validation success (%)	94	100	100
Tax. Dist.	1	1	1
Slope	0.94	0.94	0.82
Intercept	0.27	0.21	0.26

5.5.1.10.4 Water-column exposure toxicity data - chronic criterion for lead in freshwater

The chronic freshwater toxicity data in USEPA (1985d) includes results of several studies with rainbow trout and one study with brook trout. There are no empirical chronic toxicity data for fish in USEPA (1985d) of the duration required for direct calculation of a chronic NOEC or MATC under the data requirements of Stephan et al. (1985). There are several chronic toxicity studies discussed in USEPA (1985d) with rainbow trout (Davies, 1976; John P Goettl et al., 1972; Sauter, 1976), and one with brook trout (G.W. Holcombe, Benoit, Leonard, & McKim, 1976). These studies were of between 2 and 19 months duration. In all of them the chronic endpoint measured were spinal deformities, with black spots on skin also noted in several studies. None of these studies quantified chronic effects on survival, reproduction or growth.

5.5.1.10.5 Additional freshwater chronic toxicity data for lead published since 1985

We have identified no chronic toxicity studies describing chronic effects of lead on fish published since 1985.

5.5.1.10.6 Effects assessment of chronic freshwater lead criterion on listed species

In the absence of suitable empirical chronic toxicity data for lead with fish, the 96 hour LC_{50} acute toxicity concentrations for rainbow trout (224 µg/L) and brook trout (4820 µg/L) are divided by the lead acute-chronic ratio (ACR) to obtain an estimate of chronic no effect concentrations for the ESA listed species in the genus *Oncorhynchus* (rainbow trout, Chinook salmon, chum salmon) and the genus *Salvelinus* (bull trout). USEPA (1985d) uses an ACR of 51.29 to calculate the final chronic value, currently 1.2 µg/L from the final acute value of 67.54 µg/L in USEPA (1985d). As was done in the acute lead criterion assessment, all toxicity data have been normalized to a hardness of 50 mg/L $CaCO_3$ in this chronic effects assessment.

Dividing the lowest available rainbow trout 96 hour LC_{50} by the ACR results in a chronic effects assessment concentration of 4.4 µg/L for all members of the genus *Oncorhynchus*. This chronic effects assessment concentration of 4.4 µg/L is higher than the freshwater chronic lead criterion of 1.2 µg/L, and leads to a conclusion that exposure at the level of the freshwater chronic lead criterion is not likely to adversely affect rainbow trout (steelhead), Chinook salmon or chum salmon.

Dividing the lowest available brook trout 96 hour LC_{50} by the ACR results in a chronic effects assessment concentration of 94 µg/L for all members of the genus *Salvelinus*. This chronic effects assessment concentration of 94 µg/L is higher than the freshwater chronic lead criterion of 1.2 µg/L, and leads to a conclusion that exposure at the level of the freshwater chronic lead criterion is not likely to adversely affect bull trout.

The chronic effects assessment concentrations for the ESA listed species without empirical chronic lead toxicity data as follows:

Rainbow trout (steelhead) – 4.4 µg/L

Chinook salmon – 4.4 µg/L

Chum salmon – 4.4 µg/L

Bull trout – 94 µg/L

5.5.1.10.7 Summary of freshwater effect determinations for lead

EPA has made the following determinations on the acute and chronic lead criteria for the ESA listed species in the freshwater portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.5.1.11 Malathion – Freshwater

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = None

Chronic (CCC) criterion = 0.1 µg/L

CAS ID 121-75-5

5.5.1.11.1 Water-column exposure toxicity data - chronic criterion for malathion in freshwater

Malathion is a non-systemic organophosphate insecticide and acaricide used to control a wide range of pests in a variety of food crops and fruits. Malathion acts via contact, ingestion and respiratory exposure pathways. Acute and chronic toxicity of malathion derived from studies submitted by pesticide registrants along with open literature data are used to evaluate the effects of the insecticide on aquatic and terrestrial organisms. Malathion is classified as highly toxic to all taxonomic groups of aquatic animals including fish, amphibians and invertebrates (USEPA, 2009b). EPA has completed an endangered species assessment of the potential effects of malathion on 26 listed salmonids as a result of the FIFRA regulatory actions from the use of the pesticide. Long-term toxicity of malathion to two other fish species, Colorado squaw fish (*Ptychocheilus lucius*) and bonytail (*Gila elegans)* have been reported (Beyers et al., 1994). The Colorado squawfish and bonytail are large minnows that have historically

occurred throughout the Colorado River Basin (Behnke & Benson, 1980). The populations of both species have declined as a result of construction of reservoirs and other management practices in the basin (Beyers et al., 1994; C. A. Carlson & Muth, 1989). Beyers et al. (1994) reported a 32-day chronic LOEC of 3510 µg/L and chronic NOEC of 1680 µg/L for growth for Colorado squaw fish and a 32-day chronic LOEC of 2000 µg/L and chronic NOEC of 990 µg/L, for survival for the bonytail. Eaton (Beyers et al., 1994; J. G. Eaton, 1970), also reported a chronic LOEC of 7.4 pg/L for adult Colorado squawfish based on development of spinal deformities.

Other studies such as Mount and Stephan (1967) and Eaton (J. G. Eaton, 1970) reported chronic effects of malathion to bluegill (*Lepomis macrochirus*) and fathead minnows (*Pimephales promelas*) based on spinal deformities. Hermanutz (1978) reported a 30-day chronic NOEC of 8.6 µg/L for malathion on flagfish (*Jordanella* floridae) in freshwater. The chronic effects of malathion to flagfish were based on reduced growth and survival in the first generation during the first 30 days. The studies didn't find any evidence of deformities in flagfish at the chronic concentrations tested. Also, it was reported in the study that no distinct adverse effects (growth and survival) occurred in the second generation in any of the concentrations tested for flagfish. Therefore, there was no clear explanation for the absence of reduced growth in the second-generation in the higher concentrations tested (Hermanutz, 1978).

Laetz et al. (2009) conducted a 96-hr test on juvenile coho salmon (*Oncorhynchus kisutch)* with measured nominal concentrations of malathion of 89±26% and reported that no mortality was observed in any of the single chemical exposure concentrations tested. Also, there was no statistically significant difference in AChE activity between the unexposed fish and the solvent control fish. Therefore, AChE in the treatment groups were expressed as a percentage of the solvent control. The 96-hr EC50 for malathion from the test is the inverse of LogEC50 of 1.9, which is equal to 79.4 µg/L (Laetz et al., 2009).

5.5.1.11.2 Effects assessment of chronic freshwater malathion criterion on listed species

The lowest 32-day chronic effect concentration (NOEC) for fish based on the ECOTOX database is the study by Beyers et al. (1994) with the Colorado squawfish ***(****Ptychocheilus lucius****)*** and bonytail *(Gila elegans)*. The study reported a lowest chronic NOEC of 1680 µg/L for Colorado squawfish and 990 µg/L for bonytail.

The NOECs reported in the Beyers et al. (1994) study are higher than the proposed malathion freshwater chronic criterion of 0.1 µg/L. If the minimum chronic effect concentration for the remaining listed salmonids in the freshwater portions of the action area is greater than or equal to 990 µg/L (the lower of the two NOECs), exposure at the level of the proposed malathion freshwater chronic criterion of 0.1 µg/L is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, and bull trout.

5.5.1.11.3 Summary of freshwater effect determinations for malathion

EPA has made the following determinations on the chronic malathion criterion for the listed species in the freshwater portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.5.1.12 Nickel – Freshwater

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = hardness dependent (470 µg/L dissolved nickel at 100 mg/L hardness)

Chronic (CCC) criterion = hardness dependent (52 µg/L dissolved nickel at 100 mg/L hardness)

CAS ID 7440-0-20

In this biological evaluation, all freshwater effect concentrations for nickel have been normalized to a water hardness of 100 mg/L $CaCO_3$, as was done in the EPA (USEPA, 1986e). The relevant freshwater nickel toxicity studies were run at different hardness values, and hardness was normalized to facilitate comparison of toxicity results.

5.5.1.12.1 Water-column exposure toxicity data - acute criterion for nickel in freshwater

The hardness-based formulas that constitute the EPA recommendations for acute and chronic freshwater nickel criteria have not been changed since the publication of EPA (USEPA, 1986e), although EPA (C.E. Stephan et al., 1985) recommended adoption of criteria for total metals, and EPA now recommends adoption of dissolved metals criteria (USEPA, 1993, 2022). The Swinomish Tribe adopted EPA's recommended criteria for dissolved lead in Table 15 and footnotes j and l of their WQS.

EPA (USEPA, 1986e) presented toxicity data that allowed for the calculation of 18 genus mean acute values (GMAVs). Twelve of the GMAVs are for fish, the remaining six GMAVs are for invertebrates. The three most sensitive GMAVs were all for invertebrate species, ranging between 1102 – 40460 µg/L. The most sensitive GMAV for a fish species was 4312 µg/L for rock bass (Lind et al. Manuscript), which is not an ESA listed species within the action area. There is also a GMAV of 13380 µg/L for rainbow trout (Nebeker, Savonen, & Stevens, 1985), an ESA listed species within the action area.

5.5.1.12.2 Additional freshwater acute toxicity data for nickel published since 1986

Brix et al. (Brix, Keithly, DeForest, & Laughlin, 2004) evaluated the sensitivity of rainbow trout (*Oncorhynchus mykiss*) embryos to nickel. The 96-hour LC_{50} test was initiated with embryos that were 18-day-old post-swim-ups. The nickel 96-hour LC_{50} was 20.8 mg/L at a hardness of 91 mg/L as $CaCO_3$. Normalized to a hardness of 100 mg/L, the 96-hour LC_{50} was 20.8 mg/L.

Svecevičius (2010) evaluated toxicity of nickel to five fish species that included the following: rainbow trout (*Oncorhynchus mykiss*), three-spined stickleback (*Gasterosteus aculeatus*), roach (*Rutilus rutilus*), perch (*Perca fluviatilis*) and dace (*Leuciscus leuciscus*). The test species and their corresponding 96-hour LC_{50s} are listed in Table 94 (below).

*Table 94. Calculated LC50s of the Five Fish Species Tested**

Fish Species Tested	96-hr LC_{50} (mg Ni/L)	95 % Confidence Interval (mg Ni/L)
Rainbow trout (*Oncorhynchus mykiss*)	19.3	15.0 - 24.9
Three-spined stickleback (*Gasterosteus aculeatus*)	33.7	31.3 - 36.3
Roach (*Rutilus rutilus*)	48.7	43.3 - 54.9
Perch (*Perca fluviatilis*)	48.1	43.4 - 53.2
Dace (*Leuciscus leuciscus*).	61.2	56.1 - 66.8

*Hardness = 284 (271- 296) mg/L as $CaCO_3$

The lowest 96-hour LC_{50} of the test results was 19.3 mg/L for rainbow trout at a hardness of 284 mg/L $CaCO_3$. Normalized to a hardness of 100 mg/L $CaCO_3$, the 96 hour LC_{50}s are 19.3, 33.7, 48.7, 48.1 and 61.2 mg/L, for rainbow trout, three-spined stickleback, roach, perch and dace, respectively. The rainbow trout 96-hour LC_{50} from this study is the lowest available rainbow trout acute toxicity value for nickel, and is used in the remainder of this freshwater nickel acute effects assessment.

5.5.1.12.3 Effects assessment of acute freshwater nickel criterion on listed species

The only ESA listed species within the action area with empirical acute toxicity data for nickel is rainbow trout (steelhead). The lowest empirical 96-hour survival LC_{50} for rainbow trout is 19.3 mg/L from Svecevičius (2010). Dividing this 96-hour LC_{50} by 2.27 to estimate an LC_{low} results in an acute effects assessment concentration of 8.5 mg/L (19.3/2.27). The rainbow trout acute effects concentration (8.5 mg/L) is higher than the acute nickel criterion of 468.2 µg/L. This indicates that exposure at the level of the freshwater acute nickel criterion is not likely to adversely affect rainbow trout.

Without empirical toxicity data for the remaining ESA listed species, the acute effects assessment concentrations for Chinook salmon, chum salmon and bull trout will be calculated from Interspecies Correlation Estimation (ICE) models. The rainbow trout 96-hour LC_{50} of 19.3 mg/L from Svecevičius (2010) will be used to calculate effects assessment concentrations for Chinook, chum salmon and bull trout.

The ICE estimated 96-hour LC_{50}s for Chinook and chum salmon, and bull trout are shown in Table 95. Dividing the estimated 96-hour LC_{50}s by 2.27 yields the acute effects assessment concentrations for the ESA listed species without empirical acute lead toxicity data as follows:

Chinook salmon – 15.81 mg/L

Chum salmon – 20.16 mg/L

Bull trout – 17.33 mg/L

All of the calculated acute effects assessment concentrations for the ESA listed fish species without empirical toxicity data are higher than the freshwater acute nickel criterion of 260.5 µg/L. This indicates that exposure at the level of the freshwater acute nickel criterion is not likely to adversely affect Chinook salmon, chum salmon or bull trout.

Table 95. Output from Interspecies Correlation Estimation models to estimate acute Nickel toxicity to Chinook salmon, chum salmon and bull trout

Predicted Taxa	Chinook salmon (*Oncorhynchus tshawytscha*)	Oncorhynchus (used for chum salmon)	Salvelinus (used for bull trout)
Model Level	species	genus	Genus
Surrogate	Rainbow trout (*Oncorhynchus mykiss*)	Rainbow trout (*Oncorhynchus mykiss*)	Brook trout (*Salvelinus fontinalis*)
Estimated Toxicity (mg/L)	15.81	20.15	17.33
95% Confidence Intervals	9.94 - 25.16	16.63 - 24.41	7.34 – 40.92
Df (N-2)	16	60	7
R^2	0.96	0.96	0.88
p-value	0	0	0.00002
Mean Square Error (MSE)	0.074	0.077	0.054
Cross-validation success (%)	94	100	100
Tax. Dist.	1	1	1
Slope	0.94	0.94	0.82
Intercept	0.27	0.21	0.26

5.5.1.12.4 Water-column exposure toxicity data - chronic criterion for nickel in freshwater

Freshwater chronic toxicity data in EPA (USEPA, 1986e) include results of one study with rainbow trout. There are two other studies with fathead minnows, and these were either life cycle or partial life cycle and early life stage studies. There are no empirical chronic toxicity data for fish in EPA (USEPA, 1986e) of the duration required for direct calculation of a chronic NOEC or MATC under the data requirements of Stephan et al. (1985). There is only one chronic toxicity study discussed in EPA (USEPA, 1986e)with rainbow trout (Nebeker et al., 1985). A 38-day study was conducted with pre-swim-up larvae and an LC_{50} values of 1.4 mg/L was calculated and was compared to the 96-hr values. It was discovered that there were no effects of nickel on swim-up fish at any concentration (Nebeker et al., 1985). The chronic endpoint measured was mortality of swim-up fish exposed in this study. None of the studies mentioned above quantified chronic effects on survival, reproduction or growth.

5.5.1.12.5 Additional freshwater chronic toxicity data for nickel published since 1985

Brix et al. (2004) conducted a chronic toxicity study (an 85-day test) with rainbow trout and reported that measurements on day 28 of the test showed whole-egg nickel accumulated in an exposure dependent manner, with the highest concentration in chorion. In addition, Na, Ca and Mg in whole-egg, embryo and chorion showed consistent trends, indicating no ionoregulatory disruption. Thus, no significant effect on the embryo swim-up, hatching or fingerling survival or growth were observed at dissolved nickel concentrations up to 466 µg/L, the highest concentration tested. This nickel concentration is considerably higher than the only other reported NOEC (<33 µg/L) for rainbow trout (Giattina, Garton, & Stevens, 1982; Nebeker et al., 1985).

5.5.1.12.6 Effects assessment of chronic freshwater nickel criterion on listed species

Based on the Brix et al. (2004) study and in the absence of any other chronic toxicity data for nickel with fish, the NOEC of 466 µg/L (at a hardness of 89 mg/L $CaCO_3$) from the chronic toxicity study with rainbow trout will be used to estimate chronic effect concentrations for the ESA listed species in the

genus *Oncorhynchus* (rainbow trout, Chinook salmon, chum salmon) and the genus *Salvelinus* (bull trout). As was done in the acute nickel criterion assessment, all toxicity data have been normalized to a hardness of 100 mg/L $CaCO_3$ in this chronic effects assessment. The hardness adjusted NOEC is 466 µg/L at a hardness of 100 mg/L $CaCO_3$.

This chronic effects assessment concentration of 466 µg/L is higher than the freshwater chronic nickel criterion of 28.9 µg/L, and leads to a conclusion that exposure at the level of the freshwater chronic nickel criterion is not likely to adversely affect rainbow trout (steelhead), Chinook salmon or chum salmon.

Since there is no empirical chronic toxicity data for nickel available for brook trout (surrogate for bull trout), this chronic effects assessment concentration (NOEC) of 466 µg/L is higher than the freshwater chronic nickel criterion of 28.9 µg/L, and leads to a conclusion that exposure at the level of the freshwater chronic nickel criterion is not likely to adversely affect bull trout.

The chronic effects assessment concentrations for the ESA listed are as follows:

Rainbow trout (steelhead) – 466 µg/L

Chinook salmon – 466 µg/L

Chum salmon – 466 µg/L

Bull trout – 466 µg/L

5.5.1.12.7 Summary of freshwater effect determinations for nickel

EPA has made the following determinations on the acute and chronic nickel criteria for the ESA listed species in the freshwater portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Likely to adversely affect** (indirect effects - availability of prey)

Chinook salmon – **Likely to adversely affect** (indirect effects - availability of prey)

Bull trout – **Likely to adversely affect** (indirect effects - availability of prey)

Marbled murrelet – **Likely to adversely affect** (indirect effects - availability of prey)

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.5.1.13 Nonylphenol – Freshwater

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion – 28 µg/L

Chronic (CCC) criterion – 6.6 µg/L

CAS ID – 25154-52-3 (general for all nonylphenols)

84852-15-3 (branched 4-nonylphenols)

104-40-5 (4-n-nonylphenol)

5.5.1.13.1 Introduction to nonylphenols

Nonylphenol is not a single chemical structure. Nonylphenol refers to a complex mixture of nonylphenol congeners. The nine carbon nonyl group may be branched or linear and bind at various locations around the phenol ring. Most commonly the nonyl group is substituted at the para- or 4- position, but small amounts of ortho- (2-) and meta- (3-) substituted nonylphenols also exist in commercially produced nonylphenol. Many manufacturers incorrectly use the straight chain 4-nonylphenol identification (CAS ID 104-40-5) when referring to the branched nonylphenols. Straight chain 4-nonylphenol is commonly used in aquatic toxicity testing. CAS ID 84852-15-3 corresponds to the most widely produced commercially nonylphenol, a mixture of branched 4-nonylphenols. CAS ID 25154-52-3 has also been used to describe mixtures of commercially produced nonylphenol. The EPA (USEPA, 2005a) nonylphenol criteria document states that it addresses the CAS numbers 84852-15-3 and 25154-52-3, two of the CAS ID numbers used in this BE to perform ECOTOX database searches for toxicity data. In the ECOTOX database search for nonylphenol, we have also searched for straight chain 4-nonylphenol (CAS ID 104-40-5) due to its use in toxicity testing, and because it can be a congener present in commercially produced nonylphenol mixtures.

5.5.1.13.2 Water-column exposure toxicity data - acute criterion for nonylphenol in freshwater

Among the toxicity studies used to derive the EPA (USEPA, 2005a) nonylphenol freshwater acute criterion were multiple studies with salmonids. Included among the salmonid studies were two (L. T. Brooke, 1993; F James Dwyer, Sappington, Buckler, & Jones, 1995) with rainbow trout (*Oncorhynchus mykiss*), one of the listed species within the action area. Individual 96-hour LC_{50}s for rainbow trout in the Dwyer et al. (F James Dwyer et al., 1995) study ranged between 140 – 270 µg/L. Static exposure conditions were used in the F James Dwyer et al. (1995) study. The flow through study with rainbow trout performed by L. Brooke (1993) observed a 96-hour LC_{50} of 221 µg/L, the value used as the GMAV during acute criterion derivation in EPA (USEPA, 2005a). The rainbow trout 96-hour LC_{50} from L. Brooke (1993) was used by EPA as the SMAV in criterion derivation because flow through studies are considered to more accurately represent environmental exposures to chemicals in nature than are static or static renewal exposure studies.

Other salmonid species 96-hour LC_{50} values in EPA (USEPA, 2005a) used to derive the freshwater acute nonylphenol criterion include studies with Apache trout (current scientific name *Oncorhynchus apache*) in Dwyer et al. (F James Dwyer et al., 1995), with a SMAV of 170 µg/L. Studies with two subspecies of

cutthroat trout (Lahontan, *Oncorhynchus clarki henshawi* and greenback, *O. clarki stomias*), also performed by Dwyer et al. (F James Dwyer et al., 1995) had similar SMAVs of 180 µg/L and 150 µg/L, respectively.

Additional freshwater nonylphenol 96-hour LC_{50} values for salmonids presented in EPA (USEPA, 2005a), but not included in criteria derivation because the fish were fed during the studies are available for brook trout (*Salvelinus fontinalis*) and Atlantic salmon (*Salmo salar*). Brook trout are in the same genus as are bull trout (*Salvelinus confluentus*), one of the listed species within the action area. According to the Guidelines, fish fed during 96-hour LC_{50} studies are not suitable for use in acute criterion development. This is because fed fish are more tolerant of chemicals (i.e. LC_{50}s are higher) than are unfed fish. Thus, chemical toxicity in unfed fish is greater than is the toxicity to fed fish, which results in a lower (i.e. more protective) LC_{50} for use in criteria derivation. Despite this exclusion, the brook trout study is of interest in assessing the potential acute effects of nonylphenol on the closely related bull trout.

Holmes and Kingsbury (Holmes & Kingsbury, 1980) calculated a 96-hour LC_{50} of 145 µg/L for brook trout which were fed during the 96-hour exposure. Despite that the studies with *Oncorhynchus* species used to derive the EPA (USEPA, 2005a) acute criterion were performed using unfed fish, the fed brook trout 96-hour LC_{50} is comparable to the LC_{50}s observed for the unfed *Oncorhynchus* species.

McLeese et al. (McLeese, Zitko, Metcalfe, & Sergeant, 1980) fed Atlantic salmon during a 96-hour LC_{50} study, and observed a 96-hour LC_{50} of 900 µg/L. This LC_{50} is higher than all 15 of the GMAVs used to derive the freshwater acute nonylphenol criterion in EPA (USEPA, 2005a).

Of all the acute toxicity studies used to derive the freshwater acute nonylphenol criterion in EPA (USEPA, 2005a), the most sensitive species was the amphipod *Hyalella azteca*, with a GMAV of 55.72 µg/L. The most sensitive fish species used in the freshwater acute nonylphenol criterion derivation was the fathead minnow (*Pimephales promelas*), with a GMAV of 133.9 µg/L.

5.5.1.13.3 Additional freshwater acute toxicity data for nonylphenol published since 2005

There is an extensive amount of literature published since 2005 describing 96-hour LC_{50}s to a number of fish species. Spehar et al. (R. L. Spehar, Brooke, Markee, & Kahl, 2010) had the objective of determining the toxicity and bioconcentration of nonylphenol to a variety of freshwater organisms for use in deriving freshwater water quality criteria. Test species representing 10 different taxonomic families that represent a freshwater community were studied in 96-hour LC_{50} determinations. One of the 10 species studied by R. L. Spehar et al. (2010) was alevins of rainbow trout, a listed species present within the action area. The 96-hour LC_{50} for rainbow trout found by R. L. Spehar et al. (2010) was 221 µg/L.

Although R. L. Spehar et al. (2010) is the only 96-hour LC_{50} study performed on a listed species within the action area since the publication of the EPA (USEPA, 2005a) nonylphenol criteria document, several other 96-hour LC_{50} studies with fish have been published. Test results for fountain darter (*Etheostoma rubrum*), greenthroat darter (*Etheostoma lepidum*), razorback sucker (*Xyrauchen texanus*), Gila topminnow (*Poeciliopsis occidentalis*), Colorado pikeminnow (*Ptychocheilus lucius*, called Colorado squawfish in the Dwyer et al. 2005 study) and bonytail chub (*Gila elegans*) presented in Dwyer et al. (F. J. Dwyer et al., 2005) appear to be a peer reviewed presentation of the results used to derive nonylphenol criteria attributed to Dwyer et al. (F James Dwyer et al., 1995) in the EPA nonylphenol criteria document (USEPA, 2005a). Dwyer et al. (F. J. Dwyer et al., 2005) also appears to be a peer reviewed presentation of

the 96-hour LC_{50}s for rainbow trout, Apache trout, Lahontan cutthroat trout and greenback cutthroat trout attributed to Dwyer et al. (F James Dwyer et al., 1995) and used to derive the acute nonylphenol criterion in the EPA (USEPA, 2005a) nonylphenol criteria document.

Dwyer et al. (F. J. Dwyer et al., 2005) also presents 96-hour LC_{50}s for five freshwater fish species not discussed in EPA (USEPA, 2005a). The LC_{50}s for Atlantic sturgeon (*Acipenser oxyrhynchus*), shortnose sturgeon (*Acipenser brevirostrum*), shovelnose sturgeon (*Scaphirhynchus platorynchus*), Cape Fear shiner (*Notropis mekistocholas*) and spotfin chub (*Hybopsis monacha*) were 50, 80, <130, 140 and 80 µg/L, respectively.

The Atlantic sturgeon 96-hour LC_{50} of 50 µg/L is the lowest 96-hour LC_{50} we have identified for any freshwater fish species exposed to nonylphenol. According to F. J. Dwyer et al. (2005), this Atlantic sturgeon LC_{50} should be used with caution, because control survival in the acetone carrier solvent control was only 70%, lower than the ≥ 90% control survival required within the test acceptability criteria of the toxicity test methodology used with Atlantic sturgeon (ASTM, 2003). The acetone carrier solvent was required to maintain nonylphenol in solution during the toxicity test. As such, the Dwyer et al. (F. J. Dwyer et al., 2005) Atlantic sturgeon 96-hour LC_{50} is not considered to be an acceptable toxicity test for use in deriving EPA aquatic life criteria.

Additional 96-hour LC_{50}s for fathead minnow published subsequent to the nonylphenol criteria document (USEPA, 2005a) include studies by Spehar et al. (R. L. Spehar et al., 2010), Mayer et al. (F. L. Mayer, 2008), TenEyck and Markee (TenEyck & Markee, 2007) and Dwyer et al. (F. J. Dwyer et al., 2005). The 96-hour LC_{50}s from these four studies were 128, 272, 136 and 270 µg/L, respectively.

R. L. Spehar et al. (2010) also measured the 96-hour LC_{50} of nonylphenol to bluegill (*Lepomis macrochirus*) to be 209 µg/L. Stengel et al. (Stengel, Zindler, & Braunbeck, 2017) determined zebrafish (*Danio rerio*) to be tolerant of nonylphenol, with a 95-hour LC_{50} of 1500 µg/L.

Mayer et al. (F. L. Mayer, 2008) tested the acute toxicity of nonylphenol to 14 freshwater fish species, two amphibian species and two estuarine fish species. All 14 freshwater fish species tested by F. L. Mayer (2008) have also been tested by one or more other investigators. The most sensitive freshwater fish species tested by F. L. Mayer (2008) was shortnose sturgeon, with a 96-hour LC_{50} of 81 µg/L. The most tolerant freshwater fish tested in F. L. Mayer (2008) was fathead minnow, with a 96-hour LC_{50} of 272 µg/L. Among the freshwater fish species tested by F. L. Mayer (2008) were four salmonid species: rainbow trout, Apache trout, greenback cutthroat trout and Lahontan cutthroat trout, with 96-hour LC_{50} values of 191 µg/L, 161 µg/L, 153 µg/L and 163 µg/L, respectively.

96-hour LC_{50} data for several non-native fish species are also available, which are consistent with the range of 96-hour LC_{50}s described earlier in this section. Bhattacharya et al. (Bhattacharya, Xiao, & Lun, 2008) determined the 96-hour LC_{50} to rosy barb (*Puntius conchonius*), a cyprinid native to India to be 1.72 µM (379 µg/L). Senthil Kumaran et al. (Senthil Kumaran, Kavitha, Ramesh, & Grummt, 2011) found the North African catfish *Clarias gariepinus* to be among the fish species most tolerant of nonylphenol, with a 96-hour LC_{50} of 3480 µg/L.

Finally, we have found three studies in Chinese with extended English abstracts (Huang, Dong, & Ma, 2006; Lei, Tian, Tang, Zhang, & Cang, 2014; Lu, Gu, & Song, 2012) providing 96-hour LC_{50} data on a combined five originally non-native species, all of which have been introduced into North America and

have established reproducing populations. Although none of these studies provide information that is useable in the derivation of aquatic life criteria, they do help to provide a more complete evaluation of the range of nonylphenol concentrations acutely toxic to fish species residing in North American waters.

Huang et al. (Huang et al., 2006) found 24, 48, 72 and 96-hour LC_{50}s of 413.4, 351.5, 311.2 and 264.6 µg/L, respectively to a Nile tilapia x blue tilapia (*Oreochromis niloticus* x *O. aureus*) hybrid. These tilapia species are native to Africa, but the hybrid has been introduced to North America as an aquaculture species.

Lu et al. (Lu et al., 2012) determined 96-hour LC_{50}s for goldfish (*Carassius auratus*), grass carp (*Ctenopharyngodon idellus*) and silver carp (*Hypophthalmichthys molitrix*) of 251.3, 155.8 and 187 µg/L, respectively. All three of these species were originally native to Asia.

Lei et al. (Lei et al., 2014) found a 96-hour LC_{50} of 9740 µg/L for the Oriental weatherfish or pond loach (*Misgurnus anguillicaudatus*), an Asian native introduced into multiple locations within the United States, including Hawaii. This species is the most acutely tolerant to nonylphenol of any freshwater fish species for which we have toxicity information.

5.5.1.13.4 Effects assessment of freshwater acute nonylphenol criterion on listed species

The only listed species within the freshwater portions of the action area with empirical toxicity data for nonylphenol is rainbow trout (steelhead). The preferred rainbow trout study used by EPA (USEPA, 2005a) in the acute nonylphenol criterion derivation was the flow through study of Brooke L. T. Brooke (1993), with a 96-hour LC_{50} of 221 µg/L. This same 96-hour LC_{50} of 221 µg/L was also reported in Spehar et al. (R. L. Spehar et al., 2010). Individual 96-hour LC_{50}s for rainbow trout in Dwyer et al. (F James Dwyer et al., 1995), performed under static exposure conditions ranged between 140 – 270 µg/L. Dividing either the flow through 96-hour LC_{50} of 221 µg/L or the lowest static exposure 96-hour LC_{50} of 140 µg/L by 2.27 to convert these LC_{50} values to the lowest LC_{LOW} or minimum acute effect concentration for rainbow trout yields threshold acute effect concentrations of 97 µg/L and 62 µg/L, respectively. These concentrations are both higher than the nonylphenol freshwater acute criterion of 28 µg/L. Regardless of the empirical rainbow trout study used to calculate the minimum acute effect concentration, our conclusion is that exposure at the level of the freshwater acute nonylphenol criterion is not likely to adversely affect rainbow trout (steelhead).

If it is assumed that the remaining listed salmonid species in the freshwater portions of the action area have a minimum acute effect concentration greater than or equal to the 62 µg/L value for rainbow trout, exposure at the level of the nonylphenol freshwater acute criterion of 28 µg/L is not likely to adversely affect Chinook salmon, chum salmon, and bull trout. If one were to use the 96-hour LC_{50} of the freshwater fish species with the lowest empirically measured acute value, the shovelnose sturgeon 96-hour LC_{50} of 80 µg/L, the resulting minimum acute effect concentration (80/2.27 = 35 µg/L) is still greater than the acute criterion of 28 µg/L, which supports the conclusion that exposure at the level of the freshwater acute nonylphenol criterion of 28 µg/L is not likely to adversely affect these salmonids.

5.5.1.13.5 Water-column exposure toxicity data - chronic criterion for nonylphenol in freshwater

Chronic toxicity data for five species (two fish, three invertebrates) is reported in the EPA (USEPA, 2005a) nonylphenol criteria document. The lowest chronic value among the five species is the 7.861 µg/L value from Brooke (L. T. Brooke, 1993) for rainbow trout (steelhead), one of the ESA listed species

found within the action area of the Swinomish Tribe. This value is a 91 day exposure MATC for growth of rainbow trout, longer than the 28 – 32 day chronic test duration (described as a 30 day chronic test duration throughout this text for brevity) with early life stage fish suggested in the Guidelines for use in developing chronic water quality criteria. The chronic MATC of 7.861 µg/L was calculated as the geometric mean of the 91-day growth NOEC (6.0 µg/L) and the 91-day growth LOEC (10.3 µg/L).

The only other fish species with a chronic value in EPA (USEPA, 2005a) is fathead minnow. Ward and Boeri (T. J. Ward & R. L. Boeri, 1991) performed a 33 day study on embryos and larvae of fathead minnows, with hatchability, growth and survival as the monitored endpoints. Hatchability was unaffected by nonylphenol concentrations up to 7.4 µg/L, but was delayed by one day at 14 µg/L. Length and weight of larvae was unaffected at the end of the 33 day test at all five test concentrations, which ranged between 2.8 – 23 µg/L. Larval survival at the end of the 33 day exposure was reduced at exposure to 14 µg/L relative to control survival, but was not reduced relative to controls at 7.4 µg/L. The fathead minnow chronic value of 10.18 µg/L derived from Ward and Boeri (T. J. Ward & R. L. Boeri, 1991) and used in EPA (USEPA, 2005a) is based on the 33 day chronic survival portion of their study. It is the MATC of the chronic survival NOEC of 7.4 µg/L and the chronic survival LOEC of 14 µg/L.

The remaining three chronic studies used to derive the chronic freshwater nonylphenol criterion in EPA (USEPA, 2005a) are from the invertebrates *Ceriodaphnia dubia*, *Daphnia magna* and *Chironomus dilutus* (formerly *C. tentans*). Final nonylphenol chronic values for these three freshwater invertebrate species were 133.9 µg/L for *C. dubia* (D. E. England, 1995), 47.81 µg/L for *D. magna* (geometric mean of three chronic studies from Fliedner 1993, Brooke 1993 and (L. T. Brooke, 1993; Comber, Williams, & Stewart, 1993; Fliedner, 1993) and 61.82 µg/L for *C. dilutus* (Kahl, Makynen, Kosian, & Ankley, 1997).

5.5.1.13.6 Additional freshwater chronic toxicity data for nonylphenol published since 2005

A number of chronic toxicity studies with freshwater fish have been performed since the EPA (USEPA, 2005a) nonylphenol criteria were published. Information on these studies was identified through a search of the EPA ECOTOX database, as well as information in literature reviews by the National Marine Fisheries Service (NMFS, 2018), Zhang et al. (Zhang, Wei, Zhang, & Song, 2017) and Monson (Monson, 2010). The original studies identified in this review were then individually reviewed by EPA Region 10 staff for suitability for use in deriving EPA aquatic life criteria. Studies from 2004 are also discussed in this section, since they may not have been known to EPA at the time the nonylphenol criteria document was published in 2005.

The most germane of the new chronic nonylphenol toxicity studies to this biological evaluation is that of Hanson et al. (A. M. Hanson, Kittilson, Martin, & Sheridan, 2014), who monitored the feeding behavior, growth, biochemical growth factors and gene expression of rainbow trout exposed to 4-n-nonylphenol for 28 days. The only exposure concentrations tested were a control and 100 µg/L nonylphenol. Exposure had no effect on rainbow trout feeding, but did reduce growth over the 28 day exposure period. A. M. Hanson et al. (2014) is not suitable for water quality criteria derivation, because only one pollutant exposure concentration was tested, not the minimum of three pollutant exposures (plus a control) needed to provide a measure of the concentration-response relationship needed to derive a water quality criterion.

An earlier study by Hanson et al. (Andrea M Hanson, Jeffrey D Kittilson, Stephen D McCormick, & Mark A Sheridan, 2012) exposed rainbow trout to two concentrations of 4-nonylphenol (10 and 100 µg/L) in freshwater for 28 days. Hanson et al. (Andrea M Hanson et al., 2012) found a statistically significant

reduction in body weight at the end of the study for juvenile trout exposed to 100 µg/L, but not at 10 µg/L nonylphenol. However, the main objective of Hanson et al. (Andrea M Hanson et al., 2012) was to determine if nonylphenol exposure in freshwater inhibits the ability of outmigrating juvenile rainbow trout (steelhead) to adapt to seawater.

The process by which younger salmon hatched in freshwater adapt to life in marine systems is called smoltification. In preparation for outmigration from streams to marine systems, juvenile salmon residing in freshwater transform into smolts prior to and during the transition from life in freshwater to life in marine systems. The transformation of smolts into individuals that can survive in salt water includes changes in the biochemical, morphological, physiological and behavioral phenotypes that, once transformation is complete, can survive and grow in marine systems.

The experimental design of Hanson et al. (Andrea M Hanson et al., 2012) exposed rainbow trout to nonylphenol in freshwater for 28 days, then transferred the exposed fish to seawater with a salinity of 20‰. Results of this study indicated that activation of GH-IGF (growth hormone – insulin-like growth factor) system components, including increased expression of GHR 1, GHR 2 (growth hormone receptors 1 and 2), IGF-1, IGF-2, IGFR1A, and IGFR1B (insulin-like growth factor mRNA 1A and 1B), accompany salinity adaptation of rainbow trout. The results also indicate that exposure of rainbow trout to 10 µg/L nonylphenol suppresses the expression of multiple GH-IGF system components and leads to reduced growth, reduced hypo osmoregulatory ability, and a reduced ability of salmonid smolts to successfully transition from fresh water to seawater during smoltification.

Several other studies published since release of the EPA (USEPA, 2005a) nonylphenol criteria document also evaluated nonylphenol effects on salmonid smoltification, primarily Atlantic salmon, using experimental designs similar to that of Andrea M Hanson et al. (2012). Like Andrea M Hanson et al. (2012) these studies exposed salmonid smolts to nonylphenol in freshwater for an extended period, then switched the smolts gradually into salt water, where the effects of nonylphenol were recorded. Although the adverse nonylphenol effects were observed in salt water, the nonylphenol exposure was in freshwater. Based on the freshwater exposure to nonylphenol, we evaluate these studies in our assessment of the freshwater chronic nonylphenol criterion. This group of studies have also confirmed that the estrogenic effect of nonylphenol on smoltification is associated with alterations to one or more components of the GH-IGF system or products of endocrine receptor (ER) associated gene expression, such as ERα mRNA or Vtg (vitellogenin, a protein that is a precursor to egg yolk formation). In addition to the (Andrea M Hanson et al., 2012) study with rainbow trout, Luo et al. (2005) observed that these same biochemical pathways may adversely affect smoltification in sockeye salmon (*Oncorhynchus nerka*), another salmonid.

Although many of these studies provide information on water column concentrations which can adversely affect smoltification and subsequent fish health once in salt water, their experimental designs preclude their use in deriving EPA aquatic life criteria. This is because they do not meet one or more of the requirements in the Guidelines EPA uses to identify studies with acceptable data quality for criteria development. Limitations include chronic exposure to nonylphenol durations shorter than the required 30 day duration (Arsenault et al., 2004; Breves, Duffy, Einarsdottir, Bjornsson, & McCormick, 2018; Duffy, Iwanowicz, & McCormick, 2014; Ishibashi et al., 2006; Lerner, Bjornsson, & McCormick, 2007; Luo et al., 2005; McCormick, O'Dea M, Moeckel, Lerner, & Bjornsson, 2005), fewer than three exposure concentrations (Arsenault et al., 2004; Breves et al., 2018; Lerner et al., 2007; Luo et al., 2005; Madsen,

Skovbolling, Nielsen, & Korsgaard, 2004), and pollutant exposure routes that do not represent exposure to nonylphenol dissolved in the water column, such as dietary (Keen, Higgs, Hall, & Ikonomou, 2005) or intraperitoneal injection (IP injection (Luo et al., 2005; Madsen et al., 2004; McCormick et al., 2005)) exposures. Because of the importance of the smoltification process in the life history of many salmonid species, findings of these studies are summarized in the next several paragraphs, despite being unusable in water quality criteria development according to the Guidelines.

Breves et al. (2018) exposed Atlantic salmon fry in freshwater to two nonylphenol concentrations (8.8 and 44 µg/L) for 21 days. Weight of the fry was significantly reduced at 8.8 µg/L relative to control weights. Fry length was unaffected at 8.8 µg/L, but was significantly reduced relative to the length of controls at 44 µg/L. Fry survival was unaffected by either exposure concentration.

Breves et al. (2018) also exposed smolts in freshwater to three concentrations (0.88, 8.8 and 88 µg/L) of nonylphenol for four days, and monitored survival, growth, ionoregulation and gene expression of multiple genes within the insulin-like growth factor (IGF) system. No effects were observed on smolt survival or growth in this portion of the study. Gene expression of multiple insulin-like growth factors was inhibited at 8.8 µg/L. Despite inhibition of gene expression in the IGF system known to be a precursor to smolt adaptation to sea water, Breves et al. (2018) observed no effect on the osmoregulatory processes required for smolts to adapt to sea water.

Duffy et al. (2014) exposed Atlantic salmon smolts to three nonylphenol concentrations (0.88, 8.8 and 88 µg/L) for four days, then exposed to seawater. They observed no effect on survival at any nonylphenol concentration, and no effect on growth expressed as change in hepatosomatic index (liver weight / whole body weight) x 100.

Lerner et al. (2007) exposed yolk sac fry of Atlantic salmon to mean measured concentrations of 6.5 and 79.9 µg/L nonylphenol for 21 days, then monitored the response of the fry one year later after they had matured into smolts in nonylphenol-free water. Fry exposed to the 79.9 µg/L concentration exhibited nearly 50 % mortality during the 21-day nonylphenol exposure, with delayed mortality post-exposure resulting in 100% mortality within 120 days after termination of the 79.9 µg/L nonylphenol exposure. Fry exposed to 6.5 µg/L nonylphenol suffered no mortality during the initial 21-day exposure, but showed delayed mortality of roughly 30% in the 60-day period after their transfer from 6.5 µg/L nonylphenol into nonylphenol-free water. During their transition into nonylphenol-free salt water one year later, the surviving smolts exhibited reduced development, a reduced preference for salt water, and took longer to enter into salt water than did the control smolts.

Luo et al. (2005) treated two groups of smolting sockeye salmon with either 15 mg/kg or 150 mg/kg nonylphenol via IP injection twice within a 4-day period. While the sockeye salmon smolts experienced normal physiological changes during smoltification, changes in gene expression of ERα mRNA were observed in fish exposed to both 15 and 150 mg/kg nonylphenol.

McCormick et al. (2005) intraperitoneally injected juvenile Atlantic salmon living in freshwater with four separate injections over an 11 day period with one of five doses (0.5, 2, 10, 40 or 150 µg/g) of branched 4-nonylphenols. Fourteen days after the first injection, the juvenile salmon were transferred to seawater for 24 hours to assess their salinity. The 150 µg/g exposure was the only one of the five doses resulting in a significant reduction in salinity tolerance, which was identified by a significant increase in plasma sodium concentration. No mortality was observed in any of the nonylphenol exposures. The authors

concluded that if this detrimental effect of elevated nonylphenol exposure was continued, it would likely compromise the normal parr-smolt transformation of Atlantic salmon.

Arsenault et al. (2004) exposed Atlantic salmon parr-smolts to a single 20 µg/L nonylphenol concentration in freshwater for 12 – 14 days, then allowed the fish to gradually adapt to seawater. After a 51 - 52 day residence in nonylphenol-free seawater, smolt weight and specific growth rates were measured. The nonylphenol-exposed smolt had statistically significantly reduced growth and IGF-1 activity relative to controls. This impact of nonylphenol is believed by Arsenault et al. (2004) to have adverse long-term effects on growth and the optimal completion of the smoltification process.

Madsen et al. (2004) performed an integrated laboratory and field study of nonylphenol effects on Atlantic salmon smoltification, downstream migration and survival of smolts. A group of 150 smolts in freshwater were injected six times over a 20 day period with 120 µg/g 4-nonylphenol. Two 24-hour salt water challenge tests were performed prior to the start of nonylphenol injections, and a third salt water challenge was given at the end of the 20-day nonylphenol injection period. All smolts were then individually tagged with passive integrated transponder (PIT) tags at the end of the nonylphenol injection period. A total of 100 smolts nonylphenol injected and 100 control smolts were then transferred to the head of a 3.2 km long freshwater stream two days after the last nonylphenol injection. Smolts were released into the stream and then their ability to migrate downstream was monitored. Control fish immediately began their downstream migration, while the start of downstream migration of the nonylphenol-injected smolts was delayed by 6 – 8 days compared to that of the controls. After 34 days in the stream, 81% of the control smolts were captured alive in a trap at the downstream end of the 3.2 km stream, while only 12% of the nonylphenol-injected smolts were captured alive in the downstream trap. Madsen et al. (Madsen et al., 2004) speculated that the difference in survival between the nonylphenol exposed and control smolts during downstream migration was due to a combination of impaired smolt development, behavioral changes and increased predation by herons and mink on the nonylphenol-exposed smolts.

Madsen et al. (2004) maintained the remaining 50 control and nonylphenol exposed smolts in laboratory freshwater for the same length of time the other smolts were present in the 3.2 km stream. Subsets of these 50 smolts retained in laboratory were salt water challenged the same day smolts were released outdoors into the stream: only one nonylphenol-exposed smolt survived this 24-hour salt water challenge. All remaining control and nonylphenol-exposed smolts kept in laboratory freshwater during the time smolts were present in the 3.2 km stream survived to the end of the study.

Finally, one study of nonylphenol effects on salmonid smoltification evaluated dietary exposures and reported water column nonylphenol concentrations, as opposed the studies above that exposed salmonids via either water column exposure or IP injection for exposures shorter than 28 days. Keen et al. (2005) fed juvenile coho salmon (*Oncorhynchus kisutch*) with one of four dietary concentrations (0.002, 0.2, 20 or 2000 mg/kg of 4-nonylphenol) plus a control for 28 days in freshwater. Fish growth in salt water was then observed for two successive 6-week intervals. Dietary exposure to nonylphenol had no effect on survival, growth or smoltification of coho salmon, even when fed the highest dietary concentration of 2000 mg/kg nonylphenol. Keen et al. (2005) attributed this lack of toxicity on the ability of coho salmon to eliminate the ingested nonylphenol via the biliary-fecal pathway after their transfer into salt water. This observation of no toxicity from dietary ingestion of nonylphenol conflicts with the studies of water column nonylphenol exposure on smoltification, where adverse effects were observed.

A minimal amount of nonylphenol leached from the contaminated food into the water column during the 28-day freshwater feeding portion of the study, with a maximum detected concentration of 2.183 µg/L found in a water drain sample from a 2000 mg/kg dietary exposure tank.

Schoenfuss et al. (2008) exposed fathead minnows (*Pimephales promelas*) for 28 days in two separate exposure groups to a range of nonylphenol concentrations, and monitored the ability of exposed males to defend nests (defined as chasing off other male minnows from the nest site) and compete with control males for access to females. Both of these behaviors indicate effects of nonylphenol on reproductive behaviors. In the first group of fish exposed, the nonylphenol concentration tested of 0.15 µg/L resulted in exposed fish being able to outcompete control, nonexposed fish for access to nest sites. This statistically significant hermetic response of fathead minnows to nonylphenol is the lowest concentration affecting any freshwater fish species of which we are aware. The three higher nonylphenol concentrations tested with the first group of fish (0.25, 0.63 and 3.2 µg/L) all resulted in the nonylphenol exposed fish being unable to compete with the control fish for nest sites and access to females. All three of these exposures resulted in a statistically significant reduction of exposed minnows ability to compete with control minnows for access to nest sites and females.

In the second group of fathead minnows Schoenfuss et al. (2008) exposed to nonylphenol, the same general trend was observed as in their first experiment. However, the effect concentrations were much higher in this second exposure study. Minnows exposed to 0.3 and 5.0 µg/L nonylphenol outcompeted control minnows for access to females and nest sites. Minnows exposed to 11 and 15 µg/L nonylphenol for 28 days were unable to successfully compete with control minnows for access to females and their ability to access and defend nest sites. The authors were unable to explain the reasons for the difference in responses between the two exposure group experiments, and why adverse effects were exhibited at lower nonylphenol exposure concentrations in the first exposure group compared to the second exposure group. Schoenfuss et al. (2008) speculated that because the fish in the second exposure group were from a different year class than the fish in the first exposure group, and the second exposure was at a water temperature 3°C higher than the first exposure, that these factors may have played a role in the differing sensitivities of fathead minnows in the two exposure groups. But the authors also speculated that the differences may also simply reflect the complexity that live organisms introduce into evaluating results from toxicity tests with endocrine disrupting chemicals.

Sun et al. (2017) performed a three generation chronic study with the introduced zebrafish (*Danio rerio*). Their primary objective was to determine if nonylphenol altered transcriptional expression of sexual differentiation related genes, but also measured effects on growth and sex ratio of the test populations. A 125 day exposure to a mean 182.6 µg/L of nonylphenol to the F_1 generation reduced the length of zebrafish, but the same exposure duration to means of 2.9 and 20.27 µg/L nonylphenol had no effect on F_1 generation zebrafish growth. The same growth effects were observed in the F_2 generation zebrafish. The length of zebrafish was statistically significantly reduced after a 140 day exposure to a nominal 200 µg/L, but no growth effects were observed in the F_2 generation exposed to a nominal 2 or 20 µg/L nonylphenol for 140 days.

Sun et al. (2017) also observed that 20 µg/L nonylphenol altered the sex ratio of the F_1 generation zebrafish. The F_1 generation had a sex ratio of 80% females to 20% males at the end of the 125 day exposure, statistically significantly different from the expected 50:50 ratio. No effect on sex ratio of the F_1 generation fish was observed after 125 days of exposure to either 2 or 200 µg/L nonylphenol. n the F_2

generation exposed for 140 days, the sex ratio of the zebrafish exposed to 20 µg/L again showed an elevated proportion of females to males, but was not statistically significantly elevated compared to the sex ratio of the controls or fish exposed to 2 or 200 µg/L nonylphenol.

Lin & Janz (2006) also observed that nonylphenol altered the sex ratio of zebrafish populations. They exposed zebrafish from 2 to 60 days post hatch (dph) to a nominal 10 or 100 µg/L nonylphenol, then held the fish in nonylphenol-free water until 240 dph. At the end of the approximately 6 month exposure in nonylphenol-free water, the reproductive fitness of the zebrafish was evaluated. No statistically significant differences in survival or growth relative to controls were observed in the 10 and 100 µg/L exposures in the 60 dph zebrafish. Both 10 and 100 µg/L nonylphenol significantly altered sex ratios of 60 dph zebrafish, with the ratio of females to males elevated above the female:male ratio in controls. No males were identified in the 100 µg/L exposure zebrafish population at 60 dph.

Several delayed effects on the 240 dph zebrafish exposed to 100 µg/L nonylphenol between 2 and 60 dph were observed by Lin and Janz (Lin & Janz, 2006). Offspring (F_1 generation) of adult fish (F_0 generation) in the 100 µg/L exposure group exhibited statistically significantly reduced F_1 swim-up success, as well as an altered sex ratio relative to control F_1 generation fish. Unlike the sex ratios of the F_0 generation after 58 day exposure to nonylphenol, where females were more abundant than males, the F_1 generation fish from parents exposed to 100 µg/L nonylphenol had an elevated ratio of males to females. The 240 dph F_0 generation fish exposed to 10 µg/L exhibited no significant differences in survival, growth, sex ratio, amount of successful breeding and egg production compared to controls. The viability, hatchability and swim-up success of the F_1 generation offspring from 10 µg/L exposed F_0 parents was also statistically significantly unaffected by 10 µg/L nonylphenol. Except for the effects on swim-up success and sex ratio alteration, the 100 µg/L exposure to nonylphenol had no discernable effects on any other measured survival, reproduction or growth on either the F_0 or F_1 generation fish. To our knowledge, Sun et al. (Sun et al., 2017) and Lin and Janz (Lin & Janz, 2006) are the only two studies with nonylphenol to investigate its chronic effects on a population level endpoint (sex ratio) in fish.

Ishibashi et al. (2006) performed a two generation study on nonylphenol effects on reproduction in Japanese medaka (*Oryzias latipes*), a fish native to Asia. Several introductions of this species to the continental United States have been extirpated. It is uncertain whether an introduced population to Hawaii is still viable. Breeding pairs of medaka (F_0 generation) were exposed to 10, 50 or 100 µg/L nonylphenol for 21 days, and their survival, growth, fecundity and egg viability were monitored. No adverse nonylphenol effects were observed on any endpoint in the 21 day F_0 generation exposures to 10 and 50 µg/L. No effects on medaka length or weight were observed at any exposure concentration. While all females survived the exposure to 100 µg/L, 40% of males did not survive the 21 day exposure to 100 µg/L nonylphenol. Both the total number of eggs produced by the F_0 generation parents and their fertility was significantly reduced in the 21-day 100 µg/L nonylphenol exposure.

Ishibashi et al. (2006) then performed a second medaka 21-day experiment using only a 100 µg/L exposure to nonylphenol, since no toxicity to the F_0 generation was observed in the earlier 10 and 50 µg/L exposures. The second study was designed to evaluate nonylphenol effects on the offspring (F_1 generation) of the parent F_0 generation medaka. Hatchability of the eggs was significantly reduced compared to control eggs, while their time to hatching was significantly longer than the length of time needed for the control eggs to hatch. Once hatched, F_1 generation medaka maintained in clean water

for 90 days after hatching exhibited no effects on F_1 generation survival, length, weight or sex ratio of the offspring.

Although not cited as a chronic study in the EPA (USEPA, 2005a) nonylphenol criteria document, Liber et al. (Liber, Knuth, & Stay, 1999) exposed brook stickleback (*Culaea inconstans*) to nonylphenol for 122 days in littoral enclosures within a pond outside of Duluth, MN. This study was not used in criteria derivation for, among other reasons, not being performed in a controlled laboratory setting. However, the 122 day survival chronic NOEC of 243 µg/L indicates that at least some fish species can survive extended exposures to nonylphenol at concentrations higher than the aquatic life criteria without suffering any adverse effects.

Several chronic nonylphenol toxicity studies on fish species without reproducing populations in North America were identified. Zha et al. (Zha, Wang, Wang, & Ingersoll, 2007) observed a 28 day growth NOEC (for both length and weight) of Chinese rare minnow (*Gobiocypris rarus*) of 30 µg/L, the highest of four nonylphenol exposure concentrations tested (control, 3, 10 and 30 µg/L).

Adverse effects on reproduction shown by reduced gamete production were observed by Kaptaner and Ünal (Kaptaner & Unal, 2011) in tarek (*Alburnus tarichi*), an endangered cyprinid found only in Lake Van, Turkey. During a 32 day exposure to 10, 60 and 200 µg/L nonylphenol, increased apoptosis (cell death) in both male and female reproductive organs was observed at 200 µg/L, but not at either 10 or 60 µg/L.

5.5.1.13.7 Acute-chronic ratio for nonylphenol

The final ACR for nonylphenol given in EPA (USEPA, 2005a) is 8.412. This ACR was derived from three species mean ACR values of 2.023, 8.412 and 28.11 for *Daphnia magna*, *Americamysis bahia* and rainbow trout, respectively. However, because there is an empirical species specific ACR for rainbow trout, one of the ESA listed species within the action area, we will use the rainbow trout ACR of 28.11 for all listed salmonids evaluated in this section. This rainbow trout ACR was derived from the freshwater acute and chronic toxicity studies performed by Brooke (L. T. Brooke, 1993), which are used in this nonylphenol effects assessment to directly evaluate the toxicity of nonylphenol to rainbow trout.

The Guidelines stipulate that if the available species mean ACRs seems to increase or decrease as the SMAVs for the same species increase, the Final ACR (FACR) should be based on the ACRs for species whose SMAVs are close to the FAV. The Guidelines also permit the FACR to be derived as the geometric mean of all available species mean ACRs, the approach most commonly used for chemicals for which EPA has derived FACR values in criteria documents.

Examination of the three available species mean ACRs (2.023, 8.412 and 28.11) relative to the SMAVs indicates that the more acutely sensitive species (*D. magna* and *A. bahia*) have 3 to 14-fold lower species mean ACRs than does the less acutely sensitive rainbow trout, indicating a general trend of increasing species mean ACR with increasing SMAV. Therefore, the FACR should be based on the species mean ACR(s) for species whose SMAVs are close to the FAV. The *A. bahia* SMAV (51.05 µg/L) is closest to both the freshwater FAV (55.49 µg/L) and the saltwater FAV (13.93 µg/L). This was the rationale in EPA (USEPA, 2005a) for using the *A. bahia* species mean ACR of 8.412 as the nonylphenol FACR for all species, as opposed to the more commonly used approach of calculating a geometric mean of the three available species mean ACRs in EPA (USEPA, 2005a) as the FACR, which would have been 7.821 for the existing nonylphenol ACR data.

After the publication of the EPA (USEPA, 2005a) nonylphenol criteria document, Isidori et al. (Isidori, Lavorgna, Nardelli, & Parrella, 2006) published results of a series of two day acute and seven day chronic toxicity studies of nonylphenol with *Ceriodaphnia dubia*, which permit the calculation of a fourth ACR value for nonylphenol. When combined with the chronic *Ceriodaphnia* toxicity data of England (D. E. England, 1995), a *Ceriodaphnia dubia* ACR of 9.083 was calculated. This *C. dubia* ACR is comparable to both the ACR of 8.412 recommended in EPA (USEPA, 2005a) and a recalculated four species geometric mean ACR of 8.119, which includes the new *C. dubia* ACR of 9.083.

5.5.1.13.8 Effects assessment of freshwater chronic nonylphenol criterion on listed species

The empirical 91 day growth chronic NOEC for rainbow trout measured by Brooke (L. Brooke, 1993; L. T. Brooke, 1993) was 7.861 µg/L. The exposure duration of the Brooke (L. T. Brooke, 1993) study was approximately three times the recommended exposure duration for chronic studies with fish recommended in the Guidelines. It is over 22 times longer than the four day exposure duration specified in the definition of chronic exposure in both the EPA aquatic life criteria and the Swinomish Tribe water quality criteria. Even with the extended exposure duration of rainbow trout to nonylphenol in the Brooke (L. T. Brooke, 1993) study, the chronic NOEC is higher than the chronic freshwater nonylphenol criterion of 6.6 µg/L. Brooke (L. T. Brooke, 1993) is also the longest duration chronic study where any fish species of the genera *Oncorhynchus* or *Salvelinus*, the two genera containing all of the ESA listed freshwater fish within the action area were continuously exposed to nonylphenol. As such, exposure at the level of the freshwater chronic nonylphenol criterion is not likely to adversely affect rainbow trout (steelhead), based on empirical toxicity data.

No empirical chronic toxicity data for nonylphenol exists for the remaining listed salmonid species within the freshwater portions of the action area. Assuming that this 7.861 µg/L threshold chronic effect concentration or chronic NOEC for rainbow trout is less than or equal to that for all ESA listed fish species in the freshwater portions of the action area, exposure at the level of the freshwater nonylphenol chronic criterion of 6.6 µg/L is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon and bull trout.

5.5.1.13.9 Summary of freshwater effect determinations for nonylphenol

EPA has made the following determinations on the acute and chronic nonylphenol criteria for the listed species in the freshwater portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Likely to adversely affect** (indirect effects - availability of prey)

Chinook salmon – **Likely to adversely affect** (indirect effects - availability of prey)

Bull trout – **Likely to adversely affect** (indirect effects - availability of prey)

Marbled murrelet – **Likely to adversely affect** (indirect effects - availability of prey)

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.5.1.14 Silver – Freshwater

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion – hardness dependent (3.2 µg/L at 100 mg/l hardness)

Chronic (CCC) criterion – none

CAS ID 7761-88-8 (silver nitrate)

7783-90-6 (silver chloride)

5.5.1.14.1 Introduction to silver

Silver occurs naturally in the pure form and in ores. It is primarily used in photographic materials, electroplating, as a conductor, in dental alloys, solder and brazing alloys, paints, jewelry, silverware, coinage, and mirror production. Silver has also been used as an antibacterial agent. Silver is found in extremely low concentrations in the aquatic environment. In a study conducted by Kharkar, et al. (Kharkar, Turekian, & Bertine, 1968) on 10 U.S. rivers, the study reported that silver was detected in concentrations ranging from 0.092 to 0.55 µg/L. In addition, J. D. Hem (1970) also cites studies of public drinking water supplies and river waters which reported median concentrations of 0.23 and 0.09 µg/L, respectively. The geochemistry of silver has been extensively reviewed by Boyle (1968).

The primary sources of silver in surface water include industrial and smelting wastes, wastes in jewelry manufacturing, electrical supply equipments and most importantly, from the production and disposal of photographic materials (EPA, 1987). Silver comes in four oxidation states (0, +1, +2 and +3), but only the 0 and +1 states occur to any extent in the environment. Silver may exist in water as simple hydrated monovalent ions completely dissociated from anions that at one time could have been part of its crystalline salt mesh. Additionally, monovalent silver ions may exist in various forms in association with large number of inorganic ions, such as sulfate, bicarbonate, and nitrate, to form numerous compounds with a range of solubilities and potentials for hydrolysis or other reactions (EPA, 1980).

5.5.1.14.2 Silver toxicity to aquatic organisms

Silver is among the list of most toxic metals to aquatic life, particularly in freshwater. The majority of toxicity studies conducted with silver have been with silver nitrate, because it is the most soluble and a good source of free soluble silver ions; insoluble silver salts are less toxic than silver nitrate (EPA, 1980). A similar acute silver criterion, 3.4 ug/L at 100 mg/L hardness as $CaCO_3$, was assessed as part of the ESA consultation on Idaho's toxics criteria, and NLAA determinations were made for Idaho's silver criterion based on the lesser toxicity of the forms of silver found in the environment than the silver nitrate

commonly used for toxicity tests. USFWS ((USFWS, 2015a), pages 224-225) provides the following summary for the toxicity of silver compounds:

> Ionic silver is the primary form responsible for causing acute toxicity in freshwater fish (Wood 2011b)[(C.M. Wood, 2011) in reference section of this BE]. Toxicity varies widely depending on the anion present: silver nitrate has a much higher toxicity than silver chloride or silver thiosulfate, by approximately four orders of magnitude (Hogstrand et al. 1996)[(Hogstrand, Galvez, & Wood, 1996) in reference section of this BE]. Documented effects of silver toxicity in fish from silver nitrate include interruption of ionoregulation at the gills, cell damage in the gills, altered blood chemistry, interference with zinc metabolism, premature hatching, and reduced growth rates (Hogstrand and Wood 1998)[(C. Hogstrand & C.M. Wood, 1998) in reference section of this BE].
>
> Silver is sparingly soluble and rare in aquatic environments. … Even in highly contaminated areas, silver concentrations rarely exceed 0.1 to 0.3 µg/L. In nature, silver is unlikely to be found in its ionic form. Given the extremely high affinity of silver for reduced sulfur, most silver in the environment is expected to occur as silver sulfides, even in oxygenated waters (Wood 2011b). Even in Idaho's Silver Valley where 100+ years of silver mining resulted in one of the largest Superfund cleanup projects in the nation, silver is not a contaminant of concern (National Research Council (NRC) 2005). Although silver sulfides are the form most likely found in the environment, the form of silver usually used in most toxicity tests is silver nitrate because it dissolved better. Silver nitrate is much more toxic than the sparingly soluble silver sulfides (Wood 2011b).

5.5.1.14.3 Silver toxicity to aquatic life and hardness

In freshwater, the acute aquatic life criterion is that the concentration (in µg/L) of silver may not exceed the numerical value given by [$e^{(1.72[\ln(hardness)]-6.52)}*0.85$] at any time. Based on available data it has been determined that silver toxicity in freshwater is influenced by water hardness. For example, at hardness of 50, 100, and 200 mg/L as $CaCO_3$, the concentration of silver should not exceed 1.0, 3.2, and 11 µg/L, respectively, at any time. Available data indicate that chronic toxicity of silver to freshwater aquatic life may occur at concentrations as low as 0.12 µg/L (EPA, 1980). However, the greater toxicity of silver nitrate, most commonly used in toxicity tests, relative to the silver sulfides and other compounds commonly present in the environment must be taken into consideration when evaluating a criterion as described below.

5.5.1.14.4 Water-column exposure toxicity data - Acute silver toxicity in freshwater

Silver, when present as ionic silver (Ag^+), can be acutely toxic to freshwater fish. The 96-hr LC_{50} values range between 5 and 70 µg/L total silver](Christer Hogstrand & Chris M Wood, 1998). In juvenile and adult rainbow trout, acute toxicity occurs through noncompetitive inhibition of active Na^+ and Cl^- uptake at the gills, a consequence of impairment of the gill enzymes sodium, potassium-adenosine triphosphatase ($Na^+ K^+$-ATPase) and carbonic anhydrase by Ag^+ (I. J. Morgan, Henry, & Wood, 1997). Inhibition of active ion uptake leads to reductions in plasma Na^+ and Cl^- levels, which in turn initiate a suite of secondary effects that ultimately lead to circulatory collapse and mortality (Chris M Wood, Hogstrand, Galvez, & Munger, 1996).

5.5.1.14.5 Effect determinations for silver

A review of studies in the ECOTOX database for acute silver toxicity was completed. Data are summarized in Appendix G.

The toxicity data point to the acute toxicity study by Bianchini et al. (Bianchini, Grosell, Gregory, & Wood, 2002) on Rainbow Trout (*Oncorhynchus mykiss*) as having the lowest toxicity value. The 96-hr LC_{50} mortality result at hardness of 150 mg/L $CaCO_3$ was reported to be approximately 0.0003 mg/L. After the toxicity values was hardness adjusted at 100 mg/L ($CaCO_3$), the resulting hardness-adjusted value is 0.0001 mg/L (0.1 µg/L). This value is lower than the Swinomish Tribe's acute silver criterion, 3.2 µg/L (for 100 mg/L $CaCO_3$), which implies that exposure of salmonids to silver at the criterion level would be likely to result in adverse effects.

However, data reviewed in connection with the ESA consultation on EPA's approval of the same acute silver criterion in Idaho demonstrated that consideration must be given to the greater toxicity of silver nitrate, commonly used in toxicity tests, than the silver species that are prevalent in surface waters. For example, USFWS (2015, pp. 224-226) indicated that silver "is sparingly soluble and rare in aquatic environments" and provided the following analysis:

> Even in highly contaminated areas, silver concentrations rarely exceed 0.1 to 0.3 µg/L. In nature, silver is unlikely to be found in its ionic form. Given the extremely high affinity of silver for reduced sulfur, most silver in the environment is expected to occur as silver sulfides, even in oxygenated waters (Wood 2011b) [(C.M. Wood, 2011) in reference section of this BE]. Even in Idaho's Silver Valley where 100+ years of silver mining resulted in one of the largest Superfund cleanup projects in the nation, silver is not a contaminant of concern (National Research Council (NRC) 2005). Although silver sulfides are the form most likely found in the environment, the form of silver usually used in most toxicity tests is silver nitrate because it dissolved better. Silver nitrate is much more toxic than the sparingly soluble silver sulfides (Wood, 2011b). Chronic toxicity to freshwater aquatic life from silver nitrate may occur at concentrations as low as 0.12 µg/L (EPA 1980a) and the published silver criterion ranges from 0.07 to 11 µg/L over a range of water hardness values from 10 to 200 mg/L.
>
> …
>
> A key point from the environmental chemistry and aquatic toxicology literature for silver is overwhelming differences in toxicity between free ionic silver and complexed silver compounds. Most laboratory toxicity tests with silver used silver nitrate because it readily disassociates into ionic silver which tends to remain in solution (Hogstrand and Wood 1998). In contrast, in rivers, streams, lakes, and effluents, ionic silver tends to be vanishingly low, and measurable silver in natural waters and effluents occurs as either silver sulfide, silver chloride, silver thiosulfate, or as complexes with natural DOC (Adams and Kramer 1999; Kramer et al. 1999) [(Adams & Kramer, 1999; Kramer, Adams, Manalopoulos, & Collins, 1999) in reference section of this BE]. The differences in effects concentrations obtained between tests using silver nitrate and other forms of silver may be on the orders of magnitude. For instance, Hogstrand et al. (1996) obtained a 7-day LC50 with rainbow trout and silver nitrate of 9 µg silver/L, but silver chloride and silver thiosulfate LC50s were >100,000 µg silver/L. Similarly, with the fathead minnow, compared to the free silver ion resulting from silver nitrate additions, silver chloride complexes were about 300 times less toxic and silver sulfide was at least 15,000 times less toxic (Leblanc et al. 1984)

> [(Leblanc et al., 1984) in reference section of this BE]. When very low and environmentally realistic levels of sulfide were added to a test water (0.0016 mg/L), the LC50 of Daphnia magna was increased by a factor of 5.5 (Bianchi et al. 2002) [(Bianchi et al., 2002) in reference section of this BE].

The lowest toxicity value identified for silver was 0.1 µg/L, based on Bianchini et al. (Bianchini et al., 2002) as described above, while the Swinomish Tribe's silver criterion is 3.2 µg/L. Multiplying the toxicity value of 0.1 µg/L by 300, an adjustment factor that accounts for the relative toxicity of the free silver ion used for toxicity tests and the silver chloride complexes found in the environment as identified for fathead minnow by Leblanc et al. (Leblanc et al., 1984), yields 30 µg/L, a toxicity value that is above the criterion of 3.2 µg/L. On this basis, exposure of listed salmonid species to waterborne silver would not be likely to result in adverse effects.

USFWS (2015) found that exposure at the silver criteria level may have adverse effects on daphnids, which are prey items for bull trout. However, given the above considerations and the diversity of prey that bull trout may consume, USFWS (2015, p. 229) determined that "the Service does not expect significant adverse effects to the bull trout to be caused by the proposed acute criterion for silver." NMFS (2014, p. 186) similarly concluded that "it appears unlikely that acute toxicity to salmonids at criterion concentrations will occur" and summarized the rationale as follows (NMFS 2014, p. 186):

> The potential inadequacies and underprotectiveness of the silver criterion are mitigated by the fact that in the environment, silver occurs in a less toxic form than that used in most of the toxicity tests published in the literature. Significant food chain biomagnification by fish is also possible, but all of these effects appear unlikely to occur because of the low silver concentrations typically encountered in the aquatic environment.

No silver sources to the Swinomish Tribe's fresh waters have been identified, and the considerations provided by NMFS (2014) and USFWS (2015) with respect to EPA's approval of Idaho's criteria would apply to the Swinomish Tribe's waters and criteria as well.

5.5.1.14.6 Summary of freshwater effect determinations for silver

Based on the analysis provided by NMFS (NMFS, 2014b) and USFWS (USFWS, 2015a) for the approval of Idaho's silver criterion, EPA has determined that its proposed approval of the Swinomish Tribe's acute freshwater silver criterion is not likely to adversely affect listed species in the action area. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.5.1.15 Zinc – Freshwater

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = hardness dependent (120 µg/L at 100 mg/L hardness)

Chronic (CCC) criterion = hardness dependent (120 µg/L at 100 mg/L hardness)

CAS ID 7440-66-6

For the purposes of this zinc biological evaluation, we have normalized all freshwater effect concentrations to a water hardness of 100 mg/L $CaCO_3$. Because nearly all freshwater zinc toxicity literature studies have been run at different hardness values and the freshwater zinc criteria values are hardness dependent, the hardness normalization is done so that zinc toxicity results are comparable to each other, and can be compared to a single criterion concentration.

5.5.1.15.1 Water-column exposure toxicity data - acute criterion for zinc in freshwater

EPA's 1987 aquatic life criteria document for zinc (USEPA, 1987)[24] contains empirical 96-hour acute toxicity data for rainbow trout, Chinook salmon, coho salmon, cutthroat trout and sockeye salmon, all five species of which are in the genus *Oncorhynchus*. The empirical data for rainbow trout and Chinook salmon allows a direct assessment of zinc toxicity to these two ESA listed species present in the action area. Empirical data for the five *Oncorhynchus* species permits calculation of a chum salmon 96-hour LC_{50} using an ICE model. Empirical toxicity data for the brook trout (*Salvelinus fontinalis*) permits the use of an ICE model to estimate acute toxicity for bull trout (*Salvelinus confluentus*), the remaining ESA listed fish species in the action area.

Twenty four 96 hour LC_{50} values for rainbow trout were used in EPA (USEPA, 1987) to calculate the species mean acute value of 1243 µg/L at 100 mg/L hardness (691 µg/L at 50 mg/L hardness as published in EPA (USEPA, 1987). The lowest of the 24 96 hour LC_{50} values used to calculate the rainbow trout species mean acute value, and the only one of the 24 lower than the zinc acute criterion of 120 µg/L at 100 mg/L hardness was a study by Goettl et al. (J.P. Goettl, Sinley, & Davies, 1974). Rainbow trout weighing 72 grams were found by J.P. Goettl et al. (1974) to have a 96 hour LC_{50} of 98 µg/L at a hardness of 102 mg/L (= 96 µg/L at a hardness of 100 mg/L). The next lowest rainbow trout 96 hour LC_{50} of 93 µg/L at a hardness of 23 mg/L (= 323 µg/L at a hardness of 100 mg/L) was from a study by Chapman (G. A. Chapman, 1975) with swim-up alevins.

Four available 96 hour LC_{50} values for Chinook salmon are presented in EPA (USEPA, 1987). The species mean acute value for Chinook salmon is 437 µg/L at a hardness of 50 mg/L (= 1518 µg/L at a hardness of 100 mg/L). The lowest of the four 96 hour LC_{50} values for Chinook salmon is 84 µg/L at a hardness of 21 mg/L (= 315 µg/L at a hardness of 100 mg/L) for juvenile Chinook salmon (Finlayson & Verrue, 1982). All of the available Chinook salmon 96 hour LC_{50} values are higher than the acute zinc criterion of 120 µg/L at a hardness of 100 mg/L.

[24] EPA's current criteria recommendation for zinc is included in *1995 Updates: Water Quality Criteria Documents for the Protection of Aquatic Life in Ambient Water* (EPA-820-B-96-001, September 1996). No additional toxicity data for salmonids or other fish species were identified in EPA (1996), and the more detailed data summary in EPA (1987) was used for the zinc assessment in this BE.

The six brook trout 96 hour LC_{50} values in EPA (USEPA, 1987), which can be used in ICE models as a surrogate genus for the ESA listed bull trout yield a species mean acute value of 2133 µg/L at a hardness of 50 mg/L, corresponding to a brook trout species mean acute value of 3838 µg/L at a hardness of 100 mg/L. None of the individual brook trout 96 hour LC_{50} values are lower than the acute zinc criterion of 120 µg/L at a hardness of 100 mg/L.

5.5.1.15.2 Additional freshwater acute toxicity data for zinc published since 1985

Stratus Consulting (Consulting, 1999) performed acute toxicity tests on both bull trout and rainbow trout with zinc and cadmium. These studies were part of an injury assessment at the Coeur d'Alene Superfund site designed to estimate zinc and cadmium toxicity under water quality conditions found in the Coeur d'Alene River. The Stratus Consulting (1999) report contains the raw data reported in the peer-reviewed Hansen et al. (James A Hansen, Paul G Welsh, Josh Lipton, Dave Cacela, & Anne D Dailey, 2002) study, which did not contain the 96 hour LC_{50} results for bull trout or rainbow trout present in the Stratus report. A series of toxicity tests were performed at combinations of water quality conditions: nominal hardness values of 30 or 90 mg/L as CaCO3, at pH 7.5 or 6.5, and at water temperatures of 8° or 12°C.

Three of the eight rainbow trout replicate 96 hour LC_{50}s from Stratus were lower than the 120 µg/L at a hardness of 100 mg/L freshwater criterion once normalized to 100 mg/L hardness. Three of the eight bull trout replicate 96 hour LC_{50}s from Stratus were also lower than the 120 µg/L at a hardness of 100 mg/L freshwater criterion once normalized to 100 mg/L hardness. Stratus Consulting (1999) thus becomes the second study with acute toxicity results for rainbow trout that are lower than the freshwater acute zinc criterion. As the only empirical bull trout study with zinc, Stratus Consulting (1999) also provides some evidence that the acute toxicity of zinc to bull trout is also lower than the freshwater acute zinc criterion.

Four additional studies containing 96 hour LC_{50}s for rainbow trout were also identified. Alsop and Wood (2000) observed a 96 hour LC_{50}s of 3000 µg/L at a hardness of 120 mg/L (= 2570 µg/L at a hardness of 100 mg/L) with juvenile rainbow trout. De Schamphelaere and Janssen (De Schamphelaere & Janssen, 2004) performed four separate 96 hour LC_{50} studies with swim-up rainbow trout fry at 25 mg/L hardness with a mean LC_{50} of 570 µg/L when converted to a hardness of 100 mg/L. Gündoğdu (2008) using juvenile rainbow trout weighing 3.02 grams determined a 96 hour LC_{50} of 12,880 µg/L at a hardness of 249.6 mg/L (= 5934 µg/L at a hardness of 100 mg/L). Mebane et al. (Christopher A Mebane, Hennessy, & Dillon, 2008) measured 96 hour LC_{50}s of 117 and 130 µg/L at hardness of 16 and 24 mg/L, respectively. When converted to 96 hour LC_{50}s at hardness of 100 mg/L, the mean 96 hour LC_{50} for the swim-up fry used by Mebane et al. (Christopher A Mebane et al., 2008) at a hardness of 100 mg/L is 495 µg/L.

When the five new mean 96 hour LC_{50} values are appended to the 24 96 hour LC_{50} values in EPA (USEPA, 1987) and a new species mean acute value is calculated, the new rainbow trout SMAV is 1190 µg/L at 100 mg/L hardness. This new SMAV is comparable to the rainbow trout SMAV of 1243 µg/L at 100 mg/L hardness in EPA (USEPA, 1987).

5.5.1.15.3 Effects assessment of acute freshwater zinc criterion on listed species

All available Chinook salmon 96 hour LC_{50}s when normalized to 100 mg/L hardness exceed the acute zinc criterion of 120 µg/L at 100 mg/L hardness. The species mean acute value for Chinook salmon is 1518 µg/L at 100 mg/L hardness. The Chinook salmon SMAV divided by 2.27 translates the SMAV to an

effects assessment concentration of 669 µg/L at 100 mg/L hardness. Based on empirical toxicity data, the freshwater acute zinc criterion is not likely to adversely affect Chinook salmon.

Of the 29 literature citations containing 96 hour LC_{50} data for rainbow trout, 27 have 96 hour LC_{50} values greater than the freshwater acute zinc criterion. None of these 27 acute toxicity values, when divided by 2.27 to calculate the effects assessment concentrations for these 27 studies resulted in acute effects assessment concentrations lower than the freshwater acute zinc criterion.

The remaining two rainbow trout 96 hour LC_{50} studies ((J.P. Goettl, Davies, & Sinley, 1976; J.P. Goettl et al., 1974), Stratus Consulting, 1999) contain acute toxicity results where the 96 hour LC_{50} is lower than the freshwater zinc acute criterion. Stratus had multiple toxicity test results at 30 mg/L hardness where the 96 hour LC_{50} was lower than the acute criterion at the test hardness, and a single 96 hour LC_{50} at 90 mg/L hardness where the effects assessment concentration (i.e. LC_{50}/2.27) was lower than the freshwater zinc acute criterion of 120 µg/L.

Although the rainbow trout species mean acute value of 1190 µg/L, based on 29 individual study 96 hour LC_{50}s and the species mean acute effects assessment concentration of 524 µg/L, both at 100 mg/L hardness are higher than the acute criterion of 120 µg/L at 100 mg/L hardness, the existence of two studies with multiple 96 hour LC_{50}s lower than the acute zinc criterion raises concerns. Zinc exposures at hardnesses between 30 and 102 mg/L resulted in rainbow trout 96 hour LC_{50}s lower than the acute criterion in Goettl et al. (J.P. Goettl et al., 1976; J.P. Goettl et al., 1974) and Stratus Consulting (1999), so it does not appear that the highest sensitivity of rainbow trout to zinc is limited to a small range of water hardness values. Sufficient information exists from two different rainbow trout acute toxicity studies with zinc to support a conclusion that the freshwater zinc acute criterion is likely to adversely affect rainbow trout (steelhead).

The one available study on acute zinc toxicity to bull trout (Stratus Consulting, 1999) observed that two of eight replicate 96 hour LC_{50}s were lower than the acute freshwater zinc criterion. An additional two of the eight replicate bull trout 96 hour LC_{50}s have acute effects assessment concentrations (i.e. LC_{50}/2.27) lower than the freshwater acute zinc criterion. The geometric mean of the eight individual bull trout 96 hour LC_{50}s (268 µg/L) in Stratus Consulting (1999) divided by 2.27 is 116 µg/L, also lower than the 120 µg/L acute zinc criterion. As four of the eight replicate empirical effects assessment concentrations for bull trout in Stratus are lower than the freshwater acute zinc criterion, as is the geometric mean assessment effects concentration for bull trout, sufficient evidence exists to conclude that exposure at the level of the freshwater acute zinc criterion is likely to adversely affect bull trout.

There is no empirical zinc toxicity data for chum salmon (*Oncorhynchus keta*). An estimated 96 hour LC_{50} for chum salmon can be obtained using Interspecies Correlation Estimation (ICE) modeling. Given the number of 96 hour LC_{50} values for other *Oncorhynchus* species (29 alone for rainbow trout, plus four for Chinook salmon, two for coho salmon, and one each for sockeye salmon and cutthroat trout), the selection of the 96 hour LC_{50} to use as the surrogate species from which to estimate the chum salmon 96 hour LC_{50} is somewhat arbitrary. As the two studies with rainbow trout with 96 hour LC_{50}s discussed earlier are the only *Oncorhynchus* species studies with 96 hour LC_{50}s lower than the acute zinc criterion for any of the five *Oncorhynchus* species with empirical acute toxicity data for zinc, it appears that the most representative LC_{50} to use as a surrogate for chum salmon would be a measure of central tendency for all of the acute zinc toxicity data to members of the genus *Oncorhynchus*. Using all available 96 hour

LC_{50} data for zinc acute toxicity to all *Oncorhynchus* species (N = 36), the genus mean acute value is 1243 µg/L.

Assuming that the *Oncorhynchus* genus mean acute value of 1243 µg/L zinc is the central tendency and best estimate of acute zinc toxicity to all *Oncorhynchus* species, we would prefer to use this genus mean acute value as the surrogate toxicity value in the Interspecies Correlation Estimation model to estimate the acute toxicity of zinc to chum salmon. Unfortunately, the only ICE model that predicts toxicity to chum salmon uses sockeye salmon as the surrogate species. Fortunately, the empirical sockeye salmon 96 hour LC_{50} of 2702 µg/L at 100 mg/L hardness for zinc (G. A. Chapman, 1975) is available for use as a surrogate species data in ICE. The ICE estimated 96 hour LC_{50} for chum salmon is 2029 µg/L at 100 mg/L hardness (Table 96). The calculated effects assessment concentration for chum salmon is 894 µg/L. The chum salmon effects assessment concentration is higher than the acute zinc criterion of 120 µg/L at 100 mg/L hardness, supporting a conclusion that exposure at the level of the freshwater zinc acute criterion is not likely to adversely affect chum salmon.

Table 96. Interspecies correlation estimation result for acute zinc toxicity to chum salmon

Predicted Taxa	Chum salmon (*Oncorhynchus keta*)
Model Level	Species
Surrogate	Sockeye salmon (*Oncorhynchus nerka*)
Estimated Toxicity (mg/L)	2028.69
95% Confidence Intervals	1141.6 – 3605.06
Df (N-2)	2
R^2	0.975
p-value	0.0125
Mean Square Error (MSE)	0.0134
Cross-validation success (%)	100
Tax. Dist.	1
Slope	1.01
Intercept	-0.181

5.5.1.15.4 Water-column exposure toxicity data - chronic criterion for zinc in freshwater

Empirical chronic zinc toxicity data exists for both rainbow trout and Chinook salmon, two of the four ESA listed fish species within the freshwater portions of the action area. Empirical chronic toxicity data also exists for brook trout, a surrogate species in the same genus as the ESA listed bull trout.

Sinley et al. (Sinley, Goettl, & Davies, 1974) observed chronic effects on both survival and growth, with survival generally being the most sensitive endpoint, at concentrations between 140 and 547 µg/L at a mean of 26 mg/L hardness (438 and 1713 µg/L at 100 mg/L hardness). Sinley et al. (Sinley et al., 1974) is one of several studies that identified survival as the most sensitive endpoint in chronic zinc toxicity tests, a primary reason that the acute and chronic zinc criteria are numerically equivalent to two significant figures.

Cairns et al. (Cairns, Garton, & Tubb, 1982) was primarily concerned with zinc effects on rainbow trout respiration over a 4 day exposure period, although survival was observed for as long as 72 days. Minimal

effects on survival were observed at zinc concentrations between 444 – 819 µg/L at 25 mg/L hardness (1437 - 2651 µg/L at 100 mg/L hardness).

The G. A. Chapman (1975) chronic study with Chinook salmon also observed survival to be the most sensitive endpoint during chronic exposure to zinc. A chronic survival MATC of 132 µg/L at 25 mg/L hardness (427 µg/L at 100 mg/L hardness) can be calculated from the data in G. A. Chapman (1975), although an MATC was not calculated by G. A. Chapman (1975).

5.5.1.15.5 Additional freshwater chronic toxicity data for zinc published since 1985

Several chronic NOEC studies with rainbow trout have been published since 1985. De Schamphelaere and Janssen (2004) measured 30 day survival LC_{10}s as low as 34.5 µg/L at 29 mg/L hardness (= 98 µg/L at 100 mg/L hardness). The results of De Schamphelaere and Janssen (2004) indicate that low levels of rainbow trout mortality can occur at zinc concentrations lower than the chronic zinc criterion of 120 µg/L at 100 mg/L hardness.

Mebane et al. (Christopher A Mebane et al., 2008) performed a 69 day chronic survival test with rainbow trout. At 30 days exposure, the lowest zinc concentration tested (117 µg/L at 29 mg/L hardness = 334 µg/L at 100 mg/L hardness) elicited approximately 15% mortality.

5.5.1.15.6 Effects assessment of freshwater chronic zinc criterion on listed species

The potential for unacceptable levels of mortality to both rainbow trout and bull trout when exposed to 120 µg/L zinc at 100 mg/L hardness for 96 hours (the acute zinc criterion) would if anything underestimate mortality to these two species when exposed to 120 µg/L zinc at 100 mg/L hardness (the chronic zinc criterion) for chronic exposure durations. Several investigators have also noted that survival is the most sensitive endpoint in chronic zinc toxicity tests. These observations would warrant likely to adversely affect determinations for the chronic zinc criterion to both rainbow trout and bull trout. The findings of low levels of rainbow trout mortality in 30 day exposures at concentrations lower than the chronic zinc criterion (De Schamphelaere & Janssen, 2004) provide additional support for a likely to adversely affect determination for rainbow trout.

The empirical chronic zinc toxicity data for Chinook salmon (G. A. Chapman, 1975) indicated that chronic NOEC survival effects occur at concentrations higher than the 120 µg/L zinc chronic criterion. The Chinook salmon chronic NOEC value from EPA (USEPA, 1987 normalized to 100 mg/L hardness is 1200 µg/L. This empirical evidence supports a not likely to adversely affect determination for Chinook salmon for the freshwater chronic zinc criterion.

5.5.1.15.7 Summary of freshwater effect determinations for zinc

EPA has made the following determinations on the freshwater acute and chronic zinc criteria for the ESA listed species in the freshwater portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Likely to adversely affect** (exposure to water)

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Likely to adversely affect** (exposure to water)

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Likely to adversely affect** (exposure to water)

Chinook salmon – **Not likely to adversely affect**

Bull trout – **Likely to adversely affect** (exposure to water)

Marbled murrelet – **Not likely to adversely affect**

5.5.1.15.8 Considerations related to ESA consultation on Idaho's zinc criteria

ESA consultation on the same zinc criteria was completed in Idaho. NMFS (2014b, p. 193) summarized effects of salmonid exposure to zinc at criteria levels as follows: "Adverse effects were found at sub-criteria values in tests conducted at hardnesses less than 25 mg/L, a few other tests at moderately low hardness of 35 mg/L with the most sensitive size fish tested (Figure 2.4.10.2), and multiple tests reported by Hansen et al. (2002c) with rainbow trout. The preponderance of the information reviewed indicate that in waters with hardness less than about 25 mg/L as $CaCO_3$ the Idaho zinc criteria would not be sufficiently protective of listed Snake River salmon and steelhead if they were exposed at their most sensitive life stages. If alternatively, the current IDEQ zinc criteria were determined using the actual water hardness, instead of the assumed hardness of 25 mg/L, most of those data indicate that the criteria would then be sufficient to avoid harm in most of the studies reviewed. This would be sufficient to avoid population level effects to Snake River salmon and steelhead."

The Idaho WQS included a hardness floor of 25 mg/L $CaCO_3$ to be used in calculating the hardness-based metals criteria such as zinc. I.e., for waters with a hardness less than 25 mg/L, the hardness value of 25 mg/L would be used to calculate the zinc criteria. The Services did not consider this hardness floor to be protective, as the discussion of the zinc criteria reflected, and NMFS (2014b, p. 281) provided an RPA for Idaho to remove the hardness floor from its WQS and calculate hardness-based metals criteria using the actual hardness of waters, including when the hardness is below 25 mg/L $CaCO_3$. The Swinomish Tribe has not adopted a hardness floor, in effect implementing this Idaho RPA. When the zinc criteria are applied to the Swinomish Tribe's waters, criteria values must be calculated using the existing hardness level, including hardness values below 25 mg/L $CaCO_3$.[25]

USFWS (2015a) found that EPA approval of Idaho's zinc criteria (i.e., exposure of bull trout and its prey to zinc at criteria levels throughout its range in Idaho) would cause jeopardy to bull trout populations in Idaho. USFWS (2015a, p. 281) provided the following as an interim measure prior to full implementation of the RPA: "Until new criteria are adopted, a zone of passage sufficient to allow unimpeded passage of adult and juvenile bull trout and sturgeon must be maintained around any mixing zone for discharges

[25] For reference, hardness values identified for the Tribe's fresh waters were all higher than 25 mg/L (see Table 24).

that include zinc … If the regulatory mixing zone is limited to less than or equal to 25 percent of seasonal flow conditions, then a sufficient zone of passage is presumed to be present."

In its RPA analysis, USFWS (2015a, p. 282) indicated that "limiting mixing zone fractions to 1/4 (25 percent) of the receiving water discharge in flowing waters is effectively similar to reducing the criteria by about 0.25X (NMFS, 2014b). While adverse effects were identified at or below the existing criteria concentrations, few if any adverse effects to listed species or habitats would be expected at about 0.25X the criteria concentrations."

As discussed above for the cadmium and copper criteria, unlike Idaho, the Swinomish Tribe has prohibited the use of mixing zones for NPDES permits in fresh waters in WQS section 19-06.240(B)(7), and there are currently no permitted point source discharges to the Swinomish Tribe's fresh waters (section 4.2.2). If a NPDES permit were to be written for a fresh water point source in the future, the zinc criteria would be met at "end of pipe," and the effluent would be diluted upon entering the receiving water. In addition, EPA would prepare any NPDES permit, and ESA consultation would be completed if required.

Given the above restrictions, although exposure at the criteria level may adversely affect listed salmonids, the Tribe's prohibition of a mixing zone and EPA's responsibility to prepare NPDES permits and complete ESA consultation as required, are likely to limit the extent of any fresh waters under Tribal jurisdiction that may have zinc levels near the zinc criteria.

An additional consideration for bull trout is that only subadult/adult bull trout would be present in action area fresh waters, and this life stage is likely to be less sensitive than the earlier life stages that were used for the USFWS assessment. USFWS (2015a, p. 202) states:

> "This pattern of increasing sensitivity with increasing size/developmental stage of juveniles has been seen in other studies with salmonids (Hedtke et al. 1982; Mebane et al. 2012) and we interpret the bull trout results by discounting the results obtained with the life stages that are apparently more resistant to zinc and emphasizing the more sensitive life stage results. This apparent pattern of increasing sensitivity to zinc toxicity with increasing fish size likely only holds within the juvenile stage. At some point as fish age and grow, they probably become more resistant to metals. No data supporting this conclusion are available for the bull trout, but with steelhead and Chinook salmon, 96-hr LC50s with smolts of 38 to 68g in weight were >7 times higher than those obtained with fish in the swim-up stage that were <1g in weight (Chapman 1978)."

USFWS (2015a) bases its jeopardy determination for bull trout in Idaho on toxicity studies that were completed using the most sensitive life stages, and these are not found in action area waters. Table 9 of the Idaho BiOp (page 203) provides the weights of the most sensitive juvenile bull trout tested by Hansen (J. A. Hansen, Lipton, & Welsh, 1999; J. A. Hansen, P. G. Welsh, J. Lipton, D. Cacela, & A. D. Dailey, 2002) and LC50 values from Hansen et al. (2002). The lowest LC_{50} for zinc that Hansen et al. (2002) report is 30 µg/L, while the chronic zinc criterion for the test water is 43 µg/L. The subadult and adult bull trout in action area fresh waters can be extrapolated to be about 7 times less sensitive to zinc as indicated above, with LC_{50} values several times higher than the Tribe's criteria level. On this basis, exposure to zinc at the criteria level would not be likely to adversely affect bull trout in the action area.

EPA is currently collaborating with eight metals associations in a cooperative research and development association (CRADA) to develop new EPA aquatic life criteria recommendations for a number of metals, including zinc, consistent with the USFWS's RPA relative to bull trout. EPA expects states and tribes to adopt new criteria recommendations during the next triennial review after the recommendations are issued.

5.5.2 Dissolved oxygen, pH, and temperature

5.5.2.1 Dissolved oxygen (freshwater)

The Swinomish DO criterion for fresh water is 9.5 mg/L, expressed as a 1-day minimum. This is the same as the State of Washington's DO criterion for the core summer salmonid habitat and char spawning and rearing designated uses.

EPA completed ESA consultation on the State of Washington's freshwater criteria for DO, including the 9.5 mg/L criterion which applies to all freshwater bodies in the action area, in 2008. This consultation addressed all of the species that could be present in freshwater bodies in the action area.

EPA's effects analysis for the 9.5 mg/L DO criterion appears in section 5.H.12 of the Biological Evaluation of the Revised Washington Water Quality Standards (USEPA, 2007b). EPA concluded that the 9.5 mg/L criterion was LAA Puget Sound Chinook salmon, Hood Canal summer run salmon, and Puget Sound steelhead.

The EPA (2007) LAA finding for the 9.5 mg/L criterion in Washington was based on potential effects to salmon embryos and larvae, since it would not necessarily ensure that DO concentrations were at least 11 mg/L, (which is equivalent to 8.0 mg/L intergravel DO per EPA (1986)) or 95% saturation during salmon incubation periods. EPA (2007) stated that the Washington action had no effect on bull trout because bull trout spawning does not occur in waters where the DO criteria had changed in the subject action. USFWS (2008) evaluated the effects of Washington's DO criteria on bull trout and found that approval of the 9.5 mg/L criterion would have adverse effects on bull trout. Similar to EPA's effects analysis for anadromous salmon, USFWS's finding of adverse effects was based on a failure of the criterion to ensure adequate intergravel DO to protect early life stages of bull trout. However, salmonid spawning and incubation do not occur in freshwaters within the action area. Spawning and incubation are neither designated nor existing uses and it is therefore not necessary for the Tribe's DO criteria to protect those life stages.

EPA (1986) states that salmonid life stages other than embryos and larvae will have no production impairment at a DO concentration of 8.0 mg/L. Since salmonid spawning and incubation does not occur in freshwaters in the action area and the Tribe's criterion is more protective than the concentration resulting in no production impairment of salmonid life stages other than embryos and larvae, the 1-day minimum DO criterion of 9.5 mg/L for the "Salmon and Trout fish use" adopted by the Tribe is **not likely to adversely affect** Puget Sound Chinook salmon, Puget Sound steelhead, and coastal/Puget Sound bull trout in fresh water.

5.5.2.2 pH (freshwater)

The Swinomish pH criterion for fresh water is: May not be less than 6.5 or greater than 8.5, with a human-caused variation within the above range of less than 0.2 units. The Tribe's pH criterion is

identical to Washington State's criterion for the core summer salmonid habitat and char spawning and rearing uses (WAC 173-201A-200(1)(g)).

The life stages of fish most sensitive to effects from pH are spawning, egg incubation, and alevin/fry development. These life stages of salmon and bull trout are not present in the action area.

EPA's recommended criteria for pH for freshwater aquatic life are a range of 6.5 - 9.0 standard units. The Swinomish Tribe's pH criteria are more protective than the recommended criteria. A review of the effects of pH on fresh water fish published by the European Advisory Commission states that although there is no definite range at which a fishery is unharmed and outside which is damaged, the pH range which is not directly lethal to fish is 5.0 - 9.0 (EPA, 1986). Although pH in the range of 5.0 - 6.5 is unlikely to be harmful to fish, it may be harmful if free CO_2 concentrations are greater than 20 ppm or if the water contains iron salts which are precipitated as ferric hydroxide.

In the development of EPA's (1976, 1986) criteria (6.5 - 9.0, freshwater chronic exposure), two bioassay references on freshwater fish cited by EPA showed a lower limit of about 6.5 for normal development (EIFAC, 1969; Mount 1973, in EPA, 1986). Vulnerable life stages of chinook salmon are sensitive to pH levels below 6.5 and possibly at pH levels greater than 9.0 ((Marshall, 1992)). Adult salmonids are at least as sensitive as most other fish to low pH; these species include rainbow, brook and brown trout, and chinook salmon ((ODEQ, 1995)). The critical value of pH for rainbow trout presence, at the low end, is about 5.5 (Baker et al., 1990). Considering the salmonid food base, some insect larvae including those of the mayfly, stonefly, and caddis fly are sensitive to pH levels in the range of 5.5 to 6.0 (ODEQ, 1995).

At the higher end of the pH scale, even less is known regarding effects on fish. In EPA's review for water quality criteria development, the upper limit of 9.0 for the recommended criteria was obtained from only one reference (EIFAC, 1969). The larvae of aquatic insects were apparently more tolerant than fish. No recent data exist, but studies conducted earlier in the early 1900's show salmonids, including both trout and salmon species, to be sensitive to pH levels in the range of 9.2 to 9.7, depending on the life stage (ODEQ, 1995).

Non-salmonid fishes are, with some exceptions, more tolerant of high pH, with sensitivity appearing at or over pH 10 for most species tested (EIFAC, 1969). Levels of pH greater than 9.0 may adversely affect benthic invertebrate populations, thereby altering the food base for salmonids. A pH of 9.0 seems to be the cutoff for the start of noticeable adverse effects for some species of salmonids and invertebrates. The Tribe's criterion for pH provides an upper limit of 8.5, as opposed to the recommended criterion which has an upper bound of 9.0. In addition, the Tribe's WQS limit changes in pH due to anthropogenic activities to no more than 0.2 pH units.

Based on the above information, the criterion for pH is **not likely to adversely affect** Puget Sound Chinook salmon, Puget Sound steelhead, and coastal/Puget Sound bull trout in fresh water.

5.5.2.3 Temperature (freshwater)

Temperature is a critical water quality parameter for health and survival of salmonids in the Pacific Northwest (USEPA, 2003). The Swinomish Tribe's water quality criterion for freshwater is a 7-day average of daily maximum temperatures (7-DADMax) of 16 °C. This is the same as the State of Washington's criterion for the core summer salmonid habitat use category (WAC 173-201A-200(1)(c)).

The Tribe's freshwater temperature criterion is expressed as a 7-DADMax. The 7-DADMax is the measure of the maximum temperatures in a stream, averaged over a seven-day period. This metric integrates more information into one value, relative to an instantaneous maximum. The 7-DADMax metric is not overly influenced by the maximum temperature of any single day as it reflects an average temperature that fish are exposed to over a week-long period. This metric is recommended by EPA for water temperature criteria because it protects aquatic life against acute effects (e.g. lethality) and incorporates daily maximum temperatures (USEPA, 2003). The metric is also protective of chronic effects to aquatic life (e.g. reduced growth) because it describes the thermal exposure over 7 days.

EPA completed ESA consultation on the State of Washington's freshwater criteria for temperature, including the 16 °C 7-DADMax criterion which applies to all freshwater bodies in the action area, in 2008. This consultation addressed all of the species that could be present in freshwater bodies in the action area.

EPA's effects analysis for the 16 °C 7-DADMax criterion appears in section 5.H.4 of the Biological Evaluation of the Revised Washington Water Quality Standards (EPA 2007). As explained in EPA (2007), the 16 °C 7-DADMax criterion is designed to:

- Protect juvenile salmon and steelhead from lethal temperatures [23 to 26°C (73 to 79°F) constant];
- Provide conditions during the period of summer maximum temperature and other times of the year that are in the optimal range when food is limited for juvenile growth [10 to 16°C (50 to 61F) constant];
- Protect against temperature-induced elevated disease rates [14 to 17°C (57 to 63F) constant];
- Provide temperatures that juvenile salmon and trout prefer, as demonstrated by studies indicating fish in high densities at these temperatures [10 to 17°C (50 to 63F) constant or less than 18C (64F) 7DADM];
- Protect salmon and steelhead from competitive disadvantage with cool and warm water species which can occur when average temperatures are greater than 15°C and maximum temperatures exceed 17-18°C (see Ecology 2002 pp. 67);
- Provide conditions during the period of summer maximum temperatures that protect adult and sub-adult foraging and migration [less than 15°C] (see USEPA 2003 pg 27; and Bull Trout Peer Review 2002); and
- Provide conditions that protect chinook salmon that are holding over the summer (see USEPA 2003).

The Swinomish Indian Tribal Community has provided EPA with hourly temperature data for four freshwater streams. Data are summarized in Tables 68 to 71. These data show that diurnal variability is relatively small, with differences between the 7-DADMax temperature and the weekly mean temperature generally less than 1 °C. Diurnal variability is larger in Lone Tree Creek, which is an intermittent stream, which has a maximum difference between the 7-DADMax and the weekly mean of 5.78 °C.

Based on optimal growth studies, EPA concluded that when the mean temperature is above the optimal growth temperature for salmon, the mid-point between the mean and maximum temperatures is the

“equivalent” constant temperature. The “equivalent” constant temperature is the relevant value used in assessing the effects of temperature in salmon and trout streams in the PNW (EPA 2007).

The 16°C 7-DADMax is protective of the Tribe’s “Salmon and Trout Fish Use” because it is within the range of temperatures that are used by the salmonid life stages observed in freshwaters in the action area, including adult holding, summer rearing, and foraging by adult and sub-adult salmonids. Salmonid spawning and incubation do not occur in freshwaters within the action area. Spawning and incubation are neither designated nor existing uses and it is therefore not necessary for the Tribe’s temperature criteria to protect those life stages.

The 16 °C 7-DADMax criterion applies year-round and throughout all freshwater bodies in the action area. Assuming this criterion is met at the warmest times of the year and in the warmest parts of the freshwater bodies in the action area (see section 4.3.5.2), then 7-DADMax temperatures will be cooler than 16 °C most of the time.

For these reasons stated above, EPA has determined that its approval of the 16 °C 7-DADMax criterion is **not likely to adversely affect** Puget Sound Chinook salmon, Puget Sound steelhead.

Information to substantiate the thermal needs of char during the sub-adult foraging and migration life history phases was limited (McCullough et al. 2001). Based on this limited information it is believed that bull trout temperature preferences in mainstem systems is <15°C. It is likely but unknown that the 16°C 7-DADMax criterion would be protective of sympatric char species. Assuming char species would be unlikely to be exposed to average temperatures above 15°C and would likely find and use coldwater refugia due to the significant influence of groundwater in streams within the action area (Dorst 1979; Gendaszek and Opatz, 2013) the application of the 16°C criterion would result in substantial area of colder water useable by char. Also, the upstream contributing stream reaches that the char would be migrating to would have colder temperatures because of higher elevation. For these reasons, EPA has determined that its approval of the 16 °C 7-DADMax criterion is **not likely to adversely affect** coastal/Puget Sound bull trout.

5.5.2.4 Aquatic-dependent wildlife effects analysis for freshwater DO, pH, and temperature criteria

The non-fish species that have some possibility of exposure route are the marbled murrelet and killer whale.

Unlike the listed anadromous and freshwater fish species addressed above in this effects analysis, these species would not be directly physically exposed to the water DO concentration, pH, or temperatures that are regulated by the Swinomish WQS. However, fish species that would be exposed (including the listed species addressed above) are a substantial portion of the prey base for these two species.

Unlike toxic chemicals or metals that accumulate in the tissue of fish and then accumulate in the tissue of the predator species, DO, pH, and temperature exposure to prey species will have no direct effect on the predator species. The only possible effect to these two predator species is if the approval of the water quality criteria resulted in effects to the fish populations resulting in reduced numbers of prey items to bird and mammal species.

All of the above effect determinations found that the DO, pH, and temperature criteria are NLAA threatened and endangered fish species. None of these standards are a substantial deviation from water

quality limits derived from the literature that are found to be protective of salmonids. Thus, any possible adverse effects would not result in suppression of entire populations that would reduce the prey base for the listed bird and marine mammal species.

Therefore, the EPA has determined that its approval of the Swinomish Tribe's DO, pH, and temperature criteria for fresh waters is **not likely to adversely affect** marbled murrelet or killer whales.

5.6 Saltwater Criteria Assessments

5.6.1 Toxic substances

Each chemical section below, for the assessment category 4 criteria, contains the acute and/or chronic effects assessments, depending on whether each chemical has an acute criterion, a chronic criterion, or both (section 2.2.1, Table 6). Tables 2 and 3 summarize the results of all assessments.

5.6.1.1 Ammonia – Marine

This section is reserved. The assessment of the Tribe's ammonia criteria for Chinook salmon, steelhead, chum salmon, bull trout, bocaccio, yelloweye rockfish, and marbled murrelet is in progress and will be provided as an addendum to the BE when the assessment has been completed.

5.6.1.1.1 The Swinomish Tribe's ammonia criteria

The Tribe's ammonia criteria, as submitted to EPA for CWA approval, are provided in footnote k to Table 15, *Water Quality Criteria for Toxic Pollutants*, as follows:

> k. To develop total ammonia criteria use Ambient Water Quality Criteria for Ammonia (Saltwater) – 1989 (EPA 440/5-88-04). Ammonia criteria for saltwater are for unionized ammonia. Unionized ammonia is the most toxic form of ammonia to aquatic life. In saltwater, the fraction of the total ammonia that is unionized depends mainly on the pH, temperature and salinity of the water. For implementation purposes, the unionized ammonia criteria are generally converted to total ammonia to be consistent with standard water quality monitoring methods.

This footnote incorporates EPA's CWA section 304(a) criteria for ammonia (EPA 1989) by reference, as follows:

Acute (CMC) criterion = 0.233 mg/L

Chronic (CCC) criterion = 0.035 mg/L

CAS ID 7664-41-7

The unionized ammonia criteria may be implemented as total ammonia. Formulas and a table of calculated ammonia criteria for a range of temperature, pH, and salinity conditions are provided in EPA (1989), and the Tribe's footnote k to Table 15 also incorporates these conversion formulas into the Tribe's WQS by reference.

5.6.1.2 Arsenic – Marine

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = 69 µg/L

Chronic (CCC) criterion = 36 µg/L

CAS ID 7440-38-2

5.6.1.2.1 Introduction to arsenic

Arsenic in marine systems is primarily found in one of four chemical forms. Two forms are inorganic, trivalent arsenic (As^{+3}, arsenite) and pentavalent arsenic (As^{+5}, arsenate). The other two forms are organic, methylarsonic acid (MAA) and dimethylarsinic acid (DMAA). The two inorganic forms of arsenic are more toxic to aquatic life than are the two organic forms of arsenic. Arsenite is more toxic to marine aquatic life than is arsenate. In aerobic waters at circumneutral pH, As^{+3} is slowly oxidized to the less toxic As^{+5}: this reaction can be reversed in anaerobic waters.

According to EPA (USEPA, 2022) and the Swinomish Tribe's WQS, the arsenic marine water quality criteria were derived from toxicity data for the more toxic trivalent arsenic but are applied to the total arsenic concentration in marine waters. Both the acute and chronic marine arsenic criteria are expressed in terms of the dissolved metal in the water column.

Chemical speciation of arsenic, which affects its toxicity, is much more studied in marine species and food webs than it has been in freshwater systems. Neff (Neff, 1997) has reviewed the ecotoxicology of arsenic in marine systems. Many of the organic forms of arsenic found in marine invertebrates and fish are present due to the metabolic transformation of arsenite by marine algae to a number of organoarsenicals, which are then either released to surface waters or transferred to higher trophic level organisms through food web transfer. Organoarsenicals have a low order of toxicity to invertebrates and fish relative to arsenite and arsenate. The organoarsenicals tend to be rapidly eliminated from the tissues of invertebrates and fish, contributing to the low toxicity of arsenic in invertebrates and fish relative to the toxicity of arsenic to algae. Arsenic toxicity in marine species follows the order of algae > invertebrates > fish.

5.6.1.2.2 Water-column exposure toxicity data - acute criterion for arsenic in marine systems

The numeric values for both the acute and chronic marine arsenic criteria have not been changed since the publication of EPA (USEPA, 1985a). The EPA (USEPA, 1996a) update of a number of criteria did not report any marine arsenic toxicity data not described in the EPA (USEPA, 1985a) arsenic criteria document.

The marine acute toxicity data presented in the EPA (USEPA, 1985a) arsenic criteria document allows for the calculation of 11 genus mean acute values (GMAVs), of which only three are for marine fish. The three marine fish GMAVs are 16,030 µg/L for Atlantic silverside (*Menidia menidia*), 14,950 µg/L for fourspine stickleback (*Apeltes quadracus*) and 12,700 µg/L for sheepshead minnow (*Cyprinodon variegatus*). These three fish are the most arsenic tolerant of the 11 marine species with GMAVs. None of these three fish species are in either the family *Salmonidae* or *Sebastidae*, the two families containing all of the ESA listed fish species in the action area. By contrast, the most sensitive marine invertebrate species to arsenic was the Dungeness crab (*Cancer magister*) with a GMAV of 232 µg/L.

5.6.1.2.3 Additional marine acute toxicity data for arsenic published since 1985

Shaw et al. (J. R. Shaw et al., 2007; T. Shaw, 1979) reported a 96 hour LC_{50} of 29,247 µg/L for mummichog (*Fundulus heteroclitus*). Mayer (Mayer Jr., 1987) reported a 48 hour LC_{50} > 30,000 µg/L for the closely related longnose killifish (*Fundulus similis*). Although a shorter exposure duration than the 96 hour LC_{50}s normally used in this effects assessment, the longnose killifish LC_{50} implies that a 1-hour LC_{50} (a 1-hour exposure duration is part of the definition of an acute criterion) for this species would be as high or higher a concentration than 30,000 µg/L, meaning the marine acute arsenic criterion of 69 µg/L would be protective from short term arsenic toxicity to the longnose killifish.

5.6.1.2.4 Effects assessment of acute marine arsenic criterion on listed species

As no empirical acute toxicity data are available for any of the marine ESA listed fish species within the action area, the effects assessment concentration has been calculated from a species sensitivity distribution yielding the 5th percentile of the available acute toxicity data for marine fish species. The 5th percentile of the SSD for 96 hour LC_{50}s is 8862 µg/L. Dividing this 96 hour LC_{50} by 2.27 to estimate an LC_{low} results in an acute effects assessment concentration of 3904 µg/L. The calculated marine acute effects assessment concentrations for the ESA listed fish species without empirical toxicity data is well in excess of the marine acute arsenic criterion of 69 µg/L. This indicates that exposure at the level of the marine acute arsenic criterion is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio or yelloweye rockfish.

5.6.1.2.5 Water-column exposure toxicity data - chronic criterion for arsenic in marine systems

The EPA (USEPA, 1985a, 1996a) aquatic life criteria for arsenic contain no empirical chronic toxicity data for any fish species.

5.6.1.2.6 Additional chronic marine toxicity data for arsenic published since 1985

EPA has identified no chronic duration toxicity studies with arsenic for any marine fish species.

5.6.1.2.7 Effects assessment of chronic marine arsenic criterion on listed species

As no empirical chronic arsenic toxicity data for any marine fish species is available, EPA has applied the arsenic acute-chronic ratio (ACR) of 3.904 to the empirical 96 hour LC_{50} values for the four marine fish species with acceptable quality acute toxicity data. Application of the ACR to 96 hour LC_{50} data provides an estimate of a chronic no observed effect concentration (NOEC) for each species with acute toxicity data. By fitting a species sensitivity distribution to the estimated chronic NOEC data, the 5th percentile of the fitted SSD can be used as an estimate of the chronic NOEC for all fish species, including the ESA listed species within the action area.

The 5th percentile of the fitted chronic NOEC SSD using the calculated chronic NOECs for mummichog (7492 µg/L), Atlantic silverside (4196 µg/L), fourspine stickleback (3829 µg/L) and sheepshead minnow (3253 µg/L) is 2270 µg/L, with 95% confidence limits between 975 – 5286 µg/L. The chronic NOEC estimate of 2270 µg/L for all fish species is higher than the chronic marine arsenic criterion of 36 µg/L. This indicates that exposure at the level of the marine chronic arsenic criterion is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio or yelloweye rockfish.

5.6.1.2.8 Summary of effect determinations for marine arsenic criteria

EPA has made the following determinations on the acute and chronic arsenic criteria for the ESA listed species in the marine portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.6.1.3 Carbaryl – Marine

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = 1.6 µg/L

No chronic (CCC) criterion adopted

CAS ID 63-25-2

5.6.1.3.1 Water-column exposure toxicity data - acute criterion for carbaryl in marine waters

EPA (USEPA, 2012) listed two marine acute fish toxicity tests with carbaryl that were used in the derivation of the marine acute criterion. The most sensitive of the two fish species tested was the sheepshead minnow (*Cyprinodon variegatus*) with a mean 96 hour LC_{50} of 2600 µg/L. This mean 96 hour LC_{50} was based on studies with acceptable data quality from Surprenant (D. Surprenant, 1985) and Lintott (Lintott, 1992b). The second fish species tested was the threespine stickleback (*Gasterosteus aculeatus*), with a 96 hour LC_{50} of 3990 µg/L (Katz, 1961). Neither the sheepshead minnow nor the threespine stickleback are members of the family *Salmonidae* (salmonids) or *Sebastidae* (rockfishes), the two fish families containing all of the ESA listed fish species found in the marine portions of the action area.

Acute toxicity of carbaryl to a third fish species, striped bass (*Morone saxatilis*) was reported by Korn and Earnest (Korn & Earnest, 1974). The flow-through bioassays were 4-day tests using proportional diluters and carbaryl had a 96-hr LC_{50} of 1,000 µg/l. Although this 96 hour LC_{50} shows striped bass to be more sensitive to carbaryl than the sheepshead minnow and the threespine stickleback, Korn and Earnest (Korn & Earnest, 1974) was not used to assess ESA listed species in this BE because control survival was not reported.

The remainder of the marine acute data used in EPA (USEPA, 2012) to derive the acute carbaryl criterion are from crustaceans and bivalves. The six most sensitive genera of the 11 used to derive the EPA (USEPA, 2012) acute carbaryl marine criterion were all invertebrates. A mysid (*Americamysis sp.*) was the most sensitive marine species tested, with a SMAV of 7.188 µg/L, based on the studies of Thursby and Champlin (G. B. Thursby & Champlin, 1991a, 1991b), Nimmo et al. (Nimmo, Hamaker, Matthews, & Moore, 1981) and Lintott (Lintott, 1992a).

As is the case for carbaryl toxicity to freshwater species, fish are more tolerant of the toxic effects of carbaryl than are invertebrates (see the assessment for the freshwater carbaryl criteria, section 5.5.1.3). This is likely due in part to the intended uses of carbaryl to control insects and molluscs.

5.6.1.3.2 Additional marine water acute toxicity data for carbaryl published since 2012

EPA has identified no additional marine acute toxicity test results with fish meeting data quality requirements for use in this BE since the publication of the carbaryl aquatic life criteria (USEPA, 2012).

5.6.1.3.3 Effects assessment of acute marine carbaryl criterion on listed species

The lowest 96 hour LC_{50} acute effect threshold concentration for marine fish species based on the available studies meeting data quality requirements for use in criteria derivation is the 2600 µg/L value for sheepshead minnows reported by Lintott (1992b). To estimate the acute NOEC for all fish, this LC_{50} is divided by 2.27; therefore, 2600 µg/L / 2.27 = 1145 µg/L. This value is higher than the carbaryl marine acute criterion of 1.6 µg/L. If the minimum acute effect concentration for all ESA listed fish species in the marine portions of the action area is greater than or equal to 1145 µg/L, exposure at the level of the carbaryl marine acute criterion of 1.6 µg/L is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio and yelloweye rockfish.

If it is assumed that the toxicity of carbaryl to fish is comparable in freshwater and marine waters, the effect determinations for carbaryl in the freshwater portion of this BE (section 5.5.1.3) further support our conclusions for marine fish species. The lowest GMAV for any fish species exposed to carbaryl in freshwater is 889 µg/L, the GMAV for the genus *Salmo* in the family *Salmonidae*. This GMAV is the

geometric mean of 96 hour LC_{50}s for Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*) of 1129 and 700 µg/L, respectively. The freshwater GMAVs for five members of the genus *Oncorhynchus* and two members of the genus *Salvelinus* are 1810 and 1269 µg/L, respectively. All three of the freshwater GMAVs for salmonids are higher than the marine acute criterion of 1.6 µg/L. The freshwater GMAVs for the genera *Oncorhynchus* and *Salvelinus* cover all of the ESA listed salmonid species within the marine portions of the action area.

5.6.1.3.4 Summary of effect determinations for marine carbaryl criterion

EPA has made the following determinations on the acute carbaryl criterion for the listed species in the marine portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.6.1.4 Chlorine – Marine

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = 13 µg/L

Chronic (CCC) criterion = 7.5 µg/L

5.6.1.4.1 Introduction

Introductory and contextual information for chlorine is provided in the chlorine criteria assessment for freshwater, section 5.5.1.4.1. The complete list of Chemical Abstracts Service identification numbers (CAS IDs) used in our literature search for toxic effects of chlorine on aquatic life are presented in Table 90.

The EPA aquatic life criteria for chlorine describes the toxicity of total residual chlorine (TRC), which is the combined concentration of different chemical forms of chlorine able to react with other substances, or which can interconvert among each other. Within the literature, TRC is generally synonymous with reactive chlorine (RC), combined residual chlorine (CRC), and total available chlorine (TAC).

Total residual chlorine includes free available chlorine (FAC; hypochlorous acid [HOCl] and the hypochlorite ion [OCl^-]; also referred to as free residual chlorine [FRC]) and combined available chlorine (CAC; organic and inorganic chloramines [NH_2Cl or monochloramine, $NHCl_2$ or dichloramine, and NCl_3 or nitrogen trichloride]). Chloramines are also often termed N-chloramides.

In salt water, where bromine is also present, the addition of chlorine also produces hypobromous acid (HOBr), hypobromous ion (OBr^-), and bromamines. Collectively, these products are referred to as "chlorine-produced oxidants," and chlorine toxicity data is therefore referred to as total residual chlorine (TRC) or chlorine-produced oxidants (CPO) (USEPA, 1985).

As described section 5.5.1.4.1 for the Tribe's chlorine criteria for freshwater, the toxicity assessment should focus on surface water concentrations of chlorine that affect the respiratory surfaces of aquatic species. Dietary ingestion of and dermal contact with chlorine are insignificant to non-existent exposure routes for both fish and their prey.

5.6.1.4.2 Water-column exposure toxicity data - acute criterion for chlorine in marine systems

The marine acute toxicity data presented in the EPA (USEPA, 1985b) chlorine criteria document allows for the calculation of 21 genus mean acute values (GMAVs), of which 11 are for marine fish. These values range from 37 to 270 µg/L, with coho salmon (*Oncorhynchus kisutch*), tidewater silverside (*Menidia peninsulae)*, and Atlantic silverside (*Menidia menidia*) being the most sensitive to CPO with Species Mean Acute Values of 47, 37, and 54 µg/L, respectively (Buckley, 1976; Larry R. Goodman, Middaugh, Hansen, Higdon, & Cripe, 1983; Roberts, Diaz, Bender, & Huggett, 1975; Thatcher, 1978). Overall, fish are more sensitive to CPO than invertebrates, but the two most sensitive species are a copepod, *Acartia tonsa* and the Eastern oyster (*Crassostera virginica*) with SMAVs of 29 and 26 µg/L, respectively (USEPA, 1985b). The most tolerant invertebrate species is the grass shrimp (*Palaemonetes pugio*), with a SMAV of 220 µg/L (Roberts Jr, Diaz, Bender, & Huggett, 1975).

5.6.1.4.3 Additional marine acute toxicity data for chlorine published since 1985

For inland silversides (*Menidia beryllina*), reported a 96-h LC_{50} of 128 µg/L, and Fisher, Burton, Yonkos, Turley, and Ziegler (1999) published a 96-h LC_{50} value of 65 µg/L. Fisher et al. (1999); G. Holland (1960); (G. A. Holland, J.E. Lasater, E.D. Neumann and W.E. Eldridge, 1960) reported a 96-h LC_{50} for juvenile pink salmon (*Oncorhynchus gorbusha*) of 50 µg/L. For mollusks and invertebrates, (Fisher et al., 1999) reported a 96-h LC_{50} value of 62 µg/L in *Mysidopsis bahia*.

5.6.1.4.4 Effects assessment of acute marine chlorine criterion on listed species

Acute toxicity data are available for 5 marine fish species and include the 96-hour LC_{50}s for juvenile Pacific herring (*Clupea pallasii*) (65 µg/L), juvenile coho salmon (32 µg/L), Atlantic silverside (37 µg/L), juvenile tidewater silverside (54 µg/L), threespine stickleback (*Gasterosteus aculeatus*) (167 µg/L), northern pipefish (*Syngnathus fuscus*) (270 µg/L), shiner perch (*Cymatogaster aggregata*) (71 µg/L), Pacific sand lance (*Ammodytes hexapterus*) (82 µg/L), juvenile naked goby (*Gobiosoma bosc*) (80 µg/L), and juvenile English sole (*Parophrys vetulus*) (73 µg/L). Additional data includes 96-hour LC_{50}s for 8-day old inland silversides (65 µg/L), and larval striped bass (*Morone saxatilis*) (140 µg/L).

WebICE was used to estimate LC_{50} values for the ESA listed salmonid species within the marine portions of the action area, because coho salmon is the only genus *Oncorhynchus* or family *Salmonidae* species with empirical marine acute toxicity data for chlorine. With coho salmon as a genus surrogate with an

empirical 96-hour LC_{50} of 32 µg/L, the ICE predicted 96-hour LC_{50} for rainbow trout is 22.1 µg/L and for Chinook salmon is 17.17 µg/L. At a family level, with coho salmon as a surrogate, the predicted 96-hour LC_{50} for *Oncorhynchus,* which includes chum salmon, is 23.09 µg/L and for *Salvelinus*, which includes bull trout, is 18.33 µg/L. Dividing all of the ICE estimated salmonid LC_{50}s by 2.27 estimated LC_{low} concentrations as the assessment effect concentrations of chlorine. The acute assessment effects concentrations were 9.7 µg/L for rainbow trout (steelhead), 7.6 µg/L for Chinook salmon, 10 µg/L for chum salmon and 8.1 µg/L for bull trout. All of these acute assessment effect concentrations are lower than the marine acute criterion of 13 µg/L for chlorine, resulting in a conclusion that exposure at the level of the marine acute chlorine criterion is likely to adversely affect rainbow trout, Chinook salmon, chum salmon and bull trout.

No empirical chlorine toxicity data exists for either bocaccio or yelloweye rockfish. Nor are there any empirical toxicity data for species taxonomically related to rockfish that permit use of ICE to estimate 96 hour LC_{50}s for the two rockfish species. This absence of empirical data requires us to use a species sensitivity distribution of acute chlorine toxicity data to estimate a 96 hour LC_{50} for the two rockfish species.

The 5th percentile of the SSD for marine acute chlorine 96-hour LC_{50}s is 28.5 µg/L. Dividing this 96-hour LC_{50} by 2.27 to estimate an LC_{low} results in an acute effects assessment concentration of 13 µg/L (12.56 µg/L to four significant figures). The SSD calculated marine acute effects assessment concentrations for the ESA listed fish species equals the marine acute chlorine criterion of 13 µg/L (12.62 µg/L to the four significant figures used in aquatic life criteria calculations (Charles E Stephan et al., 1985). In this BE, an assessment effects concentration equal to or lower than a criterion is a basis for a likely to adversely affect determination. Water quality criteria values are calculated to four significant figures, then reported to two significant figures in criteria documents and tables. The chlorine marine acute criterion was calculated as 12.62 µg/L in EPA (USEPA, 1985b). To four significant figures the 5th percentile of the marine acute chlorine SSD using data updated from EPA (USEPA, 1985b) is 12.56 µg/L, marginally lower than the criterion to four significant figures. Both of these calculations lead to a determination that exposure at the level of the marine acute chlorine criterion is likely to adversely affect bocaccio and yelloweye rockfish.

5.6.1.4.5 Effects assessment of chronic marine chlorine criterion on listed species

There is no empirical chronic chlorine marine toxicity data available for fish that fit the criterion used in this evaluation. The effects assessment concentration calculated from the species sensitivity distribution (SSD) yielding a value of 28.5 µg/L for the 5th percentile of the available acute toxicity data for marine fish species. The 1985 EPA chlorine guidance document presented a final acute-chronic ratio of 3.345. Using this value, the final chronic value is 28.5/3.345 = 8.5 µg/L. This value is higher than the chronic marine chlorine criterion of 7.5 µg/L; therefore, chlorine exposure at the level of the chronic criterion is **not likely to adversely affect** rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio or yelloweye rockfish.

5.6.1.4.6 Summary of effect determinations for marine chlorine criteria

EPA has made the following determinations on the acute and chronic chlorine criteria for the ESA listed species in the marine portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above, and the indirect effects analysis for prey availability provided in section 5.8 (the multiple routes of exposure analysis was

not relevant as indicated in section 5.7) . The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Likely to adversely affect** (exposure to water)

Chinook salmon – **Likely to adversely affect** (exposure to water)

Chum salmon – **Likely to adversely affect** (exposure to water)

Bull trout – **Likely to adversely affect** (exposure to water)

Bocaccio – **Likely to adversely affect** (exposure to water)

Yelloweye rockfish – **Likely to adversely affect** (exposure to water)

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.6.1.5 Chlorpyrifos – Marine

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = 0.011 µg/L

Chronic (CCC) criterion = 0.0056 µg/L

CAS ID – 2921-88-2

5.6.1.5.1 Introduction

Chlorpyrifos is an organophosphate insecticide historically used on a wide variety of crops including apples, oranges, strawberries, corn, wheat, citrus, and others. Its registration calls for language on product labels stating that chlorpyrifos should not be directly applied to water. Its mode of toxic action is inhibition of acetylcholinesterase (AChE) activity. AChE inhibition adversely affects the nervous

system, and can lead to convulsions, muscular paralysis, bronchial constriction, depressed respiration and ultimately death.

5.6.1.5.2 Water-column exposure toxicity data - acute criterion for chlorpyrifos in marine water

The EPA (USEPA, 1986c) chlorpyrifos aquatic life criterion document listed 12 genera of marine species with acute toxicity data meeting all EPA quality assurance requirements for use in deriving water quality criteria. In general, invertebrate species are more sensitive to chlorpyrifos than are fish. Of the 12 genera with data of acceptable quality for acute criterion derivation, the four most sensitive genera are all invertebrates, as is the most tolerant of the 12 genera. The remaining seven genera are fish: Striped bass (*Morone saxatilis*); California grunion (*Leuresthes tenuis*); inland, Atlantic and tidewater silversides (*Menidia beryllina, M. menidia and M. peninsulae*); mummichog and longnose killifish (*Fundulus heteroclitus* and *F. similis*); striped mullet (*Mugil cephalus*); sheepshead minnow (*Cyprinodon variegatus*) and Gulf toadfish (*Opsanus beta*).

None of the seven ESA listed fish species within the marine portions of the action area have empirical acute toxicity data available in the chlorpyrifos criterion document (USEPA, 1986c). Nor are there taxonomic relatives of the ESA listed salmonid or rockfish species at the genus (*Oncorhynchus*, *Salvelinus*, *Sebastes*), family (*Salmonidae*, *Sebastidae*) or order (*Salmoniformes*, *Scorpaeniformes*) level with empirical acute toxicity data in the chlorpyrifos criterion document (USEPA, 1986c). The nearest taxonomic relatives of the ESA listed fish species under assessment are members of the class *Actinopterygii* (ray-finned fishes). *Actinopterygii* account for nearly 99% of the > 33,000 known fish species.

The most acutely sensitive fish GMAV in EPA (USEPA, 1986c) is *Morone*, based on the 96 hour SMAV of 0.58 µg/L chlorpyrifos to striped bass (Earnest, 1970; Korn & Earnest, 1974). Unfortunately, neither Korn and Earnest (Korn & Earnest, 1974) or Earnest (Earnest, 1970) report control survival, thus making these two studies and the striped bass SMAV and GMAV unsuitable for use in EPA water quality criteria development. The next most acutely sensitive fish species in EPA (USEPA, 1986c), the California grunion with a 96 hour SMAV of 1.068 µg/L, does meet the current quality assurance requirements for use in criteria development. The most tolerant fish species was the Gulf toadfish (*Opsanus beta*) with a 96 hour LC_{50} of 520 µg/L (D. J. Hansen, Goodman, Cripe, & Macauley, 1986).

By comparison, the most sensitive genus of any marine species to chlorpyrifos listed in EPA (USEPA, 1986c) is the opossum shrimp *Americamysis bahia* (formerly *Mysidopsis bahia*). The 96 hour GMAV and SMAV for *Americamysis bahia* is 0.035 µg/L, based on the work of Schimmel et al. (S. C. Schimmel, Garnas, Patrick, & Moore, 1983).

5.6.1.5.3 Additional marine water acute toxicity data for chlorpyrifos published since 1986

EPA (USEPA, 2017a) prepared a national biological evaluation for chlorpyrifos exposure to all ESA listed species in the United States. This biological evaluation was performed as part of the registration review for chlorpyrifos uses by EPA's Office of Pesticide Programs. This national BE included a review of ECOTOX data on chlorpyrifos toxicity to all aquatic species that was available as of the date of the national BE. Once identified and compiled, EPA (USEPA, 2017a) then performed a data quality review of the ECOTOX data, and compiled it into a table of toxicity data of acceptable data quality for use in water quality criteria development (Data Appendix 2-2.xlsx of (USEPA, 2017a)). Data from the national chlorpyrifos BE was used to identify additional toxicity information not presented in EPA (USEPA, 1986c).

None of the studies described in this section were cited in the References section of the EPA (USEPA, 1986c) chlorpyrifos water quality criterion document.

There are no acute toxicity studies of chlorpyrifos on any of the seven ESA listed marine fish species within the marine portions of the action area in the data published between 1986 and 2017. Nor are there any empirical acute toxicity studies available for surrogate species for the listed species at the genus, family or order taxonomic level.

Two new species worth of empirical acute toxicity data with acceptable data quality not listed in EPA (USEPA, 1986c) were identified in the literature review in EPA (USEPA, 2017a). Varo et al. (Varo et al., 2000) determined a chlorpyrifos 96 hour LC_{50} of 483.6 µg/L for the mosquitofish (*Gambusia affinis*), while Mayer (Mayer Jr., 1987) measured a 96 hour LC_{50} of 1.8 µg/L for Gulf killifish (*Fundulus grandis*). EPA (USEPA, 2017a) also reported four additional new 96 hour LC_{50}s for species with one or more 96 hour LC_{50} values already discussed in EPA 1986: (96 hour LC_{50}s of 68 µg/L, 2.7 µg/L, 1.7 µg/L and 0.96 µg/L for Gulf toadfish, California grunion, Atlantic silverside and tidewater silverside, respectively.

Because no *Salmoniformes* toxicity data were available for the chlorpyrifos assessment, EPA incorporated SSD modeling to generate a toxicity value (i.e., HC_5) to be used in the effects analysis. The combined high quality 96 hour LC_{50} data found in EPA (USEPA, 1986c) and EPA (USEPA, 2017a) results in a total of 11 species in seven genera worth of acute toxicity data to enter in a SSD calculation (Table 97). For species with multiple high quality 96 hour LC_{50} results, the lowest of the available 96 hour LC_{50} studies for each species was used to populate Table 97.

Table 97. Acute 96 hour LC50 data for fish species in the class Actinopterygii used as surrogates to assess chlorpyrifos toxicity to ESA listed species in the marine portions of the action area

Rank	Species	Lowest 96 hour LC_{50} (µg/L)	Citation
11	Mosquitofish (*Gambusia affinis*)	483.6	(Varo et al., 2000)
10	Sheepshead minnow (*Cyprinodon variegatus*)	136	(S. C. Schimmel et al., 1983)
9	Gulf toadfish (*Opsanus beta*)	68	(Mayer Jr., 1987)
8	Striped mullet (*Mugil cephalus*)	5.4	(S. C. Schimmel et al., 1983)
7	Mummichog (*Fundulus heteroclitus*)	4.65	(Thirugnanam & Forgash, 1977)
6	Inland silverside (*Menidia beryllina*)	4.2	(Clark, Patrick Jr, Middaugh, & Moore, 1985)
5	Longnose killifish (*Fundulus similis*)	4.1	(S. C. Schimmel et al., 1983)
4	Gulf killifish (*Fundulus grandis*)	1.8	(Mayer Jr., 1987)
3	California grunion (*Leurestes tenuis*)	1.0	(P. W. Borthwick, J. M. Patrick, Jr., & D. P. Middaugh, 1985)

Rank	Species	Lowest 96 hour LC_{50} (µg/L)	Citation
2	Atlantic silverside (*Menidia menidia*)	0.5	(P. W. Borthwick et al., 1985)
1	Tidewater silverside (*Menidia peninsulae*)	0.4	(P. W. Borthwick et al., 1985)

5.6.1.5.4 Effects assessment of acute marine chlorpyrifos criterion on listed species

The complete lack of empirical chlorpyrifos acute toxicity for any of the seven ESA listed species under assessment, or for species closely related taxonomically to the listed species means that under the methodology used in this BE the assessment method will be generation of a SSD based on 96 hour LC_{50} data of acceptable quality from class *Actinopterygii* fish species. This approach is the 7th and lowest tier of the hierarchy for performing acute effects assessments and is used only when there are no acute empirical toxicity studies for either the listed species themselves, or for surrogate species at the order or lower taxonomic level.

The acute toxicity data in Table 97 was used to generate the SSD for acute toxicity of chlorpyrifos to marine fish species. For species with multiple high quality 96 hour LC_{50} results, the lowest of the available 96 hour LC_{50} studies for each species was used in the generation of the SSD. The 5th percentile of the acute toxicity data in Table 97 divided by 2.27 is assumed to be the acute effects assessment concentration (i.e. the highest chlorpyrifos concentration not associated with any acute toxicity to any fish species, including the seven ESA listed marine fish species within the action area).

The 5th percentile 96 hour LC_{50} of the 11 acute toxicity studies listed in Table 97 is 0.115 µg/L, as derived from the SSD of the 11 acute studies. Dividing 0.115 µg/L by 2.27 to convert a 96 hour LC_{50} to an LC_{low}, considered to be the highest chlorpyrifos concentration with no acute effect on survival of any marine fish species results in an acute effects assessment concentration of 0.0507 µg/L.

Because chlorpyrifos is not expected to elicit any short term acutely toxic effects on any marine fish species at a chlorpyrifos concentration of 0.0507 µg/L or lower, exposure at the level of the marine acute chlorpyrifos criterion of 0.011 µg/L is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio and yelloweye rockfish.

5.6.1.5.5 Acute-chronic ratio for chlorpyrifos

The chlorpyrifos ACR in EPA (USEPA, 1986c) is 4.064. This ACR was derived from the geometric mean of five species specific ACRs ranging between 1.374 and 12.5.

5.6.1.5.6 Water-column exposure toxicity data - chronic criterion for chlorpyrifos in marine waters

The chlorpyrifos criteria document (USEPA, 1986c) lists six marine fish species with chronic toxicity data of acceptable data quality for development of a chronic water quality criterion. All of the measured toxic endpoints were reductions in growth. None of the fish species with acceptable chronic toxicity data are close taxonomic relatives of the ESA listed salmonids and rockfish in the action area. The species listed in Table 98 are only related to the ESA listed species at the class *Actinopterygii* level.

Table 98. Chronic toxicity data for fish species in the class Actinopterygii used as surrogates to assess chlorpyrifos toxicity to ESA listed species in the marine portions of the action area

Rank	Species	Chronic Value (µg/L)	Citation
6	Gulf toadfish (*Opsanus beta*)	2.276	(D. J. Hansen et al., 1986)
5	Sheepshead minnow (*Cyprinodon variegatus*)	2.258	(Cripe, Hansen, Macauley, & Forester, 1986)
4	Inland silverside (*Menidia beryllina*)	1.162	(L. Goodman, Hansen, Middaugh, Cripe, & Moore, 1985)
3	Tidewater silverside (*Menidia peninsulae*)	0.5444	(L. Goodman, Hansen, Middaugh, et al., 1985)
2	Atlantic silverside (*Menidia menidia*)	0.3666	(L. Goodman, Hansen, Middaugh, et al., 1985)
1	California grunion (*Leurestes tenuis*)	0.2049	(L. Goodman, Hansen, Cripe, Middaugh, & Moore, 1985)

All six of the chronic NOEC values for fish in Table 97 are higher than the chronic chlorpyrifos criterion of 0.0056 µg/L. By comparison, the lowest chronic value for an invertebrate is 0.0028 µg/L for the mysid *Americamysis bahia* (formerly *Mysidopsis bahia*), which is lower than the chronic chlorpyrifos criterion.

5.6.1.5.7 Additional marine chronic toxicity data for chlorpyrifos published since 1986

No chronic toxicity studies with fish of a sufficiently long duration for use in chronic water quality criteria development published after 1986 have been identified.

5.6.1.5.8 Effects assessment of marine water chronic chlorpyrifos criterion on listed species

There are no empirical chronic toxicity data for chlorpyrifos for any of the ESA listed species under assessment. In the hierarchy of biological evaluation methods for assessment of chronic water quality data (Table 82, section 5.4), the 13th tier (5th percentile of a SSD generated from chronic NOEC data for class-level surrogate species for the listed species) may be applicable. Table 98 contains surrogate species data for only six fish species.

Guidance documents for SSD development vary in their recommendations for the minimum number of species needed to generate useful SSDs. This variation in the required minimum number of species can result from differences in intended uses of the SSD (e.g. regulatory standard, risk assessment), quality of the data used to calculate the SSD, level of protectiveness desired (e.g. EC_{20}, EC_{10}, or EC_5), taxonomic considerations or statistical reliability considerations. Recommended minimum numbers of species for SSD generation in environmental applications have ranged between four (Aldenberg & Luttik, 2002), five (Batley et al., 2018), eight (C.E. Stephan et al., 1985), 10 (Posthuma, van Gils, Zijp, van de Meent, & de Zwart, 2019), and 15 – 50 (Newman et al., 2000).

Given the uncertainty surrounding the minimum number of species recommended for SSD generation, we have calculated the chronic effects assessment concentration using both the 13th and 16th tier (5th percentile of a SSD generated from acute LC_{50} data divided by the ACR for class-level surrogate species for the listed species) of the hierarchy in the methodology for assessing chronic toxicity to ESA listed species. The 16th tier of the BE methodology is the lowest tier of the chronic effects assessment

methodology, used only when there are no toxicity data for the listed species under evaluation, and the closest taxonomic relatives to the listed species are in the same class as the listed species.

The 5th percentile chronic NOEC of the six chronic fish toxicity studies listed in Table 98 is 0.142 µg/L, as derived from the SSD of the six empirical chronic marine fish studies. The 5th percentile chronic NOEC derived from the 11 96 hour LC_{50}s in Table 97, divided by the ACR of 4.064 is 0.028 µg/L, as derived from the SSD of the 11 chronic values estimated from 96 hour LC_{50}s and ACRs. Both chronic NOECs of 0.142 µg/L and 0.028 µg/L are higher than the marine chronic chlorpyrifos criterion of 0.0056 µg/L.

Because chlorpyrifos is not expected to elicit any long-term chronic toxic effects on any marine fish species at a chlorpyrifos concentration of 0.028 µg/L or lower, exposure at the level of the chronic chlorpyrifos criterion of 0.0056 µg/L is not likely to adversely affect rainbow trout, Chinook salmon, chum salmon, bull trout, bocaccio and yelloweye rockfish. To make this conclusion, we assume that the two calculated 5th percentile SSD concentrations are protective of all marine fish species from long-term chronic chlorpyrifos toxicity, and is not just protective of the six or 11 fish species used to generate the two SSDs.

5.6.1.5.9 Summary of effect determinations for marine chlorpyrifos criteria

EPA has made the following determinations on the acute and chronic chlorpyrifos criteria for the listed species in the marine portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.6.1.6 Chromium, hexavalent – Marine

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) Criterion = 1100 µg/L

Chronic (CCC) Criterion = 50 µg/L

CAS ID 10588-01-9

5.6.1.6.1 Water-column exposure toxicity data – acute criterion for hexavalent chromium in marine water

Acute toxicity data for hexavalent chromium have been reported for twenty marine fish and invertebrate species. The acute LC50 values for fish species range from 12,400 ug/L for the Atlantic silverside (*Menidia menidia*) to 91,000 ug/L for the murmichog (*Fundulus heteroclitus*). Acute toxicity values for a polychaete worm and mysid shrimp ranged from 2,000 ug/L to 105,000 ug/L for the mud snail. The most sensitive species were the polychaete annelids (2,000-8,000 ug/L), the mysid shrimp (2,000-4,400 ug/L), and two copepods (3,650 and 6,600 ug/L) (*Ambient aquatic life water quality criteria for chromium*, 1985; Schatzow, 1980).

Hutchinson et al. (Hutchinson, Williams, & Eales, 1994) conducted a 7-day comparative study where they evaluated the toxicity of hexavalent chromium to larvae of sheepshead minnow (*Cyprinodon variegatus*). Toxicity to fish larvae was measured in terms of survival and growth. For fish larvae, 96 hr. LC50 values (based on mean measured concentrations of total metal ion) were 31.6 mg $Cr6^+$/L, for survival of larvae and 23.2 mg $Cr6^+$/L, for survival of juvenile. Subchronic values for larval fish survival and growth after 7 days were <10.0 mg $Cr6^+$/L.

Jop (Jop, 1989) reported a 96-hr LC50 for sheepshead minnow of 31.6 mg $Cr6^+$/L and a subchronic (7 days) value < 2.65 mg $Cr6^+$/L for survival and growth of larvae.

Jop et al. (Jop, Rodgers, Dorn, & Dickson, 1986), also reported a 96-hr LC50 of 23.2 mg $Cr6^+$/L for survival of juvenile.

Taylor et al. (Taylor, Maddock, & Mance, 1985) reported 96-hr LC50s for two marine fish species: dab (*Limanda limanda*) and grey mullet (*Chelon labrosus*) of 47.7 mg $Cr6^+$/L and 47.0 mg $Cr6^+$/L for juvenile, respectively (see Table 99 below).

Marine fish larvae were selected for the toxicity test because of their importance in the aquatic and human food chains, and their reported sensitivity to a variety of chemicals (J. Weis & Weis, 1989).

Holland et al. (Holland, Lasater, Neumann, & Eldridge, 1960) conducted a 5-day study with hexavalent chromium (chromic acid, potassium salt (1:2)) on coho salmon (*Oncorhynchus kisutch*) and reported a subchronic NOEC of 17.8 mg/L. The effect measured was mortality.

Table 99. Comparative Toxicity of Hexavalent Chromium to Early Life-Stages of Juveniles of Marine Fish

Test Species	Test Type[a]	Toxicity Data (mg/L)				References
		LC_{50}[b] (96-hr.)	NOEC[c] (7 days)	LOEC[c] (7 days)	Subchronic Value[c] (7 days)	
Chelon labrosus	FT/JU	47.7	-	-	-	(Taylor et al., 1985)
Cyprinodon variegatus	ST/LA	31.6	<10.0	10.0	<10.0	(Hutchinson et al., 1994).
Cyprinodon variegatus	SR/LA	-	3.2[d]	32.0[d]	10.2[d]	(McCulloch & Rue, 1989)
Cyprinodon variegatus	SR/LA	31.6	<2.65	2.65	<2.65	(Jop, 1989)
Cyprinodon variegatus	S/JU	23.2	-	-	-	(Jop et al., 1986)
Limanda limanda	FT/JU	47.0	-	-	-	(Taylor et al., 1985)

[a]**FT**, Flowthrough; **JU**, Juvenile; **SR**, Static-Renewal; **LA**, Laval; **ST**, Static
[b]Based on Survival
[c]Lowest value based on survival, growth or development
[d]Mean of three tests

Source: Adapted from Hutchinson et al. (1994).

5.6.1.6.2 Effects assessment of acute marine criterion for hexavalent chromium on listed species

The lowest acute marine toxicity value for fish obtained from ECOTOX data pull for hexavalent chromium is from a study by Hutchinson et al. (1994) with sheepshead minnow (*Cyprinodon variegatus*) larvae. The study reported a 96-hr LC50 of 31.6 mg/L (31,600 µg/L) for growth and survival of larvae. This study met all the quality assurance criteria and the data is suitable for use in the Swinomish BE. Therefore, this value when compared to the acute marine criterion of 1,100 µg/L proposed for the Swinomish BE, is higher. If the minimum acute effect concentration for the remaining listed fish (including salmonids) in the marine portions of the action area is greater than or equal to 1,100 µg/L, exposure at the level of the hexavalent chromium chronic criterion of 1,100 µg/L is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout and rock fish.

5.6.1.6.3 Water-column exposure toxicity data – chronic criterion for hexavalent chromium in marine water

Chronic toxicity of hexavalent chromium in marine fish and invertebrate species have been reported. Results have been published of chronic exposures of saltwater polychaete (*Neanthes arenaceodentata*) to chromium VI for 56 and 59 days with LC50s of 200 ug/L (Mearns, Oshida, Sherwood, Young, & Reish, 1976; Oshida & Reish, 1975). The sublethal effects measured from these studies were inhibition of tube formation at 79 ug/L. Chronic results have also been reported for mysid (*Mysidopsis bahia*). Chronic toxicity reported for the life-cycle tests for the mysid was 132 ug/L (*Ambient aquatic life water quality criteria for chromium*, 1985).

Mearns et al. (Mearns et al., 1976) reported 21 days LC_{50} for hexavalent chromium on speckled sanddabs (*Citharichthys stigmaeus*) of 5.0 mg/L. Sherwood (Sherwood, 1975), also reported 21 days LC_{50}s of 5.4 and 2.2 mg/L, respectively, for speckled sanddabs.

Holland et al. (Holland et al., 1960) conducted a 3-day subchronic study with hexavalent chromium (potassium chromate) on coho salmon (*Oncorhynchus kisutch*) and reported a subchronic NOEC of 31.8 mg/L. The effect measured was mortality. This reported concentration is similar to the SMAV for the speckled sanddab of 30.5 mg/L (Mearns et al., 1976), but twice that reported for Atlantic silverside (*Menidia menidia*) of 15.28 mg/L (J. A. Cardin, 1982).

Holland et al. (1960) also conducted an 11-day subchronic study with hexavalent chromium (potassium chromate) on coho salmon (*Oncorhynchus kisutch*) and reported a subchronic NOEC of 17.8 mg/L. The effect measured was also mortality.

Table 100. Chronic Toxicity of Hexavalent Chromium to Marine Fish and Polychaetes

Species Tested	Effect Measured	Toxicity Values (ug/L)				References
		NOECs				
		Chronic Toxicity Value	Test Duration (Days)	Subchronic Toxicity Value	Test Duration (Days)	
Neanthes arenaceodentata	Inhibition of Tube Formation	200	56	-	-	(Mearns et al., 1976).
Neanthes arenaceodentata	Inhibition of Tube Formation	200	59	-	-	(Oshida & Reish, 1975).
Mysidopsis bahia	Mortality	132	28	-	-	(Schatzow, 1980)
Citharichthys stigmaeus	Mortality	5,000	21	-	-	(Mearns et al., 1976).
Citharichthys stigmaeus	Mortality	5,400	21	-	-	(Sherwood, 1975).
Citharichthys stigmaeus	Mortality	2,200	21	-	-	(Sherwood, 1975).
Citharichthys stigmaeus	Mortality	30,500	21	-	-	(Mearns et al., 1976).
Oncorhynchus kisutch	Mortality	-	-	31,800	3	(Holland et al., 1960).
Oncorhynchus kisutch	Mortality	-	-	17,800	11	(Holland et al., 1960).
Menidia menidia	Mortality	20,100	-			(J. A. Cardin, 1982)

5.6.1.6.4 Effects assessment of chronic marine criterion for hexavalent chromium on listed species

The lowest Chronic marine toxicity value for fish obtained from ECOTOX data pull for hexavalent chromium is from a study by Holland et al. (Holland et al., 1960) with Coho salmon (*Oncorhynchus kisutch*). The study reported an 11-day subchronic NOEC of 17.8 mg/L (17,800 µg/L) for mortality. This study met all the quality assurance criteria and the data is suitable for use in the Swinomish BE. In addition, the study was incorporated in the derivation of the EPA Ambient Water Quality Criteria for chromium (*Ambient aquatic life water quality criteria for chromium*, 1985). Therefore, this value when compared to the chronic marine criterion of 50 µg/L proposed for the Swinomish BE, is higher. If the

minimum chronic effect concentration for the remaining listed fish (including salmonids) in the marine portions of the action area is greater than or equal to 50 µg/L, exposure at the level of the hexavalent chromium chronic criterion of 50 µg/L is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio, and yelloweye rockfish.

5.6.1.6.5 Summary of effect determinations for marine hexavalent chromium criteria

EPA has made the following determinations on the acute and chronic hexavalent chromium criteria for the listed species in marine portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Likely to adversely affect** (indirect effects – availability of prey)

Chinook salmon – **Likely to adversely affect** (indirect effects – availability of prey)

Chum salmon – **Likely to adversely affect** (indirect effects – availability of prey)

Bull trout – **Likely to adversely affect** (indirect effects – availability of prey)

Bocaccio – **Likely to adversely affect** (indirect effects – availability of prey)

Yelloweye rockfish – **Likely to adversely affect** (indirect effects – availability of prey)

Marbled murrelet – **Likely to adversely affect** (indirect effects – availability of prey)

Chronic criterion:

Rainbow trout (steelhead) – **Likely to adversely affect** (indirect effects – availability of prey)

Chinook salmon – **Likely to adversely affect** (indirect effects – availability of prey)

Chum salmon – **Likely to adversely affect** (indirect effects – availability of prey)

Bull trout – **Likely to adversely affect** (indirect effects – availability of prey)

Bocaccio – **Likely to adversely affect** (indirect effects – availability of prey)

Yelloweye rockfish – **Likely to adversely affect** (indirect effects – availability of prey)

Marbled murrelet – **Likely to adversely affect** (indirect effects – availability of prey)

5.6.1.7 Cyanide – Marine

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion – 1.0 µg/L

Chronic (CCC) criterion – 1.0 µg/L

CAS ID – 57-12-5 (cyanide anion CN^-)

74-90-8 (hydrogen cyanide = hydrocyanic acid)

143-33-9 (sodium cyanide)

151-50-8 (potassium cyanide)

5.6.1.7.1 Introduction to cyanide

Cyanide occurs in water as hydrocyanic acid (HCN), the cyanide ion (CN^-), simple cyanides such as sodium or potassium cyanide, metallocyanide complexes, and nitriles (any organic compound with a CN functional group) (EPA 1985). Free cyanide is defined as the sum of the cyanide present as HCN and as CN^-. The relative concentrations of these two forms depend mainly on pH and temperature. When pH is below 8 and temperature is below 25°C, at least 94 percent of the free cyanide exists as HCN. When pH or temperature or both are higher, a greater percentage of free cyanide exists as CN^-. Although free cyanide is chemically defined as (HCN + CN^-), EPA aquatic life criteria for cyanide are somewhat confusingly expressed also as free cyanide (defined by EPA as CN^-only) because free cyanide (CN^- only) is believed to be a more reliable index of toxicity to aquatic life than total cyanide (EPA 1985). Total cyanide can include nitriles and relatively stable metallocyanide complexes in addition to the simple cyanides and free cyanides.

Simple cyanides such as sodium cyanide and potassium cyanide readily dissociate and hydrolyze to form CN^- and HCN, and are the chemical forms most commonly used in aquatic toxicity tests. Metallocyanides have a wide range of stabilities, from near complete dissociation (Zn and Cd), to pH dependent dissociation (Cu and Ni) to little or no dissociation (Fe). Because the dissociation products of metallocyanides are the toxic HCN and/or CN^-, the direct toxic effects of free cyanide should be sufficient to evaluate toxicity of the metallocyanides.

Because of the definition of free cyanide used in the EPA (1985) criteria document, there may be a need to convert the measured cyanide concentrations in some literature studies to the free cyanide (CN^- only) chemical form in which the EPA cyanide aquatic life criteria are expressed.

All cyanide concentrations reported in the 1985 EPA aquatic life criteria document in are in terms of free cyanide expressed as CN. Thus, data reported in the original literature in terms of free cyanide expressed as CN did not have to be adjusted. However, when free cyanide was expressed as HCN, KCN, NaCN, etc., the results were adjusted using the molecular weights of the compound and [CN]. When data were reported in the original literature in terms of [HCN], rather than in terms of free cyanide, the data were converted from molecular HCN to free cyanide as CN as follows:

$$(\mu g \text{ of free cyanide as CN/L}) = (\mu g \text{ of HCN/L})\,(1 + 10^{pH - pK_{HCN}}) \times \frac{\text{mol. wt. CN}}{\text{mol. wt. HCN}}$$

where $pK_{HCN} = 1.3440 + \frac{2347.2}{T + 273.16}$ (Izatt, Christensen, Pack, & Bench, 1962)

and T = degrees Celsius.

5.6.1.7.2 Water-column exposure toxicity data - acute criterion for cyanide in saltwater

Relative to freshwater, few toxicity studies have evaluated the effects of cyanide on marine species. As was the case for freshwater species, most cyanide toxicity studies with marine species were performed in the 1970s and 1980s.

The USEPA (1985c) cyanide criteria document contains only three 96 hour LC_{50} values for marine fish, shown below.

Table 101. Marine fish species with reported 96 hour LC_{50} values in the EPA (1985) aquatic life criteria document

Species	96 hour LC_{50}, µg/L	Citation
Winter flounder *Pseudopleuronectes americanus*	372	Cardin 1980
Sheepshead minnow *Cyprinodon variegatus*	300	Schimmel et al. 1981
Atlantic silverside *Menidia menidia*	59	Gardner and Berry 1981

None of the marine fish acute toxicity studies in USEPA (1985c) are for ESA listed fish species within the marine portions of the action area. The remaining six marine species with cyanide acute toxicity data in USEPA (1985c) are invertebrates. The most sensitive marine species in USEPA (1985c) is larvae of the rock crab (*Cancer irroratus*) with a species mean acute value (SMAV) of 4.893 µg/L for cyanide (Johns and Gentile, 1981).

5.6.1.7.3 Additional marine acute toxicity data for cyanide published since 1985

We have identified additional 96 hour LC_{50} values for three marine fish species. Unfortunately, none of the three species discussed in this section are resident in North America, one of the requirements in C.E. Stephan et al. (1985) for using a study to derive EPA aquatic life criteria. Because of the lack of other empirical acute toxicity data with marine fish species, the additional data we have identified that was published after the USEPA (1985c) aquatic life criteria document was issued.

Table 102. Marine fish species with reported 96 hour LC_{50} values published after the EPA (1985) cyanide aquatic life criteria document was issued

Species	96 hour LC_{50}, µg/L	Citation
Boddart's goggle-eyed goby *Boleophthalamus boddarti*	290	Chew and Ip 1992
Australian bass *Macquaria novemaculaeta*	109	Pablo et al. 1996
Black bream *Acanthopagrus butcheri*	70	Pablo et al. 1996

All three of the species with published acute toxicity published since the cyanide criteria document (USEPA, 1985c) was published are native to Australia and southeast Asia.

5.6.1.7.4 Effects assessment of marine acute cyanide criterion on listed species

As only three acute toxicity results with resident marine fish species in North America are available, none of the seven tiers of the hierarchy in the BE methodology can be applied to estimate a cyanide acute assessment effects concentration. However, combining the three cyanide 96 hour LC_{50}s using Australian species with the three 96 hour LC_{50}s with North American resident fish species provides enough data to construct a species sensitivity distribution to calculate a 5th percentile concentration of the available acute cyanide toxicity data. The calculated 5th percentile of the SSD can then be divided by 2.27 to obtain an acute assessment effects concentration for comparison to the cyanide marine acute criterion. This approach is comparable to the seventh and lowest tier of the hierarchy for using class-level surrogate species data to calculate the acute assessment effects concentration, with the exception that not all 96 hour LC_{50} data in the SSD will be from North American resident fish species.

The fitted SSD to the combined six available fish 96 hour LC_{50}s estimated the 5th percentile of the distribution to be 36.8 µg/L (N = 6, d.f. = 4, m = 2.619, b = -0.745, r^2 = 0.880), with 95% confidence limits of 14.1 – 95.7 µg/L. Dividing the 5th percentile of the 96 hour LC_{50} and its lower 95% confidence limit results in a central tendency and a conservative cyanide acute assessment effects concentration of 16.2 µg/L and 6.2 µg/L respectively, for marine fish species. Both of these estimates of the acute assessment effects concentration for the marine acute cyanide criterion are higher than the acute cyanide criterion of 1.0 µg/L. As such, we have determined that exposure at the level of the acute marine cyanide criterion of 1.0 µg/L is not likely to adversely affect any of the ESA listed marine fish species (steelhead, Chinook salmon, chum salmon, bull trout, bocaccio and yelloweye rockfish) in the marine portions of the action area.

5.6.1.7.5 Acute-chronic ratio for cyanide

The acute-chronic ratio (ACR) for cyanide in USEPA (1985c) is 8.568. USEPA (2006a) calculated an ACR of 10.57 for fish species. The fish ACR from USEPA (2006a) is the ACR that is used in this BE, as all of the ESA listed fully aquatic species evaluated in this BE are fish.

5.6.1.7.6 Water-column exposure toxicity data - chronic criterion for cyanide in marine waters

The only chronic toxicity study with a marine fish in the cyanide criteria document (USEPA, 1985c) is an early life stage study with the sheepshead minnow (*Cyprinodon variegatus*) performed by Schimmel et al. (1981). Growth of sheepshead minnow was not significantly reduced at a cyanide concentration of 462 µg/L. Long-term survival, however, was significantly reduced at cyanide concentrations ≥ 45 µg/L but not at ≤ 29 µg/L. Thus, the chronic value for sheepshead minnow was calculated as the geometric mean of the chronic survival NOEC and the chronic survival LOEC, resulting in a chronic MATC of 36.12 µg/L.

5.6.1.7.7 Additional marine chronic toxicity data for cyanide published since 1985

EPA has been unable to identify any chronic toxicity data for cyanide for any marine fish species that is of acceptable data quality for use in deriving aquatic life criteria published since 1985.

5.6.1.7.8 Effects assessment of marine chronic cyanide criterion on listed species

In the absence of empirical chronic toxicity data for any of the ESA listed species or taxonomically related species at the genus or family level within the marine portions of the action area, EPA is limited to using the available order-level surrogate species acute toxicity data to derive chronic NOEC estimates for the ESA listed species. The specific approach used here is to use a species sensitivity distribution 5th

percentile acute LC_{50} derived from order-level surrogate species for the listed species divided by the ACR for the chemical. This is the 15th tier of the 16 available tiers in the BE methodology to estimate chronic toxicity assessment effects concentrations.

The 5th percentile of the SSD from the six available 96 hour LC_{50}s for marine fish species is 36.8 µg/L, with a 95% lower confidence limit of 14.1 µg/L. Dividing these estimates of the 5th percentile of all 96 hour LC_{50}s for marine fish species by the acute-chronic ratio of 10.57 for fish, a chronic assessment effects concentration of 3.48 µg/L with a 95% lower confidence limit of 1.33 µg/L is calculated for all marine fish species. Both of these estimates of the chronic assessment effects concentration for cyanide are higher than the marine chronic cyanide criterion of 1.0 µg/L. As such, we have determined that exposure at the level of the chronic marine cyanide criterion of 1.0 µg/L is not likely to adversely affect any of the ESA listed marine fish species (steelhead, Chinook salmon, chum salmon, bull trout, bocaccio and yelloweye rockfish) in the marine portions of the action area.

5.6.1.7.9 Summary of effect determinations for marine cyanide criteria

EPA has made the following determinations on the acute and chronic cyanide criteria for the listed species in the marine portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above, and the indirect effects analysis for prey availability provided in section 5.8 (the multiple routes of exposure analysis was not relevant as indicated in section 5.7). The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.6.1.8 Diazinon – Marine

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = 0.82 µg/L

Chronic (CCC) criterion = 0.82 µg/L

CAS ID 333-41-5

5.6.1.8.1 Water-column exposure toxicity data - acute criterion for diazinon in marine water

The data in the EPA (USEPA, 2005b) aquatic life criteria for diazinon used marine acute toxicity information for two fish species and seven invertebrate species to derive the marine acute diazinon criterion. The acute toxicity to invertebrates ranged from 2.57 µg/L for the copepod *Acartia tonsa* (Khattat & Farley, 1976) to > 9,600 µg/L for embryos of the sea urchin *Arbacia punctulata*.

EPA has been unable to find any toxicity data for diazinon with any of the ESA listed salmonid or rockfish species in marine water within the action area. For saltwater fish, the two species tested and included in the marine acute criterion derivation were the inland silverside (*Menidia beryllina*) and sheepshead minnow (*Cyprinodon variegatus*). Test results indicated that the inland silverside is relatively insensitive to diazinon compared to arthropods with a 96-hour LC_{50} of 1,170 *µg/*L (G. Thursby & Berry, 1988), whereas the sheepshead minnow exhibited slightly less sensitivity with a 96-hour LC_{50} of 1,400 µg /L (Larry R Goodman, Hansen, Coppage, Moore, & Matthews, 1979). The much greater toxicity of diazinon to arthropod invertebrate species than to any other group of animals is not surprising, and is in keeping with the primary use of diazinon as an insecticide and acaricide.

5.6.1.8.2 Additional acute toxicity data for diazinon published since 2005

A search of the ECOTOX database found two acute toxicity studies of diazinon on marine fish species published between the 2005 issuance of EPA's diazinon criteria and present day. Couillard et al. (Couillard, Lebeuf, Légaré, & Trottier, 2008) observed no mortality during a four day exposure of larval mummichog (*Fundulus heteroclitus*) to 12.9 µg/L diazinon starting immediately after hatch. The eggs from which the mummichog larvae hatched had been topically treated immediately after fertilization with 0.1 µL of a 100 pg/µL solution of 3,3',4,4',5-pentachlorobiphenyl (PCB126). The egg exposure to PCB 126 resulted in a whole body PCB concentration of 710 pg/g wet weight in the newly hatched larvae. The combined exposure of mummichog larvae to PCB and diazinon makes the Couillard et al. (2008) study a mixture study not suitable for EPA aquatic life criteria derivation.

Mhadhbi and Boumaiza (Mhadhbi & Boumaiza, 2012b) made a number of observations on diazinon effects on turbot (*Scophthalmus maximus*, called by its older scientific name *Psetta maxima* in the study). Mhadhbi and Boumaiza (2012b) reported a 96-hour LC_{50} of 1,230 µg/L for larval turbot exposed to diazinon. They also reported 24, 48 and 72-hour LC_{50}s of 8000, 3300 and 2100 µg/L, respectively, which would permit calculation of a 1-hour LC_{50}, the exposure duration for all acute criteria of the Swinomish Tribe if needed for this effects assessment. While the Mhadhbi and Boumaiza (2012b) study

with turbot meets the data quality criteria EPA requires for data used to derive aquatic life criteria, turbot are a non-native species without a breeding population in North America. As such, the Guidelines (C.E. Stephan et al., 1985) preclude the use of non-native species without North American breeding populations in the derivation of aquatic life criteria. In any event, the sensitivity of turbot to diazinon is comparable to that of the native inland silverside (96-hour LC_{50} = 1170 µg/L) and sheepshead minnow (96-hour LC_{50} = 1400 µg/L), and its non-inclusion in the data used in this effects assessment will have minimal impact on the marine acute diazinon effects assessment in this BE.

The evaluation of diazinon effects on its acute toxicity to listed fish species is therefore based on the toxicity information available during the derivation of the EPA (USEPA, 2005b) diazinon criteria.

5.6.1.8.3 Effects assessment of acute marine/estuarine diazinon criterion on listed species

The lowest 96-hr acute LC_{50} for any marine fish species with acceptable data quality is the G. Thursby and Berry (1988) study with inland silverside. The 96-hour LC_{50} for inland silverside was 1179 µg/L. To estimate the acute NOEC or minimum acute effect concentration for all marine fish from the available empirical toxicity data, the lowest LC_{50} is divided by 2.27: 1170 µg/L /2.27 = 515 µg/L. This value is higher than the diazinon acute freshwater criterion of 0.82 µg/L. If it is assumed that the minimum acute effect concentration for the listed fish species in the marine portions of the action area is greater than 515 µg/L, exposure at the level of the diazinon freshwater acute criterion of 0.82 µg/L is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio and yelloweye rockfish.

5.6.1.8.4 Water-column exposure toxicity data – chronic criterion for diazinon in marine/estuarine water

Chronic toxicity of diazinon to marine/estuarine fish and invertebrates have been determined for sheepshead minnow (*Cyprinodon variegatus*) and in a life-cycle test with the invertebrate mysid (*Americamysis bahia).* Nimmo et al. (1981) conducted a 22-day test with mysid and the result was compared to those reported by Berry (W. J. Berry, 1989; Nimmo et al., 1981). The comparison of the results indicated no statistical difference in survival observed in both tests at any of the concentrations tested (0.54, 1.2, 2.1, 4.4 µg/L). Additionally, the number of young per female was not significantly reduced relative to controls at diazinon concentrations lower than 2.1 µg/L. Female mysids exposed to the highest concentration of diazinon of 4.4 µg/L did not produce no young ones. Thus, based on reproduction, the geometric mean of the chronic limits, 2.1 and 4.4 µg/L, are the chronic values for the mysid and the calculated ACR is 1.586 (USEPA, 2005b).

For saltwater fish, Goodman et al. (Larry R Goodman et al., 1979) reported that reproduction in sheepshead minnow (*Cyprinodon variegatus*) was significantly reduced in all diazinon exposure concentrations during a partial life-cycle test. The number of eggs spawned per female in the 0.47, 0.98, 1.8, 3.5 and 6.5 µg/L (average measured) diazinon concentrations were 69, 50, 50, 55 and 45 percent of control fish, respectively. Neither survival nor growth was affected by diazinon exposures lower than 6.5 µg/L. Based on the observed reproductive effects (reduction of eggs spawned), the chronic value for sheepshead minnow is lower than 0.47 µg/L and the calculated ACR is >2979 (USEPA, 2005b).

5.6.1.8.5 Effects assessment of chronic marine/estuarine diazinon criterion on listed species

The lowest toxicity value for saltwater fish based on ECOTOX data pull is from a study by Mhadhbi and Boumaiza (2012b) with Flounder, *Turbot* (*Psetta maxima*). The study reported a 6-day NOEC of 200 µg/L for survival of embryos/larvae. This study met all the quality assurance criteria and the data is suitable

for use in the Swinomish BE. To estimate the chronic NOEC for all fish, the NOEC is divided by 2.27: 200 µg/L /2.27 = 88.1 µg/L. This value is higher than the diazinon chronic marine/estuarine criterion of 0.82 µg/L. If the minimum acute effect concentration for the remaining listed fish (including salmonids) in the marine portions of the action area is greater than or equal to 88.1 µg/L, exposure at the level of the diazinon marine/estuarine chronic criterion of 0.82 µg/L is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout and rock fish.

5.6.1.8.6 Summary of effect determinations for marine diazinon criteria

EPA has made the following determinations on the acute and chronic diazinon criteria for the listed species in marine/estuarine portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.6.1.9 Hydrogen sulfide – Marine

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion – No criterion

Chronic (CCC) criterion – 2.0 µg/L

CAS ID – 7783-06-4 (hydrogen sulfide)

1313-82-2 (sodium sulfide)

16721-80-5 (sodium bisulfide)

5.6.1.9.1 Introduction to hydrogen sulfide

The EPA aquatic life criteria document for hydrogen sulfide was originally published in EPA (USEPA, 1976d, 1976e), commonly known as the "Red Book." The most recently published hydrogen sulfide aquatic life criteria document was published in EPA (USEPA, 1986e, 1986f), commonly known as the "Gold Book." The EPA (USEPA, 1986e, 1986f) hydrogen sulfide criterion is a largely verbatim republication in a more legible form of EPA (USEPA, 1976d, 1976e). As there are no differences in the chronic criterion concentration or the literature cited to derive both the EPA (USEPA, 1976d, 1976e) and EPA (USEPA, 1986e, 1986f) hydrogen sulfide criteria documents, we cite EPA (USEPA, 1976d, 1976e) as the source of the hydrogen sulfide criterion in this BE.

A description of the chemical and physical properties of hydrogen sulfide that affect measurement and toxicity in surface waters is found in the freshwater discussion of hydrogen sulfide in this BE (section 5.5.1.10).

5.6.1.9.2 Water-column exposure toxicity data – acute hydrogen sulfide toxicity in marine waters

The Swinomish Tribe is not adopting an acute criterion for hydrogen sulfide; as such, this BE does not include an assessment of its potential effect. However, a discussion of acute toxicity data in the EPA (USEPA, 1976e) hydrogen sulfide criteria document is warranted, and the EPA (USEPA, 1976e) chronic hydrogen sulfide criterion under assessment in this BE was based largely on acute toxicity data where short-term survival was the measured endpoint. To this day, relatively little empirical hydrogen sulfide chronic toxicity data for aquatic life is available for aquatic species. Thus, the chronic toxicity evaluation presented below is largely based on empirical acute toxicity data translated into modeled chronic toxicity data.

Although not described as a saltwater toxicity test, EPA (USEPA, 1976e) briefly discusses results of a study by Holland et al. (1960) exposing 280 day old juvenile Chinook salmon (*Oncorhynchus tshawytscha*) to three concentrations of hydrogen sulfide in saltwater for a three day exposure duration. Results of Holland et al. (Holland et al., 1960) were no effect on survival at 1000 µg/L, a LOEC for survival of 1780 µg/L, and 100% mortality at the end of the three day exposure to a concentration of 3180 µg/L. The MATC (the geometric mean of the measured NOEC and LOEC) is 1330 µg/L.

The three day exposure duration used in Holland et al. (1960) does not meet the required four day exposure duration for an acute toxicity study to be used in the derivation of EPA acute criteria according to the Guidelines (C.E. Stephan et al., 1985). To our knowledge, Holland et al. (1960) is the only toxicity test in saltwater with an ESA listed fish species known to be present within the action area.

The freshwater acute toxicity data used in EPA (USEPA, 1976e) in support of the chronic hydrogen sulfide criterion of 2 µg/L is presented in the remainder of this section. It appears that an unstated assumption in EPA (USEPA, 1976e) is that the toxic concentrations of H_2S in fresh and saltwater are comparable to each other, thus not requiring a separate saltwater toxicity test database to develop a chronic H_2S criterion applicable to only marine species. The chronic EPA (USEPA, 1976e) hydrogen sulfide criterion was derived using both fresh and saltwater toxicity data, and is described in EPA (USEPA, 1976e) as applicable to both fresh and marine waters.

The acute toxicity data cited in EPA (USEPA, 1976e) for acute toxicity of H_2S to freshwater fish is summarized in the table below. When multiple 96 hour LC_{50}s are reported for a species within a single study, only the lowest 96 hour LC_{50} is shown below.

Table 103. Summary of the acute toxicity data cited in EPA (USEPA, 1976e) for acute toxicity of H2S to freshwater fish. When multiple 96 hour LC50s are reported for a species within a single study, only the lowest 96 hour LC50 is shown

Species	**96-hour LC_{50} (µg/L)**	**Citation**
Walleye (*Sander vitreum*)	7 (Sac fry)	(L. Smith & Oseid, 1972)
White sucker (*Catostomus commersoni*)	13 (Sac fry)	(L. Smith & Oseid, 1972)
Rainbow trout (*Oncorhynchus mykiss*)	< 5 (Sac fry)	(L. Smith & Oseid, 1972)
Northern pike (*Esox lucius*)	8 (Sac fry)	(Adelman & Smith, 1970)
Fathead minnow (*Pimephales promelas*)	7 (Fry)	(L. Smith & Oseid, 1974)
Brook trout (*Salvelinus fontinalis*)	17 (Juvenile)	(L. Smith & Oseid, 1974)

5.6.1.9.3 Additional marine acute toxicity data for hydrogen sulfide published since 1976

There are three summaries of the literature regarding hydrogen sulfide toxicity to aquatic life published since the hydrogen sulfide criterion was issued (USEPA, 1976e). EPA (USEPA, 1976b) describes results of acute and chronic toxicity test results for seven fish and eight invertebrate species. These studies, performed at EPA's Duluth, Minnesota aquatic toxicology laboratory, were apparently not completed in time for inclusion in the EPA (USEPA, 1976e) hydrogen sulfide criteria document. EPA (USEPA, 2009c) and Environment and Climate Change Canada and Health Canada (ECCC, 2017) both contain updated literature reviews of toxicity studies performed since 1976. These reviews were used to identify and obtain papers for use in this toxicity assessment of hydrogen sulfide.

Of the three reviews of hydrogen sulfide toxicity data published since the publication of the EPA (USEPA, 1976f) chronic marine criterion of 2 µg/L, EPA (USEPA, 1976b) contains no marine toxicity data. EPA (USEPA, 2009c) describes results from Bagarinao and Vetter (Bagarinao & Vetter, 1993), which contains a 96 hour LC_{50} for the marine California killifish (*Fundulus parvipinnis*) of 1430 µg/L. EPA (USEPA, 2009c) also contains a citation from Ivanov et al. (Ivanov, Usenko, & Parkhomenko, 1976) with a 24 hour LC_{100} of 2900 µg/L for the Black Sea turbot (*Rhombus maeoticus*). This species is not native to North America, and as the exposure duration was only 24 hours, Ivanov et al. (Ivanov et al., 1976) does not meet EPA data quality requirements for use in a biological evaluation of ESA listed species. Environment and Climate Change Canada and Health Canada (ECCC, 2017) does not contain any acute toxicity data for

marine fish not already discussed in this section, but does contain several newer studies on H_2S toxicity to marine invertebrates.

None of the reviews of hydrogen sulfide toxicity published since 1976 cite a study by Bagarinao and Vetter (1989) that performed a series of short-term toxicity tests on 13 species of tidal marsh and nearshore marine fish from the vicinity of San Diego, California. Eleven of the species were exposed to elevated concentrations of hydrogen sulfide for 24 hours or less, to measure time to event resulting in complete mortality. Two of the tidal marsh resident species, the long-jawed mudsucker (*Gillichthys mirabilis*) and the California killifish (*Fundulus parvipinnis*) were also tested to determine 96 hour LC_{50} concentrations. The 96 hour LC_{50}s for these two species were determined to be 625 µg/L and 833 µg/L for the mudsucker and killifish, respectively. These LC_{50}s are much higher than those for most other fish species, both freshwater and marine.

Bagarinao and Vetter (1993) have speculated that marine fish that live in close association with bottom sediments, particularly in salt marshes or estuaries where elevated concentrations of sulfide are naturally generated and occur, have adapted to survive in elevated H_2S concentrations in water. Species living in such habitats are able to tolerate higher H_2S concentrations than are more pelagic fish species, which are rarely or never exposed to elevated H_2S concentrations.

To summarize the literature review of acute toxicity data for hydrogen sulfide to marine fish species, EPA has been unable to identify any acute toxicity studies with mortality as the endpoint for any members of the *Salmonidae* or *Sebastidae*, the two families containing all of the ESA listed fish species within the action area. Only a few acute toxicity studies with other marine fish species with acceptable data quality have been found for other marine fish species.

Although there are multiple studies of the acute toxicity of hydrogen sulfide in freshwater published since the EPA (USEPA, 1976f) H_2S aquatic life criteria was published, among the most pertinent for this BE is that of Fung and Bewick (Fung & Bewick, 1980). Fung and Bewick (1980) measured 96 hour LC_{50}s for multiple life stages of four freshwater fish species: lake whitefish (*Coregonus clupeaformis*), yellow perch (*Perca flavescens*), largemouth bass (*Micropterus salmoides*) and rainbow trout (*Oncorhynchus mykiss*). Rainbow trout are one of the ESA listed species within the action area, while lake whitefish is also a member of the family Salmonidae, the same family to which all ESA listed salmonids in the action area belong.

The measured 96 hour LC_{50} for juvenile rainbow trout from Fung and Bewick (1980) was 7 µg/L hydrogen sulfide, the lowest 96 hour LC_{50} of any rainbow trout lifestage tested in this study. Fung and Bewick (1980) observed a general trend of lifestage sensitivity where the youngest rainbow trout lifestages were the most tolerant of hydrogen sulfide (e.g. 96 hour LC_{50} of 47 µg/L for sac fry) to the oldest lifestages being the most sensitive to hydrogen sulfide (e.g. the previously mentioned 96 hour LC_{50} of 7 µg/L for juveniles and a 12 µg/L 96 hour LC_{50} for adult rainbow trout.

Lake whitefish are even more sensitive to hydrogen sulfide than are rainbow trout, with a sac fry 96 hour LC_{50} of 2 µg/L and a juvenile 96 hour LC_{50} of 12 µg/L. Fung and Bewick (1980) did not report a 96 hour LC_{50} for adult whitefish. Notably, the lake whitefish 96 hour LC_{50} of 2 µg/L is identical to the EPA (USEPA, 1976f) chronic hydrogen sulfide criterion of 2 µg/L. As it is likely that the chronically toxic concentration of hydrogen sulfide to lake whitefish and other members of the family Salmonidae is lower than the empirically measured acutely toxic concentration of 2 µg/L, the acute toxicity results in

Fung and Bewick (1980) for Salmonidae provide evidence that the EPA (USEPA, 1976f) chronic hydrogen sulfide criterion of 2 µg/L may not be protective of ESA listed Salmonidae within the action area.

Although not a member of the Salmonidae, Fung and Bewick (1980) found yellow perch sac fry to be even more sensitive to hydrogen sulfide than lake whitefish or rainbow trout. The yellow perch 24 hour LC_{50} was 2 µg/L, the same concentration as was the 96 hour LC_{50} of 2 µg/L for lake whitefish sac fry. The 48, 72 and 96 hour LC_{50}s for yellow perch sac fry were all <2 µg/L. Finally, Fung and Bewick (1980) found largemouth bass (*Micropterus salmoides*) to be the most tolerant of the four species they tested, with a 96 hour LC_{50} of 11 µg/L for sac fry. Juvenile largemouth bass had a 96 hour LC_{50} of 63 µg/L, the highest 96 hour LC_{50} of any lifestage of any species tested.

Several other acute toxicity studies with salmonids are available. Smith and Oseid (L. Smith & Oseid, 1975), in a study not reported in EPA (USEPA, 1976a) observed a 96 hour LC_{50} of 21.6 µg/L for brook trout swim-up fry. EPA (USEPA, 1976b) reported a brook trout sac fry 96 hour LC_{50} as low as 13.8 µg/L. Brook trout are in the same genus (*Salvelinus*) as bull trout, and are a taxonomically close relative to one of the ESA listed species within the action area. Reynolds and Haines (F. A. Reynolds & Haines, 1980) reported a 96 hour LC_{50} of 7 µg/L for brown trout (*Salmo trutta*) sac fry. EPA (USEPA, 1976b) reported a juvenile rainbow trout 96 hour LC_{50} of 12.5 µg/L, within a factor of two of the 7 µg/L 96 hour LC_{50} for juvenile rainbow trout reported by Fung and Bewick (1980).

Other 96 hour LC_{50} values for various fish species reported in EPA (USEPA, 1976b) include a relatively low 8.6 µg/L for bluegill (*Lepomis macrochirus*) swim-up fry, and a relatively high 25 µg/L for goldfish fry (*Carassius auratus*).

5.6.1.9.4 Water-column exposure toxicity data - chronic hydrogen sulfide toxicity in marine water

EPA (1976a) contains no information on the chronic toxicity of hydrogen sulfide to any marine fish species.

The chronic criterion of 2.0 µg/L in EPA (1976a) is derived from a chronic study with bluegills reported in Smith and Oseid (L. Smith & Oseid, 1974). Egg production was significantly reduced after a 46 day exposure to 1.4 µg/L hydrogen sulfide. Juvenile white suckers showed growth reductions from exposure to 1 µg/L hydrogen sulfide according to EPA (1976a), although the original literature citation for the white sucker growth reduction was not provided in EPA (1976a).

5.6.1.9.5 Additional marine water chronic toxicity data for hydrogen sulfide published since 1976

The only H_2S chronic toxicity study with a marine fish species EPA has been able to identify published since 1976 is an 18 week study of Atlantic salmon (*Salmo salar*) smolt growth published by Kiemer et al. (Kiemer, Black, Lussot, Bullock, & Ezzi, 1995). This study employed a pulsed exposure system to simulate tidal cycles. Two exposure tanks plus a third control tank containing 100 smolts each were supplied with sodium sulfide solution by a peristaltic pump which was connected to a timer programmed to activate the pump every 6 hours for 10 minutes. After each 10 minute dose the hydrogen sulfide concentration in the tanks reached its maximum of 7.8 µM (approximately 265 µg/L, the only exposure concentration tested). The water supply was adjusted such that the concentration of hydrogen sulfide dropped to zero after 3 hours. The 6 hour interval was chosen to simulate the tidal rhythm. Smolt growth as weight was measured every two weeks during the experiment. No significant difference in smolt growth was

observed by Kiemer et al. (1995) between the exposure tanks and the control tank after four weeks (28 days), nor at any other exposure duration except for the eight week and 18 week exposures. Due to the pulsed exposure to H_2S and the exposure to only one H_2S concentration during Kiemer et al. (1995), this study does not meet the data quality requirements of the Guidelines for use in deriving or assessing chronic H_2S toxicity to marine fish.

The remainder of this section describes the chronic toxicity results with freshwater fish that can be used as a surrogate for marine fish toxicity test results to estimate H_2S chronic toxicity to marine fish species.

EPA (USEPA, 1976b) measured the chronic toxicity of hydrogen sulfide to seven species of freshwater fish: fathead minnows, goldfish, bluegill, walleye, white sucker, brook trout, and rainbow trout. Long-term effects on survival, growth, reproduction, and behavior were evaluated in studies whose durations ranged from 45 days (brook trout reproduction) to 826 days (bluegill multi-generation study on survival, growth and reproduction). Recommendations from EPA (USEPA, 1976b) included that 0.002 mg/liter H_2S be considered a safe limit for protection of all fish species and bottom-inhabiting invertebrates in areas of lakes and streams not used for spawning by nesting fish, and that in areas used for reproduction by nesting fish concentrations of 0.001 mg/liter H_2S not be exceeded.

A summary of the lowest chronic MATCs, (calculated as the geometric mean of the NOEC and LOEC published in Table 133 of EPA (USEPA, 1976b) for six fish species is presented in the table below.

Table 104. Lowest chronic MATCs, for hydrogen sulfide for six fish species

Species	Lifestage	MATC (µg/L)	Endpoint
Fathead minnow	Sac fry	0.0047	Growth, reproduction
Goldfish	Adult	0.0071	Reproduction
Bluegill	Adult	<0.0007	Reproduction
Rainbow trout	Sac fry	0.0040	Growth
Brook trout	Adult	<0.0055	Reproduction
Walleye	Juvenile	0.0040	Survival

Confidence limits were not provided in EPA (USEPA, 1976b) for any of the measured endpoints, only the range of measured exposure concentrations in each study, the observed NOEC and the observed LOEC values. Thus, it is unknown whether any of the chronic MATCs in the above table, except for that of the bluegill, have lower confidence limits below the 0.002 µg/L hydrogen sulfide chronic criterion.

The lowest chronic LOEC identified in a freshwater fish since the publication of the chronic hydrogen sulfide criterion (USEPA, 1976f) is a 5.4% growth reduction in a 28 day study of juvenile Asian redtail catfish (*Hemibagrus nemurus*) exposed to 0.5 µg/L hydrogen sulfide (Hoque et al., 1998). The 0.5 µg/L H_2S concentration was the lowest of the four H_2S exposure concentrations tested, meaning a chronic NOEC cannot be calculated for this species. As the Asian redtail catfish is native to southeast Asia, and has not been introduced into North America, the toxicity test results for this species cannot be used in the derivation of EPA aquatic life criteria according to the Guidelines. But the 0.5 µg/L chronic LOEC in Hoque et al. (1998) is the lowest known chronic toxicity effect concentration for hydrogen sulfide of any freshwater fish species.

5.6.1.9.6 Acute-chronic ratio for hydrogen sulfide

EPA water quality criteria documents do not contain any estimates of or empirically determined ACRs for use with hydrogen sulfide toxicity data. EPA (USEPA, 1976b) does summarize several application factors for several taxonomic groups of fish derived from the empirical toxicity test data presented in the report. For trout, EPA (USEPA, 1976b) reports ACRs between 3 - 5 (converted from the application factors originally reported in the document), and ACRs between 15 – 20 in warmwater fish.

However, the European Centre for Ecotoxicology and Toxicology of Chemicals (ECETOC, 2003) has calculated ACRs for a number of chemicals, including hydrogen sulfide using an ACR derivation approach that combines toxicity data from mixed species (i.e. where an acute toxicity point for a chemical from one or more species could be compared with a chronic value from one or more species, not necessarily including the species used to derive the acute value). This approach differs from the EPA approach to ACR determination, which requires both empirical acutely toxic concentrations and empirical chronic no effect concentrations for the same species, although not necessarily from the same study to derive the ACR.

For freshwater fish, ECETOC (ECETOC, 2003) has estimated a hydrogen sulfide ACR of 8.2, within the range of ACRs for fish species described in EPA (USEPA, 1976b). As the ECETOC estimated hydrogen sulfide ACR is also comparable to the median ACR of 8.3 calculated from a compilation of 456 same species pairs ACRs by Raimondo et al. (S. Raimondo et al., 2007), used in this BE's methodology as a default ACR for chemicals without a published empirically determined ACRs, we employ the ECETOC (ECETOC, 2003) hydrogen sulfide ACR of 8.2 for converting empirical acute toxicity LC_{50}s to estimated chronic MATCs or NOECs in this BE.

5.6.1.9.7 Effects assessment of chronic hydrogen sulfide criterion on listed species

Environment and Climate Change Canada and Health Canada (ECCC, 2017) suggest that marine organisms generally are less sensitive to hydrogen sulfide than freshwater organisms. The limited marine toxicity data on aquatic species appears to support this suggestion, with the caveat that most empirical marine toxicity data for H_2S was derived from studies with much shorter exposure durations than the chronic toxicity data for freshwater species.

In the near complete absence of empirical chronic H_2S toxicity test results with marine fish, and only a limited number of acceptable quality acute toxicity studies with marine fish, EPA concludes that the toxicity assessment for ESA listed fish in the marine portions of the action area should be based primarily on toxicity test results with freshwater fish species. This approach is the same implicitly assumed by EPA during the development of the original EPA (USEPA, 1976f) hydrogen sulfide chronic criterion for marine waters.

EPA does not assume that the near complete absence of empirical marine toxicity data leads to a conclusion that adverse effects of hydrogen sulfide at the criterion concentration also are absent.

The 96 hour LC_{50} of 2 µg/L for lake whitefish, a member of the family *Salmonidae*, equivalent to the chronic criterion for all species is indicative that the chronic hydrogen sulfide criterion may not be protective of both lake whitefish and other *Salmonidae* during chronic exposures. Further evidence of this lack of protectiveness for members of the *Salmonidae* can be evaluated by dividing the lowest measured 96 hour LC_{50} for each salmonid species for which empirical acute toxicity data are available by the assumed ACR for hydrogen sulfide of 8.2. This information is shown in the table below.

Table 105. Further evidence of lack of protectiveness for members of the Salmonidae evaluated by dividing the lowest measured 96 hour LC50 for each salmonid species for which empirical acute toxicity data are available by the assumed ACR for hydrogen sulfide of 8.2

Species	**H_2S 96 hour LC_{50}**	**Citation**	**H_2S Chronic NOEC (96 hour LC_{50} / ACR**
Brown trout (*Salmo trutta*)	7	(F. A. Reynolds & Haines, 1980)	0.85
Rainbow trout (*Oncorhynchus mykiss*)	< 5	(L. Smith & Oseid, 1972)	<0.61
Brook trout (*Salvelinus fontinalis*)	13.8	(USEPA, 1976b)	1.7
Lake whitefish (*Coregonus clupeaformis*)	2	(Fung & Bewick, 1980)	0.24

The above table contains estimated chronic NOEC concentrations for species within four genera of *Salmonidae* (*Coregonus*, *Oncorhynchus*, *Salmo*, *Salvelinus*) that are all lower than the chronic hydrogen sulfide criterion of 2 µg/L. All ESA listed salmonids within the action area are members of either the genus *Oncorhynchus* or genus *Salvelinus*. Thus, evidence indicates that the chronic hydrogen sulfide criterion may not protect multiple salmonid species from chronic toxicity of hydrogen sulfide.

Evidence further indicates that members of the *Salmonidae* may not be the only fish species where the chronic criterion for hydrogen sulfide is not protective of fish from chronic effects of hydrogen sulfide. The 96 hour LC_{50} of < 2 µg/L determined by Fung and Bewick (1980) for yellow perch sac fry indicates that the chronic criterion may adversely affect short term survival of yellow perch (family *Percidae*), as well as resulting in longer term adverse chronic effects of hydrogen sulfide. The 96 hour LC_{50} of 7 µg/L determined for walleye (also a member of the *Percidae*) by Smith and Oseid (L. Smith & Oseid, 1972) divided by the hydrogen sulfide ACR of 8.2 results in a chronic walleye NOEC of 0.85 µg/L, lower than the chronic 2 µg/L criterion. Other taxonomic families where either the empirical or estimated chronic LOEC or NOEC of one or more members of the family is lower than the 2 µg/L chronic criterion include members of the *Catostomidae* (white sucker), *Centrarchidae* (bluegill), *Cyprinidae* (fathead minnow) and *Esocidae* (northern pike).

The estimation of chronic NOEC hydrogen sulfide concentrations for multiple fish species from multiple taxonomic families provides support for the following conclusion. Exposure at the level of the Swinomish Tribe's chronic criterion of 2 µg/L for hydrogen sulfide is likely to adversely affect listed species as shown below.

5.6.1.9.8 Summary of effect determinations for marine hydrogen sulfide criterion

EPA has made the following determinations on the chronic hydrogen sulfide criterion for the listed species in the marine portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above, and the indirect effects analysis for prey availability provided in section 5.8 (the multiple routes of exposure analysis was not relevant as indicated in section 5.7). The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Chronic criterion:

Rainbow trout (steelhead) – **Likely to adversely affect** (exposure to water)

Chinook salmon – **Likely to adversely affect** (exposure to water)

Chum salmon – **Likely to adversely affect** (exposure to water)

Bull trout – **Likely to adversely affect** (exposure to water)

Bocaccio – **Likely to adversely affect** (exposure to water)

Yelloweye rockfish – **Likely to adversely affect** (exposure to water)

Marbled murrelet – **Not likely to adversely affect**

5.6.1.10 Lead – Marine

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = 210 µg/L

Chronic (CCC) criterion = 8.1 µg/L

CAS ID 7439-92-1

5.6.1.10.1 Introduction

In marine waters, the predominant form of lead is lead carbonate (Hart, 1982). Lead occurs in the +2 and +4 valency states, although elemental lead is relatively soluble in soft and acidic water (Fergusson, 1990), and therefore, plays a significant role in the input of lead into the aquatic environment. The solubility of most lead compounds in water is poor; therefore, much of the lead entering marine systems are from precipitation and runoff. The toxicity of dissolved lead to various marine species is dependent on its speciation and the possibility of bioconcentration, bioaccumulation and biomagnification in these species. The distribution and forms of lead present in the marine environment is important and could be used when predicting its ecological effects (Branica & Konrad, 2015).

5.6.1.10.2 Water-column exposure toxicity data - acute criterion for lead in marine water

The numeric values for both the acute and chronic marine lead criteria have not been changed since the publication of EPA (USEPA, 1985d).

Acute toxicity data for lead to saltwater organisms are available for four fish species and nine invertebrates. Cardin (J.A. Cardin, 1981) conducted toxicity tests with two fish species, Sheepshead minnow *(Cyprinodon variegatus*) and Inland silverside (*Menidia beryllina*) and reported identical LC_{50}s of 2986 µg/L for each test. In the two tests, less than 50 percent of the test fish were killed at 2986 µg/L, which is the solubility of lead in saltwater under the test conditions. Also, the acute toxicity value (LC_{50}) for the Mummichog (*Fundulus heteroclitus*) was 299565 µg/L (Dorfman, 1977). The LC_{50} for Atlantic silverside (*Menidia menidia*) was > 9510 µg/L (USEPA, 1985d).

5.6.1.10.3 Additional marine acute toxicity data for lead published since 1985

A toxicity study by Rajkumar (2012) reported a 96-hour LC_{50} for Striped mullet (*Mugil cephalus*) of 12230 µg/L. Another acute toxicity test by Toxscan Inc. (1991) reported a 96-hour LC_{50} of 44761 µg/L for Inland Silverside (*Menidia beryllina*). In addition, EPA (USEPA, 2008b) reported an EC50 of 108610 µg/L for a mullet (*Liza vaigiensis*). The other two marine invertebrate species with reported 96-hour LC_{50}s in the study by Rajkumar (2012) are the crustacean (*Panaeus monodon*) and the green clam (*Perna viridis*) of 390 µg/L and 3160 µg/L, respectively. There are no empirical marine acute toxicity data for Swinomish ESA listed species.

5.6.1.10.4 Effects assessment of acute marine lead criterion on listed species

Acute toxicity data are available for 6 marine fish species which include the 96-hour LC50s for Inland silverside (44761µg/L), for adult Mummichog (299565 µg/L), Striped mullet (12230 µg/L), Atlantic silverside (>9510 µg/L), Sheepshead minnow (2986 µg/L) and the 108610 µg/L for the mullet *(Liza vaigiensis*). Therefore, effects assessment concentration has been calculated from a species sensitivity distribution yielding the 5th percentile of the available acute toxicity data for marine fish species. The 5th percentile of the SSD for 96-hour LC50s is 1082.26 µg/L. Dividing this 96-hour LC50 by 2.27 to estimate an LC_{low} (1082.26/2.27) results in an acute effects assessment concentration of 476.8 µg/L. The calculated marine acute effects assessment concentrations for the ESA listed fish species is well above the marine acute lead criterion of 210 µg/L. This indicates that exposure at the level of the marine acute lead criterion is protective and not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio or yelloweye rockfish**.**

5.6.1.10.5 Water-column exposure toxicity data - chronic criterion for lead in marine systems

The EPA (USEPA, 1985d) aquatic life criteria for lead contain no empirical chronic toxicity data for any fish species. The mysid (Mysidopsis bahia) is the only saltwater species with a chronic test data (USEPA, 1985d). The most sensitive observed adverse effect was reduced spawning and the resulting chronic value was 25.08 µg/L. The 96-hour LC50 for the same species in the same study was 3130 µg/L, producing and acute-chronic ratio of 124.8.

5.6.1.10.6 Additional chronic marine toxicity data for lead published since 1985

A study by Hoang (2015) reported an EC_{20} of 55 µg/L for a 28-day chronic survival and growth test on topsmelt (*Atherinops affinis*). Also, Parametrix (2010) published an EC_{20} of 265 µg/L for a 28-day chronic study on growth and survival for Cyprinodon variegatus (*sheepshead minnow*). The Parametrix (2010) study literature was not publicly available and as such was not used in this assessment because it did not meet one of the criteria for use in the ESA Assessment.

5.6.1.10.7 Effects assessment of chronic marine lead criterion on listed species

There is no empirical chronic lead toxicity data available for fish except the 28-day chronic survival value of 55 µg/L for topsmelt. There is no toxicity data available for ESA listed species within the action area.

The effects assessment concentration has been calculated from a species sensitivity distribution (SSD) yielding the 5th percentile of the available acute toxicity data for marine fish species. The 5th percentile of the SSD for 96-hour LC50s is 1082.26 µg/L. The acute-chronic ratio (ACR) is 51.29 (USEPA, 1985d). Therefore, the final chronic value is 1082.26 /51.29 = 21.1 µg/L. The final chronic value estimate of 21.1 µg/L for all fish species is higher than the chronic marine lead criterion of 8.1 µg/L. This indicates

that exposure at the level of the marine chronic lead criterion is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio or yelloweye rockfish.

5.6.1.10.8 Summary of effect determinations for marine lead criterion

EPA has made the following determinations on the acute and chronic lead criteria for the ESA listed species in the marine portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.6.1.11 Malathion – Marine

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = None

Chronic (CCC) criterion = 0.1 µg/L

CAS ID 121-75-5

5.6.1.11.1 Water-column exposure toxicity data - chronic criterion for malathion in marine waters

Malathion is reported to be highly toxic to all taxonomic groups of aquatic animals including fish, amphibians and invertebrates (USEPA, 2009). There are limited studies on the effects of malathion on aquatic species in marine water; however, life-cycle toxicity tests have been conducted in marine water on sheepshead minnow (*Cyprinodon variegatus)* with malathion. Hansen and Parrish (D. J. Hansen & P. R. Parrish, 1977) conducted a 20-week partial life cycle test on sheepshead minnow with malathion in saltwater. The sheepshead minnows were exposed continuously to measured concentrations of 4, 9, 18, 37 and 86 µg/L of malathion from the reproductive portion of their life cycle through the growth of progeny to determine the MATCs because no other estuarine fish has been used for this purpose (D. J. Hansen & P. R. Parrish, 1977).

Results indicated that malathion affected the survival of parental fish (adult fish) at 37 and 86 µg/L, and F_1 generation (fry) exposed to concentrations of 9 and 18 µg/L had increased mortality. The mortality of F_1 fry in 9 and 18 µg/L of malathion was significantly greater than that of the F_1 control fry. The study by D. J. Hansen and P. R. Parrish (1977) reported MATCs for malathion as > 4 or < 9 µg/L.

5.6.1.11.2 Effects assessment of chronic malathion criterion on listed species in marine waters

It should be noted that because of 'low confidence' in exposure concentrations predicted for saltwater habitats, it was conservatively (i.e., defer to the protection of the species in the face of uncertainty) concluded during the pesticide consultation that salmonids were most at risk while in coastal, estuarine areas where they spend a small portion of their lives (NMFS, 2017a). In other words, this conclusion relied on exposure assumptions rather than toxicity data.

There are no current studies on the effects of malathion on salmonids or rockfish. The lowest available chronic effect concentration for malathion in marine water in fish was the study by Hansen and Parrish (D. J. Hansen & P. R. Parrish, 1977), conducted with sheepshead minnow (*Cyprinodon variegatus).* The study reported a NOEC of 4 µg/L (<140 days) for sheepshead minnow.

The NOEC reported in the study by D. J. Hansen and P. R. Parrish (1977) is higher than the proposed malathion marine chronic criterion of 0.1 µg/L. Assuming the minimum chronic effect concentration for the listed salmonids and rockfish in the marine portions of the action area is greater than or equal to 4 µg/L, exposure at the level of the proposed malathion marine chronic criterion of 0.1 µg/L is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio, and yelloweye rockfish.

5.6.1.11.3 Summary of effect determinations for marine malathion criterion

EPA has made the following determinations on the chronic malathion criterion for the listed species in the marine portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.6.1.12 Nickel – Marine

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = 74 µg/L

Chronic (CCC) criterion = 8.2 µg/L

CAS ID 7440-0-20

5.6.1.12.1 Introduction

In open ocean areas, total Ni levels range from 0.2 to 2 mg/L (Boyden, 1975; WHO, 1991). In coastal areas, however, where impacts of industrial and diffuse sources may be observed, dissolved Ni can reach concentrations up to 250 mg/L (Boyden, 1975; Eisler, 1998; Ezekwe & Edoghotu, 2015; Knauer, 1977; Wells, Smith, & Bruland, 2000), with industrially contaminated water ranges upwards of 2 mg/L (Chau & Kulikovsky-Cordeiro, 1995).

In freshwater settings the toxicity of the trace metal nickel (Ni) is relatively well understood. However, little is known regarding Ni toxicity in waters of higher salinity, where factors such as water chemistry and the physiology of estuarine and marine biota would be expected to alter toxicological impact (Blewett & Leonard, 2017). As in freshwater, three main mechanisms of Ni toxicity exist: ionoregulatory impairment, inhibition of respiration, and promotion of oxidative stress. But, unlike in freshwater biota, where mechanisms of toxicity are largely class-specific, the delineation of toxic mechanisms between different species is less defined. In general, despite changes in Ni speciation in marine waters, organism physiology appears to be the main driver of toxic impact, this will need to be taken into consideration when adapting regulatory tools (such as bioavailability normalization) from freshwater to estuarine and marine environments (Blewett and Leonard, 2017).

5.6.1.12.2 Water-column exposure toxicity data - acute criterion for nickel in marine water

Numeric values for both the acute and chronic marine nickel criteria have not been changed since the publication of EPA (USEPA, 1986e). The EPA (USEPA, 1995) update of several criteria, including freshwater and saltwater for nickel resulted in a change to 470 µg/L for freshwater acute and 52 µg/L for chronic criteria at 100 mg/L hardness.

EPA (USEPA, 1986e) reported acute toxicity of nickel for 18 marine species of invertebrates and 4 fish. The toxicity for the invertebrates ranged from 152 µg/L for juveniles of a mysid, *Heteromysis formosa* (Gentile, Gentile, Hairston, & Sullivan, 1982) to 1100 mg/L for clams, *Macoma balthica* (V. Bryant, Newbery, McLusky, & Campbell, 1985). The 96-hour LC_{50}s for fish range from 7958 µg/L for Atlantic silverside, *Menidia menidia* larvae (Cardin, 1985) to 350000 µg/L for adult mummichog, *Fundulus heteroclitus* (Eisler & Hennekey, 1977). In addition, the acute toxicity of nickel appears to be related to salinity, but the form of the relationship appears to be species-dependent (USEPA, 1986e).

5.6.1.12.3 Additional marine acute toxicity data for nickel published since 1986

Hunt et al. (2002)Hunt et al. (2002) reported 96-hour LC50s of 25650 µg/L and 148.6 µg/L for topsmelt (*Atherinops affinis*) and mysid (*Mysidopsis intii*), and 48-hour EC50 of 145.5 µg/L for red abalone (*Haliotis rufescens*), respectively. Since the available 96-hour LC50 for marine fish is 25650 µg/L, which is higher than the acute criterion of 74 µg/L, this means that the criterion will be protective.

5.6.1.12.4 Effects assessment of acute marine nickel criterion on listed species

Acute toxicity data are available for 5 marine fish species which include the 96-hour LC50s for Atlantic silverside (7958 µg/L), adult mummichog (149900 µg/L), striped bass (21000 µg/L), tidewater silverside (38000 µg/L) and topsmelt (25650 µg/L). The effects assessment concentration has been calculated from a species sensitivity distribution yielding the 5th percentile of the available acute toxicity data for marine fish species. The 5th percentile of the SSD for 96-hour LC50s is 6249 µg/L. Dividing this 96-hour LC50 by 2.27 to estimate an LC_{low} results in an acute effects assessment concentration of 2753 µg/L. The calculated marine acute effects assessment concentrations for the ESA listed fish species is well in excess of the marine acute nickel criterion of 74 µg/L. This indicates that exposure at the level of the marine acute nickel criterion is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio or yelloweye rockfish.

5.6.1.12.5 Water-column exposure toxicity data - chronic criterion for nickel in marine systems

The EPA (USEPA, 1986e) aquatic life criteria for nickel contain no empirical chronic toxicity data for any fish species. The mysid (*Mysidopsis bahia*) is the only saltwater species with an acceptable chronic test data (USEPA, 1986e). Lussier et al. (S. Lussier, Gentile, & Walker, 1985) reported that chronic exposure to nickel reduced survival in a number of young mysid at 141 µg/L and above and not at 61 µg/L and lower. Thus, the chronic nickel value for Mysid is 92.74 µg/L, with an acute-chronic ratio of 5.478.

5.6.1.12.6 Additional chronic marine toxicity data for nickel published since 1986

A study by Hunt et al. (2002) reported a 40-day chronic survival value of 4279 µg/L for topsmelt (*Atherinops affinis*); a 20-day chronic growth value of 26.43 µg/L for red abalone (*Haliotis rufescens*) and a 28-day chronic survival value of 22.09 µg/L for mysid (*Mysidopsis bahia*). In addition, acute-chronic ratios (ACRs) were also reported for these toxicity values as follows: 6.220 for topsmelt, 5.505 for red abalone and 6.727 for mysid.

5.6.1.12.7 Effects assessment of chronic marine nickel criterion on listed species

There is no empirical chronic nickel toxicity data available for fish except the 40-day chronic survival value of 4279 µg/L for topsmelt. There is no toxicity data available for ESA listed species within the action area.

The effects assessment concentration has been calculated from a species sensitivity distribution (SSD) yielding the 5th percentile of the available acute toxicity data for marine fish species. The 5th percentile of the SSD for 96-hour LC50s is 6249 µg/L. The acute-chronic ratio (ACR) is 17.99. Therefore, the final chronic value is 6269/17.99 = 347.4 µg/L. The final chronic value estimate of 347.4 µg/L for all fish species is higher than the chronic marine nickel criterion of 8.2 µg/L. This indicates that exposure at the level of the marine chronic nickel criterion is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio or yelloweye rockfish.

5.6.1.12.8 Summary of effect determinations for marine nickel criteria

EPA has made the following determinations on the acute and chronic nickel criteria for the ESA listed species in the marine portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.6.1.13 Nonylphenol – Marine

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion – 7.0 µg/L

Chronic (CCC) criterion – 1.7 µg/L

CAS ID – 25154-52-3 (general for all nonylphenols)

84852-15-3 (branched 4-nonylphenols)

104-40-5 (4-n-nonylphenol)

5.6.1.13.1 Introduction to nonylphenols

The term nonylphenol refers to a complex mixture of nonylphenol congeners. The nine carbon nonyl group may be branched or linear and bind at various locations around the phenol ring. Most commonly the nonyl group is substituted at the para- or 4- position, but small amounts of ortho- (2-) and meta- (3-) substituted nonylphenols also exist in commercially produced nonylphenol. Many manufacturers incorrectly use the straight chain 4-nonylphenol identification (CAS ID 104-40-5) when referring to the branched nonylphenols. Straight chain 4-nonylphenol is commonly used in aquatic toxicity testing. CAS ID 84852-15-3 corresponds to the most widely produced commercial nonylphenol, a mixture of branched 4-nonylphenols. CAS ID 25154-52-3 has also been used to describe mixtures of commercially produced nonylphenol. The EPA (USEPA, 2005a) nonylphenol criteria document states that it addresses the CAS numbers 84852-15-3 and 25154-52-3, two of the CAS ID numbers used in this BE to perform ECOTOX database searches for toxicity data. In the ECOTOX database search for nonylphenol, we have also searched for straight chain 4-nonylphenol (CAS ID 104-40-5) due to its use in toxicity testing, and because it can be a congener present in commercially produced nonylphenol mixtures.

5.6.1.13.2 Water-column exposure toxicity data - acute criterion for nonylphenol in marine waters

None of the toxicity studies used to derive the USEPA (2005a) nonylphenol marine acute criterion were members of either the *Salmonidae* or *Sebastidae* families, the two families of ESA listed fish species with members of these families present within the action area. The only three fish species with toxicity data used to derive the marine acute nonylphenol criterion were winter flounder (*Pseudopleuronectes americanus*), sheepshead minnow (*Cyprinodon variegatus*) and inland silversides (*Menidia beryllina*).

Of the 11 species used to derive the USEPA (2005a) marine acute nonylphenol criterion, winter flounder was the most sensitive of the 11 species, with a 96 hour LC_{50} of 17 µg/L (S.M. Lussier, D. Champlin, J. Livolsi, S. Poucher, & R.J. Pruell, 2000). Lussier et al. (2000) also determined the 96 hour LC_{50} for inland silversides (70 µg/L) used to derive the marine acute nonylphenol criterion. This inland silversides 96 hour LC_{50} was the median (i.e. 6th) most sensitive of the 11 species used in the derivation of the acute nonylphenol criterion. Lussier et al. (2000) also performed one of the two studies with sheepshead minnow used to derive the acute nonylphenol criterion, measuring a 96 hour LC_{50} of 142 µg/L. The other sheepshead minnow 96 hour LC_{50} used in criteria derivation was performed by Ward and Boeri (1990), who measured a 96 hour LC_{50} of 310 µg/L. Combining the two 96 hour LC_{50}s for sheepshead minnow resulted in a species mean acute value of 210 µg/L.

A marine nonylphenol 96-hour LC_{50} value (McLeese et al., 1980) for one salmonid, the Atlantic salmon (*Salmo salar*) was evaluated for use in USEPA (2005a), but not included in criteria derivation for several reasons. Reasons included the lack of reported control responses, as well as the nonylphenol exposure concentrations not being reported. USEPA (2005a) reports four 96 hour LC_{50}s of 130 µg/L, 160 µg/L, 190

µg/L and 900 µg/L from McLeese et al. (1980). However, the current record of the McLeese study in ECOTOX as well as our review of the study only report the 900 µg/L 96 hour LC_{50} for Atlantic salmon.

5.6.1.13.3 Additional marine acute toxicity data for nonylphenol

There is literature published since 2005 describing 96-hour LC_{50}s to several marine fish species. There is also a European study published in the early 1970s that is not included in USEPA (2005a), but which appears suitable for use in this BE. F.L. Mayer et al. (2008) reported a 96 hour LC_{50} of 132 µg/L to the Leon Springs pupfish (*Cyprinodon bovinus*). Mayer et al. (2008) also reported two additional 96 hour LC_{50}s for sheepshead minnows: 300 µg/L at 15 ‰ salinity, and 472 µg/L at 2 ‰ salinity. Combining the two sheepshead minnow 96 hour LC_{50}s given in USEPA (2005a) with the two sheepshead minnow 96 hour LC_{50}s published by Mayer et al. (2008) yield a species mean acute value of 281 µg/L for sheepshead minnows.

Hamlin, Marciano, and Downs (2015) determined a 96 hour LC_{50} of 175 µg/L for the orchid dottyback (*Pseudochromis fridmani*), a coral reef fish. Kelly and Di Giulio (2000) determined a 96 hour LC_{50} of 204 µg/L to larval mummichog (*Fundulus heteroclitus*) and a 96 hour LC_{50} of 5000 µg/L for embryos of mummichogs. Swedmark, Braaten, Emanuelsson, and Granmo (1971) report 96 hour adult LC_{50}s of 2500 µg/L for Atlantic cod (*Gadus morhua*) and of 3000 µg/L for European flounder (*Platichthys flesus*), a European native fish which has been introduced into the United States. Breves et al. (2018) saw no effects on Atlantic salmon smolts exposed to 0.88, 8.8 and 88 µg/L of nonylphenol for four days.

5.6.1.13.4 Effects assessment of marine acute nonylphenol criterion on listed species

No listed species within the marine portions of the action area have empirical toxicity data for any species within the same genera, family or order as the listed species. The closest taxonomic relatives to rockfish and salmonids are in the class *Actinopterygii*. The geometric mean 96 hour LC_{50}s (i.e. Species Mean Acute Values) for the eight species in the same class as the Swinomish Tribe ESA listed species are presented in the table below.

Table 106. Species mean acute values for nonylphenol exposure of marine species

Common name	Scientific name	Species mean acute value, µg/L
Winter flounder	*Pseudopleuronectes americanus*	17
Inland silversides	*Menidia beryllina*	70
Leon Springs pupfish	*Cyprinodon bovinus*	132
Orchid dottyfish	*Pseudochromis fridmani*	175
Mummichog	*Fundulus heteroclitus*	204
Sheepshead minnow	*Cyprinodon variegatus*	281
Atlantic cod	*Gadus morhua*	2500
European flounder	*Platichthys flesus*	3000

According to the methods employed in this BE, the 5th percentile of a species sensitivity distribution (SSD) of the class level species mean acute values, divided by 2.27 to convert the 5th percentile 96 hour LC_{50}s to a 96 hour LC_{low} is compared to the acute nonylphenol criterion of 7 µg/L. This is the 7th and lowest tier of the approaches used to assess acute toxicity data in this BE.

As shown in the figure below, the 5th percentile of the SSD for the empirical fish acute toxicity data for nonylphenol is 12.18 µg/L. Dividing the 5th percentile of the fitted SSD (r^2 = 0.929) by 2.27 results in an LC_{low} or acute effects assessment concentration of 5.37 µg/L.

Proportion	Probit	Log Central Tendency	SSQ	Log Upper PI	Log Lower PI	Central Tendency	Upper PI	Lower PI
0.05	3.355	1.086	0.077	1.627	0.545	12.180	42.319	3.506
0.1	3.718	1.372	0.069	1.883	0.862	23.574	76.422	7.272
0.2	4.158	1.720	0.062	2.202	1.237	52.445	159.403	17.255
0.4	4.747	2.184	0.057	2.647	1.722	152.785	443.177	52.673
0.5	5.000	2.384	0.056	2.844	1.924	242.142	699.028	83.877
0.7	5.524	2.798	0.058	3.267	2.329	628.091	1850.359	213.201
0.8	5.842	3.048	0.062	3.531	2.566	1117.972	3397.965	367.827
0.9	6.282	3.396	0.069	3.906	2.885	2487.164	8062.823	767.223
0.95	6.645	3.682	0.077	4.223	3.142	4813.810	16725.377	1385.485

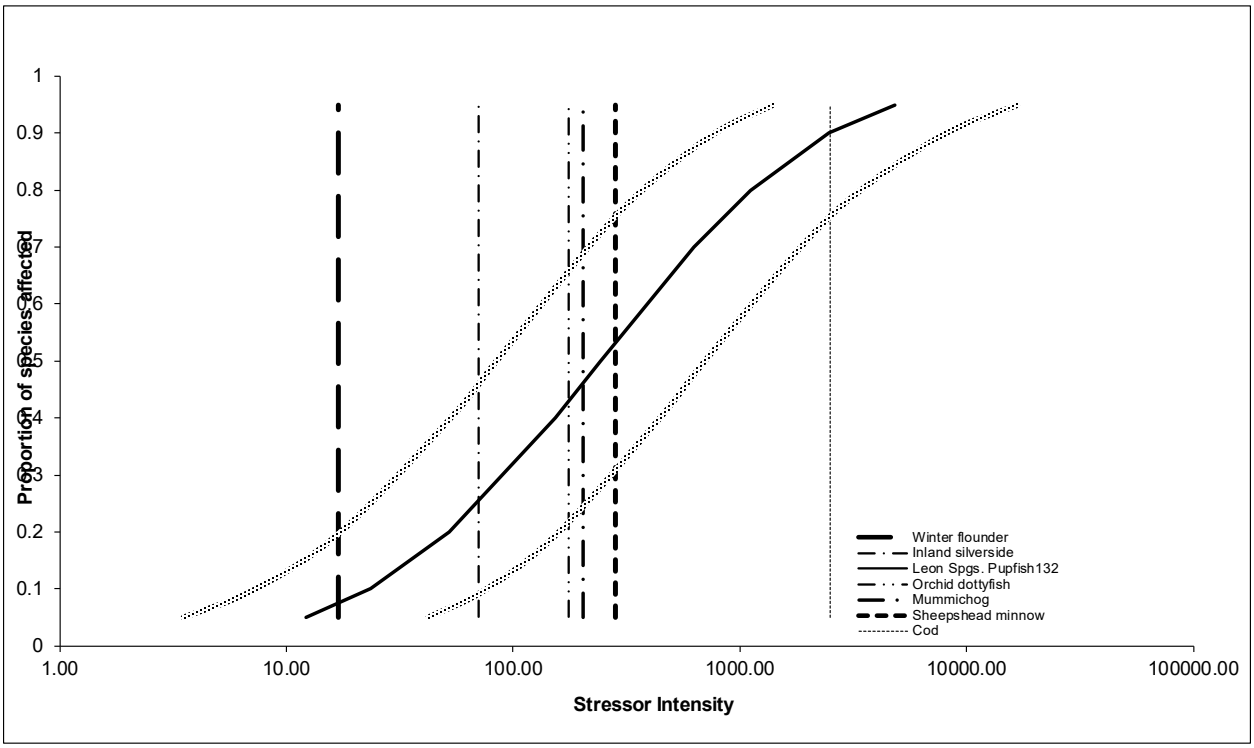


Figure 34: Species sensitivity distribution for the acute toxicity of nonylphenol to marine fish

The calculated acute effects assessment LC_{low} of 5.37 µg/L is lower than the nonylphenol marine acute criterion of 7 µg/L. Our conclusion is that exposure at the level of the marine acute nonylphenol criterion is likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio and yelloweye rockfish.

5.6.1.13.5 Acute-chronic ratio for nonylphenol

The final ACR for nonylphenol given in USEPA (2005a) is 8.412. This ACR was derived from three species mean ACR values of 2.023, 8.412 and 28.11 for *Daphnia magna*, *Americamysis bahia* and rainbow trout, respectively.

After the publication of the USEPA (2005a) nonylphenol criteria document, Isidori et al. (2006) published results of a series of two day acute and seven day chronic toxicity studies of nonylphenol with *Ceriodaphnia dubia*, which permit the calculation of a fourth ACR value for nonylphenol. When combined with the chronic *Ceriodaphnia* toxicity data of D. E. England (1995), a *Ceriodaphnia dubia* ACR of 9.083 was calculated. This *C. dubia* ACR is comparable to both the ACR of 8.412 recommended in USEPA (2005a) and a recalculated four species geometric mean ACR of 8.119, which includes the new *C. dubia* ACR of 9.083. The recalculated four species geometric mean ACR of 8.119 will be used in the assessment of the marine chronic nonylphenol criterion.

5.6.1.13.6 Water-column exposure toxicity data - chronic criterion for nonylphenol in marine waters

No chronic toxicity data for any marine fish species is reported in the USEPA (2005a) nonylphenol criteria document. The only marine chronic data in USEPA (2005a) are two studies with the mysid *Americamysis bahia*, resulting in a genus mean chronic value of 7.84 µg/L.

5.6.1.13.7 Additional marine chronic toxicity data for nonylphenol published since 2005

A substantial number of chronic duration studies with marine fish have been performed with nonylphenol since the publication of the USEPA (2005a) aquatic life criteria for nonylphenol. Information on these studies was identified through a search of the EPA ECOTOX database, as well as information in literature reviewed by the National Marine Fisheries Service (NMFS, 2018) and Gao, Guo, Li, and Gibson (2015). The original studies identified in this review were then individually reviewed by EPA Region 10 staff for suitability for use in deriving EPA aquatic life criteria. Studies from 2004 are also discussed in this section, since they may not have been known to EPA at the time the nonylphenol criteria document was developed prior to its publication in 2005.

Toxicity tests show that 4-nonylphenol disrupts endocrine systems by mimicking the female hormone 17β-estradiol. Exposure of fish resulted in abnormal gonad development, changes in reproductive behavior, altered sex ratio of offspring, and the production of yolk proteins (vitellogenin) by immature male fish.

Most toxicity tests with marine fish performed since 2005 have been either relatively short exposure duration biochemical and physiological studies of endocrine system disruption or longer term exposure duration studies on nonylphenol effects on the ability of anadromous fish to successfully transition from living in freshwater to living in estuarine or marine systems. The biochemical studies are generally of too short a duration (10 days or less) to qualify as chronic duration according to the Guidelines.

The studies on the ability of anadromous fish to successfully transition from fresh to marine waters during nonylphenol exposure have in almost all cases exposed the fish to nonylphenol during their freshwater residency, then measured adverse effects after the fish were transitioned into salt water and nonylphenol exposure was terminated. While of chronic duration and with an ecologically realistic experimental design, strictly speaking the nonylphenol exposures are not marine toxicity studies because the nonylphenol exposure were limited to freshwater, not marine waters. This study design

results in the observed toxic effects of nonylphenol being delayed effects in marine systems of a freshwater exposure to nonylphenol.

The Guidelines (Stephan et al., 1985) provide no recommendations on how to treat this study design in the derivation of aquatic life criteria but do permit studies not compliant with data quality requirements for criteria derivation to be used as additional information to support criteria. Even though these studies do not fully meet data quality requirements for criteria development, the ecological realism of these studies makes them useful in a supporting role in the evaluation of the protectiveness of the marine chronic nonylphenol criterion for ESA listed species. Another common limitation of these studies is that they often use only one exposure concentration of nonylphenol plus a control, instead of the required minimum of three exposure concentrations plus a control. The preponderance of this type of study in the literature published since 2005 has resulted in few of the newer studies being fully compliant with the data quality requirements of the Guidelines for studies useful in deriving aquatic life criteria and evaluation of the criteria in biological evaluations under ESA. The two primary shortcomings are test duration (often 21 days instead of the 30 day requirement), and only one exposure concentration plus control evaluated instead of the required minimum of three different exposure concentrations plus control. The studies reporting nonylphenol adverse effects in salt water after fish were exposed to nonylphenol in freshwater are described after the discussion of the chronic studies on fish performed solely in marine waters.

Saravanan, Nam, Eom, Lee, and Rhee (2019) exposed two species of marine fish, red seabream (*Pagrus major*) and black rockfish (*Sebastes schlegelli*) to three nominal concentrations of nonylphenol (1, 10 and 50 µg/L, plus control) for 60 days, and monitored effects on reproductive hormones after 30 and 60 days exposure. Although neither species is a North American resident (the commonly found black rockfish in Washington state is a different species, *Sebastes melanops*), the results for black rockfish are particularly relevant to this BE, as black rockfish are in the same genus as the two ESA listed rockfish species present in the action area: bocaccio (*Sebastes paucispinis*) and yelloweye rockfish (*Sebastes ruberrimus*). To our knowledge, Saravanan et al. (2019) is the first chronic toxicity study with any member of the rockfish genus *Sebastes* for any chemical. In their 14 day preliminary rangefinding study, 500 µg/L nonylphenol was associated with 25% mortality in red seabream and 30% mortality in black rockfish.

The lowest tested concentration of 1 µg/L had no effect on gonadosomatic index (GSI), hepatosomatic index (HSI), on concentrations of plasma 17β estradiol (E2), plasma 11- ketotestosterone (KT-11), plasma vitellogenin (VTG), plasma cortisol, or on VTG mRNA expression in red seabream after 30 days exposure. These were all of the endocrine system related endpoints measured by Saravanan et al. (2019). Exposure to 10 µg/L nonylphenol for 30 days resulted in increased plasma E2, plasma VTG, plasma cortisol and VTG mRNA expression of red seabream.

Black rockfish plasma E2 and VTG mRNA expression were significantly elevated after 30 days at the lowest exposure concentration of 1 µg/L. Additional effects observed on black rockfish during the 10 µg/L exposure included elevated plasma VTG and elevated plasma cortisol. Finally, the 30 day exposure to 50 µg/L resulted in a reduced gonadosomatic index relative to controls.

While Saravanan et al. (2019) did not report any information on nonylphenol effects or either red seabream or black rockfish reproduction or growth, several of the observed effects have been associated with reduced growth and reproductive output in fish. Decreased gonadosomatic indices are

indicative of a reduction in gonad mass and output and can affect sexual maturation in fish. Vitellogenin is an egg yolk precursor protein, which is mainly produced in the liver by E2 secreted from the ovary of female fish. Presence of VTG in juvenile and male fish is indicative of the estrogenic effects of nonylphenol on plasma E2, plasma VTG levels and the induction of VTG mRNA expression.

Martin-Skilton et al. (2006) exposed juvenile Atlantic cod (*Gadus morhua*) and turbot (*Scophthalmus maximus*) to 29 µg/L nonylphenol for 21 days. No effect on the growth (length, weight and condition factor) of cod was observed at the end of the study relative to controls. And while turbot length was unaffected by nonylphenol, turbot weight and condition factor were reduced by the nonylphenol exposure. The primary shortcoming of this study precluding its inclusion in a database used to derive aquatic life criteria is that only one exposure concentration of nonylphenol was used, along with the exposure duration being shorter than desired for use in evaluating chronic toxicity to fish. Cod are native to the United States while turbot are not native.

A.M. Hanson, J.D. Kittilson, S.D. McCormick, and M.A. Sheridan (2012) exposed rainbow trout to two concentrations of 4-nonylphenol (10 and 100 µg/L) in freshwater for 28 days. Hanson et al. (2012) found a statistically significant reduction in body weight at the end of the study for juvenile trout exposed to 100 µg/L, but not at 10 µg/L nonylphenol. However, the main objective of Hanson et al. (2012) was to determine if nonylphenol exposure in freshwater inhibits the ability of outmigrating juvenile rainbow trout (steelhead) to adapt to seawater.

The process by which younger salmon hatched in freshwater adapt to life in marine systems is called smoltification. In preparation for outmigration from streams to marine systems, juvenile salmon residing in freshwater transform into smolts prior to and during the transition from life in freshwater to life in marine systems. The transformation of smolts into individuals that can survive in salt water includes changes in the biochemical, morphological, physiological and behavioral phenotypes that, once transformation is complete, can survive and grow in marine systems.

The experimental design of Hanson et al. (2012) exposed rainbow trout to nonylphenol in freshwater for 28 days, then transferred the exposed fish to seawater with a salinity of 20‰. Results of this study indicated that activation of GH-IGF (growth hormone – insulin-like growth factor) system components, including increased expression of GHR 1, GHR 2 (growth hormone receptors 1 and 2), IGF-1, IGF-2, IGFR1A, and IGFR1B (insulin-like growth factor mRNA 1A and 1B), accompany salinity adaptation of rainbow trout. The results also indicate that exposure of rainbow trout to 10 µg/L nonylphenol suppresses the expression of multiple GH-IGF system components and leads to reduced growth, reduced hypoosmoregulatory ability, and a reduced ability of salmonid smolts to successfully transition from fresh water to seawater during smoltification.

Hanson et al. (2012) did observe growth reductions in rainbow trout in salt water at their lowest freshwater exposure concentration of 10 µg/L, but the exposure to nonylphenol occurred in freshwater, not marine waters. In addition, only two exposure concentrations plus control were used, not the minimum required three exposure concentrations plus control. The combination of lack of nonylphenol exposure in salt water and the lack of a third exposure concentration renders Hanson et al. (2012) unsuitable for use in evaluating nonylphenol toxicity in marine waters for any purpose other than supporting information for other, more suitable studies.

A. M. Hanson et al. (2014) monitored the feeding behavior, growth, biochemical growth factors and gene expression of rainbow trout exposed to 4-n-nonylphenol for 28 days. The only exposure concentrations tested were a control and 100 µg/L nonylphenol. Exposure had no effect on rainbow trout feeding but did reduce growth over the 28 day exposure period. Hanson et al. (2014) is not suitable for water quality criteria derivation, because only one contaminant exposure concentration was tested, not the minimum of three contaminant exposures (plus a control) needed to provide a measure of the concentration-response relationship needed to derive a water quality criterion.

Several other studies published since release of the USEPA (2005a) nonylphenol criteria document also evaluated nonylphenol effects on salmonid smoltification, primarily Atlantic salmon, using experimental designs similar to that of Hanson et al. (2012). Like Hanson et al. (2012) these studies exposed salmonid smolts to nonylphenol in freshwater for an extended period, then switched the smolts gradually into salt water, where the effects of nonylphenol were recorded. Although the adverse nonylphenol effects were observed in salt water, the nonylphenol exposure was in freshwater. Based on the freshwater exposure to nonylphenol, we evaluated these studies in our assessment of the freshwater chronic nonylphenol criterion. In this evaluation of the chronic marine nonylphenol criterion, they are used as supporting information for our evaluation, not as primary data sources to evaluate the chronic criteria.

This group of "Hanson-like" studies have also confirmed that the estrogenic effect of nonylphenol on smoltification is associated with alterations to one or more components of the GH-IGF system or products of endocrine receptor (ER) associated gene expression, such as ERα mRNA or Vtg (vitellogenin, a protein that is a precursor to egg yolk formation). In addition to the Hanson et al. (2012) study with rainbow trout, Luo et al. (2005) observed that these same biochemical pathways may adversely affect smoltification in sockeye salmon (*Oncorhynchus nerka*), another salmonid.

Although many of these studies provide information on water column concentrations which can adversely affect smoltification and subsequent fish health once in salt water, their experimental designs preclude their use in deriving EPA marine aquatic life criteria. This is because they do not meet one or more of the requirements in the Guidelines EPA uses to identify studies with acceptable data quality for criteria development. Limitations include chronic exposure to nonylphenol durations shorter than the required 30 day duration (Arsenault et al., 2004; Breves et al., 2018; Duffy et al., 2014; Lerner et al., 2007; Luo et al., 2005; McCormick et al., 2005), fewer than three exposure concentrations (Breves et al. 2018; Hanson et al. 2014; Hanson et al. 2012; Lerner et al. 2007; Luo et al. 2005; Arsenault et al. 2004, Madsen et al. 2004), and contaminant exposure routes that do not represent exposure to nonylphenol dissolved in the water column, such as dietary (Keen et al., 2005) or intraperitoneal injection (IP injection, Luo et al. 2005; McCormick et al. 2005; Madsen et al. 2004) exposures. Because of the importance of the smoltification process in the life history of many salmonid species, findings of these studies are summarized in the next several paragraphs, despite being unusable in water quality criteria development according to the Guidelines.

Breves et al. (2018) exposed Atlantic salmon fry in freshwater to two nonylphenol concentrations (8.8 and 44 µg/L) for 21 days. Weight of the fry was significantly reduced at 8.8 µg/L relative to control weights. Fry length was unaffected at 8.8 µg/L, but was significantly reduced relative to the length of controls at 44 µg/L. Fry survival was unaffected by either exposure concentration.

Breves et al. (2018) also exposed smolts in freshwater to three concentrations (0.88, 8.8 and 88 µg/L) of nonylphenol for four days, and monitored survival, growth, ionoregulation and gene expression of

multiple genes within the insulin-like growth factor (IGF) system. No effects were observed on smolt survival or growth in this portion of the study. Gene expression of multiple insulin-like growth factors was inhibited at 8.8 µg/L. Despite inhibition of gene expression in the IGF system known to be a precursor to smolt adaptation to sea water, Breves et al. (2018) observed no effect on the osmoregulatory processes required for smolts to adapt to sea water.

Duffy et al. (2014) exposed Atlantic salmon smolts to three nonylphenol concentrations (0.88, 8.8 and 88 µg/L) for four days, then exposed the smolts to seawater. They observed no effect on survival at any nonylphenol concentration, and no effect on growth expressed as change in hepatosomatic index (liver weight / whole body weight) x 100.

Lerner et al. (2007) exposed yolk sac fry of Atlantic salmon to mean measured concentrations of 6.5 and 79.9 µg/L nonylphenol for 21 days, then monitored the response of the fry one year later after they had matured into smolts in nonylphenol-free water. Fry exposed to the 79.9 µg/L concentration exhibited nearly 50 % mortality during the 21-day nonylphenol exposure, with delayed mortality post-exposure resulting in 100% mortality within 120 days after termination of the 79.9 µg/L nonylphenol exposure. Fry exposed to 6.5 µg/L nonylphenol suffered no mortality during the initial 21-day exposure, but showed delayed mortality of roughly 30% in the 60-day period after their transfer from 6.5 µg/L nonylphenol into nonylphenol-free water. During their transition into nonylphenol-free salt water one year later, the surviving smolts exhibited reduced development, a reduced preference for salt water, and took longer to enter into salt water than did the control smolts.

Luo et al. (2005) treated two groups of smolting sockeye salmon with either 15 mg/kg or 150 mg/kg nonylphenol via IP injection twice within a 4-day period. While the sockeye salmon smolts experienced normal physiological changes during smoltification, changes in gene expression of ERα mRNA were observed in fish exposed to both 15 and 150 mg/kg nonylphenol.

McCormick et al. (2005) intraperitoneally injected juvenile Atlantic salmon living in freshwater with four separate injections over an 11 day period with one of five doses (0.5, 2, 10, 40 or 150 µg/g) of branched 4-nonylphenols. Fourteen days after the first injection, the juvenile salmon were transferred to seawater for 24 hours to assess their salinity tolerance. The 150 µg/g exposure was the only one of the five doses resulting in a significant reduction in salinity tolerance, which was identified by a significant increase in plasma sodium concentration. No mortality was observed in any of the nonylphenol exposures. The authors concluded that if this detrimental effect of elevated nonylphenol exposure was continued, it would likely compromise the normal parr-smolt transformation of Atlantic salmon.

Arsenault et al. (2004) exposed Atlantic salmon parr-smolts to a single 20 µg/L nonylphenol concentration in freshwater for 12 – 14 days, then allowed the fish to gradually adapt to seawater. After a 51 - 52 day residence in nonylphenol-free seawater, smolt weight and specific growth rates were measured. The nonylphenol-exposed smolt had statistically significantly reduced growth and IGF-1 activity relative to controls. This impact of nonylphenol is believed by Arsenault et al. (2004) to have adverse long-term effects on growth and the optimal completion of the smoltification process.

Madsen et al. (2004) performed an integrated laboratory and field study of nonylphenol effects on Atlantic salmon smoltification, downstream migration and survival of smolts. A group of 150 smolts in freshwater were injected six times over a 20 day period with 120 µg/g 4-nonylphenol. Two 24-hour salt water challenge tests were performed prior to the start of nonylphenol injections, and a third salt water

challenge was given at the end of the 20-day nonylphenol injection period. All smolts were then individually tagged with passive integrated transponder (PIT) tags at the end of the nonylphenol injection period. A total of 100 smolts nonylphenol injected and 100 control smolts were then transferred to the head of a 3.2 km long freshwater stream two days after the last nonylphenol injection. Smolts were released into the stream and then their ability to migrate downstream was monitored. Control fish immediately began their downstream migration, while the start of downstream migration of the nonylphenol-injected smolts was delayed by 6 – 8 days compared to that of the controls. After 34 days in the stream, 81% of the control smolts were captured alive in a trap at the downstream end of the 3.2 km stream, while only 12% of the nonylphenol-injected smolts were captured alive in the downstream trap. Madsen et al. (2004) speculated that the difference in survival between the nonylphenol exposed and control smolts during downstream migration was due to a combination of impaired smolt development, behavioral changes and increased predation by herons and mink on the nonylphenol-exposed smolts.

Madsen et al. (2004) maintained the remaining 50 control and nonylphenol exposed smolts in laboratory freshwater for the same length of time the other smolts were present in the 3.2 km stream. Subsets of these 50 smolts retained in laboratory were salt water challenged the same day smolts were released outdoors into the stream: only one nonylphenol-exposed smolt survived this 24-hour salt water challenge. All remaining control and nonylphenol-exposed smolts kept in laboratory freshwater during the time smolts were present in the 3.2 km stream survived to the end of the study.

Finally, one study of nonylphenol effects on salmonid smoltification evaluated dietary exposures and reported water column nonylphenol concentrations, as opposed the studies above that exposed salmonids via either water column exposure or IP injection for exposures shorter than 28 days. Keen et al. (2005) fed juvenile coho salmon (*Oncorhynchus kisutch*) with one of four dietary concentrations (0.002, 0.2, 20 or 2000 mg/kg of 4-nonylphenol) plus a control for 28 days in freshwater. Fish growth in salt water was then observed for two successive 6-week intervals. Dietary exposure to nonylphenol had no effect on survival, growth or smoltification of coho salmon, even when fed the highest dietary concentration of 2000 mg/kg nonylphenol. Keen et al. (2005) attributed this lack of toxicity on the ability of coho salmon to eliminate the ingested nonylphenol via the biliary-fecal pathway after their transfer into salt water. This observation of no toxicity from dietary ingestion of nonylphenol conflicts with the studies of water column nonylphenol exposure on smoltification, where adverse effects were observed. A minimal amount of nonylphenol leached from the contaminated food into the water column during the 28-day freshwater feeding portion of the study, with a maximum detected concentration of 2.183 µg/L found in a water drain sample from a 2000 mg/kg dietary exposure tank.

5.6.1.13.8 Effects assessment of marine chronic nonylphenol criterion on listed species

The paucity of chronic toxicity studies with fish meeting data quality requirements for use in developing chronic aquatic life criteria, and the complete absence of acceptable chronic nonylphenol toxicity data explicitly describing growth or reproductive effects for any of the ESA listed fish species within the action area limits this effects assessment to dividing the nonylphenol acute effects assessment concentration from this BE (5.37 µg/L) by the nonylphenol acute-chronic ratio derived in this BE (8.119) to derive the chronic effects assessment concentration. The nonylphenol chronic effects assessment concentration so calculated (0.6614 µg/L) is lower than the marine chronic nonylphenol criterion of 1.0 µg/L. Our conclusion is that exposure at the level of the marine chronic nonylphenol criterion is likely

to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio and yelloweye rockfish.

5.6.1.13.9 Summary of effect determinations for marine nonylphenol criteria

EPA has made the following determinations on the acute and chronic nonylphenol criteria for the listed species in the marine portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Likely to adversely affect** (exposure to water)

Chinook salmon – **Likely to adversely affect** (exposure to water)

Chum salmon – **Likely to adversely affect** (exposure to water)

Bull trout – **Likely to adversely affect** (exposure to water)

Bocaccio – **Likely to adversely affect** (exposure to water)

Yelloweye rockfish – **Likely to adversely affect** (exposure to water)

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Likely to adversely affect** (exposure to water)

Chinook salmon – **Likely to adversely affect** (exposure to water)

Chum salmon – **Likely to adversely affect** (exposure to water)

Bull trout – **Likely to adversely affect** (exposure to water)

Bocaccio – **Likely to adversely affect** (exposure to water)

Yelloweye rockfish – **Likely to adversely affect** (exposure to water)

Marbled murrelet – **Not likely to adversely affect**

5.6.1.14 Pentachlorophenol – Marine

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) Criterion – 13 µg/L

Chronic (CCC) Criterion – 7.9 µg/L

CAS ID – 87-86-5 (pentachlorophenol)

CAS ID – 131-52-2 (sodium pentachlorophenate)

5.6.1.14.1 Water-column exposure toxicity data - acute criterion for pentachlorophenol in marine water

The most recent published marine criteria for PCP are in EPA (USEPA, 1995, 1996b). There are no published acute toxicity data for any salmonid or rockfish species in marine waters reported in EPA (USEPA, 1995, 1996b). EPA (USEPA, 1995, 1996b) is confusingly titled "1995 Updates", so when referring to the 1995 updates of aquatic life criteria in this section we cite both EPA 1995 and EPA 1996. EPA 1996 provides more detailed information on studies used to modify or update the earlier pentachlorophenol criteria document (USEPA, 1986b) than does the 1995 criteria update. Both pentachlorophenol and sodium pentachlorophenate have been used in aquatic toxicity testing, thus the two CAS ID numbers above.

5.6.1.14.2 Additional acute toxicity data for pentachlorophenol published since 1995

For fish, several 4-day acute toxicity studies of PCP in marine water have been published since the most recent EPA (USEPA, 1995, 1996b) PCP criteria document. Unfortunately, none of them have been performed on any of the marine listed salmonid or rockfish species within the action area. Additional acute toxicity data for PCP to marine fish were published in Smith et al. 1994 (Dover sole, *Solea solea*, 4-day LC50 = 450 µg/L), Smith et al. 1994 (European flounder, *Platichthys flesus*, geometric mean 4-day LC50 = 729 µg/L), two studies with the Leon Springs pupfish (*Cyprinodon bovinus*), Sappington et al. 2001 (4-day LC50 = 80 µg/L) and Mayer et al. 2008 (4-day LC50 = 107 µg/L), and two additional 4-day LC50s for sheepshead minnow (*Cyprinodon variegatus*) published in Sappington et al. 2001 (4-day LC50 = 50 µg/L) and Mayer et al. 2008 (4-day LC50 = 191 µg/L). Smith et al. 1994 and Sappington et al. 2001 were both cited in the Oregon Toxics BE (USEPA, 2008)(B. Shephard, Zodrow, Keenan, & Palmer, 2008) but were used as a source of information for chronic toxicity data, not acute toxicity data. Mayer et al. 2008 was not cited in the Oregon Toxics BE.

The acute toxicity data for the Leon Springs pupfish, Dover sole and European flounder are all new marine acute 4-day LC50 values not listed in the most recent EPA (USEPA, 1995, 1996b) criteria document update for PCP. The new sheepshead minnow 4-day LC50 values result in a change in the SMAV for sheepshead minnow from 442 µg/L to 225 µg/L. There are currently eight SMAVs for fish exposed to pentachlorophenol, ranging between 25.3 µg/L for Pacific herring and 729 µg/L for European flounder.

5.6.1.14.3 Effects assessment of acute pentachlorophenol criterion on listed species

As there are no empirical acute toxicity studies on any salmonid or rockfish species found in the EPA ECOTOX database, determinations regarding the acute PCP criterion were based on adverse acute effects to all marine fish species with empirical PCP acute toxicity data. After combining the available marine fish toxicity data from the 1995 and 1996 PCP criteria documents and ECOTOX literature compiled between 1995 and present day, the lowest 4-day LC_{50} remains the 25.3 µg/L concentration observed by Vigers et al. (Vigers, Marleave, Janssen, & Borgmann, 1978) as cited in the EPA (USEPA, 1986b) pentachlorophenol criteria document.

Dividing the Pacific herring 25.3 µg/L SMAV by 2.27 to convert this LC_{50} to the lowest LC_{LOW} or minimum acute effect concentration for any marine fish species yields a threshold acute effect concentration of 11.1 µg/L. This concentration is lower than the pentachlorophenol marine acute criterion of 13 µg/L. Assuming that this threshold acute effect concentration is the same as that for all ESA listed fish species in the marine portions of the action area, exposure at the level of the marine pentachlorophenol acute

criterion of 13 µg/L is likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio and yelloweye rockfish.

5.6.1.14.4 Water-column exposure toxicity data - chronic criterion for pentachlorophenol in marine water

The chronic marine PCP criterion was evaluated in the Oregon Toxics BE (USEPA, 2008)(B. Shephard et al., 2008). Based on a SSD of the existing chronic toxicity data for marine fish evaluated in EPA (2008)(B. Shephard et al., 2008), EPA determined at that time that the chronic PCP marine criterion was not likely to adversely affect any listed salmonid species in marine waters. Rockfish species were not evaluated in the Oregon Toxics BE. The only empirical chronic NOEC for any fish species exposed to PCP available for use in both the Oregon Toxics BE and this BE is a 47 µg/L embryo hatchability NOEC for sheepshead minnow (Parrish et al., 1978).

5.6.1.14.5 Additional chronic toxicity data for pentachlorophenol published since 2008

As discussed earlier, the PCP acute toxicity studies with fish published since the EPA (USEPA, 1995, 1996b) pentachlorophenol criterion document (Sappington et al., 2001; S. Smith, Furay, Layiwola, & Menezes - Filho, 1994) were used to estimate chronic NOEC concentrations in the Oregon Toxics BE (USEPA, 2008)(B. Shephard et al., 2008). The Leon Springs pupfish and sheepshead minnow acute toxicity data contained in Mayer et al. 2008 were used in this BE to estimate chronic NOEC values for these species. No more recent empirical chronic toxicity data for pentachlorophenol effects on any marine fish species was identified in an ECOTOX search.

5.6.1.14.6 Acute-chronic ratio for pentachlorophenol

The ACR for pentachlorophenol was calculated to be 3.166 in EPA (USEPA, 1986b). This ACR of 3.166 was calculated as the geometric mean of five empirically measured species mean ACRs ranging between 0.8945 and 6.873. This ACR was used to convert empirical 4-day LC50 data to 30-day chronic NOEC data for use in evaluating the chronic toxicity of PCP. This conversion is done by dividing the LC50 by the ACR.

5.6.1.14.7 Effects assessment of chronic pentachlorophenol criterion on listed species

The lowest chronic NOEC for any marine fish species exposed to PCP was derived by dividing the Pacific herring 25.3 µg/L SMAV by the PCP ACR of 3.166. This calculation converts this LC50 to the lowest chronic NOEC or minimum chronic effect concentration for any marine fish species. The calculation yields a threshold chronic effect concentration of 8.0 µg/L. This value is higher than the pentachlorophenol marine chronic criterion of 7.9 µg/L. Assuming that this 8.0 µg/L threshold chronic effect concentration is less than or equal to that for all ESA listed fish species in the marine portions of the action area, exposure at the level of the marine pentachlorophenol chronic criterion of 7.9 µg/L is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio and yelloweye rockfish.

5.6.1.14.8 Summary of effect determinations for marine pentachlorophenol criteria

EPA has made the following determinations on the acute and chronic pentachlorophenol criteria for the listed species in the marine portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Likely to adversely affect** (exposure to water)

Chinook salmon – **Likely to adversely affect** (exposure to water)

Chum salmon – **Likely to adversely affect** (exposure to water)

Bull trout – **Likely to adversely affect** (exposure to water)

Bocaccio – **Likely to adversely affect** (exposure to water)

Yelloweye rockfish – **Likely to adversely affect** (exposure to water)

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.6.1.15 Phosphorus (elemental) – Marine

The Tribe submitted the following criteria to EPA for CWA approval:

Chronic (CCC) Criterion – 0.1 µg/L

CAS ID – 7723-14-0

5.6.1.15.1 Water-column exposure toxicity data - chronic criterion for elemental phosphorus in marine water

EPA (USEPA, 1976d, 1976e) Quality Criteria for Water (The Red Book) contains the originally recommended elemental phosphorus chronic marine criterion of 0.1 µg/L. The elemental phosphorus marine chronic criterion is also shown in what is commonly known as the 'Gold Book' (USEPA, 1986d, 1986e, 1986f), whose text is essentially that originally published in EPA (USEPA, 1976d, 1976e, 1976f), and in all subsequent EPA aquatic life criteria tables until 2014, where no value for the chronic marine elemental phosphorus criterion was given. The most recent EPA National Recommended Aquatic Life Criteria Table (USEPA (2022), found online at https://www.epa.gov/wqc/national-recommended-water-

quality-criteria-aquatic-life-criteria-table) lists no value for the marine chronic elemental phosphorus criterion, but includes a link to EPA's Gold Book (USEPA, 1986e, 1986f).

The criterion of 0.10 ug/L elemental phosphorus for marine or estuarine waters was derived in EPA (USEPA, 1976d, 1976e) as 0.1 of demonstrated lethal levels to important aquatic organisms and of levels that have been found to result in significant bioaccumulation. None of the listed species within the marine portions of the action area have any available acute or chronic toxicity data described in the ECOTOX database. There also appears to be no empirical chronic toxicity data with elemental phosphorus with any fish species.

One study with a salmonid in marine water was available and used by EPA in the development of the EPA (USEPA, 1976d, 1976e) marine chronic elemental phosphorus criterion. Fletcher and Hoyle (Fletcher & Hoyle, 1972) exposed Atlantic salmon (*Salmo salar*) to various concentrations of elemental phosphorus and determined the time required for 50% mortality (LT_{50}) to occur in each exposure concentration. The lowest concentration tested of 0.79 µg/L resulted in an LT_{50} of 8.125 days. The LT_{50} closest to a 4-day LC_{50} came from an exposure of Atlantic salmon to 11 µg/L elemental phosphorus for an LT_{50} duration of 3.5458 days (approximately equal to 3 days 13 hours). Fletcher and Hoyle (Fletcher & Hoyle, 1972) also studied elemental phosphorus toxicity to Atlantic cod (*Gadus morhua*). Their lowest concentration tested of 1.89 µg/L resulted in an Atlantic cod LT_{50} of 5.2 days (125 hours). The remaining limited amount of elemental phosphorus toxicity data to fish is for freshwater fish, not marine fish.

5.6.1.15.2 Additional chronic toxicity data elemental phosphorus published since 1975

The only toxicity study on any marine species published since 1975 available in the ECOTOX database is that of Maddock and Taylor (Maddock & Taylor, 1976), who measured a 48-hour LC_{50} of 14.4 µg/L to Atlantic cod (*Gadus morhua*). Bentley et al. (Bentley et al., 1978) published the results of a series of elemental phosphorus toxicity studies performed on freshwater algae, aquatic invertebrates and freshwater fish species, including rainbow trout (*Oncorhynchus mykiss*).

The Bentley et al. (1978) work was performed under contract to the U.S. Army Medical Research and Development Command. It was not available when the EPA (USEPA, 1976d, 1976e) Red Book was published, and apparently was only available in draft form when the American Fisheries Society (Society, 1979) review of the water quality criteria in the Red Book was published. Although not a chronic toxicity study in marine water, the rainbow trout freshwater 96-hour LC_{50} of 22 µg/L (95% confidence limits of 16 – 48 µg/L) calculated by Bentley et al. (1978) is of interest for the purposes of this BE. It is not, however, applicable to the effects assessment of a marine water quality criterion.

5.6.1.15.3 Acute-chronic ratio for elemental phosphorus

There are no published ACRs for elemental phosphorus. Bentley et al. (1978) published three application factors (the inverse of the ACR) in their work with the freshwater cladoceran *Daphnia magna*, larvae of the midge *Chironomus tentans* and the fathead minnow (*Pimephales promelas*). The ACRs calculated from the inverse values of the published application factors ranged between 3.4 for *D. magna*, > 50 for fathead minnow up to 2000 for *C. tentans*. The range between the available ACRs of 588x is too wide to permit estimation of a single ACR applicable to multiple aquatic species according to the Guidelines (C.E. Stephan et al., 1985). Therefore, there is no ACR available that can convert a 96-hour LC_{50} value into a chronic NOEC value.

5.6.1.15.4 Effects assessment of chronic elemental phosphorus criterion on listed species

No additional marine toxicity data for elemental phosphorus effects on any marine species has been published since 1976. No more recent information is available which may lead EPA to conclude that the marine chronic criterion of 0.1 µg/L is not protective of marine species, including ESA listed marine species. Therefore, exposure at the level of the marine elemental phosphorus chronic criterion of 0.1 µg/L is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio or yelloweye rockfish.

5.6.1.15.5 Summary of effect determinations for marine elemental phosphorus criteria

EPA has made the following determinations on the chronic elemental phosphorus criterion for the listed species in the marine portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above, and the indirect effects analysis for prey availability provided in section 5.8:

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.6.1.16 Zinc – Marine

The Tribe submitted the following criteria to EPA for CWA approval:

Acute (CMC) criterion = 90 µg/L

Chronic (CCC) criterion = 81 µg/L

CAS ID 7440-66-6

5.6.1.16.1 Water-column exposure toxicity data - acute criterion for zinc in marine water

The USEPA (1987) Ambient Water Quality Criteria for Zinc document contains 28 species mean acute values used to derive the marine acute criterion for zinc. None of the available SMAVs are for any ESA listed species within the marine portions of the action area. No additional marine acute toxicity data was compiled in the most recent update of the zinc criteria (USEPA, 1996a).

EPA (USEPA, 1980), an earlier version of USEPA (1987) does report marine 48 hour LC_{50}s for rainbow trout (Herbert and Wakeford, 1964) at four different salinities: 3, 7, 14 and 26‰. The respective 48 hour LC_{50}s were 15,000, 25,000, 85,000 and 35,000 µg/L. Although not useable in acute criterion

development because the 48 hour exposure duration was too short to meet the requirement of using 96 hour exposures for acute criteria derivation, Herbert and Wakeford (1964) does provide some sense of *Oncorhynchus* species sensitivity to zinc under short term exposures.

Of the 28 marine species with zinc SMAVs in USEPA (1987), seven were fish. The species mean acute values for the seven marine fish were 38,000 µg/L for spot (*Leiostomus xanthurus*), 36,630 µg/L for mummichog (*Fundulus heteroclitus*), 9467 µg/L for winter flounder (*Pseudopleuronectes americanus*), 5600 µg/L for tidewater silverside (*Menidia peninsulae*), 3640 µg/L for Atlantic silverside (*Menidia menidia*) and 430 µg/L for striped bass (*Morone saxatilis*).

The seventh fish SMAV in USEPA (1987) was from Dinnel et al. (1983) with cabezon (*Scorpaenichthys marmoratus*). Dinnel, Link, Stober, Letourneau, and Roberts (1989), a peer reviewed paper based on the Dinnel et al. (1983) report used in USEPA (1987) performed a 96 hour LC_{50} study on post-hatch larvae of cabezon. The cabezon 96 hour LC_{50} of 191 µg/L is the lowest zinc acute toxicity value for any of the 28 SMAVs for marine species in EPA (1987).

5.6.1.16.2 Additional marine acute toxicity data for zinc published since 1987

Four additional marine acute fish toxicity studies with zinc have been identified. Moreau, Klerks, and Haas (1999) measured a 96 hour LC_{50} of 11,300 µg/L for sheepshead minnow (*Cyprinodon variegatus*). Shenker and Cherr (1990) determined a 96 hour LC_{50} of 14,500 µg/L for English sole (*Parophrys vetulus*). A 96 hour LC_{50} of 7730 µg/L for striped mullet (*Mugil cephalus*) was found by (Rajkumar, 2012). Dung, Cuong, Huyen, and Duc Cu (2005) reported a 96 hour LC_{50} of 685 µg/L for larval cobia (*Rachycentron canadum*).

5.6.1.16.3 Effects assessment of acute marine zinc criterion on listed species

No empirical acute toxicity data exists for any of the ESA listed marine fish species in the marine portions of the action area. A combined total of 11 species worth of acceptable quality 96 hour LC_{50} studies with zinc for marine fish species: seven studies from USEPA (1987) and four studies identified in the literature published since 1987. This amount of data permits calculation of an acute assessment effects concentration using a species sensitivity distribution.

Results of the species sensitivity distribution calculation for the acute zinc toxicity data for marine fish is presented in Table 111. The 5th percentile 96 hour LC_{50} of the SSD is 216 µg/L. When divided by 2.27 to convert the 5th percentile 96 hour LC_{50} to the 5th percentile LC_{low} used as the acute zinc assessment effects concentration in this BE, the acute zinc assessment effects concentration is 95 µg/L. The acute zinc assessment effects concentration is slightly higher than the 90 µg/L marine acute zinc criterion. However the 95% lower confidence limit of the 216 µg/L central tendency estimate of the 5th percentile of the SSD is 67 µg/L, which is lower than the marine acute zinc criterion of 90 µg/L. Based on the central tendency value of the 5th percentile of the species sensitivity distribution for all 11 marine fish 96 hour LC_{50} values, adverse effects to listed salmonids as a result of exposure at the criterion level are unlikely.

Table 107. Species sensitivity distribution for the acute toxicity of zinc to marine fish

Slope	1.240
Intercept	0.460
R^2	0.921
GrandMean	3.662
SumSQ	153.390
CSSQ	5.871
MSE	0.086
Tcrit	1.833
N	11
df	9

Proportion	Probit	Log Central Tendency	SSQ	Log Upper PI	Log Lower PI	Central Tendency	Upper PI	Lower PI
0.05	3.355	2.335	0.077	2.845	1.825	216.456	700.389	66.896
0.1	3.718	2.628	0.071	3.116	2.141	425.012	1307.051	138.201
0.2	4.158	2.983	0.065	3.451	2.516	962.146	2824.065	327.799
0.4	4.747	3.458	0.061	3.911	3.005	2869.010	8144.575	1010.638
0.5	5.000	3.662	0.061	4.114	3.210	4592.796	12994.017	1623.345
0.7	5.524	4.085	0.062	4.543	3.627	12163.230	34912.451	4237.576
0.8	5.842	4.341	0.065	4.809	3.873	21923.671	64349.747	7469.297
0.9	6.282	4.696	0.071	5.184	4.208	49630.990	152631.392	16138.457
0.95	6.645	4.989	0.077	5.499	4.479	97450.853	315323.255	30117.248

5.6.1.16.4 Water-column exposure toxicity data - chronic criterion for zinc in marine waters

No empirical chronic toxicity data for zinc in marine waters was reported in USEPA (1987) for any marine fish species. The only reported chronic toxicity data for zinc to any marine species in USEPA (1987) is a chronic study of survival and reproduction of the mysid *Americamysis bahia* (S. Lussier et al., 1985). The chronic NOEC from S. Lussier et al. (1985) for survival and reproduction was 120 µg/L, the chronic reproductive LOEC was 231 µg/L which resulted in a MATC of 166.5 µg/L.

5.6.1.16.5 Additional marine chronic toxicity data for zinc published since 1987

We have identified no acceptable quality empirical chronic toxicity for zinc to marine fish species published since 1987. Although several studies on zinc effects on growth and reproduction of marine fish have been found, none of them are of the required 30 day exposure duration (C.E. Stephan et al., 1985) needed to derive chronic criteria. The longest duration chronic toxicity study was a 21 day zinc exposure to up to 170 µg/L for black sea bream *Acanthopagrus schlegeli* and the grunt *Terapon jarbua* (Zhang & Wang, 2005). No effect on survival was observed, although there was an increase in metallothionine concentrations at the end of the exposure.

5.6.1.16.6 Effects assessment of marine chronic zinc criterion on listed species

In the absence of empirical chronic zinc toxicity data for any marine fish species, the empirical fish acute 96 hour LC_{50} values were divided by the zinc acute-chronic ratio of 2.208 (USEPA, 1987) to estimate chronic NOEC assessment effect concentrations for the 11 marine fish species with empirical 96 hour LC_{50} values as shown in Table 108.

Table 108. Estimated zinc chronic NOECs for marine fish with empirical 96 hour LC50 values but without empirical chronic NOECs

Species	96 hour LC_{50}, µg/L	Chronic NOEC (LC_{50}/2.208), µg/L
Spot	38,000	17,210
Mummichog	36,630	16,590
English sole	14,500	6567
Sheepshead minnow	11,300	5118
Winter flounder	9467	4288
Striped mullet	7730	3501
Tidewater silverside	5600	2536
Atlantic silverside	3640	1649
Cobia	685	310
Striped bass	430	195
Cabezon	191.4	86.7

The calculated chronic NOECs for zinc toxicity to marine fish were then fit to a species sensitivity distribution to estimate the 5th percentile of the chronic zinc NOECs for marine fish. This 5th percentile (Table 109) is used as the assessment effects concentration for chronic zinc toxicity to of the ESA listed marine fish species in the action area.

Table 109. Species sensitivity distribution for the chronic toxicity of zinc to marine fish

Slope	1.240
Intercept	0.886
R^2	0.921
GrandMean	3.318
SumSQ	126.979
CSSQ	5.870
MSE	0.086
Tcrit	1.833
N	11
df	9

Proportion	Probit	Log Central Tendency	SSQ	Log Upper PI	Log Lower PI	Central Tendency	Upper PI	Lower PI
0.05	3.355	1.991	0.077	2.501	1.482	98.061	317.292	30.306
0.1	3.718	2.285	0.071	2.772	1.797	192.536	592.099	62.608
0.2	4.158	2.639	0.065	3.107	2.172	435.841	1279.246	148.492
0.4	4.747	3.114	0.061	3.567	2.661	1299.537	3689.073	457.783
0.5	5.000	3.318	0.061	3.770	2.866	2080.274	5885.443	735.296
0.7	5.524	3.741	0.062	4.199	3.283	5508.903	15812.071	1919.294
0.8	5.842	3.997	0.065	4.465	3.529	9929.170	29143.297	3382.885
0.9	6.282	4.352	0.071	4.840	3.864	22476.555	69121.420	7308.813
0.95	6.645	4.645	0.077	5.155	4.135	44130.980	142792.597	13638.966

The 5th percentile chronic NOEC of the SSD is 98 µg/L. This final chronic value estimate for all fish species is higher than the chronic marine zinc criterion of 81 µg/L, which indicates that exposure at the level of the marine chronic zinc criterion is not likely to adversely affect rainbow trout (steelhead), Chinook salmon, chum salmon, bull trout, bocaccio or yelloweye rockfish.

5.6.1.16.7 Summary of effect determinations for marine zinc criteria

EPA has made the following determinations on the acute and chronic zinc criteria for the ESA listed species in the marine portions of the action area. These determinations are based on the effects assessment for exposure to this chemical at criteria concentrations as indicated above and in section 5.7, and the indirect effects analysis for prey availability provided in section 5.8. The determinations for marbled murrelet are based on effects to prey availability (see sections 5.4.3.4 and 5.4.5).

Acute criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

Chronic criterion:

Rainbow trout (steelhead) – **Not likely to adversely affect**

Chinook salmon – **Not likely to adversely affect**

Chum salmon – **Not likely to adversely affect**

Bull trout – **Not likely to adversely affect**

Bocaccio – **Not likely to adversely affect**

Yelloweye rockfish – **Not likely to adversely affect**

Marbled murrelet – **Not likely to adversely affect**

5.6.2 Dissolved oxygen, pH, and temperature

Dissolved oxygen, pH, and temperature are interrelated in terms of dynamics and effects, since extremes in all three tend to coincide (higher algal productivity associated with warmer marine temperatures, leading to swings in pH, and drawdown of dissolved oxygen once the algal biomass decays). In addition, dissolved oxygen saturation is controlled in part by temperature. It has been documented that water temperature affects fish more than any other abiotic variable (Beitinger & Fitzpatrick, 1979). Water temperature has been determined to be a controlling factor for many biochemical and physiological processes and exerts strong influence on salmonid behavioral cues (USEPA, 2001, 2003). There are a suite of drivers of diversity leading to different life histories of

salmonids, including: habitat selection, intraspecies size segregation, interspecies niche differentiation, isolating mechanisms, predator avoidance, prey location, escape reactions, and migrations (W. W. Reynolds, 1977). Rockfish are also sensitive to temperature, pH and dissolved oxygen as well as other water quality and habitat features, for the larval through adult lifestages present in the sound (NMFS, 2017a). Overall, the higher maintenance costs from stressful abiotic environments can inhibit efficient energy allocation and reproduction and growth for sensitive fish species in Puget Sound.

Salmon and steelhead in Puget Sound have virtually the same PBFs/PCES for marine waters, including estuarine areas free of obstruction and excessive predation with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater. For bull trout, the relevant PBF is sufficient water quality and quantity to sustain normal reproduction, growth, and survival. The physical and biological features (PBFs) of nearshore habitats to support Puget Sound Georgia Basin rockfish include sufficient quantity, quality, and availability of prey species and suitable water quality and dissolved oxygen to support growth, survival, reproduction, and feeding opportunities.

The methods to determine the effects of the Agency's proposed approval of the Swinomish Tribe's temperature, dissolved oxygen, and pH criteria are similar. They comprise comparing the Swinomish Tribe's criteria to current EPA recommendations, a comparison of the criteria with recent literature on thresholds relevant for each ESU and DPS, or surrogate organisms, where available, inclusion of direct and indirect (prey-based) effects, and, for certain criteria and lifestages an assessment of data within or near the action area that displays the temporal and spatial variability in temperature, DO and pH. The latter enables the determination of finer-scale effects based on likely exposures, in contrast to presuming constant exposure at criteria concentrations/levels on a year-round/constant basis.

5.6.2.1 Dissolved oxygen (marine)

5.6.2.1.1 Dissolved oxygen/hypoxia in marine waters

The effects of hypoxia on salmonids, rockfish, and their prey in the action area can be myriad. Sublethal effects from low oxygen levels can include alterations in metabolic response; reproduction - resulting in a lack of emergence from eggs, egg malformation; reduced growth, respiration; and ultimately mortality above acute thresholds. Although many mobile fish species can escape, so long as there is refuge in the water column available, benthic sessile members of the estuarine ecosystem and/or less mobile species may experience high mortality, depending on the rate of decrease in dissolved oxygen. D. C. Miller, Poucher, and Coiro (2002) provides an LC50 of 1.5 to 3 mg/L for several coastal and estuarine fish. For planktonic zooplankton that are the prey for salmonids and other fish, there is typically a progressive increase in stress and mortality starting at 3 mg/L to 0 mg/L. Copepods and other crustaceans appear to be meiofaunal groups that are very sensitive to hypoxia and also serve as major food sources for salmon and trout (Grego, Riedel, Stachowitsch, & De Troch, 2014; Roman, Stephen, Houde, & Pierson, 2019).

Global ocean DO levels are impacted by both climate change and increased nutrient export from land to marine environments, resulting in a steady decline (Tomasetti & Gobler, 2020). Such drivers as rising sea surface temperatures can impact dissolved oxygen concentrations in multiple ways, including increasing stratification strength (leading to hypoxia and anoxia at depth) and decreasing the solubility of surface water DO. However, the direct influence of anthropogenic nutrient loading on coastal water bodies is significant, as coastal water DO decline has generally been greater than that of the open ocean. In coastal and estuarine environments, ecologically and economically important fisheries are now often

dually impacted by hypoxia and acidification (Tomasetti & Gobler, 2020). As noted by Tomasetti and Gobler (2020), DO at/around the concentration allowed by Oregon's estuarine water quality criteria, 6.5 mg/L, may be protective of additional acidification stress with hypoxia on fisheries, whereas criteria < ~5 mg/L are unlikely to provide this additional protection from this stressor (acidification). However, this depends upon how coincident waters are in time and space that meet both criteria/ranges.

Areas of hypoxia occur in a large swath of Puget Sound and are due to a combination of natural dynamics (Brandenberger, Louchouarn, & Crecelius, 2011) as well as anthropogenic sources (Ecology, 2019). Hypoxia may be found in the terminal and internal basins at mid-depth through bottom waters of Puget Sound, with hypoxia most pronounced in the South Sound, Hood Canal, and Bellingham Bay. Brandenberger et al. (Brandenberger et al., 2011) reported that samples taken from sediment cores from Hood Canal revealed that hypoxia occurred in Hood Canal before European settlement in the area. Therefore, there is an indication that hypoxia can be related to natural forcing such as natural productivity in combination with long residence times for water masses in Hood Canal and other areas of Puget Sound. More recent studies have demonstrated that a combination of anthropogenic stressors and natural conditions contribute to hypoxia within the Puget Sound, at certain times and locations. The anthropogenic stressors exacerbate the extent, magnitude and duration of hypoxia and anoxia, particularly in many of the basins in the South Sound and those with longer residence times for water exchange in the Puget Sound (Ecology, 2019).

Excessive concentrations of nutrients may result in marine waters becoming hypoxic, and hypoxic conditions have been observed throughout Puget Sound (Ecology, 2019). Nutrients are essential for organisms' metabolism, growth, and other functions. Humans add excess nutrients to the waters of Puget Sound through sources such as sewage, septic tank drainage, and other point and non-point sources of pollution (Paulson et al., 2006). The addition of human-derived nutrients can stimulate the growth of algae during the summer months through a process called eutrophication. The algae die, sink to the bottom and decompose, a process that utilizes dissolved oxygen. Therefore, increased levels of nutrients may lead to lower levels of dissolved oxygen in places such as Hood Canal. Dissolved oxygen levels vary seasonally in Puget Sound, with lower levels (hypoxic) at depth during the summer months and higher levels (normoxic) near the surface during the winter (NOAA, USCG, USEPA, & API, 2010).

Fish that are acutely exposed to low DO conditions face a trade-off between increasing swimming activity and energy expenditure to locate hospitable waters with sufficient DO, or decreasing their activity to reduce oxygen requirements (Domenici, Steffensen, & Batty, 2005). Fish eggs and larvae may be particularly vulnerable to low oxygen because of limitations in behavioral avoidance and, in some cases, higher oxygen requirements than adults.

The aquatic life criterion for dissolved oxygen for fish use in marine water is a 1-day minimum of 6.0 mg/L (19-06.130(C)(2)(c), Table 11). There is DO data available for marine stations collected by the Swinomish Tribe in the 1970s and 1990s, as well as more recent data from WADOE's EIM database in the Swinomish ESA action area and they are summarized in section 4.3.4.6 of the Swinomish BE.

The lower quartile of DO concentrations reported by the Swinomish Tribe is above the criterion of 6 mg/L at all marine stations, but a few stations reported minimum DO concentrations below 6 mg/L (Figure 23). DO data reported for the Skagit County Boat Ramp (Figure 25), located in Swinomish Channel (near La Conner), show a range in variability over the year, with monthly median DO concentrations varying between just below 7 mg/L to 10 mg/L, and minima consistently above 6 mg/L.

5.6.2.1.2 Effects analysis of dissolved oxygen/hypoxia on salmon and trout in the action area

Since salmonid spawning and incubation do not occur in marine waters, for the effects to juvenile and adult salmon and trout in the marine portions of the action area are analyzed here. Nationally recommended criteria for dissolved oxygen (USEPA, 1986a) to protect salmonids and other coldwater and warmwater species are as follows:

Table 110. Dissolved oxygen criteria recommendations as presented in EPA 1986 (USEPA, 1986a)

	Coldwater Criteria		Warmwater Criteria	
	Early Life Stages[1,2]	Other Life Stages	Early Life Stages[2]	Other Life Stages
30 Day Mean	NA[3]	6.5	NA	5.5
7 Day Mean	9.5 (6.5)	NA	6.0	NA
7 Day Mean Minimum	NA	5.0	NA	4.0
1 Day Minimum[4,5]	8.0 (5.0)	4.0	5.0	3.0

[1] These are water column concentrations recommended to achieve the required intergravel dissolved oxygen concentrations shown in parentheses. The 3 mg/l differential is discussed in the criteria document. For species that have early life stages exposed directly to the water column, the figures in parentheses apply.

[2] Includes all embryonic and larval stages and all juvenile forms to 30-days following hatching.

[3] NA (not applicable).

[4] For highly manipulatable discharges, further restrictions apply (see page 37)

[5] All minima should be considered as instantaneous concentrations to be achieved at all times.

The criteria are derived from the production impairment estimates which are based primarily upon growth data and information on temperature, disease, and pollutant stresses. The EPA 1986 guidance values are indicative of effects concentrations between no production impairment and slight production impairment. The Swinomish Tribe's one day-minimum of 6 mg/L for lifestages other than early life stages of salmonids far exceeds (is more stringent than) the EPA's 1986 recommendation of instantaneous minimum of 4 mg/L.

In more recent studies on salmonids, the effects data demonstrate that there are higher dissolved oxygen requirements for juveniles and adults that could be present in marine waters in the action area, than the minima identified in EPA's 1986 recommendations for salmonids. However, effects are generally below the Swinomish marine criteria of 6 mg/L. There were some minor reductions in juvenile growth and respiration at 6 mg/L DO from the literature, however, above 6 mg/L, little to no reduction in growth or respiration, and no mortality were reported in these studies for juveniles or adults. For juveniles there was no mortality found at DO levels from 3-13 mg/L, however there was reduced growth

at 3 and 5 mg/L dissolved oxygen. For *Oncorhynchus tshawytscha* (Chinook salmon), growth was reduced in juveniles reared in constant DO concentrations of 3, 4, 6, and 7 (but not 8 or 9) mg/L (1% reduction in growth at 7 mg/L, 7% reduction at 6 mg/L, 29% reduction at 4 mg/L, and 47% reduction at 3 mg/L). Juvenile mortality occurred at DO concentrations of 1.9 and 3 mg/L, and respiratory distress was seen at a DO concentration of 4.9 mg/L (Ahearn, Viers, Mount, & Dahlgren, 2006; Birtwell, 1989; Clarke, Cameron, Stonecypher, Hayes, & Carmichael, 2009; Houde et al., 2019). For *Oncorhynchus kisutch* (Coho Salmon), juvenile mortality occurred after 15 days at DO concentrations of 25% air saturation. Mortality was also observed at 1.6 mg/L. Growth was decreased at 3 (37% reduction), 4 (21% reduction), 6 (4% reduction), and 7 (1% reduction) mg/L DO but not at 8 or 9 mg/L ((Martinez et al., 2020);(B. U. Waller, Black, Burt, Groot, & Rosenthal, 2000);(K. Carter, 2005; U. H. Waller, Rosenthal, & Black, 1997)). For *Oncorhynchus nerka* (Sockeye Salmon), survival of juvenile Sockeye was 90-97% when they were kept in water for 24 hours with DO concentrations between 4.5 and 9.0 mg/L, survival was 45% at levels between 3.0-3.3 mg/L, and there was no survival from brief exposure to a DO concentration of 2.3 mg/L (Ruggerone, 2000).

Juvenile steelhead (*O. mykiss*) require DO concentrations of approximately 7.0 mg/L for proper growth and food conversion efficiency; food consumption declines below 6.0 mg/L (Pedersen, 1987). Maximum swimming speeds for juvenile salmonids are reduced by up to 43% at DO concentrations between 5.7 mg/L to 5.9 mg/L (Jones, 1971) with hampered anaerobic metabolism below 5.1 mg/L (Kutty, 1968) which is required for rapid escape from predators.

For *Oncorhynchus mykiss* (Rainbow Trout, steelhead), growth was reduced in juveniles reared in lower DO conditions (7.3, 6.3, 4.7 mg/L) as compared to higher DO conditions (10.5 mg/L) (Waldrop, Summerfelt, Mazik, Kenney, & Good, 2020). In another recent study of rainbow trout, there was reported 100% survival of juveniles exposed to DO levels of 7, 6, and 5 mg/L, 66.7% survival to exposures of DO at 4 mg/L, and 27.8% survival at a DO level of 3 mg/L (6 hour exposures) (Hou et al., 2020). There was 100% survival of juveniles in DO concentrations of 5 mg/L after 48 hours, and roughly 80% survival at 3 mg/L after 36 hours (Dean & Richardson, 1999). After exposure of juveniles to 1 mg/L for 1 hour, 50% mortality occurred, and 100% mortality after 4 hours of exposure to 1 mg/L (Dean & Richardson, 1999). Adult fish were found to be present in waters with DO concentrations ranging from 4.3 to 9.3 mg/L but absent in waters with DO concentrations of 3.9 mg/L or less (Matthews & Berg, 1996). Adult fish consumed less food, had lower oxygen consumption rates, had lower energy levels, lower lipid contents, and lower heat production under hypoxic conditions (4.5 mg/L) compared to normoxic conditions (7.9 mg/L) (Magnoni et al., 2018).

Juvenile coho and Chinook salmon have been found to avoid DO concentrations of 1.5, 3, and 4.5 mg/L (summer mean temperatures 20.7-22.8C) but not 6 mg/L at a mean temperature of 18.4C (or lower) (K. Carter, 2008). At concentrations below 3 mg/L over long-term (20 - 30 days) exposure, 50% mortality of juvenile salmonids was observed (WDOE, 2002). Mortality was found to be avoided if daily minimum dissolved oxygen concentration remained above 3.9 mg/L (WDOE, 2002).

The available marine water quality data from within or near the action area show that in general, dissolved oxygen concentrations exceed 6 mg/L, with infrequent excursions below 6 mg/L, and all marine stations with data in or near the action area exhibited median dissolved oxygen concentrations at or above 7 mg/L (Figures 22 and 23).

Based on the above review of the effects of dissolved oxygen on salmonid lifestages (adult and juvenile rearing, migration, foraging) present in the marine waters of the action area in comparison to the Swinomish Tribe's dissolved oxygen criteria applicable to marine waters, first that mortality will be avoided for all DPS and ESUs, and that any limited sublethal effects will be mitigated given the variability in DO – as the DO data within the action area show that meeting 6 mg/L as a daily minimum will result in water quality greater than 7 mg/L on average at the marine sites within the action area, the EPA's proposed approval of the Tribe's 1-day minimum DO criterion of 6.0 mg/L to protect the "Salmon and Trout fish use" for marine water **is not likely to adversely affect (NLAA)** Puget Sound Chinook salmon, Puget Sound steelhead, and Puget Sound bull trout and essential fish habitat, and is not likely to adversely modify or destroy bull trout and Chinook designated critical habitat.

5.6.2.1.3 Effects analysis of dissolved oxygen /hypoxia on rock fish in the action area

Few studies of rockfish or bocaccio laboratory exposures to hypoxia are available. Observationally, low dissolved oxygen (DO) events in Hood Canal affect the habitat suitability for listed rockfish (NMFS, 2016a). This includes portions of southern Hood Canal that have had episodic periods of low DO found to be detrimental to rockfish and other fish, including an incident when 26 percent of the local rockfish population were killed due to wind-induced upwelling of hypoxic waters that the rockfish could not escape (Palsson et al., 2009). Increased nutrients from human sources in combination with existing natural sources of nitrogen, may be causing the hypoxia that adversely affects rockfish populations (Palsson et al., 2009). Rock fish have been reported to avoid areas with low DO (<2 mg/L) (Palsson et al., 2009). Brooding rockfish females may be at higher risk from hypoxia due to increased oxygen demand and cascade from hemoglobin to ovary (McClatchie, Goericke, Cosgrove, Auad, & Vetter, 2010).

Although juvenile copper and blue rockfish from the CA current showed some metabolic and behavioral impairment at DO concentrations <6 mg/L (hypoxic, constant conditions) compared to 9 mg/L (normoxic) dissolved oxygen constant conditions in the laboratory, there were significant species-specific differences in tolerance, which indicates that impacts to other species are likely to vary (Mattiasen et al., 2020). Juvenile copper and blue rockfish were able to maintain performance at oxygen concentrations of 4-6 mg/L but were impaired at hypoxic DO levels between 2-4 mg/L (Mattiasen et al., 2020). As discussed above, where 6 mg/L excursions occur in Skagit Bay, based on the date in hand, they are rare occurrences, not constant conditions, and median concentrations of DO are much higher. For Puget Sound, WDFW defines adequate dissolved oxygen habitat in Puget Sound for rockfish as habitat with DO concentrations > 5 mg/L (WDFW).

The marine DO criteria will protect rockfish from mortality (Palsson et al., 2009), the variability over the day and seasonally in DO will ensure that sublethal impacts are minimized (identified for other taxa of rockfish at DO concentrations below 7 mg/L), and the 6 mg/L minimum target will preserve high quality habitat for rockfish, since it is considerably higher than the metric used (5 mg/L) at WA Dept. of Fish and Wildlife to define high quality habitat. Therefore, the EPA's proposed approval of the Tribe's 1-day minimum marine DO criterion of 6.0 mg/L for "Salmon and Trout fish use" **is not likely to adversely affect** yelloweye rock fish and bocaccio in the Swinomish ESA action area, essential fish habitat, and is not likely to adversely modify or destroy bull trout and Chinook designated critical habitat.

5.6.2.2 pH (marine)

5.6.2.2.1 pH in marine waters and organismal effects

Anthropogenic influences such as nutrient enrichment due to runoff from urbanized areas/development, fertilizers and animal waste from agricultural sources, and septic tank releases can affect pH levels. Excess nutrient addition causes increased algal growth, reducing the water column CO_2 concentration, which raises pH during the day. At night, algal respiration releases CO_2, decreasing pH and typically causing large diurnal pH swings in productive waters. The corrosivity of waters in Puget Sound varies seasonally with less corrosive waters present in summer and more corrosive waters present in fall.

The chemistry of marine waters differs from that of freshwater because of the large concentration of salts present (USEPA, 1986e). Some marine communities are more sensitive to pH change than others (NAS, 1974). Normal pH values in seawater are 8.0 to 8.2 at the surface, decreasing to 7.7 to 7.8 with increasing depth (Capurro, 1970). The oceans generally have higher alkalinity due to the higher carbonate content and thus have a greater ability to buffer free hydrogen ions (Middelburg, Soetaert, & Hagens, 2020). Because of the buffering system present in seawater, the naturally occurring variability of pH is less in marine water than in freshwater (USEPA, 1986e). Most marine fish life stages are adapted to this relatively restricted range in variability, and therefore are sensitive to pH changes. Outside of circumneutral ranges, or at the low end of the range, the solubility of metals can increase (USEPA, 2018b). As the level of hydrogen ions increases and pH decreases, or for certain metals at the upper end of the range when pH increases, metal cations which can become more toxic to aquatic life, such as aluminum, lead, copper, and cadmium, and are more likely to be released into the water instead of being adsorbed into the sediment.

Organisms rely on pH to regulate ion transport, and the energy they must expend to maintain intra- and extracellular pH depends on ambient pH. Certain organisms that build calcium carbonate shells/structures are vulnerable to OA, which results in calcium carbonate dissolution, but whether the response is lethal or sublethal is organism-dependent. Organisms that employ aragonite structures are more vulnerable to OA than other calcium carbonate forms. Recent studies on marine fish have reported a range of effects of acidification on neurological responses, otoliths (which are made of aragonite) and homing behaviors, and blood ion balance, however the responses are species-dependent (Haigh, Ianson, Holt, Neate, & Edwards, 2015).

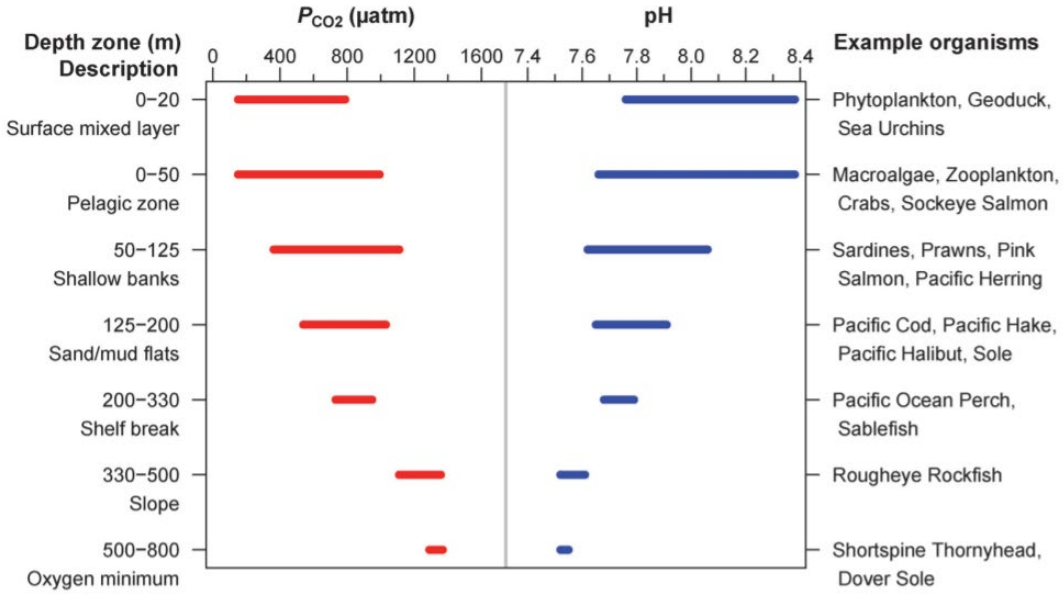


Fig 2. Estimated present-day ranges of P_{CO2} (red) and pH (blue) during spring [40] and summer [27] for various depth zones along the outer BC continental shelf, with typical species found in each zone (see Methods). There are numerous data above 50m and few below 125 m. The number of values in each depth zone from top to bottom are: 70, 116, 33, 45, 5, 4 and 2, respectively. Above 50 m, the distributions of values are skewed, such that high P_{CO2} (low pH) extremes occur less often than the low P_{CO2} (high pH) extremes. Data and R code for this figure are provided as Supporting Information (S2 Code).

Figure 35: pH ranges for outer BC continental shelf aquatic organisms as summarized in Haigh et al. (2015). Excerpted Figure 2[26]

Acidification of marine waters can lead to a variety of impacts on organisms, disrupting physiological processes including acid-base status of extracellular fluids, shell-making and repair, if relevant, depressed metabolism, growth and fitness, depending on their ability to acclimate to the changes (Tomasetti & Gobler, 2020). The likely main path for fish to be affected by acidification, since they can ion-regulate, is indirectly through food-web interactions. For example, many northeast Pacific species including Chinook Salmon (Oncorhynchus tshawytscha) are all largely dependent on krill production (Haigh et al., 2015). Another consideration is that Puget Sound and the northeast Pacific are hot spots for the presence of domoic acid. These neurotoxins can be transferred to higher trophic levels and with increasing acidification, the toxicity of these blooms has been reported to increase, based on laboratory experiments (Haigh et al., 2015).

Prey:

A decline in pteropod abundance (Bednaršek et al., 2019; Bednaršek, Pelletier, Ahmed, & Feely, 2020) linked to acidification in Puget Sound, could lead to a shift in diet toward greater predation on juvenile fish such as salmon, and pteropods are also an important food source for Pink Salmon in the first year of marine life (Haigh et al., 2015). Feely, Alin, Sabine, Warner, and Devol (2010) reported that some of the most abundant primary producers such as coccolithophores and pteropods (marine phytoplankton) which are the base of the food web for upper trophic levels, including marine fish larvae and salmon, are very vulnerable to dissolution by ocean acidification. Surface waters of Puget Sound are typically under-

[26] Note Publication is open-access- Copyright: © 2015 Haigh et al. This is an open access article distributed under the terms of the Creative Commons Attribution License, which permits unrestricted use, distribution, and reproduction in any medium, provided the original author and source are credited.

saturated with respect to aragonite during October–March, and super-saturated during April–September. The highest pCO2 and lowest pH occurred during the corrosive October–March period, while the lowest pCO2 and highest pH occurred during the super-saturated April–September period (Pelletier, Roberts, Keyzers, & Alin, 2018). At some locations in Puget Sound, during summer through fall, daily variability in pH can span 0.8 pH units (Pacella, Brown, Waldbusser, Labiosa, & Hales, 2018).

Indirect effects – prey base for salmonids and rockfish

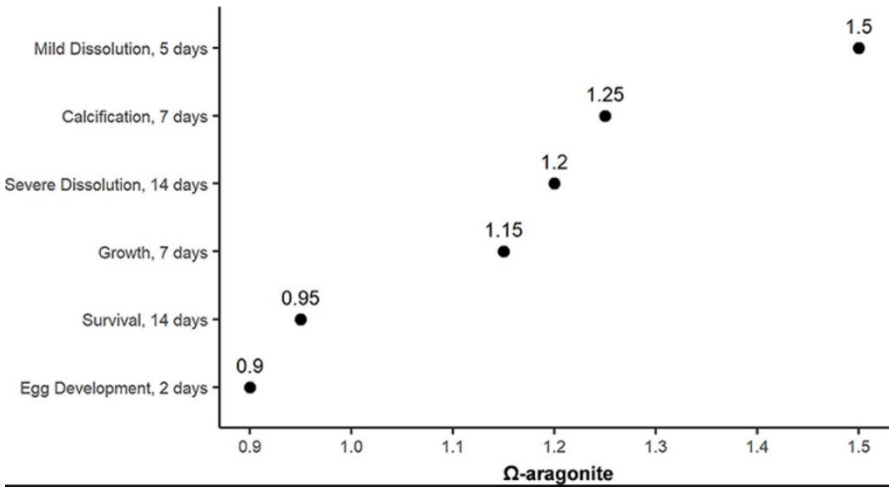


Figure 36: Synthesis of thresholds of ocean acidification impacts on calcifying pteropods excerpted from Bednaršek et al. (2019)[27]

[27] Open access publication: Copyright © 2019 Bednaršek, Feely, Howes, Hunt, Kessouri, León, Lischka, Maas, McLaughlin, Nezlin, Sutula and Weisberg. This is an open-access article distributed under the terms of the Creative Commons Attribution License (CC BY). The use, distribution or reproduction in other forums is permitted, provided the original author(s) and the copyright owner(s) are credited and that the original publication in this journal is cited, in accordance with accepted academic practice. No use, distribution or reproduction is permitted which does not comply with these terms.

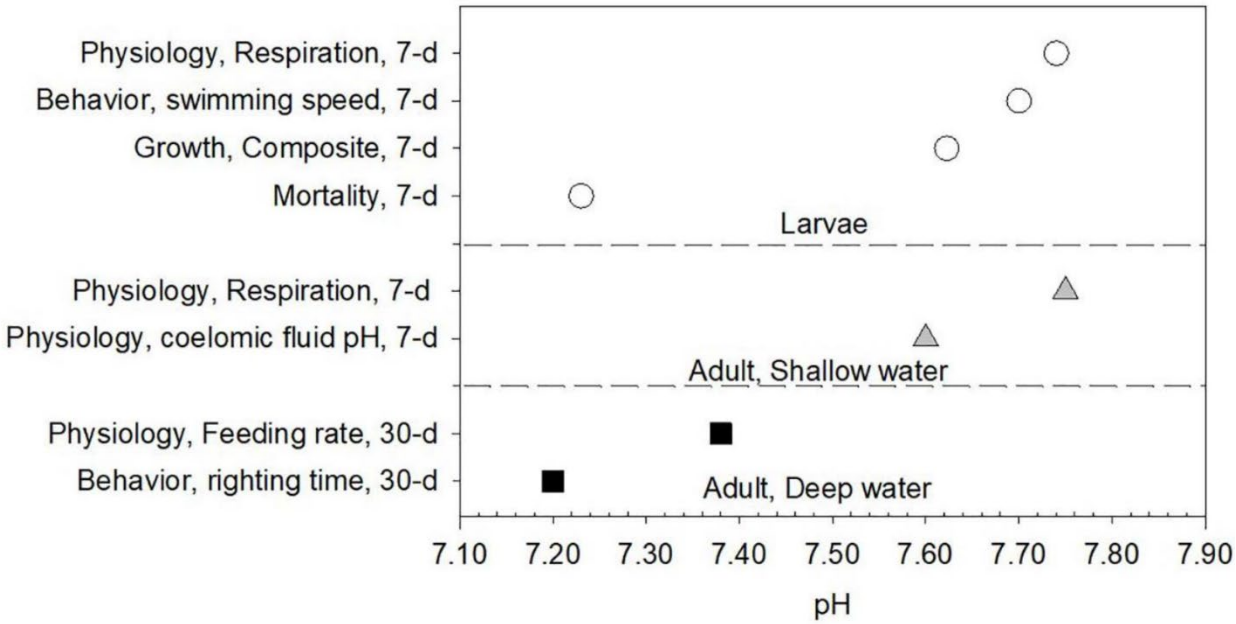


Figure 37: Echinoderm effects thresholds at a range of pH values. Sublethal larval effects at pH = 7.75 at a duration of 7 days or more (Bednaršek et al., 2021).

The survival probability for pteropods was highest under supersaturated conditions at Ωar > 1.2 (~pH = 7.8) constant exposures, declining linearly with saturation state. (See Figure 36). Likewise, echinoderm larval sublethal effects were observed at exposures of 7 days or more at pH 7.75 or less (Bednaršek et al., 2021). (See Figure 37).

5.6.2.2.2 pH data

The Tribe collected pH data at marine stations in the Swinomish action area, as summarized in section 4.3.4.7 (Table 67) of the BE. The lowest value of 6.2 was measured at unnamed tributary No. 2 to Skagit Bay near La Conner and the highest value of 8.8 was measured at WHITMARSH-2662-SW-05.

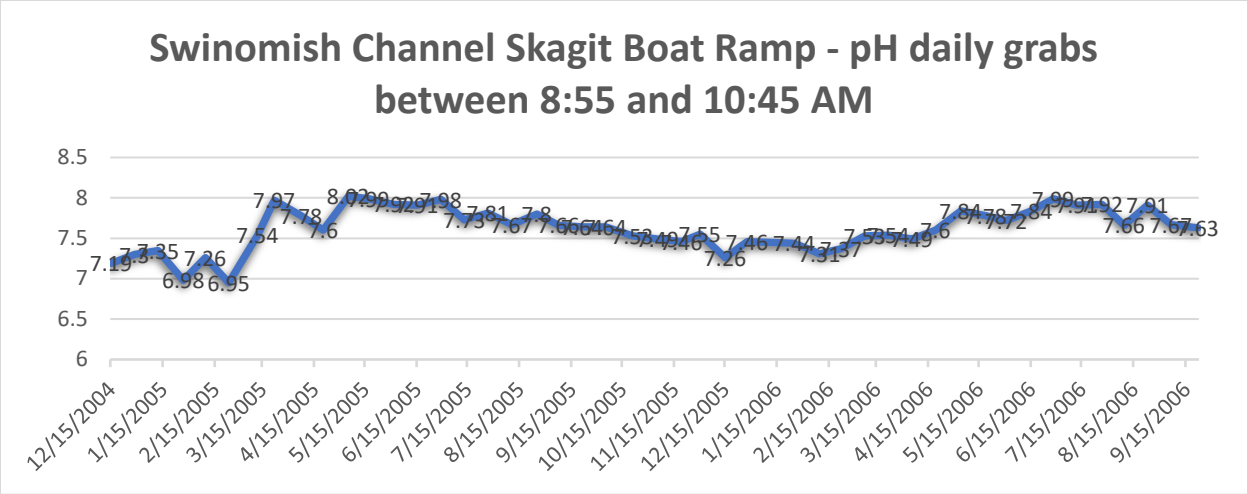


Figure 38: Data for two years (morning grabs) at the Skagit Boat Ramp demonstrate the lower-end pH interannual variability. Although it is likely that pH fell below the lowest values overnight through dawn, these data demonstrate that starting from 8-10AM and for the rest of the day, pH met or exceeded the lower bound of the range of the Swinomish Tribe's criteria over the two-year period. This information on daily variability demonstrates that aquatic life in the action area are not exposed to constant pH at the criteria values.

The Swinomish Tribe's aquatic life criterion for pH for fish use in marine water is, "may not be less than 7.0 or greater than 8.5, with a human-caused variation within above range of 0.2 units" (19-06.130(C)(2)(d), Table 12).

5.6.2.2.3 Effects analysis of marine pH on salmon and trout in the action area

EPA's recommended pH criteria for marine aquatic life are to be within a range of 6.5 to 8.5 pH units. In open ocean waters where the depth is substantially greater than the euphotic zone, additionally, the pH should not deviate more than 0.2 units outside of the naturally occurring variation together with the range of 6.5 to 8.5 (USEPA, 1986e). Therefore, the Tribe's pH criteria are more conservative than the EPA criteria at the low end, 7.0 instead of 6.5 pH units, and equivalent at the high end of the EPA range, 8.5 pH units, and additional include the EPA's recommended additional requirement for pH to remain within 0.2 pH units of the naturally varying pH. Ammonia toxicity can increase with decreasing pH, and salmonids are particularly sensitive to ammonia (EPA's (USEPA, 1989) recommended saltwater ammonia criteria are for a pH range of 7.0-9.0 for that reason), and therefore pH values should allow for appropriate calculation of criteria to protect aquatic life from ammonia toxicity.

Chronic low pH effects may alter salmonid spawning behavior, reduce egg viability, decrease hatch, and reduced survival of the early life stages (Jordahl & Benson, 1987). However, salmonid early life stages are not present in the action area. Vulnerable life stages of chinook salmon are sensitive to pH below 6.5, including otolith formation in early life stages (Geen, Neilson, & Bradford, 1985). In freshwaters, the critical value of pH for rainbow, brook, and brown trout metabolic effects, at the low end, is about 5.5 (J. Baker et al., 1990). Rainbow trout exposed to pH 4.0 for 72 hours had impaired oxygen transport and reduced scope for activity (Ye, Randal, & He, 1991), and exposure of rainbow trout to pH 5.0 reduced critical swimming velocity to 67% compared to pH 7.0 (Ye & Randal, 1991).

High pH values of 9.0 or above have been found to significantly impact salmonids (K. Carter, 2008), however, the Swinomish Tribe's range upper limit is 8.5, well below the 9.0 effects level.

Considering the salmonid food base in marine waters, pteropod exposures for two weeks at pH of ~7.8 or less, constant exposures, resulted in severe dissolution and mortality for pteropods (Bednaršek et al., 2019). Less is known about shorter-duration exposures, however, two-day exposures at lower end values found in marine waters have been found to cause egg mortality (Bednaršek et al., 2019). Since pteropods are planktonic, water mass mixing and transport play a role ultimately in their exposures and Skagit Basin was identified as a location (P4) vulnerable to acidification's effects on pteropods (Bednaršek et al., 2020).

The pH effects data show that the pH ranges are not likely to result in direct lethal or sublethal effects for threatened and endangered salmonids present in the action area, however, there would likely be significant effects to the prey base for salmonids from short duration constant exposures to the minimum threshold of 7 through ~ 7.8 pH units within the action area. In addition, in fall through spring, the data show that pH in the action area is consistently lower than the protective pH levels for the prey for salmonids, and there are many times of day that also are impactful to prey. EPA has determined that the Tribe's supplementary narrative criterion comprising a cap of 0.2 pH units on anthropogenic sources within this pH range could sufficiently limit any excursions below pH thresholds that affect species, given that federal permitting will provide detailed review/consultation on individual permits and could account for the diurnal and seasonal natural variability in pH. If the prey base apportionment could be better constrained between e.g. species that are less likely to be affected by lower end pH values (such as krill), and those that are more affected (e.g., echinoderm larvae, pteropods), this could be a mitigating factor, however EPA did not have the data available to identify diet apportionment. Likewise, the action area is spatially constrained, however, it is difficult to constrain the extent to which waters and planktonic prey may be circulating within Skagit and Padilla Bays and may be affected. Should this water movement and duration of exposure be better quantified to a narrower impact, it could be a mitigating factor, however EPA did not have the data to make such a determination. Therefore, EPA's proposed approval of the Swinomish pH criteria **is likely to adversely affect** Puget Sound Chinook salmon, bull trout and Puget Sound steelhead, and Chinook salmon and bull trout critical habitat.

5.6.2.2.4 Effects analysis of pH on rockfish in the action area

Few studies on the direct effects of acidification on rockfish are available. There is evidence that future anticipated ocean acidification could have serious consequences on behavior and sensory functions important to recruitment, settlement, prey and predator detection, and overall survival in other fishes (e.g., Munday et al. (2009) Chung, Marshall, Watson, Munday, and Nilson (2014)). Hamilton, Holcombe, and Tresguerres (2014) found that exposure to expected future surface ocean acidified conditions caused anxiety in juvenile split-nose rockfish (Sebastes diploproa), comprising a one-week continuous exposure at pH at 7.75. After being returned to the control tank conditions, anxiety remained for 7 days but was completely resolved after 12 days. In studies of certain rockfish species of the CA Current system, it has been found that for copper rockfish (S. caurinus), reduced critical swimming speed, depressed aerobic scope, changes in metabolic enzyme activity, and increases in the expression of genes were documented at high pCO2 exposure. Blue rockfish (S. mystinus), in contrast, only exhibited significant changes in the expression of muscle structural genes as a function of pCO2, indicating greater acclimation potential than copper rockfish, so it is likely that effects of acidification will be species-

specific. Reproduction and larval size of prey items for larval and juvenile rockfish are negatively affected by acidification (Love et al. 1991; Love et al. 2002).

The average pH for marine water in the Swinomish action area is 8.1 and EPA's recommended criteria for pH for marine aquatic life are a range of 6.5 – 8.5. The Tribe's criterion for pH for marine water is a range of 7-8.5, with an upper limit of 8.5 pH units.

Per the Puget Sound Georgia Basin Recovery Plan for Rockfish (NMFS, 2017a): "There have been very few published studies to date on direct effects of ocean acidification on rockfish. In other fishes, there is evidence that ocean acidification could have serious consequences on behavior and sensory functions important to recruitment, settlement, prey and predator detection, and overall survival (e.g., Chung et al. (2014); Munday et al. (2009); Simpson et al. (2011)). In a laboratory setting, OA did result in changes to juvenile splitnose rockfish (Sebastes diploproa) behavior (Hamilton et al., 2014). More research is needed to better understand the effects of ocean acidification on rockfish and their ecosystems and to evaluate if this threat is limiting listed rockfish recovery."

The pH effects data show that pH ranges are not likely to cause mortality in rockfish. However, it is possible that the exposures to the criteria values at the low end of the range will result in sublethal effects for rockfish. In addition, the prey base will likely be affected at the low end of the criteria ranges between 7 through ~ 7.8 pH units. If the prey base apportionment could be better constrained between e.g. species that are less likely to be affected by lower end pH values (such as krill), and those that are more affected (e.g., echinoderm larvae, pteropods), this could be a mitigating factor, however EPA did not have the data available to identify diet apportionment. Likewise, the action area is spatially constrained, however, it is difficult to constrain the extent to which waters and planktonic prey may be circulating within Skagit and Padilla Bays and may be affected. Should this water movement and duration of exposure be better quantified to a narrower impact, this could be a mitigating factor, however EPA did not have the data to make such a determination. Should this water movement and duration of exposure be better quantified to a narrower impact, it could be a mitigating factor, however EPA did not have the data to make such a determination. Therefore, EPA's proposed approval of the Swinomish Tribe's marine water pH criteria **is likely to adversely affect** Puget Sound yelloweye rock fish and bocaccio and their critical habitat.

5.6.2.3 Temperature (marine)

Water temperatures significantly affect the distribution, health, and survival of native salmonids in the Pacific Northwest (USEPA, 2003). Salmonids are ectothermic (cold-blooded) and experience adverse health effects when exposed to temperatures outside their optimal range. In streams and rivers in the Pacific Northwest, salmonids survived and were greater in population under historical water temperature patterns (i.e., prior to significant anthropogenic impacts that altered temperature patterns) in Pacific Northwest streams and rivers (USEPA, 2003). Mortality can result from acutely lethal exposures. Associated chronic and sub-lethal effects include but are not limited to reduced juvenile growth, malformed or reduced emergence of from eggs and malformed alevins, increased incidence of disease, reduced viability of gametes in adults prior to spawning, increased susceptibility to predation and competition, and suppressed or reversed smoltification. For vulnerable fish populations, such as the endangered or threatened salmonids of the Pacific Northwest, there are concerns that chronic and sub-lethal effects have and will continue to reduce the overall health and size of salmonid populations (USEPA, 2003). For example, a steady decline in the marine survival for coho salmon smolts over the

past four decades is correlated with warming in the Georgia Straight, where sea surface and sea bottom temperatures have increased by approximately 1°C since 1970 (R.J. Beamish et al., 2010).

Marine temperature data are available in the Water Quality Portal, provided by the Swinomish Tribe for reservation waters. Results are summarized in Figures 27-29 of the Swinomish BE and they indicate that the upper quartile or 75th percentile of the temperature data are below 16°C at most marine stations. Historical temperature data (1994-2014) from nearby Padilla Bay nearshore areas show that daily mean sea surface temperatures are typically less than 15°C during the summer maximum period with daily fluctuations in excess of 10°C, and summer maximum temperatures in recent years exceeding those daily means by up to two degrees (2015-2019) (PSEMP, 2020). In addition, available surface temperature data for Skagit Bay, Swinomish Channel, ranged from 7.3–13.9°C from March – May, 2006 (USGS, 2007).

The Tribe's 1-DADmax aquatic life (fish use) criterion for marine water is 16°C (60.8°F).

Note that the State of Washington's aquatic life temperature criteria applicable to marine water of excellent quality is 16 ° C (60.8°F) as a 1-DADmax; this criterion is in effect for state waters adjacent to the Tribe's waters.

5.6.2.3.1 Effects analysis of temperature on salmon and trout migration corridors

The marine portions of the Swinomish ESA action area begin where freshwater discharges into the Skagit Bay. These estuarine waters comprise Swinomish Channel, Sullivan Slough and the Skagit River in the action area which serve as a migration corridors for adult salmon and trout migrating into Puget Sound. Juvenile salmon outmigrate through the same pathways and therefore may be affected. Salmon and steelhead smoltification take place in nearshore areas of the Puget Sound, including Skagit Bay.

The Swinomish Tribe's adopted temperature criterion to protect aquatic life uses in marine water is a 1-day maximum temperature (1-DADMax) of 16°C (19-06.130(C)(2)(b), Table 10) to protect the fish use designation of salmon and trout migration corridors. The EPA Region 10 recommended temperature criteria (USEPA, 2003) to protect salmonid migration corridors only is 20°C as a 7D. To protect juvenile rearing and migration together, the criterion recommendation is 18°C as a 7DADMax, and to protect bull trout foraging, migration, and overwintering the criterion recommendation is 16°C as a 7DADMax.

<u>PS Bull trout DPS and assessment of other salmonid adult and juvenile migration:</u> Since the Tribe's criterion is 16°C as a 1-day maximum is more conservative than the EPA's Temperature Guidance (USEPA, 2003) recommended temperature criteria to protect salmonid migration corridors, migration corridors and juvenile rearing, and bull trout foraging, migration, and overwintering. Although exact timing and location of occupancy of juvenile bull trout in Puget Sound estuarine waters is not well known, in a recent study of the Elwha River estuary, for example, juvenile bull trout were caught in every month except for August (Lincoln, Shaffer, & Quinn, 2018). Although bull trout generally prefer temperatures of less than 15°C (USEPA, 2003), USFWS (2015) determined in its opinion on EPA's approval of Oregon's temperature WQS that Oregon's criteria to protect bull trout foraging, migration and overwintering (core cold water habitats) of 16°C as a 7DADMax would not jeopardize bull trout or adversely modify or destroy critical habitat, and the USFWS cited to multiple studies indicating that bull trout can be found using habitats with stream temperatures approaching 20°C when foraging and migrating(P. J. Howell, Dunham, & Sankovich, 2009; M. C. Nelson & Nelle, 2008; Ott & Maret, 2003). In addition, a 16°C one-day maximum criterion is more conservative than a 16°C 7DADMax criterion,

insofar as the latter could theoretically allow up to half the days within a given week to exceed the criterion, given the averaging period, instead of the 16°C one-day maximum, which is not-to-be-exceeded. For these reasons, adverse temperature effects to bull trout migration through Swinomish marine waters are unlikely to occur from the Swinomish Tribe's criterion of 16°C as a 1-day maximum, and because bull trout are the most sensitive of all the salmonids migrating as adults and juveniles through the action area, the criterion should be fully protective of the other ESU/DPS.

In addition to the presence of migrating adults and juveniles at different times of year in the Sound, salmon and steelhead smoltification take place within Puget Sound nearshore environments prior to when salmon and steelhead enter into the open ocean outside of the Sound. In particular, preceding winter sea surface temperatures have been linked to successful coho smoltification and transition (Logerwell, Mantua, Lawson, Francis, & Agostini, 2003), with cooler temperatures promoting smoltification. The recommended criterion from the Temperature Guidance (USEPA, 2003) to protect salmon and steelhead smoltification is 14°C as a 7DADMax. Although the Swinomish Tribe's criterion is 16°C as a one-day maximum, smoltification during juvenile outmigration would be protected, as 14°C as a 7DADMax (recommendation from the Temperature Guidance (USEPA, 2003)) would be met in spring (the latest smoltification likely to take place is in June) when juvenile outmigration occurs for salmon and steelhead, if the 16°C criterion is met at the peak of the summer (See environmental baseline section, Figure 29).

Effects to the prey base from EPA's proposed action are negligible. Benthic invertebrates are sensitive to temperature with acclimation potential linked to the typical range of conditions experienced naturally. As described above, the maximum temperature allowed by the criterion comports with the natural sea surface temperature maximum in summer. For fish serving as prey, the effects are not likely to be different than those for rockfish (see below), salmon, and trout. the EPA's proposed action to approve the 16°C criterion is not likely to adversely affect the prey base for salmon, trout, and rockfish. However, note that under warming scenarios of 1-4°C, significant impacts to eastern Pacific marine foodwebs are expected to occur, both for forage fish, invertebrates and other species (e.g., Cavole et al. 2016).

Based on the analysis above that the Swinomish Tribe's marine criterion is as or more protective than EPA's Temperature Guidance Recommendations (USEPA, 2003) for migration corridors for salmon and steelhead, and migration, foraging and overwintering for bull trout, and that if the criterion is met in summer, sufficiently cold waters during the smoltification period will be available, and further, that the PCEs/PBFS of each ESU and DPS, to provide sufficient water quality in estuarine environments to promote survival, growth, and reproduction, will be met, the EPA has determined that the Agency's approval of the Tribe's "Salmon and Trout Fish Use" freshwater 16 °C 7-DADMax criterion for estuarine and marine waters is **not likely to adversely affect** coastal, marine and Puget Sound salmon and trout, or essential fish habitat for salmon, and not likely to adversely modify or destroy bull trout and Chinook designated critical habitat.

5.6.2.3.2 Effects analysis of temperature on rockfish in the action area

There is limited data on rockfish effects thresholds for temperature. From field data and modeling on an interannual scale, good recruitment years for bocaccio were found to correspond to cooler marine temperatures, specifically cooler during the fall and winter and when upwelling was stronger during the fall, seemingly promoting reproductive success (Tolimieri & Levin, 2005). This relationship is attributed to the correspondence between colder temperatures, increased upwelling, and higher ocean

productivity and support of larvae or biological cues and larval growth rates (indirect and/or direct effects). Mean annual and seasonal temperatures for the warm-shifted PDO (Pacific Decadal Oscillation) are up to 3.1°C higher than for the cold phase (Hartmann & Wendler, 2005), and within the Salish Sea, upwards of 1°C increases in surface, intermediate, and bottom water temperatures since the 1970s have been observed (R.J. Beamish et al., 2010). Precipitous rockfish and bocaccio declines over the last five decades have been attributed to environmental factors such as warming temperatures, as well as overfishing and related decreases in diversity (NMFS, 2017a). The Swinomish Tribe's criterion of 16°C as a daily maximum, however, appears to comport well with historical data on summer maxima in Puget Sound, given the available data (PSEMP, 2020), and if met in summer, the seasonally modulated temperature progression will result in targeting historically colder temperatures in the shoulder seasons and winter.

For NPDES permitting, it is important that implementation ensure the protection of the above-mentioned naturally colder temperatures. Narrative criteria and/or antidegradation policies could be applied in order to protect waters that are naturally colder than criteria, at the times when they occur. Regarding implementation of the Tribe's temperature criteria, EPA is the NPDES permitting authority for tribal waters, and effective implementation of the antidegradation requirements would ensure that water temperatures are not degraded outside of the summer maximum period. In addition, if EPA approves the Tribe's marine temperature criterion, the decision document will include a recommendation that the Tribe adopt seasonal temperature criteria for its marine waters that explicitly protect all life stages of rockfish and bocaccio.

As mentioned above, negligible impacts to the prey base from the EPA's action are expected to occur. There are some data to indicate that marine temperature regime shifts to warmer conditions may not be wholly harmful to rockfish in Puget Sound, due to cascading trophic effects that can reduce predation on rockfish and increase rockfish prey abundance, but more study is needed.

For the above reasons, the Tribe's 16 °C 1-DADMax criterion is **not likely to adversely affect** Puget Sound yelloweye rockfish , bocaccio, their critical habitat and essential fish habitat.

5.6.2.4 Aquatic-dependent wildlife effects analysis for marine dissolved oxygen, pH and Temperature

5.6.2.4.1 Effects analysis of dissolved oxygen/hypoxia on aquatic-dependent wildlife in the action area

Unlike toxic chemicals or metals that accumulate in the tissue of fish and then accumulate in the tissue of the predator species, DO exposure to prey species will have no direct effect on the predator species. Any effects to aquatic-dependent wildlife species resulting from the approval of the water quality criteria would be effects to the prey population such that numbers of prey items were reduced in availability to these aquatic-dependent species. Therefore, all effects to aquatic-dependent wildlife are expected to be indirect effects from effects to the prey base.

Marbled murrelets

Marbled murrelets have a diverse diet comprising a variety of invertebrates and fish species. As described above, invertebrates and sensitive cold-water fish species (rockfish, salmon, trout) are not likely to be adversely affected by the EPA's proposed approval of the Swinomish Tribe's marine

dissolved oxygen criteria. Therefore, the Agency's proposed action to approve the Swinomish Tribe's marine dissolved oxygen criteria is **not likely to adversely affect** marbled murrelets.

Humpback Whales

Humpback whales have a diverse diet comprising krill and a variety of fish species. As described above, invertebrates and sensitive cold-water fish species (rockfish, salmon, trout) are not likely to be adversely affected by the EPA's proposed approval of the Swinomish Tribe's marine dissolved oxygen criteria. Therefore, the Agency's proposed action to approve the Swinomish Tribe's marine dissolved oxygen criteria is **not likely to adversely affect** humpback whales.

SRKW (*Orcinus orca*)

SRKW prefer Chinook salmon as prey, although they also feed on other fish and squid. Chinook salmon make up the majority of SRKW prey and are not likely to be adversely affected by the Agency's action to approve the Swinomish Tribe's marine dissolved oxygen criteria. Therefore, the EPA's proposed action to approve the Swinomish Tribe's marine dissolved oxygen criteria likewise, is **not likely to adversely affect** SRKW and its critical habitat.

5.6.2.4.2 Effects analysis of pH on aquatic-dependent wildlife in the action area

Unlike toxic chemicals or metals that accumulate in the tissue of fish and then accumulate in the tissue of the predator species, pH exposure to prey species will have no direct effect on the predator species. Any effects to aquatic-dependent wildlife species resulting from the approval of the water quality criteria would be effects to the prey population such that numbers of prey items were reduced in availability to these aquatic-dependent species. Therefore, all effects to aquatic-dependent wildlife are expected to be indirect effects from effects to the prey base.

Marbled murrelets

Marbled murrelets prey upon a diverse assortment of fish and invertebrates, and given that diversity, there is a high likelihood that there is some dependence on prey species that are likely to be adversely affected at the low end of the pH range of the Tribe's criteria. Therefore, the Agency's action to approve the Swinomish Tribe's marine pH criteria is **likely to adversely affect** marbled murrelets. EPA is making this determination based on a general knowledge of impacts to prey for the marbled murrelets, however, a more detailed apportionment of prey and better-defined relative impacts of effects for each of the individual prey species of marbled murrelets may provide more specific insight.

Humpback Whales

The Swinomish Tribe's pH criteria include a cumulative allowance for anthropogenic sources of 0.2 pH units, which could allow pH of 6.8-7.0 at the low end of the allowable pH range. Humpback Whales feed on zooplankton, primarily krill, and a variety of fish. The evidence for impacts to krill at the lower end of the Tribe's criteria pH range is mixed. Crustaceans overall have been found to be more resilient to acidification than other animal species (Wittmann & Portner, 2013). For *Euphausia pacifica* (the dominant krill in the northeastern Pacific) some studies have shown resilience at pH levels below 8.0, for example juveniles tolerant of pH of 7.54 for 7 days (Yamada & Ikeda, 1999). However, below 7.0 under constant conditions, adverse impacts have been observed for adults and embryos. For example, constant exposures of *E. pacifica* at pH conditions at or below 6.96 pH units resulted in significant

reductions in adult growth, with a critical mortality threshold at 6.92, and 100% mortality at pH 6.89 after 10 days of exposure (Cooper et al., 2016). Between pH of 6.96 and 6.92, sublethal impacts were observed. Although impacts to larvae are unknown, significant adverse effects to emergence and larval development under constant conditions under acidification (typically high pCO2), pH at or below 7.7, have been observed in Antarctic krill (e.g., Kurihara 2008; Kawaguchi et al. 2011, 2013; Rollands et al. 2021). Prey for Humpback whales are a diverse assortment, and given that diversity, it is likely that there is some dependence on prey species that are likely to experience adverse effects at the low end of the pH range of the Tribe's criteria. However, as described in section 5.1.2, the action area constitutes a small fraction of habitat available for prey of humpback whale, and any adverse effects to potential prey species in the action area would not be expected to have a significant effect on the prey base of humpback whale. Based on the absence of sightings of humpback in the action area (section 5.1.2) and on the small fraction of potential prey that would be affected by pH levels at the low end of the Tribe's pH criteria, EPA has determined that its approval of the Swinomish Tribe's pH criteria for marine waters is **not likely to adversely affect** humpback whales.

SRKW (*Orcinus orca*)

SRKW prefer Chinook salmon as prey, although they also feed on other salmonids, fish species, and squid. Chinook salmon make up the majority of SRKW prey; Chinook in the action area are likely to experience adverse effects at the low end of the pH range of the Tribe's criteria. However, as described in section 5.1.2, north Puget Sound Chinook stocks account for approximately 5% of the fall/early winter and mid-winter/early spring diets of SRKW (Hanson et al., 2021), and Chinook that enter the action area would represent a fraction of this small percentage. Based on the small fraction of potential prey that would be affected by pH levels at the low end of the Tribe's pH criteria, EPA has determined that its approval of the Swinomish Tribe's pH criteria for marine waters is **not likely to adversely affect** SRKW and its critical habitat.

5.6.2.4.3 Effects analysis of temperature on aquatic-dependent wildlife in the action area

Unlike toxic chemicals or metals that accumulate in the tissue of fish and then accumulate in the tissue of the predator species, temperature exposure to prey species will have no direct effect on the predator species. Any effects to aquatic-dependent wildlife species resulting from the approval of the water quality criteria would be effects to the prey population such that numbers of prey items were reduced in availability to these aquatic-dependent species. Therefore, all effects to aquatic-dependent wildlife are expected to be indirect effects from effects to the prey base.

Marbled murrelets

Marbled murrelets have a diverse diet comprising a variety of invertebrates and fish species. As described above, invertebrates and sensitive cold-water fish species (rockfish, salmon, trout) are not likely to be adversely affected by the EPA's proposed approval of the Swinomish Tribe's marine temperature criteria. Therefore, the Agency's proposed action to approve the Swinomish Tribe's marine temperature criteria is **not likely to adversely affect** marbled murrelets.

Humpback Whales

Humpback whales have a diverse diet comprising krill and a variety of fish species. As described above, invertebrates and sensitive cold-water fish species (rockfish, salmon, trout) are not likely to be adversely

affected by the EPA's proposed approval of the Swinomish Tribe's marine temperature criteria. Therefore, the Agency's proposed action to approve the Swinomish Tribe's marine temperature criteria is **not likely to adversely affect** humpback whales.

SRKW (*Orcinus orca*)

SRKW prefer Chinook salmon as prey, although they also feed on other fish and squid. Chinook salmon make up the majority of SRKW prey and are not likely to be adversely affected by the Agency's action to approve the Swinomish Tribe's marine temperature criteria. Therefore, the EPA's proposed action to approve the Swinomish Tribe's marine temperature criteria likewise, is **not likely to adversely affect** SRKW and its critical habitat.

5.7 Multiple Routes of Exposure for Fish Species

The multiple routes of exposure assessment evaluates water quality criteria by considering all possible exposure pathways of chemicals to aquatic species: inhalation and uptake across gills or other respiratory surfaces, dietary ingestion, and dermal exposure. These exposure pathways are evaluated by comparing the internal chemical concentration in tissues, which integrates chemical exposure from all exposure pathways to a single concentration of the chemical in the whole body of aquatic species.

The measured whole body tissue concentration of a chemical is compared to a tissue screening concentration (TSC) derived by multiplying the lower of the freshwater or marine chronic aquatic life criterion for each chemical by the bioconcentration factor for that chemical. The primary assumption behind this approach is that since aquatic life criteria are designed to be protective of 95% of all aquatic genera from adverse effects on survival, reproduction and growth, the tissue residue bioaccumulated from exposure to criteria concentrations should also be lower than residues that adversely affect survival, reproduction and growth of 95% of aquatic genera. If the TSC is lower than the tissue residue in a species associated with adverse effects on survival, reproduction, growth or behavior of that species, the TSC represents a water column concentration expressed as a tissue residue that does not adversely affect the receptor species. The tissue residues used in this assessment are all presented in or derived from measured residue-effects studies of exposure to each chemical individually. No mixture studies or field exposures to multiple chemicals were used to identify residues associated with adverse effects on aquatic species. All tissue residue concentrations in this section are in units of mg/kg whole body wet weight.

A detailed description of this multiple routes of exposure analysis and the literature supporting it is presented in the methodology section of this BE, section 5.4.3.3. This approach utilizes the same TSC for both freshwater and marine waters. Thus, the conclusions of this multiple routes of exposure assessment are equally applicable to fully aquatic species in both freshwater and marine waters. Water column based aquatic life criteria are separately derived for freshwater and marine waters because, among other reasons, water column parameters such as pH, hardness and salinity affect the bioavailability of chemicals from water into aquatic species. Once accumulated into tissues, however, the tissue residue of a chemical associated with a given toxic effect is comparable in both freshwater and marine species.

Multiple routes of exposure analyses were completed for the chronic criteria assuming exposure at the chronic criteria level and steady state bioaccumulation conditions as described in section 5.4.3.3. A

multiple routes of exposure assessment was completed for the following chronic criteria chemicals in assessment category 4 (see Table 2 and Table 3):

- Arsenic (marine CCC)
- Carbaryl (freshwater and marine CCCs)
- Chlorpyrifos (freshwater and marine CCCs)
- Chromium VI (freshwater and marine CCCs)
- Diazinon (freshwater and marine CCCs)
- Lead (freshwater and marine CCCs)
- Malathion (freshwater and marine CCCs)
- Nickel (freshwater and marine CCCs)
- Nonylphenol (freshwater and marine CCCs)
- Pentachlorophenol (marine CCC)
- Zinc (freshwater and marine CCCs).

Of the remaining six chemicals assessed in this BE, chlorine, hydrogen cyanide and hydrogen sulfide are gases under ambient conditions, with little bioaccumulative potential. The high vapor pressure and Henry's Law constant for acrolein indicate it will volatilize into air when released into the environment, making it unlikely to bioaccumulate in aquatic species to any appreciable extent. Elemental phosphorus and iron have no residue-effects information available. These chemicals are therefore not assessed via a multiple routes of exposure evaluation.

Tables 2 and 3 (section 1.6.1) and the assessment summaries for each criterion in sections 5.5 and 5.6 incorporate the findings of the multiple routes of exposure assessment. Since the only aquatic ESA listed species in this BE are fish species, the residue-effects literature review has been limited to fish species. Raw data tables with the available whole body residue-effects literature for the evaluated chemicals are presented in Appendix C. Of the assessed chemicals, only exposure to arsenic and silver at the criteria level may have adverse effects on ESA listed aquatic species in action area. Individual chemical assessments are presented in the following paragraphs.

5.7.1 Arsenic

The marine chronic arsenic criterion of 36 µg/L multiplied by the bioconcentration factor from the criteria document of 44 L/kg yields a TSC of 1.6 mg/kg wb/ww. Two NOERs were found and compared to the TSC. The first is 0.14 mg/kg from a brook trout exposure that assessed physiological effects (Harper, Farag, Hogstrand, & MacConnell, 2009); the second study EPA reviewed provides a range of 0.07 to 0.20 mg/kg based on mortality in lake trout swim up fry (Fitzsimons, Huestis, & Williston, 1995). The available residue-effects data indicates exposure to arsenic at chronic criteria levels appears likely to result in bioaccumulation of arsenic to levels associated with toxicity to aquatic species.

5.7.2 Carbaryl

Only two residue-effect studies are available for carbaryl for comparison to the 0.0064 mg/kg TSC. One of the two studies (McKim and Schmieder 1991) determined a lethal body burden of 36 mg/kg to rainbow trout, an ESA listed species within the Swinomish Tribe action area, the other being a 1.5 mg/kg survival NOER to golden ide (*Leuciscus idus*). The limited available residue-effects data indicates

exposure to carbaryl at chronic criteria levels appears unlikely to result in bioaccumulation of carbaryl to levels associated with toxicity to aquatic species.

5.7.3 Chlorpyrifos

None of the 20 LOERs or the single available chlorpyrifos NOER occurred at a residue lower than the chlorpyrifos TSC of 0.0073 mg/kg. Although none of the available fish species LOERs were for any of the ESA listed fish within the Swinomish Tribe action area, the number of LOERs higher than the TSC is indicative that exposure to chlorpyrifos at chronic criteria levels appears unlikely to result in bioaccumulation of chlorpyrifos to levels associated with toxicity to aquatic species.

5.7.4 Chromium VI

None of the six Cr^{+6} fish NOERs or eight LOERs are lower than the Cr^{+6} TSC of 0.25 mg/kg. This indicates that hexavalent chromium is not likely to adversely affect the ESA listed fish species within the Swinomish Tribe action area from a multiple routes of exposure scenario. Among the chromium residue-effects data are three separate NOERs for Chinook salmon egg hatchability, larval and juvenile survival, and juvenile growth and behavior at residues between 0.31 – 0.53 (Patton et al. 2007), a Chinook salmon growth NOER of 1.28 mg/kg (Farag et al. 2006), a Chinook salmon survival LOER of 1.3 mg/kg (Farag et al. 2006), and two rainbow trout lethal body burdens of 2.8 mg/kg (Fromm and Stokes 1962) and 10.5 mg/kg (van der Putte et al. 1981). These data indicate that exposure to chromium (VI) at chronic criteria levels appears unlikely to result in bioaccumulation of chromium (VI) to levels associated with toxicity to aquatic species.

Chromium is an essential micronutrient in fish that is required for the proper functioning of carbohydrate metabolism and insulin function. Shiau and Shy (1998) determined residues associated with nutritional deficiency, nutritional sufficiency and toxicity in red tilapia (*Oreochromis niloticus* x *O. aureus*), a hybrid between Nile tilapia and blue tilapia. The whole body residues associated with nutritional deficiency, nutritional sufficiency and toxicity (expressed as reduced growth) were 0.62, 1.96 and 9.38 mg/kg, respectively. It is likely that the nutritionally essential form of chromium is trivalent chromium, not the hexavalent chromium evaluated in this BE. None of the residue-effect studies evaluated in this section determined chromium speciation in fish tissue. But if typical of other fish species nutritional requirement of chromium, the red tilapia data may be indicative of a range of chromium residues in fish in natural aquatic systems representative associated with essential concentrations of chromium.

5.7.5 Diazinon

All of the 13 diazinon residue-effect results (2 NOERs, 11 LOERs) occur at tissue residues higher than the diazinon TSC of 0.019 mg/kg. Although the data set contains no data for any of the ESA listed fish within the action area, the LOER range between 0.05 – 2435 mg/kg indicates exposure to diazinon at chronic criteria levels appears unlikely to result in bioaccumulation of diazinon to levels associated with toxicity to aquatic species.

5.7.6 Lead

Lead BCFs are inversely proportional to the concentration of lead in water. At a hardness of 50 mg/L, the regression between lead concentration and BCF for all species yields an estimated BCF of 200 L/kg at the

chronic lead criterion concentration of 1.2 µg/L (McGeer et al., 2003). Multiplying the chronic lead criterion of 1.2 µg/L at 50 mg/L hardness by a bioconcentration factor for lead at a hardness of 50 mg/L (McGeer et al., 2003) yields a TSC of 0.24 mg/kg wb-ww.

Of the 82 available no effect or adverse effect residues on North American resident species, 18 are no effect residues on survival, reproduction or growth. NOER tissue residues ranged between 0.18 mg/kg for survival, growth and swimming behavior for the topsmelt (*Atherinops affinis*), and 18,932 mg/kg for a 7-day survival NOER for a dragonfly (*Pachydiplax longipennis*). The only ESA listed surrogate species within the action area with several residue-effect studies is rainbow trout. Three survival NOERs for rainbow trout of 0.25, 0.68 and 2.4 mg/kg are available (Appendix C, Excel spreadsheet *Pb multiple routes of exposure 101620.xlsx*), with exposure durations between 10 – 60 days. The only available acute effects tissue residue value for rainbow trout is 10 mg/kg for a 48-hour exposure. These data are all greater than the TSC of 0.24 mg/kg and therefore indicate that exposure to lead at chronic criteria levels appears unlikely to result in bioaccumulation of lead to levels associated with toxicity to aquatic species.

5.7.7 Malathion

All of the malathion residue-effect studies (2 NOERs, 4 LOERs) are at concentrations higher than the malathion TSC of 0.0023 mg/kg. Of the four LOERs, one (McKim and Schmieder 1991) is a rainbow trout lethal body burden of 49 mg/kg. The available evidence indicates that exposure to malathion at chronic criteria levels appears unlikely to result in bioaccumulation of malathion to levels associated with toxicity to aquatic species.

5.7.8 Nickel

Nickel BCFs are inversely proportional to the nickel concentration in water. At a hardness of 100 mg/L, and a BCF of 47 L/kg (USEPA, 1986g) and nickel concentration at the chronic nickel criterion concentration of 28.9 µg/L (McGeer et al., 2003), a TSC of 0.39 mg/kg wb-ww is calculated.

The available nickel residue-effects data indicates that nearly all nickel residues associated with adverse effects to aquatic life are higher than the TSC of 0.39 mg/kg (Appendix C, Excel spreadsheet *Nickel_Multiple Routes of Exposure_112020.xlsx*); the only exception to this finding is a Fathead minnow (Pimephales promelas) NOER of 0.096 mg/kg. Of the 209 available no effect or adverse effect residues on North American resident species, 58 are no effect residues on survival, reproduction or growth. These NOERs range from 0.096 mg/kg for survival or growth of the Fathead minnow (*Pimephales promelas*) to 862 mg/kg for 20% mortality in 8 days, reduced swimming speed, and increased excretion in a nickel exposed Guppy (*Poecilia reticulata*). The only ESA listed surrogate species within the action area with residue-effect studies is rainbow trout. There are no nickel NOERs reported for rainbow trout, but there are two available acute effect tissue residue concentrations for rainbow trout: 0.845 mg/kg (lethal to 10% of fish in 96 hours) and 0.99 mg/kg (lethal to 50% of fish in 96 hours (Leonard, Banerjee, D'Silva, & Wood, 2014)). Both effects concentrations are higher than the TSC of 0.39 mg/kg. Therefore, exposure to nickel at chronic criteria levels appears unlikely to result in bioaccumulation of nickel to levels associated with toxicity to aquatic species.

5.7.9 Nonylphenol

None of the three available nonylphenol NOERs or seven LOERs occur at concentrations lower than the TSC of 0.23 mg/kg. Although none of the available residue-effect studies are with an ESA species present

in the Swinomish Tribe action area, there is one survival LOER of 6.9 mg/kg for a surrogate salmonid species, the Atlantic salmon (McLeese et al., 1981). Exposure to nonylphenol at chronic criteria levels appears unlikely to result in bioaccumulation of nonylphenol to levels associated with toxicity to aquatic species.

5.7.10 Pentachlorophenol

Pentachlorophenol has more residue-effects data available than does any other chemical evaluated in this BE. Three of the 11 NOERs were observed at residues lower than the pentachlorophenol TSC of 3.1 mg/kg. Two of the three NOERs below the TSC are for salmonids: a rainbow trout growth NOER of 0.06 mg/kg (Niimi and Cho 1983) and an Arctic charr (*Salvelinus alpinus*) survival and behavioral NOER of 0.786 mg/kg (Cravedi et al. 1995). Arctic charr is the same genus as the ESA listed bull trout (*Salvelinus confluentus*) within the Swinomish Tribe action area, and can be used as a surrogate species for bull trout. There is also a rainbow trout growth NOER of 29.8 mg/kg. All 18 LOERs are higher than the pentachlorophenol TSC of 3.1 mg/kg. The lowest of the 18 pentachlorophenol LOERs is a 3.83 mg/kg survival LOER resulting when rainbow trout were exposed to 200 µg/L pentachlorophenol for 7 – 8 hours (Tsuda and Kariya 1963). Four additional rainbow trout survival LOERs between 13.8 – 37.3 mg/kg are also available, as is a rainbow trout growth LOER of 72.3 mg/kg. The residue-effects data indicates that exposure to pentachlorophenol at chronic criteria levels appears unlikely to result in bioaccumulation of pentachlorophenol to levels associated with toxicity to aquatic species.

5.7.11 Zinc

Unless water column zinc concentrations are elevated above naturally occurring concentrations, most aquatic species can regulate their internal body burdens to the levels needed to meet their nutritional requirements. However, it can be bioaccumulated to some degree. Therefore, a TSC of 10.4 mg/kg wb-ww was calculated using the freshwater chronic criterion at 50 mg/L hardness of 65.7 µg/L multiplied by a bioconcentration factor of 158 L/kg .

Of the 113 studies where whole body zinc residues are associated with toxic effects on survival, reproduction, growth or behavior (Appendix C, Excel spreadsheet *Zinc multiple routes of exposure.xlsx*) , only one residue lower than the zinc benchmark was associated with toxicity. That occurred in the marine copepod *Acartia tonsa*, where a 46% reduction in egg production relative to controls was observed at 1.3 mg/kg whole body wet weight (wbww) (Hook & Fisher, 2002). Of the 10 zinc residues lower than the 10.4 mg/kg zinc benchmark, four are NOERs (no effect residues), four are associated with nutritional deficiency, and one is associated with nutritional sufficiency. All but one of the remaining residues associated with nutritional deficiency is 17.4 mg/kg or lower. Two studies (Ogino & Yang, 1978; Spry, Hodson, & Wood, 1988) note that whole body zinc residues between 6.1 and 8.1 mg/kg are nutritionally deficient for the rainbow trout. Nutritionally sufficient zinc levels in rainbow trout have been found to be between 18.8 mg/kg (Ogino & Yang, 1978) and 44 – 52 mg/kg (Kucukbay et al., 2006). The lowest whole body rainbow trout residue associated with toxicity is 101 mg/kg, which resulted in a reduction in percent viable hatch (Leland, 1983). Based on residues where adverse effects begin to be seen in aquatic species, a threshold residue for toxicity appears to be between 20 – 25 mg/kg. Therefore, exposure to zinc at chronic criteria levels appears unlikely to result in bioaccumulation of zinc to levels associated with toxicity to aquatic species.

5.8 Effects to Prey of Listed Species

Effects of exposure at criteria levels to prey of listed species were assessed as described in section 5.4.5 for the toxics criteria. These assessments apply to all listed species addressed in this BE. The findings presented in this section will be incorporated into the final effects determinations that are provided in the summary section for each criterion chemical in sections 5.5 and 5.6.

Effects to prey for temperature, dissolved oxygen, and pH criteria are assessed in sections 5.5.2 and 5.6.2.

5.8.1 Indirect effects analysis for acrolein

The analysis of indirect effects on listed species of the proposed acute and chronic freshwater criteria for acrolein is presented below. ECOTOX was searched in February of 2020 for data relevant to the indirect effects analysis approach. Results of the data analysis are presented below in Table 111 and Table 112. The findings presented in this section are summarized in section 5.8.20.

5.8.1.1 Freshwater data summaries

Table 111. Acute freshwater toxicity data (µg/L) for acrolein. The proposed criterion is 3.0 µg/L.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Insects	5	1386	510 – 5920	510	*Chironomus riparius*	(Horne & Oblad, 1983)
Molluscs	4	434.3	200 – 1000	200	*Biomphalaria glabrata*	(Hopf & Muller, 1962)
Fish	14	102.8	14.0 – 2500	14.0	*Pimephales promelas*	(R. Spehar, 1989)
Crustaceans	4	744.3	57.0 – 160,000	13.0	*Daphnia magna*	(Macek et al., 1976)
Amphibians	2	12.7	7.0 – 23.0	7.0	*Xenopus laevis*	(Gary W Holcombe, Gary L Phipps, Abdul H Sulaiman, & Alex D Hoffman, 1987)
Aquatic Invertebrates	No Data					

The acute toxicity of acrolein in freshwater was evaluated in five of six prey categories. See Table 110. The lowest PCLTV was for amphibians and was 7.0 µg/L. As this PCLTV was greater than the proposed criterion of 3 µg/L, exposure at the level of the acute freshwater criterion is **not likely to result in reductions** in the community of freshwater prey species.

Table 112. Chronic freshwater toxicity data (µg/L) for acrolein. The proposed criterion is 3.0 µg/L.

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Insects	1	1000	1000 – 1000	1000	*Chironomus sp.*	(Venturino, Montagna, & de D'Angelo, 2007)
Crustaceans	2	125.0	100 – 200	180	*Ceriodaphnia dubia*	(Union_Carbide_Corporation, 1997)
Fish	3	15.1	14.0 – 26.0	14.0	*Pimephales promelas*	(R. Spehar, 1989)
Aquatic Invertebrates	No Data					
Molluscs	No Data					
Insects	No Data					

The chronic toxicity of acrolein in freshwaters was evaluated in three of six prey categories. See Table 112. The PCLTVs for fish (14.0 µg/L), crustaceans (125.0 µg/L), and insects (1000 µg/L) were higher than the proposed chronic freshwater criterion of 3.0 µg/L. Therefore, exposure at the level of the chronic freshwater criterion is **not likely to result in reductions** in the community of freshwater prey species.

5.8.1.2 Freshwater acrolein criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

- Acute Freshwater Criterion Magnitude: *Prey reductions unlikely*
- Chronic Freshwater Criterion Magnitude: *Prey reductions unlikely*

5.8.2 Indirect effects analysis for ammonia

This section is reserved. The assessment of the Tribe's ammonia criteria for Chinook salmon, steelhead, chum salmon, bull trout, bocaccio, yelloweye rockfish, and marbled murrelet is in progress and will be provided as an addendum to the BE when the assessment has been completed.

5.8.3 Indirect effects analysis for arsenic

The analysis of indirect effects on listed species of the proposed acute freshwater criterion and the acute and chronic marine criteria for arsenic is presented below. ECOTOX was searched in May of 2022 for data relevant to the indirect effects analysis approach. Results of the data analysis are presented below in Table 113 and Table 114. The findings presented in this section will be used to evaluate the indirect effects on listed species in section 5.8.20.

5.8.3.1 Freshwater data summary

Table 113. Acute freshwater toxicity data (µg/L) for arsenic. The proposed criterion is 340 µg/L.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Amphibians	1	40.0	40.0	40.0	*Gastrophryne carolinensis*	(Birge, Black, & Westerman, 1979)
Fish	3	4703	170 – 1200	170.0	*Oncorhynchus mykiss*	(Birge et al., 1979)
Crustaceans	2	1833.0	293 – 3500	293.0	*Hyalella azteca*	(Uwe Borgmann, Couillard, Doyle, & Dixon, 2005)
Insects	No Data					
Molluscs	No Data					
Aquatic Invertebrates	No Data					

The acute toxicity of arsenic in freshwater was evaluated in three of six prey categories. See Table 113. The lowest PCLTV was for amphibians and was 40.0 µg/L. As this PCLTV was less than the proposed criterion of 340 µg/L, the acute freshwater criteria may not be protective of prey species relevant to listed species. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because 17% of species mean acute values were less than the criterion (<20% threshold, or 1 of 6 species), the acute freshwater criterion is **not likely to result in reductions** in the community of freshwater prey species.

5.8.3.2 Marine data summaries

The EPA (USEPA, 1985a) acute marine criterion for arsenic is based on results from 11 species: eight invertebrate species and three fish species. The three fish species were the three most tolerant species to arsenic. Because EPA (USEPA, 1985a) identified only one empirical chronic toxicity study of acceptable quality to directly derive a chronic criterion, the marine arsenic chronic criterion was derived using acute-chronic ratios with the 11 acceptable acute toxicity test results.

Several additional acute and chronic arsenic toxicity studies with both fish and invertebrates have been published since 1985 that are of acceptable data quality for use in criteria derivation, as well as the assessment of arsenic effects on marine prey species of the ESA listed marine species evaluated in this section. Acceptable quality empirical acute toxicity data published since 1985 has been found for zebrafish (*Danio rerio*, (Liu, Wang, & Theodorakis, 2006)), mummichog (*Fundulus heteroclitus*, J. R. Shaw et al. (2007)) and the copepod *Tigriopus japonicus* (Lee et al., 2008). Acceptable quality empirical chronic toxicity data published since 1985 has been found for zebrafish (Liu et al., 2006) and *Tigriopus japonicus* (Lee et al., 2008).

This section evaluates the potential for adverse effects on ESA listed species due to direct arsenic toxicity to their prey, followed by the loss of food items of listed species from the aquatic system. Results are presented in Table 114 for prey of aquatic and aquatic-dependent species and are expressed

as a range of acute toxicity and chronic toxicity values for various categories of prey species. Briefly, the toxicity values compiled in Table 114 were grouped into common categories (e.g., insects, molluscs, fish, etc.). Each respective acute toxicity LC_{50} value used to generate the range of acute effect assessment concentrations was divided by 2.27 to generate the acute no effect concentration range shown in Table 114. Chronic data in Table 114 are either the empirical long-term chronic NOEC or MATC data (*Americamysis bahia*, *Tigriopus japonica* and zebrafish), or the acceptable quality acute toxicity studies divided by the arsenic acute-chronic ratio of 3.803. The number of prey species with toxicity data within each category of prey species is also presented.

Table 114. Range of adverse arsenic acute and chronic effect assessment concentrations for categories of prey species of the ESA listed marine species within the action area (number of species in each category in parentheses).

Arsenic Effect Concentrations on Food Items of ESA Listed Species Acute Criterion = 69 µg/L, Chronic Criterion = 36 µg/L		
Organism Type	**Acute Effect Assessment Concentration (LC_{50}/2.27), µg/L**	**Chronic Effect Assessment Concentration, µg/L**
Fish	5595 – 119,824 (N =5)	3389 – 10,000 (N = 5)
Amphibians	No data	No data
Aquatic Invertebrates	102 – 4458 (N = 10)	61 – 2661 (N = 10)
Aquatic Insects	No data	No data
Crustaceans	102 - 4273 (N = 5)	61 – 2104 (N = 5)
Zooplankton	508 - 9700 (N = 3)	100 - 895 (N = 3)
Molluscs	144 – 3304 (N = 4)	86 – 1972 (N = 4)

The species with the lowest empirical acute toxicity value is zoea of the Dungeness crab (*Cancer magister*), with a 96 hour LC_{50} of 232 µg/L. The species with the lowest empirical chronic toxicity value is the copepod *Tigriopus japonica*, with a reproductive NOEC of 100 µg/L. None of either the acute or chronic effects assessment concentrations in Table 114 is lower than the corresponding acute or chronic marine arsenic criterion. This observation leads to a conclusion that exposure at the level of the acute and chronic marine criteria is **not likely to result in reductions** to the prey of ESA listed species in the action area.

5.8.3.3 Freshwater criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

- Acute Freshwater Criterion Magnitude: *Prey reductions unlikely*

5.8.3.4 Marine arsenic criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for

minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

- Acute Marine Criterion Magnitude: *Prey reductions unlikely*
- Chronic Marine Criterion Magnitude: *Prey reductions unlikely*

5.8.4 Indirect effects analysis for carbaryl

The analysis of indirect effects on listed species of the proposed acute and chronic freshwater and acute marine criteria for carbaryl is presented below. ECOTOX was searched in February of 2020 for data relevant to the indirect effects analysis approach. Results of the data analysis are presented below in Table 115 and Table 116. The findings presented in this section are summarized in section 5.8.20.

5.8.4.1 Freshwater data summaries

Table 115. Acute freshwater toxicity data (µg/L) for carbaryl. The proposed criterion is 2.1 µg/L.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Aquatic Invertebrates	6	12,320	4493 – 46,256	4493	*Paramecium caudatum*	(Lejczak, 1977)
Amphibians	19	5184	258 – 220,264	258	*Rana boylii*	(Kerby, 2006)
Molluscs	22	5959	194 – 25,520	194	*Lymnaea acuminata*	(Singh & Agarwal, 1981)
Fish	74	2705	61.7 – 127,753	61.7	*Ictalurus punctatus*	(K. Brown, Anderson, Jones, Deuel, & Price, 1979)
Insects	42	63.1	0.75 – 1453	0.75	*Pteronarcella badia*	(F. L. Mayer & M. R. Ellersieck, 1986a)
Crustaceans	39	50.3	0.40 – 401,322	0.40	*Daphnia pulex*	(Nishiuchi & Hashimoto, 1967)

The acute toxicity of carbaryl in freshwater was evaluated in all six prey categories. See Table 115. The PCLTV for crustaceans and insects were 0.40 and 0.75 µg/L, respectively. As these PCLTVs were both less than the proposed criterion of 2.1 µg/L, the criterion may not be protective of prey species relevant to listed species. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because 0.98% of SMAVs were less than the criterion (<20% threshold, or 2 of 204 species), exposure at the level of the acute freshwater criterion is **not likely to result in reductions** in the community of freshwater prey species.

Table 116. Chronic freshwater toxicity data (µg/L) for carbaryl. The proposed criterion is 2.1 µg/L.

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Fish	9	1570	400 – 11,000	400	*Pimephales promelas*	(Norberg-King, 1989b)
Molluscs	7	514.6	33.5 – 9300	33.5	*Planorbella trivolvis*	(Rohr, Raffel, Sessions, & Hudson, 2008)
Insects	5	37.4	5.0 – 7,800	5.0	*Coenagrion puella*	(Campero, Ollevier, & Stoks, 2007)
Amphibians	16	2233	5.0 – 20,122	5.0	*Ambystoma barbouri*	(Rohr et al., 2003)
Aquatic Invertebrates	5	196	1.0 – 10,000	1.0	*Echinoparyphium sp.*	(Hua, Buss, Kim, Orlofske, & Hoverman, 2016)
Crustaceans	10	16.1	0.20 – 5000	0.20	*Daphnia magna*	(Toumi, Burga-Perez, & Ferard, 2016)

The chronic toxicity of carbaryl in freshwaters was evaluated in all prey categories. See Table 116. The PCLTVs for crustaceans (0.20 µg/L) and aquatic invertebrates (1.0 µg/L) were lower than the proposed chronic freshwater criterion of 2.1 µg/L. Therefore, the proposed criterion may not be protective for prey of listed species. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because 1.9% of SMCVs were less than the criterion (<20% threshold, or 1 of 52 species), exposure at the level of the chronic freshwater criterion is **not likely to result in reductions** in the community of freshwater prey species.

5.8.4.2 Marine data summaries

Table 117. Acute marine toxicity data (µg/L) for carbaryl. The proposed criterion is 1.6 µg/L.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Molluscs	1	16,727	1696 – 819,383	1696	*Clinocardium nuttallii*	(Butler, Millemann, & Stewart, 1968)
Fish	8	2010	888.5 – 7324	888.5	*Cyprinodon bovinus*	(USEPA, 2008b)
Crustaceans	11	26.3	0.88 – 12,144	0.881	*Cancer magister*	(Buchanan, Millemann, & Stewart, 1970)
Aquatic Invertebrates	**No Data**					
Insects	**No Data**					
Amphibians	**No Data**					

The acute toxicity of carbaryl in marine water was evaluated in three prey categories. See Table 117. The PCLTV for crustaceans was 0.881 µg/L. As this PCLTV was less than the proposed criterion of 1.6 µg/L, the criterion may not be protective of prey species relevant to listed species. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because 0% of SMAVs were less than the criterion (<20% threshold, or 0 of 20 species), exposure at the level of the acute marine criterion is **not likely to result in reductions** in the community of marine prey species.

5.8.4.3 Freshwater criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

- Acute Freshwater Criterion Magnitude: *Prey reductions unlikely*
- Chronic Freshwater Criterion Magnitude: *Prey reductions unlikely*

5.8.4.4 Marine criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Acute Marine Criterion Magnitude: *Prey reductions unlikely*

5.8.5 Indirect effects analysis for chlorine

The list of chemicals and Chemical Abstracts Service identification numbers (CAS IDs) used in our literature search for toxic effects of chlorine on aquatic life is provided in Table 86 (section 5.5.1.4.2).

5.8.5.1 Effects of chlorine on prey of ESA listed freshwater species

This section evaluates the potential for adverse effects on ESA listed species due to direct chlorine toxicity to their prey, followed by the loss of food items of listed species from the aquatic system. Results are presented in Table 118 for prey of aquatic and aquatic-dependent species and are expressed as a range of acute toxicity and chronic toxicity values for various categories of prey species. Briefly, the toxicity values for freshwater organisms compiled in Table 118 were grouped into common categories (e.g., insects, molluscs, fish, etc.). Each respective acute toxicity LC_{50} value used to generate the range of acute effect assessment concentrations was divided by 2.27 to generate the acute no effect concentration range shown in Table 118.

Table 118. Toxicity of Chlorine to Food Items of ESA Listed Fish Species

Assessment Exposure Concentrations (C_A): Acute = 19 µg/L, Chronic = 11 µg/L		
Organism Type	**Acute EC_A Range (µg/L)**	**Chronic EC_A Range (µg/L)**
Fish	20 – 313	13 - 212
Amphibians	No data	No data
All aquatic invertebrates	5.1 - 1410	3.5 - 957
Aquatic insects	5.1 - 1410	3.5 - 957
Crustaceans	5.9 – 297	4.0 – 201
Zooplankton	12 – 34	8.3 – 23
Molluscs	31 – 105	21 – 71

No fish species had acute or chronic EC_A values lower than the respective acute or chronic water quality criteria. This finding supports a conclusion that the Tribe's chlorine criteria should not have any adverse effect on prey of adult bull trout, which normally feed on fish.

Juvenile and subadult salmonids feed on a variety of invertebrate species before switching over to the primarily fish diet of adults. The favored prey appears to be mayflies and dipteran larvae. Table 118 indicates that both the lowest calculated acute and chronic EC_A values are lower than the respective acute and chronic chlorine criteria for aquatic insects, crustaceans and zooplankton. Among aquatic insects, data for two of the six available insect species, both of which are mayflies, yielded both acute and chronic assessment effects concentrations lower than the respective acute and chronic criteria. A third mayfly species had an acute and chronic assessment effects concentration higher than the acute and chronic criteria, as did a caddisfly and two beetle species. Of the remaining 12 invertebrate species with available data (three zooplankton species, three molluscs and six non-zooplankton crustaceans), only one zooplankton species (*Daphnia magna*, the single most sensitive species to chlorine) and one crustacean (*Gammarus minus*) had calculated acute and chronic EC_A concentrations lower than the respective acute and chronic water quality criteria.

Most salmonids tend to be opportunistic feeders. Numerous alternative prey species exist for the ESA listed salmonids with acute and chronic assessment effects concentrations above the chlorine criteria. This would minimize the potential for adverse effects on them from chlorine toxicity to their prey. In addition, any areas with elevated chlorine concentrations are expected to be small because of the transient nature of chlorine, and fish will be able to move to adjacent areas with lower chlorine concentrations and greater food abundance. Therefore, exposure at the levels of the acute and chronic freshwater chlorine criteria is **not likely to result in in reductions** in the available prey for ESA listed salmonids within the action area.

5.8.5.2 Effects of chlorine on prey of ESA listed marine species

This section evaluates the potential for adverse effects on ESA listed species due to direct chlorine toxicity to their prey, followed by the loss of food items of listed species from the aquatic system. Results are presented in Table 119 for prey of aquatic and aquatic-dependent species and are expressed as a range of acute toxicity and chronic toxicity values for various categories of prey species. Briefly, the toxicity values for organisms compiled in Table 119 were grouped into common categories (e.g., insects, molluscs, fish, etc.). Each respective acute toxicity LC_{50} value used to generate the range of acute effect

assessment concentrations was divided by 2.27 to generate the acute no effect concentration range shown in Table 119. Chronic data in Table 119 are either the acceptable quality acute toxicity studies divided by the chlorine acute-chronic ratio of 3.345. The number of prey species with toxicity data within each category of prey species is also presented.

Table 119. Range of adverse chlorine acute and chronic effect assessment concentrations for categories of prey species of the ESA listed fish species within the action area (number of species in each category in parentheses).

Chlorine Effect Concentrations on Food Items of ESA Listed Fish Species Acute Criterion = 13 µg/L, Chronic Criterion = 7.5 µg/L		
Organism Type	**Acute Effect Assessment Concentration (LC_{50}/2.27), 5.7 µg/L**	**Chronic Effect Assessment Concentration, 7.5 µg/L**
Fish	14.1 – 118.9 (N =12)	9.6-80.7 (N = 15)
Amphibians	No data	No data
Aquatic Invertebrates (Worms)	No data	No data
Crustaceans	27.3-624.7 (N =11)	18.5-423.9 (N = 11)
Zooplankton	12.8 (N=1)	8.7 (N = 1)
Mollusks	11.5 (N = 1)	7.8 (N=1)

The species with the lowest empirical acute toxicity value is Eastern oyster larvae, with a 96 hour LC_{50} of 26 µg/L. No prey categories had effects concentrations below the acute or chronic criteria. This observation leads to a conclusion that exposure at the level of the acute and chronic marine criteria is **not likely to result in reductions** to the prey of ESA listed species in the action area.

5.8.5.3 Freshwater criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

- Acute Freshwater Criterion Magnitude: *Prey reductions unlikely*
- Chronic Freshwater Criterion Magnitude: *Prey reductions unlikely*

5.8.5.4 Marine criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

- Acute Marine Criterion Magnitude: *Prey reductions unlikely*
- Chronic Marine Criterion Magnitude: *Prey reductions unlikely*

5.8.6 Indirect effects analysis for chlorpyrifos

The analysis of indirect effects on listed species of the proposed acute and chronic freshwater and marine criteria for chlorpyrifos is presented below. The data used in EPA's Biological Evaluation of chlorpyrifos[28] were used for this assessment. Results of the queries are presented below in Table 120 through Table 123. Using the best available commercial and scientific data, the analysis was conducted for the effect of each WQS magnitude on surrogate prey items potentially used by listed species. The findings presented in this section are summarized in section 5.8.20.

5.8.6.1 Freshwater toxicity data summary

Table 120. Acute freshwater toxicity data (µg/L) for a range of listed species prey items. The proposed chlorpyrifos freshwater chronic criterion is 0.083 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine acute values (PCMAV) are geometric means of all species acute toxicity values.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Aquatic Invertebrates	1	12000	12000	12000	*Brachionus calyciflorus*	(Snell & Moffat, 1992)
Molluscs	4	792.9	431 – 1322	431	*Pomacea canaliculata*	(Tejada et al., 1994)
Amphibians	13	153.1	4.41 – 10162	4.41	*Hoplobatrachus tigerinus*	(Abbasi & Soni, 1991)
Fish	26	67.5	0.44 – 4145	0.44	*Oncorhynchus mykiss*	(F. Mayer & M. Ellersieck, 1986)
Crustaceans	25	1.02	0.01 - 21366	0.011	*Hyalella azteca*	(Ding, Landrum, You, Harwood, & Lydy, 2012)
Aquatic Insects	79	3.49	0.004 – 9559	0.004	*Culex quinquefasciatus*	(Aguilera, Marquetti, Navarro, & Bisset, 1995)

The acute toxicity of chlorpyrifos in freshwater was evaluated in all prey categories. See Table 120. Two of six PCLTVs were lower than the proposed criterion. The PCLTV for aquatic insects was 0.004 µg/L in *Culex quinquefasciatus* and 0.011 µg/L in *Hyalella azteca*. As these were lower than the proposed criterion of 0.083 µg/L, the criterion may not be protective of prey species relevant to listed species. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because only 7.5% of toxicity values were less than the criterion (<20% threshold, or 11 of 146 species), exposure at the level of the acute freshwater criterion is **not likely to result in reductions** in the community of prey species.

[28] USEPA. 2017. Biological Evaluation for Chlorpyrifos ESA Assessment. Webpage: https://www.epa.gov/endangered-species/biological-evaluation-chapters-chlorpyrifos-esa-assessment, last updated on September 13, 2021.

Table 121. Chronic fresh water toxicity data (µg/L) for a range of listed species prey items. The proposed chlorpyrifos freshwater chronic criterion is 0.041 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine chronic values (PCMCV) are geometric means of all species acute toxicity values.

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Molluscs	11	56.0	22 – 720	22.5	*Lampsilis siliquoidea*	(Bringolf et al., 2007)
Amphibians	11	129	4.74 – 992	4.74	*Lithobates sphenocephalus*	(Widder & Bidwell, 2006)
Fish	23	10.8	0.12 - 992	0.12	*Pimephales promelas*	(Jarvinen, 1982)
Aquatic Insects	34	2.39	0.01 - 250	0.01	*Chironomus riparius*	(Agra & Soares, 2009)
Aquatic Invertebrates	32	2.81	0.01 - 400	0.01	*Polyarthra remata*	(van Wijngaarden, Brock, & Douglas, 2005)
Crustaceans	43	0.95	0.005 – 44	0.005	*Daphnia carinata*	(Zalizniak & Nugegoda, 2006)

The chronic toxicity of chlorpyrifos in freshwater was evaluated in all prey categories. See Table 121. Three prey categories had PCLTVs that were lower than the proposed chronic criterion. These were crustaceans with a PCLTV of 0.005 µg/L (LOEC for reduced number of progeny) for *Daphnia carinata*, the lowest value of the 43 tested species; aquatic invertebrates with a PCLTV of 0.01 µg/L (LOEC for reduced abundance) for *Polyarthra remata*, the lowest value of the 32 tested species; and aquatic insects with a PCLTV of 0.01 µg/L (LOEL for emergence, a developmental effect) for *Chironomus riparius*, the lowest value of 34 tested species. As these were lower than the proposed criterion of 0.041 µg/L, the criterion may not be protective of prey species relevant to listed species. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because only 2.6% of toxicity values were less than the criterion (<20% threshold, or 4 of 153 studies), exposure at the level of the chronic freshwater criterion is **not likely to result in reductions** in the community of prey species.

5.8.6.2 Marine toxicity data summary

Table 122. Acute marine toxicity data (µg/L) for a range of listed species prey items. The proposed chlorpyrifos freshwater chronic criterion is 0.010 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine acute values (PCMAV) are geometric means of all species acute toxicity values.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Fish	14	5.22	0.16 – 440.5	0.16	*Menidia peninsulae*	(P. W. Borthwick, J. M. Patrick, & D. P. Middaugh, 1985)
Molluscs	No Data					

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Amphibians	No Data					
Crustaceans	No Data					
Aquatic Insects	No Data					
Aquatic Invertebrates	No Data					

The acute toxicity of chlorpyrifos in marine water was evaluate in fish only. See Table 122. The PCLTV for fish was 0.16 µg/L for *Menidia peninsulae*. Because the PCLTV is greater than the proposed acute marine criterion of 0.010 µg/L, exposure at the level of the acute marine criterion is **not likely to result in reductions** in the community of prey species.

Table 123. Chronic marine toxicity data (µg/L) for a range of listed species prey items. The proposed chlorpyrifos freshwater chronic criterion is 0.006 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine chronic values (PCMCV) are geometric means of all species acute toxicity values.

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Molluscs	1	3.51	3.51	3.51	*Paracentrotus lividus*	(Aluigi, Falugi, Mugno, Privitera, & Chiantore, 2010)
Fish	4	0.82	0.28 – 17.1	0.28	*Menidia menidia*	(Goodman, 1985)
Crustaceans	5	0.35	0.002 – 5.90	0.002	*Americamysis bahia*	(McKenney, Matthews, & Lawrence, 1981; S. Schimmel, 1981)
Aquatic Invertebrates	No Data					
Aquatic Insects	No Data					
Amphibians	No Data					

The chronic toxicity of chlorpyrifos in marine waters was evaluated in molluscs, fish, and crustaceans (Table 123). The PCLTV for molluscs was 3.51 µg/L (NOEL for larval growth) for *Paracentrotus lividus*, the only mollusk species with relevant test data. The fish PCLTV was 0.28 µg/L (the NOEC for reduced survival) for *Menidia menidia*, the lowest value of the four tested fish species. The crustacean PCLTV was 0.002 µg/L (NOEC for reduced growth), the lowest of five tested crustacean species. The chronic marine criterion for chlorpyrifos is 0.006 µg/L. Because 0.006 µg/L is less than 2 of 3 PCLTVs, the proposed chronic marine criteria may not be protective for chronic effects to marine prey items. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because only 10% of toxicity values were less than the criterion (<20% threshold, or 1 of 10 tested species), exposure at the level of the chronic marine criterion is **not likely to result in reductions** in the community of prey species.

5.8.6.3 Freshwater criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each Listed Species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Acute Freshwater Criterion Magnitude: *Prey reductions unlikely*

Chronic Freshwater Criterion Magnitude: *Prey reductions unlikely*

5.8.6.4 Marine criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each Listed Species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Acute Marine Criterion Magnitude: *Prey reductions unlikely*

Chronic Marine Criterion Magnitude: *Prey reductions unlikely*

5.8.7 Indirect effects analysis for chromium VI

The analysis of indirect effects on listed species of the proposed acute and chronic freshwater and marine criteria for hexavalent chromium (CrVI) is presented below. ECOTOX was searched in February of 2020 for data relevant to the indirect effects analysis approach. Furthermore, an open literature search using Web of Science was conducted in March of 2020 to augment the ECOTOX database, as only acute data for freshwater taxa were available in ECOTOX. Results of the data analysis are presented below in Table 124 through Table 127. The findings presented in this section are summarized in section 5.8.20.

5.8.7.1 Freshwater data summaries

Table 124. Acute freshwater toxicity data (µg/L) for CrVI. The proposed criterion is 16 µg/L.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Molluscs		425	207 – 872	207	*Lymnaea stagnalis*	(Ivey et al., 2017)
Crustaceans	7	52.1	22.0 – 264	22.0	*Simocephalus mixtus*	
Amphibians	1	13.0	13.0 – 13.0	13.0	*Hypsiboas pulchellus*	(Natale, Ammassari, Basso, & Ronco, 2006)
Fish	4	2429	2.6 – 64,185	2.6	*Pimephales promelas*	(Anderson & Norberg-King, 1991)
Insects	No Data					
Aquatic Invertebrates	No Data					

The acute toxicity of CrVI in freshwater was evaluated in all four of six prey categories. See Table 124. The PCLTV for fish and amphibians 2.6 and 13.0 µg/L, respectively. As these PCLTVs were both less than the proposed criterion of 16 µg/L, the criterion may not be protective of prey species relevant to listed species. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because 6.7% of SMAVs were less than the criterion (<20% threshold, or 1 of 15 species), exposure at the level of the acute freshwater criterion is **not likely to result in reductions** in the community of freshwater prey species.

Table 125. Chronic freshwater toxicity data (µg/L) for CrVI. The proposed criterion is 11 µg/L.

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Amphibians	3	153.3	9.0 – 2000	9.0	*Hypsiboas pulchellus*	(Natale et al., 2006)
Crustaceans	1	3.5	3.5 – 3.5	3.5	*Daphnia magna*	(Gorbi, Corradi, Invidia, Rivara, & Bassi, 2002)
Aquatic Invertebrates	No Data					
Fish	No Data					
Molluscs	No Data					
Insects	No Data					

The chronic toxicity of CrVI in freshwaters was evaluated in two of six prey categories. See Table 125. The PCLTVs for crustaceans (3.5 µg/L) and amphibians (9.0 µg/L) were lower than the proposed chronic freshwater criterion of 11 µg/L. Therefore, the proposed criterion may not be protective for prey of listed species. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because 50% of SMCVs were less than the criterion (>20% threshold, or 2 of 4 species), exposure at the level of the chronic freshwater criterion is **likely to result in reductions** in the community of freshwater prey species.

5.8.7.2 Marine data summaries

Table 126. Acute marine toxicity data (µg/L) for CrVI. The proposed criterion is 1100 µg/L.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Fish	2	49.2	30.5 – 79.4	30.5	*Pomatoschistus microps*	(Luís, Ferreira, Fonte, Oliveira, & Guilhermino, 2015)
Crustaceans	2	19.8	16.7 – 23.5	16.7	*Artemia franciscana*	(Leis et al., 2014)
Aquatic Invertebrates	No Data					

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Insects	No Data					
Amphibians	No Data					
Molluscs	No Data					

The acute toxicity of CrVI in marine water was evaluated in two of six prey categories. See Table 126. The PCLTV for crustaceans and fish was 16.7 and 30.5 µg/L, respectively. As these PCLTVs were both less than the proposed criterion of 1100 µg/L, the criterion may not be protective of prey species relevant to listed species. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because 100% of SMAVs were less than the criterion (>20% threshold, or 4 of 4 species), exposure at the level of the acute marine criterion is **likely to result in reductions** in the community of marine prey species.

Table 127. Chronic marine (freshwater) toxicity data (µg/L) for CrVI. The proposed criterion is 50 µg/L. Because there were not marine chronic data available, chronic freshwater were used.

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Amphibians	3	153.3	9.0 – 2000	9.0	*Hypsiboas pulchellus*	(Natale et al., 2006)
Crustaceans	1	3.5	3.5 – 3.5	3.5	*Daphnia magna*	(Gorbi et al., 2002)
Aquatic Invertebrates	No Data					
Fish	No Data					
Molluscs	No Data					
Insects	No Data					

The chronic toxicity of CrVI in "marine" (only freshwater data were available, see Table 127) waters was evaluated in two of six prey categories. See Table 127. The PCLTVs for crustaceans (3.5 µg/L) and amphibians (9.0 µg/L) were lower than the proposed chronic marine criterion of 50 µg/L. Therefore, the proposed criterion may not be protective for prey of listed species. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because 50% of SMCVs were less than the criterion (>20% threshold, or 2 of 4 species), exposure at the level of the chronic marine criterion is **likely to result in reductions** in the community of marine prey species.

5.8.7.3 Freshwater chromium VI criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for

minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

- Acute Freshwater Criterion Magnitude: *Prey reductions unlikely*
- Chronic Freshwater Criterion Magnitude: *Prey reductions likely*

5.8.7.4 Marine chromium VI criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

- Acute Marine Criterion Magnitude: *Prey reductions likely*
- Chronic Marine Criterion Magnitude: *Prey reductions likely*

5.8.8 Indirect effects analysis for cyanide

The analysis of indirect effects on listed species of the proposed acute and chronic saltwater criteria for cyanide is presented below. A search was conducted on the ECOTOX data base in May 2022 for relevant toxicity data and the outcome of the search are presented below. The findings presented in this section will be used to evaluate the indirect effects of cyanide on listed species in section 5.8.20.

Table 128. Acute marine toxicity data (µg/L) for cyanide. The proposed acute criterion is 1 µg/L.

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Crustaceans	1	113.0	113.0	113.0	*Americamysis bahia*	(S. Lussier et al., 1985)
Amphibians	No Data					
Aquatic Invertebrates	No Data					
Mollusks	No Data					
Aquatic Insects	No Data					
Fish	No Data					

The acute toxicity of cyanide in marine waters was evaluated in crustaceans only (1 of 6 groups) because there were no data for other prey species. See Table 128. Acute marine toxicity data (µg/L) for cyanide. The proposed acute criterion is 1 µg/L.. The PCLTV for crustaceans was 113.0 µg/L for mortality to *Americamysis bahia*. The lowest PCLTV of 113.0 µg/L for this species was higher than the acute freshwater criteria for cyanide of 1 µg/L. Thus, exposure at the level of the acute marine criterion is **not likely to result in reductions** in prey items.

Table 129. Chronic marine toxicity data (µg/L) for cyanide. The proposed acute criterion is 1 µg/L.

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Crustaceans	1	50.6	43.0 – 70.0	43.0	*Americamysis bahia*	(S. Lussier et al., 1985)
Fish	No Data					
Aquatic Invertebrates	No Data					
Mollusks	No Data					
Aquatic Insects	No Data					
Amphibians	No Data					

The chronic toxicity of cyanide in marine waters was evaluated in crustaceans only. See Table 129. The PCLTV for this group was 43.0 µg/L for reproduction of *Americamysis bahia*. As the lowest PCLTV of 43.0 µg/L is higher than the criterion of 1 µg/L, exposure at the level of the chronic marine criterion is **not likely to result in reductions** in prey items.

5.8.8.1 Marine criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each Listed Species subject to this consultation. A conclusion of protective for prey species provides evidence for minimal effects to listed species whereas a conclusion of unprotective suggests effects to listed species through prey reductions are foreseeable.

- Acute Marine Criterion Magnitude: *Prey reductions unlikely*
- Chronic Marine Criterion Magnitude: *Prey reductions unlikely*

5.8.9 Indirect effects analysis for diazinon

The analysis of indirect effects on listed species of the proposed acute and chronic freshwater and marine criteria for diazinon is presented below. The data used in EPA's Biological Evaluation of diazinon[29] were used for this assessment. Results of the queries are presented below in Table 130 through Table 133. Using the best available commercial and scientific data, the analysis was conducted for the effect of each WQS magnitude on surrogate prey items potentially used by listed species. The findings presented in this section are summarized in section 5.8.20.

[29] https://www.epa.gov/endangered-species/biological-evaluation-chapters-diazinon-esa-assessment

5.8.9.1 Freshwater toxicity data summary

Table 130. Acute freshwater toxicity data (µg/L) for a range of listed species prey items. The proposed diazinon freshwater chronic criterion is 0.17 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine acute values (PCMAV) are geometric means of all species acute toxicity values.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Aquatic Invertebrates	1	11842	11842	11842	*Brachionus calyciflorus*	(Fernandez-Casalderry, Ferrando, & Andreu-Moliner, 1992)
Molluscs	1	6238	6238	6238	*Biomphalaria alexandrina*	(Mohamed, El-Emam, Osman, Abdel-Hamid, & Ali, 2012)
Fish	32	2246	3.5 – 240088	3.5	*Cyprinus carpio*	(Kaur & Toor, 1977)
Amphibians	4	561	1.2 – 5568	1.2	*Lithobates clamitans*	(Harris, Bishop, Struger, Ripley, & Bogart, 1998)
Aquatic Insects	11	8.81	0.44 – 2952	0.44	*Hydropsyche angustipennis*	(Van der Geest et al., 1999)
Crustaceans	7	1.41	0.07 – 20.5	0.07	*Ceriodaphnia dubia*	(Deanovic, Markiewicz, Stillway, Fong, & Werner, 2013)

The acute toxicity of diazinon in freshwater was evaluated in all prey categories. See Table 130. One of six PCLTVs were lower than the proposed criterion. The PCLTV for crustaceans was 0.07 µg/L in *Ceriodaphnia dubia*. As this value is lower than the proposed criterion of 0.17 µg/L, the criterion may not be protective of prey species relevant to listed species. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because only 3.6% of toxicity values were less than the criterion (<20% threshold, or 2 of 56 species values), exposure at the level of the acute freshwater criterion is **not likely to result in reductions** in the community of prey species.

Table 131. Chronic fresh water toxicity data (µg/L) for a range of listed species prey items. The proposed diazinon freshwater chronic criterion is 0.17 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine chronic values (PCMCV) are geometric means of all species acute toxicity values.

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Aquatic Invertebrates	1	4784	4784	4784	*Brachionus calyciflorus*	(Fernandez-Casalderrey, Ferrando, & Andreu-Moliner, 1992)
Amphibians	4	20.2	2.1 – 7500	2.1	*Liobates pipiens*	(Relyea, 2009)
Fish	3	499	10 – 2419	10	*Tinca tinca*	(Máchová, Prokeš, Peňáz, Baruš, & Kroupová, 2010)

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Aquatic Insects	2	8.95	1.0 – 35.2	1.0	*Hydropsyche angustipennis*	(Van der Geest et al., 1999)
Crustaceans	9	1.52	0.10 – 2211	0.10	*Daphnia sp.*	(Buser, Spaak, & Wolinska, 2012)
Molluscs	No Data					

The chronic toxicity of diazinon in freshwater was evaluated in all prey categories except molluscs. See Table 131. One prey category had a PCLTV that was lower than the proposed chronic criterion. This was crustaceans with a PCLTV of 0.10 µg/L (EC50 for reduced reproductivity) for *Daphnia sp.*, the lowest value of the 9 tested species. As this was lower than the proposed criterion of 0.17 µg/L, the criterion may not be protective of prey species relevant to listed species. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because only 15.8% of toxicity values were less than the criterion (<20% threshold, or 3 of 19 species values), exposure at the level of the chronic freshwater criterion is **not likely to result in reductions** in the community of prey species.

5.8.9.2 Marine toxicity data summary

Table 132. Acute marine toxicity data (µg/L) for a range of listed species prey items. The proposed diazinon freshwater chronic criterion is 0.82 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine acute values (PCMAV) are geometric means of all species acute toxicity values.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Aquatic Invertebrates	1	11852	11852	11852	*Brachionus plicatilis*	(Marcial, Hagiwara, & Snell, 2005)
Molluscs	1	1013	1013	1013	*Haliotis varia*	(Kaligis & Lasut, 1997)
Crustaceans	5	38.5	1.18 – 290749	1.18	*Palaemonetes pugio*	(G. Thursby & Berry, 1988)
Fish	3	861	0.81 – 3524	0.81	*Psetta maxima*	(Mhadhbi & Boumaiza, 2012a)
Amphibians	No Data					
Aquatic Insects	No Data					

The acute toxicity of diazinon in marine water was evaluate in fish, crustaceans, molluscs, and other aquatic invertebrates. See Table 132. PCLTVs for all but one prey categories were greater than the proposed acute marine criterion of 0.82 µg/L. The PCLTV for fish was 0.81 µg/L and so the proposed marine acute criterion may not be protective for prey of listed species. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because only 10% of toxicity values were less than the criterion (<20% threshold, or 1 of 10 tested species), exposure at the level of the acute marine criterion is **not likely to result in reductions** in the community of prey species.

Table 133. Chronic marine toxicity data (µg/L) for a range of listed species prey items. The proposed diazinon freshwater chronic criterion is 0.82 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine chronic values (PCMCV) are geometric means of all species acute toxicity values.

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Aquatic Invertebrates	1	9.13	9.13	9.13	*Brachionus plicatilis*	(Marcial et al., 2005)
Crustaceans	3	8.48	2.1 – 25.7	2.1	*Americamysis bahia*	(W. Berry, 1989)
Fish	5	1079	0.47 – 8000	0.47	*Cyprinodon variegatus*	(Larry R Goodman et al., 1979)
Amphibians	No Data					
Molluscs	No Data					
Aquatic Insects	No Data					

The chronic toxicity of diazinon in marine waters was evaluated in fish, crustaceans, and other aquatic invertebrates (Table 133). The PCLTV for fish was 0.47 µg/L (LOAEL for fecundity) in *Cyprinodon variegatus*, one of five species of fish for which relevant toxicity data were available. All other PCLTVs were greater than the proposed chronic marine criterion. Because 0.47 µg/L is less than the proposed chronic marine criterion of 0.82 µg/L, it may not be protective for chronic effects to marine prey items. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because only 9% of toxicity values were less than the criterion (<20% threshold, or 1 of 11 tested species), exposure at the level of the chronic marine criterion is **not likely to result in reductions** in the community of prey species.

5.8.9.3 Freshwater criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Acute Freshwater Criterion Magnitude: *Prey reductions unlikely*

Chronic Freshwater Criterion Magnitude: *Prey reductions unlikely*

5.8.9.4 Marine criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Acute Marine Criterion Magnitude: *Prey reductions unlikely*

Chronic Marine Criterion Magnitude: *Prey reductions unlikely*

5.8.10 Indirect effects analysis for hydrogen sulfide

The analysis of indirect effects on listed species of the proposed chronic freshwater and marine criteria for hydrogen sulfide is presented below. Results of the ECOTOX data base queries are presented below in Table 134 and Table 135. Using the best available commercial and scientific data, the analysis was conducted for the effect of each WQS magnitude on surrogate prey items potentially used by listed species. The findings presented in this section are summarized in section 5.8.20.

5.8.10.1 Freshwater chronic toxicity data summary

Table 134. Freshwater chronic toxicity data (µg/L) for a range of listed species prey items are summarized below. The proposed hydrogen sulfide freshwater chronic criterion is 2 µg/L .Prey categories are sorted from highest PCLTV to lowest. Prey category mean acute values (PCMAV) are geometric means of all species chronic toxicity values.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Crustaceans	1	1.33	1.33	11	*Gammarus pseudolimnaeus*	(Daniel J Call et al., 1999)
Fish	3	3.34	1.33 – 3.34	6	*Oncorhynchus mykiss*	(Daniel J Call et al., 1999)
Amphibians	No Data					
Invertebrates	No Data					
Molluscs	No Data					

The chronic toxicity of dissolved hydrogen sulfide in freshwater was evaluated in fish and crustaceans. See Table 134. PCLTVs for two prey categories (fish and crustaceans) of 6 and 11 µg/L, respectively, were greater than the proposed freshwater chronic criterion of 2 µg/L. Thus, all PCLTVs were greater than the proposed acute criterion of 2 µg/L. Therefore, exposure at the level of the chronic freshwater criterion is **not likely to result in reductions** in the community of freshwater prey species.

5.8.10.2 Marine chronic toxicity data summary

Table 135. Marine toxicity data (µg/L) for a range of listed species prey items. The proposed chronic criterion is 2 µg/L. Prey category mean acute values (PCMAV) are geometric means of all species acute toxicity values.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Molluscs	2	269.2	275.9–437.9	275.9	*Crassostrea gigas*	(Caldwell, 1975)
Fish	1	273.49	273.49	273.49	*Oncorhynchus tshawytscha*	(Holland et al., 1960)

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Crustaceans	2	300.6	270.8 – 333.7	270.8	*Rhepoxynius abronius*	(Berry, Cantwell, Edwards, Serbst, & Hansen, 1999)
Invertebrates	1	5.87	5.87	5.87	*Lytechinus pictus*	(Thompson, Bay, Greenstein, & Laughlin, 1991)
Amphibians	No Data					
Aquatic Insects	No Data					

The chronic toxicity of dissolved hydrogen sulfide in marine water was evaluated in four of six prey categories. See Table 135. The lowest PCLTV was for invertebrates at 5.87 µg/L. All PCLTVs were greater than the proposed acute criterion of 2 µg/L. Therefore, exposure at the level of the chronic marine criterion is **not likely to result in reductions** in the community of marine prey species.

5.8.10.3 Freshwater hydrogen sulfide criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

- Chronic Freshwater Criterion Magnitude: *Prey reductions unlikely*

5.8.10.4 Marine hydrogen sulfide criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

- Chronic Marine Criterion Magnitude: *Prey reductions unlikely*

5.8.11 Indirect effects analysis for iron

5.8.11.1 Freshwater data summaries

The analysis of indirect effects on listed species of the proposed chronic freshwater criteria for iron is presented below. Data were acquired from an ECOTOX search conducted in 2019 and augmented by data collated in Pete Cadmus, Stephen F Brinkman, and Melynda K May (2018). The latter resource provided or cited a majority of the data for fish species that were not available in ECOTOX. Results of the queries are presented below in Table 136. Using the best available commercial and scientific data, the analysis was conducted for the effect of each WQS magnitude on surrogate prey items potentially used by listed species.

Table 136. Range of adverse chronic effect assessment concentrations for categories of prey species of the ESA listed fish species within the action area. 1000 µg/L is the proposed criterion.

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Aquatic Insects	2	12484	7863 – 19818	7863	*Hexagenia limbata*	(Pete Cadmus et al., 2018)
Amphibians	1	3145	3145	3145	*Bufo boreas*	(Pete Cadmus et al., 2018)
Fish	6	2624	520 – 9237	520	*Pimephales promelas*	(E. Smith, Sykora, & Shapiro, 1973)
Crustaceans	4	1587	480 – 5200	480	*Ceriodaphnia dubia*	ASTM Spec. Tech. Publ.6:551-556
Aquatic Invertebrates	No Data					
Molluscs	No Data					

Notes:
PCMCV = prey category geometric mean chronic value

The chronic toxicity of iron in fresh water was evaluated in aquatic insects, amphibians, fish, and crustaceans. The PCLTV for crustaceans was 480 ug/L (the NOEC for reduced fecundity) for *Ceriodaphnia dubia* and 520 ug/L for fish, an EC_{20} for reduced survival in *Pimephales promelas*. The PCLTVs for amphibians and aquatic insects were both higher than the proposed chronic criterion. See Table 136. Because the PCLTV for crustaceans and fish were both lower than the proposed criteria of 1000 ug/L, the criterion may not be protective of these prey items. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because 21% of toxicity values were less than the criterion (>20% threshold, or 3 of 14 species), exposure at the level of the chronic freshwater criterion is **likely to result in reductions** in the community of prey species.

5.8.11.2 Freshwater criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Chronic Freshwater Criterion Magnitude: *Prey reductions likely*

5.8.12 Indirect effects analysis for lead

The analysis of indirect effects on listed species of the proposed acute and chronic freshwater and marine criteria for lead is presented below. ECOTOX was searched in February of 2020 for data relevant to the indirect effects analysis approach. Results of the data analysis are presented below in Table 133 and Table 139.

5.8.12.1 Freshwater data summaries

Table 137. Range of adverse lead acute and chronic effect assessment concentrations for categories of prey species of the ESA listed fish species within the action area (number of species in each category in parentheses).

Lead Effect Concentrations on Food Items of ESA Listed Fish Species Acute Criterion = 65 µg/L, Chronic Criterion = 1.2 µg/L		
Organism Type	**Acute Effect Assessment Concentration (LC_{50}/2.27), µg/L**	**Chronic Effect Assessment Concentration, µg/L**
Fish	515 – 238,767 (N = 7)	18.9 – 101.8 (N = 3)
Amphibians	No data	No data
Aquatic Invertebrates	54 – 98,678 (N = 12)	12.3 - 128 (N = 4)
Aquatic Insects	98,678 (N = 1)	No data
Crustaceans	54 - 2247 (N = 10)	12.3 - 128 (N = 3)
Zooplankton	270 - 2247 (N = 7)	12.3 - 128 (N = 3)
Molluscs	590 (N = 1)	25.5 (N = 1)

The acute toxicity of lead in fresh waters was evaluated in five of six prey categories. None of the available freshwater fish data where acute LC_{low} values could be calculated had concentrations adversely affecting survival lower than the acute criterion of 65 µg/L. The only freshwater invertebrate data with an LC_{low} value below the acute criterion of 65 µg/L was the amphipod *Gammarus pseudolimnaeus*, whose LC_{low} value was 54 µg/L. As survival of less than 20% of both fish and invertebrate prey species are adversely affected by lead concentrations at or below the acute lead criterion, exposure at the level of the acute freshwater criterion is **not likely to result in reductions** in the community of freshwater prey species.

The chronic toxicity of lead in fresh waters was evaluated in four of six prey categories. None of the available chronic NOEC or MATC data showed any chronic toxicity to either fish or invertebrates at concentrations lower than the chronic lead criterion of 1.2 µg/L. As such, exposure at the level of the chronic freshwater criterion is **not likely to result in reductions** in the community of freshwater prey species.

5.8.12.2 Marine data summaries

Table 138. Range of adverse lead acute and chronic effect assessment concentrations for categories of prey species of the ESA listed fish species within the action area (number of species in each category in parentheses).

Lead Effect Concentrations on Food Items of ESA Listed Fish Species Acute Criterion = 210 µg/L, Chronic Criterion = 8.1 µg/L		
Organism Type	**Acute Effect Assessment Concentration (LC_{50}/2.27), µg/L**	**Chronic Effect Assessment Concentration (NOEC/MATC), µg/L**
Fish	1315.4 - 131967 (N = 5)	55 (N = 1)
Crustaceans	229.1 – 157103.5 (N = 9)	999 (N = 1)

Lead Effect Concentrations on Food Items of ESA Listed Fish Species Acute Criterion = 210 µg/L, Chronic Criterion = 8.1 µg/L		
Organism Type	**Acute Effect Assessment Concentration (LC_{50}/2.27), µg/L**	**Chronic Effect Assessment Concentration (NOEC/MATC), µg/L**
Aquatic Invertebrates (Worms)	212.3 - 41850 (N = 17)	No data
Molluscs	26.7 – 11311.4 (N = 12)	23 -1165 (N=9)
Zooplankton	No data	No data
Amphibians	No data	No data

The acute toxicity of lead in marine waters was evaluated in four of six prey categories. None of the available fish data where acute LC_{low} values could be calculated had concentrations adversely affecting survival lower than the acute criterion of 210 µg/L. The only freshwater invertebrate data with an LC_{low} value below the acute criterion of 210 µg/L was for mollusks. Because fewer than 20% of potential prey species had an LC_{Low} < 210 ug/L, exposure at the level of the acute freshwater criterion is **not likely to result in reductions** in the community of freshwater prey species.

The chronic toxicity of lead in marine waters was evaluated in three of six prey categories. None of the available chronic NOEC or MATC data showed any chronic toxicity to either fish or invertebrates at concentrations lower than the chronic lead criterion of 8.1 µg/L. As such, exposure at the level of the chronic freshwater criterion is **not likely to result in reductions** in the community of freshwater prey species.

5.8.12.3 Freshwater criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Acute Freshwater Criterion Magnitude: *Prey reductions unlikely*

Chronic Freshwater Criterion Magnitude: *Prey reductions unlikely*

5.8.12.4 Marine criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Acute Marine Criterion Magnitude: *Prey reductions unlikely*

Chronic Marine Criterion Magnitude: *Prey reductions unlikely*

5.8.13 Indirect effects analysis for malathion

The analysis of indirect effects on listed species of the proposed chronic freshwater and marine criteria for malathion is presented below. High quality, well documented, and relevant data from the EPA Office of Pesticides Programs biological evaluation of malathion registration[30] were used for this analysis. Results of the data analysis are presented below in Table 139 and Table 140.

5.8.13.1 Freshwater data summaries

Table 139. Chronic freshwater toxicity data (µg/L) for malathion. The proposed criterion is 0.1 µg/L.

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Aquatic Invertebrates	1	48050	48050	48050	*Euglena gracilis*	(Azizullah, Richter, & Häder, 2011)
Fish	10	846	21 – 9800	21	*Oncorhynchus mykiss*	(USEPA, MRID41422401)
Molluscs	2	67.9	9.6 – 480	9.6	*Planorbella trivolvis*	(Rohr et al., 2008)
Aquatic Insects	2	24.2	6 – 300	6	*Culex tarsalis*	(Jensen, Lawler, & Dritz, 1999)
Amphibians	4	71.9	1 – 2880	1	*Xenopus laevis*	(Webb & Crain, 2006)
Crustaceans	3	0.32	0.06 – 0.57	0.06	*Daphnia magna*	(USEPA, MRID41718401)

The chronic toxicity of malathion in freshwater was evaluated in all six prey categories. See Table 139. The PCLTV for crustaceans was 0.06 µg/L for diminished number of offspring in *Daphnia magna,* one of three tested crustacean species. As the lowest PCLTV of 0.06 µg/L was lower than the chronic criterion of 0.1 µg/L, the criterion may not be protective of prey species relevant to listed species. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because only 4.5% of toxicity values were less than the criterion (<20% threshold, or 1 of 22 species), exposure at the level of the chronic freshwater criterion is **not likely to result in reductions** in the community of freshwater prey species.

5.8.13.2 Marine data summaries

Table 140. Chronic marine toxicity data (µg/L) for malathion. The proposed criterion is 0.1 µg/L.

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Fish	2	9.91	4 – 37	4	*Cyprinodon variegatus*	(D. Hansen & P. Parrish, 1977; D. J. Hansen & P. R. Parrish, 1977)

[30] https://www.epa.gov/endangered-species/biological-evaluation-chapters-malathion-esa-assessment

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Crustaceans	2	10.2	0.58 – 180	0.58	*Americamysis bahia*	(USEPA, MRID48752901)
Aquatic Invertebrates	No Data					
Amphibians	No Data					
Molluscs	No Data					
Aquatic Insects	No Data					

The chronic toxicity of malathion in marine waters was evaluated in fish and crustaceans only. See Table 140. The PCLTVs for crustaceans (0.58 µg/L) and fish (4 µg/L) were both higher than the proposed chronic marine criterion of 0.1 µg/L. No other marine chronic toxicity data were available. Because 0.1 µg/L is lower than the lowest PCLTV, exposure at the level of the chronic marine criterion is **not likely to result in reductions** in the community of marine prey species.

5.8.13.3 Freshwater criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Chronic Freshwater Criterion Magnitude: *Prey reductions unlikely*

5.8.13.4 Marine criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Chronic Marine Criterion Magnitude: *Prey reductions unlikely*

5.8.14 Indirect effects analysis for nickel

The analysis of indirect effects on listed species of the proposed acute and chronic freshwater and marine criteria for nickel is presented below. ECOTOX was searched in February of 2020 for data

relevant to the indirect effects analysis approach. Results of the data analysis are presented below in Table 141 and Table 142.

5.8.14.1 Freshwater data summaries

Table 141. Range of adverse nickel acute and chronic effect assessment concentrations for categories of prey species of the ESA listed fish species within the action area (number of species in each category in parentheses).

Nickel Effect Concentrations on Food Items of ESA Listed Fish Species Acute Criterion = 468 µg/L, Chronic Criterion = 52.0 µg/L		
Organism Type	**Acute Effect Assessment Concentration (LC_{50}/2.27), µg/L**	**Chronic Effect Assessment Concentration, µg/L**
Aquatic Insects	52463.2 (N = 1)	10100 (N = 1)
Fish	9162.9 (N =1)	2269 (N = 3)
Molluscs	58– 263.9 (N = 3)	No data
Crustaceans	881(N = 1)	400– 10100 (N = 2)
Amphibians	No data	No data
Aquatic Invertebrates	No data	No data

The acute toxicity of nickel in fresh waters was evaluated in four of six prey categories. None of the available freshwater fish data where acute LC_{low} values could be calculated had concentrations adversely affecting survival lower than the acute criterion of 468 µg/L. The only freshwater invertebrate data with an LC_{low} value below the acute criterion of 468 µg/L was the mollusc, whose LC_{low} value was 58 µg/L. As survival of more than 20% of prey species are adversely affected by nickel concentrations at or below the acute nickel criterion, exposure at the level of the acute freshwater criterion is **likely to result in reductions** in the community of freshwater prey species.

The chronic toxicity of nickel in fresh waters was evaluated in three of six prey categories. None of the available chronic NOEC or MATC data showed any chronic toxicity to either fish or invertebrates at concentrations lower than the chronic nickel criterion of 52 µg/L. As such it is unlikely that the chronic freshwater criterion will have adverse effects on prey species of the ESA listed salmonid species within the action area; therefore, exposure at the level of the chronic freshwater criterion is **not likely to result in reductions** in the community of freshwater prey species.

5.8.14.2 Marine data summaries

Table 142. Range of adverse nickel acute and chronic effect assessment concentrations for categories of prey species of the ESA listed fish species within the action area (number of species in each category in parentheses).

Nickel Effect Concentrations on Food Items of ESA Listed Fish Species Acute Criterion = 74 µg/L, Chronic Criterion = 8.2 µg/L		
Organism Type	**Acute Effect Assessment Concentration (LC_{50}/2.27), µg/L**	**Chronic Effect Assessment Concentration, µg/L**
Fish	3505.7 - 154185 (N = 8)	7513 (N = 1)
Amphibians	No data	No data

Nickel Effect Concentrations on Food Items of ESA Listed Fish Species Acute Criterion = 74 µg/L, Chronic Criterion = 8.2 µg/L		
Organism Type	**Acute Effect Assessment Concentration (LC_{50}/2.27), µg/L**	**Chronic Effect Assessment Concentration, µg/L**
Aquatic Invertebrates (Worms)	7489 -140969 (N = 4)	17000 – 25000 (N =2)
Crustaceans	223.8 – 49339.2 (N = 9)	213 – 497 (N = 2)
Zooplankton	No data	No data
Mollusks	3171,8 - 140969 (N = 2)	100 (N=1)

The acute toxicity of nickel in marine waters was evaluated in four of six prey categories. All of the available acute LC_{low} values for potential prey species were above the acute criterion of 74 µg/L. Based on these data, exposure at the level of the acute marine criterion is **not likely to result in reductions** in the community of marine prey species.

The chronic toxicity of nickel in marine waters was evaluated in four of six prey categories. None of the available chronic NOEC or MATC data showed any chronic toxicity to either fish or invertebrates at concentrations lower than the chronic nickel criterion of 8.2 µg/L. As such, exposure at the level of the chronic marine criterion is **not likely to result in reductions** in the community of marine prey species.

5.8.14.3 Freshwater criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Acute Freshwater Criterion Magnitude: *Prey reductions likely*

Chronic Freshwater Criterion Magnitude: *Prey reductions unlikely*

5.8.14.4 Marine criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Acute Marine Criterion Magnitude: *Prey reductions unlikely*

Chronic Marine Criterion Magnitude: *Prey reductions unlikely*

5.8.15 Indirect effects analysis for nonylphenol

The analysis of indirect effects on listed species of the proposed acute and chronic freshwater and marine criteria for nonylphenol (NP) is presented below. ECOTOX was queried in January of 2020 for relevant data. Results of the queries are presented below in Table 143 through Table 146.

5.8.15.1 Freshwater data summaries

Table 143. Acute freshwater toxicity data (ug/L) for nonylphenol. The proposed criterion is 28 ug/L.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Fish	2	13.7	13.5 – 13.8	13.5	*Lepomis macrochirus*	(L. Brooke, 1993)
Aquatic Invertebrates	No Data					
Amphibians	No Data					
Molluscs	No Data					
Aquatic Insects	No Data					

The acute toxicity of NP in freshwaters was evaluated in fish only. See Table 143. The PCLTV for fish was 13.5 ug/L for mortality to *Lepomis macrochirus* of two tested fish species. As the lowest PCLTV of 13.5 ug/L was lower than the criterion of 28.0 ug/L, the criterion may not be protective of prey species relevant to listed species. Therefore, EPA calculated the percent of species with toxicity values less than the criterion and found that because 2 of 2 (100%; >20% threshold) species toxicity values were greater than the criterion, exposure at the level of the acute freshwater criterion is **likely to result in reductions** in the community of prey species.

Table 144. Chronic freshwater toxicity data (ug/L) for nonylphenol. The proposed criterion is 6.6 ug/L.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Aquatic Insects	1	21.0	21.0	21.0	*Chironomus tentans*	(D. England & Bussard, 1993)
Fish	3	37.2	7.40 – 77.5	7.40	*Pimephales promelas*	(T. Ward & R. Boeri, 1991b)
Crustaceans	1	5.00	5.00	5.00	*Gammarus fossarum*	(Geffard et al., 2010)
Aquatic Invertebrates	No Data					
Amphibians	No Data					
Molluscs	No Data					

The chronic toxicity of NP in freshwater was evaluated in aquatic insects, fish, and crustaceans. See Table 144. The PCLTV for aquatic insects was 21.0 ug/L for diminished growth to the only tested insect species, *Chironomus tentans*; 7.40 ug/L for survival to *Pimephales promelas* (fathead minnow) of three tested fish species; and, 5.00 ug/L for reduced gamete production to *Gammarus fossarum*, the only tested crustacean species. As the lowest PCLTV of 5.00 ug/L was lower than the chronic criterion of 6.6

ug/L, the criterion may not be protective of prey species relevant to listed species. Therefore, the percent of species with toxicity values less than the criterion was calculated. Because only 11% of toxicity values were less than the criterion (<20% threshold, or 1 of 9 studies), exposure at the level of the chronic freshwater criterion is **not likely to result in reductions** in the community of prey species.

5.8.15.2 Marine data summaries

Table 145. Acute marine toxicity data (ug/L) for nonylphenol. The proposed criterion is 7.0 ug/L.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Fish	3	85.8	70.0 – 320	70.0	*Pleuronectes americanus*	(Suzanne M Lussier, Denise Champlin, Joseph LiVolsi, Sherry Poucher, & Richard J Pruell, 2000)
Crustaceans	4	58.4	43.0 – 71.0	43.0	*Americamysis bahia*	(Ward & Boeri, 1990)
Aquatic Invertebrates	No Data					
Amphibians	No Data					
Molluscs	No Data					
Aquatic Insects	No Data					

The acute toxicity of NP in marine waters was evaluated in fish and crustaceans only. See Table 145. The PCLTV for fish was 70.0 ug/L for mortality to *Pleuronectes americanus* of three tested fish species; 43.0 ug/L for mortality to *Americamysis bahia* of four tested crustacean species. As the lowest PCLTV of 43.0 ug/L was higher than the acute marine criteria for NP of 7.0 ug/L, exposure at the level of the acute marine criterion is **not likely to result in reductions** in the community of marine prey species.

Table 146. Chronic marine toxicity data (ug/L) for nonylphenol. The proposed criterion is 1.7 ug/L.

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Crustaceans	1	6.20	3.90 – 9.10	3.90	*Americamysis bahia*	(T. Ward & R. Boeri, 1991a)
Aquatic Invertebrates (rotifers)	No Data					
Molluscs	No Data					
Aquatic Insects	No Data					
Amphibians	No Data					
Fish	No Data					

The chronic toxicity of NP in marine waters was evaluated in crustaceans and the PCLTV was 3.90 ug/L (the NOEC for reduced growth) for *Americamysis bahia*, the only tested species. No other marine chronic toxicity data were available. See Table 146. The chronic marine criterion for NP is 1.7 ug/L. Because 1.7 ug/L is less than the PCLTV for all available prey categories, exposure at the level of the chronic marine criterion is **not likely to result in reductions** in the community of marine prey species.

5.8.15.3 Freshwater criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Acute Freshwater Criterion Magnitude: *Prey reductions likely*

Chronic Freshwater Criterion Magnitude: *Prey reductions unlikely*

5.8.15.4 Marine criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Acute Marine Criterion Magnitude: *Prey reductions unlikely*

Chronic Marine Criterion Magnitude: *Prey reductions unlikely*

5.8.16 Indirect effects analysis for pentachlorophenol

The analysis of indirect effects on listed species of the proposed acute freshwater and marine criteria for pentachlorophenol (PCP) is presented below. Results of the ECOTOX data base queries performed in 2019 and 2020 are presented below in Table 147 through Table 150. Using the best available commercial and scientific data, the analysis was conducted for the effect of each WQS magnitude on surrogate prey items potentially used by listed species.

5.8.16.1 Freshwater toxicity data summary

Table 147. Acute freshwater toxicity data (µg/L) for a range of listed species prey items. The proposed acute freshwater criterion is 19 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category mean acute values (PCMAV) are geometric means of all species acute toxicity values.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Insects	5	1058	533 – 43172	533	*Pteronarcys dorsata*	(D.J. Call & Brooke, 1982)
Molluscs	7	146.6	62.6 – 608	62.6	*Aplexa hypnorum*	(Gary L Phipps & Gary W Holcombe, 1985)

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Crustaceans	9	232.6	26.4 – 23348	26.4	*Daphnia magna*	(Adema, 1978)
Amphibians	2	103.9	22.0 – 229	22.0	*Xenopus laevis*	(Fort & Stover, 1996)
Fish	22	86.3	7.93 – 1321	7.93	*Oncorhynchus mykiss*	(Van Leeuwen, Griffioen, Vergouw, & Maas-Diepeveen, 1985)
Invertebrates	**No Data**					

The acute toxicity of PCP in freshwater was evaluated in five of six prey categories. See Table 147. PCLTVs for one prey category (fish) was 7.93µg/L, which is less than the proposed acute freshwater criterion of 19 µg/L and so may not be protective for prey of listed species. Thus, the percent of species with toxicity values less than the criterion was calculated (1 out of 45 or 2%). Because 2% of SMAVs were less than the criterion (<20% threshold), exposure at the level of the acute freshwater criterion is **not likely to result in reductions** in the community of freshwater prey species.

Table 148. Chronic freshwater toxicity data (µg/L) for a range of listed species prey items. The proposed chronic freshwater criterion is 15 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category mean chronic values (PCMCV) are geometric means of all species acute toxicity values.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Insects	1	250	250	250	*Chironomus riparius*	(Morales et al., 2014)
Molluscs	5	85.4	11.0 – 377	11.0	*Pyrgulopsis idahoensis*	(Besser, Hardesty, Greer, & Ingersoll, 2009)
Amphibians	1	31.5	3.0 – 2500	3.0	*Xenopus laevis*	(Fort & Stover, 1996)
Fish	14	79.9	0.10 – 640.0	0.10	*Zebra Danio*	(Cheng, Ekker, & Chan, 2015)
Crustaceans	8	16.4	0.05 – 600	0.05	*Daphnia magna*	(Chen et al., 2014)
Invertebrates	**No Data**					

The chronic toxicity of PCP in freshwater was evaluated in five of six prey categories. See Table 148. PCLTVs for four of five prey categories (crustaceans, fish, amphibians, and molluscs) were less than the proposed chronic freshwater criterion of 15 µg/L and so it may not be protective for prey of listed species. Thus, the percent of species with toxicity values less than the criterion was calculated (5 out of 29 or 17%). Because 17% of SMAVs were less than the criterion (<20% threshold), exposure at the level of the chronic freshwater criterion is **not likely to result in reductions** in the community of freshwater prey species.

5.8.16.2 Marine toxicity data summary

Table 149. Acute marine toxicity data (µg/L) for a range of listed species prey items. The proposed acute criterion is 13 µg/L. Prey category mean acute values (PCMAV) are geometric means of all species acute toxicity values.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Crustaceans	13	279.3	70 – 8200	70.0	*Nitocra spinipes*	(Bengtsson & Bergström, 1987)
Molluscs	4	200.9	31.3 – 705	31.3	*Crassostrea virginica*	(Davis & Hidu, 1969)
Invertebrates	2	18.2	16.3 – 20.3	16.3	*Leptosynapta inhaerens*	(Tagatz & Stanley, 1987)
Fish	7	69.9	10.1 – 770	10.1	*Odontesthes regia*	(Silva, Troncoso, Bay-Schmith, & Larrain, 2001)
Amphibians	**No Data**					
Aquatic Insects	**No Data**					

The acute toxicity of PCP in marine water was evaluated in four of six prey categories. See Table 149. The lowest PCLTV was for fish at 10.1 µg/L, which is lower than the proposed acute criterion of 13 µg/L. Therefore, the percent of species with toxicity values less than the criterion was calculated (1 out of 26 or 3.8%). Because 3.8% of SMAVs were less than the criterion (<20% threshold), exposure at the level of the acute marine criterion is **not likely to result in reductions** in the community of marine prey species.

Table 150. Chronic toxicity data (µg/L) for a range of listed species prey items. The proposed chronic criterion is 7.9 µg/L. Prey category mean chronic values (PCMCV) are geometric means of all species acute toxicity values.

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Crustaceans	2	52.0	20 – 10000	20	*Nitocra spinipes*	(Bengtsson & Bergström, 1987)
Invertebrates	**No Data**					
Molluscs	**No Data**					
Fish	**No Data**					
Amphibians	**No Data**					
Aquatic Insects	**No Data**					

The chronic toxicity of PCP in marine water was evaluated in one of six prey categories. See Table 150. The lowest PCLTV was for crustaceans at 20.0 µg/L, which is higher than the proposed acute criterion of 7.9 µg/L. Therefore, exposure at the level of the chronic marine criterion is **not likely to result in reductions** in the community of marine prey species.

5.8.16.3 Freshwater criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Acute Freshwater Criterion Magnitude: *Prey reductions unlikely*

Chronic Freshwater Criterion Magnitude: *Prey reductions unlikely*

5.8.16.4 Marine criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Acute Marine Criterion Magnitude: *Prey reductions unlikely*

Chronic Marine Criterion Magnitude: *Prey reductions unlikely*

5.8.17 Indirect effects analysis for elemental phosphorus

The analysis of indirect effects on listed species of the proposed chronic marine criteria for elemental phosphorus (EP) is presented below. ECOTOX was queried for relevant data in December 2019, but because the most recent data were from 1978, a Web of Knowledge search was conducted on February 24, 2020; however, no additional data were found. Results of the queries are presented below in Table 151.Using the best available commercial and scientific data, the analysis was conducted for the effect of the WQS magnitude on surrogate prey items potentially used by listed species.

5.8.17.1 Marine toxicity data summary

Table 151. Chronic marine toxicity data (µg/L) for a range of listed species prey items. The proposed EP marine chronic criterion is 0.10 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category marine acute values (PCMAV) are geometric means of all species acute toxicity values.

Prey Category	N Species	PCMCV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Fish	1	1.73[1]	1.73	1.73	*Gadus morhua*	(Maddock & Taylor, 1976)
Crustaceans	No Data					
Aquatic Invertebrates	No Data					
Molluscs	No Data					
Amphibians	No Data					
Aquatic Insects	No Data					

1, The chronic value was determined by dividing the acute toxicity value of 14.4 µg/L by 8.3, a widely applicable ACR (Sandy Raimondo, Brian J Montague, & Mace G Barron, 2007).

The chronic toxicity of EP in marine water was evaluated in fish only. See Table 151. The PCLTV for fish was 1.73 µg/L. Because the PCLTV for fish was greater than the proposed acute marine criterion of 0.10 µg/L, exposure at the level of the chronic marine criterion is **not likely to result in reductions** in the community of prey species.

5.8.17.2 Marine criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

- Chronic Marine Criterion Magnitude: *Prey reductions unlikely*

5.8.18 Indirect effects analysis for silver

The analysis of indirect effects on listed species of the proposed acute freshwater and marine criteria for silver is presented below. Results of the ECOTOX data base queries are presented below in Table 152 and Table 153. Using the best available commercial and scientific data, the analysis was conducted for the effect of each WQS magnitude on surrogate prey items potentially used by listed species. Using (EPA, 2006; Swinomish Indian Tribal Community, 2016) all toxicity values and criteria values were adjusted to a hardness of 100 mg/L $CaCO_3$.

5.8.18.1 Freshwater toxicity data summary

Table 152. Acute Freshwater toxicity data (µg/L) for a range of listed species prey items. The proposed silver freshwater acute criterion is based on the hardness adjusted toxicity value to prey species. The CMC for silver at a hardness of 100 mg CaC1003/L is 3.22 µg/L. Prey categories are sorted from highest PCLTV to lowest. Prey category mean acute values (PCMAV) are geometric means of all species acute toxicity values.

	N			Prey Category Lowest Toxicity Value (PCLTV)		
Prey Category	**Species**	**PCMAV**	**Range**	**PCLTV**	**Species**	**Reference**
Aquatic Insects	1	78.6	78.6	78.6	*Chironomus tentans*	(Daniel J Call et al., 1999)
Crustaceans	2	4.3	4.3 – 7.3	1.9	*Daphnia pulex*	(Marcial et al., 2005; G. Thursby & Berry, 1988)
Fish	3	13.9	3.9 – 153.8	0.49	*Pimephales promelas*	(Marcial et al., 2005)
Amphibians	**No Data**					
Invertebrates	**No Data**					
Molluscs	**No Data**					

The acute toxicity of dissolved silver in freshwater was evaluated in fish, aquatic insects and crustaceans. See Table 148. PCLTVs for two prey categories (Aquatic insects and crustaceans) of 78.6 and 1.9 µg/L, respectively, were greater than the proposed hardness-based acute freshwater criterion of 3.22 µg/L.

However, the PCTLV for fish was 0.49 µg/L is lower than the proposed freshwater criterion and so may not be protective for prey of listed species. Thus, the percent of species with toxicity values less than the criterion was calculated (1 out of 6 or 0.16 %). Because 0.16 % of SMAVs were less than the criterion (<20% threshold), exposure at the level of the acute freshwater criterion is **not likely to result in reductions** in the community of freshwater prey species.

5.8.18.2 Marine toxicity data summary

Table 153. Marine toxicity data (µg/L) for a range of listed species prey items. The proposed acute criterion is 1.9 µg/L. Prey category mean acute values (PCMAV) are geometric means of all species acute toxicity values.

Prey Category	N Species	PCMAV	Range	Prey Category Lowest Toxicity Value (PCLTV)		
				PCLTV	Species	Reference
Crustaceans	9	48,528	8.81 – 2797	8.81	*Ampelisca abdita*	(Berry et al., 1999)
Invertebrates	1	17,574	6.65 – 90.3	6.65	*Brachionus plicatilis*	(Saunders, 2012)
Molluscs	4	28,139	2.56 – 1762	2.56	*Crassostrea virginica*	(A. Calabrese, Collier, Nelson, & MacInnes, 1973)
Fish	13	153,251	2.07 – 308,370	2.07	*Paralichthys dentatus*	(S. M. Lussier & Cardin, 1985)
Amphibians	**No Data**					
Aquatic Insects	**No Data**					

The acute toxicity of dissolved silver in marine water was evaluated in four of six prey categories. See Table 153. The lowest PCLTV was for fish at 2.07 µg/L. All PCLTVs were greater than the proposed acute criterion of 1.9 µg/L. Therefore, exposure at the level of the acute marine criterion is **not likely to result in reductions** in the community of marine prey species.

5.8.18.3 Freshwater criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

- Acute Freshwater Criterion Magnitude: *Prey reductions unlikely*

5.8.18.4 Marine criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

- Acute Marine Criterion Magnitude: *Prey reductions unlikely*

5.8.19 Indirect effects analysis for zinc

The analysis of indirect effects on listed species of the proposed acute and chronic freshwater and marine criteria for zinc is presented below. Methods as described in section 5.4.5 were followed. ECOTOX was searched in February of 2020 for data relevant to the indirect effects analysis approach. Results of the data analysis are presented below in Table 154 and Table 155.

5.8.19.1 Freshwater data summaries

A total of 53 species representing 44 genera have empirical zinc acute toxicity data. For this prey species evaluation, the empirical 96 or 48 hour LC_{50}s were divided by 2.27 to calculate an LC_{low} for each species. These LC_{low}s were used as the assessment effects concentration for each prey species.

Nine species from seven genera have empirical chronic toxicity data. One of the species with empirical chronic toxicity data, the caddisfly (*Clistoronia magnifica*) has empirical chronic toxicity data but no empirical acute toxicity data. This results in a total of 54 species with chronic toxicity data used to assess zinc risks. For the rest of the species with empirical acute toxicity data, a chronic NOEC was estimated by dividing the LC_{50} by the zinc acute-chronic ratio (ACR) of 2.0 (EPA 1995). Chronic assessment effects concentrations were calculated for species without empirical chronic toxicity data by dividing the acute LC_{50} by 2.0.

Table 154. Range of adverse freshwater zinc acute and chronic effect assessment concentrations for categories of prey species of the ESA listed fish species within the action area (number of species in each category in parentheses).

Zinc Effect Concentrations on Food Items of ESA Listed Fish Species Acute Criterion = 120 µg/L, Chronic Criterion = 120 µg/L (both at 100 mg/L hardness)		
Organism Type	**Acute Effect Assessment Concentration (LC_{50}/2.27), µg/L**	**Chronic Effect Assessment Concentration (LC_{50}/ACR), µg/L**
Fish	92.6 – 25,246 (N = 28)	105 – 28,655 (N = 28)
Amphibian	14,864 (N = 1)	16,871 (N = 1)
Aquatic Invertebrates	39.3 – 68,956 (N = 24)	44.6 – 78,265 (N = 25)
Worms	980 – 14,263 (N = 3)	1112 – 16,188 (N = 3)
Aquatic Insects	11,435 – 68,956 (N = 2)	12,979 – 78,265 (N = 3)
Crustaceans	39.3 – 15,348 (N = 10)	44.6 – 17,420 (N = 10)
Zooplankton	39.3 - 276 (N = 4)	44.6 - 313 (N = 4)
Molluscs	256 – 13,038 (N = 5)	291 – 14,798 (N = 5)

Although the number of zooplankton species with empirical acute toxicity data (4) and empirical chronic toxicity data (1) is low, they appear as a group to be the most zinc sensitive of all freshwater prey categories evaluated. The two *Ceriodaphnia* species with empirical acute toxicity data both have acute LC_{50} values lower than the 120 µg/L freshwater acute zinc criterion: 117 µg/L for *Ceriodaphnia dubia* and 89.21 µg/L for *Ceriodaphnia reticulata*. A third zooplankter, *Daphnia magna*, has an empirically measured chronic NOEC of 100.1 µg/L, lower than the chronic freshwater zinc criterion of 120 µg/L. The two *Ceriodaphnia* species' acute LC50s are also lower than the chronic zinc criterion, meaning three of

the four zooplankton species with available data may be adversely affected by chronic exposure to zinc at or below the chronic criterion.

Individual fish species may also be adversely affected by exposure at either the acute or chronic zinc criterion. Of the 28 fish species with empirical acute toxicity data, both bull trout and striped bass have acute assessment effects concentrations and calculated chronic assessment effects concentrations lower than the acute and chronic zinc criteria (please see section 5.5.1.15.8 for considerations related to zinc toxicity to bull trout and other aquatic species). Individual replicate acute and chronic toxicity tests of zinc with rainbow trout also have assessment effects concentrations lower than the acute and chronic zinc criteria, although the geometric means of all available rainbow trout toxicity studies with zinc is higher than the acute and chronic zinc criteria.

Overall when all prey taxa are combined, fewer than 8% of tested prey species are adversely affected by zinc exposure at the acute or chronic criteria values. This is lower than the 20% reduction in species richness where adverse effects on community structure begins to be observable (see section 5.4.5). Planktonic zooplankton appears to be the only taxonomic group where potential adverse effects on species richness may occur at exposure to zinc at criteria levels. Based on these observations, exposure at the level of the acute and chronic freshwater criteria is **not likely to result in reductions** in the community of freshwater prey species.

5.8.19.2 Marine data summaries

In addition to the 11 marine fish species with empirical zinc acute toxicity data, there are 41 invertebrate species with empirical zinc acute toxicity data. As noted earlier, we have identified no marine fish species with acceptable empirical chronic zinc toxicity data. We are aware of only two marine invertebrate species with acceptable empirical chronic zinc toxicity data: *Americamysis bahia* and *Acanthomysis costata*, both mysids (opossum shrimps). These empirical results will be used as part of the assessment of the chronic marine zinc criterion.

The toxicity values for marine organisms compiled in Table 155 were grouped into common categories (e.g., insects, molluscs, fish, etc.). Each respective acute toxicity LC_{50} value used to generate the range of acute effect assessment concentrations in Table 155 was divided by 2.27 to generate the acute no effect concentration range shown in Table 155. Chronic data in Table 155 are long-term chronic NOEC or MATC data. To obtain additional data with which an assessment of the marine chronic zinc criterion can be made, all acceptable marine fish and invertebrate empirical acute toxicity LC_{50} data (96 hour exposures for fish, 48 or 96 hour exposures for invertebrates) will be divided by the zinc acute-chronic ratio of 2.208 to obtain estimates of chronic NOEC concentrations. The number of prey species with toxicity data within each category of prey species is also presented.

Table 155. Range of adverse marine zinc acute and chronic effect assessment concentrations for categories of prey species of the ESA listed fish species within the action area (number of species in each category in parentheses).

Zinc Effect Concentrations on Food Items of ESA Listed Fish Species Acute Criterion = 90 µg/L, Chronic Criterion = 81 µg/L		
Organism Type	**Acute Effect Assessment Concentration (LC_{50}/2.27), µg/L**	**Chronic Effect Assessment Concentration, µg/L**
Fish	84.3 – 16,740 (N = 11)	86.7 – 17,210 (N = 11)
Aquatic Invertebrates	22.4 – 133,523 (N = 41)	23.1 - 137,273 (N = 41)
Worms	560 – 58,177 (N = 10)	576 – 59,811 (N = 10)
Aquatic Insects	No data	No data
Crustaceans	40.1 - 6610 (N = 17)	63.46 - 6796 (N = 17)
Zooplankton	40.1 - 1698 (N = 7)	63.46 - 1745 (N = 7)
Molluscs	22.4 – 133,523 (N = 11)	23.1 - 137,273 (N = 11)

Five of the 52 species for which empirical acute zinc toxicity in marine waters exist have assessment effects concentrations (= 96 hour LC_{50} / 2.27) lower than the marine acute zinc criterion of 90 µg/L. The species whose assessment effects concentration is lower than the 90 µg/L acute zinc criterion include the cabezon (84.3 µg/L), the only fish species with an acute assessment effects concentration lower than the acute criterion. Two molluscs, the quahog (*Mercenaria mercenaria*) and the red abalone (*Haliotis rufescens*) had assessment effect concentrations of 81.3 µg/L and 22.4 µg/L, respectively. The red abalone has the lowest acute zinc assessment effects concentration of the 52 species with empirically measured zinc acute toxicity data. The last two invertebrate species with assessment effects concentrations lower than the acute zinc criterion are the crab *Nanosesarma sp*. and the mysid *Acanthomysis costata*, with assessment effect concentrations of 52.1 µg/L and 40.1 µg/L, respectively.

Overall, 9.6% of all marine species with empirical acute toxicity data for zinc have assessment effects concentrations lower than the acute marine zinc criterion of 90 µg/L. The taxonomic group with the highest proportion of species with assessment effects concentrations lower than the acute zinc criterion was molluscs, where two of 11 species (18%) have assessment effects concentrations lower than the 90 µg/L acute zinc criterion. Since none of the taxonomic groups with empirical acute toxicity data have more than 20% of the species in the group with assessment effects concentrations lower than the acute zinc criterion, exposure at the level of the acute marine criteria is **not likely to result in reductions** in the community of marine prey species.

Because there are only two marine species with empirical chronic zinc toxicity data, we have divided the acute toxicity 96 hour LC_{50}s for all species with empirical acute toxicity data by the acute-chronic ratio for zinc of 2.208 to estimate chronic NOEC data for marine species. Comparing the estimated chronic NOEC data and the two empirical chronic NOECs to the chronic marine zinc criterion of 81 µg/L, there are three species with assessment effects concentrations lower than the chronic criterion. One of the three, the mysid *Acanthomysis costata* has an empirically measured chronic NOEC of 63.46 µg/L. The other two invertebrates with estimate chronic NOEC assessment effects concentrations lower than the 81 µg/L chronic marine zinc criterion are the crab *Nanosesarma sp*. and the red abalone, with assessment effect concentrations of 53.56 µg/L and 23.08 µg/L respectively. Neither the proportion of total prey species (3 of 52) or species within the taxonomic groups shown in Table 155 reach 20%

reduction in species richness; on this basis, exposure at the level of the chronic marine criterion is **not likely to result in reductions** in the community of marine prey species.

5.8.19.3 Freshwater criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Acute Freshwater Criterion Magnitude: *Prey reductions unlikely*

Chronic Freshwater Criterion Magnitude: *Prey reductions unlikely*

5.8.19.4 Marine criteria conclusions of protectiveness for prey of listed species

The conclusions presented here will be used to make indirect effects determinations for each listed species subject to this consultation. A conclusion of *prey reductions unlikely* reflects evidence for minimal effects to listed species whereas a conclusion of *prey reductions likely* suggests effects to listed species through prey reductions are foreseeable if chemical concentrations reach criteria levels.

Acute Marine Criterion Magnitude: *Prey reductions unlikely*

Chronic Marine Criterion Magnitude: *Prey reductions unlikely*

5.8.20 Summary of indirect effects assessment conclusions

The conclusions of the indirect effects assessments are summarized in Table 156. Entries of "prey reductions unlikely" indicate that exposure at the criterion level for a chemical is not likely to result in reduction of available prey for a listed species. Entries of "prey reductions likely" indicate that exposure at the criterion level for a chemical may result in reduction of available prey for a listed species. In combination with the direct effects analysis and other factors, a determination of "prey reduction unlikely" supports an overall NLAA determination for a species, whereas a determination of "prey reduction likely" results in an overall LAA determination. Each conclusion for effects to prey applies to the prey of all listed species included in this consultation.

Table 156. Summary of Indirect Effects Assessments

Chemical	**Freshwater**		**Marine**	
	Acute	**Chronic**	**Acute**	**Chronic**
Acrolein	*Prey reductions unlikely*	*Prey reductions unlikely*	NA	NA
Arsenic	*Prey reductions unlikely*	NA	*Prey reductions unlikely*	*Prey reductions unlikely*
Carbaryl	*Prey reductions unlikely*	*Prey reductions unlikely*	*Prey reductions unlikely*	NA
Chlorine	*Prey reductions unlikely*	*Prey reductions unlikely*	*Prey reductions unlikely*	*Prey reductions unlikely*
Chlorpyrifos	*Prey reductions unlikely*	*Prey reductions unlikely*	*Prey reductions unlikely*	*Prey reductions unlikely*

Chemical	Freshwater		Marine	
	Acute	**Chronic**	**Acute**	**Chronic**
Chromium VI	NA	*Prey reductions likely*	*Prey reductions likely*	*Prey reductions likely*
Cyanide	NA	NA	*Prey reductions unlikely*	*Prey reductions unlikely*
Diazinon	*Prey reductions unlikely*	*Prey reductions unlikely*	*Prey reductions unlikely*	*Prey reductions unlikely*
Hydrogen sulfide	NA	*Prey reductions unlikely*	NA	*Prey reductions unlikely*
Iron	NA	*Prey reductions likely*	NA	NA
Lead	*Prey reductions unlikely*	*Prey reductions unlikely*	*Prey reductions unlikely*	*Prey reductions unlikely*
Malathion	NA	*Prey reductions unlikely*	NA	*Prey reductions unlikely*
Nickel	*Prey reductions likely*	*Prey reductions unlikely*	*Prey reductions unlikely*	*Prey reductions unlikely*
Nonylphenol	*Prey reductions likely*	*Prey reductions unlikely*	*Prey reductions unlikely*	*Prey reductions unlikely*
Pentachlorophenol	NA	NA	*Prey reductions unlikely*	*Prey reductions unlikely*
Elemental Phosphorus	NA	NA	NA	*Prey reductions unlikely*
Silver	*Prey reductions unlikely*	NA	NA	NA
Zinc	*Prey reductions unlikely*	*Prey reductions unlikely*	*Prey reductions unlikely*	*Prey reductions unlikely*
NA – Not applicable. The Tribe has not adopted a criterion, or the criterion is not currently part of this ESA consultation.				

5.9 Narrative Criteria

Sections 19-06.100, 19-06.140 and 19-06.150 of the Swinomish Tribe's WQS provide narrative criteria that protect the aquatic life and aquatic-dependent wildlife uses (see section 2.4.5). The narrative criteria are qualitative statements that provide general water quality requirements to protect aquatic life (and other water uses), including for example a general prohibition against pollutants in amounts that are toxic to aquatic life (19-06.140(B)), or against any substances that are deleterious to aquatic life (19-06.150(B)(3)). Additional narrative criteria provide protections from specific substances, such as floating solids and bottom deposits, turbidity, and color, and protection of instream flow for aquatic life use.

These narrative criteria require aquatic life to be protected in all waters regulated under these standards. Where numeric criteria have not been adopted, the narrative criteria provide a mechanism for the Tribe or EPA to address a specific water quality problem by interpreting the narrative criteria on a case by case basis. They are broad in scope so can be used to address the protection of a particularly sensitive species or a unique set of circumstances.

The narrative criteria include protections that directly apply to ESA listed species, and they are applied to all of the Tribe's waters. Section 19-06.120 of the Swinomish Tribe's WQS provides a water use

classification specifically for salmon and trout in fresh waters, including listed species, and designates all of the named fresh waters for protection of salmon and trout in table 1 of the WQS. Section 19-06.130(C)(1) defines similar uses for marine waters, including salmonids and other native species, which would include bocaccio and yellow-eye rockfish. Section 19-06.130(B)(1) defines a use for all indigenous "fish and non-fish aquatic life," and sections 19-06.120(G)(1)(c) and 19-06.130(F)(1)(c) define wildlife uses in fresh and marine waters, which would include whales and marbled murrelet. The fish and wildlife uses and narrative criteria are applied to all of the Tribe's fresh and marine waters (WQS tables 1, 2, 3, and 9). As a result, the narrative criteria must be applied to protect aquatic life and aquatic-dependent wildlife, including the ESA-listed species, in all of the Tribe's waters.

The plain reading of the narrative criteria ensures the condition of the waters will avoid adverse effects to listed salmonids and other listed species within the Tribe's waters, and EPA will use this interpretation in applying the criteria. Application of the narrative criteria would occur primarily through NPDES permits, which would implement the site-specific interpretation of the narrative criteria. For example, these narrative criteria would be used as a basis for effluent limitations in NPDES permits, to the extent an effluent limitation is required, to address pollutants that do not have numeric criteria. On a more limited basis, application could occur in the issuance of a TMDL, which sets allocations to guide non-point source controls and waste load allocations to establish effluent limits in NPDES permits.

EPA administers the NPDES and TMDL programs for waters in Indian country. Therefore, application of narrative criteria would be part of the consultation for issuance of the NPDES permits if the EPA determines that the action may affect ESA-listed threatened or endangered species. Given the plain language of the narrative criteria to provide water quality characteristics that avoid adverse effects to listed species and the requirement for the EPA to consult with the Services on EPA-issued permits as described above, the EPA has determined that approval of the narrative criteria is **not likely to adversely affect** chinook salmon, chinook salmon critical habitat, chum salmon, steelhead, bocaccio, yelloweye rockfish, bocaccio and yelloweye rockfish critical habitat, humpback whale, SRKW, SRKW critical habitat, marbled murrelet, bull trout, and bull trout critical habitat.

5.10 Critical Habitat Assessments

Section 7 of the ESA and its implementing regulations at 50 CFR Part 402 require the action agency to determine whether the action, i.e., EPA's approval of the Swinomish Tribe water quality criteria, is likely to "destroy or adversely modify the designated critical habitat of the listed species." The ESA consultation regulations define the statutory term "destruction or adverse modification" of critical habitat to mean:

"A direct or indirect alteration that appreciably diminishes the value of critical habitat for both the survival and recovery of a listed species. Such alterations include, but are not limited to, alterations adversely modifying any of those physical or biological features that were the basis for determining the habitat to be critical."

The critical habitat assessment in this section evaluates physical or biological features essential to the conservation of the ESA listed species evaluated in this BE. This is defined in 50 CFR §402.02 as:

"The features that occur in specific areas and that are essential to support the life-history needs of the species, including but not limited to, water characteristics, soil type, geological features, sites, prey, vegetation, symbiotic species, or other features. A feature may be a single habitat characteristic, or a

more complex combination of habitat characteristics. Features may include habitat characteristics that support ephemeral or dynamic habitat conditions. Features may also be expressed in terms relating to principles of conservation biology, such as patch size, distribution distances, and connectivity."

Maps and descriptions of the designated critical habitat for the ESA listed species assessed in this BE are presented in section 3 of this BE. With the possible exceptions of water characteristics and prey, approval of the Swinomish Tribe water quality criteria are expected to have no effect on the remaining features of critical habitat described in 50 CFR §402.02. Water characteristics, which include the effects of criteria chemicals at the criteria concentrations within the action area, and the effects of chemicals at criteria concentrations on prey of ESA listed species, are evaluated below.

5.10.1 Analysis of effects of criteria for toxic pollutants on critical habitat for Chinook salmon, bull trout, and killer whale

The action area as defined in section 2.1 of this BE does not overlap with the Services' designated critical habitat for chum salmon, steelhead, humpback whale, and marbled murrelet. Therefore, EPA's proposed action to approve the Swinomish Tribe water quality criteria will have **no effect on the critical habitat of humpback whale, chum salmon, steelhead and marbled murrelet.**

The marine portions of the action area outside of the Swinomish Reservation boundary include critical habitat for Chinook, bocaccio, yelloweye rockfish, SRKW, and bull trout. Habitat requirements of several of the listed species may limit or possibly preclude their use of the designated critical habitat within the action area. As discussed in section 3, SRKW critical habitat occurs only in waters ≥ 6.1 meters (20 feet) in depth. Orca have not been observed within the action area. The use of critical habitat for the Puget Sound/Georgia Basin DPS for yelloweye rockfish is limited to waters deeper than 15 meters (49 feet).

The Swinomish reservation lies within the action area and includes 27 miles (43 km) of marine shoreline habitat (http://www.npaihb.org/member-tribes/swinomish-tribe/). The action area overlaps with only a very small proportion of the designated critical habitat for SRKW, Chinook salmon, bull trout, bocaccio and yelloweye rockfish. Of this, the 11 mile long (18 km) Swinomish Channel is dredged for navigation use, most recently in 2018. The surface waters, including portions the Swinomish Channel, that lie within the Reservation boundary are not part of the designated critical habitat for the ESA listed species evaluated in this BE. The surface water area which lies within the action area but outside the exterior boundary of the Tribe's regulated surface waters consists of about 515 acres (about 0.8 square mile).

For SRKW, EPA determined that its action is not likely to adversely affect the species for all criteria as summarized in Table 3 (section 1.6.1) and described in section 5.1.2, including effects of exposure to waterborne chemicals and the quality and availability of prey. Similarly, EPA's action is not likely to adversely affect PCEs related to water quality and prey quality and availability, and therefore is **not likely to adversely affect critical habitat of SRKW** in the action area.

Evaluation of criteria effects on prey of ESA listed species was evaluated in the indirect effects sections of this BE. With the exception of the acute and chronic criteria for hexavalent chromium, the Swinomish Tribe's water quality criteria for marine waters were determined not likely to adversely affect prey availability for the ESA listed fish species (Table 156, section 5.8.20). With respect to prey quality and bioaccumulation considerations, arsenic (all fish species) and possibly selenium (Chinook salmon and bull trout) were the only chemicals for which exposure of prey at chronic criteria levels may result in

bioaccumulation in the listed fish species to levels that may have adverse effects. Although exposure of prey to hexavalent chromium, arsenic, or selenium at criteria levels may result in adverse effects to critical habitat features associated with abundance and quality of prey species, water quality criteria are otherwise not likely to adversely affect the prey of Chinook salmon, bull trout, bocaccio, and yelloweye rockfish.

Marine water quality criteria for four chemicals have LAA determinations for any of the ESA listed fish species in marine waters as a result of waterborne exposure, including the acute chlorine criterion, the chronic hydrogen sulfide criterion, the acute and chronic nonylphenol criteria, and the acute pentachlorophenol criterion (Table 3). Since exposure to these chemicals at criteria level is likely to adversely affect the listed fish species, critical habitat features related to water quality would also be adversely affected.

Based on the species and prey assessments described above, if chemicals reach criteria levels in critical habitat waters, EPA approval of the chronic criteria for selenium is likely to adversely affect critical habitat of Chinook salmon and bull trout. Additionally, approval of the following criteria is likely to adversely affect critical habitat of Chinook salmon, bull trout, bocaccio, and yelloweye rockfish if chemicals reach criteria levels: chronic arsenic, acute chlorine, acute and chronic hexavalent chromium, chronic hydrogen sulfide, acute and chronic nonylphenol, and acute pentachlorophenol. EPA approval of the remaining criteria is not likely to adversely affect critical habitat of Chinook salmon, bull trout, bocaccio, and yelloweye rockfish.

However, the following additional factors are provided for consideration regarding these criteria and critical habitat for the fish species. Hydrogen sulfide concentrations in oxygenated marine waters are expected to be low, and the half-life of hydrogen sulfide in marine waters ranges between several minutes and several hours, depending on water temperature, dissolved oxygen concentration, pH and the initial hydrogen sulfide concentration (Siang, Tahir, Malek, & Isa, 2017). Given the well oxygenated status of marine waters in the action area (Figure 17) and tidal movements in the action area, any adverse effects of the hydrogen sulfide chronic marine criterion would are expected to be insignificant. It is not likely that one would be able to meaningfully measure, detect, or evaluate hydrogen sulfide impacts on any critical habitat within the action area, which aligns with the definition of insignificant effects within both NMFS and USFWS ESA guidance.

Chlorine in action area marine waters would be present as part of wastewater treatment plant discharges to which chlorine is added as a disinfectant (see sections 4.2.2 and 4.2.3). Degradation rates of chlorine species in natural waters are generally rapid, ranging between seconds and hours (see section 5.5.1.4.1). Three waste water treatment plants discharge to the action area (section 4.2.2) under NPDES permits. Two of these discharge to Reservation waters and are administered by EPA, and are therefore subject to ESA consultation. Given the localized sources of chlorine and the transient nature of chlorine species in marine waters, effects of chlorine to water quality-related feature of critical habitat are likely to be small.

The pentachlorophenol marine acute criterion LAA determination was not based on effects to any of the ESA listed fish species within the action area. Instead it was based on 96 hour acute toxicity to Pacific herring (*Clupea pallasii*) of 25.3 µg/L (Vigers et al., 1978), which when divided by 2.27 resulted in an acute threshold toxicity value of 11.1 µg/L, lower than the marine acute pentachlorophenol criterion of 13 µg/L. The Pacific herring 96 hour LC50 is the only one of 17 96-hour LC50s for marine species in the

EPA (USEPA, 1986b) pentachlorophenol criteria document with a calculated acute threshold toxicity value lower than the 13 µg/L marine acute criterion. Given the available toxicity data showing limited adverse effects on marine species of pentachlorophenol at the acute criterion concentration combined with the small size of the action area within the area of the critical habitats for the ESA listed species, the effects of the Swinomish Tribe water quality criteria on critical habitat are likely to be insignificant. One would not be able to meaningfully measure, detect, or evaluate pentachlorophenol impacts on any critical habitat within the action area.

Similarly to the pentachlorophenol marine acute criterion LAA determination, the nonylphenol marine acute criterion LAA call was not based on effects to any of the ESA listed fish species within the action area. Instead it was based on the 5th percentile of a SSD of eight 96 hour LC50 values for marine fish, five of which were found in a review of the literature published since the EPA (USEPA, 2005a) nonylphenol criteria document was issued. The 5th percentile of the fitted SSD (12.18 µg/L) divided by 2.27 resulted in a calculated acute toxicity threshold value of 5.37 µg/L, lower than the marine acute nonylphenol criterion of 7 µg/L. The same considerations apply to the chronic criterion, which was derived from the acute criterion. The nonylphenol chronic effects assessment concentration (0.6614 µg/L) is lower than the marine chronic nonylphenol criterion (1.0 µg/L). Given the available toxicity data showing limited adverse effects on marine species of nonylphenol at the acute criterion concentration combined with the small size of the action area within the area of the critical habitats for the ESA listed species, it is likely that the effects of the Swinomish Tribe water quality criteria on critical habitat are insignificant. One would not be able to meaningfully measure, detect, or evaluate nonylphenol impacts on any critical habitat within the action area.

In general, as described above, the extent of critical habitat within the action area is small, and if any of the criteria chemicals were to reach criteria levels in reservation waters, the effects to critical habitat for any of the species addressed in this consultation would be spatially limited.

5.11 Cumulative Effects

This section provides an overview of the potential cumulative effects on listed species and their critical habitats that are related to future, non-federal (i.e., state, tribal, municipal, or private) actions reasonably certain to occur in the action area. This section seeks to address changes caused by multiple activities in a region causing multiple stressors that directly and indirectly affect multiple interconnected aspects of an ecosystem. The species that utilize the action area are susceptible to impacts within and throughout the greater Puget Sound.

For this BE, impacts or stressors are discussed in relation to the species for which the effects analysis concluded “neither insignificant nor discountable and are therefore measurable and potentially adverse.” These species include the salmonids, rockfish, bull trout, humpback whale, southern resident killer whale, and marbled murrelet (section 1.4). Cumulative impacts within the action area include impacts from the following categories: land and marine environment use changes; permitted discharges to surface water; commercial, recreational, and subsistence fish harvest; and, global climate change. All these activities may have an incremental or compounding effect when experienced together and are known to often result in impacts to the listed species of interest in this BE.

5.11.1 Land and marine environment use change

Land use change from urban and agricultural development is the most urgent threat to the action area for this BE (PSRC.org/puget-sound-trends). The Puget Sound region added 324,000 people between 2012-2017 and expected continued population growth in the Puget Sound region has and will continue to result in increased urban and agricultural development that degrades functional habitat even as regional and local programs seek to produce long-term, environmentally sustainable benefits. Pressure to accommodate a growing population is leading to re-designation of lands through the expansion of urban growth boundaries into previously undeveloped areas. The Puget Sound region has experienced widespread depletion, fragmentation, and modification of marine and freshwater habitat at an increasing rate over the last 25 years. Individually managed projects supporting this unprecedented growth have an incremental or compounding effect when experienced together and result in significant direct and indirect impacts to listed species considered in this BE.

Existing and future human land and marine environment use impacts include degraded floodplain and in-river channel structure; degraded estuarine conditions and loss of estuarine habitat; degradation of riparian habitat and loss of in-river large woody debris; excessive sedimentation (especially in spawning gravels); degraded nearshore conditions; reduction of prey species; impaired passage for migrating fish; and altered hydrology (Thomas P. Good et al., 2005; NMFS, 2016b). In addition, increased human activity in the marine environment increases key stressors for many species and especially marbled murrelet, humpback whale and southern resident killer whale. For example, increased (Williams, Lusseau, & Hammond, 2009); physical disturbance including entrapment in gear and displacement from key habitats; increases the rate of ship strikes; increases general harassment of individuals and groups of species (Williams, Lusseau, & Hammond, 2006) and, increases pollution input to water and air (Canada, 2014; James V. Carretta et al., 2014; Fretwell, 2017; Williams et al., 2006, 2009).

5.11.2 Permitted discharges

Toxic pollutants are considered a priority threat in Puget Sound and changes in land and marine environment use affects the amount of pollution introduced into freshwater, estuarine, and marine habitats from private (e.g., industrial) and public (e.g., municipal wastewater) sources. Although the NPDES permitting program is overseen by EPA at the federal level, many states, including Washington have implementation responsibilities over the permitting process to limit discharges of wastewater to surface waters within the jurisdictions of the respective states. Effects resulting from exposures to permitted discharges within Washington state fresh and marine waters are considered cumulative.

Overall water and sediment quality in the Puget Sound is degraded (Long, Dutch, Aasen, Welch, & Hameedi, 2005), eopugetsound.org)(L. Johnson et al., 2010). Elevated concentrations of pollutants in the Puget Sound and elsewhere have been linked to elevated concentrations in salmon and in SRKW (B. E. Hickie, Ross, Macdonald, & Ford, 2007; M. M. Krahn et al., 2007; Krahn et al., 2009; Lachmuth et al., 2010). Once in the environment, many pollutants accumulate in tissues, and some biomagnify up the food chain, reaching high levels in long-lived apex predators like SRKWs. Persistent bioaccumulative toxins tend to accumulate in industrial and urbanized habitats near their sources. Maternal transfer of persistent and bioaccumulative pollutants 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 (M. M. Krahn et al., 2007; Krahn et al., 2009).

Potential discharges throughout the Puget Sound region will continue to contribute to cumulative effects for both prey species (salmonids, smaller fish, and invertebrates) and predatory species (salmonids, SRKW, Humpback Whales). By consuming contaminated fish or invertebrates, marbled murrelet may in turn be exposed to and accumulate contamination in their bodies (Fry, 1995). The accumulation of pollutants such as dioxins/furans, PCBs, and mercury (among many others) in birds can cause toxic effects resulting in reduced growth, reproduction, and survival (e.g., (Augspurger et al., 2008; Burgess & Meyer, 2008; Henning, Robinson, McKay, Sullivan, & Bruckert, 2003; Scheuhammer, 1987).

The permits within the action area are described above in section 4.2.2 and listed in Table 15.

5.11.3 Commercial, recreational, and subsistence fishing

Commercial, recreational, and subsistence fishing, which are managed by state agencies and tribes, are important activities for many people in the Pacific Northwest. For example, as of 2006, the fishing industry (excluding tribal harvest) in Washington State provided more than 16,000 jobs and contributed $540 million to personal income (TCW, 2008). These jobs were mostly related to recreational fisheries. In 2006, 109.4 million pounds of non-tribal commercial fish and shellfish were harvested in Washington. Salmon and trout are the most frequently targeted saltwater and freshwater species, respectively among anglers. Fishing is also vitally important to NW Native American communities in terms of diet, personal income, and cultural identity.

Harvest of salmon from all types of fisheries has had a substantial adverse effect on salmon populations, particularly after the introduction of high-yield harvest methods such as gillnets, fish wheels, or horse seining in the late 19th century (Smithsonian, 2018). In recent decades, harvest management has substantially reduced the impact of harvest on salmon populations (IMST, 2000). Some populations are improving, while others are not. These limitations are due in small part to continuing harvest and in large part to the lack of high-quality spawning and rearing habitat as noted in sections above. Commercial and recreational fishing occurs within and around the action area.

5.11.4 Climate change

Average sea surface temperature are expected to increase by 4.9° ± 1.3°F (2.7° ± 0.7°C) in 2100 compared to temperatures in the 20th century (oceans have absorbed 93% of the excess heat from human-induced warming), and due to the absorption of carbon dioxide, surface ocean acidity is expected to increase 100-150% (conditions that are unprecedented in the last 66 million years; USGCRP 2018). Concomitant increases in the extent and duration of hypoxic conditions will continue, driven by increases in sea surface temperatures, rising sea levels, and changing patterns of precipitation, winds, nutrients, and ocean circulation. The Puget Sound Science Review describes the pressures that climate change exerts in Puget Sound as “six general classes of processes that affect its structure and functioning: (1) water cycle changes; (2) weather/temperature change, (3) ocean thermal expansion/melting of land ice; (4) large and local scale atmospheric forcing; (5) ocean acidification; and (6) ultraviolet irradiance. Each in turn contributes to changes in ecosystem states. More than one

pressure can contribute to a given state change; similarly, many system-level impacts are driven by multiple state changes (eopugetsound.org)."

For example, there is potential to cause significant adverse changes to freshwater and marine fish habitats in the action area, including but not limited to reductions in water quantity and quality and altered prey resources. Reduced snowpack and increased glacier and snow melt (as a result of extended warm periods) will eventually lead to less available water and warmer streams; this will be exacerbated by other stressors like water diversion. Warming waters are likely to reduce fish fitness by accelerating the development of embryos, the timing of hatching, and emergence of juveniles. Accelerating these processes too much could result in asynchrony between the emergence of fish and the availability of prey items in the marine environment, which would reduce feeding, growth, and survival (NMFS, 2016h). Warming oceans tend to be more acidic and less productive with can lead to a decline in food availability (NMFS, 2015b). Although specific effects to listed rockfish and bocaccio are as yet unknown (NMFS, 2016a), the temperatures of marine waters of Puget Sound have increased over the last several decades, with additional increases expected in the future (Mauger et al., 2015), and rockfish productivity has been linked to colder-phase conditions of the Pacific Decadal Oscillation (PDO). Such supportive conditions have been predicted to decline as a result of changes in atmosphere-ocean forcing linked to climate change (USGCRP, 2018). Increased temperatures can result in more widespread invasive species, increased disease rates for native species, hypoxia, harmful algal blooms increasing extent, duration, and toxicity, and other changes to the food web that can impact the overall productivity of the marine ecosystem. Increased coastal acidification as anthropogenic sources continue to increase will result in the crossing of thresholds to adversely affect calcification for many organisms in Puget Sound (Pacella et al., 2018), which can in turn alter the distribution of prey for a marine life, including rockfish (NMFS, 2016).

Salmon physiology and behavior are adapted to local environmental conditions and long-term climate change is likely to influence the species long-term success (NMFS, 2015b). Possible effects on salmon from climate change include increase in mortality from heat stress, changes in growth and development rates, and disease resistance. Behavioral effects may include shifts in the timing of important life history events such as adult and juvenile migration, spawn timing, and fry emergence. 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; Bruce E Rieman et al., 2007). Nearly all anadromous salmonids on the west coast are directly and indirectly (via prey) vulnerable to projected changes in sea surface temperature and ocean acidification as well as summer stream temperatures, and in Puget Sound loss of snowpack and the switch to rain-dominated hydrology in particular will play a role in vulnerability (Crozier et al. 2019).

The fitness of marbled murrelet depends on their ability to obtain prey of sufficient quality and quantity, and the availability of specific prey items is driven by oceanic conditions (Becker et al., 2007). Cooler conditions favor high productivity along the eastern boundary upwelling system off the US West Coast, which in turn generates a greater biomass of murrelet prey items (e.g., euphausiids). Warming ocean temperatures could reduce productivity in the eastern Pacific Ocean, which would likely reduce marbled murrelet fitness. Pacific herring are an important murrelet prey species. Herring have a narrow temperature tolerance and increases in water temperature will result in mortality and reduced reproduction. Shifting conditions in the polar regions of the Earth will continue to result in the melting of

sea ice and an increasing elevation of sea levels. Other important murrelet prey species, the sand lance and surf smelt are beach spawners and sea level rise will likely result a loss of spawning habitat when tideline shift reducing beach habitat in the action area. This condition is already exacerbated by the significant area of shoreline modified consisting of bulk heads, sea walls and other hardscaping in Puget Sound. As residents attempt to combat sea level rise these shoreline modifications will only increase affecting nearshore hydrodynamics, sediment transport and beach nourishment. If current climate change models and predictions for Pacific Northwest aquatic habitats are relatively accurate, the action area and species evaluated in this BE will be affected.

5.12 Effects from Interrelated/Interdependent actions

The ESA section 7 implementing regulations (50 CFR 402.02 and 50 CFR 402.33) as updated on September 26, 2019 define effects of a federal action in part as: "Effects of the action are all consequences to listed species or critical habitat that are caused by the proposed action, including the consequences of other activities that are caused by the proposed action. A consequence is caused by the proposed action if it would not occur but for the proposed action and it is reasonably certain to occur." This definition of "effects of the action" simplified the previous definition by collapsing the terms "direct, "indirect," interrelated," and "interdependent" into the term "consequences" and by applying a two-part test of "but for" and "reasonably certain to occur."

The historical definitions (USFWS & NMFS, 1998) of interrelated actions and interdependent actions were as follows:

Interrelated actions - actions that are part of a larger action and depend on the larger action for their justification.

Interdependent actions - actions having no independent utility apart from the proposed action.

There are no interdependent or interrelated actions expected as a result of EPA approval of the proposed Swinomish Tribe water quality criteria for toxic pollutants. Thus EPA's approval of the Swinomish Tribe water quality for toxic pollutants has no consequences for listed species and critical habitat from interrelated/interdependent actions.

6 Essential Fish Habitat Assessment

6.1 Introduction

The Magnuson Stevens Fishery Conservation and Management Act (MSFCMA), as amended by the Sustainable Fisheries Act of 1996 (Public Law 104-267), through the Essential Fish Habitat (EFH) provision, protects the waters and substrate necessary for the spawning, breeding, feeding, or growth to maturity of certain commercially managed fisheries species (16 U.S.C. § 1802(10)). Federal agencies are required to consult with NMFS on proposed actions authorized, funded, or undertaken by the agency that may adversely affect EFH (section 305(b)(2)). NMFS is required to provide conservation recommendations for any federal activity that would adversely affect EFH (section 305(b)(4)(A)). Adverse effect means any impact that reduces the quality and/or quantity of EFH, and may include direct, indirect, site-specific or habitat-wide impacts, including individual, cumulative, or synergistic consequences of actions (50 CFR § 600.810).

Pursuant to the MSFCMA, the Pacific Fisheries Management Council (PFMC) has designated EFH for federally managed species within the waters of Washington, Oregon, and California. The waters of the greater Puget Sound are designated EFH for coastal pelagic, Pacific coast groundfish, and Pacific coast salmon species (PFMC, 2016 a-c respectively). The Pacific groundfish fishery includes 82 species, approximately two thirds of which occur in Washington State. The coastal pelagic fishery includes four fin fishes (Pacific sardine (*Sardinops sagax*), Pacific [chub] mackerel, northern anchovy (*Engraulis mordax*), and jack mackerel) and the invertebrate market squid (*Doryteuthis opalescens*). The Pacific salmon fishery includes Chinook, coho, and Puget Sound pink salmon (*O. gorbuscha*).

In addition to EFH designations, areas called Habitat Areas of Particular Concern (HAPC) are also designated by the regional Fishery Management Councils (FMCs). Designated HAPC are discrete subsets of EFH that provide extremely important ecological functions or are especially vulnerable to degradation (50 CFR §§ 600.805-600.815). Regional FMCs may designate a specific habitat area as an HAPC based on one or more of the following reasons: (1) importance of the ecological function provided by the habitat; (2) the extent to which the habitat is sensitive to human-induced environmental degradation; (3) whether, and to what extent, development activities are, or will be, stressing the habitat type; and (4) rarity of the habitat type (67 FR 2343-2383). Categorization as a HAPC does not confer additional protection or restriction to the designated area.

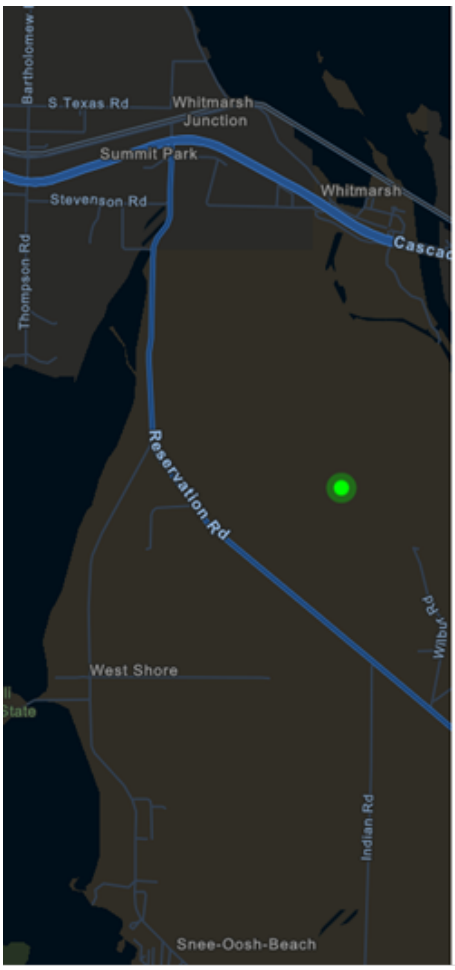
The objective of this assessment is to determine whether the proposed action “may adversely affect” or “will not adversely affect” designated EFH for relevant federally managed commercial species within the action area. The assessment focused on the effects of EPA’s proposed approval under the CWA section 303(c) of the Swinomish Tribe’s water quality standards for the protection of aquatic life. Refer to section 1.2 of the BE for a summary of EPA’s action subject to the consultation.

EPA considered the direct and indirect effects to listed species from exposure to the aquatic life criteria within the action area, which is the Swinomish Reservation and the full width of the Swinomish Channel on the east side of the reservation. For purposes of the BE analysis and this EFH assessment, EPA incorporated a 400-foot “buffer” area around the rest of the Reservation boundary. Located in the northwestern Washington near La Conner, the Reservation includes approximately 7,450 acres of

uplands and approximately 3,000 acres are tidelands within Skagit Bay and Padilla Bay (see Figures 1 and 2, section 2.1 of the BE).

EPA's CWA action applies only to the surface waters under the jurisdiction of the Swinomish Tribe. The Tribe's water quality standards apply to pollutant levels in these waters. Outside of these waters, Washington state standards apply (WAC 173-201A). Waters entering the Reservation must meet the Tribe's water quality standards at the reservation boundary, and waters leaving the reservation must meet Washington's water quality standards at the reservation boundary.

Below are the reports generated from the online EFH mapping tool.[31] Based on the two hydraulic unit codes (HUC) that overlay the Swinomish Reservation land area for the salmon EFH (the Strait of Georgia and the Puget Sound), Chinook Salmon, Coho Salmon, Puget Sound Pink Salmon (in all lifestages) are found in both HUCs.



Query Results

Degrees, Minutes, Seconds: Latitude = 48° 26' 13" N, Longitude = 123° 28' 32" W
Decimal Degrees: Latitude = 48.437, Longitude = -122.525

The query location intersects with spatial data representing EFH and/or HAPCs for the following species/management units.

EFH

No Essential Fish Habitats (EFH) were identified at the report location.

Salmon EFH

Link	HUC Name	Species/Management Unit	Lifestage(s) Found at Location	Management Council	FMP
	Strait Of Georgia	Chinook Salmon, Coho Salmon, Puget Sound Pink Salmon	All	Pacific	Pacific Coast Salmon Plan

HAPCs

No Habitat Areas of Particular Concern (HAPC) were identified at the report location.

EFH Areas Protected from Fishing

No EFH Areas Protected from Fishing (EFHA) were identified at the report location.

[31] https://www.habitat.noaa.gov/apps/efhmapper/?page=page_4

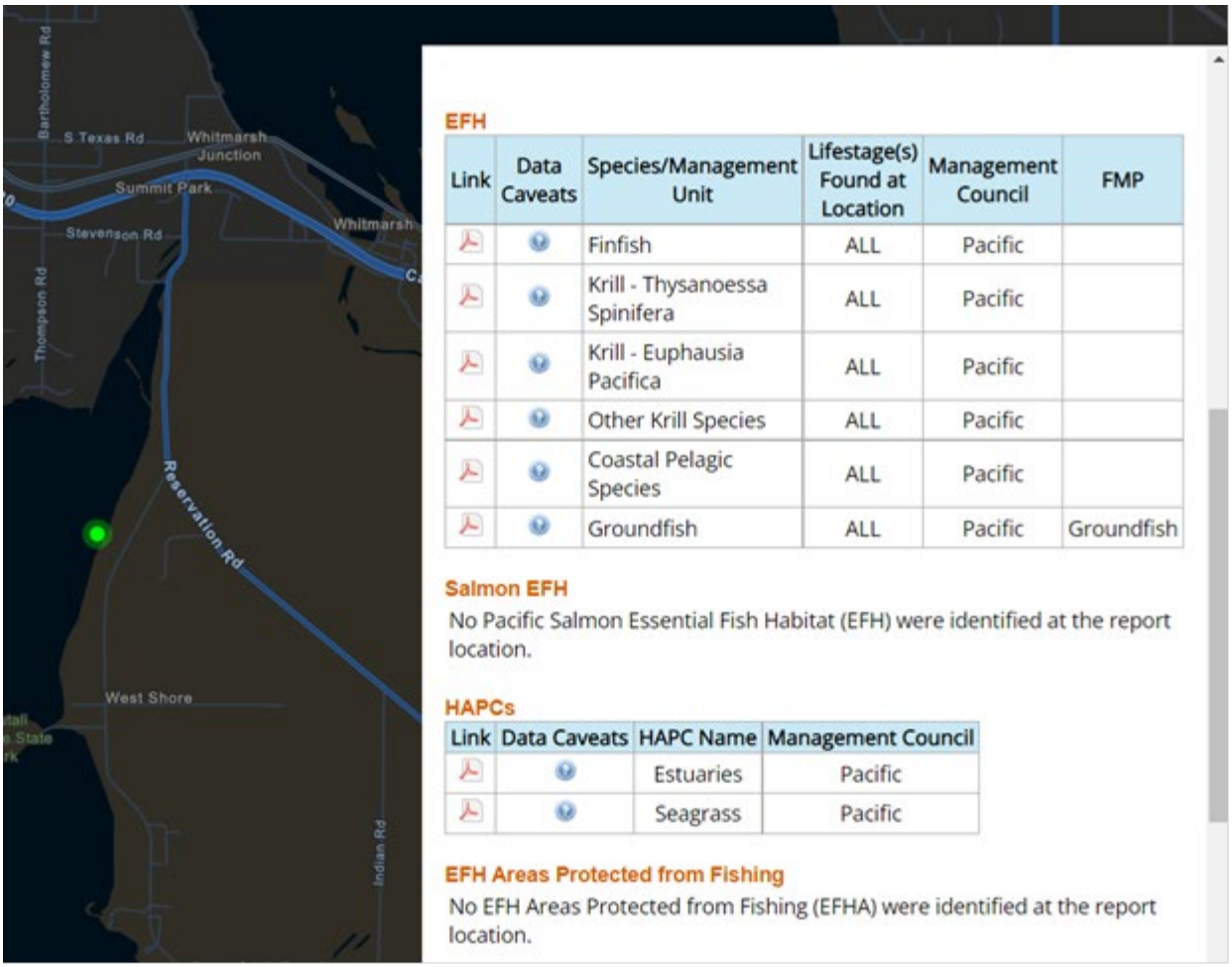


Figure 39: The HAPCs generated by the EFH Mapper Tool (seagrass and estuaries). Both are listed in the Pacific Coast Groundfish Fishery Management Plan (FMP).[32]

6.2 Essential Fish Habitat Designations

6.2.1 Coastal pelagic species

The Pacific Council's Coastal Pelagic Species FMP specifies a management framework for northern anchovy – central and northern subpopulations (*Engraulis mordax*), Pacific sardine, Pacific (chub) mackerel (*Scomber japonicus*), and jack mackerel (*Trachurus symmetricus*). In October 2006, the Coastal Pelagic Species FMP was amended to include all krill species. In July 2009, Amendment 12 to the Coastal Pelagic Species FMP prohibited the harvest of krill within California, Oregon, and Washington waters (74 FR 33372). No krill harvest will occur as a result of the EPA's Proposed Action. EFH for non-krill coastal pelagic species addresses four pelagic species that are treated as a single species complex because of similarities in life histories and habitat requirements: Northern anchovy, Pacific sardine, Pacific (chub) mackerel, and jack mackerel. Two of these coastal pelagic species are known to occur in the greater Puget Sound: northern anchovy and Pacific mackerel. The definition for coastalpelagic species EFH is

[32] https://www.pcouncil.org/documents/2016/08/pacific-coast-groundfish-fishery-management-plan.pdf/#page=117

based on the geographic range and in-water temperatures where these species are present during a particular life stage (67 FR 2343-2383). EFH for these species includes all estuarine and marine waters above the thermocline where sea surface temperatures range from 50 to 68 °F. These boundaries include all marine waters located within the action area.

Coastal pelagic species have value to commercial Pacific fisheries, and are also important as food for other fish, marine mammals, and birds (63 FR 13833). Coastal pelagic species are considered sensitive to overfishing, loss of habitat, reduction in water and sediment quality, and changes in marine hydrology, including entrainment through water intakes.

The general descriptions of northern anchovy and Pacific mackerel are provided in the FMP (PFMC, 2016a) and were reviewed for information on designated EFH pertinent to consideration of effects from the proposed action. Northern anchovy are small, short-lived fish that are typically found in schools near the surface. They eat phytoplankton and zooplankton and spawn year-round with peaks from February to April. They frequently occur in estuaries through the Pacific coast and may be associated with the neritic portion of the nearshore environment within the current-swept waterways of the Puget Sound Basin (Penttila, 2007). All life stages are preyed on by a variety of predators, including salmon and numerous fishes.

6.2.2 Pacific coast groundfish

Pacific coast groundfish species are considered sensitive to over-fishing, the loss of habitat, and water andsediment quality (PFMC, 2020).[33] The groundfish EFH consists of the aquatic habitat necessary to allow for groundfish production to support long-term sustainable fisheries for groundfish and for groundfish contributions to a healthy ecosystem. The PFMC (2020) identifies the overall area designated as groundfish EFH for all species covered in the FMP as all waters and substrate within "depths less than or equal to 3,500 m [~ 11,500 feet] to MHHW level or the upriver extent of saltwater intrusion, defined as upstream and landward to where ocean-derived salts measure less than 0.5 parts per thousand during the period of average annual low flow." Furthermore, the PFMC has also designated EFH for each individual groundfish species by life stage.

The management unit in the Pacific Coast Groundfish FMP includes over 90 groundfish species. Of these, 37 were identified through the analysis of the Habitat Use Relational Database as having EFH designated in the greater Puget Sound. Marine species that are likely to occur within the aquatic environment of the action area are expected to be the same as those that typically occur within the greater Puget Sound.

Based on the analysis, the primary habitats designated as EFH for groundfish include: the epipelagic zoneof the water column, including macrophyte canopies and "drift algae"; unconsolidated sediments consisting of mud, sand, or mixed mud/sand; hard-bottom habitats composed of boulder, bedrock, cobble,gravel, or mixed gravel/cobble; mixed sediments composed of sand and rocks; and vegetated bottoms consisting of algal beds, macrophytes, or rooted vascular plants.

6.2.3 Pacific coast salmon

The Pacific salmon management unit includes Chinook, coho, and pink salmon. The EFH designation for the Pacific salmon fishery in estuarine and marine environments in the state of Washington extends

[33] https://www.pcouncil.org/documents/2016/08/pacific-coast-groundfish-fishery-management-plan.pdf/

from nearshore and tidal submerged environments within state territorial waters out to the full extent of the exclusive economic zone (200 miles) offshore. In addition to marine and estuarine waters, salmon species have a defined freshwater EFH, which includes all lakes, streams, ponds, rivers, wetlands,and other bodies of water that have been historically accessible to salmon.

Pacific salmon EFH is primarily affected by the loss of suitable spawning habitat, barriers to fish migration (habitat access), reduction in water and sediment quality, changes in estuarine hydrology, and decreases in prey food source. The most abundant Puget Sound forage fish species for salmonids include Pacific herring, surf smelt, and Pacific sand lance.

6.3 Habitat Areas of Particular Concern

Designated HAPCs are regarded as essential for protection of federally managed species. EFH guidelines published in Federal regulations (50 CFR 600.815(a)(8)) identify HAPCs as types or areas of habitat within EFH that are identified based on one or more of the following considerations (PFMC, 2020):

- The importance of the ecological function provided by the habitat;
- The extent to which the habitat is sensitive to human-induced environmental degradation;
- Whether, and to what extent, development activities are or will be stressing the habitat type; and
- The rarity of the habitat type.

HAPCs have not been formally designated for coastal pelagic species. There are four HAPCs designated for groundfish that include (PFMC, 2020):

1. Estuaries – Protected nearshore areas such as bays, sounds, inlets, and river mouths, influenced by ocean and freshwater. Estuaries tend to be shallow, protected, nutrient-rich, and are biologically productive, providing important habitat for marine organisms, including groundfish. The inland extent of the estuary HAPC is defined as MHHW, or the upriver extent of saltwater intrusion, defined as upstream and landward to where ocean-derived salts measure less than 0.5 ppt during the period of average annual low flow.

2. Canopy Kelp – Kelp forests are of primary importance to the ecosystem and serve as important groundfish habitat. Kelp forest communities are found relatively close to shore along the open coast.

3. Seagrass – Seagrass species found on the West Coast of the U.S. include eelgrass species (*Zostera* spp.), widgeongrass (*Ruppia maritima*), and surfgrass (*Phyllospadix* spp.). These grasses form dense beds of leafy shoots year-round in the lower intertidal and subtidal areas. Eelgrass is found on soft-bottom substrates in intertidal and shallow subtidal areas of estuaries.

4. Rocky Reefs – Rocky habitats are generally categorized as either nearshore or offshore in reference to the proximity of the habitat to the coastline. Rocky habitat may be composed of bedrock, boulders, or smaller rocks, such as cobble and gravel. Hard substrates are one of the least abundant benthic habitats, yet they are among the most important habitats for groundfish.

6.4 EFH near the Swinomish Reservation

Coastal pelagic EFH is present in the waters of Swinomish Channel, Skagit Bay and Padilla Bay. The mix of sand and silt bottom substrate provides EFH for groundfish species (PFMC, 2020). Additionally, Pacific

coast salmon EFH for various life stages of Chinook, coho, and Puget Sound pink salmon also exist in these waters. The Chinook, coho, and pink salmon could utilize this area as migration corridor to spawning and rearing streams located in various areas in Puget Sound (WDFW, 2016).

The Tribe's water quality standards are used to control pollutant levels in tribal waters and based on EPA's analysis of exposure levels contained in the BE, EPA's proposed CWA action to approve the Swinomish Tribe's water quality standards for the protection of aquatic life *will not adversely affect EFH* near the Swinomish Reservation.

7 References

Abbasi, S., & Soni, R. (1991). Evaluation of water quality criteria for four common pesticides on the basis of computer-aided studies. *Indian Journal of Environmental Health, 33*(1), 22-24.

ACOE, U.-., USEPA, Ecology, DNR, W., ODEQ, IDEQ, . . . USFWS. (2005). *Sediment Evaluation Framework for the Pacific Northwest*. ACOE Seattle District, Portland District, Walla Walla District and Northwestern Division: U.S. Army Corps of Engineers

Adams, N. W. H., & Kramer, J. R. (1999). Silver speciation in wastewater effluent, surface waters, and pore waters. *Environmental Toxicology and Chemistry, 18*(12), 2667-2673. Retrieved from http://dx.doi.org/10.1002/etc.5620181203

Adelman, I. R., & Smith, L. L. (1970). Effect of Hydrogen Sulfide on Northern Pike Eggs and Sac Fry. *Transactions of the American Fisheries Society, 99*(3), 501-509. doi:10.1577/1548-8659(1970)99<501:Eohson>2.0.Co;2

Adema, D. (1978). Daphnia magna as a test animal in acute and chronic toxicity tests. *Hydrobiologia, 59*(2), 125-134.

Agra, A. R., & Soares, A. M. (2009). Effects of two insecticides on survival, growth and emergence of Chironomus riparius Meigen. *Bulletin of environmental contamination and toxicology, 82*(4), 501-504.

Aguilera, L., Marquetti, M. d. C., Navarro, A., & Bisset, J. (1995). Effects of three organophosphorus insecticides in the reproductive potential of Culex quinquefasciatus. *Mem. Inst. Oswaldo Cruz*, 411-413.

Ahearn, D. S., Viers, J. H., Mount, J. F., & Dahlgren, R. A. (2006). Priming the productivity pump: flood pulse driven trends in suspended algal biomass distribution across a restored floodplain. *Freshwater Biology, 51*, 1417-1433. Retrieved from https://doi.org/10.1111/j.1365-2427.2006.01580.x

Ahmed, M. K., Habibullah-Al-Mamun, M., Parvin, E., Akter, M. S., & Khan, M. S. (2013). Arsenic induced toxicity and histopathological changes in gill and liver tissue of freshwater fish, tilapia (Oreochromis mossambicus). *Exp Toxicol Pathol, 65*(6). doi:10.1016/j.etp.2013.01.003

Ainley, D. G., Allen, S. G., & Spear, L. B. (1995). Chapter 34. Offshore occurrence patterns of marbled murrelets in central California. In *Ecology and conservation of the Marbled Murrelet. USDA Forest Service General Technical Report PSW-152* (pp. 361-369). Albany, CA: Pacific Southwest Research Station, Forest Service, US Department of Agriculture.

Aldenberg, T., & Luttik, R. (2002). Extrapolation factors for tiny toxicity data sets from species sensitivity distributions with known standard deviation. In L. Posthuma, G. W. S. II, & T. P. Traas (Eds.), *Species Sensitivity Distributions in Ecotoxicology* (pp. p. 103 – 118). Boca Raton, FL: Lewis Publishers.

Allison, D. T., & Hermanutz, R. O. (1977). *Toxicity of diazinon to brook trout and fathead minnows*: Environmental Protection Agency, Office of Research and Development ….

Alsop, D. H., & Wood, C. M. (2000). Kinetic analysis of Zinc accumulations in the gills of juvenile rainbow trout: Effects of zinc acclimation and implications for Biotic Ligand Modeling. *Environmental Toxicology and Chemistry, 19*(7), 1911-1918.

Aluigi, M. G., Falugi, C., Mugno, M. G., Privitera, D., & Chiantore, M. (2010). Dose-dependent effects of chlorpyriphos, an organophosphate pesticide, on metamorphosis of the sea urchin, Paracentrotus lividus. *Ecotoxicology, 19*(3), 520-529.

Ambient aquatic life water quality criteria for chromium. (EPA 440/5-84-029; PB85-227478). (1985). Washington, D.C.: U.S. Environmental Protection Agency, Office of Water Regulations and Standards, Criteria and Standards Division Retrieved from https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=60000XZI.PDF

Amelung, M. (1982). Effects of Dissolved Iron Compounds on Developing Eggs and Larvae of Salmo gairdneri (Richardson). *Archiv fur Fischereiwissenscchaft, 32*(1-3), 77-87.

Anderson, S. L., & Norberg-King, T. J. (1991). Precision of short-term chronic toxicity tests in the real world. *Environmental Toxicology and Chemistry: An International Journal, 10*(2), 143-145.

Ankley, G. T., Dierkes, J. R., Jensen, D. A., & Peterson, G. S. (1991). Piperonyl butoxide as a tool in aquatic toxicological research with organophosphate insecticides. *Ecotoxicology and Environmental Safety, 21*(3), 266-274.

ANZECC, & ARMCANZ. (2000). *Australian and New Zealand Guidelines for Fresh and Marine Water Quality, Volume 1, The Guidelines (chapters 1–7).* Canberra, Australia: Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand

Arsenault, J. T., Fairchild, W. L., MacLatchy, D. L., Burridge, L., Haya, K., & Brown, S. B. (2004). Effects of water-borne 4-nonylphenol and 17beta-estradiol exposures during parr-smolt transformation on growth and plasma IGF-I of Atlantic salmon (Salmo salar L.). *Aquat Toxicol, 66*(3), 255-265. doi:10.1016/j.aquatox.2003.09.005

ASTM. (2003). Standard guide for conducting acute toxicity tests on test materials with fishes, macroinvertebrates, and amphibians. In *Annual book of ASTM Standards* (Vol. Vol. 11.05, pp. 179-200). West Conshohocken, PA: American Society for Testing and Materials.

ATSDR. (2016). *Toxicological Profile for Hydrogen Sulfide and Carbonyl Sulfide*. Atlanta, GA: Public Health Service; Agency for Toxic Substances and Disease Registry

Augspurger, T. P., Tillitt, D. E., Bursian, S. J., Fitzgerald, S. D., Hinton, D. E., & Di Giulio, R. T. (2008). Embryo toxicity of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin to the wood duck (*Aix sponsa*). *Arch Environ Contam Toxicol, 55*(4), 659-669. doi:10.1007/s00244-008-9198-2

Ayres, K. L., Booth, R. K., Hempelmann, J. A., Koski, K. L., Emmons, C. K., Baird, R. W., . . . Wasser, S. K. (2012). Distinguishing the impacts of inadequate prey and vessel traffic on an endangered killer whale (*Orcinus orca*) population. *PLoS ONE, 7*(6), e36842. doi:10.1371/journal.pone.0036842

Azizullah, A., Richter, P., & Häder, D.-P. (2011). Comparative toxicity of the pesticides carbofuran and malathion to the freshwater flagellate Euglena gracilis. *Ecotoxicology, 20*(6), 1442-1454.

Bagarinao, T., & Vetter, R. D. (1989). Sulfide tolerance and detoxification in shallow-water marine fishes. *Marine Biology, 103*(3), 291-302. doi:10.1007/bf00397262

Bagarinao, T., & Vetter, R. D. (1993). Sulphide tolerance and adaptation in the California killifish, Fundulus parvipinnis, a salt marsh resident. *Journal of Fish Biology, 42*(5), 729-748. doi:10.1111/j.1095-8649.1993.tb00381.x

Bailey, H. C., Deanovic, L., Reyes, E., Kimball, T., Larson, K., Cortright, K., . . . Hinton, D. E. (2000). Diazinon and chlorpyrifos in urban waterways in northern California, USA. *Environmental Toxicology and Chemistry: An International Journal, 19*(1), 82-87.

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. *Can J Zool, 83*, 257-267.

Baker, C. S. (1985). *The population structure and social organization of humpback whales (Megaptera novaeangliae) in the central and eastern North Pacific.* (Ph.D. dissertation). University of Hawaii, Honolulu, HI.

Baker, C. S., & Herman, L. M. (1985). Whales that go to extremes. *Nat Hist, 94*(10), 52-61.

Baker, C. S., Herman, L. M., Perry, A., Lawton, W. S., Straley, J. M., & Straley, J. H. (1985). Population characteristics and migration of summer and late-season humpback whales (*Megaptera novaeangliae*) in southeastern Alaska. *Mar Mam Sci, 1*(4), 304-323. Retrieved from http://www.dolphin-institute.org/our_research/whale_research/abstracts/1985Bakeretal.html

Baker, J., Bernard, D., Christensen, S., Sale, M., Freda, J., Heltcher, K., . . . Suter, G. (1990). *Biological effects of changes in surface water acid-base chemistry*. Retrieved from

Balcomb, K., & Nichols, G. (1978). Western North Atlantic humpback whales. *Rep Int Whal Commn, 28*, 159-164.

Banaee, M., Sureda, A., Mirvaghefi, A., & Ahmadi, K. (2011). Effects of diazinon on biochemical parameters of blood in rainbow trout (Oncorhynchus mykiss). *Pesticide biochemistry and physiology, 99*(1), 1-6.

Bargmann, G., Palsson, W., Burley, C., Friedel, D., & Tsou, T. (2011). *Final Environmental Impact Statement for the Puget Sound Rockfish Conservation Plan Including Preferred Range of Actions*. Olympia, WA: Washington Department of Fish and Wildlive Retrieved from https://wdfw.wa.gov/publications/00035

Barrett-Lennard, L. G., & Heise, K. A. (2007). The natural history and ecology of killer whales. In J. Estes (Ed.), *Whales, Whaling, and Ocean Ecosystems* (pp. 161-171).

Bathe, R., Sachsse, K., Ullmann, L., Hoermann, W., Zak, F., & Hess, R. (1975). *The evaluation of fish toxicity in the laboratory.* Paper presented at the Proceedings of the European Society of Toxicology.

Batley, G. E., Dam, R. A. v., Warne, M. S., Chapman, J. C., Fox, D. R., Hickey, C. W., & Stauber, J. L. (2018). *Technical Rationale for Changes to the Method for Deriving Australian and New Zealand Water Quality Guideline Values for Toxicants*. Canberra, ACT: Australian and New Zealand Governments and Australian State and Territory Governments

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.

Beamer, E., Henderson, R., & Wolf, K. (2004). *Bull Trout use of Swinomish Reservation Waters*. Retrieved from LaConner, WA: http://skagitcoop.org/wp-content/uploads/Report-of-Bull-Trout-Use-of-Swinomish-Reservation-Waters.pdf

Beamer, E. M., Zackey, W., Marks, D., Teel, D., Kuligowski, D., & Henderson, R. (2013). *Juvenile chinook salmon rearing in small non-natal streams draining into the Whidbey Basin*. Retrieved from LaConner, WA: http://skagitcoop.org/wp-content/uploads/EB2752_Beamer-et-al_2013.pdf

Beamer, E. M., Zackey, W. T., Marks, D., Teel, D., Kuligowski, D., & Henderson, R. (2013). *Juvenile Chinook salmon rearing in small non-natal streams draining into the Whidbey Basin*. Retrieved from LaConner, Washington: http://skagitcoop.org/wp-content/uploads/EB2752_Beamer-et-al_2013.pdf

Beamish, R. J., & Bouillon, D. R. (1993). Pacific salmon production trends in relation to climate. *Can J Fish Aquat Sci, 50*, 1002-1016.

Beamish, R. J., Sweeting, R. M., Lange, K. L., Noakes, D. J., Preikshot, D., & Nevile, C. M. (2010). Early Marine Survival of Coho Salmon in the Strait of Georgia Declines to Very Low Levels. *Marine and Coastal Fisheries, 2*(1). doi:https://doi.org/10.1577/C09-040.1

Beasley, A., Belanger, S. E., Brill, J. L., & Otter, R. R. (2015). Evaluation and comparison of the relationship between NOEC and EC10 or EC20 values in chronic Daphnia toxicity testing. *Environmental Toxicology and Chemistry, 34*(10), 2378-2384.

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:doi:10.1577/1548-8659(2001)130<0204:MSTIOA>2.0.CO;2

Becker, B. H., Peery, M. Z., & Beissinger, S. R. (2007). Ocean climate and prey availability affect the trophic level and reproductive success of the marbled murrelet, and endangered seabird. *Mar Ecol Prog Ser, 329*, 267-279.

Bednaršek, N., Calosi, P., Feely, R. A., Ambrose, R., Byrne, M., Chan Kit Yu, K., . . . Weisberg, S. B. (2021). Synthesis of Tresholds of Ocean Acidification Impacts on Echinoderms. *Frontiers in Marine Science, 8*, 261. doi:10.3389/fmars.2021.602601

Bednaršek, N., Feely, R. A., Howes, E. L., Hunt, B., Kessouri, F., & León, P. (2019). Systematic review and meta-analysis towards synthesis of thresholds of ocean acidification impacts on calcifying pteropods and interactions with warming. *Frontiers in Marine Science, 6*, 227. doi:doi: 10.3389/fmars.2019.00227

Bednaršek, N., Pelletier, G., Ahmed, A., & Feely, R. A. (2020). Chemical Exposure Due to Anthropogenic Ocean Acidification Increases Risks for Estuarine Calcifiers in the Salish Sea: Biogeochemical Model Scenarios. *Frontiers in Marine Science, 7*, 580. doi:10.3389/fmars.2020.00580

Behnke, R., & Benson, D. (1980). Endangered and threatened fishes of the Upper Colorado River Basin: Fort Collins. *Colorado State University Cooperative Extension Service Bulletin A, 503*, 34.

Beitinger, T., & Fitzpatrick, L. (1979). Physiological and Ecological Correlates of Preferred Temperature in Fish. *Integrative and Comparative Biology*. doi:DOI:10.1093/ICB/19.1.319

Bengtsson, B.-E., & Bergström, B. (1987). A flowthrough fecundity test with Nitocra spinipes (harpacticoidea crustacea) for aquatic toxicity. *Ecotoxicology and Environmental Safety, 14*(3), 260-268.

Bentley, R. I., Dean, J. W., Hollister, T. A., Leblanc, C. A., Sauter, S., III, B. B. S., & Wilson, W. C. (1978). *Laboratory Evaluation of the Toxicity of Elemental Phosphorus (P4) to Aquatic Organisms*. Retrieved from 790 Main Street, Wareham, MA:

Berry, W. (1989). [Recalculation of Del Nimmo's Flow-Through Chronic Values for Mysids with Diazinon]. 85670.

Berry, W. J. (1989, April 27, 1989). [Recalculation of Del Nimmo's flow-through chronic values for mysids with diazinon.].

Berry, W. J., Cantwell, M. G., Edwards, P. A., Serbst, J. R., & Hansen, D. J. (1999). Predicting toxicity of sediments spiked with silver. *Environmental Toxicology and Chemistry: An International Journal, 18*(1), 40-48.

Besser, J., Hardesty, D., Greer, I., & Ingersoll, C. (2009). Sensitivity of freshwater snails to aquatic contaminants: Survival and growth of endangered snail species and surrogates in 28-day exposures to copper, ammonia and pentachlorophenol. *US Geological Survey, Columbia, MO*.

Beyers, D. W., Carlson, C. A., & Keefe, T. J. (1994). Toxicity of carbaryl and malathion to two federally endangered fishes, as estimated by regression and ANOVA. *Environmental Toxicology and Chemistry: An International Journal, 13*(1), 101-107.

Bhattacharya, H., Xiao, Q., & Lun, L. (2008). Toxicity studies of nonylphenol on rosy barb (Puntius conchonious): a biochemical and histopathological evaluation. *Tissue Cell, 40*(4), 243-249. doi:10.1016/j.tice.2007.12.004

Bianchi, A., Bowles, K. C., Brauner, C. J., Gorsuch, J. W., Kramer, J. R., & Wood, C. M. (2002). Evaluation of the effect of reactive sulfide on the acute toxicity of silver (I) to *Daphnia magna*. part 2: toxicity results. *Environmental Toxicology and Chemistry, 21*(6), 1294-1300.

Bianchini, A., Grosell, M., Gregory, S. M., & Wood, C. M. (2002). Acute silver toxicity in aquatic animals is a function of sodium uptake rate. *Environmental science & technology, 36*(8), 1763-1766.

Birge, W., Black, J., & Westerman, A. (1979). Evaluation of aquatic pollutants using fish and amphibian eggs as bioassay organisms. *Animals as monitors of environmental pollutants/sponsored by Northeastern Research Center for Wildlife Diseases, University of Connecticut, Registry of Comparative of Lab Animal Resources, National Academy of Sciences*.

Birtwell, I. K. (1989). Comments on the sensitivity of salmonids to reduced levels of dissolved oxygen, and to pulp mill pollution in Neroutsos Inlet, British Columbia. *Canadian Technical Report of Fisheries and Aquatic Sciences*.

Bishop, S., & Morgan, A. (1996). *Critical habitat issues by basin for natural Chinook salmon stocks in the coastal and Puget Sound areas of Washington State*. Retrieved from Olympia, WA:

Blewett, T. A., & Leonard, E. M. (2017). Mechanisms of nickel toxicity to fish and invertebrates in marine and estuarine waters. *Environmental Pollution, 223*, 311-322.

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. (1992). Notes on the nomenclature and distribution of the bull trout and the effects of human activity on the species. In. Corvallis, OR: Oregon Chapter of the American Fisheries Society

Boran, H., Altinok, I., & Capkin, E. (2010). Histopathological changes induced by maneb and carbaryl on some tissues of rainbow trout, Oncorhynchus mykiss. *Tissue and Cell, 42*(3), 158-164.

Borgmann, U., Couillard, Y., Doyle, P., & Dixon, D. G. (2005). Toxicity of sixty-three metals and metalloids to Hyalella azteca at two levels of water hardness. *Environmental Toxicology and Chemistry: An International Journal, 24*(3), 641-652.

Borgmann, U., Norwood, W. P., Reynoldson, T. B., & Rosa, F. (2001). Identifying cause in sediment assessments: bioavailability and the Sediment Quality Triad. *Canadian Journal of Fisheries and Aquatic Sciences, 58*(5), 950-960.

Borthwick, P. W., Patrick, J. M., Jr., & Middaugh, D. P. (1985). Comparative acute sensitivities of early life stages of atherinid fishes to chlorpyrifos and thiobencarb. *Arch Environ Contam Toxicol, 14*(4), 465-473. doi:10.1007/BF01055533

Borthwick, P. W., Patrick, J. M., & Middaugh, D. P. (1985). Comparative acute sensitivities of early life stages of atherinid fishes to chlorpyrifos and thiobencarb. *Archives of environmental contamination and toxicology, 14*(4), 465-473.

Borton, S. F., & Miller, B. S. (1980). *Geographical distribution of Puget Sound fishes: maps and data source sheets*. Retrieved from https://digital.lib.washington.edu/researchworks/handle/1773/4282

Boyden, C. (1975). Distribution of some trace metals in Poole Harbour, Dorset. *Marine Pollution Bulletin, 6*(12), 180-187.

Boyle, R. W. (1968). *The geochemistry of silver and its deposits.* Retrieved from Ottawa, Canada.:

Brandenberger, J. M., Louchouarn, P., & Crecelius, E. A. (2011). Natural and post urbanization signatures of hypoxia in two basins of Puget Sound: Historical reconstruction of redox sensitive metals and organic matter inputs. *Aquatic Geochemistry, 17*(4), 645-670.

Branica, M., & Konrad, Z. (2015). *Lead in the Marine Environment: Proceedings of the International Experts Discussion on Lead Occurrence, Fate and Pollution in the Marine Environment, Rovinj, Yugoslavia, 18-22 October 1977*: Elsevier.

Brenner, F. J., & Cooper, W. L. (1978). Effect of suspended iron hydroxide on the hatchability and embryonic development of the coho salmon.

Breves, J. P., Duffy, T. A., Einarsdottir, I. E., Bjornsson, B. T., & McCormick, S. D. (2018). In vivo effects of 17alpha-ethinylestradiol, 17beta-estradiol and 4-nonylphenol on insulin-like growth-factor binding proteins (igfbps) in Atlantic salmon. *Aquat Toxicol, 203*, 28-39. doi:10.1016/j.aquatox.2018.07.018

Brewin, P. A., & Brewin, M. K. (1997). Distribution maps for bull trout in Alberta. *Friends of the Bull trout in Alberta*, 209-216.

Bringolf, R. B., Cope, W. G., Barnhart, M. C., Mosher, S., Lazaro, P. R., & Shea, D. (2007). Acute and chronic toxicity of pesticide formulations (atrazine, chlorpyrifos, and permethrin) to glochidia and juveniles of Lampsilis siliquoidea. *Environmental Toxicology and Chemistry: An International Journal, 26*(10), 2101-2107.

Brix, K. V., Keithly, J., DeForest, D. K., & Laughlin, J. (2004). Acute and chronic toxicity of nickel to rainbow trout (Oncorhynchus mykiss). *Environmental Toxicology and Chemistry: An International Journal, 23*(9), 2221-2228.

Brooke, L. (1993). Accumulation and lethality for two freshwater fishes (fathead minnow and bluegill) to nonylphenol. *Report to the US EPA for Work Assignment No. 1-12 of Contract No. 68-C1-0034*.

Brooke, L., & Smith, G. (2005). *Aquatic life ambient water quality criteria: diazinon. United States Environmental Protection Agency. Office of Water, Office of Science and Technology, Washington, DC*. Retrieved from

Brooke, L. T. (1993). *Acute and chronic toxicity of nonylphenol to ten species of aquatic organisms* (Report to the U.S. EPA for Work Assignment No. 02 of Contract No. 68-C1-0034). Retrieved from Superior, WI:

Brooks, E. G., Holland, R. A., Darwall, W. R., Eigenbrod, F., & Tittensor, D. (2016). Global evidence of positive impacts of freshwater biodiversity on fishery yields. *Glob Ecol Biogeogr, 25*(5), 553-562. doi:10.1111/geb.12435

Brown, K., Anderson, D., Jones, S., Deuel, L., & Price, J. (1979). The relative toxicity of four pesticides in tap water and water from flooded rice paddies. *International Journal of Environmental Studies, 14*(1), 49-53.

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)* (94-04). Retrieved from Olympia, WA:

Brueggeman, J. J., Green, G. A., Grotefendt, R. A., & Chapman, D. G. (1987). *Aerial surveys of endangered cetaceans and other marine mammals in the northwestern Gulf of Alaska and southeastern Bering Sea. Final report to Outer Continental Shelf Environmental Assessment Program*. Retrieved from Bellevue, WA:

Brueggeman, J. J., Green, G. A., Grotefendt, R. A., & Chapman, D. G. (1988). *Shipboard surveys of endangered cetaceans in the northwestern Gulf of Alaska. Final report to Outer Continental Shelf Environmental Assessment Program*. Retrieved from Bellevue, WA:

Bryant, P. J., Nichols, G., Bryant, T. B., & Miller, K. (1981). Krill availability and the distribution of humpback whales in southeast Alaska. *J Mammal, 62*, 427-430.

Bryant, V., Newbery, D. M., McLusky, D. S., & Campbell, R. (1985). Effect of temperature and salinity on the toxicity of nickel and zinc to two estuarine invertebrates *(Corophium volutator, Macoma balthica)*. *Marine Ecology - Progress Series, 24*, 139-153. Retrieved from http://www.int-res.com/articles/meps/24/m024p139.pdf

Buchanan, D. V., Millemann, R. E., & Stewart, N. E. (1970). Effects of the insecticide Sevin on various stages of the Dungeness crab, Cancer magister. *Journal of the Fisheries Board of Canada, 27*(1), 93-104.

Buckley, J. A. (1976). Acute Toxicity of Residual Chlorine in Wastewater to Coho Salmon (Oncorhynchus kisutch) and Some Resultant Hematologic Changes. *Journal of the Fisheries Research Board of Canada, 33*(12), 2854-2856. doi:10.1139/f76-344

Buhl, K. J., & Hamilton, S. J. (1991). Relative Sensitivity of early life stages of Artic grayling, Coho salmon, and rainbow trout to nine inorganics. *Ecotoxicology Environmental Safety, 22*(2). doi:10.1016/0147-6513(91)90058-w

Bulthuis, D., A. (2010). *The Ecology of Padilla Bay, Washington: An Estuarine Profile of a National Estuarine Research Reserve* Retrieved from

Burger, A. E. (1995). Chapter 29. Marine distribution, abundance, and habitats of marbled murrelets in British Columbia. In *Ecology and conservation of the Marbled Murrelet. USDA Forest Service General Technical Report PSW-152* (pp. 295-312). Albany, CA: Pacific Southwest Research Station, Forest Service, US Department of Agriculture.

Burgess, N. M., & Meyer, M. W. (2008). Methylmercury exposure associated with reduced productivity in common loons. *Ecotoxicology, 17*(2), 83-91. doi:10.1007/s10646-007-0167-8

Burkett, E. E. (1995). *Chapter 22: Marbled Murrelet Food Habits and Prey Ecology. In: Ralph, C. John; Hunt, George L., Jr.; Raphael, Martin G.; Piatt, John F., Technical Editors. 1995. Ecology and conservation of the Marbled Murrelet. Gen. Tech. Rep. PSW-GTR-152*. Albany, CA Retrieved from https://www.fs.usda.gov/treesearch/pubs/27904

Buser, C. C., Spaak, P., & Wolinska, J. (2012). Disease and pollution alter Daphnia taxonomic and clonal structure in experimental assemblages. *Freshwater Biology, 57*(9), 1865-1874.

Butler, J. A., Millemann, R. E., & Stewart, N. E. (1968). Effects of the insecticide Sevin on survival and growth of the cockle clam Clinocardium nuttalli. *Journal of the Fisheries Board of Canada, 25*(8), 1621-1635.

Cadmus, P., Brinkman, S. F., & May, M. K. (2018). Chronic toxicity of ferric iron for North American aquatic organisms: Derivation of a chronic water quality criterion using single species and mesocosm data. *Archives of environmental contamination and toxicology, 74*(4), 605-615.

Cadmus, P., Brinkman, S. F., & May, M. K. (2018). Chronic Toxicity of Ferric Iron for North American Aquatic Organisms: Derivation of a Chronic Water Quality Criterion Using Single Species and Mesocosm Data. *Arch Environ Contam Toxicol, 74*(4), 605-615. doi:10.1007/s00244-018-0505-2

Cairns, M. A., Garton, R. R., & Tubb, R. A. (1982). Use of fish ventilation frequency to estimate chronically safe toxicant concentrations. *Transactions of the American Fisheries Society, 111*(1), 70-77.

Calabrese, A., Collier, R., Nelson, D., & MacInnes, J. (1973). The toxicity of heavy metals to embryos of the American oyster Crassostrea virginica. *Marine Biology, 18*(3), 162-166.

Calabrese, E. J., & Baldwin, L. A. (1993). *Performing Ecological Risk Assessments*. Boca Raton, FL: Lewis Publishers.

Calambokidis, J., Steiger, G. H., Curtice, C., Harrison, J. D., Ferguson, M. C., Becker, E., . . . Van Parijs, S. M. (2015). Biologically important areas for selected cetaceans within U.S. waters - West Coast region. *Aquat Mamm, 41*(1), 39-53.

Caldwell, R. S. (1975). Hydrogen sulfide effects on selected larval and adult marine invertebrates.

Call, D. J., & Brooke, L. T. (1982). *Report on Stonefly Toxicity Tests with Priority Pollutants*. Duluth, MN: Ctr.for Lake Superior Environ.Stud., Univ.of Wisconsin-Superior, Superior, WI (Memo to R.E.Siefert, U.S.EPA, Duluth, MN):2 p.

Call, D. J., Polkinghorne, C. N., Markee, T. P., Brooke, L. T., Geiger, D. L., Gorsuch, J. W., & Robillard, K. A. (1999). Silver toxicity to Chironomus tentans in two freshwater sediments. *Environmental Toxicology and Chemistry: An International Journal, 18*(1), 30-39.

Campero, M., Ollevier, F., & Stoks, R. (2007). Ecological relevance and sensitivity depending on the exposure time for two biomarkers. *Environmental Toxicology: An International Journal, 22*(6), 572-581.

Canada, N. (2014). *BC Nature to challenge Cabinet decision to approve Northern Gateway*.

Capurro, L. R. A. (1970). *Oceanography for practicing engineers*. New York: Barnes and Noble Inc.

Cardin, J. A. (1981). [Memorandum to John H. Gentile].

Cardin, J. A. (1982). [Memorandum to JH Gentile. US Environmental Protection Agency].

Cardwell, R. D., Foreman, D. G., Payne, T. R., & Wilbur, D. J. (1976). *Acute Toxicity of Selected Toxicants to Six Species of Fish*. (EPA-600/3-76-008). Duluth, MN: Environmental Research Laboratory

Carlson, C. A., & Muth, R. T. (1989). The Colorado River: lifeline of the American southwest. *Canadian Special Publication of Fisheries and Aquatic Sciences, 106*, 220-239.

Carlson, H. R., & Straty, R. R. (1981). Habitat and nursery grounds of Pacific rockfish, *Sebastes* spp., in rocky coastal areas of southeastern Alaska. *Mar Fish Rev, 43*, 13-19.

Carr, M. H. (1983). *Spatial and temporal patterns of recruitment of young-of-the-year rockfishes (genus Sebastes) into a Central California kelp forest.* (Master of Arts). San Francisco State University, San Francisco, CA.

Carretta, J. V., Forney, K. A., Oleson, E. M., Weller, D. W., Lang, A. R., Baker, J., . . . Brownell, R. L., Jr. (2016). *U.S. Pacific draft marine mammal stock assessments: 2016*.

Carretta, J. V., Oleson, E. M., Weller, D. W., Lang, A. e. R., Forney, K. A., Baker, J. D., . . . Hill, M. C. (2013). *U.S. Pacific marine mammal stock assessment, 2012*. Retrieved from https://repository.library.noaa.gov/view/noaa/4464

Carretta, J. V., Oleson, E. M., Weller, D. W., Lang, A. e. R., Forney, K. A., Baker, J. D., . . . Mattila, D. K. (2014). *U.S. Pacific marine mammal stock assessments, 2013*. Retrieved from https://repository.library.noaa.gov/view/noaa/4772

Carter, H. R., & Sealy, S. G. (1986). Year-round use of coastal lakes by Marbled Murrelets. *The Condor, 88*(4), 473-477.

Carter, K. (2005). *The effects of dissolved oxygen on Steelhead Trout, Coho Salmon, and Chinook Salmon biology and function by life stage*.

Carter, K. (2008). *Effects of Temperature, Dissolved Oxygen/Total Dissolved Gas, Ammonia, and pH on Salmonids*. Retrieved from California:

Cascadia Research. (2017, June 4, 2017). Return of humpback whales to the Salish Sea. Retrieved from http://www.cascadiaresearch.org/projects/return-humpback-whales-salish-sea

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.

Cearley, J. E. (1971). *Toxicity and bioconcentration of cadmium, chromium and silver in Micropterus salmoides and Lepomis macrochirus.*

Chamberlin, J. W., Kagley, A. N., Fresh, K. L., & Quinn, T. P. (2011). Movements of Yearling Chinook salmon during the First Summer in Marine Waters of Hood Canal, Washington. *Transactions of the American Fisheries Society*, 429-439. Retrieved from https://doi.org/10.1080/00028487.2011.572006

Chapman, G. A. (1975). *Toxicity ot copper, cadmium and zinc to Pacific Northwest salmonids - Interim Report*. (Task 002 ROAP 10CAR). Corvallis, OR: National Environmental Research Center

Chapman, P. M., Wang, F., Janssen, C. R., Goulet, R. R., & Kamunde, C. N. (2003). Conducting ecological risk assessments of inorganic metals and metalloids: current status. *Human and ecological risk assessment, 9*(4), 641-697.

Chau, Y., & Kulikovsky-Cordeiro, O. (1995). Occurrence of nickel in the Canadian environment. *Environmental Reviews, 3*(1), 95-120.

Chemistry, T. R. S. o. (1999). *Metabolic Pathways of Agrochemicals. Part 2. Insecticides and Fungicides* (T. R. Roberts, D.H. Hutson, P.J. Jewess, P.W. Lee, P.H. Nicholls and J.R. Plimmer Ed.). Cambridge, United Kingdom.

Chen, Y., Huang, J., Xing, L., Liu, H., Giesy, J. P., Yu, H., & Zhang, X. (2014). Effects of multigenerational exposures of D. magna to environmentally relevant concentrations of pentachlorophenol. *Environmental Science and Pollution Research, 21*(1), 234-243.

Cheng, Y., Ekker, M., & Chan, H. M. (2015). Relative developmental toxicities of pentachloroanisole and pentachlorophenol in a zebrafish model (Danio rerio). *Ecotoxicology and Environmental Safety, 112*, 7-14.

Chung, W. S., Marshall, N. J., Watson, S. A., Munday, P. L., & Nilson, G. E. (2014). Ocean acidification slows retinal function in a damselfish through interference with GABA(A) receptors. *Journal of Experimental Biology, 217*, 323-326.

Clark, J. R., Patrick Jr, J. M., Middaugh, D. P., & Moore, J. C. (1985). Relative sensitivity of six estuarine fishes to carbophenothion, chlorpyrifos, and fenvalerate. *Ecotoxicology and Environmental Safety, 10*(3), 382-390.

Clarke, L. R., Cameron, W. A., Stonecypher, R. W., Jr., Hayes, M. C., & Carmichael, R. W. (2009). Performance of spring Chinook Salmon reared in Standard Raceways and in Michigan Raceways with oxygen supplementation. *North American Journal of Aquaculture, 71*, 339-347. Retrieved from https://doi.org/10.1577/A08-067.1

Clearwater, S. J., Farag, A. M., & Meyer, J. S. (2002). Bioavailability and toxicity of dietborne copper and zinc to fish. *Comp Biochem Physiol C Toxicol Pharmacol, 132*(3), 269-313. doi:10.1016/s1532-0456(02)00078-9

Clifford, M. A., Eder, K. J., Werner, I., & Hedrick, R. P. (2005). Synergistic effects of esfenvalerate and infectious hematopoietic necrosis virus on juvenile chinook salmon mortality. *Environ Toxicol Chem, 24*(7), 1766-1772. doi:10.1897/04-463r.1

Collins, B. (2000). *Mid-19th century stream channels and wetlands interpreted from archival sources for three north Puget Sound estuaries*. Retrieved from Seattle, WA.

Collins, B., & Sheikh, A. J. (2005). *Historical construction, classification and change analysis of Puget Sound tidal marshes*. Retrieved from Seattle, WA.

Collins, B. D., & Sheikh, A. J. (2002). *Historical riverine dynamics and habitats of the Nooksack River. Interim report to the Nooksack Indian Tribe Natural Resources Department.* Retrieved from Deming, WA.

Comber, M., Williams, T., & Stewart, K. (1993). The effects of nonylphenol on Daphnia magna. *Water Research, 27*(2), 273-276.

Connell, D., Chaisuksant, Y., & Yu, J. (1999). Importance of internal biotic concentrations in risk evaluations with aquatic systems. *Marine Pollution Bulletin, 39*(1-12), 54-61.

Consulting, S. (1999). *Sensitivity of bull trout (Salvelinus confluentus) to cadmium and zinc in water characteristic of the Coeur d'Alene River Basin: Boulder, Colorado*. Retrieved from

Cook, P. M., Carlson, A. R., & Lee II, H. (1992). *Tissue Residue Approach*. Washington, D.C.: U.S. Environmental Protection Agency

Coombs, C. I. (1979). *Reef fishes near Depoe Bay, Oregon: movement and the recreational fishery.* (Master of Science). Oregon State University.

Cooper, W. J., Jones, A. C., Whitehead, R. F., & Zika, R. G. (2007). Sunlight-induced photochemical decay of oxidants in natural waters: implications in ballast water treatment. *Environ Sci Technol, 41*(10), 3728-3733. doi:10.1021/es062975a

Cope, O. B. (1965). *Effects of Pesticides on Fish and Wildlife –1964 - Research Findings of the Fish and Wildlife Service*. Retrieved from Washington, DC.

Corporation, C.-G. (1976). *Reports of investigations made with respect to fish and wildlife requirements for diazinon and its formulated products*.

Couillard, C., Lebeuf, M., Légaré, B., & Trottier, S. (2008). Effects of diazinon on mummichog (Fundulus heteroclitus) larvae produced from eggs differentially treated with PCB126. *Archives of environmental contamination and toxicology, 54*(2), 283-291.

Cripe, G. M., Hansen, D. J., Macauley, S. F., & Forester, J. (1986). Effects of diet quantity on sheepshead minnows (Cyprinodon variegatus) during early life-stage exposures to chlorpyrifos. In T. M. Poston & R. Purdy (Eds.), *Aquatic Toxicology and Environmental Fate: Ninth Volume* (Vol. ASTM STP 921, pp. 450-460). Philadelphia, PA: American Society for Testing and Materials (ASTM).

CWR. (2016, December 31, 2016). Southern resident killer whale population. Retrieved from http://www.whaleresearch.com/orca-population

D'Vincent, C. G., Nilson, R. M., & Hanna, R. E. (1985). Vocalization and coordinated feeding behavior of the humpback whale in southeastern Alaska. *Sci Rep Whales Res Inst Tokyo, 36*, 41-48.

Daam, M. A., Teixeira, H., Lillebø, A. I., & Nogueira, A. J. (2019). Establishing causal links between aquatic biodiversity and ecosystem functioning: status and research needs. *Science of the Total Environment, 656*, 1145-1156.

Dave, G. (1984). Effects of copper on growth, reproduction, survival and hemoglobin in *Daphnia magna*. *Comp. Biochem. Physiol, 78C*, 439.

Davies, P. H., et. al. (1976). Acute and chronic toxicity of lead to rainbow trout (*Salmo gairdneri*) in hard and soft water. *Water Research, 10*(199).

Davis, H. C., & Hidu, H. (1969). Effects of pesticides on embryonic development of clams and oysters and on survival and growth of the larvae. *Fish. Bull, 67*(2), 393-404.

De Schamphelaere, K. A., & Janssen, C. R. (2004). Bioavailability and chronic toxicity of zinc to juvenile rainbow trout (Oncorhynchus mykiss): comparison with other fish species and development of a biotic ligand model. *Environmental science & technology, 38*(23), 6201-6209.

De Zwart, D. (2002). Observed regularities in species sensitivity distributions *Species Sensitivity Distributions in Ecotoxicology*, 133-154.

Dean, T. L., & Richardson, J. (1999). Responses of seven species of native freshwater fish and a shrimp to low levels of dissolved oxygen. *New Zealand Journal of Marine and Freshwater Research, 1*(33), 99-106. doi:DOI: 10.1080/00288330.1999.9516860

Deanovic, L. A., Markiewicz, D., Stillway, M., Fong, S., & Werner, I. (2013). Comparing the effectiveness of chronic water column tests with the crustaceans Hyalella azteca (order: Amphipoda) and Ceriodaphnia dubia (order: Cladocera) in detecting toxicity of current-use insecticides. *Environmental Toxicology and Chemistry, 32*(3), 707-712.

DeMott, G. E. (1983). *Movement of tagged lingcod and rockfishes off Depoe Bay, Oregon.* (Master of Science). Oregon State University.

Ding, Y., Landrum, P. F., You, J., Harwood, A. D., & Lydy, M. J. (2012). Use of solid phase microextraction to estimate toxicity: Relating fiber concentrations to toxicity—Part I. *Environmental Toxicology and Chemistry, 31*(9), 2159-2167.

Dinnel, P. A., Link, J. M., Stober, Q. J., Letourneau, M. W., & Roberts, W. E. (1989). Comparative sensitivity of sea urchin sperm bioassays to metals and pesticides. *Archives of Environmental Contamination and Toxicology, 18*, 748-755.

Dinnel, P. A., Slober, Q. J., Link, J. M., Letourneau, M. W., Roberts, W. E., Felton, S. P., & Nakatani, R. E. (1983). *Methodology and validation of a sperm cell toxicity test for testing toxic substances in marine waters*. Retrieved from Seattle, WA.

Dolphin, W. F. (1987). Observations of humpback whale, *Megaptera novaeangliae*, killer whale, *Orcinus orca*, interactions in Alaska: comparison with terrestrial predator-prey relationships. *Can Field Nat, 101*, 70-75.

Domagalski, J. L., Dubrovsky, N. M., & Kratzer, C. R. (1997). Pesticides in the San Joaquin River, California: Inputs from dormant sprayed orchards. *Journal of Environmental Quality, 26*(2), 454-465.

Domenici, P., Steffensen, J. F., & Batty, R. S. (2005). The effect of progressive hypoxia on swimmin activity and schooling in Atlantic herring. *Journal of Fish Biology, 57*(6). Retrieved from https://doi.org/10.1111/j.1095-8649.2000.tb02229.x

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

Dorfman, D. (1977). Tolerance of Fundulus heteroclitus to different metals in salt waters. *Bull NJ Acad Sci, 22*(2), 21-23.

Drake, J. S., Berntson, E. A., Gustafson, R. G., Holmes, E. E., Levin, P. S., Tolimieri, N., . . . Cope, J. M. (2010). *Status review of five rockfish species in Puget Sound, Washington : bocaccio (Sebastes*

paucispinis), canary rockfish (S. pinniger), yelloweye rockfish (S. ruberrimus), greenstriped rockfish (S. elongatus), and redstripe rockfish (S. proriger). U.S. Dept. of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service Retrieved from https://repository.library.noaa.gov/view/noaa/3777

Drost, B. W. (1979). *Water resources of the Swinomish Indian Reservation, Washington* (79-12). Retrieved from http://pubs.er.usgs.gov/publication/wri7912

Duarte, C. M. (2002). The future of seagrass meadows. *Environ Conserv, 29*(2), 192-206.

Duffy, T. A., Iwanowicz, L. R., & McCormick, S. D. (2014). Comparative responses to endocrine disrupting compounds in early life stages of Atlantic salmon, Salmo salar. *Aquat Toxicol, 152*, 1-10. doi:10.1016/j.aquatox.2014.03.015

Dung, L. Q., Cuong, N. M., Huyen, N. T., & Duc Cu, N. (2005). Acute toxicity test to determine the effects of copper, zinc and cyanide on cobia (*Rachycentron canadum*) resources in North Vietnam. *Australasian Journal of Ecotoxicology, 11*, 163-166. Retrieved from http://www.leusch.info/ecotox/aje/archives/vol11p163.pdf

Dwyer, F. J., Mayer, F. L., Sappington, L. C., Buckler, D. R., Bridges, C. M., Greer, I. E., . . . Neuderfer, G. N. (2005). Assessing contaminant sensitivity of endangered and threatened aquatic species: part I. Acute toxicity of five chemicals. *Arch Environ Contam Toxicol, 48*(2), 143-154. doi:10.1007/s00244-003-3038-1

Dwyer, F. J., Sappington, L. C., Buckler, D. R., & Jones, S. B. (1995). *Use of surrogate species in assessing contaminant risk to endangered and threatened fishes*. (EPA/600/R-96/029). Springfield, VA: National Technical Information Service.

Dyer, S. D., White-Hull, C. E., & Shephard, B. K. (2000). Assessments of Chemical Mixtures via Toxicity Reference Values Overpredict Hazard to Ohio Fish Communities. *Environmental Science & Technology, 34*(12), 2518-2524. doi:10.1021/es991160h

Earnest, R. (1970). Effects of pesticides on aquatic animals in the estuarine and marine environment. In *Progress in Sport Fisheries Research 1970* (Vol. Resource Publication 106, pp. 10-13). Washington, D.C.: U.S. Bureau of Sport Fisheries and Wildlife.

Eaton, D. L., & Klaassen, C. D. (1995). Principles of Toxicology. In C. D. Klaassen (Ed.), *Casarett & Doull's Toxicology, the Basic Science of Poisons* (5th ed., pp. 13 - 33). New York, NY: McGraw Hill.

Eaton, J. G. (1970). Chronic malathion toxicity to the bluegill (Lepomis macrochirus Rafinesque). *Water Research, 4*(10), 673-684.

ECETOC. (2003). *Aquatic Hazard Assessment II*. Retrieved from

Ecology. (2012). *Preparing for a changing climate. Washington State's integrated climate response strategy*.

Ecology. (2013). *Sediment Management Standards, Chapter 173-204 WAC*. (13-09-055). Olympia, WA: Washington State Department of Ecology Retrieved from https://fortress.wa.gov/ecy/publications/documents/1309055.pdf

Ecology. (2019). *Puget Sound Nutrient Source Reduction Project*. Retrieved from https://apps.ecology.wa.gov/publications/documents/1903001.pdf

Eder, K. J., Clifford, M. A., Hedrick, R. P., Kohler, H. R., & Werner, I. (2008). Expression of immune-regulatory genes in juvenile Chinook salmon following exposure to pesticides and infectious hematopoietic necrosis virus (IHNV). *Fish Shellfish Immunol, 25*(5), 508-516. doi:10.1016/j.fsi.2008.07.003

Eisler, R. (1998). *Nickel hazards to fish, wildlife, and invertebrates : a synoptic review*: U.S. Dept. of the Interior, U.S. Geological Survey, Patuxent Wildlife Research Center.

Eisler, R., & Hennekey, R. J. (1977). Acute toxicities of Cd 2+, Cr+ 6, Hg 2+, Ni 2+ and Zn 2+ to estuarine macrofauna. *Archives of Environmental Contamination and Toxicology, 6*(1), 315-323.

England, D., & Bussard, J. (1993). Toxicity of nonylphenol to the midge Chironomus tentans. *Analytical Bio-Chemistry Laboratories, Inc. Report*(40597).

England, D. E. (1995). *Chronic toxicity of nonylphenol to Ceriodaphnia dubia* (Report No. 41756). Retrieved from Columbia, MO.

Environment and Climate Change Canada and Health Canada (ECCC, H. (2017). Draft screening assessment Hydrogen Sulfide (H_2S), Sodium Sulfide (Na(SH)) and Sodium Sulfide (Na_2S). *Environment and Climate Change Canada, Health Canada*, 117.

EPA. (1980). *Ambient Water Quality Criteria for Silver*. Washington, DC: United States Environmental Protection Agency.

EPA. (1985). Ambient Water Quality Criteria for Chlorine - 1984.

EPA. (1987). *Ambient Aquatic Life Water Quality Criteria for Silver*. Narragansett, RI

EPA. (1994). *Chemical Summary for Chlorine*. Washington, D.C.: Office of Pollution Prevention and Toxics

EPA. (2006). *National Recommended Water Quality Criteria*. Washington, DC

EPA. (2009). *Pesticides - Reregistration Eligibility Decision (RED)*. Washington DC: United States Environmental Protection Agency

Escher, B. I., & Hermens, J. L. (2004). Internal exposure: linking bioavailability to effects. *Environ Sci Technol, 38*(23), 455A-462A. doi:10.1021/es0406740

Ezekwe, C. I., & Edoghotu, M. I. (2015). Water quality and environmental health indicators in the Andoni River estuary, Eastern Niger Delta of Nigeria. *Environmental Earth Sciences, 74*(7), 6123-6136.

Falxa, G., Baldwin, J., Lynch, D., Nelson, S. K., Miller, S. L., Pearson, S. F., . . . Huff, M. H. (2008). *Marbled murrelet effectiveness monitoring, Northwest Forest Plan: 2004-2007 summary report*. Retrieved from https://wdfw.wa.gov/publications/00116

Feder, H. M., Turner, C. H., & Limbaugh, C. (1974). *Observations on fishes associated with kelp beds in Southern California*. Retrieved from http://content.cdlib.org/view?docId=kt9t1nb3sh&brand=calisphere&doc.view=entire_text

Feely, R. A., Alin, S. R., Sabine, C. L., Warner, M., & Devol, A. (2010). The combined effects of ocean acidification, mixing and respiration on pH and carbonate saturation in an urbanized estuary. *Estuary Coast Shelf Science, 88*, 442-449. doi:doi: 10.1016/j.ecss.2010.05.004

Fergusson, J. E. (1990). *The heavy elements: chemistry, environmental impact adn health effects\Jack E. Fergusson*.

Fernandez-Casalderrey, A., Ferrando, M., & Andreu-Moliner, E. (1992). Effect of sublethal diazinon concentrations on the demographic parameters of Brachionus calyciflorus Pallas (Rotifera). *Bulletin of environmental contamination and toxicology, 48*(2), 202-208.

Fernandez-Casalderry, A., Ferrando, M., & Andreu-Moliner, E. (1992). Acute toxicity of several pesticides to rotifer (Brachionus calyciflorus). *Bulletin of environmental contamination and toxicology, 48*(1), 14-17.

Finlayson, B. J., & Verrue, K. M. (1982). Toxicities of copper, zinc, and cadmium mixtures to juvenile Chinook salmon. *Transactions of the American Fisheries Society, 11*(5), 645-650. Retrieved from http://dx.doi.org/10.1577/1548-8659(1982)111<645:TOCZAC>2.0.CO;2

Fisher, D. J., Burton, D. T., Yonkos, L. T., Turley, S. D., & Ziegler, G. P. (1999). The relative acute toxicity of continuous and intermittent exposures of chlorine and bromine to aquatic organisms in the presence and absence of ammonia. *Water Research, 33*(3), 760-768. doi:https://doi.org/10.1016/S0043-1354(98)00278-4

Fitzsimons, J. D., Huestis, S., & Williston, B. (1995). Occurrence of a swim-up syndrome in Lake Ontario lake trout in relation to contaminants and cultural practices. *Journal of Great Lakes Research, 21*, 277-285.

Fletcher, G., & Hoyle, R. (1972). Acute toxicity of yellow phosphorus to Atlantic cod (Gadus morhua) and Atlantic salmon (Salmo salar) smolts. *Journal of the Fisheries Research Board of Canada, 29*(9), 1295-1301.

Fliedner, A. (1993). *Daphnia magna, Reproduction test (OECD No. 202)* (Report No. UBA-002/4-22). Retrieved from Schmallenberg-Grafschaft, Germany:

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., 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. *Can J Zool, 76*, 1456-1471.

Ford, M. J., Hempelmann, J., Hanson, M. B., Ayres, K. L., Baird, R. W., Emmons, C. K., . . . Park, L. K. (2016). Estimation of a Killer Whale (Orcinus orca) Population's Diet Using Sequencing Analysis of DNA from Feces. *PLoS One, 11*(1), e0144956. doi:10.1371/journal.pone.0144956

Fort, D. J., & Stover, E. L. (1996). Effect of low-level copper and pentachlorophenol exposure on various early life stages of Xenopus laevis. In *Environmental Toxicology and Risk Assessment: Biomarkers and Risk Assessment: Fifth Volume*: ASTM International.

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. Retrieved from <Go to ISI>://WOS:A1989AT02400001

Fresh, K. L. (2006). *Juvenile Pacific Salmon in Puget Sound*. (2006-06). Seattle, WA Retrieved from https://apps.dtic.mil/docs/citations/ADA477860

Fretwell, K. (2017). *Exploring the issue of cumulative shipping impacts in the Salish Sea though a systematic focal species assessment framework [graduate project].* Retrieved from

Frierson, T., Hillier, L., LeClair, L., Lowry, D., Bean, J., Blaine, J., & Wright, E. (2015, January 28, 2015). [Assessment of ESA-listed rockfish (Sebastes spp.) and other rockfish presence, abundance, habitat, and prey base adjacent to NBK-Bremerton, NUWC-Keyport, NBK-Bangor, NAVMAG-Indian Island, NAS-Whidbey Island: 2014 Initial Survey Results].

Fry, D. M. (1995). Chapter 25. Pollution and fishing threats to marbled murrelets. In C. J. Ralph, G. L. Hunt, Jr, M. G. Raphael, & J. F. Piatt (Eds.), *Ecology and Conservation of the Marbled Murrelet. General Technical Report PSW-GTR-152* (pp. 257-260). Albany, CA: Pacific Southwest Research Station.

Fung, D. K., & Bewick, P. H. (1980). Short-Term Toxicity of Aqueous Hydrogen Sulfide to Representative Fish Species of Lake Huron. In J. G. Eaton, P. R. Parrish, & A. C. Hendricks (Eds.), (pp. 377-396). West Conshohocken, PA: ASTM International.

Gao, P., Guo, L., Li, Z., & Gibson, M. (2015). The derivation of water quality criteria for nonylphenol considering its endocrine disrupting features. *Water Quality Res. J. Canada, 50*, 268-278.

Geen, G. H., Neilson, J. D., & Bradford, M. (1985). Effects of pH on the early development and growth and otolith microstructure of chinook salmon, *Onocorhynchus tshawytscha*. *Canadian Journal of Zoology, 63*(1). doi:https://doi.org/10.1139/z85-005

Geffard, O., Xuereb, B., Chaumot, A., Geffard, A., Biagianti, S., Noël, C., . . . Charmantier-Daures, M. (2010). Ovarian cycle and embryonic development in Gammarus fossarum: application for reproductive toxicity assessment. *Environmental Toxicology and Chemistry, 29*(10), 2249-2259.

Gendaszek, A. S., & Opatz, C. C. (2013). *Thermal profiles for reaches of Snee-Oosh and Fornsby Creeks, Swinomish Indian Reservation, northwestern Washington, July 2013* (807). Retrieved from Reston, VA: http://pubs.er.usgs.gov/publication/ds807

Gentile, J., Gentile, S., Hairston, N., & Sullivan, B. (1982). The use of life-tables for evaluating the chronic toxicity of pollutants to Mysidopsis bahia. In *Ecology of Mysidacea* (pp. 179-187): Springer.

Giattina, J. D., Garton, R. R., & Stevens, D. G. (1982). Avoidance of copper and nickel by rainbow trout as monitored by a computer-based data acquisition system (Salmo gairdneri, pollutants, behaviour). *Transactions of the American Fisheries Society, 111*(4), 491-504.

Gilardi, K. V., Carlson-Bremer, D., June, J. A., Antonelis, K., Broadhurst, G., & Cowan, T. (2010). Marine species mortality in derelict fishing nets in Puget Sound, WA and the cost/benefits of derelict net removal. *Mar Pollut Bull, 60*(3), 376-382. doi:10.1016/j.marpolbul.2009.10.016

Goettl, J. P., Davies, P. H., & Sinley, J. R. (1972). *Water pollution studies*: Department of Natural Resources, Colorado Division of Wildlife.

Goettl, J. P., Davies, P. H., & Sinley, J. R. (1976). *Water Pollution Studies*. (Review No. 8). Fort Collins, CO: Colorado Division of Wildlife

Goettl, J. P., Sinley, J. R., & Davies, P. H. (1974). *Water Pollution Studies*. (Review No. 9). Fort Collins, CO: Colorado Division of Wildlife

Goetz, F. (1989). *Biology of the bull trout, a literature review*. Eugene, Oregon

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. Technical Memorandum NMFS-NWFSC-66*. Retrieved from

Good, T. P., Waples, R. S., & Adams, P. B. (2005). *Updated status of federally listed ESUs of West Coast salmon and steelhead*. U.S. Dept. of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center Retrieved from https://repository.library.noaa.gov/view/noaa/3413

Goodman. (1985). *MRID 154718*. Washington, D.C.

Goodman, L., Hansen, D., Cripe, G., Middaugh, D., & Moore, J. (1985). A new early life-stage toxicity test using the California grunion (Leuresthes tenuis) and results with chlorpyrifos. *Ecotoxicology and Environmental Safety, 10*(1), 12-21.

Goodman, L., Hansen, D., Middaugh, D., Cripe, G., & Moore, J. (1985). Method for early life-stage toxicity tests using three atherinid fishes and results with chlorpyrifos. In R. D. Cardwell, R. Purdy, & R. C. Bahner (Eds.), *Aquatic Toxicology and Hazard Assessment: Seventh Symposium* (Vol. ASTM STP 854, pp. 145-154). Philadelphia, PA: American Society for Testing and Materials.

Goodman, L. R., Hansen, D. J., Coppage, D. L., Moore, J. C., & Matthews, E. (1979). Diazinon (R): Chronic Toxicity to, and Brain Acetylcholinesterase Inhibition in, the Sheepshead Minnow, Cyprinodon variegatus. *Transactions of the American Fisheries Society, 108*(5), 479-488.

Goodman, L. R., Middaugh, D. P., Hansen, D. J., Higdon, P. K., & Cripe, G. M. (1983). Early life-stage toxicity test with tidewater silversides (Menidia Peninsulae) and chlorine-produced oxidants. *Environmental Toxicology and Chemistry, 2*(3), 337-342. doi:10.1002/etc.5620020310

Gorbi, G., Corradi, M. G., Invidia, M., Rivara, L., & Bassi, M. (2002). Is Cr (VI) toxicity to Daphnia magna modified by food availability or algal exudates? The hypothesis of a specific chromium/algae/exudates interaction. *Water Research, 36*(8), 1917-1926.

Grego, M., Riedel, B., Stachowitsch, M., & De Troch, M. (2014). Meiofauna winners and losers of coastal hypoxia: case study harpacticoid copepods. *Biogeosciences, 11*. doi:10.5194/bg-11-281-2014

Griffith, J., Tribe, S., Zackey, T., & Tribes, T. (2006). HABITAT AND FISH USE OF POCKET ESTUARIES IN THE WHIDBEY BASIN AND NORTH SKAGIT COUNTY BAYS, 2004 AND 2005.

Grunblatt, J., Meyer, B. E., & Wipfli, M. S. (2019). Invertebrate prey contributions to juvenile Coho Salmon diet from riparian habitats along three Alaska streams: Implications for environmental change. *Journal of Freshwater Ecology, 34*(1), 617-631.

Gündoğdu, A. (2008). Acute toxicity of Zinc and Copper for rainbow trout (*Onchorhyncus mykiss*). *Journal of Fisheries Sciences, 2*(5), 711-721. doi:10.3153/jfscom.2008039

Haigh, R., Ianson, D., Holt, C. A., Neate, H. E., & Edwards, A. M. (2015). Effects of Ocean Acidification on Temperate Coastal Marine Ecosystems and Fisheries in the Northeast Pacific. *PLoS ONE, 10*(2). doi:doi:10.1371/journal.pone.0117533

Hain, J. H. W., Carter, G. R., Kraus, S. D., Mayo, C. A., & Winn, H. E. (1982). Feeding behavior of the humpback whale, *Megaptera novaeangliae*, in the western North Atlantic. *Fish Bull, 80*, 259-268.

Hamazaki, T. (2002). Spatiotemporal prediction models of cetacean habitats in the mid-western North Atlantic Ocean (from Cape Hatteras, North Carolina, USA to Nova Scotia, Canada). *Mar Mam Sci, 18*(4), 920-937.

Hamilton, T. J., Holcombe, A., & Tresguerres, M. (2014). CO2-induced ocean acidification increases anxiety in rockfish via alteration of GABAA receptor functioning. *Proc Biol Sci, 281*. doi:doi: 10.1098/rspb.2013.2509

Hamlin, H. J., Marciano, K., & Downs, C. A. (2015). Migration of nonylphenol from food-grade plastic is toxic to the coral reef fish species Pseudochromis fridmani. *Chemosphere, 139*, 223-228.

Hankin, S. (2001). Chemicals in Drinking Water: Chloramines. *Scottish Centre for Infection and Environmental Health, Glasgow, Scotland*.

Hansen, D., & Parrish, P. (1977). *Suitability of Sheepshead Minnows (Cyprinodon variegatus) for Life-Cycle Toxicity Tests (Test ID 1062811).* Paper presented at the ASTM STP 634, Aquatic Toxicology and Hazard Evaluation, Philadelphia, PA.

Hansen, D. J., Goodman, L. R., Cripe, G. M., & Macauley, S. F. (1986). Early life-stage toxicity test methods for gulf toadfish (Opsanus beta) and results using chlorpyrifos. *Ecotoxicology and Environmental Safety, 11*(1), 15-22.

Hansen, D. J., & Parrish, P. R. (1977). Suitability of Sheepshead Minnows (Cyprinodon variegatus) for Life-Cycle Toxicity Tests. In F. L. Mayer & J. L. Hamelink (Eds.), (pp. 117-126). West Conshohocken, PA: ASTM International.

Hansen, J. A., Lipton, J., & Welsh, P. G. (1999). *Sensitivity of bull trout (Salvelinus confluentus) to cadmium and zinc in water characteristic of the Coeur d'Alene River Basin*. Retrieved from Seattle, WA.

Hansen, J. A., Welsh, P. G., Lipton, J., Cacela, D., & Dailey, A. D. (2002). Relative sensitivity of bull trout (Salvelinus confluentus) and rainbow trout (Oncorhynchus mykiss) to acute exposures of cadmium and zinc. *Environ Toxicol Chem, 21*(1), 67-75. doi:10.1897/1551-5028(2002)021<0067:Rsobts>2.0.Co;2

Hansen, J. A., Welsh, P. G., Lipton, J., Cacela, D., & Dailey, A. D. (2002). Relative sensitivity of bull trout (Salvelinus confluentus) and rainbow trout (Oncorhynchus mykiss) to acute exposures of cadmium and zinc. *Environmental Toxicology and Chemistry: An International Journal, 21*(1), 67-75.

Hanson, A. M., Kittilson, J. D., Martin, L. E., & Sheridan, M. A. (2014). Environmental estrogens inhibit growth of rainbow trout (Oncorhynchus mykiss) by modulating the growth hormone-insulin-like growth factor system. *Gen Comp Endocrinol, 196*, 130-138. doi:10.1016/j.ygcen.2013.11.013

Hanson, A. M., Kittilson, J. D., McCormick, S. D., & Sheridan, M. A. (2012). Effects of 17β-estradiol, 4-nonylphenol, and β-sitosterol on the growth hormone-insulin-like growth factor system and seawater adaptation of rainbow trout (Oncorhynchus mykiss). *Aquaculture, 362-363*, 241-247.

Hanson, A. M., Kittilson, J. D., McCormick, S. D., & Sheridan, M. A. (2012). Effects of 17β-estradiol, 4-nonylphenol, and β-sitosterol on the growth hormone-insulin-like growth factor system and seawater adaptation of rainbow trout (Oncorhynchus mykiss). *Aquaculture*(362-363), 241-247.

Hanson, M. B., Emmons, C. K., Ford, M. J., Everett, M., Parsons, K., Park, L. K., . . . Jacobsen, J. K. (2021). Endangered predators and endangered prey: Seasonal diet of Southern Resident killer whales. *PloS one, 16*(3), e0247031. Retrieved from C:\Users\MJankows\OneDrive - Environmental

Protection Agency (EPA)\Washington\NH3\Section 5 working folder\Lit and Data\Hanson et al 2021.pdf

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. *J Accoust Soc Am, 134*(5), 3486-3495.

Hard, J. J. (2007). *Status review of Puget Sound steelhead (Oncorhynchus mykiss)*. U.S. Dept. of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service Retrieved from https://repository.library.noaa.gov/view/noaa/3513

Harper, D. D., Farag, A. M., Hogstrand, C., & MacConnell, E. (2009). Trout density and health in a stream with variable water temperatures and trace element concentrations: Does a cold-water source attract trout to increased metal exposure? *Environmental Toxicology and Chemistry: An International Journal, 28*(4), 800-808.

Harris, M. L., Bishop, C. A., Struger, J., Ripley, B., & Bogart, J. P. (1998). The functional integrity of northern leopard frog (Rana pipiens) and green frog (Rana clamitans) populations in orchard wetlands. II. Effects of pesticides and eutrophic conditions on early life stage development. *Environmental Toxicology and Chemistry: An International Journal, 17*(7), 1351-1363.

Hart, B. T. (1982). Australian water quality criteria for heavy metals.

Hartmann, B., & Wendler, G. (2005). The Significance of the 1976 Pacific Climate Shift in the Climatology of Alaska. *Journal of Climate, 18*(22). doi:https://doi.org/10.1175/JCLI3532.1

Hartwell, S. I., Jin, J. H., Cherry, D. S., & Cairns Jr, J. (1989). Toxicity versus avoidance response of golden shiner, Notemigonus crysoleucas, to five metals. *Journal of Fish Biology, 35*(3), 447-456. Retrieved from https://doi.org/10.1111/j.1095-8649.1989.tb02995.x

Hayes, M. C., Rubin, S. P., Reisenbichler, R. R., Goetz, F. A., Jeanes, E., & McBride, A. (2011). Marine Habitat Use by Anadromous Bull Trout from the Skagit River, Washington. *Marine and Coastal Fisheries, 3*(1), 394-410.

Hays, H., Winn, H. E., & Petrecig, R. (1985). Anomalous feeding behavior of a humpback whale. *J Mammal, 66*, 819-826.

Heinz, G. H., Hoffman, D. J., & Gold, L. G. (1989). Impaired reproduction of mallards fed an organic form of selenium. *The Journal of Wildlife Management*, 418-428.

Heinz, G. H., Hoffman, D. J., Krynitsky, A. J., & Weller, D. M. (1987). Reproduction in mallards fed selenium. *Environmental Toxicology and Chemistry: An International Journal, 6*(6), 423-433.

Hem, J., & Cropper, W. (1962). *Chemistry of iron in natural water*. Retrieved from

Hem, J. D. (1970). *Study and interpretation of the chemical characteristics of natural water*: US Government Printing Office.

Henning, M. H., Robinson, S. K., McKay, K. J., Sullivan, J. P., & Bruckert, H. (2003). Productivity of American robins exposed to polychlorinated biphenyls, Housatonic River, Massachusetts, USA. *Environ Toxicol Chem, 22*(11), 2783-2788. doi:10.1897/02-536

Herbert, D. W. M., & Wakeford, A. C. (1964). Susceptibility of salmonid fish to poison under estuarine conditions. I. Zinc sulphate. *Int. Journal Air Water Pollut., 8*, 251-256.

Herman, L. M., Baker, C. S., Forestell, P. H., & Antinoja, R. C. (1980). Right whale *Balaena glacialis* sightings near Hawaii: a clue to the wintering grounds? *Mar Ecol Prog Ser, 2*, 271-275.

Hermanutz, R. O. (1978). Endrin and malathion toxicity to flagfish (Jordanella floridae). *Arch Environ Contam Toxicol, 7*(2), 159-168. doi:10.1007/BF02332046

Hickie, B. E., Ross, P. S., Macdonald, R. W., & Ford, J. K. (2007). Killer whales (*Orcinus orca*) face protracted health risks associated with lifetime exposure to PCBs. *Environ Sci Technol, 41*(18), 6613-6619. doi:10.1021/es0702519

Hickie, B. E., S, R. P., MacDonald, R. W., & Ford, J. K. B. (2007). Killer whales (*Orcinus orca*) face protracted health risks associated with lifetime exposure to PCBs. *Environ Sci Tech, 41*, 6613-6619.

Hoang, T. (2015). *Chronic toxicity of Lead to Topsmelt (Atherinops affinis)*. Retrieved from Durham, NC, USA.

Hobson, K. A. (1990). Stable isotope analysis of marbled murrelets: evidence for freshwater feeding and determination of trophic level. *The Condor, 92*(4), 897-903.

Hoelscher, B., & Bjornn, T. (1989). *Habitat, densities, and potential production of trout and char in Pend Oreille Lake Tributaries. Idaho Department of Fish and Game, Job Completion Report*.

Hogstrand, C., Galvez, F., & Wood, C. M. (1996). Toxicity, silver accumulation and metallothionein induction in freshwater rainbow trout during exposure to different silver salts. *Environmental Toxicology and Chemistry, 15*(7), 1102-1108. Retrieved from http://dx.doi.org/10.1002/etc.5620150713

Hogstrand, C., & Wood, C. M. (1998). Toward a better understanding of the bioavailability, phyisiology, and toxicity of silver in fish: Implications for water quality criteria. *Environmental Toxicology and Chemistry, 17*(4), 547-561. Retrieved from https://doi.org/10.1002/etc.5620170405

Hogstrand, C., & Wood, C. M. (1998). Toward a better understanding of the bioavailability, physiology, and toxicity of silver in fish: implications for water quality criteria. *Environmental Toxicology and Chemistry: An International Journal, 17*(4), 547-561.

Holcombe, G. W., Benoit, D. A., Leonard, E. N., & McKim, J. M. (1976). Long-Term Effects of Lead Exposure on Three Generations of Brook Trout (*Salvenilus fontinali*s). *Journal of the Fisheries Research Board of Canada, 33*(8), 1731-1741. Retrieved from https://doi.org/10.1139/f76-220

Holcombe, G. W., Phipps, G. L., Sulaiman, A. H., & Hoffman, A. D. (1987). Simultaneous multiple species testing: acute toxicity of 13 chemicals to 12 diverse freshwater amphibian, fish, and invertebrate families. *Archives of environmental contamination and toxicology, 16*(6), 697-710.

Holcombe, G. W., Phipps, G. L., Sulaiman, A. H., & Hoffman, A. D. (1987). Simultaneous multiple species testing: acute toxicity of 13 chemicals to 12 diverse freshwater amphibian, fish, and invertebrate families. *Arch Environ Contam Toxicol, 16*(6), 697-710. doi:10.1007/BF01055420

Holcombe, G. W., Phipps, G. L., & Tanner, D. K. (1982). The acute toxicity of kelthane, dursban, disulfoton, pydrin, and permethrin to fathead minnows Pimephales promelas and rainbow trout Salmo gairdneri. *Environmental Pollution Series A, Ecological and Biological, 29*(3), 167-178.

Holland, G. (1960). Toxic effects of organic and inorganic pollutants on young salmon and trout. 278.

Holland, G., Lasater, J., Neumann, E., & Eldridge, W. (1960). *Toxic effects of organic and inorganic pollutants on young salmon and trout*. Seattle, WA.

Holland, G. A., J.E. Lasater, E.D. Neumann and W.E. Eldridge. (1960). Toxic Effects of Organic and Inorganic Pollutants on Young Salmon and Trout. *Res. Bull. No. 5, State of Washington Dept. Fish., Seattle, WA*, 263.

Holmes, S. B., & Kingsbury, P. D. (1980). *The environmental impact of nonylphenol and the Matacil® formulation. Part 1: Aquatic ecosystems* (Report FPMX-35). Retrieved from Sault Ste. Marie, Ontario, Canada:

Hook, S. E., & Fisher, N. S. (2002). Relating the reproductive toxicity of five ingested metals in calanoid copepods with sulfur affinity. *Marine Environmental Res., 53*, 161-174.

Hooper, D. U., Adair, E. C., Cardinale, B. J., Byrnes, J. E., Hungate, B. A., Matulich, K. L., . . . O'Connor, M. I. (2012). A global synthesis reveals biodiversity loss as a major driver of ecosystem change. *Nature, 486*(7401), 105-108. doi:10.1038/nature11118

Hopf, H., & Muller, R. (1962). Laboratory breeding and testing of Australorbis glabratus for molluscicidal screening. *Bulletin of the World Health Organization, 27*(6), 783.

Hoque, M. T., Yusoff, F. M., Law, T., & Syed, M. A. (1998). Effect of hydrogen sulphide on liver somatic index and Fulton's condition factor in Mystus nemurus. *Journal of Fish Biology, 52*(1), 23-30. doi:10.1111/j.1095-8649.1998.tb01549.x

Horne, J. D., & Oblad, B. R. (1983). *Aquatic Toxicity Studies of Six Priority Pollutants* (4380). Retrieved from Houston, TX.

Hou, Z., Wen, H., Li, J., He, F., Li, Y., & Qi, X. (2020). Environmental hypoxia causes growth retardation, osteoclast differentiation and calcium dyshomeostasis in juvenile rainbow trout (Oncorhynchus mykiss). *Science of the Total Environment*(705). Retrieved from https://doi.org/10.1016/j.scitotenv.2019.135272

Houde, A. L. S., Akbarzadeh, A., Günther, O. P., Li, S., Patterson, D. A., Farrel, A. P., . . . Miller, K. M. (2019). Salmonid gene expression biomarkers indicative of physiological responses to changes in salinity and temperature, but not dissolved oxygen. *Journal of Experimental Biology, 222*. doi:10.1242/jeb.198036

Howell, P., & Buchanan, D. V. (1992). *Proceedings of the Gearhart Mountain Bull Trout Workshop.* Paper presented at the American Fisheries Society (USA). Oregon Chapter. Gearhart Mountain Bull Trout Workshop. Gearhart Mountain Wilderness, Or.(USA). 1989.

Howell, P. J., Dunham, J. B., & Sankovich, P. M. (2009). Relationships between watertemperatures and upstream migration, cold water refuge use, and spawning of adult bull trout from the Lostine River, Oregon, USA. *Ecology of Freshwater Fish, 19*, 96-106.

Hua, J., Buss, N., Kim, J., Orlofske, S. A., & Hoverman, J. T. (2016). Population-specific toxicity of six insecticides to the trematode Echinoparyphium sp. *Parasitology, 143*(5), 542-550.

Huang, C., Dong, Q., & Ma, R. (2006). Nonylphenol: a Toxicant to Hybrid Tilapia (Oreochromis niloticus x O. aureus) (in Chinese with English abstract). *Oceanologia Et Limnologia Sinica, 37*(4), 309-315.

Huff, M. H., Raphael, M. G., Miller, S. L., Nelson, S. K., & Baldwin, J. (2006). *Northwest Forest Plan—the first 10 years (1994-2003): status and trends of populations and nesting habitat for the marbled murrelet.* (PNW-GTR-650). Portland, OR Retrieved from https://www.fs.usda.gov/treesearch/pubs/23161

Hull, C. L., Kaiser, G. W., Lougheed, C., Lougheed, L., Boyd, S., & Cooke, F. (2001). Intraspecific variation in commuting distance of marbled murrelets (*Brachyramphus marmoratus*): ecological and energetic consequences of nesting further inland. *The Auk, 118*(4), 1036-1046.

Hunt, J. W., Anderson, B. S., Phillips, B. M., Tjeerdema, R. S., Puckett, H. M., Stephenson, M., . . . Watson, D. (2002). Acute and chronic toxicity of nickel to marine organisms: Implications for water quality criteria. *Environmental Toxicology and Chemistry, 21*, 2423-2430. Retrieved from https://doi.org/10.1002/etc.5620211122

Hunter, R., Faulkner, L., Culver, F., & Hill, J. (1985). Draft user manual for the QSAR system. Bozeman, MT: Montana State University.

Hutchinson, T. H., Williams, T. D., & Eales, G. J. (1994). Toxicity of cadmium, hexavalent chromium and copper to marine fish larvae (Cyprinodon variegatus) and copepods (Tisbe battagliai). *Marine Environmental Research, 38*(4), 275-290. doi:10.1016/0141-1136(94)90028-0

IMST. (2000). *Salmon abundances and effects of harvest: implications for rebuilding stocks of wild coho salmon in Oregon. Technical report 2000-3*.

Ingebrigtsen, A. (1929). Whales caught in the North Atlantic and other seas. *Rapports et Process-verbaux des reunions, Conseil Permanent International pour l'Exploration de la Mer, LVI*, 1-26.

Ishibashi, H., Hirano, M., Matsumura, N., Watanabe, N., Takao, Y., & Arizono, K. (2006). Reproductive effects and bioconcentration of 4-nonylphenol in medaka fish (Oryzias latipes). *Chemosphere, 65*(6), 1019-1026. doi:10.1016/j.chemosphere.2006.03.034

Isidori, M., Lavorgna, M., Nardelli, A., & Parrella, A. (2006). Toxicity on crustaceans and endocrine disrupting activity on Saccharomyces cerevisiae of eight alkylphenols. *Chemosphere, 64*(1), 135-143. doi:10.1016/j.chemosphere.2005.10.047

Ivanov, V., Usenko, T., & Parkhomenko, A. (1976). Effect of Hydrogen Sulfide on the Survival Rate of Eggs and Embryonal Mitoses of the Black Sea Turbot. *Hydrobiological Journal, 12*, 60 - 62.

Ivey, C. D., Besser, J. M., Ingersoll, C. G., Wang, N., Rogers, D. C., Raimondo, S., . . . Hammer, E. J. (2017). Acute sensitivity of the vernal pool fairy shrimp, Branchinecta lynchi (Anostraca; Branchinectidae), and surrogate species to 10 chemicals. *Environmental Toxicology and Chemistry, 36*(3), 797-806. doi:10.1002/etc.3723

Izatt, R. M., Christensen, J. J., Pack, R. T., & Bench, R. (1962). Thermodynamics of Metal-Cyanide Coördination. I. pK, ΔH0, and ΔS0 Values as a Function of Temperature for Hydrocyanic Acid Dissociation in Aqueous Solution. *Inorganic Chemistry, 1*(4), 828-831.

Janz, D. M., DeForest, D. K., Brooks, M. L., Chapman, P. M., Gilron, G., Hoff, D., . . . Palace, V. P. (2010). Selenium toxicity to aquatic organisms. *Ecological assessment of selenium in the aquatic environment*, 141-231.

Jarvinen. (1982). *MRID 154721*.

Jarvinen, A. W., & Ankley, G. T. (1999). *Linkage of effects to tissue residues: Development of a comprehensive database for aquatic organisms exposed to inorganic and organic chemicals*. Pensacola, FL: SETAC Press.

Jarvinen, A. W., Nordling, B. R., & Henry, M. E. (1983). Chronic toxicity of Dursban (chlorpyrifos) to the fathead minnow (Pimephales promelas) and the resultant acetylcholinesterase inhibition. *Ecotoxicol Environ Saf, 7*(4), 423-434. doi:10.1016/0147-6513(83)90008-8

Jarvinen, A. W., & Tanner, D. K. (1982). Toxicity of selected controlled release and corresponding unformulated technical grade pesticides to the fathead minnow Pimephales promelas. *Environmental Pollution Series A, Ecological and Biological, 27*(3), 179-195.

Jensen, T., Lawler, S. P., & Dritz, D. A. (1999). Effects of ultra-low volume pyrethrin, malathion, and permethrin on nontarget invertebrates, sentinel mosquitoes, and mosquitofish in seasonally impounded wetlands. *Journal of the American Mosquito Control Association, 15*(3), 330-338.

Johnson, D. W. (2006). Predation, habitat complexity, and variation in density-dependent mortality of temperate reef fishes. *Ecology, 87*(5), 1179-1188. doi:10.1890/0012-9658(2006)87[1179:phcavi]2.0.co;2

Johnson, J. H., & Wolman, A. A. (1984). The humpback whale, *Megaptera novaeangliae*. *Mar Fish Rev, 46*(4), 30-37.

Johnson, L., Bravo, C., O'Neill, S., West, J., Myers, M. S., Ylitalo, G., . . . Collier, T. K. (2010). *A toxics-focused biological observing system for Puget Sound. Developed by the Washington Department of Fish and Wildlife and NOAA Fisheries for the Puget Sound Partnership.* . Retrieved from Olympia, WA:

Johnson, S. W., Murphy, M. L., & Csepp, D. J. (2003). Distribution, habitat, and behavior of rockfishes, *Sebastes* spp., in nearshore waters of southeastern Alaska: observations from a remotely operated vehicle. *Env Biol Fish, 66*, 259-270.

Johnson, W. W., & Finley, M. T. (1980). *Handbook of Acute Toxicity of Chemicals to Fish and Aquatic Invertebrates*. (Resour. Publ. 137). Washington, D.C.: U.S. Fish and Wildlife Service

Johnston, R. K., Halkola, H., & George, R. (2003). Assessing the Ecological Risk of Creating Artificial Reefs from Ex-Warships. In C. Gauthier, R. Wild, W. Bell, & M. Martore (Eds.), *MTS/IEEE Proceedings OCEANS 2003* (Vol. 2, pp. 804 - 811). Columbia, MD: Marine Technology Society and Institute of Electrical and Electronics Engineers.

Jones, D. R. (1971). Theoretical analysis of factors which may limit the maximum oxygen uptake of fish: the oxygen cost of the cardiac and branchial pumps. *J. Theor. Biol, 32*, 341-349.

Jop, K. M. (1989). Acute and Rapid-Chronic Toxicity of Hexavalent Chromium to Five Marine Species. In U. M. Cowgill & L. R. Williams (Eds.), (pp. 251-260). West Conshohocken, PA: ASTM International.

Jop, K. M., Rodgers, J. H., Dorn, P. B., & Dickson, K. L. (1986). Use of Hexavalent Chromium as a Reference Toxicant in Aquatic Toxicity Tests. In T. M. Poston & R. Purdy (Eds.), (pp. 390-403). West Conshohocken, PA: ASTM International.

Jordahl, D. M., & Benson, A. (1987). Effect of Low pH on Survival of Brook Trout Embryos and Yolk-Sac Larvae in West Virginia Streams. *Trans Amer Fish, 116*(6), 807-816.

Jurasz, C. M., & Jurasz, V. P. (1979). Feeding modes of the humpback whale, *Megaptera novaeangliae*, in southeast Alaska. *Sci Rep Whales Res Inst, 31*, 69-83.

Kahl, M. D., Makynen, E. A., Kosian, P. A., & Ankley, G. T. (1997). Toxicity of 4-Nonylphenol in a Life-Cycle Test with the MidgeChironomus tentans. *Ecotoxicology and Environmental Safety, 38*(2), 155-160.

Kaligis, F., & Lasut, M. (1997). Effects of salinity and diazinon on the abalone Haliotis varia(Gastropoda: Haliotidae). *Spec. Publ. Phuket Mar. Biol. Cent., 17*(1), 115-120.

Kaptaner, B., & Unal, G. (2011). Effects of 17alpha-ethynylestradiol and nonylphenol on liver and gonadal apoptosis and histopathology in Chalcalburnus tarichi. *Environ Toxicol, 26*(6), 610-622. doi:10.1002/tox.20585

Katz, M. (1961). Acute toxicity of some organic insecticides to three species of salmonids and to the threespine stickleback. *Transactions of the American Fisheries Society, 90*(3), 264-268.

Kaur, K., & Toor, H. (1977). Toxicity of pesticides to embryonic stages of Cyprinus carpio communis Linn. *Indian journal of experimental biology*.

Keen, P. L., Higgs, D. A., Hall, K. J., & Ikonomou, M. (2005). Effects of dietary exposure of 4-nonylphenol on growth and smoltification of juvenile coho salmon (Oncorhynchus kisutch). *Sci Total Environ, 349*(1-3), 81-94. doi:10.1016/j.scitotenv.2005.01.035

Kelly, S. A., & Di Giulio, R. T. (2000). Developmental Toxicity of Estrogenic Alkylphenols in Killifish (*Fundulus heteroclitus*). *Environmental Toxicology and Chemistry, 19*, 2564-2570.

Kennedy, C. J., & Ross, P. S. (2012). Stress syndromes: heightened bioenergetic costs associated with contaminant exposure at warm temperatures in teleosts. *Integrated Environmental Assessment and Management, 8*(1), 202-204.

Kerby, J. L. (2006). *Pesticide effects on amphibians : a community ecology perspective.* (Ph.D.). University of California, Davis, Davis, CA. Retrieved from http://books.google.com/books?id=jkd3cJJ0dd0C Available from http://worldcat.org /z-wcorg/ database.

Kharkar, D., Turekian, K., & Bertine, K. (1968). Stream supply of dissolved silver, molybdenum, antimony, selenium, chromium, cobalt, rubidium and cesium to the oceans. *Geochimica et Cosmochimica Acta, 32*(3), 285-298.

Khattat, F. H., & Farley, S. (1976). *Acute toxicity of certain pesticides to Acartia tonsa Dana*: US Environmental Protection Agency, Office of Research and Development.

Kiemer, M. C. B., Black, K. D., Lussot, D., Bullock, A. M., & Ezzi, I. (1995). The effects of chronic and acute exposure to hydrogen sulphide on Atlantic salmon (Salmo salar L.). *Aquaculture, 135*(4), 311-327. doi:10.1016/0044-8486(95)01025-4

Klumov, S. K. (1963). Food and helmonthofauna of the baleen whales (Mysteceti) in the world ocean. *Trudy Inst Okeanol Acad Sci USSR, 71*, 94-104.

Knauer, G. A. (1977). Immediate industrial effects on sediment metals in a clean coastal environment. *Marine Pollution Bulletin, 8*(11), 249-254.

Korn, S., & Earnest, R. (1974). Acute toxicity of twenty insecticides to a Striped Bass (Morone saxatilis). *California Fish and Game, 60*(3), 128-131.

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. *Mar Pollut Bull, 54*(12), 1903-1911. doi:10.1016/j.marpolbul.2007.08.015

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. *Mar Poll Bull, 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. *Mar Poll Bull, 58*, 1522-1529.

Kramer, J. R., Adams, N. W. H., Manalopoulos, H., & Collins, P. V. (1999). Silver at an old mining camp, Cobalt, Ontario, Canada. *Environmental Toxicology and Chemistry, 18*(1), 23-29. Retrieved from http://dx.doi.org/10.1002/etc.5620180104

Krieger, K., & Wing, B. L. (1984). *Hydroacoustic surveys and identification of humpback whale forage in Glacier Bay, Stephens Passage, and Frederick Sound, summer 1983. NMFSINWC-66*.

Krieger, K., & Wing, B. L. (1986). *Hydroacoustic monitoring of prey to determine humpback whale movements. NMFSNWC-89*

Ku, Y., Chang, J.-L., & Cheng, S.-C. (1998). Effect of solution pH on the hydrolysis and photolysis of diazinon in aqueous solution. *Water Air and Soil Pollution, 108*(3-4), 445-456.

Kucukbay, Z., Yazlak, H., Sahin, N., Tuzcu, M., Cakmak, M. N., Gurdogan, F., . . . Sahin, K. (2006). Zinc picolinate supplementation decreases oxidative stress in rainbow trout (Oncorhynchus mykiss). *Aquaculture, 257*(1-4), 465-469.

Kuletz, K. J. (2005). *Foraging behavior and productivity of a non-colonial seabird, the marbled murrelet (Brachyramphus marmoratus), relative to prey and habitat.* (Doctor of Philosophy). University of Victoria,

Kumari, B., Kumar, V., Sinha, A. K., Ahsan, J., Ghosh, A., Wang, H., & DeBoeck, G. (2017). Toxicology of arsenic in fish and aquatic systems. *Environmental Chemistry Letters, 15*(1), 43-64.

Kutty, M. N. (1968). Respiratory quotients in goldfish and rainbow trout. *J. Fish. Res. Board Can., 25*, 1689-1728.

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*.

Laetz, C. A., Baldwin, D. H., Collier, T. K., Hebert, V., Stark, J. D., & Scholz, N. L. (2009). The synergistic toxicity of pesticide mixtures: implications for risk assessment and the conservation of endangered Pacific salmon. *Environ Health Perspect, 117*(3), 348-353. doi:10.1289/ehp.0800096

Land, L. F., & Brown, M. F. (1996). *Water Quality assessment of the Trinity River Basin, Texas - Pesticides in streams draining an urban and an agricultural area, 1993-95* (USGS Report 96-4114). Retrieved from Austin, TX.

Landrum, P. F., Eadie, B. J., & Faust, W. R. (1991). Toxicokinetics and toxicity of a mixture of sediment-associated polycyclic aromatic hydrocarbons to the amphipod Diporeia sp. *Environmental Toxicology and Chemistry: An International Journal, 10*(1), 35-46.

Landrum, P. F., Lydy, M. J., & Lee, H. (1992). Toxicokinetics in aquatic systems: Model comparisons and use in hazard assessment. *Environmental Toxicology and Chemistry, 11*(12), 1709-1725. doi:10.1002/etc.5620111205

Leary, R. F., & F.W., A. (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:doi:10.1577/1548-8659(1997)126<0715:NGCOSB>2.3.CO

Leathe, S. A., & Graham, P. J. (1982). *Flathead Lake fish habits study - Final Report*. Retrieved from Kalispell, Montana.

Leblanc, G. A., Mastone, J. D., Paradice, A. P., Wilson, B. F., Lockhart, J., H.B., & Robillard, K. A. (1984). The influence of speciation on the toxicity of silver to fathead minnow (*Pimephales promelas*). *Environmental Toxicology and Chemistry, 3*(1), 37-46. Retrieved from https://setac.onlinelibrary.wiley.com/doi/10.1002/etc.5620030106

Lee, K.-W., Raisuddin, S., Hwang, D.-S., Park, H. G., Dahms, H.-U., Ahn, I.-Y., & Lee, J.-S. (2008). Two-generation toxicity study on the copepod model species Tigriopus japonicus. *Chemosphere, 72*(9), 1359-1365.

Lei, X., Tian, P., Tang, D., Zhang, J., & Cang, R. (2014). Acute toxicity of nonylphenol to Misgurnus anguillicaudatus (in Chinese with English abstract). *Jiangsu Agricultural Sciences, 42*, 306-307.

Leis, M., Manfra, L., Taddia, L., Chicca, M., Trentini, P., & Savorelli, F. (2014). A comparative toxicity study between an autochthonous Artemia and a non native invasive species. *Ecotoxicology, 23*(6), 1143-1145.

Lejczak, B. (1977). Effect of insecticides: chlorphenvinphos, carbaryl and propoxur on aquatic organisms. *Polish Archives of Hydrobiology, 24*(4), 583-591.

Lenarz, W. H., Larson, R. J., & Ralston, S. (1991). Depth distributions of late larvae and pelagic juveniles of some fishes of the California current. *CalCOFI Rep, 32*, 41-46.

Leonard, E. M., Banerjee, U., D'Silva, J. J., & Wood, C. M. (2014). Chronic nickel bioaccumulation and sub-cellular fractionation in two freshwater teleosts, the round goby and the rainbow trout, exposed simultaneously to waterborne and dietborne nickel. *Aquatic Toxicology, 154*, 141-153.

Lerner, D. T., Bjornsson, B. T., & McCormick, S. D. (2007). Larval exposure to 4-nonylphenol and 17beta-estradiol affects physiological and behavioral development of seawater adaptation in Atlantic salmon smolts. *Environ Sci Technol, 41*(12), 4479-4485. doi:10.1021/es070202w

Levin, P. S., Coyer, J. A., Petrik, R., & Good, T. P. (2002). Community-wide effects of nonindigenous species on temperate rocky reefs. *Ecology, 83*(11), 3182-3193.

Li, D., Lu, C., Wang, J., Cao, Z., Sun, D., & Ma, X. (2009). Developmental mechanisms of arsenite toxicity in zebrafish (Danio rerio) embryos. *Aquatic Toxicology, 91*(3). doi:10.1016/j.aquatox.2008.11.007

Liber, K., Knuth, M. L., & Stay, F. S. (1999). An integrated evaluation of the persistence and effects of 4-nonylphenol in an experimental littoral ecosystem. *Environmental Toxicology and Chemistry, 18*(3), 357-362.

Lin, L. L., & Janz, D. M. (2006). Effects of binary mixtures of xenoestrogens on gonadal development and reproduction in zebrafish. *Aquat Toxicol, 80*(4), 382-395. doi:10.1016/j.aquatox.2006.10.004

Lincoln, A. E., Shaffer, J. A., & Quinn, T. P. (2018). Opportunistic use of estuarine habitat by juvenile bull trout, *Salvelinus confluentus*, from the Elwha River before, during, and after dam removal. *Environmental Biology of Fishes, 101*, 1559-1569. doi:doi: 10.1007/s10641-018-0800-9

Lintott, D. R. (1992a). *Carbaryl technical: acute toxicity to the mysid, Mysidopsis bahia, under flow-through test conditions*. Retrieved from Research Triangle Park, NC:

Lintott, D. R. (1992b). *Carbaryl technical: acute toxicity to the sheepshead minnow, Cyprinodon variegatus, under flow-through test conditions*. Retrieved from Research Triangle Park, NC:

Liu, F.-J., Wang, J.-S., & Theodorakis, C. W. (2006). Thyrotoxicity of sodium arsenate, sodium perchlorate, and their mixture in zebrafish Danio rerio. *Environmental science & technology, 40*(10), 3429-3436.

Logerwell, E. A., Mantua, N., Lawson, P. W., Francis, R. C., & Agostini, V. N. (2003). Tracking environmental processes in the coastal zone for understanding and predicting Oregon coho (Oncorhynchus kisutch) marine survival. *Fisheries Oceanography, 12*, 554-568.

Long, E. R., Dutch, M., Aasen, S., Welch, K., & Hameedi, M. J. (2005). Spatial extent of degraded sediment quality in Puget Sound (Washington State, USA) based upon measures of the sediment quality triad. *Environmental monitoring and assessment, 111*(1), 173-222.

Lorz, H., Glenn, S., Williams, R., Kunkel, C., Norris, L., & Loper, B. (1979). *Effects of selected herbicides on smolting of coho salmon*. (EPA-600/3-79-071). Corvallis, OR: Environmental Research Laboratory

Love, M. S. (1996). *Probably more than you want to know about the fishes of the Pacific coast*. Santa Barbara, CA: Really Big Press.

Love, M. S. (2002). *Rockfishes of the northeast Pacific*: University of California Press.

Love, M. S., Carr, M. H., & Haldorson, L. J. (1991). The ecology of substrate-associated juveniles of the genusSebastes. *Environmental Biology of Fishes, 30*(1-2), 225-243. doi:10.1007/bf02296891

Love, M. S., & Yoklavich, M. (2008). Habitat characteristics of juvenile cowcod, *Sebastes levis* (Scorpaenidae), in Southern California. *Environ Biol Fish, 82*, 195-202.

Love, M. S., Yoklavich, M., & Thorsteinson, L. (2002). *Rockfishes of the Northeast Pacific*. Berkeley, CA: University of California Press.

Lowden, G., Saunders, C., & Edwards, R. (1969). Organo-chlorine insecticides in water--part II. *Water Treat. Exam., 18*, 275-287.

Lu, X., Gu, Y., & Song, Y. (2012). Toxicity and tissue accumulation of nonylphenol in Carassius auratus red variety, Grass carp and Sliver carp (in Chinese with English abstract). *Journal of Hygiene Research, 41*(5), 785-789.

Luís, L. G., Ferreira, P., Fonte, E., Oliveira, M., & Guilhermino, L. (2015). Does the presence of microplastics influence the acute toxicity of chromium (VI) to early juveniles of the common goby (Pomatoschistus microps)? A study with juveniles from two wild estuarine populations. *Aquatic toxicology, 164*, 163-174.

Luo, Q., Ban, M., Ando, H., Kitahashi, T., Bhandari, R. K., McCormick, S. D., & Urano, A. (2005). Distinct effects of 4-nonylphenol and estrogen-17β on expression of estrogen receptor α gene in smolting sockeye salmon. *Comparative Biochemistry and Physiology Part C: Toxicology & Pharmacology, 140*(1), 123-130.

Luoma, S. N., & Rainbow, P. S. (2005). Why is metal bioaccumulation so variable? Biodynamics as a unifying concept. *Environmental Science & Technology, 39*(7), 1921-1931.

Lussier, S., Gentile, J., & Walker, J. (1985). Acute and chronic effects of heavy metals and cyanide on Mysidopsis bahia (Crustacea: Mysidacea). *Aquatic Toxicology, 7*(1-2), 25-35.

Lussier, S. M., & Cardin, J. A. (1985). *Results of Acute Toxicity Tests Conducted with Silver at ERL, Narragansett*. Narragansett, RI, USA: USEPA.

Lussier, S. M., Champlin, D., Livolsi, J., Poucher, S., & Pruell, R. J. (2000). Acute toxicity of para-nonylphenol to saltwater animals. *Environmental Toxicology and Chemistry, 19*, 617-621.

Lussier, S. M., Champlin, D., LiVolsi, J., Poucher, S., & Pruell, R. J. (2000). Acute toxicity of para-nonylphenol to saltwater animals. *Environmental Toxicology and Chemistry: An International Journal, 19*(3), 617-621.

MacCall, A. D. (2002). *Status of bocaccio off California in 2002*. Retrieved from Santa Cruz, CA:

Macek, K. J., Lindberg, M. A., Sauter, S., Buxton, K. S., & Costa, P. A. (1976). *Toxicity of Four Pesticides to Water Fleas and Fathead Minnows: Acute and Chronic Toxicity of Acrolein, Heptachlor, Endosulfan, and Trifluralin to the Water Flea (Daphnia magna) and the Fathead Minnow (Pimephales promelas)*. (EPA-600/3-76-099). Duluth, MN: USEPA.

MacGregor, J. S. (1970). *Fecundity, multiple spawning, and description of the gonads in Sebastodes.* Retrieved from Washington, DC.

Máchová, J., Prokeš, M., Peňáz, M., Baruš, V., & Kroupová, H. (2010). Toxicity of Diazinon 60 EC for embryos and larvae of tench, Tinca tinca (L.). *Reviews in fish biology and fisheries, 20*(3), 409-415.

Maddock, B., & Taylor, D. (1976). The acute toxicity of dissolved elemental phosphorus to cod (Gadus morhua). *Water Research, 10*(4), 289-294.

Madsen, S. S., Skovbolling, S., Nielsen, C., & Korsgaard, B. (2004). 17-Beta estradiol and 4-nonylphenol delay smolt development and downstream migration in Atlantic salmon, Salmo salar. *Aquat Toxicol, 68*(2), 109-120. doi:10.1016/j.aquatox.2004.03.008

Magnoni, L. J., Eding, E., Leguen, I., Prunet, P., Geurden, I., Ozório, R. O. A., & Schrama, J. W. (2018). Hypoxia, but not an electrolyte-imbalanced diet, reduces feed intake, growth and oxygen consumption in rainbow trout (Oncorhynchus mykiss). *Sci Rep, 1*(8), 4965.

Mansour, M., Feicht, E. A., Behechti, A., Schramm, K. W., & Kettrup, A. (1999). Determination photostability of selected agrochemicals in water and soil. *Chemosphere, 39*(4), 575-585. doi:10.1016/s0045-6535(99)00123-x

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.

Marcial, H. S., Hagiwara, A., & Snell, T. W. (2005). Effect of some pesticides on reproduction of rotifer Brachionus plicatilis Müller. In *Rotifera X* (pp. 569-575): Springer.

Marshall, D. B. (1992). *Sensitive vertebrates of Oregon*: Oregon Department of Fish and Wildlife.

Martinez, D., De Lázaro, O., Cortés, P., Oyarzún-Salazar, R., Paschke, K., & Vargas-Chacoff, L. (2020). Hypoxia modulates the transcriptional immunological response in Oncorhynchus kisutch. *Fish & Shellfish Immunology, 106*, 1042-1051. Retrieved from https://doi.org/10.1016/j.fsi.2020.09.025

Mason, A., Burger, A. E., & Hansen, B. (2002). At-sea surveys of marbled murrelets in Clayoquot Sound, 1996-2000. In A. E. Burger & T. A. Chatwin (Eds.), *Multi-scale Studies of Populations, Distribution and Habitat Associations of Marbled Murrelets in Clayoquot Sound, British Columbia* (pp. 15-30). Victoria, BC: Ministry of Water, Land and Air Protection.

Mattiasen, E. G., Kashef, N. S., Stafford, D. M., Logan, C. A., Sogard, S., Bjorkstedt, E. P., & Hamilton, S. L. (2020). Effects of hypoxia on the behavior and physiology of kelp forest fishes. *Global Chang Biology, 6*(26), 3498-3511. doi:doi: 10.1111/gcb.15076

Mauger, G. S., Casola, J. H., Morgan, H. A., Strauch, R. L., Jones, B., Curry, B., . . . Snover, A. K. (2015). *State of Knowledge: Climate Change in Puget Sound*. Retrieved from Seattle, WA: https://cig.uw.edu/resources/special-reports/ps-sok/

Mayer, F., & Ellersieck, M. (1986). *Manual of Acute Toxicity: Interpretation and Data Base for 410 Chemicals and 66 Species of Freshwater Animals*. Washington, D.C.

Mayer, F. L. (2008). *Endangered aquatic vertebrates : comparative and probabilistic-based toxicology*. (EPA/600/R-08/045). Washington, DC: U.S. Environmental Protection Agency, Office of Research and Development Retrieved from https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=500025UB.txt

Mayer, F. L., Buckler, D. R., Dwyer, F. J., Ellersieck, M. R., Sappington, L. C., Besser, J. M., & Bridges, C. M. (2008). *Endangered Aquatic Vertebrates: Comparative and Probabilistic-Based Toxicology*. (EPA/600/R-08/045). Washington, D.C.: Office of Research and Development

Mayer, F. L., & Ellersieck, M. R. (1986a). *Manual of acute toxicity: interpretation and data base for 410 chemicals and 66 species of freshwater animals*: US Department of the Interior, Fish and Wildlife Service Washington, DC.

Mayer, F. L., & Ellersieck, M. R. (1986b). *Manual of Acute Toxicity: Interpretation and Data Base for 410 Chemicals and 66 Species of Freshwater Animals*. (Resour. Publ. No. 160). Washington, D.C.: U.S. Fish and Wildlife Service

Mayer Jr., F. L. (1987). *Acute Toxicity Handbook of Chemicals to Estuarine Organisms*. (EPA 600/8-87-017). Gulf Breeze, FL.

McBride, A. (2007). *Turners Bay Change Analysis*. Retrieved from La Conner, WA: https://srpconnect.rco.wa.gov/services/project/file?sid=280&pid=11865&id=18776

McCarty, L., Landrum, P., Luoma, S., Meador, J., Merten, A., Shephard, B., & Van Wezel, A. (2011). Advancing environmental toxicology through chemical dosimetry: External exposures versus tissue residues. *Integrated Environmental Assessment and Management, 7*(1), 7-27.

McCarty, L. S., & Mackay, D. (1993). Enhancing ecotoxicological modeling and assessment. Body residues and modes of toxic action. *Environmental Science & Technology, 27*(9), 1718-1728.

McClatchie, S., Goericke, R., Cosgrove, R., Auad, G., & Vetter, R. (2010). Oxygen in the Southern California Bight: Multidecadal trends and implications for demersal fisheries. *Geophysical Research Letters, 37*(19). doi:doi:10.1029/2010GL044497

McConnell, L. L., LeNoir, J. S., Datta, S., & Seiber, J. N. (1998). Wet deposition of current-use pesticides in the Sierra Nevada mountain range, California, USA. *Environmental Toxicology and Chemistry, 17*(10), 1908-1916.

McCormick, S. D., O'Dea M, F., Moeckel, A. M., Lerner, D. T., & Bjornsson, B. T. (2005). Endocrine disruption of parr-smolt transformation and seawater tolerance of Atlantic salmon by 4-nonylphenol and 17beta-estradiol. *Gen Comp Endocrinol, 142*(3), 280-288. doi:10.1016/j.ygcen.2005.01.015

McCulloch, W. L., & Rue, W. J. (1989). Evaluation of a Seven-Day Chronic Toxicity Estimation Test Using Cyprinodon variegatus. In U. M. Cowgill & L. R. Williams (Eds.), (pp. 355-364). West Conshohocken, PA: ASTM International.

McGeer, J. C., Brix, K. V., Skeaff, J. M., DeForest, D. K., Brigham, S. I., Adams, W. J., & Green, A. (2003). Inverse relationship between bioconcentration factor and exposure concentration for metals: implications for hazard assessment of metals in the aquatic environment. *Environmental Toxicology and Chemistry: An International Journal, 22*(5), 1017-1037.

McKenney, C., Matthews, E., & Lawrence, D. (1981). *Chronic Toxicity Testing and Physiological Studies. A. Chronic Toxicity Tests: Invertebrates. 1. Mysid Life-Cycle Toxicity Tests*. Gulf Breeze, FL

McLeese, D., Zitko, V., Metcalfe, C., & Sergeant, D. (1980). Lethality of aminocarb and the components of the aminocarb formulation to juvenile Atlantic salmon, marine invertebrates and a freshwater clam. *Chemosphere, 9*(2), 79-82.

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 Opporunitites*. Retrieved from Vancouver, BC:

Mearns, A. J., Oshida, P. S., Sherwood, M. J., Young, R., & Reish, D. J. (1976). Chromium effects on coastal organisms. *J Water Pollut Control Fed, 48*(8), 1929-1939. Retrieved from https://www.ncbi.nlm.nih.gov/pubmed/966358

Mebane, C. A. (2006). *Cadmium risks to freshwater life: Derivation and validation of low-effect criteria values using laboratory and field studies*. Retrieved from

Mebane, C. A. (2010). *Cadmium risks to freshwater life: Derivation and validation of low-effect criteria values using laboratory and field studies*. Retrieved from

Mebane, C. A., Dillon, F. S., & Hennessy, D. P. (2012). Acute Toxicity of Cadmium, Lead, Zinc, and their Mixtures to Stream-Resident Fish and Invertebrates. *Environmental Toxicology and Chemistry, 31*(6), 1334-1348. doi:https://doi.org/10.1002/etc.1820

Mebane, C. A., Hennessy, D. P., & Dillon, F. S. (2008). Developing acute-to-chronic toxicity ratios for lead, cadmium, and zinc using rainbow trout, a mayfly, and a midge. *Water, Air, and Soil Pollution, 188*(1-4), 41.

Meyer, J. S., Adams, W.J., Brix, K.V., Luoma, S.N., Mount, D.R., Stubblefield, W.A. and Wood, C.M. (2005). *Toxicity of dietborne metals to aquatic organisms*. Pensacola, FL: SETAC Press.

Mhadhbi, L., & Boumaiza, M. (2012a). Toxic effects of acute exposure of Diazinon in turbot (Psetta maxima) early life stage (ELS).

Mhadhbi, L., & Boumaiza, M. (2012b). Toxic effects of acute exposure of Diazinon in turbot (Psetta maxima) early life stage (ELS). *International Journal of Environmental Research, 6*(1), 139-144.

Micheletti, C., Critto, A., Carlon, C., & Marcomini, A. (2004). Ecological risk assessment of persistent toxic substances for the clam Tapes philipinarum in the Lagoon of Venice, Italy. *Environ Toxicol Chem, 23*(6), 1575-1582. doi:10.1897/03-331

Middelburg, J. J., Soetaert, K., & Hagens, M. (2020). Ocen Alkalinity, Buffering and Biogeochemical Processes. *Reviews of Geophysics, 58*(3). Retrieved from https://doi.org/10.1029/2019RG000681

Miller, B. S., & Borton, S. F. (1980). *Geographical distribution of Puget Sound fishes: maps and data source sheets*. Retrieved from Seattle, WA.

Miller, D. C., Poucher, S. L., & Coiro, L. (2002). Determination of lethal dissolved oxygen levels for selected marine and estuarine fishes, crustaceans and a bivalve. *Marine Biology, 140*, 287-296.

Mobley, J. R., Bauer, G. B., & Herman, L. M. (1999). Changes over a ten-year interval in the distribution and relative abundance of humpback whales (*Megaptera novaeangliae*). *Aquat Mamm, 25*, 63-72.

Mohamed, A. M., El-Emam, M. A., Osman, G. Y., Abdel-Hamid, H., & Ali, R. E. (2012). Effect of Basudin, Selecron and the phytoalkaloid Colchicine (pesticides) on biological and molecular parameters of Biomphalaria alexandrina snails. *Pesticide biochemistry and physiology, 102*(1), 68-78.

Monson, P. (2010). *Aquatic Life Water Quality Standards Technical Support Document for Nonylphenol and Ethoxylates*. St. Paul, MN

Morales, M., Martínez-Paz, P., Martín, R., Planelló, R., Urien, J., Martínez-Guitarte, J. L., & Morcillo, G. (2014). Transcriptional changes induced by in vivo exposure to pentachlorophenol (PCP) in Chironomus riparius (Diptera) aquatic larvae. *Aquatic toxicology, 157*, 1-9.

Moreau, C. J., Klerks, P. L., & Haas, C. N. (1999). Interaction between phenanthrene and zinc and their toxicity to the sheepshead minnow (*Cyprinodon variegatus*). *Archives of Environmental Contamination and Toxicology, 37*(2). doi:doi: 10.1007/s002449900512

Morgan, H. G. (1976). *Sublethal Effects of Diazinon in Stream Invertebrates.* (Ph,D, Ph.D. Dissertation). University of Guelph, Guelph, Ontario, Canada.

Morgan, I. J., Henry, R. P., & Wood, C. M. (1997). The mechanism of acute silver nitrate toxicity in freshwater rainbow trout (Oncorhynchus mykiss) is inhibition of gill Na+ and Cl− 1 transport. *Aquatic toxicology, 38*(1-3), 145-163.

Mote, P. W., & Salathé, E. P., Jr. (2010). Future climate change in the Pacific Northwest. *Climate Change, 102*(1-2), 29-50. Retrieved from https://link.springer.com/article/10.1007/s10584-010-9848-z?no-access=true

Moulton, L. L., & Miller, B. S. (1987). *Characterization of Puget Sound marine fishes: survey of available data. FRI-UW-8716. Final report*. Retrieved from Seattle, WA.

Mount, D. I., & Stephan, C. E. (1967). A Method for Establishing Acceptable Toxicant Limits for Fish—Malathion and the Butoxyethanol Ester of 2,4-D. *Transactions of the American Fisheries Society, 96*(2), 185-193. doi:10.1577/1548-8659(1967)96[185:AMFEAT]2.0.CO;2

Mumford, T. F., Jr. (2007). *Kelp and eelgrass in Puget Sound. Prepared in support of the Puget Sound Nearshore Partnership. Technical report 2007-05*. Retrieved from Seattle, WA.

Munday, P. L., Dixson, D. L., Donelson, J. M., Jones, G. P., Pratchett, M. S., Devitsina, G. V., & Døving, K. B. (2009). Ocean acidification impairs olfactory discrimination and homing ability of a marine

fish. *Proceedings of the National Academy of Sciences, 106*(6), 1848-1852. doi:DOI: 10.1073/pnas.0809996106

Myers, J. M., Hard, J. J., Connor, E. J., Hayman, R. A., Kope, R. G., Lucchetti, G., . . . Thompson, B. E. (2015). *Identifying historical populations of steelhead within the Puget Sound distinct population segment. Technical Memorandum NMFS-NWFSC-128*. Retrieved from

Myers, J. M., Kope, R. G., Bryant, G. J., Teel, D., Lierheimer, L. J., Wainwright, T. C., . . . Waples, R. S. (1998). *Status review of chinook salmon from Washington, Idaho, Oregon, and California. NOAA Technical Memorandum NMFS-NWFSC-35*. Retrieved from Seattle, WA:

Natale, G. S., Ammassari, L., Basso, N., & Ronco, A. E. (2006). Acute and chronic effects of Cr (VI) on Hypsiboas pulchellus embryos and tadpoles. *Diseases of aquatic organisms, 72*(3), 261-267.

Navy, U. (2008). *Request for letter of authorization for the incidental harassment of marine mammals resulting from Navy training and research, development, testing, and evaluation activities conducted within the Southern California range complex. Submitted to Office of Protected Resources, NMFS*. Retrieved from

Navy, U. (2011). *Gulf of Alaska Navy training activities: preliminary final environmental impact statement/overseas environmental impact statement. Vol 1*. Retrieved from Pearl Harbor, HI:

Nebeker, A. V., Savonen, C., & Stevens, D. G. (1985). Sensitivity of rainbow trout early life stages to nickel chloride. *Environmental Toxicology and Chemistry, 4*(2), 233-239. Retrieved from https://doi.org/10.1002/etc.5620040214

Neff, J. M. (1997). Ecotoxicology of arsenic in the marine environment. *Environmental Toxicology and Chemistry, 16*(5), 917-927.

Nelson, M. C., & Nelle, R. D. (2008). *Seasonal movements of adult fluvial bull trout in the Entiat River, WA 2003-2006*. Leavenworth, Washington: U.S. Fish and Wildlife Service

Nelson, S. (2020). Marbled Murrelet (Brachyramphus marmoratus), version 1.0. *Birds of the World.* Retrieved from https://birdsoftheworld.org

Nelson, S. K. (1997). Marbled murrelet (*Brachyramphus marmoratus*). In P. G. Rodewald (Ed.), *The Birds of North America*. Ithaca, NY: Cornell Lab of Ornithology.

Nemoto, T. (1957). Foods of baleen whales in the northern Pacific. *Sci Rep Whales Res Inst Tokyo, 12*, 33-89.

Nemoto, T. (1959). Foods of baleen whales with reference to whale movements. *Sci Rep Whales Res Inst Tokyo, 12*, 149-290.

Nemoto, T. (1970). The feeding pattern of baleen whales in the ocean. In J. H. Steele (Ed.), *Marine food chains* (pp. 241-252). Edinburgh, Scotland: Oliver and Boyd.

Nendza, M., Herbst, T., Kussatz, C., & Gies, A. (1997). Potential for secondary poisoning and biomagnification in marine organisms. *Chemosphere, 35*(9), 1875-1885. doi:10.1016/s0045-6535(97)00239-7

Network, O. (2016). Whale Sighting Reports. Retrieved from http://archive.constantcontact.com/fs077/1101447505873/archive/1102327155336.html

Newman, M. C., Ownby, D. R., Mézin, L. C., Powell, D. C., Christensen, T. R., Lerberg, S. B., & Anderson, B. A. (2000). Applying species-sensitivity distributions in ecological risk assessment: Assumptions of distribution type and sufficient numbers of species. *Environmental Toxicology and Chemistry, 19*(2), 508-515.

Nightingale, B., & Simenstad, C. (2001). *Overwater Structures: Marine Issues*. (WA-RD 508.1). Seattle, WA: Washington State Transportation Center (TRAC), University of Washington Retrieved from https://www.wsdot.wa.gov/research/reports/fullreports/508.1.pdf

Nimmo, D., Hamaker, T., Matthews, E., & Moore, J. (1981). An overview of the acute and chronic effects of first and second generation pesticides on an estuarine mysid [Shrimp, Mysidopsis bahia]. In F.

J. Vernberg, A. Calabrese, F. P. Thurberg, & W. B. Vernberg (Eds.), *Biological Monitoring of Marine Pollutants* (pp. 3-19). New York, NY: Academic Press.

Nishiuchi, Y., & Hashimoto, Y. (1967). Toxicity of pesticide ingredients to some fresh water organisms. *Science of Pest Control, 32*(1), 5-11.

NMFS. (1991). *Final recovery plan for the humpback whale (Megaptera novaeangliae)*. Retrieved from Silver Spring, MD.

NMFS. (2006). *Biological assessment of the Alaska groundfish fisheries and NMFS managed Endangered Species Act listed marine mammals and sea turtles*. Retrieved from Juneau, AK:

NMFS. (2008). *Recovery plan for southern resident killer whales (Orcinus orca)*. Retrieved from Seattle, WA.

NMFS. (2010). *Recovery plan for the fin whale (Balaenoptera physalus). Final*. Retrieved from Washington, DC.

NMFS. (2011a). *Endangered Species Act (ESA) Section 7(a)(2) biological opinion: (1) The US Navy's proposed training activities on the Gulf of Alaska temporary maritime training area from April 2011 to April 2016; (2) Promulgation of regulations to authorize the Navy to "take" marine mammals incidental to training on the Gulf of Alaska temporary maritime training area from April 2011 to April 2016*. Retrieved from Juneau, AK.

NMFS. (2011b). *Southern resident killer whales (Orcinus orca) 5-year review: summary and evaluation*. Retrieved from Seattle, WA.

NMFS. (2012). *Jeopardy and Adverse Modification of Critical Habitat Biological Opinion for the EPA's Proposed Approval of Certain Oregon Administrative Rules Related to Revised Water Quality Criteria for Toxic Pollutants*. (NMFS No. 2008/00148). Seattle, Washington: NOAA

NMFS. (2014a). *Designation of Critical Habitat for the Distinct Population Segments of Yelloweye Rockfish, Canary Rockfish, and Bocaccio Final Section 4(b)(2) Report*. Seattle, WA Retrieved from https://repository.library.noaa.gov/view/noaa/18690

NMFS. (2014b). *Endangered Species Act Section 7(a)(2) Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat (EFH) Consultation, Idaho Water Quality Standards for Toxic Substances*. (NMFS No. 2000-1484). Seattle, Washington: NOAA

NMFS. (2015a, January 21, 2015). Chum salmon (*Oncorhynchus keta*). Retrieved from http://www.fisheries.noaa.gov/pr/species/fish/chum-salmon.html

NMFS. (2015b). *Killer whale (Orcinus orca): eastern North Pacific southern resident stock*. Retrieved from http://www.nmfs.noaa.gov/pr/sars/species.htm#smallwhales

NMFS. (2015c). *Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Pacific Northwest*. Retrieved from

NMFS. (2016a). *5-Year Review for Yelloweye rockfish (Sebastes ruberrimus), canary rockfish (Sebastes pinniger), and bocaccio (Sebastes paucispinis) of the Pudget Sound/Georgia Basin*.

NMFS. (2016b). *2015 Adult Sockeye Salmon Passage Report*. NOAA Fisheries Retrieved from https://www.westcoast.fisheries.noaa.gov/publications/hydropower/fcrps/2015_adult_sockeye_salmon_passage_report.pdf

NMFS. (2016c). *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. (2016d). *2016 5-year review: summary & evaluation of Upper Columbia River steelhead, Upper Columbia River spring-run chinook salmon*. Retrieved from Portland, OR:

NMFS. (2016e). *California Current Integrated Ecosystem Assessment (CCIEA): State of the California Current Report* Long Beach, CA: NOAA

NMFS. (2016f). *Critical habitat Hood Canal summer-run chum salmon*. Retrieved from http://www.westcoast.fisheries.noaa.gov/publications/gis_maps/maps/salmon_steelhead/critical_habitat/chum/chum_hcs.pdf

NMFS. (2016g). *Critical habitat Puget Sound Chinook salmon*. Retrieved from http://www.westcoast.fisheries.noaa.gov/publications/gis_maps/maps/salmon_steelhead/critical_habitat/chin/chinook_pug.pdf

NMFS. (2016h). *Southern resident killer whales (Orcinus orca) 5-year review: summary and evaluation*. Retrieved from Seattle, WA:

NMFS. (2016i). Steelhead trout (*Oncorhynchus mykiss*). Retrieved from http://www.fisheries.noaa.gov/pr/species/fish/steelhead-trout.html

NMFS. (2017a). *2016 5-Year Review : Summary & Evaluation of Puget Sound Chinook Salmon Hood Canal Summer-run Chum Salmon Puget Sound Steelhead*. Portland, OR Retrieved from https://repository.library.noaa.gov/view/noaa/17015

NMFS. (2017b, January 2017). Killer whale (*Orcinus orca*). Retrieved from http://www.nmfs.noaa.gov/pr/species/mammals/whales/killer-whale.html

NMFS. (2017c). Killer whales identification tips and vocalizations. Retrieved from http://www.westcoast.fisheries.noaa.gov/protected_species/marine_mammals/killer_whale/id_vocalizations.html

NMFS. (2018). *Biological Opinion on the Environmental Protection Agency's Approval of Florida's Proposed Water Quality Criteria for 4-Nonylphenol*. Silver Spring, MD: Office of Protected Resources, National Marine Fisheries Service

NOAA, USCG, USEPA, & API. (2010). *Characteristics of response strategies: a guide for spill response planning in marine environments*. Retrieved from Seattle, WA:

Norberg-King, T. (1987). Relative sensitivity of the fathead minnow (Pimephales promelas) seven day subchronic test to the early life stage and life cycle tests. In *Environmental research Laboratory—Duluth Deliverable 7164A*.

Norberg-King, T. J. (1987a). *An Evaluation of the Fathead Minnow Seven-Day Subchronic Test for Estimating Chronic Toxicity* (M.S Thesis). University of Wyoming, Laramie, WY.

Norberg-King, T. J. (1987b, August 31, 1987). [Toxicity Data on Diazinon, Aniline, 2,4-Dimethylphenol].

Norberg-King, T. J. (1989a). An evaluation of the fathead minnow seven-day subchronic test for estimating chronic toxicity. *Environmental Toxicology and Chemistry, 8*(11), 1075-1089.

Norberg-King, T. J. (1989b). An evaluation of the fathead minnow seven-day subchronic test for estimating chronic toxicity. *Environmental Toxicology and Chemistry: An International Journal, 8*(11), 1075-1089.

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. *Aquat Mamm, 42*(2), 168-183.

NRC. (2016). *Final Report Identifying Rockfish Hot Spot Areas in Puget Sound through a Spatial Analysis of "Grey" Data*. Retrieved from https://media.fisheries.noaa.gov/2021-10/nrc-rockfishhotspotsgis-2016.pdf

NWFSC. (2015). *Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Pacific Northwest*. Retrieved from

Nysewander, D. R., Evenson, J. R., Murphie, B. L., & Cyra, T. A. (2005). *Report of marine bird and mammal component, Puget Sound Ambient Monitoring Program, for July 1992 to December 1999 period*. Olympia, WA Retrieved from https://wdfw.wa.gov/publications/01135

Odden, K. (2014). *Marine Shoreline Protection Assessment for Skagit County*. Retrieved from Skagit County, Washington.

ODEQ. (1995). Temperature: 1992-1994 water quality standards review. In: ODEQ Portland.
Ogino, C., & Yang, G. Y. (1978). Requirement of rainbow trout for dietary zinc. *Bull. Jap. Sci. Fish., 44*, 1015-1018.
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. *Rep Int Whal Commn*(Special Issue 12), 209-243.
Oris, J. T., Winner, R. W., & Moore, M. V. (1991). A four-day survival and reproduction toxicity test for Ceriodaphnia dubia. *Environmental Toxicology and Chemistry, 10*(2), 217-224.
Orr, J. W., Brown, M. A., & Baker, D. C. (2000a). *Guide to rockfishes (Scorpaenidae) of the genera Sebastes, Sebastolobus, and Adelosebastes of the Northeast Pacific Ocean, second edition. NOAA technical memorandum NMFS-AFSC-117*. Retrieved from
Orr, J. W., Brown, M. A., & Baker, D. C. (2000b). *Guide to rockfishes (Scorpaenidae) of the genera Sebastes, Sebastolobus, and Adelosebastes of the Northeast Pacific Ocean, second edition. Technical Memorandum NMFS-AFSC-117*. Retrieved from
Orrell, R. F. (1976). *Skagit Chinook Race Differentiation Study*. Retrieved from
Oshida, P. S., & Reish, D. J. (1975). Effects of chromium on reproduction in polychaetes. *So. Calif. Coastal Water. Res. Proj, 1500*, 55-65.
Ott, S., & Maret, R. (2003). *Aquatic assemblages and their relationship to temperature variables of least-disturbed streams in the Salmon River, Basin, central Idaho, 2001.* Retrieved from Idaho:
Pacella, S. R., Brown, C. A., Waldbusser, G. G., Labiosa, R. G., & Hales, B. (2018). Seagrass habitat metabolism increases short-term extremes and lon-term offset of CO_2 under future ocean acidification. *Proceedings of the National Academy of Sciences of the USA, 115*(15), 3870-3875. doi:10.1073/pnas.1703445115
Palsson, W. A., Tsou, T.-S., Bargmann, G. G., Buckley, R. M., Jim E. West, Mills, M. L., . . . Pacunski, R. E. (2009). *The Biology and Assessment of Rockfishes in Puget Sound*. Washington Department of Fish and Wildlife Retrieved from https://wdfw.wa.gov/sites/default/files/publications/00926/wdfw00926.pdf
Parametrix. (2010). *Toxicity of lead to the sheepshead minnow Cyprinodon variegatus.* Retrieved from Corvallis, OR, USA.
Parrish, P. R., Dyar, E. E., Enos, J. M., Wilson., W. G., Parrish, P. R., Hansen, D. J., . . . Wilson, W. G. (1978). *Chronic toxicity of chlordane, trifluralin, and pentachlorphenol to sheepshead minnows (Cyprinodon variegatus)*. (EPA-600/3-78-010; PB-278 269). Gulf Breeze, Fla.: Environmental Research Laboratory, Office Of Research and Development, U.S. Environmental Protection Agency Retrieved from https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=9101B3R1.PDF
Paulson, A., Konrad, C., Frans, L., Noble, M., Kendall, C., Josberger, E., . . . Olsen, T. (2006). *Freshwater and Saline Loads of Inorganic Nitrogen to Hood Canal and Lynch Cove, Washington*. Retrieved from www.pubs.usgs.gov/sir/2006/5106
Payne, P. M., Wiley, D. N., Young, S. B., Pittman, S., Clapham, P. J., & Jossi, J. W. (1990). Recent fluctuations in the abundance of baleen whales in the southern Gulf of Maine in relation to changes in selected prey. *Fish Bull, 88*, 687-696.
Pedersen, C. L. (1987). Energy budgets for juvenile rainbow trout at various oxygen concentrations. *Aquaculture*(62), 289-298. Retrieved from https://doi.org/10.1016/0044-8486(87)90171-2
Pelletier, G., Roberts, M., Keyzers, M., & Alin, S. R. (2018). Seasonal variation in aragonite saturation in surface waters of Puget Sound - a pilot study. *Elementa Science of the Anthropocene, 6*(5). doi:https://doi.org/10.1525/elementa.270
Penttila, D. (2007). *Marine Forage Fishes in Puget Sound*. (2007-03). Olympia: Washington Department of Fish and Wildlife Retrieved from https://apps.dtic.mil/docs/citations/ADA477927

Perry, A., Baker, C. S., & Herman, L. M. (1985). *The natural history of humpback whales in Glacier Bay, Alaska. Final report to the National Park Service*. Retrieved from Anchorage, AK:

PFMC. (2020). *Salmon Fishery Management Plan Impacts to Southern Resident Killer Whales, FINAL DRAFT Risk Assessment*.

Phipps, G. L. (1988, April 29, 1988). [Diazinon acute tests for criteria development].

Phipps, G. L., & Holcombe, G. W. (1985, October 22, 1985). [Memorandum to C.E. Stephan].

Phipps, G. L., & Holcombe, G. W. (1985). A method for aquatic multiple species toxicant testing: acute toxicity of 10 chemicals to 5 vertebrates and 2 invertebrates. *Environmental Pollution Series A, Ecological and Biological, 38*(2), 141-157.

Pianka, E. R. (1983). *Evolutionary Ecology* (3rd ed.). New York, NY: Harper and Row.

Posthuma, L., van Gils, J., Zijp, M. C., van de Meent, D., & de Zwart, D. (2019). Species sensitivity distributions for use in environmental protection, assessment, and management of aquatic ecosystems for 12,386 chemicals. *Environmental Toxicology and Chemistry, 38*(4), 905-917.

Pratt, K. (1992). *A review of bull trout life history.* Paper presented at the Proceedings of the Gearhart Mountain bull trout workshop. Oregon Chapter of the American Fisheries Society, Corvallis.

Presser, T. S., & Luoma, S. N. (2010). A methodology for ecosystem-scale modeling of selenium. *Integrated Environmental Assessment and Management, 6*(4), 685-710.

Pressey, R. T. (1953). *The Sport Salmon Fishery for Salmon on Puget Sound*. Olympia, WA Retrieved from https://wdfw.wa.gov/publications/00920

PSEMP. (2020). *Puget Sound marine waters: 2020 overview*. Retrieved from Seattle, WA: https://www.psp.wa.gov/PSmarinewatersoverview.php

R_Core_Team. (2020). R: A Language and Environment for Statistical Computing. Vienna, Austria: R Foundation for Statistical Computing. Retrieved from https://www.R-project.org/

Raimondo, S., Lilavois, C. R., & Barron, M. G. (2015). *Web-based Interspecies Correlation Estimation (Web-ICE) for Acute Toxicity: User Manual. Version 3.3*. (EPA/600/R-15/192). Gulf Breeze, FL: U. S. Environmental Protection Agency, Office of Research and Development

Raimondo, S., Montague, B. J., & Barron, M. G. (2007). Determinants of variability in acute to chronic toxicity ratios for aquatic invertebrates and fish. *Environmental Toxicology and Chemistry: An International Journal, 26*(9), 2019-2023.

Raimondo, S., Montague, B. J., & Barron, M. G. (2007). Determinants of variability in acute to chronic toxicity ratios for aquatic invertebrates and fish. *Environ Toxicol Chem, 26*(9), 2019-2023. doi:10.1897/07-069R.1

Rajkumar, J. S. I. (2012). Marine organisms in toxicological approach for the assessment of environmental risk associated with Cd, Cu, Pb, and Zn. *Univ. J. Eniron. Res. Technol., 2*(2), 19-25.

Ralph, C. J., Hunt, G. L., Raphael, M. G., & Piatt, J. F. (1995). *Ecology and conservation of the Marbled Murrelet*. (PSW-GTR-152). Albany, CA Retrieved from https://www.fs.usda.gov/treesearch/pubs/27863

Relyea, R. A. (2009). A cocktail of contaminants: how mixtures of pesticides at low concentrations affect aquatic communities. *Oecologia, 159*(2), 363-376.

Reynolds, F. A., & Haines, T. A. (1980). Effects of chronic exposure to hydrogen sulphide on newly hatched brown trout Salmo trutta L. *Environmental Pollution Series A, Ecological and Biological, 22*(1), 11-17. doi:https://doi.org/10.1016/0143-1471(80)90077-X

Reynolds, W. W. (1977). Temperature as a Proximate Factor in Orientation Behavior. *J. Fish. Res. Board Can., 34*, 734-739.

Rich, C. F., Jr. (1996). *Influence of abiotic and biotic factors on occurrence of resident bull trout in fragmented habitats, western Montana*. Retrieved from Bozeman, MT:

Richards, L. J., Cass, A. J., Selsby, J. R., van der Broek, R., & Loynachan, S. D. (1985). *A summary of research catch and effort data for nearshore reef-fishes collected during 1984 in MSA 15 and 16 of the Strait of Georgia. Can. MS Rep. Fish Aquat. Sci. 1833*. Retrieved from

Riedel, R., Schlenk, D., Frank, D., & Costa-Pierce, B. (2002). Analyses of organic and inorganic contaminants in Salton Sea fish. *Mar Pollut Bull, 44*(5), 403-411. doi:10.1016/s0025-326x(01)00254-5

Rieman, B. E., Isaak, D., Adams, S., Horan, D., Nagel, D., Luce, C., & Myers, D. (2007). Anticipated climate warming effects on bull trout habitats and populations across the interior Columbia River basin. *Transactions of the American Fisheries Society, 136*(6), 1552-1565.

Rieman, B. E., & McIntyre, J. D. (1993). *Demographic and habitat requirements for conservation of bull trout*. (INT-302). Retrieved from https://www.fs.usda.gov/treesearch/pubs/29778

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

Ritter, W. F., Johnson, H. P., Lovely, W. G., & Molnau, M. (1974). Atrazine, propachlor, and diazinon residues on small agricultural watersheds. Runoff losses, persistence, and movement. *Environmental Science & Technology, 8*(1), 38-42.

Roberts Jr, M. H., Diaz, R. J., Bender, M. E., & Huggett, R. J. (1975). Acute toxicity of chlorine to selected estuarine species. *Journal of the Fisheries Board of Canada, 32*(12), 2525-2528.

Roberts, M. H., Diaz, R. J., Bender, M. E., & Huggett, R. J. (1975). Acute Toxicity of Chlorine to Selected Estuarine Species. *Journal of the Fisheries Research Board of Canada, 32*(12), 2525-2528. doi:10.1139/f75-292

Rohr, J. R., Elskus, A. A., Shepherd, B. S., Crowley, P. H., McCarthy, T. M., Niedzwiecki, J. H., . . . Palmer, B. D. (2003). Lethal and sublethal effects of atrazine, carbaryl, endosulfan, and octylphenol on the streamside salamander (Ambystoma barbouri). *Environmental Toxicology and Chemistry: An International Journal, 22*(10), 2385-2392.

Rohr, J. R., Raffel, T. R., Sessions, S. K., & Hudson, P. J. (2008). Understanding the net effects of pesticides on amphibian trematode infections. *Ecological Applications, 18*(7), 1743-1753.

Roman, M. R., Stephen, B. B., Houde, E. D., & Pierson, J. J. (2019). Interactive Effects of Hypoxia and Temperature on Coastal Pelagic Zooplankton and Fish. *Frontiers in Marine Science*. Retrieved from https://doi.org/10.3389/fmars.2019.00139

Rosenthal, R. J., Haldorson, L. J., Field, L. J., O'Connell, V. M., LaRiviere, M. G., Underwood, J., & Murphy, M. C. (1982). *Inshore and shallow offshore bottomfish resources in the southeastern Gulf of Alaska 1981-82*. Retrieved from Sitka, AK:

Ruggerone, G. T. (2000). Differential survival of juvenile Sockeye and Coho Salmon exposed to low dissolved oxygen during winter. *Journal of Fish Biology*(56), 1013-1016. doi:doi:10.1006/jfbi.1999.1198

Sakuma, K. M., & Ralston, S. (1995). Distributional patterns of late larval groundfish off central California in relation to hydrographic features during 1992 and 1993. *CalCOFI Rep, 36*, 179-192.

Sanders, H. O., Finley, M. T., & Hunn, J. B. (1983). *Acute Toxicity of Six Forest Insecticides to Three Aquatic Invertebrates and Four Fishes*. (Tech. Pap. No. 110). Washington, D.C.: U.S. Fish and Wildlife Service

Sands, N. J., Rawson, K., Currens, K. P., Graeber, W. H., Ruckelshaus, M. H., Fuerstenberg, R. R., & Scott, J. B. (2009). *Determination of independent populations and viability criteria for the Hood Canal summer chum salmon evolutionarily significant unit. NOAA Technical Memorandum NMFS-NWFSC-101*.

Sappington, L. C., Mayer, F. L., Dwyer, F. J., Buckler, D. R., Jones, J. R., & Ellersieck, M. R. (2001). Contaminant sensitivity of threatened and endangered fishes compared to standard surrogate

species. *Environ Toxicol Chem, 20*(12), 2869-2876. doi:10.1897/1551-5028(2001)020<2869:csotae>2.0.co;2

Saravanan, M., Nam, S. E., Eom, H. J., Lee, D. H., & Rhee, J. S. (2019). Long-term exposure to waterborne nonylphenol alters reproductive physiological parameters in economically important marine fish. *Comp. Biochem. Physiol, 216C*, 10-18.

Saunders, C. E. (2012). *Effects of dissolved organic matter and salinity on the toxicity of individual and metal mixtures of copper with zinc and silver to the saltwater rotifer, Brachionus plicatilis*: Stephen F. Austin State University.

Sauter, S., et. al. (1976). *Effects of exposure to heavy metals on selected freshwater fish. Toxicity of copper, cadmium, chromium, and lead to eggs and fry of seven fish species.* Retrieved from

Schatzow, S. (1980). *Ambient water quality criteria for chromium*. (EPA 440/5-80-035; TD370.A4321 1980; PB81-117467). Washington, DC Springfield, Virginia: United States Environmental Protection Agency, Office of Water Regulations and Standards, Criteria and Standards Division, Available to the public through the National Technical Information Service Retrieved from https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=2000M2E8.PDF

http://catalog.hathitrust.org/api/volumes/oclc/6998630.html

Scheuhammer, A. M. (1987). The chronic toxicity of aluminium, cadmium, mercury, and lead in birds: a review. *Environ Pollut, 46*(4), 263-295. doi:10.1016/0269-7491(87)90173-4

Schimmel, S. (1981). [Memorandum to John H. Gentile].

Schimmel, S. C., Garnas, R. L., Patrick, J. M., Jr., & Moore, J. C. (1983). Acute toxicity, bioconcentration, and persistence of AC 222,705, benthiocarb, chlorpyrifos, fenvalerate, methyl parathion, and permethrin in the estuarine environment. *J Agric Food Chem, 31*(1), 104-113. doi:10.1021/jf00115a027

Schoenfuss, H. L., Bartell, S. E., Bistodeau, T. B., Cediel, R. A., Grove, K. J., Zintek, L., . . . Barber, L. B. (2008). Impairment of the reproductive potential of male fathead minnows by environmentally relevant exposures to 4-nonylphenolf. *Aquat Toxicol, 86*(1), 91-98. doi:10.1016/j.aquatox.2007.10.004

Scott, W. B., & Crossman, E. J. (1973). *Freshwater fishes of Canada. Bulletin 184*. Retrieved from Ottawa, ON, Canada:

Senthil Kumaran, S., Kavitha, C., Ramesh, M., & Grummt, T. (2011). Toxicity studies of nonylphenol and octylphenol: hormonal, hematological and biochemical effects in Clarias gariepinus. *J Appl Toxicol, 31*(8), 752-761. doi:10.1002/jat.1629

Sexauer, H. M., & James, P. W. (1997). *Microhabitat use by juvenile trout in four streams located in the eastern Cascades, Washington.* Retrieved from Calgary, AB:

Shaffer, J. A., Doty, D. C., Buckley, R. M., & West, J. E. (1995). Crustacean community composition and trophic use of the drift vegetation habitat by juvenile splitnose rockfish *Sebastes diploproa*. *Mar Ecol Prog Ser, 123*, 13-21.

Shannon, M. A., Bohn, P. W., Elimelech, M., Georgiadis, J. G., Marinas, B. J., & Mayes, A. M. (2008). Science and technology for water purification in the coming decades. *Nature, 452*(7185), 301-310. doi:10.1038/nature06599

Shaw, J. R., Gabor, K., Hand, E., Lankowski, A., Durant, L., Thibodeau, R., . . . Karlson, K. H. (2007). Role of glucocorticoid receptor in acclimation of killifish (Fundulus heteroclitus) to seawater and effects of arsenic. *American Journal of Physiology-Regulatory, Integrative and Comparative Physiology, 292*(2), R1052-R1060.

Shaw, T. (1979). A fast indicator for detecting gross pollution by hazardous chemicals. *New Zealand Journal of Marine and Freshwater Research, 13*(3), 393-394.

Shenker, J. M., & Cherr, G. N. (1990). Toxicity of zinc and bleached kraft mill effluent to larval english sole (Parophrys vetulus) and topsmelt (Atherinops affinis). *Archives of Environmental Contamination and Toxicology, 19*(5), 680-685.

Shephard, B., Zodrow, J., Keenan, D., & Palmer, J. (2008). *Biological Evaluation of Oregon's Water Quality Criteria for Toxics*. Seattle, WA: USEPA Region 10

Shephard, B. K. (1998). *Quantification of Ecological Risks to Aquatic Biota from Bioaccumulated Chemicals.* Paper presented at the National Sediment Bioaccumulation Conference: Proceedings, Washington, D.C.

Sherwood, M. J. (1975). Toxicity of chromium to fish. *Coastal Wder Res. Proj., Annual Rept, 61*.

Shiau, S. Y., & Shy, S. M. (1998). Dietary chromic oxide inclusion level required maximize glucose utilization in hybrid tilapia, *Oreochromis niloticus HO. aureus*. *Aquaculture, 161*, 357-364.

Short, F. T., & Neckles, H. A. (1999). Review. The effects of global climate change on seagrasses. *Aquat Bot, 63*, 169-196.

Siang, H. Y., Tahir, N. M., Malek, A., & Isa, M. A. M. (2017). Breakdown of hydrogen sulfide in seawater under different ratio of dissolved oxygen/hydrogen sulfide. *Malaysian Journal of Analytical Sciences, 21*(2), 1016-1027. doi:DOI:10.17576/mjas-2017-2105-03

Silva, J., Troncoso, L., Bay-Schmith, E., & Larrain, A. (2001). Utilization of Odontesthes regia (Atherinidae) from the south eastern pacific as a test organism for bioassays: Study of its sensitivity to six chemicals. *Bulletin of environmental contamination and toxicology, 66*(5), 570-575.

Simpson, S. D., Munday, P. L., Wittenrich, M. L., Manassa, R., Dixson, D. L., Gagliano, M., & Yan, H. Y. (2011). Ocean acidification erodes crucial auditory behaviour in a marine fish. Retrieved from https://doi.org/10.1098/rsbl.2011.0293

Singh, O., & Agarwal, R. (1981). Toxicity of certain pesticides to two economic species of snails in northern India. *Journal of economic Entomology, 74*(5), 568-571.

Sinley, J. R., Goettl, J. P., & Davies, P. H. (1974). The effects of zinc on rainbow trout (Salmo gairdneri) in hard and soft water. *Bulletin of environmental contamination and toxicology, 12*(2), 193-201.

Smith, C. J. (2005). *Salmon and Steelhead Habitat Limiting Factors Water Resource Inventory Areas 3 and 4, The Skagit and Samish Basisns*. Lacey, Washington: Washington State Conservation Commission Retrieved from http://www.pugetsoundnearshore.org/supporting_documents/WRIA_3and4_SkagitLFA.pdf

Smith, E., Sykora, J., & Shapiro, M. (1973). Effect of lime neutralized iron hydroxide suspensions on survival, growth, and reproduction of the fathead minnow (Pimephales promelas). *Journal of the Fisheries Board of Canada, 30*(8), 1147-1153.

Smith, E. J., & Sykora, J. L. (1976). Early Developmental Effects of Lime-Neutralized Iron Hydroxide Suspensions on Brook Trout and Coho Salmon. *Transactions of the American Fisheries Society, 105*(2), 308-312. doi:10.1577/1548-8659(1976)105<308:Edeoli>2.0.Co;2

Smith, L., & Oseid, D. (1972). Effects of hydrogen sulfide on fish eggs and fry. *Water Research, 6*(6), 711-720.

Smith, L., & Oseid, D. (1974). *Effect of hydrogen sulfide on development and survival of eight freshwater fish species.* Paper presented at the The Early Life History of Fish. Proceedings of an International Symposium, Oban, Scotland.

Smith, L., & Oseid, D. (1975). Chronic effects of low levels of hydrogen sulfide on freshwater fish. *Prog. Water Technol., 7*, 599-605.

Smith, S., Furay, V. J., Layiwola, P. J., & Menezes - Filho, J. A. (1994). Evaluation of the toxicity and quantitative structure - activity Relationships (QSAR) of chlorophenols to the copepodid stage of a marine copepod (Tisbe battagliai) and two species of benthic flatfish, the flounder (Platichthys flesus) and sole (Solea solea). *Chemosphere, 28*(4), 825-836. doi:10.1016/0045-6535(94)90234-8

Smithsonian. (2018). On the water. Commercial fishers: Columbia River salmon. Retrieved from http://americanhistory.si.edu/onthewater/exhibition/3_6.html

Smokorowski, K., & Kelso, J. (2002). Trends in fish community structure, biomass, and production in three Algoma, Ontario, lakes. *Water, Air and Soil Pollution: Focus, 2*(1), 129-150.

Snell, T. W., & Moffat, B. D. (1992). A 2-d life cycle test with the rotifer Brachionus calyciflorus. *Environmental Toxicology and Chemistry: An International Journal, 11*(9), 1249-1257.

Society, A. F. (1979). *A Review of the EPA Red Book: Quality Criteria for Water* (R. V. Thurston, R. C. Russo, C. M. F. Jr., T. A. Edsall, & Y. M. B. Jr. Eds.). Thurston, R.V., R.C. Russo, C.M. Fetterolf Jr., T.A. Edsall and Y.M. Barber Jr.: Water Quality Section.

Spehar, R. (1989). Aquatic toxicity test information on acrolein with fathead minnows (Pimephales promelas) and flagfish (Jordanella floridae). *US EPA, Duluth, MN, 5*.

Spehar, R. L., Brooke, L. T., Markee, T. P., & Kahl, M. D. (2010). Comparative toxicity and bioconcentration of nonylphenol in freshwater organisms. *Environ Toxicol Chem, 29*(9), 2104-2111. doi:10.1002/etc.262

Spry, D. J., Hodson, P. V., & Wood, C. M. (1988). Relative Contributions of Dietary and Waterborne Zinc in the Rainbow Trout, *Salmo gairdneri Canadian Journal of Fish. Aquatic Science, 45*(1). doi:https://doi.org/10.1139/f88-005

SRSC, & WDFW. (2005). *Skagit Chinook Recovery Plan*. Retrieved from http://skagitcoop.org/wp-content/uploads/Skagit-Chinook-Plan-13.pdf

Stanley Jr, T. R., Smith, G. J., Hoffman, D. J., Heinz, G. H., & Rosscoe, R. (1996). Effects of boron and selenium on mallard reproduction and duckling growth and survival. *Environmental Toxicology and Chemistry: An International Journal, 15*(7), 1124-1132.

Stengel, D., Zindler, F., & Braunbeck, T. (2017). An optimized method to assess ototoxic effects in the lateral line of zebrafish (Danio rerio) embryos. *Comp Biochem Physiol C Toxicol Pharmacol, 193*, 18-29. doi:10.1016/j.cbpc.2016.11.001

Stephan, C. E., Mount, D. I., Hansen, D. J., Gentile, J., Chapman, G. A., & Brungs, W. A. (1985). *Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses*: US Environmental Protection Agency Washington, DC.

Stephan, C. E., Mount, D. I., Hansen, D. J., Gentile, J. R., Chapman, G. A., & Brungs, W. A. (1985). *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection Of Aquatic Organisms and Their Uses*. Washington, D.C.: U.S. Environmental Protection Agency

Stephenson, R. (1983). Effects of water hardness, water temperature, and size of the test organism on the susceptibility of the freshwater shrimp, *Gammarus pulex* (L.) to toxicants. *Bull. Environ. Contam. Toxicol., 31*, 459-466.

Sun, D., Chen, Q., He, N., Diao, P. P., Jia, L. X., & Duan, S. S. (2017). Effect of environmentally-relevant concentrations of nonylphenol on sexual differentiation in zebrafish: a multi-generational study. *Sci Rep, 7*, 42907. doi:10.1038/srep42907

Surprenant, D. (1985). *Acute toxicity of sevin technical to sheepshead minnow (Cyprinodon variegatus)*. Toxicity Test Report. Springborn Bionomics, Inc. . Wareham, MA.

Surprenant, D. (1985). *The chronic toxicity of carbaryl technical to Daphnia magna under flow through conditions*. Toxicity Test Report. Springborn Bionomics, Inc. . Wareham, MA.

Suter II, G. W., Efroymson, R. A., Sample, B. E., & Jones, D. S. (2000). *Ecological risk assessment for contaminated sites*: CRC Press.

Svecevičius, G. (2010). Acute toxicity of nickel to five species of freshwater fish. *Polish Journal of Environmental Studies, 19*(2), 453-456. Retrieved from https://www.cabdirect.org/cabdirect/abstract/20103158751

Swedmark, M., Braaten, B., Emanuelsson, E., & Granmo, A. (1971). Biological effects of surface active agents on marine animals. *Marine Biology, 9*, 183-201.

Swinomish Indian Tribal Community. (2016). *2016 State of Our Watersheds Report Skagit River Basin*. Retrieved from Olympia, Washington:

Sykora, J., Smith, E., Synak, M., & Shapiro, M. (1975). Some observations on spawning of brook trout (Salvelinus Fontinalis, mitchill) in lime neutralized iron hydroxide suspensions. *Water Research, 9*(4), 451-458.

Sykora, J. L., Smith, E. J., & Synak, M. (1972). Effect of lime neutralized iron hydroxide suspensions on juvenile brook trout (salvelinus fontinalis, mitchill). *Water Research, 6*(8), 935-950. doi:10.1016/0043-1354(72)90045-0

Tagal, M., Massee, K. C., Ashton, N., Campbell, R. A., Pleasha, P., & Rust, M. B. (2002). *Larval development of yelloweye rockfish, Sebastes ruberrimus*. Retrieved from

Tagatz, M. E., & Stanley, R. S. (1987). *Sensitivity Comparisons of Estuarine Benthic Animals Exposed to Toxicants in Single Species Acute Tests and Community Tests*. Gulf Breeze, FL

Taylor, D., Maddock, B. G., & Mance, G. (1985). The acute toxicity of nine 'grey list' metals (arsenic, boron, chromium, copper, lead, nickel, tin, vanadium and zinc) to two marine fish species: Dab (Limanda limanda) and grey mullet (Chelon labrosus). *Aquatic Toxicology, 7*(3), 135-144. doi:10.1016/s0166-445x(85)80001-1

TCW. (2008). *Economic analysis of the non-treaty commercial and recreational fisheries in Washington State. Prepared for the Washington Department of Fish and Wildlife*. Retrieved from Sacramento, CA:

Tejada, A., Bajet, C., Magbauna, M., Gambalan, N., Araez, L., & Magallona, E. (1994). Toxicity of pesticides to target and non-target fauna of the lowland rice ecosystem. *Environmental Toxicology in South East Asia, VU Univ. Press, Amsterdam, Netherlands*, 89-103.

TenEyck, M. C., & Markee, T. P. (2007). Toxicity of nonylphenol, nonylphenol monoethoxylate, and nonylphenol diethoxylate and mixtures of these compounds to Pimephales promelas (fathead minnow) and Ceriodaphnia dubia. *Archives of Environmental Contamination and Toxicology, 53*(4), 599-606.

Thatcher, T. (1978). The relative sensitivity of Pacific Northwest fishes and invertebrates to chlorinated sea water. *2*.

Thirugnanam, M., & Forgash, A. J. (1977). Environmental impact of mosquito pesticides: toxicity and anticholinesterase activity of chlorpyrifos to fish in a salt marsh habitat. *Arch Environ Contam Toxicol, 5*(4), 415-425. doi:10.1007/BF02220921

Thom, R. M., & Hallum, L. (1990). *Long-term changes in the areal extent of tidal marshes, eelgras meadows and kelp forests of Puget Sound*. Retrieved from Seattle, WA:

Thomas, G. (1992). Satus of Bull Trout in Montana. In. Helena, MT.

Thompson, B., Bay, S., Greenstein, D., & Laughlin, J. (1991). Sublethal effects of hydrogen sulfide in sediments on the urchin Lytechinus pictus. *Marine Environmental Research, 31*(4), 309-321.

Thursby, G., & Berry, W. (1988). [Acute Toxicity of Diazinon to Saltwater Animals (Letter to J.Scott and D.J.Hansen)].

Thursby, G. B., & Champlin, D. (1991a, June 13, 1991). [Flow-through acute and chronic toxicity of carbaryl to Mysidopsis bahia].

Thursby, G. B., & Champlin, D. (1991b, June 13, 1991). [Static Acute Toxicity Data for Carbaryl].

Tolimieri, N., & Levin, P. S. (2005). The roles of fishing and climate in the population dynamics of bocaccio rockfish. *Ecol Appl, 15*(2), 458-468.

Tomasetti, S. J., & Gobler, C. J. (2020). Dissolved oxygen and pH criteria leave fisheries at risk. *Science, 368*. doi:10.1126/science.aba4896

Tomilin, A. G. (1967). *Mammals of the USSR and adjacent countries. Volume 9, Cetacea. Translated (1967) by Israel Program for Scientific Translations, Jerusalem. TT 65-50086*. Retrieved from Springfield, VA:

Toumi, H., Burga-Perez, K. F., & Ferard, J.-F. (2016). Acute and chronic ecotoxicity of carbaryl with a battery of aquatic bioassays. *Journal of Environmental Science and Health, Part B, 51*(1), 57-62.

Union_Carbide_Corporation. (1997). *Chronic Toxicity of Acrolein to Ceriodaphnia dubia*. (EPA/OTS Doc.#86970000815). Washington, D.C.

Updegraff, K. F., & Sykora, J. L. (1976). Avoidance of lime-neutralized iron hydroxide solutions by coho salmon in the laboratory. *Environmental Science & Technology, 10*(1), 51-54.

USEPA. (1976a). *Effect of Hydrogen Sulfide on Fish and Invertebrates. Part I - Acute and Chronic Toxicity Studies*. (EPA-600/3-76-062a). Duluth, MN: U.S. Environmental Protection Agency

USEPA. (1976b). *Effect of hydrogen sulfide on fish and invertebrates. Part I, Acute and chronic toxicity studies*. (EPA-600/3-76-062a; EPA-R-800992). Duluth, Minn. Springfield, Va.: U.S. Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratory

available through the National Technical Information Service

USEPA. (1976c). Iron. In *Quality Criteria for Water* (Vol. EPA-440/9-76-023, pp. p. 152 - 158). Washington, D.C.: Office of Water and Hazardous Materials, U.S. Environmental Protection Agency.

USEPA. (1976d). Phosphorus. In *Quality Criteria for Water* (Vol. EPA-440/9-76-023, pp. 352 - 360). Washington, D.C.: Office of Water and Hazardous Materials, U.S. Environmental Protection Agency.

USEPA. (1976e). *Quality criteria for water*. (EPA 440-R-76-001; EPA 440-9-76-023; TD223.U52 1976; PB-263 943). Washington: U.S. Environmental Protection Agency Retrieved from https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=2000IYMP.PDF; https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1008F64.PDF

USEPA. (1976f). Sulfide - Hydrogen Sulfide. In *Quality Criteria for Water* (Vol. EPA-440/9-76-023, pp. 410 - 416). Washington, D.C.: Office of Water and Hazardous Materials, U.S. Environmental Protection Agency.

USEPA. (1980). *Ambient water quality criteria for zinc*. (EPA 440/5-80-079; PB81-117897). Washington, DC Springfield, Virginia: United States Environmental Protection Agency, Office of Water Regulations and Standards, Criteria and Standards Division, Available to the public through the National Technical Information Service Retrieved from https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=2000LNKE.PDF

USEPA. (1984). *Ambient Water Quality Criteria for Copper - 1984*. (EPA 440/5-84-031). Washington, D.C.: U.S. Environmental Protection Agency

USEPA. (1985a). *Ambient aquatic life water quality criteria for arsenic*. (EPA/440/5-84-033; PB85-227445). Washington, DC Springfield, VA: Environmental Protection Agency, Regulations and Standards, Criteria and Standards Division, Available through the National Technical Information Service Retrieved from https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=00001M7S.PDF

USEPA. (1985b). *Ambient water quality criteria for chlorine - 1984*. (EPA 440/5-84-030; PB85-227429). Washington, D.C. ;: U.S. Environmental Protection Agency Retrieved from https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=20011OLZ.PDF

USEPA. (1985c). *Ambient Water Quality Criteria for Cyanide – 1984*. (EPA-440/5-84-028). Springfield, Virginia: National Technical Information Service

USEPA. (1985d). *Ambient Water Quality Criteria for Lead*. (EPA 440/5-84-027). Washington, DC: USEPA

USEPA. (1986a). *Ambient aquatic life water quality criteria for dissolved oxygen (Freshwater)*. (EPA/440/5-86-003; PB86208253). Washington, DC: U.S. Environmental Protection Agency, Office of Water Regulations and Standards, Criteria and Standards Division Retrieved from https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=00001MSS.PDF

USEPA. (1986b). *Ambient aquatic life water quality criteria for pentachlorophenol*. (EPA/440-5-86-009; PB87-105391). Washington, D.C.: U.S. Environmental Protection Agency, Office of Water Regulations and Standards, Criteria and Standards Division Retrieved from https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=2000M5GU.PDF

USEPA. (1986c). *Ambient Water Quality Criteria for Chlorpyrifos - 1986*. (EPA 440/5-86-005). Washington, D.C.: Criteria and Standards Division, U.S. Environmental Protection Agency

USEPA. (1986d). *Quality criteria for water, 1986*. (EPA/440/5-86-001). Washington, DC: U.S. Environmental Protection Agency, Office of Water Regulations and Standards Retrieved from https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=00001MGA.PDF; http://www.epa.gov/waterscience/criteria/goldbook.pdf

USEPA. (1986e). *Quality criteria for water, 1986*. Washington, DC: U. S. Environmental Protection Agency. Office of Water Regulations and Standards.

USEPA. (1986f). Sulfide - Hydrogen Sulfide. In *Quality Criteria for Water 1986* (Vol. EPA 440/5-86-001, pp. 273 - 276). Washington, D.C.: Office of Water, U.S. Environmental Protection Agency.

USEPA. (1986g). *Superfund Public Health Evaluation Manual*. (EPA/540/1-86/060). Washington, D.C.: U.S. Environmental Protection Agency

USEPA. (1987). *Ambient Aquatic Life Water Quality Criteria for Zinc* (EPA-440/5-87-003). Washington, DC: USEPA

USEPA. (1989). *Ambient aquatic life water quality criteria for ammonia (saltwater)*. (EPA/440-5-88-004; PB89-195242). Washington, DC: U.S. Environmental Protection Agency, Office of Water Regulations and Standards, Criteria and Standards Division Retrieved from https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=00001N64.PDF

USEPA. (1991). *Technical support document for water quality-based toxics control*. Environmental Protection Agency, Washington, DC. Office of the Assistant Administrator for Water. Office of Water Enforcement and Permits : Office of Water Regulations and Standards, U.S. Environmental Protection Agency Retrieved from http://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=100002CU.PDF

USEPA. (1993). *Office of Water Policy and Technical Guidance on Interpretation and Implementation of Aquatic Life Metals Criteria*. Washington D.C.: U.S. Environmental Protection Agency Retrieved from https://www.epa.gov/sites/default/files/2019-03/documents/metals-criteria-interpret-aqlife-memo.pdf

USEPA. (1995). *Quality criteria for water 1995*. (EPA 823-D-95-003). Washington, DC: U.S. Environmental Protection Agency, Office of Water, Health and Ecological Criteria Division Retrieved from https://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=910076BC.txt

USEPA. (1996a). *1995 Updates: Water Quality Criteria Documents for the Protection of Aquatic Life in Ambient Water*. (EPA-820-B-96-001). Washington, D.C.: U.S. Environmental Protection Agency

USEPA. (1996b). *1995 Updates: Water quality criteria documents for the protection of Aquatic Life in Ambient Water*. (EPA 820-B-96-001; PB98-153067). Washington, D.C. Springfield, Va.: U.S. Environmental Protection Agency, Office of Water, U.S. Dept. of Commerce, National Technical Information Service [distributor] Retrieved from https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=20002924.PDF

USEPA. (1998). *Guidelines for Ecological Risk Assessment*. (EPA/630/R-95/002F). Washington, D.C.

USEPA. (1999a). *1999 Update of ambient water quality criteria for ammonia*. (EPA 822-R-99-014; PB2005-109112). Washington, D.C.: U.S. Environmental Protection Agency, Office of Water Retrieved from https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=20003O3L.PDF

USEPA. (1999b). *National Recommended Water Quality Criteria - Correction*. (EPA 822-Z-99-001).

USEPA. (1999c). *Proposed revisions to the Water Quality Planning and Management Regulation : proposed rule*. Washington, DC: Federal Register.

USEPA. (2001). *Ambient water quality criteria recommendations : information supporting the development of state and tribal nutrient criteria for lakes and reservoirs in nutrient ecoregion III : xeric west including all or parts of the states of : Washington, Oregon, California, Nevada, Idaho, Wyoming, Montana, Utah, Colorado, New Mexico, Arizona, and Texas, and the authorized Tribes within the ecoregion*. Environmental Protection Agency, Washington, DC. Office of Water.

U.S. Environmental Protection Agency, Office of Water : Office of Science and Technology, Health and Ecological Criteria Division Retrieved from https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=20003FNV.PDF http://www.epa.gov/waterscience/criteria/nutrient/ecoregions/lakes/lakes_3.pdf

USEPA. (2003). *EPA Region 10 guidance for Pacific Northwest state and tribal temperature water quality standards*. U.S. Environmental Protection Agency, Region 10 Retrieved from http://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=P1004IUI.txt http://yosemite.epa.gov/R10/WATER.NSF/1507773cf7ca99a7882569ed007349b5/ce95a3704aeb5715882568c400784499?OpenDocument

USEPA. (2005a). *Ambient Aquatic Life Water Quality Criteria, Nonylphenol: Final*. (EPA-822-R-05-005). Washington, D.C.: U.S. Environmental Protection Agency

USEPA. (2005b). *Aquatic Life Ambient Water Quality Criteria for Diazinon*. (EPA-822-R-05-006). Washington, D.C.: U.S Environmental Protection Agency

USEPA. (2005c). *Proposed Revisions to Aquatic Life Guidelines: Tissue-Based Criteria for "Bioaccumulative" Chemicals*. Washington, D.C.: U.S. Environmental Protection Agency

USEPA. (2006a). *Biological Evaluation of Aquatic Life Criteria – Cyanide*. Washington, D.C. : U.S. Environmental Protection Agency

USEPA. (2006b). *Draft Framework for Conducting Biological Evaluations of Aquatic Life Criteria: Methods Manual*. U.S. Environmental Protection Agency

USEPA. (2006c). *EPA Science Advisory Board (SAB) Consultation on a Proposed Framework for Revising the Guidelines for Deriving Water Quality Criteria for Protection of Aquatic Life*. U.S. Environmental Protection Agency

USEPA. (2007a). *Aquatic life ambient freshwater quality criteria copper [electronic resource]*. (EPA-822-R-07-001; PB2007-107219). Washington, D.C.: Office of Water, U.S. Environmental Protection Agency Retrieved from http://www.epa.gov/waterscience/criteria/copper/2007/index.htm; http://www.epa.gov/waterscience/criteria/copper/2007/criteria200702.pdf; https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P1000PXC.PDF

USEPA. (2007b). *Biological Evaluation of the Revised Washington Water Quality Standards*. Region 10, Seattle, WA: U.S Environmental Protection Agency

USEPA. (2008a). *Biological Evaluation of Oregon's Water Quality Criteria for Toxics*. Seattle, WA: U.S. Environmental Protection Agency, Region 10

USEPA. (2008b). *Endangered Aquatic Vertebrates: Comparative and Probabilistic-Based Toxicology*. (EPA/600/R-08/045). Washington, DC: USEPA

USEPA. (2009a). *Ambient Aquatic Life Water Quality Criteria for Acrolein*. Washington, D.C.: U.S. Environmental Protection Agency

USEPA. (2009b). *Technical Review of Hydrogen Sulfide: Chemistry, Environmental Fate and Ecological Toxicity*. Washington, D.C.: U.S. Environmental Protection Agency

USEPA. (2009c). *Technical Review of Hydrogen Sulfide: Chemistry, Environmental Fate and Ecological Toxicity*. Retrieved from

USEPA. (2010). *National Pollutant Discharge Elimination System (NPDES) permit writers' manual*. Environmental Protection Agency, Washington, DC. Office of Wastewater Management. U.S. Environmental Protection Agency, Office of Wastewater Management Retrieved from http://www.epa.gov/npdes/pubs/pwm_2010.pdf http://nepis.epa.gov/Exe/ZyPURL.cgi?Dockey=P1009L35.txt

USEPA. (2012). *Aquatic Life Ambient Water Quality Criteria for Carbaryl*. (EPA-820-R-12-007). Washington, D.C.: Office of Science and Technology Health and Ecological Criteria Division, U.S. Environmental Protection Agency

USEPA. (2013a). *Aquatic Life Ambient Water Quality Criteria for Ammonia - Freshwater*. (EPA 822-R-18-002). Washington D.C. Retrieved from https://www.epa.gov/sites/default/files/2015-08/documents/aquatic-life-ambient-water-quality-criteria-for-ammonia-freshwater-2013.pdf

USEPA. (2013b). *Biological Evaluation of the Coeur d'Alene Tribe Water Quality Standards*. Seattle, WA

USEPA. (2014). *Water Quality Standards Handbook: Chapter 5: General Policies (updated 2014)*. Washington, DC: U.S. Environmental Protection Agency (EPA) Retrieved from https://www.epa.gov/sites/production/files/2014-09/documents/handbook-chapter5.pdf

USEPA. (2016a). *Aquatic Life Ambient Water Quality Criteria Cadmium*. (EPA 820-R-16-002). Washington D.C. Retrieved from https://www.epa.gov/sites/default/files/2016-03/documents/cadmium-final-report-2016.pdf

USEPA. (2016b). *Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater*. (EPA 822-R-16-006). Washington, D.C., USA

USEPA. (2016c). *Draft Technical Support Document Recommended Estimates for Missing Water Quality Parameters for Application in EPA's Biotic Ligand Model*. (820R15106). Retrieved from https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100O2MX.PDF

USEPA. (2017a). *Biological Evaluation of Chlorpyrifos Risks to Threatened and Endangered Species*. Washington, D.C. Retrieved from https://www.epa.gov/endangered-species/biological-evaluation-chapters-chlorpyrifos-esa-27-17assessment

USEPA. (2017b). *Working Draft: Biological Evaluation Cookbook For Aquatic Life Water Quality Standards*. Washington, D.C.

USEPA. (2018a). *DRAFT Aquatic Life and Aquatic-Dependent Wildlife Selenium Water Quality Criterion for Freshwaters of California*. Washington, D.C., USA

USEPA. (2018b). *Final Aquatic Life Ambient Water Quality Criteria for Aluminum*. (EPA-822-R-18-001). Washington D.C.: U.S. Environmental Protection Agency

USEPA. (2018c). *Northwest area contingency plan 2018*. Retrieved from

USEPA. (2019). National Recommended Aquatic Life Criteria table - 2019. Retrieved from https://www.epa.gov/wqc/national-recommended-water-quality-criteria-aquatic-life-criteria-table

USEPA. (2021). *Revision To: Aquatic Life Ambient Water Quality Criterion for Selenium - Freshwater 2016*. (EPA 822-R-21-006). Washington, D.C. Retrieved from https://www.epa.gov/system/files/documents/2021-08/selenium-freshwater2016-2021-revision.pdf

USEPA. (2022). National Recommended Water Quality Criteria - Aquatic Life Criteria. Retrieved from https://www.epa.gov/wqc/national-recommended-water-quality-criteria-aquatic-life-criteria-table

USEPA. (MRID41422401). *MRID41422401, Test ID 41422401*. Washington, D.C.: USEPA

USEPA. (MRID41718401). *MRID41718401 (TestID P)*. Washington, D.C.

USEPA. (MRID48752901). *MRID48752901 (TestID X)*. Washington, D.C.

USFWS. (1997). *Recovery Plan for the threatened marbled murrelet (Brachyramphus marmoratus) in Washington, Oregon and California*. Portland, OR: U.S. Fish and Wildlife Service Retrieved from https://www.fws.gov/wafwo/species/Fact%20sheets/USFWS%20Recovery%20Plan%201997.pdf

USFWS. (1998). Bull Trout Facts. In U. F. a. W. Service (Ed.). Portland, OR: US Fish and Wildlife Service Public Affairs Office.

USFWS. (2004). *Marbled murrelet 5-year review process: overview*. Retrieved from

USFWS. (2009). *Marbled Murrelet (Brachyramphus marmoratus) 5-Year Review*. Lacey, WA Retrieved from https://www.fws.gov/wafwo/species/Fact%20sheets/5%20Year%20Review%202009.pdf

USFWS. (2015a). *Biological Opinion for the Idaho Water Quality Standards for Numeric Water Quality Criteria for Toxic Pollutants*. Retrieved from Boise, Idaho:

USFWS. (2015b). *Final PROJECTS biological opinion*. Retrieved from Portland, OR:

USFWS. (2015c). *Recovery Plan for the Coterminous United States Population of Bull Trout*. Portland, Oregon Retrieved from https://www.fws.gov/pacific/bulltrout/pdf/Final_Bull_Trout_Recovery_Plan_092915.pdf

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, & NMFS. (1998). *Endangered Species Consultation Handbook: Procedures for Conducting Consultation and Conference Activities Under Section of the Endangered Species Act*. Retrieved from https://www.fisheries.noaa.gov/webdam/download/64572719

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*. Washington D.C.

USGS. (1998). *Ground-Water Age, Flow, and Quality Near a Landfill, and Changes in Ground-Water Conditions From 1976 to 1996 in the Swinomish Indian, Reservation, Northwestern Washington*. Retrieved from Tacoma, Washington: https://pubs.usgs.gov/wri/1998/4014/report.pdf

USGS. (2007). *Nearshore circulation and water column properties in the Skagit River Delta, northern Puget Sound, Washington - Juvenile Chinook salmon habitat availability in the Swinomish Channel*. Retrieved from Reston, Virginia: http:pubs.usgs.gov/sir/2007/5120/

Uthus, E. O. (1992). Evidence for arsenic essentiality. *Environ. Geochem. Health, 14*, 55-58.

Van der Geest, H. G., Greve, G. D., de Haas, E. M., Scheper, B. B., Kraak, M. H., Stuijfzand, S. C., . . . Admiraal, W. (1999). Survival and behavioral responses of larvae of the caddisfly Hydropsyche angustipennis to copper and diazinon. *Environmental Toxicology and Chemistry: An International Journal, 18*(9), 1965-1971.

van der Plas, F. (2019). Biodiversity and ecosystem functioning in naturally assembled communities. *Biol Rev Camb Philos Soc, 94*(4), 1220-1245. doi:10.1111/brv.12499

Van Leeuwen, C., Griffioen, P., Vergouw, W., & Maas-Diepeveen, J. (1985). Differences in susceptibility of early life stages of rainbow trout (Salmo gairdneri) to environmental pollutants. *Aquatic toxicology, 7*(1-2), 59-78.

van Wezel, A. P., & Opperhuizen, A. (1995). Narcosis due to environmental pollutants in aquatic organisms: residue-based toxicity, mechanisms, and membrane burdens. *Critical Reviews in Toxicology, 25*(3), 255-279.

van Wijngaarden, R. P., Brock, T. C., & Douglas, M. T. (2005). Effects of chlorpyrifos in freshwater model ecosystems: the influence of experimental conditions on ecotoxicological thresholds. *Pest management science, 61*(10), 923-935.

Vardy, D. W., Santore, R., Ryan, A., Giesy, J. P., & Hecker, M. (2014). Acute toxicity of copper, lead, cadmium, and zinc to early life stages of white sturgeon (Acipenser transmontanus) in

laboratory and Columbia River water. *Environmental Sci Pollut Res Int, 21*(13). doi:10.1007/s11356-014-2754-6

Varo, I., Serrano, R., Pitarch, E., Amat, F., Lopez, F. J., & Navarro, J. C. (2000). Toxicity and bioconcentration of chlorpyrifos in aquatic organisms: Artemia parthenogenetica(Crustacea), Gambusia affinis, and Aphanius iberus (Pisces). *Bull Environ Contam Toxicol, 65*(5), 623-630. doi:10.1007/s0012800169

Vaughn, C. C. (2010). Biodiversity Losses and Ecosystem Function in Freshwaters: Emerging Conclusions and Research Directions. *BioScience, 60*(1), 25-35. doi:10.1525/bio.2010.60.1.7

Venturino, A., Montagna, C. M., & de D'Angelo, A. M. P. (2007). Risk assessment of Magnacide® H herbicide at Río Colorado irrigation channels (Argentina). tier 3: Studies on native species. *Environmental Toxicology and Chemistry: An International Journal, 26*(1), 177-182.

Vidal, D., & Horne, A. (2003). Mercury toxicity in the aquatic oligochaete Sparganophilus pearsei: I. Variation in resistance among populations. *Archives of Environmental Contamination and Toxicology, 45*(2), 184-189.

Vigers, G. A., Marleave, J. B., Janssen, R. G., & Borgmann, P. (1978). *Use of larval herring in bioassays*. Retrieved from Ottawa, Canada:

Villarosa, L., McCormick, M., Carpenter, P., & Marriott, P. (1994). Determination of trace levels of diazinon and propetamphos in water and sewage by solid phase extraction. *International Journal of Environmental Analytical Chemistry, 54*(2), 93-103.

Waldrop, T., Summerfelt, S., Mazik, P., Kenney, P. B., & Good, C. (2020). The effects of swimming exercise and dissolved oxygen on growth performance, fin condition and survival of rainbow trout Oncorhynchus mykiss. *Aquaculture Research*(51), 2582-2589. Retrieved from https://doi.org/10.1111/are.14600

Wallace, J. R. (2007). *Update to the status of yelloweye rockfish (Sebastes ruberrimus) off the U.S. West Coast in 2007*. Retrieved from Seattle, WA.

Waller, B. U., Black, E. A., Burt, D., Groot, C., & Rosenthal, H. (2000). The reaction of young coho Oncorhynchus kisutch to declining oxygen levels during long-term exposure. *Journal of Applied Ichthyology, 16*, 14-19. Retrieved from https://doi.org/10.1046/j.1439-0426.2000.00168.x

Waller, U. H., Rosenthal, & Black, E. A. (1997). The respiration of young coho Oncorhynchus kisutch at gradually declining oxygen levels and during recovery from oxygen deprivation. *J. Appl. Ichthyol, 13*, 73-78.

Wan, M., Moul, D., & Watts, R. (1987). Acute toxicity to juvenile Pacific salmonids of Garlon 3A, Garlon 4, triclopyr, triclopyr ester, and their transformation products: 3, 5, 6-trichloro-2-pyridinol and 2-methoxy-3, 5, 6-trichloropyridine. *Bulletin of Environmental Contamination and Toxicology, 39*(4), 721-728.

Ward, T., & Boeri, R. (1990). Acute flow through toxicity of nonylphenol to the mysid (Mysidopsis bahia). *Glen Osmond, Australia: CMA*.

Ward, T., & Boeri, R. (1991a). Chronic toxicity of nonylphenol to the mysid (Mysidopsis bahia). *Report Prepared for Chemical Manufactures Association by Resource Analysts. Study*(8977-CMA).

Ward, T., & Boeri, R. (1991b). Early life stage toxicity of nonylphenol to the fathead minnow (Pimephales promelas). *Report prepared for Chemical Manufactures Association by Resource Analysts*.

Ward, T. J., & Boeri, R. L. (1991). *Early life stage toxicity of nonylphenol to the fathead minnow, Pimephales promelas* (Study Number 8979 CMA). Retrieved from Hampton, NH:

Washington Department of Fisheries, Washington Department of Wildlife, & Western Washington Treaty Indian Tribes. (1993). *1992 Washington State salmon and steelhead stock inventory*. Retrieved from Olympia, WA:

Washington, P. M. (1977). *Recreationally important marine fishes of Puget Sound, Washington*. Retrieved from Seattle, WA.

Washington, P. M., Gowan, R., & Ito, D. H. (1978). *A biological report on eight species of rockfish (Sebastes spp.) from Puget Sound, Washington*. Retrieved from Seattle, WA:

Watkins, W. A., & Schevill, W. E. (1979). Aerial observation of feeding behavior in four baleen whales: *Eubalaena glacialis*, *Balaenoptera borealis*, *Megaptera novaeangliae* and *Balaenoptera physalus*. *J Mammal, 60*, 155-163.

WDFW. (1997). *Washington State Salmonid Stock Inventory - Bull Trout/Dolly Varden*. Retrieved from Olympia, WA:

WDFW, & PNPTT. (2000). *Summer chum salmon conservation initiative*. Retrieved from Olympia, WA:

WDOE. (2002). *Evaluating Criteria for the Protection of Freshwater Aquatic Life in Washington's Surface Water Quality Standards: Dissolved Oxygen* (Number 00-10-071). Retrieved from

Webb, C. M., & Crain, D. A. (2006). Effects of ecologically relevant doses of malathion on developing Xenopus laevis tadpoles. *Bios, 77*(1), 1-6.

Weinrich, M. T. (1983). *Observations: the humpback whales of Steliwagen Bank*. Gloucester, MA: Whale Research Press.

Weis, J., & Weis, P. (1989). Effects of environmental pollutants on early fish development. *Rev. Aquat. Sci, 1*(1), 45-73.

Weis, L. J. (2004). *The effects of San Juan County, Washington, marine protected areas on larval rockfish production.* (Master of Science). University of Washington, Seattle, WA. Retrieved from https://alliance-primo.hosted.exlibrisgroup.com/permalink/f/kjtuig/CP71156512390001451

Wells, M. L., Smith, G. J., & Bruland, K. (2000). The distribution of colloidal and particulate bioactive metals in Narragansett Bay, RI. *Marine Chemistry, 71*(1-2), 143-163.

Whitehead, H. R. (1987). Updated status of the humpback whale, *Megaptera novaeangliae*, in Canada. *Can Field Nat, 101*(2), 284-294.

WHO. (1991). *Nickel, Environmental Health Criteria No. 108*. Geneva Retrieved from https://wedocs.unep.org/20.500.11822/29455.

WHO. (1998). *Environmental Health Criteria198 - Diazinon.* Retrieved from Geneva.:

Widder, P. D., & Bidwell, J. R. (2006). Cholinesterase activity and behavior in chlorpyrifos-exposed Rana sphenocephala tadpoles. *Environmental Toxicology and Chemistry: An International Journal, 25*(9), 2446-2454.

Williams, R., Lusseau, D., & Hammond, P. S. (2006). Estimating relative energetic costs of human disturbance to killer whales (Orcinus orca). *Biological Conservation, 133*(3), 301-311.

Williams, R., Lusseau, D., & Hammond, P. S. (2009). The role of social aggregations and protected areas in killer whale conservation: the mixed blessing of critical habitat. *Biological Conservation, 142*(4), 709-719.

Winn, H. E., & Reichley, N. (1985). Humpback whale - *Megaptera novaeangliae*. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of marine mammals* (Vol. 3: The sirenians and baleen whales, pp. 241-274). London, UK: Academic Press.

Wittmann, A. C., & Portner, H.-O. (2013). Sensitivities of extant animal taxa to ocean. *Nature Climate Change, 3*. Retrieved from https://www.researchgate.net/profile/Hans-Otto-Poertner/publication/259178444_Sensitivities_of_extant_animal_taxa_to_ocean_acidification/links/592aeb90aca27295a80b7506/Sensitivities-of-extant-animal-taxa-to-ocean-acidification.pdf

Wolfe-Simon, F., Switzer, J., Kulp, T. R., Gordon, G. W., Hoeft, S. E., Pett-Ridge, J., . . . Oremland, R. S. (2011). A bacterium that can grow by using arsenic instead of phosphorus. *Science, 332*. doi:10.1126/science.1197258

Wood, C. M. (2011). Silver. Fish Physiology: Homeostasis and Toxicology of Non-Essential Metals. *31*, 1-65. Retrieved from http://dx.doi.org/10.1016/S1546-5098(11)31001-1

Wood, C. M., Hogstrand, C., Galvez, F., & Munger, R. (1996). The physiology of waterborne silver toxicity in freshwater rainbow trout (Oncorhynchus mykiss) 2. The effects of silver thiosulfate. *Aquatic toxicology, 35*(2), 111-125.

Wyllie-Echeverria, T. (1987). Thirty-four species of California rockfishes: maturity and seasonality of reproduction. *Fish Bull, 85*(2), 229-250.

Xu, Q., & Pascoe, D. (1993). The bioconcentration of zinc by Gammarus pulex (L.) and the application of a kinetic model to determine bioconcentration factors. *Water Research, 27*(11), 1683-1688.

Yamada, Y., & Ikeda, T. (1999). Acute toxicity of lowered pH to some oceanic zooplankton. *Plankton Biology and Ecology, 46*, 62-67.

Yamanaka, K. L., & Kronlund, A. R. (1997). *Inshore rockfish stock assessment for the West Coast of Canada in 1996 and recommended yields for 1997. Canadian technical report of fisheries and aquatic sciences No. 2175*. Retrieved from Nanaimo, BC:

Yamanaka, K. L., & Lacko, L. C. (2001). *Inshore rockfish (Sebastes ruberrimus, S. maliger, S. caurinus, S. melanops, S. nigrocinctus, and S. nebulosus) stock assessment for the West Coast of Canada and recommendations for management*. Retrieved from Nanaimo, BC:

Yamanaka, K. L., Lacko, L. C., Withler, R., Grandin, C., Lochead, J. K., Martin, J. C., . . . Wallace, S. S. (2006). *A review of yelloweye rockfish Sebastes ruberrimus along the Pacific coast of Canada: biology, distribution and abundance trends*. Retrieved from Nanaimo, BC:

Ye, X., & Randal, D. J. (1991). The effect of water pH on swimming performance in rainbow trout (*Salmo gairdneri*, Richardson). *Fish Physiology and Biochemistry, 9*. doi:https://doi.org/10.1007/BF01987607

Ye, X., Randal, D. J., & He, X. (1991). The effect of acid water on oxygen consumption, circulating catecholamines and blood ionic and acid-base status in rainbow trout (*Salmo gairdneri*, Richardson). *Fish Physiology and Biochemistry*(9).

Yoklavich, M. M., Greene, H. G., Cailliet, G. M., Sullivan, D. E., Lea, R. N., & Love, M. S. (2000). Habitat associations of deep-water rockfishes in a submarine canyon: an example of a natural refuge. *Fish Bull, 98*, 625-641.

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. *Conserv Biol, 20*(1), 190-200.

Zalizniak, L., & Nugegoda, D. (2006). Effect of sublethal concentrations of chlorpyrifos on three successive generations of Daphnia carinata. *Ecotoxicology and Environmental Safety, 64*(2), 207-214.

Zha, J., Wang, Z., Wang, N., & Ingersoll, C. (2007). Histological alternation and vitellogenin induction in adult rare minnow (Gobiocypris rarus) after exposure to ethynylestradiol and nonylphenol. *Chemosphere, 66*(3), 488-495. doi:10.1016/j.chemosphere.2006.05.071

Zhang, L., & Wang, W. X. (2005). Effects of Zn pre-exposure on Cd and Zn bioaccumulation and metallothionein levels in two species of marine fish. *Aquatic Toxicology, 73*(4), 353-369. Retrieved from https://doi.org/10.1016/j.aquatox.2005.04.001

Zhang, L., Wei, C., Zhang, H., & Song, M. (2017). Criteria for assessing the ecological risk of nonylphenol for aquatic life in Chinese surface fresh water. *Chemosphere, 184*, 569-574. doi:10.1016/j.chemosphere.2017.06.035

Appendix A: Swinomish Tribe Water Quality Standards Code

Appendix B: Literature Search Process for Biological Evaluations

Appendix C: Effects Assessment Data for Multiple Routes of Exposure

This appendix includes the following:

Table C1. Sources and values of BCFs and log KOW used in multiple routes of exposure analysis for fish species

Excel spreadsheets that include the data and analyses for the multiple routes of exposure summaries in section 5.7 of the BE for the following criteria. The spreadsheets are provided as Microsoft Excel files separately from the BE:

C2 Arsenic

C3 Carbaryl

C4 Chlorpyrifos

C5 Chromium

C6 Diazinon

C7 Malathion

C8 Nickel

C9 Nonylphenol

C10 Lead

C11 Pentachlorophenol

C12 Silver

C13 Zinc

C14 Summary

Appendix D: Oregon Acute Cadmium Analysis

Appendix E: Freshwater Criteria Effects Assessment for Ammonia

Appendix F: Estimation of Acute and Chronic Toxicity Values for Rockfish for Criteria in Assessment Category 1

This appendix is comprised of spreadsheets that include the data and analyses used to evaluate criteria in assessment category 1 relative to rockfish (i.e., marine cadmium, copper, selenium, and silver criteria) as discussed in section 5.8.20 of the BE. The spreadsheets are provided separately as Microsoft Excel files.

Appendix G: Freshwater Toxicity Data for Silver