



UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

OFFICE OF CHEMICAL SAFETY
AND POLLUTION PREVENTION

**DRAFT BIOLOGICAL OPINIONS ISSUED UNDER THE ENDANGERED SPECIES ACT, BY THE
NATIONAL MARINE FISHERIES SERVICE, RELATED TO PESTICIDES AND PACIFIC
SALMON AND STEELHEAD SPECIES**

EPA has initiated formal consultation with the National Marine Fisheries Service on the potential effects of certain pesticides to Pacific salmon and steelhead listed under the Endangered Species Act as either threatened or endangered. As EPA receives draft Biological Opinions relative to these consultations, they will be posted to www.epa.gov/espp and included in a public docket EPA-HQ-OPP-2008-0654 so EPA may receive public input on any changes to a pesticide's registration recommended by the National Marine Fisheries Service.

BACKGROUND

The Endangered species Act (ESA) requires that Federal Agencies assess their "actions" to determine whether species listed as threatened or endangered under the ESA, may be affected by those actions, or whether critical habitat may be adversely modified. The registered uses of a pesticide constitute an EPA "action" under the ESA.

If EPA determines a pesticide's registered uses are likely to adversely affect a federally listed threatened or endangered species (listed species) or modify its critical habitat, EPA initiates "formal consultation" with the U.S. Fish and Wildlife Service or the National Marine Fisheries Service (the Service or Services), as appropriate. In response to a Federal Agency initiating formal consultation, the Service(s) develops a Biological Opinion (BO) in which it provides its opinion on whether the "action" is likely to jeopardize the continued existence of a listed species or is likely to adversely modify designated critical habitat and, if so, describes alternatives to avoid jeopardy.

PUBLIC INPUT

In 2005, EPA published in the Federal Register (FR 70 No. 211 pp. 66392-66402), a document titled *Endangered Species Protection Program Field Implementation*. That notice of how EPA intends to implement its responsibilities under the ESA, states (p 66401):

"If EPA must formally consult with the Services, after the Services issue a draft Biological Opinion, EPA will welcome input from State, Tribal and local governments **on draft reasonable and prudent measures and alternatives**. The purpose of this review would be to determine whether the alternatives or measures can be reasonably implemented and whether there are different measures that may provide adequate protection but result in less impact to pesticide users. The Agency will consider this input in developing its response to draft Biological Opinions."

EPA also is seeking input from the public in general, pesticide users, and public interest organizations on whether the alternatives or measures can be reasonably implemented and whether there are different measures that may provide adequate protection but result in less impact to pesticide users. The Agency will also consider this input in developing its response to draft Biological Opinions.

APPLICANT INPUT

Further, the Services' Consultation Handbook (pp. 2-13), supports their consultation regulations and states:

“... the Service and the action agency meet their obligations to [the applicant or pesticide registrant] as outlined in 50 CFR section 402 through the following: The applicant is entitled to **review draft Biological Opinions obtained through the action agency and to provide comments through the action agency. The Service will discuss the basis of their biological determination with the applicant and seek the applicant's expertise in identifying reasonable and prudent alternatives ...**”.

COMMENTS

Draft Biological Opinions are being included in the docket (insert Docket ID) and posted to EPA's Web site (<http://www.epa.gov/oppfead1/endanger/litstatus/effects>) to seek input on the Service's recommended reasonable and prudent measures and alternatives, as noted above. Such input should be submitted **by April 5, 2011** in order to be considered in EPA's response to the Draft Biological Opinion. Comments received by EPA on other aspects of the Draft Biological Opinion, will be forwarded to the Service for their consideration.

As stated in the Services' regulations (50 CFR 402.14(g)(5)):

“All comments on the draft Biological Opinion must be submitted to the Service through the Federal agency, although the applicant may send a copy of its comments directly to the Service.”

SUBMITTING YOUR COMMENTS

You may submit your comments, identified by the docket identification (ID) number EPAHQ-OPP-2008-0654 and the pesticide to which the Biological Opinion pertains, by one of the following methods:

Federal eRulemaking Portal: <http://www.regulations.gov>. Follow the on-line instructions for submitting comments.

Mail: Office of Pesticide Programs (OPP) Regulatory Public Docket (7502P), Environmental Protection Agency, 1200 Pennsylvania Ave., NW, Washington, DC 20460-0001.

Delivery: OPP Regulatory Public Docket (7502P), Environmental Protection Agency, Rm. S-4400, One Potomac Yard (South Bldg.), 2777 S. Crystal Dr., Arlington, VA. Deliveries are only accepted during the Docket's normal hours of operation (8:30 a.m. to 4 p.m., Monday through Friday, excluding legal holidays). Special arrangements should be made for deliveries of boxed information. The Docket Facility telephone number is (703) 305-5805.

Instructions: EPA's policy is that all comments received will be included in the docket without change and may be made available on-line at <http://www.regulations.gov>, including any personal information provided, unless the comment includes information claimed to be Confidential Business Information (CBI) or other information whose disclosure is restricted by statute. Do not submit information that you consider to be CBI or otherwise protected through regulations.gov or e-mail. The regulations.gov website is an "anonymous access" system, which means EPA will not know your identity or contact information unless you provide it in the body of your comment. If you send an e-mail comment directly to EPA without going through regulations.gov, your e-mail address will be automatically captured and included as part of the comment that is placed in the docket and made available on the Internet. If you submit an electronic comment, EPA recommends that you include your name and other contact information in the body of your comment and with any disk or CD-ROM you submit. If EPA cannot read your comment due to technical difficulties and cannot contact you for clarification, EPA may not be able to consider your comment. Electronic files should avoid the use of special characters, any form of encryption, and be free of any defects or viruses.

Docket: All documents in the docket are listed in the docket index available in regulations.gov. To access the electronic docket, go to <http://www.regulations.gov>, select "Advanced Search," then "Docket Search." Insert the docket ID number where indicated and select the "Submit" button. Follow the instructions on the regulations.gov website to view the docket index or access available documents. Although, listed in the index, some information is not publicly available, e.g., CBI or other information whose disclosure is restricted by statute. Certain other material, such as copyrighted material, is not placed on the Internet and will be publicly available only in hard copy form. Publicly available docket materials are available either in the electronic docket at <http://www.regulations.gov>, or, if only available in hard copy, at the OPP Regulatory Public Docket in Rm. S-4400, One Potomac Yard (South Bldg.), 2777 S. Crystal Dr., Arlington, VA. The hours of operation of this Docket Facility are from 8:30 a.m. to 4 p.m., Monday through Friday, excluding legal holidays. The Docket Facility telephone number is (703) 305-5805.

The Draft Reasonable and Prudent Alternatives and Measures on which EPA is seeking input begin on page 726 of this PDF file (page 722 of the document itself).



UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
Silver Spring, MD 20910

MAR - 1 2011

Mr. Steven Bradbury
Director, Office of Pesticide Programs
U.S. Environmental Protection Agency
One Potomac Yard
2777 S. Crystal Drive
Arlington, VA 22202

Dear Mr. Bradbury:

Enclosed is NOAA's National Marine Fisheries Service's (NMFS) draft biological opinion (Opinion), issued under the authority of section 7(a)(2) of the Endangered Species Act (ESA) of 1973 as amended (16 U.S.C. 1536(a)(2)), on the effects of the U.S. Environmental Protection Agency's (EPA) proposed registration of pesticide products containing the active ingredients 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil, on endangered species, threatened species, and critical habitat that has been designated for those species. This Opinion assesses the effects of all pesticides containing the above-listed ingredients on 28 listed Pacific salmonids.

After considering the status of the listed resources, the environmental baseline, and the direct, indirect, and cumulative effects of EPA's proposed action on listed species, NMFS preliminarily concludes that pesticide products containing triclopyr BEE, linuron, and captan are not likely to jeopardize the continuing existence of any listed Pacific salmonids or destroy or adversely modify designated critical habitat as described in the attached draft Opinion. NMFS also concludes that the effects of products containing diuron will destroy or adversely modify designated critical habitat for some listed Pacific salmonids as described in the attached draft Opinion. Finally, NMFS concludes that the effects of products containing 2,4-D, and chlorothalonil are likely to jeopardize the continued existence of some listed Pacific salmonids and to destroy or adversely modify designated habitat for some listed Pacific salmonids as described in the attached draft Opinion. As NMFS has not designated critical habitat for Lower Columbia River coho salmon or Puget Sound steelhead, the Opinion presents no further critical habitat analysis for these species.

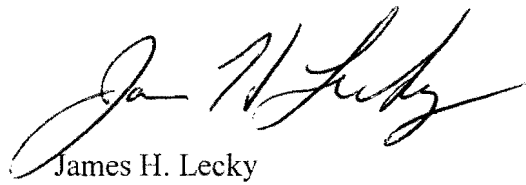
The draft Opinion includes a draft reasonable and prudent alternative that will be discussed with the Office of Pesticides Programs. We look forward to continuing our work with you and to work with applicants on the reasonable and prudent alternative and in particular to develop risk reduction measures in elements 3 and 5. NMFS must issue its final biological opinion by April 30, 2011. We request that EPA provide its comments and the applicants' comments to us no later than April 12, 2011, to allow adequate time for consideration and incorporation of comments.



This Opinion assesses effects to listed Pacific salmonids pursuant to the ESA. It does not address EPA's obligation under the Magnuson-Stevens Fishery Conservation and Management Act to consult on effects to essential fish habitat (EFH) for salmonids and other federally-managed species. Please contact Mr. Tom Bigford or Ms. Susan-Marie Stedman in NMFS' Office of Habitat Conservation at 301-713-4300 regarding the EFH consultation process.

If you have questions regarding this Opinion please contact me or Ms. Angela Somma, Chief of our Endangered Species Division at (301) 713-1401.

Sincerely,

A handwritten signature in black ink, appearing to read "Ja H Lecky", written in a cursive style.

James H. Lecky
Director, Office of Protected Resources

this page left blank

Table of Contents

List of Figures	v
List of Tables	viii
Background	3
Consultation History	6
<i>Species Addressed in the BEs</i>	<i>20</i>
Description of the Proposed Action	23
<i>The Federal Action</i>	<i>23</i>
Action Area	62
Approach to this Assessment	65
<i>Overview of NMFS' Assessment Framework</i>	<i>65</i>
<i>Evidence Available for the Consultation</i>	<i>68</i>
<i>Application of Approach in this Consultation</i>	<i>69</i>
<i>Other Considerations</i>	<i>79</i>
Status of Listed Resources	81
<i>Species Status</i>	<i>82</i>
<i>Conservation Role of Critical Habitat for the Species</i>	<i>84</i>
<i>Chinook Salmon</i>	<i>88</i>
<i>Puget Sound Chinook Salmon</i>	<i>93</i>
<i>Lower Columbia River Chinook Salmon</i>	<i>101</i>
<i>Upper Columbia River Spring-run Chinook Salmon</i>	<i>109</i>
<i>Snake River Fall-run Chinook Salmon</i>	<i>115</i>
<i>Snake River Spring/Summer-Run Chinook Salmon</i>	<i>120</i>
<i>Upper Willamette River Chinook Salmon</i>	<i>126</i>
<i>California Coastal Chinook Salmon</i>	<i>131</i>
<i>Central Valley Spring-run Chinook Salmon</i>	<i>139</i>
<i>Sacramento River Winter-run Chinook Salmon</i>	<i>147</i>
<i>Chum Salmon</i>	<i>151</i>
<i>Hood Canal Summer-run Chum Salmon</i>	<i>153</i>

<i>Columbia River Chum Salmon</i>	161
<i>Coho Salmon</i>	168
<i>Lower Columbia River (LCR) Coho Salmon</i>	171
<i>Oregon Coast Coho Salmon</i>	177
<i>Southern Oregon/Northern California Coast Coho Salmon</i>	184
<i>Central California Coast Coho Salmon</i>	187
<i>Sockeye Salmon</i>	193
<i>Ozette Lake Sockeye Salmon</i>	195
<i>Snake River Sockeye Salmon</i>	200
<i>Steelhead</i>	206
<i>Puget Sound Steelhead DPS</i>	209
<i>Lower Columbia River Steelhead</i>	215
<i>Upper Willamette River Steelhead</i>	222
<i>Middle Columbia River Steelhead</i>	229
<i>Upper Columbia River Steelhead</i>	238
<i>Snake River Steelhead</i>	244
<i>Northern California Steelhead</i>	251
<i>Central California Coast Steelhead</i>	258
<i>California Central Valley Steelhead</i>	265
<i>South-Central California Coast Steelhead</i>	273
<i>Southern California Steelhead</i>	277
<i>Environmental Baseline</i>	282
<i>Natural Mortality Factors</i>	283
<i>Anthropogenic Mortality Factors</i>	292
<i>Baseline Pesticide Detections in Aquatic Environments</i>	292
<i>Baseline Water Temperature - Clean Water Act</i>	296
<i>Baseline Habitat Condition</i>	299
<i>Baseline Pesticide Consultations</i>	302
<i>Geographic Regions</i>	303
<i>Southwest Coast Region</i>	306
<i>Other Land Uses in the Southwest Coast Region</i>	321

<i>Pacific Northwest Region</i>	<i>333</i>
<i>Columbia River Basin</i>	<i>337</i>
<i>Other Land Use in the Pacific Northwest Region.....</i>	<i>348</i>
<i>Puget Sound Region.....</i>	<i>361</i>
<i>Oregon-Washington-Northern California Coastal Drainages.....</i>	<i>377</i>
<i>Integration of the Environmental Baseline on Listed Resources.....</i>	<i>382</i>
<i>Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids ...</i>	<i>384</i>
<i>Exposure Analysis.....</i>	<i>384</i>
<i>Modeling: Estimates of Exposure to the six a.i.s.....</i>	<i>413</i>
<i>Monitoring Data: Measured Concentrations of Parent Compounds and Degradates in Surface Waters.....</i>	<i>429</i>
<i>Response Analysis.....</i>	<i>459</i>
<i>Risk Characterization</i>	<i>544</i>
<i>Integration of Exposure and Response</i>	<i>545</i>
<i>Conclusions Regarding Risk Associated with Specific a.i.s</i>	<i>601</i>
<i>Cumulative Effects.....</i>	<i>602</i>
<i>Integration and Synthesis for Threatened and Endangered Pacific Salmonids</i>	<i>606</i>
<i>Summaries of Individual a.i.s.....</i>	<i>608</i>
<i>ESU/DPS Specific Evaluations.....</i>	<i>618</i>
<i>Integration and Synthesis for Designated Critical Habitat.....</i>	<i>659</i>
<i>Designated Critical Habitat Specific Evaluations for Each a.i.....</i>	<i>662</i>
<i>Conclusion.....</i>	<i>698</i>
<i>Reasonable and Prudent Alternatives</i>	<i>702</i>
<i>Specific Elements of the Reasonable and Prudent Alternative.....</i>	<i>703</i>
<i>Incidental Take Statement.....</i>	<i>706</i>
<i>Amount or Extent of Take</i>	<i>707</i>
<i>Reasonable and Prudent Measures</i>	<i>710</i>
<i>Terms and Conditions.....</i>	<i>711</i>
<i>Conservation Recommendations.....</i>	<i>712</i>
<i>Conservation Recommendations.....</i>	<i>713</i>
<i>Reinitiation Notice</i>	<i>713</i>

<i>References</i>	<i>715</i>
<i>Appendix 1 – Population Modeling.....</i>	<i>767</i>
<i>Appendix 2. Species and Population Annual Rates of Growth.....</i>	<i>806</i>
<i>Appendix 3: Abbreviations / Acronyms.....</i>	<i>817</i>
<i>Appendix 4: Glossary</i>	<i>823</i>
<i>Appendix 5: Co-occurrence Analysis for Integration and Synthesis.....</i>	<i>832</i>
<i>Appendix 6: Generalized Average Annual Run-timing for ESA listed</i>	<i>888</i>
<i>Pacific Coast Salmon and Steelhead.....</i>	<i>888</i>
<i>Appendix 7 – Ecological Effects Data from EPA's Red Legged Frog BE's</i>	<i>920</i>
<i>Appendix 8. AgDrift Tier III aerial assessment input values for mixture calculation</i>	<i>921</i>

List of Figures

Figure 1. Stressors of the Action.....	25
Figure 2. Map showing extent of inland action area with the range of all ESU and DPS boundaries for ESA listed salmonids highlighted in gray.	64
Figure 3. Conceptual framework for assessing risks of EPA’s action to ESA listed resources.	71
Figure 4. Exposure pathways of the stressors of the action and general responses of Pacific salmonids and habitat.....	73
Figure 5. Puget Sound Chinook salmon distribution.	96
Figure 6. Puget Sound Chinook salmon Conservation Values per Sub-watershed.	100
Figure 7. Lower Columbia River Chinook salmon distribution.	102
Figure 8. Lower Columbia River Chinook salmon Conservation Values per Sub-Area.	108
Figure 9. Upper Columbia River Chinook salmon distribution.....	109
Figure 10. Upper Columbia River Spring-run Chinook salmon Conservation Values per Sub-Area	115
Figure 11. Snake River Fall-run Chinook salmon distribution.....	116
Figure 12. Snake River Spring/Summer-run Chinook salmon distribution.....	121
Figure 13. Upper Willamette River Chinook salmon distribution.....	127
Figure 14. California Coastal Chinook salmon distribution.	133
Figure 15. California Coastal Chinook salmon Conservation Values per Sub-Area.....	138
Figure 16. Central Valley Spring-run Chinook salmon distribution.....	140
Figure 17. Central Valley Spring-run Chinook salmon Conservation Values per Sub-Area	146
Figure 18. Sacramento River Winter-run Chinook salmon distribution.....	148
Figure 19. Hood Canal Summer-run Chum salmon distribution.....	155
Figure 20. Hood Canal Summer-run Conservation Values per Sub-area.....	160
Figure 21. Columbia River Chum salmon distribution.....	162
Figure 22. Columbia River Chum salmon Conservation Values per Sub-area.	167
Figure 23. LCR coho salmon distribution.....	173

Figure 24. Oregon Coast Coho salmon distribution.	179
Figure 25. Oregon Coast Coho salmon Conservation Values per Sub-area.	183
Figure 26. SONCC coho salmon distribution.	186
Figure 27. CCC Coho salmon distribution.	190
Figure 28. Ozette Lake Sockeye salmon distribution.	197
Figure 29. SR Sockeye Salmon distribution.	203
Figure 30. Puget Sound steelhead distribution.	211
Figure 31. Lower Columbia River steelhead distribution.	217
Figure 32. Lower Columbia River Steelhead Conservation Values per Sub-area.	221
Figure 33. UWR Steelhead distribution.	224
Figure 34. Upper Willamette River Steelhead Conservation Values per Sub-Area.	228
Figure 35. MCR Steelhead distribution.	231
Figure 36. Upper Willamette River Steelhead Conservation Values per Sub-area.	237
Figure 37. UCR Steelhead distribution.	239
Figure 38. Upper Columbia River Steelhead Conservation Values per Sub-area.	243
Figure 39. SR Basin Steelhead distribution.	245
Figure 40. Snake River Steelhead Conservation Values per Sub-area.	250
Figure 41. Northern California Steelhead distribution.	253
Figure 42. Northern California Steelhead Conservation Values per Sub-area.	258
Figure 43. CCC steelhead distribution.	260
Figure 44. Central California Coast Steelhead Conservation Values per Sub-area.	264
Figure 45. CCV steelhead distribution.	267
Figure 46. California Central Valley Steelhead Conservation Value per Sub-area.	272
Figure 47. South-Central California Coast Steelhead Conservation Values per Sub-area.	275
Figure 48. S-CCC steelhead distribution.	276
Figure 49. Southern California steelhead distribution.	278
Figure 50. Southern California Steelhead Conservation Values per Sub-area.	281
Figure 51. Landuse in Southwest Region.	311
Figure 52. Southwest Coast 303(d) waters, dams, and NPDEs permit sites.	324
Figure 53. Pacific Northwest Landuse.	340

Figure 54. Pacific Northwest 303(d) waters, dams, and NPDES permit sites.....	355
Figure 55. Exposure analysis.	385
Figure 56. Chemical structure of triclopyr BEE.	394
Figure 57. Chemical structure of diuron	395
Figure 58. Chemical structure of linuron.	396
Figure 59. Chemical structure of captan.	398
Figure 60. Chemical structure of chlorothalonil.	399
Figure 61. Distribution of NAWQA monitoring sites and listed Pacific salmonids.	435
Figure 62. Distribution of Washington Department of Ecology sample stations compared to the distribution of listed salmon ESUs/DPSs.	442
Figure 63. Response analysis.	459
Figure 64. Part I of a conceptual model of potential effects of herbicides aquatic communities.....	475
Figure 65. Part II of a conceptual model of potential effects of herbicides aquatic communities.....	476
Figure 66. 2,4-D acid, and example amine and ether forms.	479
Figure 67. Triclopyr acid and triclopyr BEE.	480
Figure 68. Basic phenylurea structure.	495
Figure 69. Structures of diuron and linuron.....	496
Figure 70. Toxic degradates of diuron and linuron.....	512
Figure 71. Structure of Captan.....	519
Figure 72. Captan Mode of Action	519
Figure 73. Structures of Folpet (left) and Captafol (right).....	520
Figure 74. Structure of Chlorothalonil.....	530
Figure 75. Chlorothalonil substitutions at 4 and 6 position (-SH represents any –S R group).....	531
Figure 76. Schematic of the Risk Characterization Phase.	544

List of Tables

Table 1. EPA’s effects determinations.....	22
Table 2. Chemical forms of 2,4-D products that are currently registered for use in the action area.	29
Table 3. Summary of all authorized use sites and application restrictions for active 2,4-D.	30
Table 4. Summary of all authorized use sites and application restrictions for active triclopyr BEE products.	37
Table 5. Summary of all authorized use sites and application restrictions for active diuron products.	41
Table 6. Summary of all authorized use sites and application restrictions for active linuron products.	45
Table 7. Summary of all authorized use sites and application restrictions for active captan products.	48
Table 8. Summary of all authorized use sites and application restrictions for active chlorothalonil products.	57
Table 9. Essential physical and biological features named as PCEs in all salmonid critical habitat designations.....	75
Table 10. Examples of salmonid lifestage and habitat assessment endpoints and measures.....	77
Table 11. Listed Species and Critical Habitat (denoted by asterisk) in the Action Area.	81
Table 12. Puget Sound Chinook salmon - preliminary population structure, abundances, and hatchery contributions.....	93
Table 13. Puget Sound Chinook salmon watersheds with conservation values.	98
Table 14. Lower Columbia River Chinook salmon - population structure, abundances, and hatchery contributions.....	104
Table 15. LCR Chinook salmon HUC 5 watersheds with conservation values.	107
Table 16. Upper Columbia River Spring-run Chinook salmon - preliminary population structure, abundances, and hatchery contributions.	110

Table 17. UCR Spring-run Chinook salmon watersheds with conservation values.	113
Table 18. Snake River Spring/Summer-run Chinook salmon populations, abundances, and hatchery contributions.	123
Table 19. Upper Willamette River Chinook salmon independent populations core (C) and genetic legacy (G) populations, and hatchery contributions.	129
Table 20. UWR Chinook salmon watersheds with conservation values.	131
Table 21. California Coastal Chinook salmon fall-run populations-preliminary population structure, abundances, and hatchery contributions.	134
Table 22. CC Chinook salmon CALWATER HSA watersheds with conservation values.	136
Table 23. Central Valley Spring-run Chinook salmon--preliminary population structure, historic and most recent natural production, spawner abundance, and hatchery contributions.	142
Table 24. CV Spring-run Chinook salmon CALWATER HSA watersheds with conservation values.	144
Table 25. Hood Canal Summer-run Chum salmon populations, abundances, and hatchery contributions.	153
Table 26. Hood Canal Summer-run chum salmon watersheds with conservation values.	158
Table 27. Populations within the Columbia River chum salmon ESU, their abundances, and hatchery input.	163
Table 28. CR chum salmon watersheds with conservation values.	166
Table 29. Lower Columbia River coho salmon populations, estimated natural spawner abundances, and hatchery contributions.	175
Table 30. Oregon Coast Coho salmon potential historic and estimated recent spawner abundances, and hatchery contributions.	179
Table 31. OC coho salmon watersheds with conservation values.	181
Table 32. Central California Coast Coho salmon populations, abundances, and releases of hatchery raised smolt.	191
Table 33. Geometric mean estimates of escapement of natural spawners for Puget Sound steelhead.	213

Table 34. LCR Steelhead salmon populations, historic abundances (Good, Waples et al. 2005), 1998 – 2002 and 2004 to 2005 geometric mean abundance (Good, Waples et al. 2005)(Salmon Scape Query 2009), and hatchery contributions.	218
Table 35. LCR steelhead watersheds with conservation values.	219
Table 36. Upper Willamette River steelhead salmon populations, core (C) and genetic legacy (G) populations, abundances, and hatchery contributions.....	225
Table 37. UWR steelhead watersheds with conservation values.....	226
Table 38. Middle Columbia River steelhead independent populations, abundances, and hatchery contributions.....	233
Table 39. MCR steelhead watersheds with conservation values.	235
Table 40. Upper Columbia River Steelhead salmon populations, abundances, and hatchery contributions.....	241
Table 41. UCR Steelhead watersheds with conservation values.	242
Table 42. SR Basin Steelhead salmon populations, abundances, and hatchery contributions.	246
Table 43. SR steelhead watersheds with conservation values.	248
Table 44. NC Steelhead salmon historic functionally independent populations and their abundances and hatchery contributions.	254
Table 45. NC steelhead CALWATER HSA watersheds with conservation values	256
Table 46. CCC Steelhead populations, historic population type, abundances, and hatchery contributions	261
Table 47. CCC steelhead CALWATER HSA watersheds with conservation values....	263
Table 48. CCV spring-run Chinook salmon CALWATER HSA watersheds with conservation values.....	270
Table 49. Number of South-Central California Coast steelhead CALWATER HSA watersheds with conservation values.....	274
Table 50. Southern California Steelhead salmon populations, abundances, and hatchery contributions.	279
Table 51. Southern California steelhead CALWATER HSA watersheds with conservation values.....	280

Table 52. Washington State water temperature thresholds for salmonid habitat. These temperatures are representative of limits set by California, Idaho, and Oregon.....	297
Table 53. Number of kilometers of river, stream and estuaries included in state 303(d) lists due to temperature that are located within each salmonid ESU/DPS.	299
Table 54. USGS Subregions and accounting units within the Northwest and Southwest Regions, along with ESUs/DPSs present within the area.	304
Table 55. Area of land use categories within the range Chinook and Coho Salmon ESUs in km ²	307
Table 56. Area of Land Use Categories within the Range of Steelhead Trout DPSs (km ²).	308
Table 57. Select rivers in the southwest coast region.	309
Table 58. Land uses and population density in several southwest coast watersheds. ...	310
Table 59. California's 2006 Section 303(d) List of Water Quality Limited Segments: segments listed for exceeding temperature limits.	317
Table 60. Summary of State-level limitations on pesticide use in Idaho, Oregon, and Washington. All materials are available on organization web-sites.	328
Table 61. Area of land use categories within Chinook Salmon ESUs in km ² where bolded numbers are totals of each category. Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies.....	335
Table 62. Area of land use categories within chum and coho ESUs in km ²	336
Table 63. Area of land use categories within sockeye ESUs and steelhead DPSs in km ²	337
Table 64. Select tributaries of the Columbia River.....	338
Table 65. Land use and population density in select tributaries of the Columbia River	339
Table 66. Summarized detection information from (Carpenter, Sobieszczyk et al. 2008).	346
Table 67. Summarized detection information from (Carpenter, Sobieszczyk et al. 2008). Note that percentages aren't comparable because results were pooled from multiple sources.....	347
Table 68. Examples of Water Quality Contaminants in Residential and Urban Areas.	367

Table 69. Pollutants of Concern in Puget Sound (PSAT 2005).....	372
Table 70. Change in total area (acres ²) of tidal wetlands in Oregon (tidal marshes and swamps) due to filling and diking between 1870 and 1970 (Good 2000).	380
Table 71. General life-histories of Pacific salmonids.....	386
Table 72. Chemical structures and the molecular weight ratios of various chemical forms of 2,4-D ¹	390
Table 73. Environmental fate characteristics of 2,4-D acid ¹	393
Table 74. Environmental fate characteristics of triclopyr acid ¹	394
Table 75. Environmental fate characteristics of diuron ¹	396
Table 76. Environmental fate characteristics of linuron ¹	398
Table 77. Environmental fate characteristics of captan ¹	399
Table 78. Environmental fate characteristics of chlorothalonil ¹	400
Table 79. Metabolites and degradates of 2,4-D.	401
Table 80. Metabolites and degradates of triclopyr BEE.	401
Table 81. Metabolites and degradates of diuron.	402
Table 82. Metabolites and degradates of linuron.....	402
Table 83. Metabolites and degradates of captan.	403
Table 84. Metabolites and degradates of chlorothalonil.....	404
Table 85. Co-occurrence of listed Pacific salmonids with potential application of pesticides to use sites within the salmons freshwater distribution.....	406
Table 86. Summary of use sites approved on active labels.	414
Table 87. Examples of current registered uses of the six a.i.s and the exposure method used by EPA in salmonid BEs.	414
Table 88. PRZM-EXAMS exposure estimates from EPA's salmonid BEs.	417
Table 89. Estimates of pesticide concentrations in surface waters from California red legged frog BEs.....	419
Table 90. Average initial concentration of any a.i. in surface water resulting from a direct overspray of aquatic habitat.	424
Table 91 Estimated average initial pesticide concentrations in a floodplain habitat that is 2m wide and 0.1m deep using AgDrift.....	425

Table 92. AgDrift estimated concentrations of pesticides in surface water adjacent to aerial application at the maximum labeled use rate for EPA Reg. No. 228-654 (12 lbs diuron/A and 1.5 lb imazapyr/A).	429
Table 93. Number of NAWQA sample sites within the distribution of listed Pacific salmonids.	434
Table 94. Concentrations of Parent Pesticides in NAWQA Water Samples for California, Idaho, Oregon, and Washington.	437
Table 95. Concentrations of Degradates in NAWQA Water Samples for California, Idaho, Oregon, and Washington.	437
Table 96. Surface Water Concentrations of Pesticides in CDPR Database.....	438
Table 97. Concentrations of Pesticides in Washington EIM Database.	440
Table 98. Number of stations sampled for pesticide in Washington Department of Ecology Study.	441
Table 99. Washington Department of Ecology sample sites within the distribution of listed Pacific salmonids.....	443
Table 100. Concentrations of pesticides detected in recent studies by Washington Department of Ecology (2003-2009) ¹	444
Table 101. Concentrations of diuron detected in surface water runoff.....	446
Table 102. Monitoring Data Concentrations.	448
Table 103. Examples of pesticide product ingredients.	449
Table 104. Detection and concentrations of nonionic detergent degradates in streams of the U.S. (Koplin et al 2002).	450
Table 105. Examples of label recommended tank mixtures.	451
Table 106. Chemical Concentration Ranges in Monitoring Data and Modeling.	455
Table 107. Acute toxicity data for fish.	483
Table 108. Chronic toxicity data for fish.	484
Table 109. Acute toxicity data for aquatic invertebrates.	488
Table 110. Chronic toxicity data for aquatic invertebrates.	488
Table 111. Toxicity data for aquatic plants.	490
Table 112. Degradate toxicity data for Triclopyr.	491
Table 113. Terrestrial plant data.	492

Table 114. Assessment Endpoints and Measures for 2,4-D.	493
Table 115. Assessment Endpoints and Measures for Triclopyr.	494
Table 116. Acute toxicity data for fish.	499
Table 117. Chronic toxicity data for fish.	502
Table 118. Acute toxicity data for aquatic invertebrates.	505
Table 119. Chronic toxicity data for aquatic invertebrates.	507
Table 120. Toxicity data for aquatic plants.	509
Table 121. Degradates of Diuron and Linuron.	513
Table 122. Diuron degradate toxicity data for <i>D. magna</i> and algae.	514
Table 123. Terrestrial plant data.	516
Table 124. Assessment Endpoints and Measures for Diuron.	517
Table 125. Assessment Endpoints and Measures for Linuron.	518
Table 126. Assessment Endpoints and Measures for Captan	528
Table 127. Assessment Endpoints and Measures for Chlorothalonil	543
Table 128. Summary of estimated and measured environmental concentrations, co- occurrence, and effect concentrations for 2,4-D.	548
Table 129. Summary of estimated and measured environmental concentrations, co- occurrence, and effect concentrations for triclopyr BEE.	552
Table 130. Summary of estimated and measured environmental concentrations, co- occurrence, and effect concentrations for diuron.	556
Table 131. Summary of estimated and measured environmental concentrations, co- occurrence, and effect concentrations for linuron.	560
Table 132. Summary of estimated and measured environmental concentrations, co- occurrence, and effect concentrations for captan.	563
Table 133. Summary of estimated and measured environmental concentrations, co- occurrence, and effect concentrations for Chlorothalonil.	566
Table 134. AgDrift estimated concentrations of pesticides in surface water adjacent to aerial application at the maximum labeled use rate for EPA Reg. No. 228-654 (12 lbs diuron/A and 1.5 lb imazapyr/A).	568

Table 135. AgDrift estimated point deposition of diuron and imazapyr at various distances downwind from application site at the maximum labeled use rate for EPA Reg. No. 228-654 (12 lbs diuron/A and 1.5 lb imazapyr/A).....	569
Table 136. Toxicity of diuron and imazapyr to nontarget plants.....	569
Table 137. Summary of individual-based risk hypotheses.	589
Table 138. Assessment endpoint toxicity values ($\mu\text{g/L}$) for saltwater aquatic organisms presented in salmonids BEs.	598
Table 139. Puget Sound Chinook Salmon.	620
Table 140. Lower Columbia River Chinook Salmon.	621
Table 141. Upper Columbia Spring-Run Chinook Salmon.	622
Table 142. Snake River Fall-run Chinook Salmon.	624
Table 143. Snake River Spring/Summer-run Chinook Salmon.....	626
Table 144. Upper Willamette River Chinook Salmon.....	627
Table 145. California Coastal Chinook Salmon.	629
Table 146. Central Valley Spring-run Chinook Salmon.....	630
Table 147. Sacramento Winter-run Chinook Salmon.....	632
Table 148. Hood Canal Summer-run Chum Salmon.	633
Table 149. Columbia River chum Salmon.....	634
Table 150. Lower Columbia River coho Salmon.	636
Table 151. Oregon Coast Coho Salmon.	637
Table 152. Southern Oregon/Northern California Coast Coho Salmon.	638
Table 153. Central California Coast Coho Salmon.	640
Table 154. Ozette Lake sockeye Salmon.....	642
Table 155. Snake River Sockeye Salmon.....	643
Table 156. Puget Sound steelhead.	644
Table 157. Lower Columbia River steelhead.	646
Table 158. Upper Willamette Steelhead.	647
Table 159. Middle Columbia River steelhead.	648
Table 160. Upper Columbia River steelhead.....	650
Table 161. Snake River steelhead.....	651
Table 162. Northern California steelhead.....	653

Table 163. Central California Coast steelhead.	654
Table 164. California Central Valley steelhead.	656
Table 165. South-Central California Coast steelhead.	657
Table 166. Southern California steelhead.	658
Table 167. Potential for reduction in reproduction, abundance, or distribution.	659
Table 168. Co-occurrence of land use types and likelihood of a reduction in PCEs for Puget Sound Chinook salmon.	664
Table 169. Co-occurrence of land use types and likelihood of a reduction in PCEs for Lower Columbia River Chinook salmon.	665
Table 170. Co-occurrence of land use types and likelihood of a reduction in PCEs for Upper Columbia River Chinook Salmon.	666
Table 171. Co-occurrence of land use types and likelihood of a reduction in PCEs for Snake River Fall-run Chinook Salmon.	668
Table 172. Co-occurrence of land use types and likelihood of a reduction in PCEs for Snake River Spring/Summer-run Chinook Salmon.	669
Table 173. Co-occurrence of land use types and likelihood of a reduction in PCEs for Upper Willamette River Chinook Salmon.	670
Table 174. Co-occurrence of land use types and likelihood of a reduction in PCEs for Central California Chinook Salmon.	671
Table 175. Co-occurrence of land use types and likelihood of a reduction in PCEs for Central Valley Spring-run Chinook Salmon.	673
Table 176. Co-occurrence of land use types and likelihood of a reduction in PCEs for Sacramento River Winter-run Chinook Salmon.	674
Table 177. Co-occurrence of land use types and likelihood of a reduction in PCEs for Hood Canal Summer-run Chum Salmon.	675
Table 178. Co-occurrence of land use types and likelihood of a reduction in PCEs for Columbia River Chum Salmon.	676
Table 179. Co-occurrence of land use types and likelihood of a reduction in PCEs for Oregon Coast Coho Salmon.	677
Table 180. Co-occurrence of land use types and likelihood of a reduction in PCEs for Southern Oregon Northern California Coast Coho Salmon.	679

Table 181. Co-occurrence of land use types and likelihood of a reduction in PCEs for Central California Coast Coho Salmon.....	680
Table 182. Co-occurrence of land use types and likelihood of a reduction in PCEs for Ozette Lake Sockeye Salmon.	681
Table 183. Co-occurrence of land use types and likelihood of a reduction in PCEs for Snake River Sockeye Salmon.	682
Table 184. Co-occurrence of land use types and likelihood of a reduction in PCEs for Lower Columbia River Steelhead.	683
Table 185. Co-occurrence of land use types and likelihood of a reduction in PCEs for Upper Willamette River Steelhead.	684
Table 186. Co-occurrence of land use types and likelihood of a reduction in PCEs for Middle Columbia River Steelhead.....	686
Table 187. Co-occurrence of land use types and likelihood of a reduction in PCEs for Upper Columbia River Steelhead.	687
Table 188. Co-occurrence of land use types and likelihood of a reduction in PCEs for Snake River Basin Steelhead.	688
Table 189. Co-occurrence of land use types and likelihood of a reduction in PCEs for Northern California Steelhead.	690
Table 190. Co-occurrence of land use types and likelihood of a reduction in PCEs for Central California Coast Steelhead.....	691
Table 191. Co-occurrence of land use types and likelihood of a reduction in PCEs for California Central Valley Steelhead.	693
Table 192. Co-occurrence of land use types and likelihood of a reduction in PCEs for South-Central California Coast Steelhead.	694
Table 193. Co-occurrence of land use types and likelihood of adverse effects for Southern California Steelhead.	695
Table 194. Appreciable reduction in conservation value of critical habitat.	696
Table 195. Jeopardy determinations for a.i.s.	699
Table 196. Adverse modification determinations.....	700

**National Marine Fisheries Service
Endangered Species Act Section 7 Consultation
Biological Opinion**

Agency: United States Environmental Protection Agency

Activities Considered: Authorization of pesticide products (as described by product labels) containing the active ingredients 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil, and their formulations in the United States and its affiliated territories.

Consultation Conducted by: Endangered Species Division of the Office of Protected Resources, National Marine Fisheries Service

Approved by: _____

Date: _____

Section 7(a)(2) of the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. §1531 *et seq.*) requires each federal agency to insure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. When a federal agency's action "may affect" a protected species, that agency is required to consult formally with the National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (USFWS), depending upon the endangered species, threatened species, or designated critical habitat that may be affected by the

action (50 CFR §402.14(a)). Federal agencies are exempt from this general requirement if they have concluded, with written concurrence from the U.S. Fish and Wildlife Service, NMFS or both, that an action “may affect but is not likely to adversely affect” endangered species, threatened species or designated critical habitat (50 CFR §420.14(b)).

The United States (U.S.) Environmental Protection Agency (EPA) initiated consultation with NMFS on its proposal to authorize use, pursuant to the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA), 7 U.S.C. 136 *et seq.*, of pesticide products containing the active ingredients (a.i.s) of 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil from August 1, 2003 through December 1, 2004. EPA authorization of pesticide uses are categorized as FIFRA sections 3 (new product registrations), 4 (reregistrations and special review), 18 (emergency use), or 24(c) [Special Local Needs (SLN)]. At that time, EPA determined that uses of pesticide products containing these ingredients “may affect” most of the 26 Evolutionarily Significant Units (ESUs) of Pacific salmonids listed as endangered or threatened and designated critical habitat for the ESUs. This document represents NMFS’ biological opinion (Opinion) on the impacts of EPA’s authorization of pesticide products containing the above-mentioned a.i.s on the listed ESUs, plus on two newly listed salmonids. This is a partial consultation because pursuant to the court’s order, EPA sought consultation on only this group of listed species under NMFS’ jurisdiction. However, even though the court’s order did not address the two more recently listed salmonids, NMFS analyzed the impacts of EPA’s action to them because they belong to the same taxon. NMFS analysis requires consideration of the same information. Consultation with NMFS will be completed when EPA makes effect determinations on all remaining species and consults with NMFS as necessary.

This Opinion is prepared in accordance with section 7(a)(2) of the ESA and implementing regulations at 50 CFR §402. However, consistent with the decision in Gifford Pinchot Task Force v. USFWS, 378 F.3d 1059 (Ninth Cir. 2004), we did not apply the regulatory definition of “destruction or adverse modification of critical habitat”

at 50 CFR §402.02. Instead, we relied on the statutory provisions of the ESA to complete our analysis of the effects of the action on designated critical habitat.

This Opinion is based on NMFS' review of the package of information the EPA submitted with its 2003 and 2004 requests for formal consultation on the proposed authorization of the above a.i.s. It also includes our review of recovery plans for listed Pacific salmonids, past and current research and population dynamics modeling efforts, monitoring reports from prior research, Opinions on similar research, published and unpublished scientific information on the biology and ecology of threatened and endangered salmonids in the action area, and other sources of information gathered and evaluated during the consultation on the proposed authorization of a.i.s for 2,4-D, triclopyr BEE, diuron, linuron, captan and chlorothalonil. NMFS also considered information and comments provided by EPA and by the registrants identified as applicants by EPA.

Background

On January 30, 2001, the Washington Toxics Coalition, Northwest Coalition for Alternatives to Pesticides, Pacific Coast Federation of Fishermen's Associations, and Institute for Fisheries Resources filed a lawsuit against EPA in the U.S. District Court for the Western District of Washington, Civ. No. 01-132. This lawsuit alleged that EPA violated section 7(a)(2) of the ESA by failing to consult on the effects to 26 ESUs of listed Pacific salmonids of its continuing approval of 54 pesticide a.i.s.

On July 2, 2002, the court ruled that EPA had violated ESA section 7(a)(2) and ordered EPA to initiate interagency consultation and make determinations regarding effects to the salmonids on all 54 a.i.s by December 2004. *Washington Toxics Coalition v. EPA*, C01-132C (W.D. Wash. 7/2/2002).

In December 2002, EPA and the USFWS and NMFS (referred to as the Services) began interagency discussions for streamlining EPA's court ordered consultations.

On January 24, 2003, EPA and the Services published an Advance Notice of Proposed Rulemaking seeking public comment on improving the process by which EPA and the Services work together to protect listed species and critical habitat (68 FR 3785).

Between May and December 2003, EPA and the Services reviewed EPA's ecological risk assessment methodology and earlier drafts of EPA's "Overview of the Ecological Risk Assessment Process in the Office of Pesticide Programs, U.S. Environmental Protection Agency (Overview Document)". EPA and the Services also developed counterpart regulations to streamline the consultation process.

On January 22, 2004, the court enjoined application of pesticides within 20 (for ground) and 100 (for aerial) feet (ft) of streams supporting salmon. *Washington Toxics Coalition v. EPA*, C01-132C (W.D. Wash. 1/22/2004). The court imposed several additional restrictions on pesticide use in specific settings.

On January 23, 2004, EPA finalized its Overview Document which specified EPA's conduct of ecological risk assessment on pesticide registrations.

On January 26, 2004, the Services approved EPA's procedures and methods for conducting ecological risk assessments and approved interagency counterpart regulations for EPA's pesticide registration program.

On January 30, 2004, the Services published in the Federal Register (69 FR 4465) proposed joint counterpart regulations for consultation under the ESA for regulatory actions under the FIFRA, codified at 50 C.F.R. Part 402 Subpart D.

On August 5, 2004, the Services promulgated final joint counterpart regulations for EPA's ESA-related actions taken pursuant to FIFRA. These regulations and the Alternative Conservation Agreement (ACA) under the regulations allowed EPA to conduct independent analyses of potential impacts of pesticide registration on listed species and their designated critical habitats. The ACA outlined procedures to ensure

EPA's risk assessment approach will produce effect determinations that reliably assess the effects of pesticides on listed species and designated critical habitat. Additionally, EPA and the Services agreed to meet annually, or more frequently as may be deemed appropriate. The intention of these meetings was to identify new research and other activities that may improve EPA's current approach for assessing the potential ecological risks posed by use of a pesticide to listed species or designated critical habitat.

On September 23, 2004, the Washington Toxics Coalition and others challenged the counterpart regulations in the U.S. District Court for the Western District of Washington, Civ. No. 04-1998, alleging that the regulations were not authorized by the ESA and that the Services had not complied with the Administrative Procedure Act and the National Environmental Policy Act (NEPA) in promulgating these counterpart regulations.

In January 2006, EPA and the Services developed a draft joint interagency research agenda to address several critical areas of scientific and procedural uncertainties in EPA's current effects determination process. The jointly developed document identified eight areas of risk assessment and research uncertainties.

On August 24, 2006, the court determined the Services did not implement NEPA procedures properly during their promulgation of the joint counterpart regulations for EPA actions under FIFRA. Additionally, the court determined that the "not likely to adversely affect" and emergency consultation provisions of the counterpart regulations were arbitrary and capricious and contrary to the substantive requirements of ESA section 7(a)(2). The court determined that EPA may conduct its own formal consultation with the Services' involvement. Washington Toxics Coalition v. Department of the Interior, 457 F.Supp. 2d 1158 (W.D.Wash. 2006).

On November 5, 2007, the Northwest Coalition for Alternatives to Pesticides and others filed a legal complaint in the U.S. District Court for the Western District of Washington, Civ. No. 07-1791, against NMFS for its unreasonable delay in completing the section 7 consultations for EPA's registration of 54 pesticide a.i.s.

On July 30, 2008, NMFS and the plaintiffs entered into a settlement agreement with the Northwest Coalition for Alternatives to Pesticides. NMFS agreed to complete consultation within four years on 37 a.i.s. (EPA had concluded that 17 of the 54 a.i.s at issue in the first litigation would not affect any listed salmonid species or any of their designated critical habitat, and so did not initiate consultation on those a.i.s.)

On November 18, 2008, NMFS issued its first Opinion for three organophosphates: chlorpyrifos, diazinon, and malathion.

On April 20, 2009, NMFS issued its second Opinion for three carbamates: carbaryl, carbofuran, and methomyl.

On August 31, 2010, NMFS issued its third Opinion. This third consultation evaluated 12 organophosphate insecticides: azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, methamidophos, methidathion, methyl parathion, naled, phorate, and phosmet.

The current consultation evaluates 4 herbicides: 2,4-D, triclopyr BEE, diuron and linuron; and 2 fungicides: captan and chlorothalonil. EPA consultations on pesticide products currently focus on their effects to listed Pacific salmonids. EPA consultations remain incomplete until all protected species under NMFS' jurisdiction are covered.

Consultation History

Between August 1, 2003, and December 1, 2004, the EPA transmitted letters to NMFS' Office of Protected Resources (OPR) requesting section 7(a)(2) consultation for the registration of the six a.i.'s: 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil and detailing their effects determinations on 26 ESUs of Pacific salmonids listed at that time (Puget Sound steelhead and Lower Columbia River coho were not evaluated). In the BE's, and summarized in Table __, EPA's Office of Pesticide Programs (OPP) determined that the use of 2,4-D may adversely affect 26 ESUs. EPA

determined that the continued use of triclopyr BEE may adversely affect 16 ESUs, and is not likely to adversely affect 10 ESUs. For Diuron, EPA determined its continued use may adversely affect 25 ESUs and is not likely to adversely affect Lake Ozette sockeye salmon. EPA determined the use of linuron will have no effect on 19 ESUs, and is not likely to adversely affect 7 ESUs. Considering the fungicides captan and chlorothalonil, EPA determined the continued use of captan will not affect 13 ESUs, is not likely to adversely affect 11 ESUs, but may adversely two ESUs. EPA determined that the continued use of chlorothalonil will have no affect on six ESUs, is not likely to adversely affect 11 ESUs, and may adversely affect nine ESUs

On June 28, 2005, NMFS listed the Lower Columbia River coho salmon ESU as endangered. Given this recent listing, EPA's 2003 and 2004 effects determinations for 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil on listed Pacific salmonids lack an effects determination for the Lower Columbia River coho salmon.

On May 22, 2007, NMFS listed the Puget Sound Steelhead Distinct Population Segment (DPS) as threatened. Given this recent listing, EPA's 2003 and 2004 effects determinations for 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil on listed Pacific salmonids lack an effects determination for the Puget Sound steelhead.

On December 10-12, 2007, EPA and the Services met and discussed approaches for moving forward with ESA consultations and pesticide registrations. The agencies agreed to develop methodologies for filling existing data gaps. In the interim, the Services will develop approaches within their Opinions to address these gaps. The agencies identified communication and coordination mechanisms to address technical and policy issues and procedures for conflict resolution.

On August 20, 2008, NMFS met with EPA and requested EPA to identify applicants for this and subsequent pesticide consultations.

On August 29, 2008, NMFS met with EPA and the applicants for chlorpyrifos, diazinon, and malathion. At that meeting, NMFS asked EPA to identify applicants for this and subsequent pesticide consultations.

On September 17, 2008, NMFS requested EPA approval of Confidential Business Information (CBI) clearance for certain staff members in accordance with FIFRA regulations and access to EPA's incident database so NMFS staff may evaluate CBI materials from the applicants and incident reports for the a.i.s under consultation. EPA conveyed to NMFS that no access to the incident database would be authorized and the reports will be sent directly from EPA to NMFS.

On September 23, 2008, NMFS staff received notification of CBI clearance from EPA.

On September 26, 2008, NMFS sent correspondence to EPA regarding the roles of the federal action agency and identified applicants by such agency during formal consultation. NMFS also requested incident reports and label information for subsequent pesticide consultations from EPA. The specified timeline for NMFS' receipt of incident reports and label information for the six a.i.s considered in this Opinion was July, 2010.

From September 23, 2009 through November 5, 2009, NMFS staff completed their renewal of CBI status.

January 4, 2010, NMFS mailed out to EPA computer discs containing electronic files of the salmonid life cycle population models developed by NMFS' Northwest Fishery Science Center.

.

On January 13, 2010, NMFS requested from EPA for a copy of reports associated with Cramer Fish Sciences presentation provided at the 2008 American Chemical Society meeting.

On January 19, 2010, NMFS provided computer disks containing electronic files on the salmonid population models to EPA.

On June 1, 2010, NMFS sent a confirming email to EPA that all current labels for end use products, or if available, a master label that includes all use instructions for all products including 2,4-D, triclopyr BEE, diuron, linuron, captan, or chlorothalonil should be submitted to NMFS in July 2010.

On July 13, 2010, EPA sent a letter to registrants to confirm applicant status and participation in the consultation process for the Opinion covering the herbicides 2,4-D , triclopyr BEE, , diuron and linuron; and the fungicides captan and chlorothalonil.

On July 20, 2010, Syngenta (representing GB Biosciences) via email responded to the July 16, 2010 letter from EPA confirming GB Biosciences were certified registrants of chlorothalonil (EPA Reg. No. 50534-7 for technical material), and confirming GB Biosciences (Syngenta) wanted to be considered an applicant and thus participate in the consultation process.

On July 23, 2010, NMFS received grower-provided use information data from the Washington State Department of Agriculture (supplied by the USDA's National Agricultural Statistics Service (NASS) on the known use of 2,4-D, diuron, linuron, captan, and chlorothalonil in Washington State during the 2009 growing season for a few commodities.

On July 27, 2010, Syngenta forwarded the July 20, 2010 email noted above to NMFS to verify with NMFS that Syngenta were involved as an applicant for the consultation.

On July 30, 2010, NMFS received notification from Tessenderlo Kerley, Inc. (in response to the July 13, 2010 letter sent by EPA, noted above) that they were registrants for linuron and desired applicant status for the consultation with EPA for this active ingredient.

On August 2, 2010, NMFS received notification from Dow AgroSciences LLC (in response to the July 13, 2010 letter sent by EPA, noted above) that they were registrants for triclopyr BEE and desired applicant status for the consultation with EPA for this active ingredient.

On August 2, 2010, NMFS received information from Syngenta regarding the fungicide chlorothalonil.

On August 3, 2010, NMFS received notifications from Dow AgroSciences LLC, PBI-Gordon Corporation, Atanor S.A., and AgroGor Corporation (in response to the July 13, 2010 letter sent by EPA, noted above) that they were registrants for 2,4-D and desired applicant status for the consultation with EPA for this active ingredient.

On August 3, 2010, NMFS received notifications from Arysta Lifescience North America and Mahkteshim Chemical LTD (in response to the July 13, 2010 letter sent by EPA, noted above) that they were registrants for captan and desired applicant status for the consultation with EPA for this active ingredient.

On August 4, 2010, In a letter dated August 3, 2010, NMFS received notification from Albaugh, Inc (in response to the July 13, 2010 letter sent by EPA, noted above) that they were registrants for 2,4-D and desired applicant status for the consultation with EPA for this active ingredient.

On August 4, 2010, NMFS received notification from NuFarm (in response to the July 13, 2010 letter sent by EPA, noted above) that they were registrants for both triclopyr BEE and 2,4-D, and desired applicant status for the consultation with EPA for these active ingredients.

On August 9, 2010, via email, NMFS asked EPA to clarify information related to the Red-legged frog (RLF) biological assessment for 2,4-D. In that BA, EPA used a master

label approach. NMFS wanted confirmation this approach is accurate and reflects what is in the individual labels. If this was the case, NMFS could get started more quickly on the analysis for the 4th Opinion. NMFS also asked EPA to confirm if there are or are not registrants whose labels do not conform to the master label used in the RLF BA. EPA responded on the same day via email with an attached file of the most recent master label for 2,4-D (dated June 20, 2005). EPA indicated the products will reflect the master label by September 30, 2010. In addition EPA indicated they would send ten special 24(c) labels on a CD sent in the mail. The master label and the 24(c) labels package of product labels would be complete for 2,4-D.

On August 10, 2010, In a letter dated August 5, 2010, NMFS received notification from Albaugh, Inc (in response to the July 13, 2010 letter sent by EPA, noted above) that they were registrants for triclopyr BEE and desired applicant status for the consultation with EPA for this active ingredient.

On August 11, 2010, NMFS confirmed with EPA that Syngenta had requested a meeting as applicants for the purposes of the consultation process for this Opinion. NMFS expressed a desire to hold all applicant meetings early in September, 2010.

On August 11, 2010, NMFS received via email the 24(c) label for 2, 4-D registered in California, Oregon and Washington. NMFS asked for the Section 3 labels associated with the 24(c) labels and additional information concerning labels CA040027 and OR-10016 (this label was shown as expired on 12/31/2009). NMFS wanted to know if it had or was going to be renewed (NMFS received clarification on this issue from EPA on September 16, 2010 via email).

On August 11, 2010, NMFS received notice from EPA via email that two compact discs were sent; and with the email transmittal, two special local needs labels for captan were attached. NMFS was still awaiting label information from EPA on the other active ingredients being considered for this Opinion.

On August 12, 2010, NMFS received notification from DuPont De Nemours and Co. (in response to the July 13, 2010 letter sent by EPA, noted above) that they were registrants for diuron and desired applicant status for the consultation with EPA for this active ingredient.

On August 18, 2010, via email EPA asked NMFS how they would like to proceed with the applicant meetings. For chemicals with more than one applicant NMFS was asked if we wanted to ask the applicants if they are willing to meet together with NFS as opposed to meeting individually with NMFS. On August 20, 2010, NMFS responded by email to EPA that NMFS would prefer to consolidate the meetings to the extent possible. EPA responded later this same day that they would try to proceed in the manner NMFS preferred.

On August 19, 2010, NMFS received via email from EPA most recent stamped (approved) linuron product labels and a list of the products giving the registration number, product and company name, percent active ingredient, and label stamp date. EPA informed NMFS with this email that they were still assembling the labels for diuron and chlorothalonil.

On August 19, 2010, via a separate email from EPA, NMFS was notified that EPA had received word from GB Biosciences (Syngenta) on when they can meet on chlorothalonil. EPA also informed NMFS they were waiting on Dupont's reply regarding a meeting to discuss diuron.

On August 20, 2010, NMFS received via email notice from the Chemical Review Manager for triclopyr in EPA's Office of Pesticide Program's Pesticide Re-evaluation Division that they were attempting to set up applicant meetings between NMFS, OPP, and applicants Albaugh Inc. and Nufarm Americas Inc.

On August 25, 2010, EPA confirmed via email to NMFS the applicant meeting to discuss triclopyr was set for September 23, 2010.

On August 25, 2010, NMFS received letters from NuFarm and Dow indicating that as applicants for the consultation, their initial submission of information and data is being transmitted under a joint effort with several applicants through the “Industry Task Force II on 2,4-D Research Data.” A third letter was also received on this day from the 2,4-D Task Force submitting data for the consultation. In this letter they also asked for a joint meeting with EPA and NMFS.

On August 26, 2010, NMFS confirmed via email with EPA that the initial meetings with the applicants involved in the consultation Opinion were to introduce the applicants to the ESA Section 7(a)(2) consultation process, and to describe to them their role in the process. In this email, NMFS requested EPA that the applicants provide materials explaining unique application methods or uses for their chemicals if applicable. Also, NMFS informed EPA that several applicant letters were received regarding the meetings, including the 2,4-D Task Force (this letter was attached to the email to EPA), who proposed a joint meeting with the triclopyr applicants. NMFS confirmed that NMFS was amenable to this proposal.

On August 30, 2010, NMFS received two packages via Federal Express. NuFarm sent hard copies of two triclopyr BEE labels. NuFarm suggested these labels should be used in lieu of labels dating back to 2004. The second package was from AGRO-GOR and PBI/GORDON. Each sent identical letters dated August 25, 2010 (in same FedEx envelope) that their initial submissions of information are being transmitted through the 2,4-D Task Force

On August 31, 2010, NMFS received notification via email that E.I. duPont de Nemours and Company (“DuPont”) wanted to be included as applicants and agreed to submit information to EPA and to NMFS for consideration during consultation in the development of this Opinion. This email from DuPont was in reaction to EPA’s letter dated July 13, 2010, asking if DuPont wished to participate in the consultation process. This email included 13 attachments.

On September 1, 2010, NMFS received a background report on chlorothalonil from Syngenta in advance of the September 22, 2010, applicant meeting. This report was forwarded to NMFS via email from EPA.

On September 3, 2010, NMFS received the diuron labels from EPA via UPS.

On September 7, 2010, NMFS received an email from EPA stating that EPA is still checking on the chlorothalonil labels and will get those to us in the following week.

On September 7, 2010, NMFS received additional background materials from Syngenta via email, on chlorothalonil, for review prior to the September 22, 2010 applicant meeting. Although unnecessary, because EPA was providing all the labels, Syngenta also sent labels for our review for the consultation process for this Opinion.

September 16, 2010, NMFS received requested information on 2,4-D labels via email from EPA per August 11, 2010, request noted above.

On September 17, 2010, via email NMFS sent a two-page request to EPA to clarify linuron labels and uses. NMFS was later copied on an EPA email passing the questions on internally for response.

On September 17, 2010, NMFS received from EPA via email background information for linuron from Tessenderlo Kerley, Inc (TKI), an applicant.

On September 21, 2010, NMFS, EPA and applicant TKI met and shared information for this consultation. At this meeting NMFS explained the consultation process and the role of applicants in this process. TKI provided materials on linuron to EPA and NMFS for consideration in the consultation, and the development of the Opinion.

On September 22, 2010, NMFS, EPA and applicants BG Biosciences/Syngenta and Drexel met to discuss the fungicides captan and chlorothalonil. At this meeting NMFS explained the consultation process and the role of applicants in this process. The applicants provided materials on captan and chlorothalonil to EPA and NMFS for consideration in the consultation, and the development of the Opinion.

On September 22, 2010, NMFS received via email from EPA, two files on captan referenced in the applicant meeting earlier in the day by the Captan Task Force.

On September 23, 2010, NMFS met with EPA and applicants: 2,4-D Task Force, DOW, Nufarm, Albaugh Inc. to discuss the herbicides 2,4-D and triclopyr BEE. Earlier in the day, NMFS received via email an advanced copy of DOW's presentations on 2,4-D and triclopyr BEE. At this meeting NMFS explained the consultation process and the role of applicants in this process. The applicants provided materials on 2,4-D and triclopyr BEE to EPA and NMFS for consideration in the consultation, and the development of the Opinion.

On September 24, 2010, NMFS provided EPA via email the presentation NMFS gave at each of the applicant meetings held earlier in the same week.

On September 29, 2010, NMFS, EPA and DuPont (applicant for the herbicide diuron) met and shared information for this consultation. Earlier in the day, NMFS received via email from EPA advanced copies of DuPont's power-point presentations on diuron. At this meeting NMFS explained the consultation process and the role of applicants in this process. The applicant provided materials on diuron to EPA and NMFS for consideration in the consultation, and the development of the Opinion.

On September 29, 2010, NMFS received via email from EPA electronic versions of hand-outs provided at the September 23rd meeting for those who could only attend via phone.

On September 29, 2010, NMFS received contact information via email from EPA for the PMRA drift model and how EPA's Office of Pesticide Program's uses it.

On October 1, 2010, via email NMFS requested additional information from EPA on 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil. In particular, NMFS highlighted the need to consider the potential impact of all stressors associated with the federal action to listed species and their designated critical habitat. NMFS asked for additional information on "inert" and "other" ingredients approved for use in end-use pesticide products known to be toxic to aquatic organisms. NMFS explained to EPA that inert and other ingredients are considered as potential stressors and are part of the action that NMFS must evaluate. To date NMFS had not received complete composition information (list of all ingredients and percentage of formulation) for end-use products EPA is proposing to authorize under FIFRA. NMFS reminded EPA that several of the staff involved in the consultation are annually recertified to receive classified business information (CBI). NMFS therefore requested EPA to provide NMFS with complete composition information for all of the end-use products which contain 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil to adequately complete the consultation. NMFS requested this information be received by October 31, 2010.

On October 8, 2010, NMFS received via email from EPA two documents on triclopyr BEE that were referenced during the applicant meeting on September 23, 2010.

On October 11, 2010, NMFS received via email from a representative of the Captan Task Force, a power point presentation given at the applicant meeting held with EPA and NMFS on September 22, 2010.

On October 13, 2010, the Department of Justice (DOJ), representing NMFS, submitted a request to plaintiffs for a 90-day extension to complete the Opinion. The deadline extension would allow for NMFS to complete the final by April 30 instead of January 31 as required by the settlement agreement. DOJ also inquired as whether one Opinion could cover all of the pesticides instead of completing three separate Opinions.

On October 13, 2010, NMFS and EPA received from Dow AgroSciences LLC, three files of data collected from California Pesticide Use Reporting Information on 2,4-D and triclopyr BEE.

On October 13, 2010, NMFS received via email a power point and notes from Syngenta from the September 22, 2010, applicant meeting noted above.

On October 18, 2010, NMFS requested additional information from EPA, via email, where a few more 24C labels were missing the corresponding section 3 label.

On October 20, 2010, NMFS received requested information on missing section 3 labels requested on October 18, 2010.

On October 25, 2010, plaintiffs respond to DOJ approving the October 13, 2010 request for a 90-day extension, and allowance to cover all six pesticides in one Opinion.

On October 26, 2010, NMFS notified EPA via email plaintiffs have agreed for a 90-day extension for the Opinion and agree to flexible approach to batching the chemicals into one Opinion. NMFS noted that the 90-day extension had not yet been approved by the Court.

On October 27, 2010, NMFS received via email information from Washington Department of Ecology on the use of 2,4-D in Washington State for aquatic weed control.

On October 29, 2010, U. S. District Court affirmed the 90-day extension to complete the Opinion, and allowed flexibility in the number of Opinions NMFS chooses to complete for the batch of six chemicals under consultation.

On November 1, 2010, NMFS requested via email additional information from EPA on one of the chlorothalonil labels that appeared to have conflicting information between

mixture ratios and use ratios. This needed to be cleared up in order to determine maximum seasonal use rates associated with the products in question.

On November 1, 2010, NMFS received additional information via email from the Washington State Department of Ecology on 2,4-D aquatic applications for Washington.

On November 5, 2010, NMFS received a response and clarifying information via email from EPA on questions raised in NMFS' request for additional information on November 1, 2010, noted above.

On November 11, 2010, NMFS received various state restrictions on pesticide use compiled by DuPont for the diuron consultation.

On November 16, 2010, NMFS requested via email additional information from EPA on two captan labels (CA-020017 and WA-940026).

On November 18, 2010, NMFS received a response from EPA via email regarding additional information on one of the two captan labels. EPA stated that information on the other label (WA-940026) will be sent soon.

On November 18, 2010, NMFS requested an additional 24(c) label that was not provided by EPA (EPA Reg. No. 51036-166).

On November 19, 2010, NMFS receives a response to the November 18, request noted above from EPA. NMFS was informed the CA SLN references an old registration number. EPA Reg. No. 51036-166 was transferred to Arysta in 2006 -- thus the registration number for the product changed to EPA Reg. No. 6630-234. EPA provided this label information as an attached file. EPA informed NMFS that they were checking to see if the CA SLN is still active and would let NMFS know of its status within the next few days.

On November 29, 2010, NMFS received via email from DuPont a report sent previously to EPA for “86-5 compliance” regarding consultation on diuron.

On December 6, 2010, NMFS phoned EPA to get an update on when NMFS might expect responses to questions regarding linuron label statements and uses first requested on September 17, 2010.

On December 12, 2010, NMFS received via FedEx a CD from applicant Syngenta additional new data and information for the consultation on chlorothalonil.

On December 13, 2010, NMFS received via email from EPA responses to questions about linuron labels, along with additional labels not previously provided. This information is important in understanding the scope of the proposed action and in determining any possible effects of the action to listed salmon and steelhead and their designated critical habitat.

In response to the information provided by EPA on December 13, 2010, NMFS sought additional clarifying information on linuron in an email request sent on December 15, 2010.

On December 21, 2010, NMFS received an email from EPA asking if NMFS had received a CD containing additional information on chlorothalonil from Syngenta. NMFS responded on this same day stating that the CD had arrived along with a transmittal letter dated December 15, 2010.

On December 21, 2010, NMFS sent an email to EPA requesting additional chemical fate information regarding captan.

On January 3, 2011, NMFS received two email responses from EPA, with numerous attachments, to the questions and information requests sent on December 21, 2010.

On January 4, 2011, NMFS received an email from EPA inquiring about the availability of a draft Opinion. EPA was interested in scheduling meetings with the applicants to discuss the draft.

On February 14, 2011, NMFS received via FedEx a letter and CD from Syngenta Crop Protection, LLC. Syngenta provided additional information to EPA and NMFS on non-crop uses of chlorothalonil.

On February 27, 2011, NMFS received via email additional information from Syngenta for NMFS and EPA to consider in our consultation. The information pertained to a drinking water assessment for the IR-4 registration of chlorothalonil and its specific degradation product for new uses on bulb vegetables, bushberries, and low growing berries.

On March 1, 2011, NMFS delivered via weblink a draft of this Opinion with transmittal letter covering the 6 a.i.s, 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil.

[MORE to be added as consultation history builds]

Species Addressed in the BEs

EPA's BEs considered the effects of pesticides containing the six a.i.s to 26 species of listed Pacific salmonids and their designated critical habitat ((EPA 2004), (EPA 2004), (EPA 2003), (EPA 2004), (EPA 2003), (EPA 2003). Two listed species, the Lower Columbia River coho and the Puget Sound steelhead, were not considered in the BEs. Although EPA has determined that its action of registering pesticide products containing some of the a.i.s will have no effect on some endangered or threatened Pacific salmon and steelhead, it also determined that some of the a.i.s may affect certain ESUs/DPSs (Table 1). With regard to linuron, EPA determined "no effect" or "NLAA" for all of the ESUs/DPSs it evaluated. With regard to captan and chlorothalonil, EPA determined no effect or NLAA for many of the ESUs/DPSs it evaluated. NMFS does not concur with some of these determinations. When an action agency concludes its action will not affect

any listed species or critical habitat, then no section 7 consultation is necessary (USFWS, & NMFS 1998). However, consultation on the proposed action is necessary because EPA concluded for five of the a.i.s that the proposed action may affect some listed Pacific anadromous salmonids and their designated critical habitat, and NMFS did not concur with any of EPA's "NLAA" determinations. Once NMFS enters into formal consultation it considers all species and critical habitat affected. In this Opinion, NMFS will analyze the impacts to all ESUs/DPSs of Pacific salmonids present in the action area, including those salmonid species identified by EPA as being unaffected and including the two species of salmonid listed after EPA provided its BEs to NMFS.

Table 1. EPA's effects determinations.

Species	ESU	Herbicides				Fungicides	
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chlorothalonil
Chinook	Puget Sound	may affect	may affect	may affect	no effect	NLAA	may affect
	Lower Columbia River	may affect	may affect	may affect	no effect	NLAA	NLAA
	Upper Columbia River Spring - Run	may affect	may affect	may affect	NLAA	may affect	may affect
	Snake River Fall - Run	may affect	may affect	may affect	NLAA	NLAA	may affect
	Snake River Spring/Summer - Run	may affect	may affect	may affect	NLAA	NLAA	may affect
	Upper Willamette River	may affect	may affect	may affect	no effect	NLAA	may affect
	California Coastal	may affect	NLAA	may affect	no effect	no effect	NLAA
	Central Valley Spring - Run	may affect	NLAA	may affect	no effect	no effect	NLAA
	Sacramento River Winter - Run	may affect	NLAA	may affect	no effect	no effect	NLAA
Chum	Hood Canal Summer - Run	may affect	may affect	may affect	no effect	no effect	no effect
	Columbia River	may affect	may affect	may affect	no effect	no effect	no effect
Coho	Lower Columbia River	<i>not evaluated</i>	<i>not evaluated</i>	<i>not evaluated</i>	<i>not evaluated</i>	<i>not evaluated</i>	<i>not evaluated</i>
	Oregon Coast	may affect	may affect	may affect	no effect	no effect	no effect
	Southern Oregon and Northern California Coast	may affect	NLAA	may affect	no effect	no effect	NLAA
	Central California Coast	may affect	NLAA	may affect	no effect	no effect	NLAA
Sockeye	Ozette Lake	may affect	may affect	NLAA	no effect	no effect	no effect
	Snake River	may affect	may affect	no effect	no effect	may affect	no effect
Steelhead	Puget Sound	<i>not evaluated</i>	<i>not evaluated</i>	<i>not evaluated</i>	<i>not evaluated</i>	<i>not evaluated</i>	<i>not evaluated</i>
	Lower Columbia River	may affect	may affect	may affect	no effect	no effect	NLAA
	Upper Willamette River	may affect	may affect	may affect	no effect	NLAA	may affect
	Middle Columbia River	may affect	may affect	may affect	NLAA	NLAA	may affect
	Upper Columbia River	may affect	may affect	may affect	NLAA	NLAA	may affect
	Snake River	may affect	may affect	may affect	no effect	NLAA	may affect
	Northern California	may affect	NLAA	may affect	no effect	no effect	no effect
	Central California Coast	may affect	NLAA	may affect	no effect	no effect	NLAA
	California Central Valley	may affect	NLAA	may affect	no effect	no effect	NLAA
	South-Central California Coast	may affect	NLAA	may affect	NLAA	NLAA	NLAA
	Southern California	may affect	NLAA	may affect	NLAA	NLAA	NLAA

Description of the Proposed Action

The Federal Action

The proposed action encompasses EPA's registration of the uses (as described by product labels) of all pesticides containing 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil. The purpose of the proposed action is to provide tools for pest control that do not cause unreasonable adverse effects to the environment throughout the U.S. and its affiliated territories. Pursuant to FIFRA, before a pesticide product may be sold or distributed in the U.S. it must be exempted or registered with a label identifying approved uses by EPA's OPP. Once registered, a pesticide may not legally be used unless the use is consistent with directions on its approved label (<http://www.epa.gov/pesticides/regulating/registering/index.htm>). EPA authorization of pesticide uses are categorized as FIFRA sections 3 (new product registrations), 4 (reregistrations and special review), 18 (emergency use), or 24(c) Special Local Needs (SLN).

EPA's pesticide registration process involves an examination of the ingredients of a pesticide, the site or crop on which it will be used, the amount, frequency and timing of its use, and its storage and disposal practices. Pesticide products may include a.i.s and other ingredients, such as adjuvants, and surfactants (described in greater detail below). The EPA evaluates the pesticide to ensure that it will not have unreasonable adverse effects on humans, the environment, and non-target species. An unreasonable adverse effect on the environment is defined in FIFRA as, "(1) any unreasonable risk to man or the environment, taking into account the economic, social, and environmental costs and benefits of the use of the pesticide, or (2) a human dietary risk from residues that result from a use of a pesticide in or on any food inconsistent with the standard under section 408 of the FFDCa (21 U.S.C. §346a)" 7 U.S.C. 136(b).

After registering a pesticide, EPA retains discretionary involvement and control over such registration. EPA must periodically review the registration to ensure compliance with FIFRA and other federal laws (7 U.S.C. §136d). A pesticide registration can be canceled whenever “a pesticide or its labeling or other material...does not comply with the provisions of FIFRA or, when used in accordance with widespread and commonly recognized practice, generally causes unreasonable adverse effects on the environment.”

On December 12, 2007, EPA, NMFS, and FWS agreed that **the federal action for EPA’s FIFRA registration actions will be defined as the “authorization for use or uses described in labeling of a pesticide product containing a particular pesticide ingredient.”** In order to ensure that EPA’s action will not jeopardize listed species or destroy or adversely modify critical habitat, NMFS’ analysis encompasses the impacts to listed Pacific salmonid ESUs/DPSs of all uses authorized by EPA.

Pesticide Labels. For this consultation, EPA’s proposed action encompasses all approved product labels containing the a.i.s 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil; their degradates, metabolites, and formulations, including other ingredients within the formulations; adjuvants; and tank mixtures. These activities comprise the stressors of the action (Figure 1). The six BEs indicate that the subject a.i.s are labeled for a variety of uses including applications to residential areas, industrial areas, pastures, forested areas, and crop lands (EPA 2003; EPA 2003; EPA 2003; EPA 2004; EPA 2004; EPA 2004).

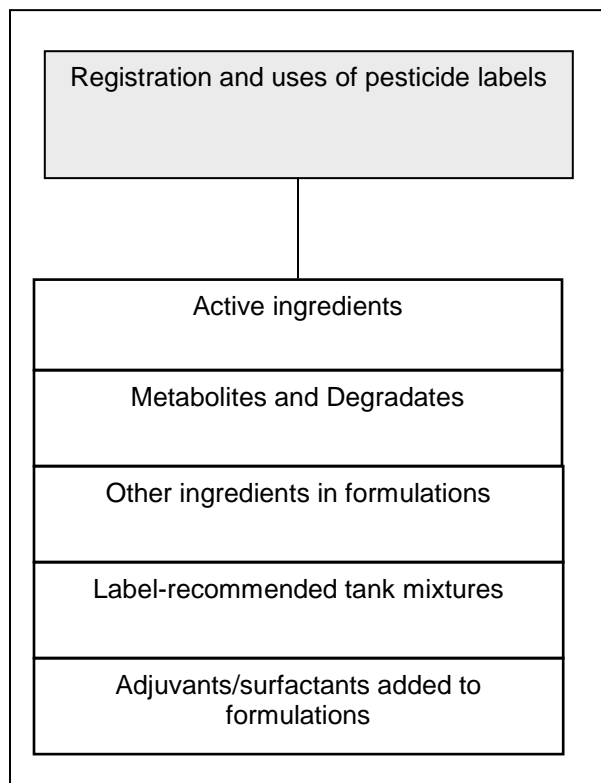


Figure 1. Stressors of the Action

Active and Other Ingredients. 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil are the a.i.s that kill or otherwise affect targeted organisms (listed on the label). However, pesticide products that contain these a.i.s also contain inert ingredients. Inert ingredients are ingredients which EPA defines as not “pesticidally” active. EPA also refers to inert ingredients as “other ingredients”. The specific identification of the compounds that make up the inert fraction of a pesticide is not required on the label. However, this does not necessarily imply that inert ingredients are non-toxic, non-flammable, or otherwise non-reactive. EPA authorizes the use of chemical adjuvants to make pesticide products more efficacious. An adjuvant aides the operation or improves the effectiveness of a pesticide. Examples include wetting agents, spreaders, emulsifiers, dispersing agents, solvents, solubilizers, stickers, and surfactants. A surfactant is a substance that reduces surface tension of a system, allowing oil-based and water-based substances to mix more readily. A common group of non-ionic surfactants is the alkylphenol polyethoxylates (APEs), which may be used in pesticides or pesticide tank

mixes, and also are used in many common household products. Nonylphenol (NP), one of the APEs, has been linked to endocrine-disrupting effects in aquatic animals.

Formulations. Pesticide products come in a variety of solid and liquid formulations. Examples of formulation types include dusts, dry flowables, emulsifiable concentrates, granulars, solutions, soluble powders, ultra-low volume concentrates, water-soluble bags, powders, and baits. The formulation type can have implications for product efficacy and exposure to humans and other non-target organisms.

Tank Mix. A tank mix is a combination by the user of two or more pesticide formulations as well as any adjuvants or surfactants added to the same tank prior to application. Typically, formulations are combined to reduce the number of spray operations or to obtain better pest control than if the individual products were applied alone. The compatibility section of a label may advise on tank mixes known to be incompatible or provide specific mixing instructions for use with compatible mixes. Labels may also recommend specific tank mixes. Pursuant to FIFRA, EPA has the discretion to prohibit tank mixtures. Applicators are permitted to include any combination of pesticides in a tank mix as long as each pesticide in the mixture is permitted for use on the application site and the label does not explicitly prohibit the mix.

Pesticide Registration. The Pesticide Registration Improvement Act (PRIA) of 2003 became effective on March 23, 2004. The PRIA directed EPA to complete REDs for pesticides with food uses/tolerances by August 3, 2006, and to complete REDs for all remaining non-food pesticides by October 3, 2008. The goal of the reregistration program is to mitigate risks associated with the use of older pesticides while preserving their benefits. Pesticides that meet today's scientific and regulatory standards may be declared "eligible" for reregistration. The eligibility for continued registration may be contingent on label modifications to mitigate risk and can include phase-out and cancellation of uses and pesticide products. The terms of EPA's regulatory decisions are summarized in RED documents (EPA 1995; EPA 1998; EPA 1999; EPA 2003; EPA 2004; EPA 2005).

Registrants can submit applications for the registration of new products and new uses following reregistration of an a.i. Several types of products are registered, including the pure (or nearly pure) active ingredient, often referred to as technical grade active ingredient (TGAI), technical, or technical product. This is generally used in manufacturing and testing, and not applied directly to crops or other use sites. Products that are applied to crops, either on their own or in conjunction with other products or surfactants in tank mixes are called end-use products (EUPs). Sometimes companies will also register the pesticide in a manufacturing formulation, intended for sale to another registrant who then includes it into a separately registered EUP. Manufacturing formulations are not intended for application directly to use sites. The EPA may also cancel product registrations. EPA typically allows the use of canceled products, and products that do not reflect RED label mitigation requirements, until those products have been exhausted. Labels that reflect current EPA mitigation requirements are referred to as “active labels.” Products that do not reflect current label requirements are referred to as “existing stocks.” EPA’s action includes all authorizations for use of pesticide products including use of existing stocks, and active labels, of products containing the six a.i.s for the duration of the proposed action.

Duration of the Proposed Action. EPA’s goal for reassessing currently registered pesticide a.i.s is every 15 years. Given EPA’s timeframe for pesticide registration reviews, NMFS’ evaluation of the proposed action is also 15 years.

Interrelated and Interdependent Activities. No interrelated and interdependent activities are associated with the proposed action.

Registration Information of Pesticide a.i.s under Consultation. As discussed above, the proposed action encompasses EPA’s registration of the uses (as described by product labels) of all pesticides containing 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil. EPA provided copies of all active product labels for triclopyr BEE, diuron, linuron, captan, and chlorothalonil; and a master label summarizing all label

restrictions for 2,4-D. The following descriptions represent information acquired from review of these labels as well as information conveyed in the EPA BEs, REDs, and other documents.

2,4-D

The herbicide 2,4-D has been used in the United States for more than 60 years (EPA 2005). It is most commonly used as a post-emergent herbicide for broadleaf control and is available in several chemical forms that are each formulated in multiple end-use products (Table 2) (EPA 2009). 2,4-D is a phenoxy herbicide that acts as a plant growth regulator. It's a synthetic auxin that disrupts normal plant growth by mimicking endogenous auxins that act as regulator hormones. Plant injuries include impacts to growth and reproduction. Symptoms may appear almost immediately in plants, but death may not occur for several weeks. Currently, Dow AgroSciences, Nufarm, Ltd., and the Agro-Gor Corporation have registrations with EPA to manufacturing use products containing 2,4-D. These products are formulated into a large number of end-use pesticides which are registered by dozens of companies and applied for a variety of uses (National Pesticide Information Retrieval System <http://ppis.ceris.purdue.edu/htbin/cnamlist.com>). In total, there are over 600 end-use products that are registered for use on over 300 distinct use sites (e.g. agricultural, residential, aquatic, etc., EPA RED DOCUMENT, 2005). Additionally, there are nine SLN registrations in California, Idaho, Oregon, and Washington (EPA Reg. No. CA040027, CA070017, CA970033, OR050016, OR-940036, WA010009, WA010038, WA070007, and WA9400032). There are no emergency use registrations (section 18) for 2,4-D in California, Idaho, Oregon, or Washington.

Table 2. Chemical forms of 2,4-D products that are currently registered for use in the action area.

EPA PC Code	CAS Number	Chemical Name
030001	94-75-7	2,4D acid
030004	2702-72-9	2,4D sodium salt
030016	5742-19-8	2,4D diethanolamine (DEA) salt
030019	2008-39-1	2,4D dimethylamine (DMA) salt
030025	5742-17-6	2,4D Isoproylamine (IPA) salt
030035	32341-80-3	2,4D triisopropanolamine (TIPA) salt
030053	1929-73-3	2,4D butoxyethyl ester (BEE)
030063	1928-43-4	2,4D 2 ethylhexyl ester (EHE)
030066	94-11-1	2,4D isopropyl ester (IPE)

Usage Information.

EPA estimated 36 million pounds of 2,4-D are applied annually in the United States for agricultural uses, with “heavy use” of 2,4-D in the Pacific Northwest and California (EPA 2004). Usage of 2,4-D in California has remained relatively stable, with over 400,000 lbs applied during each year from 1998 – 2008 (CDPR 2009). Washington State Department of Agriculture estimates total annual usage of 2,4-D for nine of registered agricultural use sites to be 183,630 – 469,843 lbs, based on recent crop patterns and use surveys (WSDA 2010). Recent usage information for Oregon and Idaho is not available.

Agricultural Uses. Cereal grains, field and pop corn, sweet corn, sorghum, soybeans, sugarcane, rice, pome fruits (e.g. apples, pears), stone fruits (e.g. cherries, peaches, plums, apricots), nut orchards, pistachios, filberts, pastures, rangeland, follow land and crop stubble, grass grown for seed or sod, irrigation ditch banks, potatoes, asparagus, hops, strawberries, blueberries, grapes, cranberries, citrus, clover, cottonwood and poplar trees grown for pulp, abandoned orchards, and forestry (site preparation, conifer release, roadsides, etc.).

Non-agricultural Uses. Grasslands not in agricultural production, ornamental turf, tree and brush control, non-cropland such as fencerows, hedgerows, roadsides, ditches, rights-

of-way, utility power lines, railroads, airports, industrial sites, and other non-crop areas, and aquatic uses to control floating/emergent aquatic weeds and submerged aquatic weeds (e.g. ponds, lakes, reservoirs, marshes, bayous, ditches, canals, slow moving rivers and streams).

Registered Formulation Types. 2,4-D products are formulated as emulsifiable concentrates, wettable powders, granules, soluble concentrate solids, soluble concentrate liquids, and water dispersible granules (dry flowables). 2,4-D products frequently contain 2 – 4 a.i.s. Other registered herbicides ingredients in currently registered 2,4-D products include atrazine, aminopyralid, bromoxynil, carfentrazone-ethyl, clopyralid, dicamba, fluroxypyr, fenoxaprop-p-ethyl, glyphosate, MCPA, MCPP, MSMA, picloram, pyraflufenethyl, quinclorac, sulfentrazone, and triclopyr.

Methods and Rates of Application.

Methods. 2,4-D can be applied using a variety of methods and equipment. It may be applied as a spot treatment or broadcast application using aircraft (fixed wing or helicopter), ground boom sprayers, granule spreaders, hand held nozzle sprayers, wick applicators, and stump injectors.

Application Rates. Application rates are limited to 1 – 2 lbs of 2,4-D /A on the majority of agricultural use sites (Table 3). Sites with the greatest application rates include forestry and several non-crop use sites that allow a maximum single application of 4 lbs 2,4-D /A. Additionally, up to 10.8 lbs of 2,4-D per acre-foot (4 parts per million) can be applied to aquatic habitats for control of submergent weeds. Multiple application are permitted on several use sites. Typically, either the maximum number of applications and/or maximum seasonal rate is specified. However, several of the SLN registrations do not specify limitations on either the number of applications or seasonal/annual use rates (CA-070017, CA-970033, OR-940036, WA-070007, and WA-940032).

Table 3. Summary of all authorized use sites and application restrictions for active 2,4-D.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Cereal grains	1.25	2/crop cycle	1.75/crop cycle	NS	Aerial, ground spray	5481-145
Field and Pop Corn	1.50	NS	3.00/season	NS	Aerial, ground spray, granule spreader	5481-145
Sweet Corn	1.00 preplant or preemergence, 0.50 post emergence	1/crop cycle	1.50/season	21	Aerial, ground spray, granule spreader	5481-145
Sorghum	1.00 for amines, acids, and salts; 0.50 for esters	1/crop cycle	1.00 for amines, acids, and salts; 0.50 for esters /season	NA	Aerial, ground spray, granule spreader	34704-120
Soybeans	1.00	NS	1.00/season	NS	Preplant aerial or ground spray	5481-145
Sugarcane	2.00 amines, acids, and salts	1/crop cycle	4.00/season	NS	Aerial, ground spray	5481-145
Rice	1.00 preplant, 1.50 post emergence amines, acids, and salts	1/crop cycle	1.50/season	NS	Aerial, ground spray	228-260
Pome fruits (e.g. apples, pears)	2.00 amines, acids, and salts	2	4.00/season	75	Post emergence Ground spray	5481-145
Stone fruits (e.g. cherries, peaches, plums, apricots)	2.00 amines, acids, and salts	2	4.00/season	75	Post emergence Ground spray	5481-145
Nut orchards, Pistachios	2.00 amines, acids, and salts	2	4.00/season	30	Post emergence Ground spray	228-260
Filberts	NS ³ amines, acids, and salts	4	NS	30	Post emergence Ground spray	34704-120
Pastures, rangeland, grasslands not in agricultural production	2.00	NS	4.00/season	30	Post emergence Aerial, ground spray	5481-145
Ornamental and residential turf	1.50	2	3.00/season	NS	Ground spray	42750-19, 961-394
Grass grown for seed or sod	2.00	NS	4.00/season	NS	Ground spray	5481-145
Follow land and crop stubble	2.00	NS	4.00/season	30	Aerial, ground spray	5481-145
Forestry:	4.00	NS	4.00 for	NS	Aerial,	5481-145

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
roadsides, site prep., conifer release			broadcast, NS for other		ground spray, tree injection	
Tree and brush control	4.00	NS	4.00 for broadcast, NS for other	NS	Aerial, ground spray, tree injection	2217-703
Non-cropland such as fencerows, hedgerows, roadsides, ditches, rights-of-way, utility power lines, railroads, airports, industrial sites, and other non-crop areas	4.00	NS	4.00/season	30	Aerial, ground spray	5481-145
Irrigation ditch banks	2.00	2	4.00/season	30	Aerial spray, boat spray- Allow no more than 2 feet overspray onto water.	5481-145
Floating/emergent aquatic weeds	4.00	2	NS	21	NS	5481-145
Submerged aquatic weeds (e.g. ponds, lakes, reservoirs, marshes, bayous, ditches, canals, slow moving rivers and streams)	10.8 lbs / Acre foot (4 parts per million)	2	NS	21	NS	5481-145
Potatoes	0.07	2	0.14/season	10-14	Post-emergent aerial or ground spray	228-139
Asparagus	2.00 amines, acids, and salts	2	4.00/season	30	NS	34704-120
Hops	0.50 amines, acids, and salts	3	1.50/season	30	Apply to ground between rows	34704-120
Strawberries (not in CA)	1.50 amines, acids, and salts	1	1.50/season	NA	Aerial, ground spray	34704-120
Low bush blueberries	NS ⁴ amines, acids,	1	NS	NA	Wipe and spot	Master label

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
	and salts				applications on weeds	
High bush blueberries	1.4 amines, acids, and salts	1/crop cycle	2.80/season	NS	Directed ground spray	Master label
Grapes (CA only)	1.36 amines, acids, and salts	1/crop cycle	NS	NA	Directed ground spray	228-260
Cranberries	4.00 granular ester 1.2 amines, acids, and salts	1 dormant season, 2 growing season	4.00 lbs in dormant season, 2.40 lbs in growing season	NS	Ground: granule spreader, spot spray, wipe	228-61
Citrus	0.1 Isopropyl ester to increase fruit size, NS ⁵ to prevent pre-harvest drop	1/crop stage	NS	NS	Aerial, ground spray	5481-145
24 (c) CA: Mandarin growth regulator	NS ²	1	NS	NA	Air blast, other ground spray	5481-145; CA040027
24 (c) CA: Ladino Clover (seed crop)	1.00	NS	NS	NS	Aerial, ground spray	34707-120; CA-070017
24 (c) CA: Citrus floor	1.64	NS, as needed	NS	NS	Directed ground spray-apply to the point of runoff	228-260; CA-970033
24 (c) OR: Blueberries	1.40	2	2.8	NS- Once in spring and once in fall	Ground spray	34704-803; OR-050016
24 (c) OR: Cottonwood and poplar trees for pulp	1.43	NS	NS	NS	Ground spray or wick application	228-145; OR-940036
24 (c) WA: Blueberries	1.40	2	2.8	NS- Once in spring and once after harvest	Ground spray	34704-803; WA-010009
24 (c) WA: Abandoned orchards	2 lb. triclopyr BEE /A + 4 lb 2,4-D	1	2 lb. triclopyr BEE /A + 4 lb 2,4-D	NA	Bark spray or hack and squirt trees	62719-260; WA-010038
24 (c) WA: Eurasian milfoil	As needed to achieve 1 – 4 mg/L maintained for 48 hrs	NS	NS	21 days for “broadcast” applications; NS for “spot treatments”	Subsurface drip application in slowly moving surface waters	62719-3; WA-070007

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
24 (c) WA: Cottonwood and poplar trees for pulp	1.43	NS	NS	NS	Ground spray or wick application	228-154; WA-940032
<ol style="list-style-type: none"> 1. NS = not specified 2. No upper limit placed on maximum application rate. Used as growth regulator not herbicide. Minimum 500 gal per acre of spray material per acre = 0.10 lbs a.i./A 3. Use rate per acre NS. Maximum rate of 1.00 lb a.i./100 gallons of spray solution. Wet leaves and stems of suckers April through August. 4. Use rate per acre NS. Maximum rate of 1.0 lb a.i./10 gallons of spray solution. 5. Use rate per acre NS. Maximum rate of 200 ppm a.i for spray solution. 						

Metabolites and Degradates.

Several degradates of parent 2,4-D have been identified in environmental fate studies including 1,2,4-benzenetriol; 2,4-dichlorophenol (2,4-DCP); 2,4 dichloroanisol (2,4-DCA); 4-chlorophenol; 2-chlorophenol; 4-chlorophenoxyacetic acid; and chlorohydroquinone (EPA 2009).

Triclopyr butoxyethyl ester (triclopyr BEE)

Triclopyr is a systemic broad-spectrum herbicide that is in the pyridinyloxyacetic acid family. It acts as a plant growth regulator and is used to control broadleaf weeds and woody plants. Triclopyr acid is formulated as a manufacturing product, and is used to formulate triclopyr BEE and triclopyr triethylamine salt (TEA) pesticides (EPA 2009). The current consultation is for approved uses of the Triclopyr BEE, and does not evaluate triclopyr TEA. Triclopyr TEA was first registered in 1979 for use on non-crop areas and forestry. Triclopyr BEE was registered in 1980 for use on the same sites. Both chemical forms of triclopyr were registered for use on turf in 1984. Triclopyr BEE is registered for use on rangeland and permanent grass pastures. Whereas, triclopyr TEA is registered for use on rice and BEE is not. Currently, Albaugh, Inc., Dow AgroSciences, Makhteshim Agan of North America, Inc., and NuFarm Americas, Inc. have registrations with EPA for manufacturing use products containing triclopyr. There are 31 active labels for end use products containing triclopyr BEE registered by 10 companies. Additionally, there is one SLN registration for control of unwanted trees in abandoned orchards (WA-010038). There are no section 18 registrations for use of triclopyr BEE products in California, Idaho, Oregon, or Washington.

Usage Information.

EPA estimates approximately 200,000 lbs of triclopyr BEE are applied within the action area each year (EPA 2004). California use reports indicate more than 70,000 lbs of triclopyr are applied annually in the state. However, the reports do not distinguish between use of triclopyr BEE and triclopyr TEA products (CDPR 2009). Recent usage information for Washington, Oregon and Idaho is not available.

Agricultural Uses. Uses include range and pasture treatments, ornamental turf (sod farms), non-crop agricultural areas such as abandoned orchards, around farm buildings, fence rows, roads, and non-irrigations ditch banks.

Non-agricultural Uses. Non-agricultural uses of triclopyr BEE include rights-of-way, forest management (site preparation and conifer release), and applications to golf course and residential turf, and industrial areas.

Registered Formulation Types. Triclopyr BEE enduse products are typically formulated as emulsifiable concentrates or ready to use liquids that are spray applied. There is also a granular fertilizer product that contains Triclopyr BEE (EPA Reg. No. 961-394). Triclopyr BEE is frequently formulated with other a.i.s. Several formulations contain 2,4-D. (*e.g.*, EPA Reg. No. 228-565, 961-394, and 34704-928). It is also formulated with fluroxypyr (EPA Reg. No. 62719-477). Two formulations include MCPA and dicamba (EPA Reg. No. 228-395, 228-317). One formulation partners triclopyr BEE with three other herbicides (sulfentrazone, 2-4,D, and dicamba; EPA Reg. No. 2217-920).

Methods and Rates of Application.

Methods. Triclopyr BEE is typically spray applied by ground application or aerial methods. A granular formulation is applied with ground spreader. Labels frequently authorize tank mixes with other herbicides (EPA Reg. No. 62719-527), liquid fertilizers (EPA Reg. No. 228-317), drift control agents (EPA Reg. No. 74779-8), and/or surfactants (EPA Reg. No. 66222-153).

Application Rates. Active labels allow a maximum single and seasonal application rate of up to 8 lbs triclopyr BEE/A to forests and several non-crop areas (Table 4). The number of applications allowed is 1, 8, or is not specified. Use sites without specifications for the number of applications limit the total amount of product that can be applied either annually or seasonally.

Table 4. Summary of all authorized use sites and application restrictions for active triclopyr BEE products.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number(s)
Range/pasture	2	1	2	NA	Aerial or ground spray	228-521; 228-552
Forests	6	1	8	NA	Aerial or ground spray	228-517
Turf (ornamental, commercial, golf course, residential)	1 (2 for spot treatments)	8	8	4 weeks	Aerial or ground spray	17545-8; 961-394; 62719-566; 66222-153
Non-Crop areas (e.g. fence rows, non-irrigation ditch banks, rights-of way, around farm buildings, industrial areas)	8	NS	8	NS	Aerial or ground spray	66222-153
Seasonably dry wetlands, flood plains, deltas, marshes, swamps, bogs and transitional areas between uplands and lowlands	8	NS	8	NS	Aerial or ground spray	66222-153
24 (c) WA: Abandoned orchards	2 lb. Triclopyr BEE;	1	2 lb. Triclopyr BEE; 4 lb. 2,4-D	NA	Ground spray	62719-260 WA010038

	4 lb. 2,4-D					
NS = not specified						

Metabolites and Degradates.

Triclopyr BEE and triclopyr TEA both rapidly degrade to for triclopyr acid. In aquatic environments, photodegradation products of the acid/anion include 5-chloro-3,6-dihydroxy-2-pyridinoloxycetic acid (TCP) and oxamic acid. In soils, TCP and 3,5,6-trichloro-2-methoxypyridine (TMP) are formed through biotic metabolism (EPA 2009).

Diuron

Diuron (N-(3,4-dichlorophenyl)-N,N-dimethylurea) is a systemic substituted phenylurea herbicide. Diuron acts by inhibiting the Hill reaction in photosynthesis which limits the production of high energy compounds such as ATP used for various metabolic processes. Diuron is primarily absorbed through plant roots. It is transported upward through the xylem, and exerts its action at the seedling stage when the newly emerged plant starts to photosynthesize. It is effective primarily on annual broadleaved weeds, annual grasses, or newly emerged perennial plants. Established perennial plants are less susceptible, which is the basis for its use in fruit and nut crops (EPA 2003). Twenty three companies currently hold active registrations for end-use or technical products that contain diuron (<http://ppis.ceris.purdue.edu/htbin/cnamlist.com>). There are currently 73 active labels for products containing diuron including 66 end use products and 7 technical/manufacturing use products. There are 11 SLN registrations in California, Idaho, Oregon, or Washington. There are no emergency use registrations for diuron in California, Idaho, Oregon, or Washington.

Usage Information.

EPA estimated approximately 8 million pounds of diuron are applied annually for domestic uses based on pesticide surveys for the years of 1990 through 1999 (EPA 2003). Slightly over half was used in non-agricultural areas; about 25% of diuron was used on

railroads; other non-agricultural sites of high usage (5-9% of total diuron) were pipelines and industrial facilities, roads, and sanitation/utilities. Among agricultural uses, the highest amounts of diuron were used on oranges (15%), cotton (10%), seed crops (9%), grapefruit (3%), and alfalfa (3%) (EPA 2003). Recent use of diuron in California has declined over the preceding decade from approximately 1.5 million pounds applied in 1998 to 730,000 lbs in 2008 (CDPR 2009). In 2008 nonagricultural uses continued to account for the largest amount of diuron applied in California. More than 283,000 lbs of diuron were applied for maintenance of right of ways. The largest agricultural use sites in California included oranges (approximately 144,000 lbs) and alfalfa (approximately 121,000 lbs). Based on recent crop patterns and surveys of typical use, Washington State Department of Agriculture estimates the total annual usage of diuron on asparagus, blueberries, iris bulbs, and cane berries is approximately 7,000 – 10,000 lbs, (WSDA 2010). Usage estimates for other crops and the nonagricultural high usage sites is not available. Recent use information for Oregon and Idaho is not available.

Agricultural Uses. Diuron is used on a variety of fruit and nut crops, grains, cotton, corn, sorghum, mint, asparagus, sugarcane, seed crops, coffee, hay, cut flowers, and for fallow, idle cropland. It may be used in irrigation and drainage systems when water is not present.

Non-agricultural Uses. Diuron is used on impervious surfaces such as paved areas. It is also used on industrial and rights-of-way areas where total vegetation control is desired; often it is combined with other herbicides for total vegetation control. Such broad-spectrum weed control includes along fence lines, rights-of-way (pipelines, powerlines, railway lines, roads), footpaths, in timber yards and storage areas, around commercial, industrial and farm buildings, electrical substations, and petroleum storage tanks. It has some use as an algicide in ornamental ponds, fountains, and aquaria, but not natural water bodies. It may be used as a mildewicide in paints used on buildings and structures.

Registered Formulation Types. Diuron is available in wettable powder, dry flowable, liquid suspension, and soluble concentrate formulations. Many of these products contain additional active ingredients. Most are herbicides, but chlorothalonil, a fungicide, is used in

the paint preservative formulations. Herbicides formulated with diuron include paraquat, thiadiazuron, bromacil, imazapyr, monosodium methanearsonate, tebuthiuron, sodium chlorate, sodium metaborate, sulfometuron-methyl, and copper sulfate (EPA 2003).

Methods and Rates of Application.

Methods. Diuron is a systemic herbicide registered for pre- and post emergent control using ground and aerial equipment. Diuron is typically applied as a pre-emergent herbicide to the soil, and needs to be watered in to be effective. It may persist in the soil throughout much of the season, thus providing continuing control of weeds. It can also be effective as a post-emergent herbicide, especially if applied during high humidity and warm temperatures, and with a surfactant added to enhance penetration into the weeds. In formulations with other herbicides, the other active ingredient(s) typically provides knockdown of established weeds, while the diuron inhibits additional weeds from becoming established (EPA 2003). Diuron is often applied in combination with other herbicides such as bromacil, hexazinone, paraquat, thiadiazuron, imazapyr, monosodium, sodium chlorate, sodium metaborate, and copper sulfate (EPA 2009).

Application Rates. Active labels allow a maximum single application rate of 12 lbs diuron/A on uncultivated agricultural areas, industrial sites, and intermittently flooded areas such as marshes, swamps, and irrigation ditches when water is not present. The maximum annual application rate at these sites is 24 lbs diuron/A. The maximum application rate in crops is 4 lbs diuron/A for single applications and 8 lbs diuron/A annually (Table 5).

Table 5. Summary of all authorized use sites and application restrictions for active diuron products.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Alfalfa (established)	2.4	1	2.4	NA	Aerial or Ground Spray; Chemigation	352-678 352-692
Alfalfa (seed) alfalfa grown for seed	1.2	1	1.2	NA	Aerial or Ground Spray; Chemigation	352-666
Apple	3.2	2	3.2	NS	Ground Spray	352-692
Artichoke in California	3.2	1	3.2	NA	Ground Spray	352-692
Asparagus	3.2	2 ²	4.8	N	Ground Spray	352-692
Barley (Western OR, Western WA)	1.6	1	1.6	NA	Aerial or Ground Spray at planting	352-678
Birdsfoot Trefoil (Western OR)	1.6	1	1.6	NA	Ground Spray	352-692
Blueberry, Caneberry, Gooseberry (Western OR, Western WA)	2.4	2	3.2	one application in Fall and Spring	Ground Spray	352-678
Blackberry, Boysenberry, Dewberry, Loganberry, Raspberry (CA)	2.4	2	3.2	one application in Fall and Spring	Ground Spray	352-692
Citrus (CA)	3.2	2	6.4	60	Ground Spray	352-692
Corn	0.8	1	0.8	NA	Ground Spray	352-692
Cotton (Preplant CA)	1.6	1	2.2	NA	Aerial or Ground Spray	352-692
Cotton (Post emergence CA)	0.6	2	2.2	NS	Ground Spray	352-692
Filberts (except CA)	2.2	2	3.2	150	Ground Spray	352-692
Grape	3.2	2	6.4	90	Ground Spray	352-692
Grass Seed Crops (OR and WA)	2.4	1	2.4	NA	Aerial or Ground Spray	352-692
Oats (spring) (drill planted in OR, WA, ID)	1.2	1	1.2	NA	Ground Spray	352-692
Oats (winter) (drill planted in OR, WA, ID)	1.6	1	1.6	NA	Ground Spray	352-692
Olives (CA)	1.6	2	3.2	NS	Ground Spray	352-692
Papaya	4.0	1	4.0	NA	Ground Spray	352-692
Peas (Austrian field) (Western OR)	1.6	1	1.6	NA	Aerial or Ground Spray	352-692
Peach	2.2 (3 in CA)	1	2.2 (3 in CA)	NA	Ground Spray	352-692
Pear	3.2	2	3.2	NS	Ground Spray	352-692
Pecan	3.2	1	3.2	NA	Ground Spray	352-692
Peppermint/Spearmint	2.4	NS	NS	NS	Ground Spray	352-692

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Red Clover (Western OR)	1.6	1	1.6	NA	Ground Spray	352-692
Sorghum (Southwestern States)	0.4	2	0.4	NS	Ground Spray	352-692
Tree Plantings (CA, OR, WA)	2.4	NS	NS	NS	Ground Spray	352-692
Walnut (English) (CA,OR, WA)	2.2 (3 in CA)	2	3.2 (3 in CA)	once in Fall, once in Spring	Ground Spray	352-692
Wheat (winter) (ID, OR, WA, East of the Cascade Range)	1.2	1	1.2	NA	Aerial or Ground Spray	352-692
Wheat (winter) (OR, WA, West of the Cascade Range)	1.6	1	1.6	NA	Aerial or Ground Spray	352-692
General Weed Control in non-crop and non-timber (e.g. rights of way, uncultivated agricultural areas, fence rows, industrial sites, intermittently flooded areas such as marshes, swamps, and bogs after water has receded)	12	2	24	90	Ground Spray or dry application of granules with ground equipment	228-654; 352-692
Irrigation and Drainage Ditches (when dry)	12	NS	NS	NS	Ground Spray or dry application of granules with ground equipment	352-692
24 (c) CA: citrus in Fresno and Tulare Counties	3.2	2	3.2	once in Fall, once in Spring	Microsprinkler irrigation	352-678; CA-050005
24 (c) CA: Lilly bulbs in Del Norte County	4	NS	4	NS	Ground spray	352-678; CA-870038
24 (c) OR: Triticale	1.2 east of Cascades; 1.6 west of Cascades	1	1.2 east of Cascades; 1.6 west of Cascades	NA	Ground spray	352-678; OR-010029
24 (c) OR: Triticale	1.2 east of Cascades; 1.6 west of Cascades	1	1.2 east of Cascades; 1.6 west of Cascades	NA	Ground spray	352-692; OR-010030
24 (c) OR: Triticale	1.2 east of Cascades; 1.6 west of Cascades	1	1.2 east of Cascades; 1.6 west of Cascades	NA	Ground spray	352-678; OR-070032

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
24 (c) OR: Field grown Easter Lilies in Curry County	4	2	6	once in Fall, once in Spring	Ground spray	352-678; OR-080020
24 (c) OR: Grasses grown for seed	2.4	1	2.4	NA	Aerial or Ground Spray	9779-329; OR-940025
24 (c) OR: Grasses grown for seed	2.4	1	2.4	NA	Aerial or Ground Spray	9779-318; OR-920023
24 (c) WA: Ryegrass grown for seed	1.6	1	1.6	NA	Aerial or Ground Spray	19713-36; WA-000034
24 (c) WA: Ryegrass grown for seed	1.6	1	1.6	NA	Aerial or Ground Spray	19713-274; WA-000033
1. NS = not specified 2. In Washington, apply a single application only						

Metabolites and Degradates.

Diuron degrades in the environment to four major (>10% of the applied parent) and four minor (<10% of the applied parent) metabolites and degradates. The major metabolites are: carbon dioxide (CO₂), N-(3,4-dichlorophenyl)-N-methylurea (DCPMU), N'-(3-chlorophenyl)-N,N-dimethylurea (MCPDMU), and 1,1-dimethyl-3-phenylurea (PDMU). The minor metabolites include: 4-dichlorophenylurea (DCPU); 3,4-dichloroaniline (3,4-DCA), N-(3-chlorophenyl)-N-methylurea (CPMU), and 3,3',4,4'-tetrachlorobenzene (TCAB) (EPA 2009). Additionally, diuron products contains two impurities from the manufacturing process, TCAB and 3,3',4,4'-tetrachloroazoxybenzene (TCAOB), both 'dioxin-like' substances. TCAB levels between 0.15 and 28 ppm have been found in diuron samples tested. TCAOB is present at lower levels (EPA 2009).

Linuron

Linuron is a substituted urea herbicide registered for use on several agricultural crops and some non-agricultural sites. It was first registered in 1966 and is currently used for preplant, preemergence, postemergence, or post-transplant weed control. Linuron is a systemic herbicide that targets grasses and broadleaf weeds by inhibiting the photosystem II reaction center (EPA 2008). Three companies currently hold active registrations for 11 products containing linuron including four technical products (EPA Reg. No. 19713-158, 19713-386, 61842-22 / 352-679, and 61842-24 / 352-726) and 7 end –use products (EPA Reg. No. 19713-97, 19713-251, 61842-20 / 352-660, 61842-21 / 352-677, 61842-23 / 352-686, 61842-24 / 352-726, 66330-218 / 51036-78). There is one SLN registration in California, and none in Idaho, Oregon, and Washington (EPA Reg. No. CA-020006). There are no emergency use registrations for linuron in California, Idaho, Oregon, or Washington.

Usage Information.

Typical use pattern suggest approximately 400,000 lbs of linuron are applied each year to agricultural use sites in the United States

(http://water.usgs.gov/nawqa/pnsp/usage/maps/show_map.php?year=02&map=m1993).

Application to sorghum, cotton, potatoes, and carrots account for about 80% of the domestic agricultural uses. In California linuron use has generally declined over the last decade from approximately 82,000 lbs in 1998 to 59,000 lbs in 2008 (CDPR 2010).

Washington State Department of Agriculture estimates total maximum usage of linuron on asparagus, carrots, and wheat at 3,367, 4,952, and 12,066 lbs, respectively (WSDA 2010). Use estimates for other crops, and non-crop areas were not available.

Agricultural Uses. Linuron use sites in the action area include asparagus, bulbs (CA), carrots, celery, corn, kenaf, marigolds grown for seed (CA), parsley grown for seed (OR, WA), parsnips, sorghum, soybeans, wheat, post-harvest crop stubble and fallow lands.

Non-agricultural Uses. Non-crop areas such as roadsides and fence rows (EPA Reg. No. 19717-97).

Registered Formulation Types. End use products containing linuron are formulated in wettable powders, flowable concentrates, water dispersible granules, and liquid suspensions. One linuron formulation also contains diuron (EPA Reg. No. 352-660). Otherwise, all active labels of diuron contain a single active ingredient.

Methods and Rates of Application.

Methods. Linuron is applied through chemigation, ground boom, or other ground application methods. Aerial applications are not permitted. Several labels provide recommendations for tank mixtures with surfactants and herbicides. For example, one label suggests possible tank mixtures with alachlor, atrazine, Prowl 3.3 EC, Lexone, gramoxone , glyphosate, metribuzin, and 2,4-D (e.g. EPA Reg. No. 19713-97). Some labels contain soil type restrictions to manage the risk of surface and ground water contamination (EPA Reg. No. 19713-97, 352-660, 352-677).

Application Rates. Active labels allow for a maximum single application rate of up to 4lbs of linuron/A and a maximum annual application rate of up to 6 lbs a.i./A. Most field crops allow 1-2 lbs a.i./A for a single application and ≤ 2 lbs a.i./A annually (Table 6).

Table 6. Summary of all authorized use sites and application restrictions for active linuron products.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Asparagus	4	NS ¹	4 / season Or 6 /year	NS	Ground Chemigation	19717-97 19713-251
Bulbs (CA only)	1	NS	NS	NS ²	Ground Chemigation	19717-97
Celery	1.5 1 (in CA)	1	NS	do not apply within 67 days	Ground	19717 -97

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Corn	1.5	NS ³	NS	NS	Chemigation	19717 -97
Parsnips	1.5	1	1.5	NA	Ground	19717 -97
Sorghum	1	2 ⁴	NS	NS	Chemigation	19717 -97
Soybeans ⁵	2	NS	NS	NS	Ground	19717 -97
Wheat, Drill planted winter (ID,OR,WA)	0.25 to 1.75 ⁶	2 (Spring and Fall)	0.5 – 3.5	At least 4 months	Ground Chemigation: Preemergent broadcast spray	19717 -97
Non-crop weed control: Roadsides, fence rows, , etc.	3	NS	NS	NS	Ground Chemigation	19717-97
Carrots	1	NS	2 / year	NS	Ground Chemigation	19710-251
Cotton (not in CA)	0.6 ⁷	Max 3	1.5 / season	21 day minimum	Ground	352-660
Kenef	1	1	1	na	Ground Chemigation	352-677
Post Harvest, Crop stubble, fallow ground, stale seedbed	2	1 (fallow season)	2	NA	Ground	352-677
24 (c) OR, WA: Parsley grown for seed	1	2 in first growing season and 2 in second growing season	NS	21 day minimum	NS	Supplemental label registration numbers not provided 352-686
24 (c) CA: Marigolds grown for seed	1	1	1	NA	Ground	CA-020006 (1812-320)
1. NS = not specified 2. Pre-emergence, only during growing season 3. Single application is specified, but not clear if that is a yearly or seasonal limit 4. One pre- and one post-emergent applications permitted 5. The soybean use directions include multiple types of applications and recommended mixtures. These are further broken down by soil texture and % organic material. 6. West of Cascades = 1.75, East of Cascades (with 10-20 in rainfall) = .25 (varies w/ rainfall) 7. This formulation includes an equal amount diuron as an additional active ingredient						

Metabolites and Degradates.

In the soil, linuron degrades to 3,4-dichlorobenzenamine (DCA), n-(3,4-dichlorophenyl)-N-methylurea (DCPMU), N-(3,4-dichlorophenyl)-N'' methoxyurea (DML), AND (3,4-dichlorophenyl)urea (DCPU). In anaerobic aqueous environments, major degradates include desmethoxy linuron and desmethoxy monolinuron (EPA 2004).

Captan

Captan was first registered in 1951 to control fungal disease in fruit crops. It is currently registered as a non-systemic fungicide in orchards, vineyard, turf, ornamentals, and a large variety of food crops. The mode of action of captan is inhibition of normal cell division on a broad spectrum of microorganisms and fungi. Captan inhibits the process of oxidative phosphorylation in fish, invertebrates, and other nontarget aquatic and terrestrial organisms (EPA 2007). There are 43 active labels for end use products containing captan that are held by nine registrants. There are six SLN registrations in California, Idaho, Oregon, and Washington (EPA Reg. No.CA020017, CA100006, CA980023, OR030029, OR070024, WA940026).

Usage Information

EPA estimates more than 5 million lbs of captan are applied annually for domestic uses (EPA 2004). Recent data from California indicate agriculture use of captan has declined from over 1.5 million lbs to approximately 350,000 lbs during the last decade (CDPR 2008). Washington State Department of Agriculture estimates total annual usage of captan in apples, blueberries, cane berries, and strawberries at 10,903 – 104,754 lbs, based on recent crop patterns and surveys of typical use (WSDA 2010). Recent usage for other crops and recent information for Oregon and Idaho is not available.

Agricultural Uses. Grasses/turf (seed crops, ornamentals, sod farms, grapes, honeydew, kale, lentils, lespedeza, lettuce, milo, mustard seed, nectarines, oats, onions, okra,

peaches, prunes, peanuts, peas, peppers, potatoes, roses, radish, raspberries, rye, rutabaga, strawberries, Swiss chard, soybeans, spinach, squash, safflower, sunflower, sesame, greenhouse, sorghum, sugar beets, tomatillo, tomatoes, turnips, wheat, lily bulbs.

Non-agricultural Uses. Active labels allow captan use on turf (golf course, lawn seed beds) and ornamentals.

Registered Formulation Types. Captan is formulated into more than forty end-use products including liquid, dust, and granular formulations. Ten formulations include mixtures of one or more other active ingredients. Captan is mixed with the fungicides PCNB (*e.g.*, EPA Reg. No. 264-949), thiophonate-methyl (*e.g.*, 264-998), trifloxystrobin (*e.g.*, 264-999), carboxin (*e.g.*, 400-555), metalaxyl (*e.g.*, 400-561), sulfur (*e.g.*, 4-355), and fenhexamid (*e.g.*, 66330-48), and the insecticides imidacloprid (*e.g.*, 400-568), malathion (*e.g.*, 4-59), and carbaryl (*e.g.*, 4-122).

Methods. Captan is a contact fungicide applied as a seed treatment, a root dip, an in-furrow application, and by various ground and aerial foliar applications. Several active labels suggest captan can or should be applied with other fungicides and/or insecticides (*e.g.*, EPA Reg. No. 4-459, 19713-235, 19713-268, 19713-362, 19713-385, 19713-405, 62575-6, 66222-1, 66222-24, 66222-66, 66330-209).

Application Rates. Active labels allow for a maximum single application rate of up to 4 lbs captan/A and an annual application rate of up to 35 lbs captan/ acre (Table 7). Many products are applied as seed treatments and consequently only applied once per year. However, up to 8 foliar applications/year are allowed in several crops.

Table 7. Summary of all authorized use sites and application restrictions for active captan products.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Alfalfa, Clover, Lespedeza,	0.1792	1	12	NS	Seed dip	19713-161

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Trefoil						
Almonds	4.5	1	20	7	Aerial or Boom Spray	66222-58 66222-66 66222-24
Apples	4	8	26.3	14	Aerial or Boom Spray	66330-27 66330-29 66330-54
Apricots	2.5	NS	12.5	NS	Aerial or Boom Spray	66330- 209
Artichoke	NS	NS	NS	NS	NS	400-568
Azaleas	2	1	NS	NS	Root dip	19713- 235 19713- 258 19713- 268 19713- 362
Barley	0.0356	1	12	NS	Seed dip	264-931 66330- 238
Beans (dry and snap)	0.0309	1	NS	NS	Seed dip	264-931 66330- 238
Beans	0.1008	1	NS	NS	Seed dip	264-931 66330- 238 400-567
Beets	0.01416	1	12	NS	Seed dip	264-928
Begonias	2	1	NS	NS	Root dip	19713- 235 19713- 258 19713- 268 19713- 362
Blackberries	2	8	10	14	Aerial or Boom Spray	19713- 258 19713- 268
Blueberries	2.5	NS	35	10	Aerial or Boom Spray	19713- 268 19713- 258
Blue Grass	0.16558	1	NS	NS	Seed dip	19713- 161

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Brassica (Cole)	0.0318	1	NS	NS	Seed dip	400-568
Cabbage	NS	1	NS	NS	Seed dip	19713-258 66330-27
Canola	.0051	1	NS	NS	Seed dip	400-568 400-567
Cauliflower	NS	1	NS	NS	Seed dip	19713-258
Camellias	2	1	NS	NS	Root dip	19713-235 19713-258 19713-268 19713-362
Cantaloupe, Cucumber	0.00304	1	12	NS	Seed dip	264-931 66330-238
Carnations	1	NS	NS	10	Root dip	19713-235 19713-258 19713-268 19713-362
Carrots	NS	1	NS	NS	Seed dip	19713-258
Cherries	2	8	14	14	Aerial or Boom Spray	19713-258 19713-268
Chrysanthemums	2	1	NS	NS	Root dip	19713-235 19713-258 19713-268 19713-362
Cilantro	NS	1	NS	NS	Seed dip	400-568 400-567
Clover	N/A	1	NS	NS	Seed dip	19713-258
Cole Crops (Broccoli, Brussels Sprouts,	0.0544	1	12	NS	Seed dip	264-931 66330-238

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Cabbage, Cauliflower)						
Conifers	NS	NS	NS	NS	Seed dip	19713-258
Cotton - Acid Delinted	0.0107	1	NS	NS	Seed dip	264-931
Cotton - Fuzzy	0.0149	1	NS	NS	Seed dip	264-931
Cotton - Machine Delinted	0.0096	1	NS	NS	Seed dip	264-931
Collards	N/A	1	NS	NS	Seed dip	19713-258
Corn - Field	0.0244	1	NS	NS	Seed dip	264-931
Corn - Sweet	0.0439	1	NS	NS	Seed dip	264-931
Cowpeas	0.0640	1	NS	NS	Seed dip	66330-238
Crucifers (mustard, radish, rape, turnips)	0.00732	1	12	NS	Seed dip	264-931 66330-238
Cucurbits	NS	1	NS	NS	Seed dip	400-568
Dewberries	2	8	10	14	Aerial or Boom Spray	19713-235 19713-258 19713-268
Dichondra	0.0125	1	NS	NS	Root dip	66222-66
Eggplant	N/A	N/A	N/A	N/A	N/A	19713-258 400-568 400-567
Flax	0.0641	1	NS	NS	Seed dip	264-931 66330-238
Gladiolus	0.75	1	NS	NS	Root dip	19713-235 19713-258 19713-268 19713-362
Gladiola Bulbs	0.037	1	NS	NS	Seed dip	19713-258 19713-268 66330-238

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Ginseng	2	8	15.6	10	Aerial or Boom Spray	19713- 235
Grasses (seed)	0.1461	1	NS	NS	Seed dip	264-931
Grasses (Ornamental)/ Turf (Golf Course)	1	2	4.3	8.6	Aerial or Boom Spray	19713- 235 19713- 258 19713- 268
Grasses (Lawn Seedbeds)/Turf (Sod Farms)	1	2	4.3	8.6	Aerial or Boom Spray	19713- 235 19713- 258 19713- 268
Grapes	2	8	12	14	Aerial or Boom Spray	19713- 258 19713- 268
Honeydew	NS	1	NS	NS	Seed dip	19713- 258
Kale	NS	1	NS	NS	Seed dip	19713- 258
Lentils	0.1098	1	12	NS	Seed dip	19713- 258 34704- 935 66330- 238
Lespedeza	0.3514	1	NS	NS	Seed dip	19713- 258 66330- 238
Lettuce	NS	1	NS	NS	Seed dip	19713- 258 400-568
Milo	0.2635	1	NS	NS	Seed dip	19713- 258 66330- 238
Mustard Seed	0.859	1	NS	NS	Seed dip	400-568
Nectarines	2	NS	24	14	Aerial or Boom Spray	19713- 258 19713- 268

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Oats	0.0475	1	NS	NS	Seed dip	264-931 66330-238
Onions	NS	1	NS	NS	Seed dip	66330-238
Onions (pelleting)	0.0409	1	NS	NS	Seed dip	19713-161
Okra	NS	1	NS	NS	Seed dip	19713-258
Paint Additive	NS	NS	NS	NS	Additive	66330-31
Peaches	4	NS	32	14	Aerial or Boom Spray	19713-258 19713-268
Plums/Fresh Prunes (Western US)	2	NS	27	14	Aerial or Boom Spray	19713-258 19713-268
Peanuts	0.0913	1	1	NS	Seed dip	66330-238
Peas	0.05562	1	12	NS	Seed dip	66330-238
Peppers (California wonder)	0.0053	1	NS	NS	Seed dip	19713-161
Peppers	0.73	1	12	NS	Seed dip	66330-238
Potatoes	1	1	NS	NS	Dusted at cutting for seed	400-568 2935-536 19713-258
Roses	1	NS	NS	14	Aerial or Boom Spray	19713-235 19713-258 19713-268 19713-362
Radish	0.0716	NS	NS	NS	Seed dip	66330-238
Raspberries	2	8	10	14	Aerial or Boom Spray	19713-258 19713-268
Rye, Triticale	0.297	1	12	NS	Seed dip	264-931 66330-238

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Rutabaga	NS	1	NS	NS	Seed dip	19713-258
Strawberries	3	8	24	7	Aerial or Boom Spray	19713-258 19713-268
Swiss Chard	0.0114	1	12	NS	Seed dip	66330-238
Soybeans	0.1627	1	12	NS	Seed dip	66330-238
Spinach	0.1627	1	12	NS	Seed dip	66330-238
Squash, Watermelon, Pumpkin, Muskmelon	0.0124	1	12	NS	Seed dip	66330-238
Safflower	NS	1	NS	NS	Seed dip	19713-258
Sunflower	0.0976	1	12	NS	Seed dip	66330-238
Sesame	NS	1	NS	NS	Seed dip	19713-258
Soil/Greenhouse Bench	0.006387	1	NS	NS	Root dip/soil condition/ Spray	66330-234 66330-239
Sorghum (Milo)	0.0087	1	12	NS	Seed dip	66330-238
Sugar Beets – Western US	0.012	1	12	NS	Seed dip	66330-238
Tomatillo	.318	1	NS	NS	Seed dip	400-568
Tomatoes	.318	1	NS	NS	Seed dip	400-568 19713-258
Turnips	0.0078	1	NS	NS	Seed dip	66330-238
Wheat	0.03213	1	12	NS	Seed dip	264-931 66330-238
24 (c) CA: Strawberry	NS	1	NA	NA	Plant dip	CA-020017; 51036-166
24 (c) CA: Lily bulbs	4	1	4	NA	Soak bulbs in solution then apply solution	CA-100006; 1973-156

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
					in-furrow	
24 (c) CA: Easter lily bulbs	NS	1	NS	NA	Soak bulbs in solution	CA-980028; 264-931
24 (c) OR: Easter lily	NS	1	NS	NA	Soak bulbs in solution	OR-030029; 19713-156
24 (c) OR: Grass Seeds for Export Only	0.8 grams carboxin and 0.76 grams captan /kg seed	NA	NA	NA	Seed treatment for export, not for use immediately prior to planting	OR-070024; 400-554
1. NS = not specified						

Metabolites and Degradates

Captan degrades in soil or water include tetrahydrophthalate (THPI), trichloromethylthio (TCMT), tetrahydrophthalimic acid (THPAm), cyclohex-4-ene-2-cyano-1-carboxylic acid (THCY), inorganic sulfur and chlorine, and thiophosgene (EPA 2003).

Chlorothalonil

Chlorothalonil is a broad spectrum pesticide, used primarily as a fungicide and registered for use on a variety of crop and noncrop sites (*e.g.* nursery, home and garden, golf course). It has also been registered for use as a wood protectant, antimold and antimildew agent, bactericide, microbiocide, algacide, insecticide, and acaricide (EPA 1999). Chlorothalonil exact mechanism of toxicity for vertebrate species is unknown, although in fungus it is reported to interfere with cellular respiration by binding glutathione (EPA 2007). More than 40 companies hold active registrations for pesticides containing chlorothalonil, and there are more than 100 enduse products containing chlorothalonil are currently registered with EPA (<http://ppis.ceris.purdue.edu/htbin/cnamlst2.com>). There are eleven SLN registrations in California, Idaho, Oregon, and Washington (EPA Reg. No. CA-030010, CA-960027, OR-000023, OR-030008, OR-990038, OR-990037, OR-990039, OR-990040, WA-

000003, WA-020012, WA 000014). There are no emergency use registrations (section 18) for chlorothalonil in California, Idaho, Oregon, or Washington.

Usage Information.

EPA reported an average domestic use of approximately 15 million lbs of chlorothalonil per year for the period 1999-2000 (EPA 2003). Recent data show use of chlorothalonil has declined in California over the last decade from approximately 1.8 million lbs in 1998 to 558 thousand lbs in 2008 (CDPR 2009). In recent years, the greatest use of chlorothalonil in California has been on tomatoes, almonds, and landscape maintenance. Washington State Department of Agriculture estimates total annual usage of chlorothalonil on potatoes exceeds 96,000 lbs and approximately 20, 000 – 28,000 lbs of chlorothalonil are used on onions. Annual use of chlorothalonil on cranberry (1,000 – 12,000 lbs), Christmas trees (1,000-18,000 lbs), and iris and tulip bulbs (600-1,200 lbs) account for a smaller proportion of current use of the chemical in Washington (WSDA 2010). Estimated use on other sites in Washington and recent usage information for Oregon and Idaho is not available.

Agricultural Uses. Chlorothalonil is approved for use on a variety of vegetables, field, orchard, turf, and ornamental crops.

Non-agricultural Uses. Non-agricultural uses of chlorothalonil include golf course, forestry, lawns around commercial and industrial buildings, athletic fields, landscaping in residential, public, parks, and recreational areas. It is also used as a wood protectant, antimold, and antimildew agent.

Registered Formulation Types. End use products containing chlorothalonil are available in a variety of liquid applied formulations including emulsifiable concentrates, wettable powders, and water dispersable granules. Chlorothalonil end-use pesticides frequently contain other active ingredients such as propiconazole (*e.g.*, EPA Reg. No. 100-1347), azoxystrobin (100-1315), mandipropamid (100-1279), mefenoxam (100-1221), and fludioxonil (100-1231).

Methods and Rates of Application.

Methods. Chlorothalonil may be spray-applied by aerial and ground application methods, including chemigation. Tank dip applications are also authorized for bulbs and several plants. Some labels specify that the product must not be applied within 150 feet for aerial and air-blast applications, or 25 feet for ground applications of marine/estuarine water bodies (EPA Reg. No. 50534-201). Several labels indicate that chlorothalonil products can be tank mixed with many commonly used insecticides, fungicides, and spray adjuvants (EPA Reg. No. 100-800).

Application Rates. Active labels allow for a maximum single application rate of up to 11.3 lbs of chlorothalonil/A on golf courses and some lawn and turf use sites (Table 8). Most agricultural applications are restricted to single application rates of 7 lbs chlorothalonil/A. However, tank-dip solution from bulb treatments can be applied to the field at a rate of 41.25 lbs a.i./A. Additionally, the maximum seasons application rates are quite high for some field crops (celery 18 lbs a.i./A), nut and fruit crops (pistachio 22.5 lbs a.i./A), ornamentals (36.4 lbs a.i./A), and golf course applications (27 and 73 lbs a.i./A for fairways and greens, respectively).

Table 8. Summary of all authorized use sites and application restrictions for active chlorothalonil products.

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Brassica, head and stem	1.5	NS	8.8 per growing season	7-10	Air, ground or chemigation	60063-7
Chinese (napa) cabbage	1.5	NS	8.8 per growing season	7-10	Air, ground or chemigation	60063-7
Curbits: Cucumber, Cantaloupe, Muskmelon, Honeydew melon, Watermelon, Squash, Pumpkin, Zucchini Additional Crops: Chayote, Chinese waxgourd,	3.0	NS	15.75 per growing season	7-10	Air, ground or chemigation	60063-7

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Gourds, Momordica spp.						
Fruiting Vegetables (does not include tomatos): Eggplant, Groundcherry, Okra, Pepino, Peppers	1.25	NS	9.0 per growing season	7-10	Air, ground or chemigation	50534-201; 50534-188
Ginseng	1.5	NS	12.0 per growing season	7-10	Air, ground or chemigation	60063-7
Horseradish	2.25	NS	18.0 per growing season	7-10	Air, ground or chemigation	60063-7
Lupine, Lentil	1.125	NS	6.0 per growing season	7-10	Air, ground or chemigation	60063-7
Persimmon	0.94	NS	4.7 per growing season	14	Air, ground or chemigation	60063-7
Rhubarb	2.25	NS	13.5 per growing season	7-10	Air, ground or chemigation	60063-7
Yam	0.94	NS	11.25 per growing season	10-14	Air, ground or chemigation	60063-7
Turf / sod farms	7.3	NS	13.0	7	Air, ground or chemigation	50534-202
Turf / lawns around commercial/industrial buildings, collegiate and professional athletic fields, ornamental turfgrass (lawns at homes, apartments, etc., excluded)	11.3	1-NS	26.0	7	NS - Spray application	50534-209
Golf courses tees, greens and fairways	11.3	1-2 depending on rate	73.0 greens; 52.0 tees; 26.0 fairways ²	Variable depending on concentration rates	Air, ground or chemigation	50534-202; 50534-209
Ornamentals	2.1 lbs./100 gallons water; 1.54 lb./ac	NS	36.4 per growing season	7-14	Air, ground or chemigation	50534-202; 66222-149
Flowering bulbs (caladium, crocus, daffodil, iris, lily, tulip)	41.25 ³	1	41.25 ³	Dip once prior to planting bulbs	Dip tank, then spray apply material to field with ground	50534-202

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
					application equipment.	
Conifers Christmas tree plantations; forestry applications.	4.1	For as long as conditions are favorable for disease.	16.5	7-21 depending on size of trees.	Air, ground spray	66222-149; 50534-201
Asparagus	3.0	NS	9.0	14-28	Air, ground	66222-149; 50534-201
Beans, Dry: Including but not limited to: Navy, Pinto, Kidney, Lima, Broad, Pink, Jack, Cow pea, Chick pea (Garbanzo), Black-eyed pea, Southern Pea	1.5	NS	6.0	7-10	Air, ground spray or chemigation	66222-149; 50534-201
Beans, Snap	2.25	NS	9.0	7	Air, ground spray	66222-149; 50534-201
Blueberry	3.0	NS	9.0	10-14	Air, ground spray	66222-149; 50534-201
Cabbage, Broccoli, Cauliflower, Brussels sprouts, Chinese Mustard Cabbage	1.5	NS	12.0	7-10	Air, ground spray	66222-149; 50534-188
Carrot	1.5	NS	15.0	7-10	Air, ground spray or chemigation	66222-149; 50534-201
Celery	2.25	NS	18.0	7	Air, ground spray or chemigation	66222-149; 50534-188
Corn	1.5	NS	9.0	7	Air, ground spray	66222-149; 50534-201
Cranberry	5.0 ⁴	NS	15.0	10-14	Air, ground spray or chemigation	66222-149; 50534-201
Grasses grown for seed	1.5	NS	4.5	14	Air, ground spray or chemigation	66222-149; 50534-201
Mango	2.64	NS	24.0	7-14	Air, ground spray	66222-149; 50534-201
Mint (Oregon)	1.0	NS	3.0	7-10	Air, ground spray	66222-149
Onion (dry bulb), and Garlic	2.25	NS	15.0	7-10	Air, ground spray	50534-201
Onion (green bunching) Leek, Shallot, Onion, Garlic grown for seed	2.25	NS	6.75	7-10	Air, ground spray or chemigation	66222-149; 50534-201; 50534-188
Papaya	2.25	NS	6.75	14	Air, ground spray	66222-149; 50534-201; 50534-188
Parsnip	2.2	NS	6.0	7-10	Air, ground spray	66222-149

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
					or chemigation	
Peanut	1.1	NS	9.0	14	Recommended to alternate chemigation with air or ground spray	66222-149; 50534-201
Potato	1.2	NS	11.25	5-10	Air, ground spray or chemigation	66222-149; 50534-201
Soybean	1.8	NS	4.5	10-14	Air, ground spray or chemigation	66222-149; 50534-201
Tomato	2.2	NS	15.1	7-14	Air, ground spray or chemigation	66222-149; 50534-201
Strawberry	1.2	NS	15.0	10-14	Air, ground spray or chemigation	66222-149
Almonds	3.0	NS	18.75	NS	Air, ground spray	66222-149; 50534-201
Filberts	3.0	NS	9.0	14-28	Air, ground spray	66222-149; 50534-201
Fruit Trees: Apricot, Cherry (sweet and tart), Nectarine, Peach Plum, Prune	3.1	NS	15.5	10-14	Air, ground spray	66222-149; 50534-201
Pistachio	4.125	NS	22.5	28	Air, ground spray	66222-149; 50534-201
Passion Fruit	1.5	NS	7.5	14	Air, ground spray	50534-201
Wood protectant and antimildew and antimildew agent	16.25 ⁵	NS	NS	NS	Brush, spray, or dip application	577-544
24 (c) CA: Strawberry Transplants	1.125 / 100 gallons water	One dip per season	NS	NS	Tank dip only	CA-960027; 50534-188
24 (c) CA: Strawberry (non-bearing nurseries)	1.125 / 100 gallons water	NS	NS	10-14	Ground spray or chemigation	CA-960027; 50534-188
24 (c) WA: treatment for "bulb rot"	4.125 / 100 gallons water	NS	NS	NS	Tank dip only	WA-000003; 50534-202
24 (c) OR: Sugar beets (seed production only)	1.3	NS	NS	NS	Air, ground spray or chemigation	OR-990040; re:50534-188
24 (c) OR: Mint	1.1	3	3.3	7-10	Air, ground spray or chemigation	OR-990038; 50534-188-10182
24 (c) OR:	1.0	3	3.0	7-10	Air, ground spray	OR-

Use(s)	Max. Single App. Rate (lbs a.i./A)	Number of App. per Year	Annual App. Rate (lbs a.i./A)	App. Interval (days)	App. Method	Label Number
Mint					or chemigation	990037; 50534-201
24 (c) OR: Ornamental Bulbs	4.125 / 100 gallons water	NS	NS	NS	Tank dip only	OR- 000023; 50534- 202-100
24 (c) WA: Chickpeas	1.5	NS	6.0 per growing season	7-10	Air, ground or chemigation	WA- 020012; 60063-7
24 (c) WA: Conifer seedlings forest tree nursery and green house management.	4.1	For as long as conditions are favorable for disease.	16.5 per growing season	7-14	Ground spray	WA 000014; 50534-202
24 (c) OR: Chickpeas	1.5	5	6.0 per growing season	7-10	Air, ground spray or chemigation	OR- 030008; 66222-149
24 (c) OR: Sugar beets (seed production only)	1.3	NS	NS	NS	Air, ground spray or chemigation	OR- 990039; 0534-201
24 (c) CA: Gladiolus corms	2.0 / 100 gallons water	One dip per season	NS	NS	Tank dip only	CA- 030010; 50534-209
<ol style="list-style-type: none"> 1. NS = not specified 2. Label discrepancy allows for much higher annual use rate than could be achieved considering single application maximum and limits on number of applications 3. Bulbs dipped in solution at rate of up to 5.0 lbs. product / 100 gallons of water. This ratio is equivalent to 41.25 lbs. a.i. / acre as calculated assuming 100 gallons of solution applied/A. 4. Label specifies not to apply to bogs when flooded or allow release of irrigation water from bogs for at least 3 days following application. 5. Wood stain product containing 0.7% chlorothalonil and 0.3% bis(tributyltin) oxide. Apply at rates of up to 1 gallon / 150 ft². Assumed net weight of 8 lbs/gallon as net weight not provided. 						

Metabolites and Degradates.

EPA has identified 4-hydroxy-2,5,6-trichloro-1,3-dicyanobenzene (SDS-3701) as a degradate of concern for terrestrial organisms due to its elevated toxicity and persistence relative to chlorothalonil (EPA 2007). It is ubiquitous in the environment and consistently found at 10 – 40% of the applied parent. Four other degradates/metabolites (SDS-19221,

SDS-46851, SDS-47523/SDS-47524, and SDS-47525) have been identified as products in aerobic soil or anaerobic aquatic conditions (EPA 1999).

Action Area

The action area is defined as all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action (50 CFR §402.02).

Given EPA's nationwide authorization of these pesticides, the action area would encompass the entire U.S. and its territories. These same geographic areas would include all listed species and designated critical habitat under NMFS jurisdiction.

In this instance, as a result of the 2002 order in Washington Toxics Coalition v. EPA, EPA initiated consultation on its authorization of 37 pesticide a.i.s and their effects on listed Pacific salmonids under NMFS' jurisdiction and associated designated critical habitat in the states of California, Idaho, Oregon, and Washington. Consequently, for purposes of this Opinion, the action area consists of the entire range and most life history stages of listed salmon and steelhead and their designated critical habitat in California, Idaho, Oregon, and Washington. The action area encompasses all freshwater, estuarine, marsh, swamps, nearshore, and offshore marine surface waters of California, Oregon, and Washington. The action area also includes all freshwater surface waters in Idaho (Figure 2).

2,4-D, triclopyr BEE, diuron, linuron, captan and chlorothalonil are the fourth set of pesticides identified in the consultation schedule established in the settlement agreement and are analyzed in this Opinion. NMFS' analysis focuses only on the effects of EPA's action on listed Pacific salmonids in the above-mentioned states. It includes the effects of these pesticides on the recently listed Lower Columbia River coho salmon, Puget Sound steelhead, and Oregon Coast coho salmon. The Lower Columbia River coho salmon was listed as endangered in 2005. The Puget Sound steelhead and the Oregon Coast coho salmon were listed as threatened in 2007 and 2008, respectively.

EPA's consultation with NMFS remains incomplete until it analyzes the effects of its authorization of pesticide product labels with these six compounds for all remaining threatened and endangered species under NMFS' jurisdiction. EPA must ensure its action does not jeopardize the continued existence or result in the destruction or adverse modification of critical habitat for other listed species and designated critical habitat under NMFS' jurisdiction throughout the U.S. and its territories.

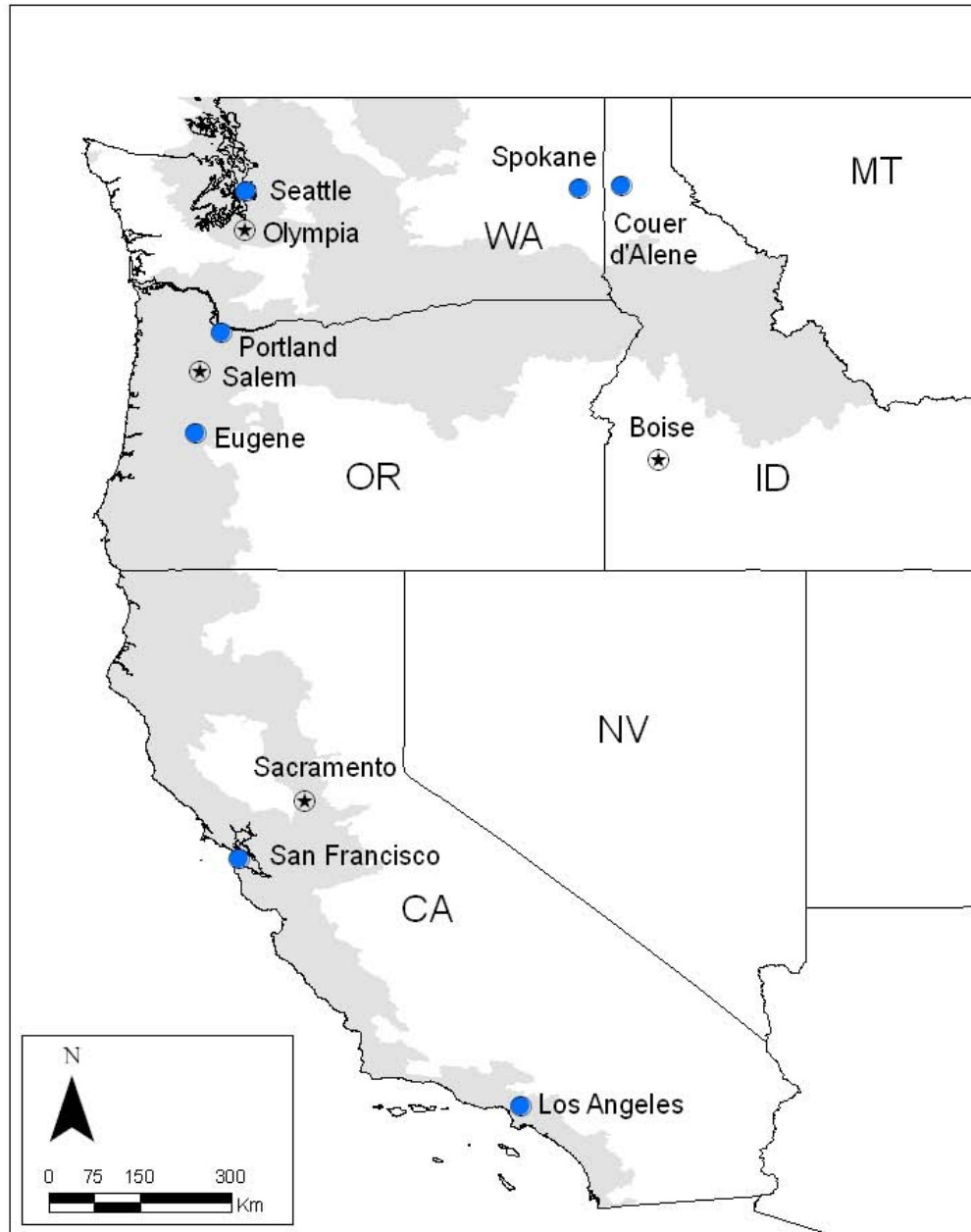


Figure 2. Map showing extent of inland action area with the range of all ESU and DPS boundaries for ESA listed salmonids highlighted in gray.

Approach to this Assessment

Overview of NMFS' Assessment Framework

NMFS uses a series of steps to assess the effects of federal actions on endangered and threatened species and designated critical habitat. The first step of our analysis identifies those physical, chemical, or biotic aspects of proposed actions that are likely to have individual, interactive, or cumulative direct and indirect effects on the environment (we use the term “potential stressors” for these aspects of an action). As part of this step, we identify the spatial extent of any potential stressors and recognize that the spatial extent of those stressors may change with time. The spatial extent of these stressors is the “action area” for a consultation.

The second step of our analyses identifies the listed resources (endangered and threatened species and designated critical habitat) that are likely to occur in the same space and at the same time as these potential stressors. If we conclude that such co-occurrence is likely, we then try to estimate the nature of co-occurrence (these represent our *Exposure Analyses*). In the exposure analysis, we try to identify the life stage and life history of the individuals that are likely to be exposed to an action’s effects and the populations or subpopulations those individuals represent. Spatial analyses are used to overlay each species range with land types that pesticides are used on including agriculture, urban/residential, forested, and right of ways, to evaluate co-occurrence of pesticides and salmonids.

Once we identify which listed resources are likely to be exposed to potential stressors associated with an action and the nature of that exposure, in the third step of our analysis we examine the scientific and commercial data available to determine whether and how those listed resources are likely to respond given their exposure (these represent our

Response Analyses). We integrate the exposure and response analyses within the *Risk Characterization section* to assess the risk to listed individuals and their habitat from the stressors of the action.

In the *Risk Characterization Section*, , we also determine whether population level effects are anticipated (these analyses are conducted within the risk characterization phase).

NMFS' analysis is ultimately a qualitative assessment that draws on a variety of quantitative and qualitative tools and measures to address risk to listed resources.

In the final steps of our analyses, we establish the risks posed to listed species and to designated critical habitat. This part of the analysis is found within the *Integration and Synthesis section* where spatial analyses are used to determine overall risk to listed resources.

Our jeopardy determinations for listed species must be based on an action's effects on the continued existence of threatened or endangered species as those "species" have been listed, which can include true biological species, subspecies, or distinct population segments of vertebrate species. Because the continued existence of listed species depends on the fate of the populations that comprise them, the viability (that is, the probability of extinction or probability of persistence) of listed species depends on the viability of the populations that comprise the species. Similarly, the continued existence of populations are determined by the fate of the individuals that comprise them; populations grow or decline as the individuals that comprise the population live, die, grow, mature, migrate, and reproduce (or fail to do so).

The structure of our risk analyses reflects the relationships between listed species, the populations that comprise each species, and the individuals that comprise each population. Our risk analyses begin by identifying the probable risks actions pose to listed individuals that are likely to be exposed to an action's effects. Our analyses then integrates those individual-level effects to identify consequences to the populations those individuals represent. Our analyses conclude by determining the consequences of those population-level risks to the species those populations comprise.

We evaluate risks to listed individuals by measuring the individual's "fitness" defined as changes in an individual's growth, survival, annual reproductive success, or lifetime reproductive success. In particular, we examine the scientific and commercial data available to determine if an individual's probable response to an action's effect on the environment (which we identify in our *Response Analyses*) are likely to have consequences for the individual's fitness.

Reductions in abundance, reproduction rates, or growth rates (or increased variance in one or more of these rates) of the populations those individuals represent is a *necessary* condition for reductions in a population's viability, which is itself a *necessary* condition for reductions in a species' viability. On the other hand, when listed plants or animals exposed to an action's effects are *not* expected to experience reductions in fitness, we would not expect that action to have adverse consequences on the viability of the population those individuals represent or the species those populations comprise ((Anderson, Phillips et al. 2006), (Mills and Beatty 1979), (Stearns 1982)). If we conclude that listed species are *not* likely to experience reductions in their fitness, we would conclude our assessment because an action that is not likely to affect the fitness of individuals is not likely to jeopardize the continued existence of listed species.

If, however, we conclude that listed plants or animals are likely to experience reductions in their fitness, our assessment determines if those fitness reductions are likely to be sufficient to reduce the viability of the populations those individuals represent (measured using changes in the populations' abundance, reproduction, spatial structure and connectivity, growth rates, or variance in these measures to make inferences about the population's extinction risks). In this step of our analyses, we use the population's base condition (established in the *Status of Listed Resources* and *Environmental Baseline* sections of this Opinion) as our point of reference. Finally, our assessment determines if changes in population viability are likely to be sufficient to reduce the viability of the species those populations comprise.

The critical habitat analysis focuses on reductions in the quality, quantity, or availability of primary constituent elements (PCEs - Table 9) from exposure to the stressors of the action. Since chemicals are the stressors of the action for this Opinion, PCEs potentially affected are freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, and nearshore marine areas. The PCE attributes of prey availability and water quality are the primary assessment endpoints addressed when evaluating the effects of the action on designated critical habitat. Information evaluated for effects to prey include prey survival, prey growth, prey drift, prey reproduction, abundance of prey, health of invertebrate aquatic communities, and recovery of aquatic communities following pesticide exposure. Information evaluated for degradation of water quality include anticipated exposure concentrations leading to toxic responses within aquatic organisms (including salmonids and their prey) as well as instances of water bodies not meeting local, state, or federal water quality standards and criteria.

Evidence Available for the Consultation

We search, compile and use a variety of resources to conduct our analyses including:

- EPA's BEs, REDs, IREDS, other documents developed by EPA
- Peer-reviewed literature
- Gray literature
- Books
- Available pesticide labels
- Any correspondence (with EPA or others)
- Available monitoring data and other local, county, and state information
- Pesticide registrant generated data
- Online toxicity databases (PAN, EXTOTOXNET, ECOTOX, USGS, NPIC)
- Pesticide exposure models run by NMFS
- Population models run by NMFS
- Information and data provided by the registrants identified as applicants
- Comments on the draft Opinion from EPA, stakeholders, and any applicants
- Incident reports

Collectively, this information provides the basis for our determination as to whether and to what degree listed resources under our jurisdiction are likely to be exposed to EPA's action and whether and to what degree the EPA can ensure that its authorization of

pesticides is not likely to jeopardize the continued existence of threatened and endangered species or is not likely to result in the destruction or adverse modification of designated critical habitat.

Application of Approach in this Consultation

For this consultation, we adapt our general approach to incorporate elements of EPA's ecological risk assessment framework (EPA 1998). Figure 3 shows the overall framework used in this Opinion. This risk assessment framework organizes the available information in three phases: problem formulation, analysis of exposure and response, and risk characterization (EPA 1998). We adapted the EPA framework to address ESA-specific considerations. The NMFS framework follows a process for organizing, evaluating, and synthesizing the available information on listed resources and the stressors of the action. We separately evaluate the risk to listed species and the risk to designated critical habitat from the stressors of the action (See *Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids* and *Effects of the Proposed Action to Designated Critical habitat*). Below, we briefly describe the problem formulation phase in the general framework.

Problem Formulation Phase

Here we generate conceptual models from our initial evaluation of the relationships between stressors of the action (pesticides and other identified chemical stressors) and potential receptors (listed species and habitat). These relationships are presented in conceptual model diagrams and written risk hypotheses (Species Risk Hypotheses and Critical Habitat Risk Hypotheses). A conceptual model diagram is constructed to illustrate potential pesticide exposure pathways and associated listed resources' responses. The conceptual model for Pacific salmonids is presented in Figure 4. In it, we illustrate where the pesticides generally reside in the environment following application, how pesticides may co-occur with listed species and their habitats, and how the individuals/habitat may respond upon exposure. In the case of Pacific salmonids, we

ascribe exposure and response to specific life stages of individuals and then assess individual fitness endpoints sensitive to the action's stressors.

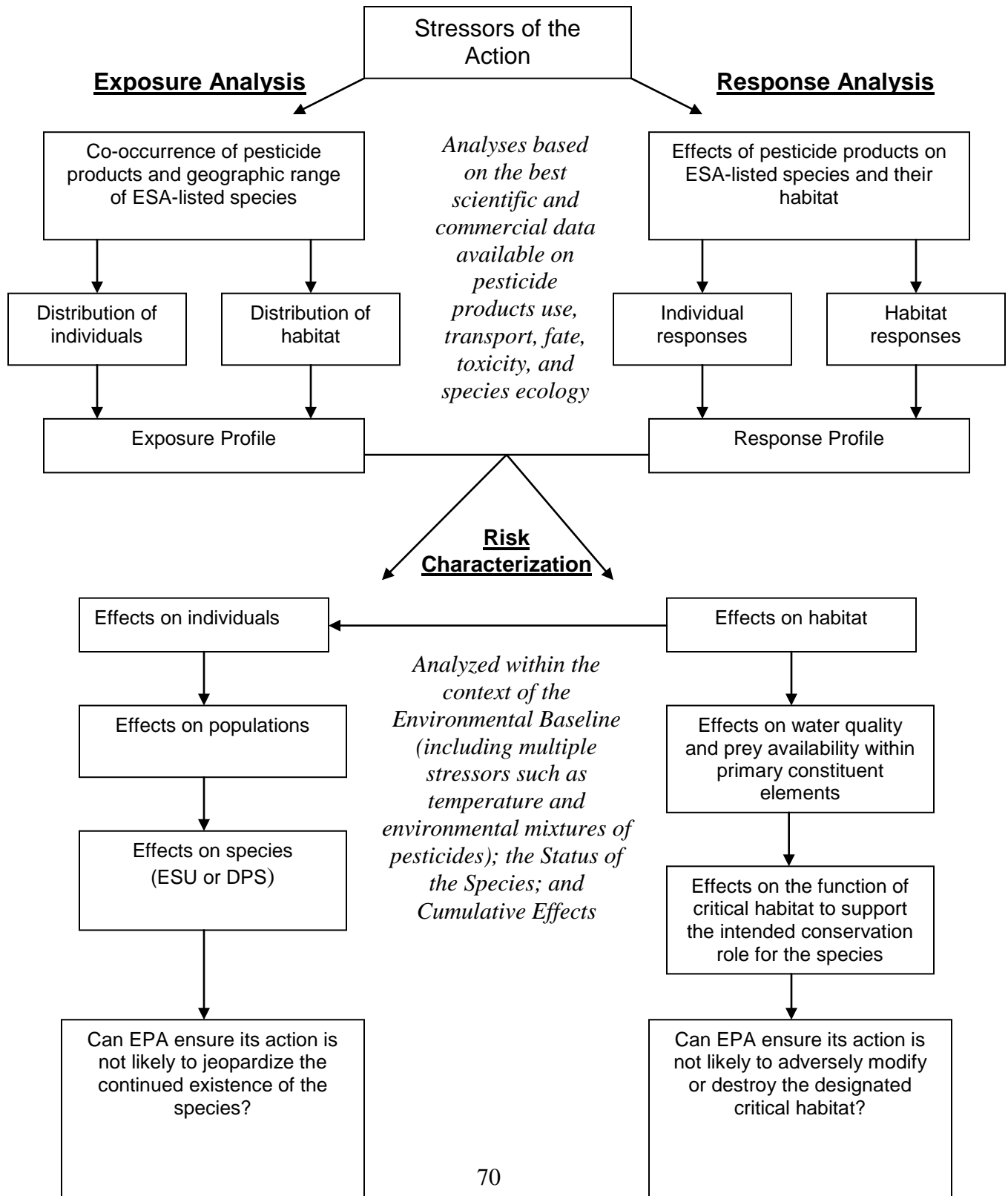


Figure 3. Conceptual framework for assessing risks of EPA's action to ESA listed resources.

In the problem formulation phase, we also identify the toxic mode and mechanism of action of chemical stressors, particularly for the pesticide a.i.s- if known. This information helps us understand what an organism's physiological consequences may be following exposure. It also helps us evaluate whether mixture toxicity occurs because we identify other pesticides that share similar modes of action and the likelihood for co-occurrence in listed species habitats. A similar mode of action with other pesticides is a key determinant of the likelihood of mixture toxicity. Based on the problem formulation, we then search, compile, and review the available toxicity information for the identified stressors to ascertain which physiological systems are known to be affected and to what degree. This analysis is contained in the *Response Analysis* in the *Effects of the Proposed Action to Threatened and Endangered Salmonids* section.

Species Risk Hypotheses

We develop risk hypotheses by identifying biological requirements or assessment endpoints for listed resources in the action area that are potentially affected by the stressors of the action. Table 10 shows examples of salmonid assessment endpoints and measures. We designate assessment endpoints as those biological properties of species and their habitat essential for successful completion of an individual's life cycle. We integrate the listed resources information with what is known about the stressors of the action, including their physical properties, use, presence in aquatic habitats, and their toxicity. We then evaluate how listed salmonids and their habitat are potentially affected by the stressors of the action and integrate this information with exposure information to develop risk hypotheses.

Species risk hypotheses are evaluated in the *Risk Characterization* section of *Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids*. Uncertainties pertaining to toxicity and exposure are identified and discussed as they are encountered in the analysis. Effects of these uncertainties on the ultimate conclusions are discussed in relevant sections of the Opinion.

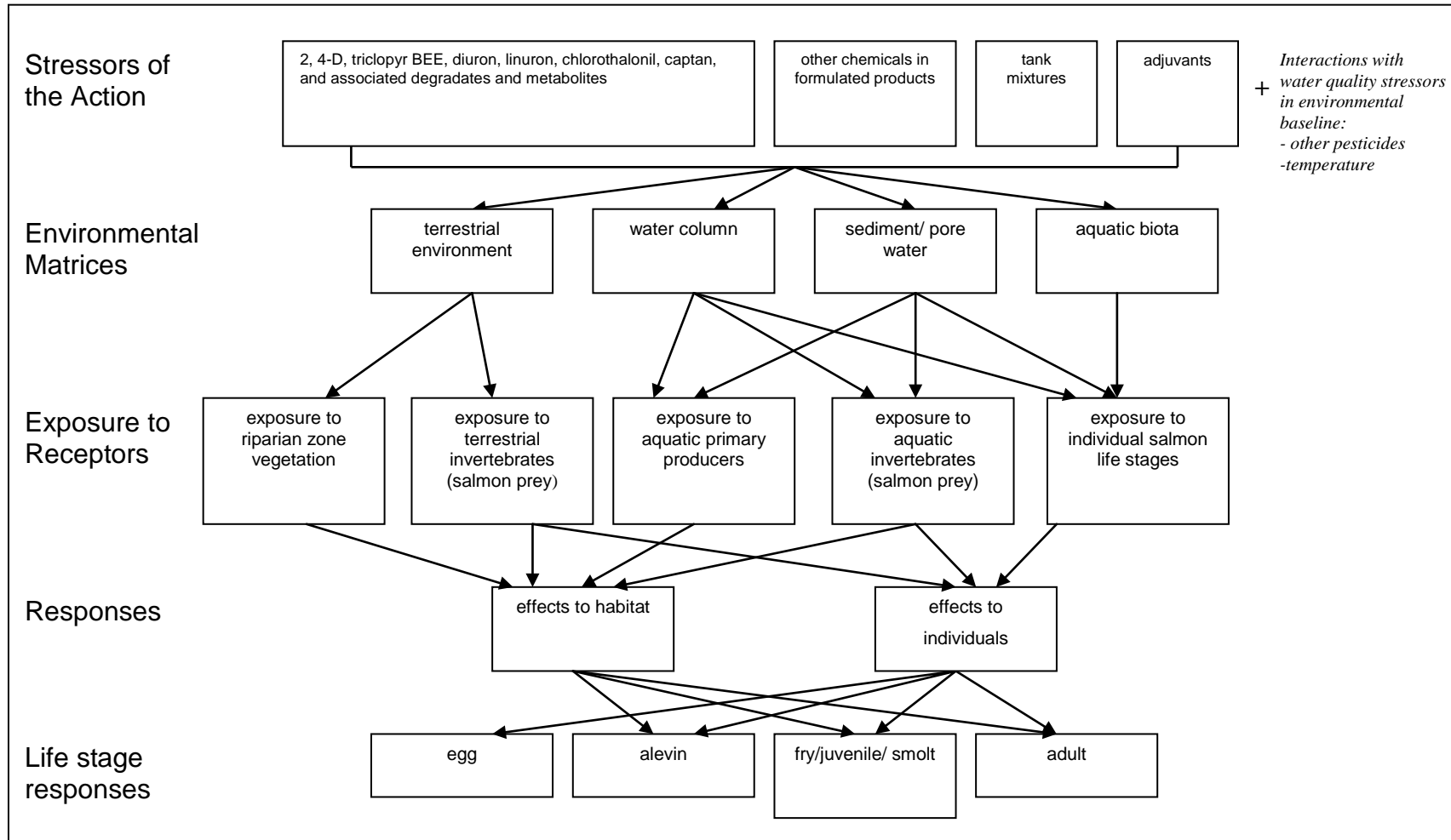


Figure 4. Exposure pathways of the stressors of the action and general responses of Pacific salmonids and habitat.

These risk hypotheses will be evaluated in the Risk Characterization Section:

1. Exposure to the stressors of the action is sufficient to:
 - a. kill salmonids from direct, acute exposure;
 - b. reduce salmonid survival through impacts to growth or development;
 - c. reduce salmonid growth through impacts to salmonid prey;
 - d. reduce survival, migration, and reproduction through impacts to olfactory-mediated behaviors; and
2. Exposure to the herbicides 2, 4-D, triclopyr BEE, diuron, and linuron is sufficient to:
 - a. reduce aquatic primary producers thereby affecting salmonid prey communities and salmonids and natural cover;
 - b. reduce riparian vegetation to such an extent that stream temperatures are elevated, erosion increases, and reductions in natural coverage results through reduced inputs of woody debris and other organic matter.
3. Exposure to mixtures of diuron and linuron can act in combination to increase adverse effects to salmonids and salmonid habitat.
4. Exposure to active ingredient degradates, adjuvants, tank mixtures, and other active and other ingredients in pesticide products containing 2, 4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil cause adverse effects to salmonids and their habitat.
5. Exposure to other pesticides present in the action area can act in combination with 2, 4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil to increase effects to salmonids and their habitat.
6. Exposure to elevated temperatures can enhance the toxicity of the stressors of the action.

Critical Habitat Risk Hypotheses

To determine potential effects to designated critical habitat, NMFS evaluates the effects of the action by first looking at whether PCEs of critical habitat are affected by the stressors of the action (Table 9). Effects to PCEs include changes to the functional condition of salmonid habitat caused by the action in the action area. Properly

functioning salmonid PCEs are important to the conservation of the ESU/DPS. The stressors of the action for this Opinion are chemicals introduced into the environment by application of pesticide products. As such, the key PCEs that are potentially affected are freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, estuarine areas, and nearshore marine areas where exposure is anticipated.

Table 9. Essential physical and biological features named as PCEs in all salmonid critical habitat designations.

Site	Essential Physical and Biological features	Species Life Stage
Freshwater Spawning	Water quality, water quantity, and substrate	Spawning, incubation larval development
Freshwater rearing	Water quantity and floodplain connectivity	Juvenile growth and mobility
	Water quality and forage	Juvenile growth and development
	Natural cover ^a	Juvenile mobility and survival
Freshwater migration	Free of obstructions, water quality and quantity, and natural cover ^a	Juvenile and adult mobility and survival
	forage	Juvenile growth and development
Estuarine areas	Free of obstruction, water quality and quantity, and salinity	Juvenile and adult physiological transitions between salt and freshwater
	Natural cover ^a and forage ^b and water quantity	Growth and maturation
Nearshore Marine areas	Free of obstruction, water quality and quantity, natural cover ^a and forage ^b	Growth and maturation, survival
Offshore marine areas	Water quality and forage ^b	Growth and maturation

^a Natural cover includes shade, large wood, log jams, beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks.

^b Forage includes aquatic and terrestrial invertebrates and fish and shellfish species that support growth and maturation.

Primary attributes of these PCEs that are potentially affected by the stressors of the action are prey availability, water quality, and natural cover. Below are the risk hypotheses (written as affirmative statements) which we evaluate in the *Effects of the Proposed Action to Designated Critical Habitat* section:

1. Exposure to the stressors of the action is sufficient to degrade water quality, and substrate in freshwater spawning sites;
2. Exposure to the stressors of the action is sufficient to degrade water quality, reduce prey availability (forage), and/or reduce natural cover in rearing sites;
3. Exposure to the stressors of the action is sufficient to degrade water quality, prey availability, and/or reduce natural cover in freshwater migration corridors;
4. Exposure to the stressors of the action is sufficient to degrade water quality, prey availability, and/or reduce natural cover in estuarine areas;
5. Exposure to the stressors of the action is sufficient to degrade water quality, prey availability and/or reduce natural cover in nearshore marine areas.

These hypotheses are evaluated using the best scientific and commercial data available, and are presented in the *Risk Characterization* section of the *Effects of the Proposed Action to Designated Critical Habitat*. If the available evidence supports a risk hypothesis, then NMFS evaluates whether the potential reductions in PCEs overlap with habitats rated as high, medium, or low conservation value. Conservation values were determined by Biological Review Teams (BRTs). This portion of the analysis is conducted in the *Integration and Synthesis* section.

Table 10. Examples of salmonid lifestage and habitat assessment endpoints and measures.

Salmonid Life Stage	Assessment Endpoint (individual fitness)	Assessment Measure (measures of changes in individual fitness)
Egg* * Is the egg permeable to pesticides (measured by pesticide concentrations in eggs)?	Development Survival	size, hatching success, morphological deformities viability (percent survival)
Alevin (yolk-sac fry)	Respiration Swimming: predator avoidance site fidelity Yolk-sac utilization: growth rate size at first feeding Development Survival	gas exchange, respiration rate swimming speed, orientation, burst speed predator avoidance assays rate of absorption, growth weight and length weight and length morphology, histology LC ₅₀ (dose-response slope). Percent dead at a given concentration
Fry, Juvenile, Smolt	First exogenous feeding (fry)– post yolk- sac absorption Survival Growth Feeding Swimming: predator avoidance behavior migration use of shelter Olfaction: kin recognition predator avoidance imprinting feeding Smoltification (smolt) Development	time to first feeding, starvation LC ₅₀ (dose-response slope). Percent dead at a given concentration weight, length stomach contents, weight, length, starvation, prey capture rates swimming speed, orientation, burst swimming speed predator avoidance assays swimming rate, downstream migration fish monitoring, bioassays electro-olfactogram measurements, behavioral assays behavioral assays behavioral assays behavioral assays Na/K ATPase activity, sea water challenge tests length, weight, malformations
Returning adult	Survival Feeding Swimming: predator avoidance migration spawning feeding Sexual development Olfaction: Predator avoidance Homing Spawning	LC ₅₀ (dose-response slope). Percent dead at a given concentration stomach contents behavioral assays numbers of adult returns, behavioral assays numbers of eggs fertilized stomach contents histological assessment of ovaries/testis electro-olfactogram measurements, measurements of intersex behavioral assays behavioral assays behavioral assays
Habitat	Aquatic primary producers salmonid prey abundance dissolved oxygen and pH	growth inhibition bioassays (EC50) survival (LC ₅₀); field measured community metrics direct measurements

	natural cover for salmonids Riparian zone vegetation natural cover for salmonids sedimentation temperature	growth inhibition bioassays (EC50) salmonid monitoring (field) direct measurements direct measurements
--	--	---

Analysis Plan

Status of the Species

In this section, we present information regarding each of the ESUs and DPSs considered in this Opinion. We discuss life history, population abundance and trends and overall viability of the species. This provides part of the context in which we evaluate the effect of the proposed action.

Environmental Baseline

In this section we discuss all stressors affecting salmon populations including natural predators, events and disease; and anthropogenic effects such as pollution and habitat modification. This also provides part of the context in which we evaluate the effect of the proposed action.

Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids

In the *Exposure* section we discuss life histories of the various species which may make them more or less likely to be exposed to stressors of the actions. Then we evaluate measured and estimated environmental concentrations of the stressors from various sources. The *Response* section details toxicity information for the assessment endpoints identified in the problem formulation. In the *Risk Characterization* sections for listed species and designated critical habitat, we integrate the exposure and response information and evaluate the risk hypotheses. *Risk Characterization* may also include population-level analyses to determine if effects on an individual fitness are sufficiently large to affect population parameters. Finally, we conclude the *Effects of the Proposed*

Action to Threatened and Endangered Pacific Salmonids with a summary of risk associated with each of the a.i.s. This summary presumes use sites are proximate to salmon populations and habitat. Note that an ESU/DPS specific co-occurrence analysis to show where use sites for the pesticide products overlap with salmon populations is done in the *Exposure section*.

Integration and Synthesis

In separate sections for listed species and critical habitat, we combine risk conclusions regarding the effects of the proposed action with information in the *Status of the Species* and *Environmental Baseline* to determine potential effects on populations and species.

Conclusion

Based on the potential effects for each species, we determine if the effects of the proposed action is likely to jeopardize the survival and recovery of the species or cause destruction or adverse modification of designated critical habitat.

Other Considerations

In this Opinion, we evaluated lines of evidence constructed as species-specific risk hypotheses to ensure relevant endpoints were addressed. The analysis weighs each line of evidence by evaluating the best commercial and scientific data available that pertain to a given risk hypothesis. Overall, the analysis is a qualitative approach that uses some quantitative tools to provide examples of potential risks to listed salmonids and their habitat. Multiple methods and tools currently exist for addressing contaminant-induced risk to the environment. Hazard-based assessments, probabilistic risk assessment techniques, combinations of the two, and deterministic approaches such as screening level assessments have been applied to questions of risk related to human health and the environment.

In recent pesticide risk assessments, probabilistic techniques have been used to evaluate the probability of exceeding a “toxic” threshold for aquatic organisms by combining pesticide monitoring data with species sensitivity distributions (Geisy, Solomon et al. 1999; Giddings 2009). There is utility in information generated by probabilistic approaches if supported by robust data. NMFS considered the use of probabilistic risk assessment techniques for addressing risk at population and species (ESU and DPS) scales for the stressors of the action. However, we encountered significant limitations in available data that suggested the information was not sufficient to define exposure and/or response probabilities necessary to determine the probability of risk. Probabilistic techniques were not used in the Opinion due to issues with data collection, paucity of data, non-normal distributions of data, and quality assurance and quality control. For example, it was not deemed appropriate to pair the salmonid prey responses with exposure probabilities based on monitoring results given the limitations of that data set discussed in the *Effects of the Proposed Action*. To evaluate population consequences associated with potential lethality from pesticide exposure in salmon, NMFS selected the lowest reported salmonid LC₅₀ from the available information to ensure risk was not underestimated. When we consider the data limitations coupled with the inherent complexity of EPA’s proposed action in California, Idaho, Oregon, and Washington, we find that probabilistic assessments at population and species scales introduce an unquantifiable amount of uncertainty that undermines confidence in derived risk estimates. These same studies do not factor the status of the existing health and baseline conditions of the environment into their assessment. At this time, the best available data do not support such an analysis and conclusions from such an analysis would be highly speculative.

Status of Listed Resources

The purpose of this section is to characterize the condition of the 28 salmonid species¹ under consultation relative to their likelihood of viability and to describe the conservation role and function of their respective critical habitats. NMFS has determined that the following species and critical habitat designations may occur in the action area for EPA's registration of 2,4-D, triclopyr BEE, diuron, linuron, captan and chlorothalonil - containing products (Table 11). More detailed information on the status of these species and critical habitat are found in a number of published documents including recent recovery plans, status reviews, stock assessment reports, and technical memorandums. Many are available on the Internet at <http://www.nmfs.noaa.gov/pr/species/>.

Table 11. Listed Species and Critical Habitat (denoted by asterisk) in the Action Area.

Common Name (Distinct Population Segment or Evolutionarily Significant Unit)	Scientific Name	Status
Chinook salmon (Puget Sound*)	<i>Oncorhynchus tshawytscha</i>	Threatened
Chinook salmon (Lower Columbia River*)		Threatened
Chinook salmon (Upper Columbia River Spring-run*)		Endangered
Chinook salmon (Snake River Fall-run*)		Threatened
Chinook salmon (Snake River Spring/Summer-run*)		Threatened
Chinook salmon (Upper Willamette River*)		Threatened
Chinook salmon (California Coastal*)		Threatened
Chinook salmon (Central Valley Spring-run*)		Threatened
Chinook salmon (Sacramento River Winter-run*)		Endangered
Chum salmon (Hood Canal Summer-run*)	<i>Oncorhynchus keta</i>	Threatened
Chum salmon (Columbia River*)		Threatened
Coho salmon (Lower Columbia River)	<i>Oncorhynchus kisutch</i>	Threatened
Coho salmon (Oregon Coast*)		Threatened
Coho salmon (Southern Oregon & Northern California Coast*)		Threatened
Coho salmon (Central California Coast*)		Endangered
Sockeye salmon (Ozette Lake*)	<i>Oncorhynchus nerka</i>	Threatened
Sockeye salmon (Snake River*)		Endangered
Steelhead (Puget Sound)	<i>Oncorhynchus mykiss</i>	Threatened

¹ We use the word "species" as it has been defined in section 3 of the ESA, which include "species, subspecies, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature (16 U.S. C 1533)." Pacific salmon that have been listed as endangered or threatened were listed as "evolutionarily significant units (ESU), which NMFS uses to identify distinct population segments of Pacific salmon. Any ESU or DPS is a "species" for the purposes of the ESA.

Common Name (Distinct Population Segment or Evolutionarily Significant Unit)	Scientific Name	Status
Steelhead (Lower Columbia River*)		Threatened
Steelhead (Upper Willamette River*)		Threatened
Steelhead (Middle Columbia River*)		Threatened
Steelhead (Upper Columbia River*)		Threatened
Steelhead (Snake River*)		Threatened
Steelhead (Northern California*)		Threatened
Steelhead (Central California Coast*)		Threatened
Steelhead (California Central Valley*)		Threatened
Steelhead (South-Central California Coast*)		Threatened
Steelhead (Southern California*)		Endangered

The following narratives summarize the biology and ecology of threatened and endangered Pacific salmonids that are relevant to EPA's proposed action. This includes a description of the timing and duration of each life stage such as adult river entry, spawning, egg incubation, freshwater rearing, smolt outmigration, and ocean migration. These summaries provide a foundation for NMFS' evaluation of the effects of the proposed action on listed salmonids. We also highlight information related to the viability of salmonid populations and the primary constituent elements (PCEs) of designated critical habitat.

Species Status

The status of an ESU or DPS is determined by the degree that it (1) maintains sufficient genetic and phenotypic diversity to ensure continued fitness in the face of environmental change, (2) maintains spatial distribution of populations so that not all populations would be affected by a catastrophic event, and (3) maintains sufficient connectivity among populations within the ESU or DPS to maintain long-term demographic and evolutionary processes ((ICTRT 2007), (McElhany, Ruckelshaus et al. 2000), (Spence, Bjorkstedt et al. 2008)). We describe the current condition of the spatial structure and major life histories within the ESUs or DPSs. In order to maintain a spatial distribution and diversity that support a viable ESU or DPS, a species must maintain multiple viable populations that are sustainable in the long-term in the face of environmental variability.

Before assessing population viability, we first identify the historic and current populations that constitute a species. How NMFS defines a population and its function are found in McElhany *et al.* (McElhany, Ruckleshaus et al. 2000) and in Bjorkstedt *et al.* (Bjorkstedt, Spence et al. 2005). NMFS' Pacific salmon Technical Recovery Teams (TRTs) have identified historic populations within ESUs/DPSs. These historical populations have been categorized based on their distribution and demographic role (*i.e.*, functionally independent, potentially independent, or dependent). Functionally independent (independent) populations were sufficiently large to be viable in isolation, (*i.e.*, a negligible extinction risk). Potentially independent populations were potentially viable in isolation, but were likely influenced by immigrants from adjacent populations. Dependent populations were unlikely to persist over a 100-year time period in isolation. However, immigration from other nearby populations reduced the extinction risk for dependent populations. The historical conditions of the populations for each ESU/DPS serve as a point of reference for evaluating the current viability of populations² and the status of the species. The current viability is used as the base condition from which the effects of the proposed action on individuals are evaluated to determine whether these effects are likely to increase the probability of extinction of the populations those individuals represent.

In our *Approach to the Assessment* section, NMFS introduced the VSP concept and its four criteria. We restate that a VSP is an independent population (a population of which extinction probability is not substantially affected by exchanges of individuals with other populations) with a negligible risk of extinction, over a 100-year period, when threats from random catastrophic events, local environmental variation, demographic variation, and genetic diversity changes are taken into account (McElhany, Ruckleshaus et al. 2000). The four factors defining a viable population are a population's: (1) spatial

² The TRTs did not propose that historical conditions are the criteria or benchmark for evaluating population or ESU viability (extinction risk).

structure; abundance; (3) annual growth rate, including trends and variability of annual growth rates; and (4) diversity (McElhany, Ruckleshaus et al. 2000).

A population's tendency to increase in abundance and its variation in annual population growth defines a viable population ((McElhany, Ruckleshaus et al. 2000), (Morris and Doak 2002)). A negative long-term trend in average annual population growth rate will eventually result in extinction. Further, a weak positive long-term growth rate will increase the risk of extinction as it maintains a small population at low abundances over a longer time frame. A large variation in the growth rates also increases the likelihood of extinction ((Lande 1993), (Morris and Doak 2002)).

Thus, in our status reviews of each listed salmonid species, we provide information on population abundance and annual growth rate of extant and extirpated populations. We use the median annual population growth rate (denoted as lambda, λ) from available time series of abundance for independent populations (Good, Waples et al. 2005). Several publications provide a detailed description of the calculation of lambda ((Good, Waples et al. 2005), (McClure, Holmes et al. 2003)). The lambda values for salmonid populations presented in these papers are summarized in *Appendix 2*.

Conservation Role of Critical Habitat for the Species

The action area for this consultation contains designated critical habitat. Critical habitat is defined as the specific areas within the geographical area occupied by the species, at the time it is listed, on which are found those physical or biological features that are essential to the conservation of the species, and which may require special management considerations or protection. Critical habitat can also include specific areas outside the geographical area occupied by the species at the time it is listed that are determined by the Secretary to be essential for the conservation of the species (ESA of 1973, as amended, section 3(5)(A)).

The primary purpose in evaluating the status of critical habitat is to identify for each ESU or DPS the function of the critical habitat to support the intended conservation role for each species. Such information is important for an adverse modification analysis as it establishes the context for evaluating whether the proposed action results in negative changes in the function and role of the critical habitat for species conservation. NMFS bases its critical habitat analysis on the areas of the critical habitat that are affected by the proposed action and the area's physical or biological features that are essential to the conservation of a given species, and not on how individuals of the species will respond to changes in habitat quantity and quality.

In evaluating the status of designated critical habitat, we consider the current quantity, quality, and distribution of those primary constituent elements or PCEs that are essential to the conservation of the species [50 CFR 424.12(b)]. NMFS has identified PCEs of critical habitat for each life stage (*e.g.*, migration, spawning, rearing, and estuary) common for each species. To fully understand the conservation role of these habitats, specific physical and biological habitat attributes (*e.g.*, water temperature, water quality, forage, etc.) were identified for each life stage. Specifically, during all freshwater life stages, salmonids require cool water that is free of contaminants. During the juvenile life stage, salmonids also require stream habitat that provides excess forage (*i.e.*, prey abundance). Besides potential toxicity, water free of contaminants is important as contaminants can disrupt normal behavior necessary for successful migration, spawning, and juvenile rearing. Sufficient forage is necessary for juveniles to maintain growth that reduces freshwater predation mortality, increase overwintering success, initiate smoltification, and increase ocean survival. A description of the past, ongoing, and continuing activities that threaten the functional condition of PCEs and their attributes are described in the *Environmental Baseline* section of this Opinion.

NMFS has identified six common PCEs for 7 California listed Chinook salmon and steelhead (70 FR 52488), 12 ESUs of Oregon, Washington, and Idaho salmon (chum,

sockeye, Chinook) and steelhead (70 FR 52630), and for the Oregon Coast coho salmon (73 FR 7816). They are:

(1) Freshwater spawning sites with water quantity and quality, and suitable substrate size as attributes necessary to support spawning, incubation and larval development;

(2) Freshwater rearing sites with the following attributes: (i) Water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; (ii) Water quality and forage supporting juvenile development; and (iii) Natural cover such as shade, submerged and overhanging large wood, log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks.

(3) 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 supporting juvenile and adult mobility and survival.

(4) Estuarine areas free of obstruction and excessive predation with:

(i) Water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; (ii) Natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels; and (iii) Juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation.

(5) Nearshore marine areas free of obstruction and excessive predation with:

(i) Water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and (ii) Natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels.

(6) Offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.

NMFS similarly developed the following list of species habitat requirements and PCEs for coho salmon ESUs (64 FR 24049). They are:

1. Juvenile summer and winter rearing areas,
2. Juvenile migration corridors,
3. Areas for growth and development to adulthood,
4. Adult migration corridors, and
5. Spawning areas.

Within these areas, essential habitat attributes of coho salmon critical habitat include adequate: (1) substrate, (2) water quality, (3) water quantity, (4) water temperatures, (5) water velocity, (6) cover/shelter, (7) food, (8) riparian vegetation, (9) space, and (10) safe passage conditions. Riparian vegetation refers to its role in providing essential habitat for coho salmon such as instream woody debris and submerged vegetation for holding and shelter, low water temperature through shading, functional channel bottom substrate for development of eggs and alevins by stabilizing stream banks and capturing fine sediment in runoff, and food by providing nutrients to streams and production of terrestrial insects.

In this section, we also identify the conservation values of watersheds located within the critical habitat designated for a species. If the effects on PCEs are important at the watershed scale, then the conservation value for the watershed is used to assess the conservation role of that watershed in the context of range wide critical habitat. The conservation value of a particular watershed was determined by Critical Habitat Analytical Review Teams (CHARTs). These teams considered the presence of PCEs within each occupied area of a watershed and the activities that potentially affect the PCEs, and assigned conservation values for watersheds within designated critical habitat.

Each watershed was scored as low, moderate, or high conservation value. High value watersheds/areas have a high likelihood of promoting species conservation, while low value watersheds/areas are less important for species conservation. Scores were based on: (1) a comparison of current quantity of PCEs within a watershed relative to other watersheds and probable historic quantity of PCEs within the watershed; (2) existing quality of PCEs in watersheds; (3) the likelihood of achieving PCE potential in a watershed; (4) the PCEs support of rare genetic or life history characteristics or rare/important habitat types in the watershed; (5) considerations of the PCEs support of variable-sized populations relative to other watersheds and the probable historical levels in the watershed; and (6) considerations of the PCE support of spawning or rearing of varying numbers of populations.

Chinook Salmon

Description of the Species

Chinook salmon are the largest of the Pacific salmon and historically ranged from the Ventura River in California to Point Hope, Alaska in North America, and in northeastern Asia from Hokkaido, Japan to the Anadyr River in Russia (Healey 1991). Chinook salmon prefer streams that are deeper and larger than those used by other Pacific salmon species. We discuss the distribution, life history, status, and critical habitat of nine species² of endangered and threatened Chinook salmon separately.

Chinook salmon are generally described as one of two races, within which there is substantial variation ((Healey 1991), (Groot and Margolis 1991)). One race, the “stream-type,” resides in fresh water for a year or more following emergence from gravel nests. Juveniles migrate to sea as yearlings. Stream-type Chinook salmon normally returns in late winter and early spring (spring-run) as immature adults and reside in deep pools during summer before spawning in fall. The other race, the “ocean-type,” migrate to the ocean within their first year (sub-yearlings) and usually return as full mature adults in fall (fall-run). Fall-run adults spawn soon after river entry.

The timing of return to fresh water, and ultimately spawning, often provides a temporal isolating mechanism for populations with different life histories. Return timing is often related to spawning location. Thus, differences in the timing of spawning migration also serve as a geographic isolating mechanism. Fall-run Chinook salmon generally spawn in the mainstem of larger rivers and are less dependent on flow, although early autumn rains and a drop in water temperature often provide cues for movements to spawning areas. Spring-run Chinook salmon take advantage of high flows from snowmelt to access the upper reaches of rivers.

Successful incubation depends on several factors including dissolved oxygen (DO) levels, temperature, substrate size, amount of fine sediment, and water velocity. Chinook salmon egg incubation time is highly correlated with water temperature (McCullough 1999). Spawning sites have larger gravel and more water flow up through the gravel than the sites used by other Pacific salmon. Maximum survival of incubating eggs and the pre-emergent alevins occurs at water temperatures between about 5.5° and 13.5°C. Development time is influenced by degree days with fertilization to emergence taking up to 325 days at 2°C and about 50 days at 16°C (McCullough 1999). Fry emergence commonly begins in December and continues into mid April (Leidy 1984). When emerging from the redd, fry move through the interstitial spaces in the redd substrate to escape the gravel. However, a high content of fines and sand in the redd substrate can severely hinder fry emergence and cause high mortality (Bjornn and Reiser 1991). Optimal temperatures for both Chinook salmon fry and fingerlings range from 12° to 14°C (Boles 1988). Temperatures above 15°C increase the risk of diseases and lower the tolerance to other stressors (McCullough 1999). At about 19°C, Chinook salmon cease to eat. In the laboratory, 50% mortality during a 24 hour period is observed at 24° to 25°C ((Brett 1952), (Hanson 1997)) the exact lethal temperature being somewhat dependent on the temperature that the fish has been acclimated to.

Chinook salmon alevins, as is the case for other salmonids, rely on yolk for nutrition until the onset of active feeding. It is important that the young start feeding at the proper time since failure to start feeding can retard growth and lead to behavioral or developmental problems that reduce survival. In Chinook salmon, alevins may start feeding immediately upon emergence even if they have not yet absorbed all of the egg yolk (Linley 2001). During freshwater residence, Chinook salmon juveniles feed in the water column and from the water surface. Food items include a variety of small terrestrial and aquatic insects and aquatic crustaceans; the prey species of juveniles depend on availability (habitat and months), prey size distribution, and the size of the fish ((Koehler, Fresh et al. 2006), (Rondorf, Gray et al. 1990)). The coarse bottom substrate found in faster flowing riverine habitats supports drift of larger aquatic insects such as caddisflies (*Trichoptera*), mayflies (*Ephemeroptera*), stoneflies (*Plecoptera*), and other benthic organisms when they are present in the water column during high flow events. These taxa, when present, are important food items in terms of biomass for Chinook salmon juveniles. Terrestrial insects and midges (*Diptera: Chironomidae*) often dominate the diet in slower moving water with finer bottom substrate such as floodplains, off-channel ponds, sloughs, and in lakes/reservoirs ((Miller and Simenstad 1997), (Rondorf, Gray et al. 1990), (Sommer, Nobriga et al. 2001), (Tabor, Gearns et al. 2006)). In addition, copepods and daphnia may make up a high proportion of the diet in ponds, reservoirs and lakes, and in the mainstems of large rivers ((Koehler, Fresh et al. 2006), (Rondorf, Gray et al. 1990), (Sommer, Nobriga et al. 2001)). At periods, swarming terrestrial insects such as ants can make up a substantial portion of the diet of Chinook salmon rearing in floodplains, ponds and reservoirs ((Rondorf, Gray et al. 1990). In estuaries, scuds, mysids, and gammarid amphipods may be major prey (Miller and Simenstad 1997).

Studies of stream habitat use show that there are velocity thresholds for rearing fry and juveniles, that fish move to faster and deeper water as they grow, and that fish use substrate and cover as refuge from high velocities ((Chapman and Bjornn 1969), (Everest and Chapman 1972), (Johnson, Thedinga et al. 1992)). In the mainstem of large rivers and in lakes, fry and juveniles rear along the river margins and in nearshore areas that are

less than one meter deep and have low lateral bank slopes ((Sergeant and Beauchamp 2006), (Tiffan, Clark et al. 2006)). Juveniles tend to avoid the elevated water velocities found in the thalweg of river channels. As they grow larger, their habitat preferences change; juveniles move away from stream margins and begin to use deeper water ((Everest and Chapman 1972), (Tabor, Gearns et al. 2006)). When the river channel is greater than 9- to 10-ft in depth, juvenile salmon tend to inhabit the surface waters (Healey 1982).

Chinook salmon fry may also move into non-natal tributaries (*i.e.*, streams other than those where they incubated) to rear ((Limm and Marchetti 2009), (Teel, Baker et al. 2009)). In both the Columbia River and Sacramento River, California, fry and juveniles move into seasonally inundated floodplains and off-channel water bodies to rear as they move downstream ((Limm and Marchetti 2009), (Sommer, Nobriga et al. 2001), (Teel, Baker et al. 2009)). However, Chinook salmon use of floodplain and off-channel habitat depend on availability of these habitats, the life history of the race, time of year, flow, and temperatures. Up to a certain limit, distribution in floodplain habitat is positively correlated with water temperatures ((Limm and Marchetti 2009), (Sommer, Nobriga et al. 2001), (Teel, Baker et al. 2009)). Floodplain wetlands and off-channel habitat also often have higher prey densities. Several studies have shown that fry rearing on large floodplains experience a higher growth rate, and possibly higher survival, than fry remaining in the main channel ((Jeffres, Opperman et al. 2008), (Limm and Marchetti 2009), (Sommer, Nobriga et al. 2001)). The increased growth rate is likely caused by the higher water temperatures as well as the higher prey densities in these habitats. Having sufficient growth during the juvenile stage is critical as some studies indicate that size at smolting influence survival during the first year in the ocean. As flow decreases and water temperature increases in summer, juveniles move out of the inundated floodplain habitat or succumb to lethal temperatures and stranding.

Many Chinook salmon populations use the estuary intensively for rearing, and a downstream movement of large numbers of fry is typical for many populations ((Reimers

1973), (Sazaki 1966), (Thorpe 1994)). Estuaries can provide a productive environment and additional growth, refuge from predators, and a transition to marine waters; availability of unmodified estuaries is correlated with difference between rivers in survival of hatchery reared fish from smolt to maturity (Magnusson and Hilborn 2003). Ocean-type Chinook salmon migrate downstream as fry immediately after emerging from spawning beds (Healey 1991). These smaller fry and sub-yearlings extensively use shallow water habitat and sloughs within the estuary to rear to the smolt stage (Fresh, Casillas et al. 2005). Yearling juveniles of the river-type life history enter the estuaries at the smolting stage; they usually spend less time in estuaries and use deeper water than fry or sub-yearlings (Fresh, Casillas et al. 2005).

Upon entering the marine environment, immature Chinook salmon maintain close proximity to nearshore areas. The highest ocean mortality of immature Chinook salmon occurs during the first year after entering the ocean. Expected survival during this period depends both on the condition of the fish such as size and the physical conditions of the marine environment. Ocean condition such as coastal upwelling and atmospheric condition such as El Niño have a significant influence on returning run size. Because of the annual variability in ocean and climatic conditions, the stock-recruitment relationship in Chinook salmon is weak.

Immature Chinook salmon of the ocean- and river-type may have different dispersal and migration patterns during their first marine year (Healey 1991). The larger stream-type immature fish disappear from the surface waters of the Strait of Georgia in early summer. In contrast, during their first ocean year, ocean-type fish are abundant in the sheltered surface waters and estuaries of the Strait of Georgia and the Puget Sound from July through November and some continue to be present throughout winter. Estuaries provide the only shelter along the open coasts of Washington, Oregon, and California; in these areas, ocean-type fry remain longer in their native estuaries. After ocean entry, immature Chinook salmon may move into large estuaries and bays as they migrate along the coast. Chinook salmon remain at sea for one to six years (more commonly two to four years),

with the exception of a small proportion of yearling males (called jack salmon) which mature in fresh water or return after two or three months in salt water.

Status and Trends

Chinook salmon face natural threats from flooding, changes in ocean productivity, and predation. Chinook salmon have declined from overharvests, loss of genetic integrity by mixing with hatchery reared fish, retracted distribution by migration barriers such as dams, mortality and loss of rearing habitat from gravel mining, degradation of riparian habitat, and modified stream function and reduced water quality from land use practices (logging, agriculture, and urbanization).

Climate change also poses significant hazards to the survival and recovery of salmonids. They included elevated water temperature, earlier spring runoff and lower summer flows, and winter flooding.

Puget Sound Chinook Salmon

The Puget Sound ESU (Figure 5) includes all runs of Chinook salmon in the Puget Sound region from the North Fork Nooksack River to the Elwha River on the Olympic Peninsula. Thirty-six hatchery populations were included as part of the ESU and five were considered essential for recovery and listed (Table 12). They were spring Chinook salmon from Kendall Creek, the North Fork Stillaguamish River, White River, and Dungeness River, and fall run fish from the Elwha River. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within the ESU.

Table 12. Puget Sound Chinook salmon - preliminary population structure, abundances, and hatchery contributions (Good et al 2005).

Independent Populations	Historical Abundance	Mean Number of Spawners	Hatchery Abundance Contributions
Nooksack-North Fork	26,000	1,538	91%

Independent Populations	Historical Abundance	Mean Number of Spawners	Hatchery Abundance Contributions
Nooksack-South Fork	13,000	338	40%
Lower Skagit	22,000	2,527	0.2%
Upper Skagit	35,000	9,489	2%
Upper Cascade	1,700	274	0.3%
Lower Sauk	7,800	601	0%
Upper Sauk	4,200	324	0%
Suiattle	830	365	0%
Stillaguamish-North Fork	24,000	1,154	40%
Stillaguamish-South Fork	20,000	270	Unknown
Skykomish	51,000	4,262	40%
Snoqualmie	33,000	2,067	16%
Sammamish	Unknown	Unknown	Unknown
Cedar	Unknown	327	Unknown
Duwamish/Green			
Green	Unknown	8,884	83%
White	Unknown	844	Unknown
Puyallup	33,000	1,653	Unknown
Nisqually	18,000	1,195	Unknown
Skokomish	Unknown	1,392	Unknown
Mid Hood Canal Rivers			
Dosewallips	4,700	48	Unknown
Duckabush	Unknown	43	Unknown
Hamma Hamma	Unknown	196	Unknown
Mid Hood Canal	Unknown	311	Unknown
Dungeness	8,100	222	Unknown
Elwha	Unknown	688	Unknown

Life History

Puget Sound Chinook salmon populations exhibit both early-returning (August) and late-returning (mid-September and October) Chinook salmon spawners (Healey 1991).

Juvenile Chinook salmon within the Puget Sound generally exhibit an “ocean-type” life history. However, substantial variation occurs with regard to juvenile residence time in freshwater and estuarine environments. Hayman (Hayman, Beamer et al. 1996) described three juvenile life histories for Chinook salmon with varying freshwater and estuarine residency times in the Skagit River system in northern Puget Sound. In this system, 20% to 60% of sub-yearling migrants rear for several months in freshwater habitats while the remaining fry migrate to rear in the Skagit River estuary and delta (Beamer, Hayman et al. 2005). Juveniles in tributaries to Lake Washington exhibit both

a stream rearing and a lake rearing strategy. Lake rearing fry are found in highest densities in nearshore shallow (<1 m) habitat adjacent to the opening of tributaries or at the mouth of tributaries where they empty into the lake (Tabor, Gearns et al. 2006). Puget Sound Chinook salmon also has several estuarine rearing juvenile life history types that are highly dependent on estuarine areas for rearing (Beamer, Hayman et al. 2005). In the estuaries, fry use tidal marshes and connected tidal channels including dikes and ditches developed to protect and drain agricultural land. During their first ocean year, immature Chinook salmon use nearshore areas of Puget Sound during all seasons and can be found long distances from their natal river systems (Brennan, Higgins et al. 2004).

Puget Sound Chinook ESU Sub-Basin Range and Distribution



Figure 5. Puget Sound Chinook salmon distribution.

Status and Trends

NMFS listed Puget Sound Chinook salmon as threatened in 1999 (64 FR 14308) and reaffirmed its status as threatened on June 28, 2005 (70 FR 37160). Historically, the ESU included 31 rivers or river systems that supported historic independent populations. Of the historic populations, only 22 are extant (Ruckelshaus, Currens et al. 2006) (Table 12). A disproportionate loss of an early-run life history represents a significant loss of the evolutionary legacy of the ESU (Ruckelshaus, Currens et al. 2006).

The spatial structure of the ESU is compromised by extinct and weak populations being disproportionably distributed to the mid- to southern Puget Sound and the Strait of Juan de Fuca. A large portion (at least 11) of the extant runs is sustained, in part, through artificial propagation. Of the populations with greater than 1,000 natural spawners, only two have a low fraction of hatchery fish. Populations known to contain significant natural production are found in the northwest Puget Sound.

Estimates of the historic abundance range from 1,700 to 51,000 potential Puget Sound Chinook salmon spawners per population. During the period from 1996 to 2001, the geometric mean of natural spawners in populations of Puget Sound Chinook salmon ranged from 222 to just over 9,489 fish. Thus, the historical estimates of spawner capacity are several orders of magnitude higher than spawner abundances currently observed throughout the ESU (Good, Waples et al. 2005). Long-term trends in abundance and median population growth rates for naturally spawning populations indicate that approximately half of the populations are declining and the other half are increasing in abundance over the length of available time series. However, the median overall long-term trend in abundance is close to 1 for most populations that have a lambda exceeding 1, indicating that most of these populations are barely replacing themselves. Eight of 22 populations are declining over the short-term, compared to 11 or 12 populations that have long-term declines (Good, Waples et al. 2005). Populations with the greatest long-term population growth rates are the North Fork Nooksack and White rivers.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). It includes 1,683 km of stream channels, 41 square km of lakes, and 3,512 km of nearshore marine habitat. Of 61 watersheds (5th field Hydrological Units or HUC 5) reviewed in NMFS' assessment of critical habitat for the Puget Sound ESU, 9 watersheds were rated as having a medium conservation value, 12 were rated as low, and the remaining watersheds (40), where the bulk of federal lands overlap with this ESU, were rated as having a high conservation value for Puget Sound Chinook salmon (Figure 6). The 19 nearshore marine areas were all given a high conservation value rating. (Table 13).

Table 13. Puget Sound Chinook salmon watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Strait of Georgia	0		0		3	(3, 1, 2)
Nooksack	4	(1, 3, 2)	1	(3, 1)	0	
Upper Skagit	4	(1, <3)	1	(3)	0	
Sauk	4	(1, 2, 3)	0		0	
Lower Skagit	2	(3, 1, 2)	0		0	
Stillaguamish	3	(1, 3)	0		0	
Skykomish	5	(1, 3)	0		0	
Snoqualmie	2	(1, 3, 2)	0		0	
Snohomish	1	(1,2,3)	1	(1, 2, 3)		
Lake Washington	1	(1)	3	(1, 3, <2)	0	
Duwamish	2	(3, 1, 2)	1	(3)	0	
Puyallup	5	(3, 2, 1)	0		0	
Nisqually	2	(1, <3)	0		0	
Deschutes	0		0		2	(1, 3)
Skokomish	1	(1, 3)	0		0	
Hood Canal	2	(1)	1	(1)	3	(1, <3,<2)
Kitsap	0		0		4	(3, 1)
Dungeness/Elwha	2	(1)	1	(3, 1)	0	
Totals	40		9		12	

¹ Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

Forestry practices have heavily impacted migration, spawning, and rearing PCEs in the upper watersheds of most rivers systems within critical habitat designated for the Puget Sound Chinook salmon. Degraded PCEs include reduced conditions of substrate supporting spawning, incubation and larval development caused by siltation of gravel; and degraded rearing habitat by removal of cover and reduction in channel complexity. Urbanization and agriculture in the lower alluvial valleys of mid- to southern Puget Sound and the Strait of Juan de Fuca have reduced channel function and connectivity, reduced available floodplain habitat, and affected water quality. Thus, these areas have degraded spawning, rearing, and migration PCEs. Hydroelectric development and flood control also obstruct Puget Sound Chinook salmon migration in several basins. The most functional PCEs are found in northwest Puget Sound: the Skagit River basin, parts of the Stillaguamish River basin, and the Snohomish River basin where federal land overlap with critical habitat designated for the Puget Sound Chinook salmon. However, estuary PCEs are degraded in these areas by reduction in the water quality from contaminants, altered salinity conditions, lack of natural cover, and modification and lack of access to tidal marshes and their channels.

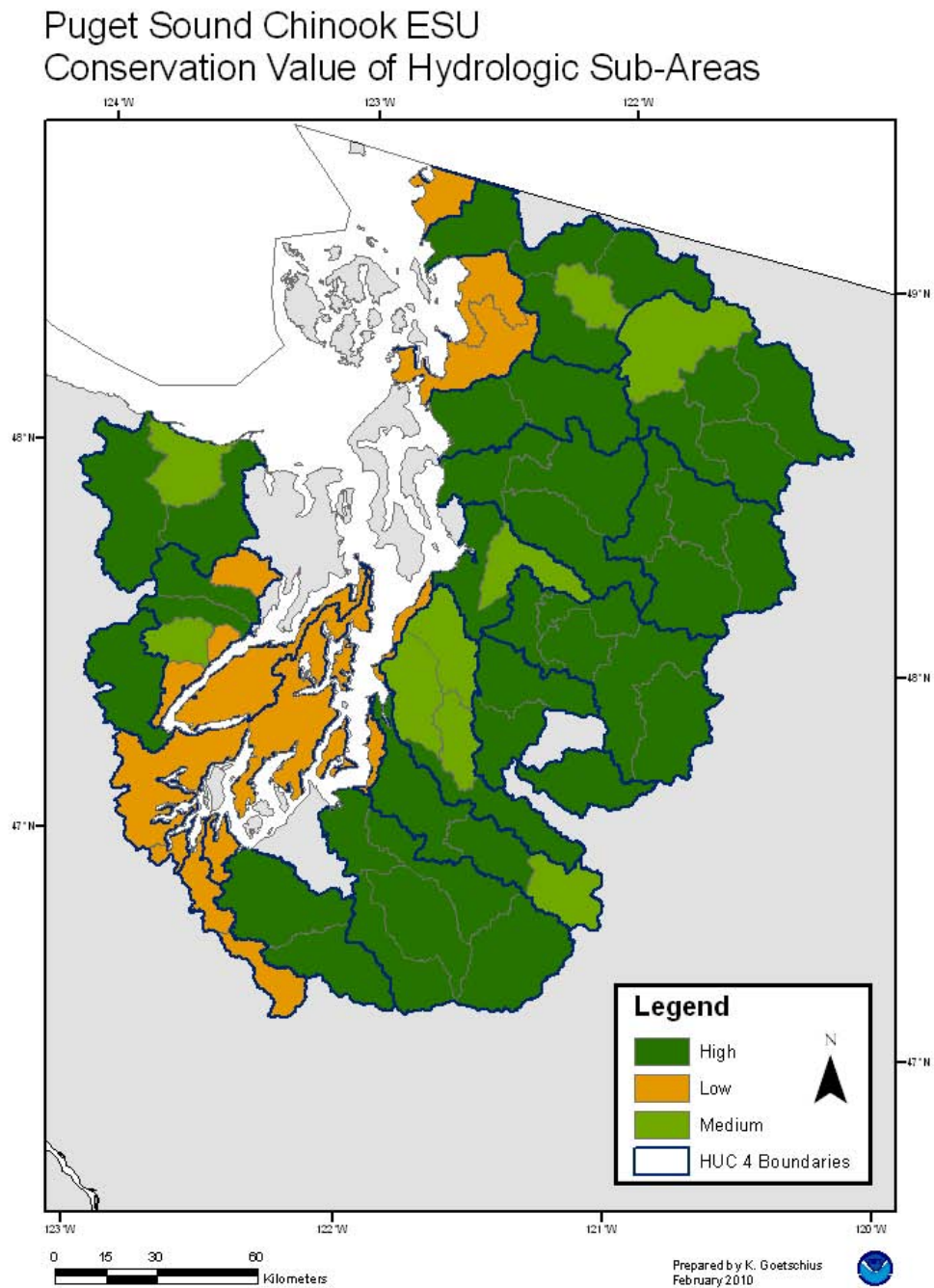


Figure 6. Puget Sound Chinook salmon Conservation Values per Sub-watershed.

Lower Columbia River Chinook Salmon

The Lower Columbia River (LCR) Chinook salmon ESU (Figure 7) includes all naturally-spawned populations of fall-run and spring-run Chinook salmon from the Columbia River and its tributaries from its mouth at the Pacific Ocean upstream to a transitional point between Oregon and Washington, east of the Hood River and the White Salmon River. The eastern boundary for this species occurs at Celilo Falls, which corresponds to the edge of the drier Columbia Basin Ecosystem. It also includes the Willamette River to Willamette Falls, Oregon, exclusive of spring-run Chinook salmon in the Clackamas River. Seventeen artificial propagation programs are included in the ESU (70 FR 37160). These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Lower Columbia River Chinook ESU Sub-Basin Range and Distribution

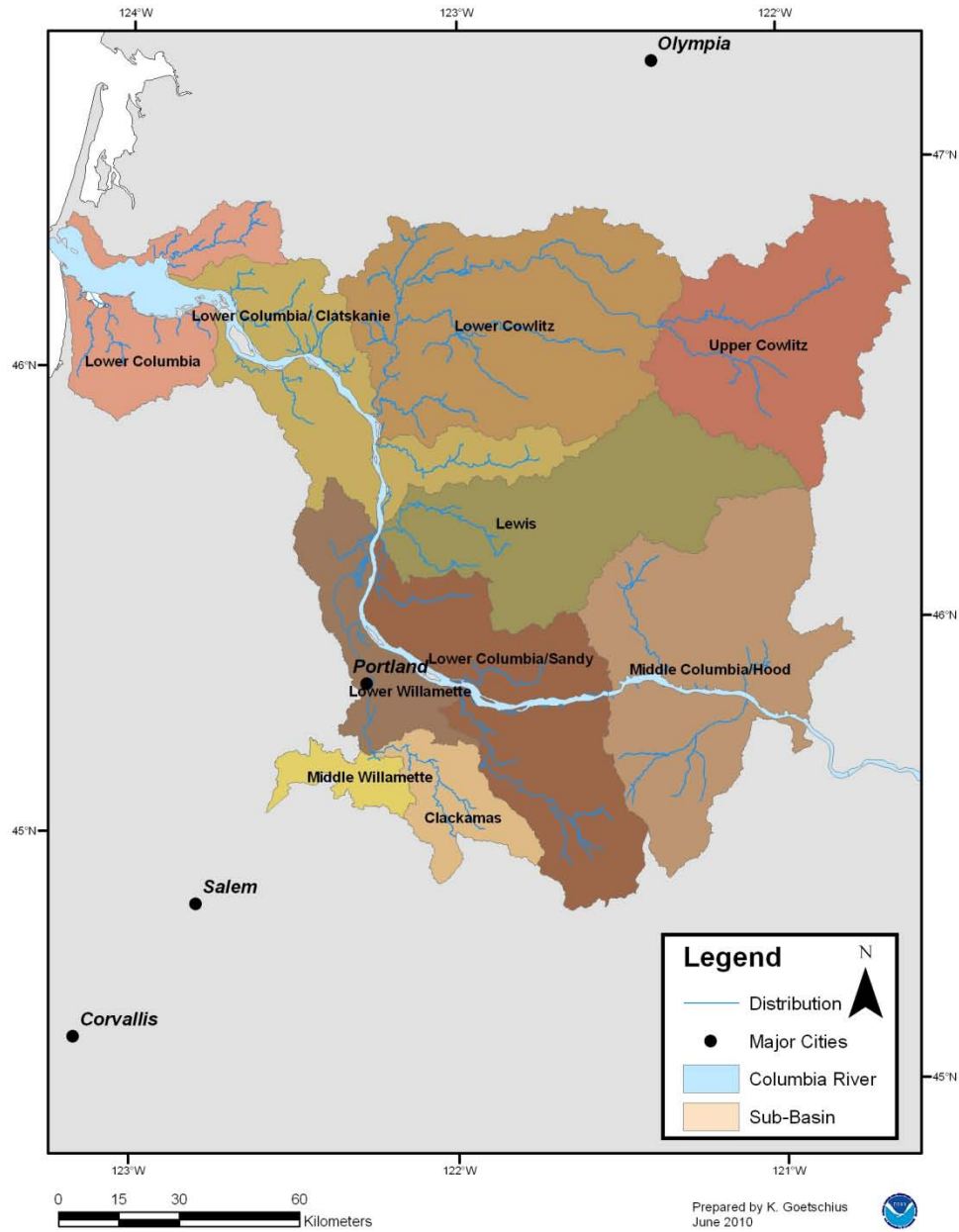


Figure 7. Lower Columbia River Chinook salmon distribution.

Life History

LCR Chinook salmon display three run types including early fall-runs, late fall-runs, and spring-runs. Presently, the fall-run is the predominant life history type. Spring-run Chinook salmon were numerous historically. Fall-run Chinook salmon enter fresh water typically in August through October. Early fall-run spawn within a few weeks in large river mainstems. The late fall-run enters in immature conditions, has a delayed entry to spawning grounds, and resides in the river for a longer time between river entry and spawning. Spring-run Chinook salmon enter fresh water in March through June to spawn in upstream tributaries in August and September.

Offspring of fall-run spawning may migrate as fry to the ocean soon after yolk absorption (*i.e.*, ocean-type), at 30–45 mm in length (Healey 1991). In the Lower Columbia River system, however, the majority of fall-run Chinook salmon fry migrate either at 60-150 days post-hatching in the late summer or autumn of their first year. Offspring of fall-run spawning may also include a third group of yearling juveniles that remain in fresh water for their entire first year before emigrating. The spring-run Chinook salmon migrates to the sea as yearlings (stream-type) typically in spring. However, the natural timing of LCR spring-run Chinook salmon emigration is obscured by hatchery releases (Myers, Busack et al. 2006).

Once at sea, the ocean-type LCR Chinook salmon tend to migrate along the coast, while stream-type LCR Chinook salmon appear to move far off the coast into the central North Pacific Ocean ((Healey 1991), (Myers, Busack et al. 2006). Adults return to tributaries in the lower Columbia River predominately as three- and four-year-olds for fall-run fish and four- and five-year-olds for spring-run fish.

Status and Trends

NMFS originally listed LCR Chinook salmon as threatened on March 24, 1999 (64 FR 14308), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). Thirty-one independent Chinook salmon populations – 22 fall- and late fall-runs and 9 spring-

runs – are estimated to have existed historically in the Lower Columbia River (Myers, Busack et al. 2006). The Willamette/Lower Columbia River Technical Review Team (W/LCRTRT) has estimated that 8-10 historic populations have been extirpated, most of them spring-run populations. The fall-run Chinook salmon historically occurred throughout the Lower Columbia River basin, while spring-run Chinook salmon only occurred in the upper portions of Lower Columbia Basins that consist of snowmelt driven flow regimes. The Cowlitz, Kalama, Lewis, White Salmon, and Klickitat Rivers are the major river systems on the Washington side, and the lower Willamette and Sandy Rivers are foremost on the Oregon side.

The basin wide spatial structure has remained generally intact. However, the loss of about 35% of historic habitat has affected distribution within several Columbia River subbasins. Currently, only one population appears self sustaining (Good, Waples et al. 2005). Table 14 identifies populations within the LCR Chinook salmon ESU, their abundances, and hatchery input.

Table 14. Lower Columbia River Chinook salmon - population structure, abundances, and hatchery contributions (Good et al 2005, Myers et al 2006).

Run	Population	Historical Abundance	Mean Number of Spawners	Hatchery Abundance Contributions
F-R	Grays River (WA)	2,477	99	38%
	Elochoman River (WA)	Unknown	676	68%
	Mill, Abernathy, and German Creeks (WA)	Unknown	734	47%
	Youngs Bay (OR)	Unknown	Unknown	Unknown
	Big Creek (OR)	Unknown	Unknown	Unknown
	Clatskanie River (OR)	Unknown	50	Unknown
	Scappoose Creek (OR)	Unknown	Unknown	Unknown
F-R	Lower Cowlitz River (WA)	53,956	1,562	62%
	Upper Cowlitz River (WA)	Unknown	5,682	Unknown
	Coweeman River (WA)	4,971	274	0%
	Toutle River (WA)	25,392	Unknown	Unknown
	Salmon Creek and Lewis River (WA)	47,591	256	0%
	Washougal River (WA)	7,518	3,254	58%
	Kalama River (WA)	22,455	2,931	67%
	Clackamas River (OR)	Unknown	40	Unknown
	Sandy River (OR)	Unknown	183	Unknown

Run	Population	Historical Abundance	Mean Number of Spawners	Hatchery Abundance Contributions
LF-R	Lewis R-North Fork (WA)	Unknown	7,841	13%
	Sandy River (OR)	Unknown	504	3%
S-R	Upper Cowlitz River (WA)	Unknown	Unknown	Unknown
	Tilton River (WA)	Unknown	Unknown	Unknown
	Cispus River (WA)	Unknown	1,787*	Unknown
	Toutle River (WA)	2,901	Unknown	Unknown
	Kalama River (WA)	4,178	98	Unknown
	Lewis River (WA)	Unknown	347	Unknown
	Sandy River (OR)	Unknown	3,085	3%
F-R	Upper Columbia Gorge (WA)	2,363	136	13%
	Big White Salmon R (WA)	Unknown	334	21%
	Lower Columbia Gorge (OR)	Unknown	Unknown	Unknown
	Hood River (OR)	Unknown	18	Unknown
S-R	Big White Salmon R (WA)	Unknown	334	21%
	Hood River (OR)	Unknown	18	Unknown

*Arithmetic mean

Recent 5-year spawner abundance (up to 2001) and historic abundance over more than 20 years is given as a geometric mean, and include hatchery origin Chinook salmon.

F-R is fall run, LF-R is late fall run, and S-R is spring run Chinook salmon.

Historical records of Chinook salmon abundance are sparse. However, cannery records suggest a peak run of 4.6 million fish [43 million lbs see (Lichatowich 1999) in 1883]. Historically, the number of spring-run Chinook salmon returning to the Lower Columbia River may have almost equaled that of fall-run Chinook salmon (Myers, Busack et al. 2006). Today, the majority of spring-run LCR Chinook salmon populations are extirpated and total returns are substantially lower than for the fall-run component.

Trend indicators for most populations are negative. The majority of populations for which data are available have a long-term trend of <1 ; indicating the population is in decline ((Bennet 2005), (Good, Waples et al. 2005)). Only the late-fall run population in Lewis River has an abundance and population trend that may be considered viable (McElhany, Chilcote et al. 2007). The Sandy River is the only stream system supporting a natural production of spring-run Chinook salmon of any amount. However, the population is at risk from low abundance and negative to low population growth rates (McElhany, Chilcote et al. 2007).

The genetic diversity of all populations (except the late fall-run Chinook salmon) has been eroded by large hatchery influences and periodically by low effective population sizes. The near loss of the spring-run life history type remains an important concern for maintaining diversity within the ESU.

The ESU is at risk from generally low abundances in all but one population, combined with most populations having a negative or stagnant long-term population growth. However, fish from conservation hatcheries do help to sustain several LCR Chinook salmon runs in the short-term though this is unlikely to result in sustainable wild populations in the long-term. Having only one population that may be viable puts the ESU at considerable risk from environmental stochasticity and random catastrophic events. The loss of life history diversity limits the ESU's ability to maintain its fitness in the face of environmental change.

Critical Habitat

NMFS designated critical habitat for LCR Chinook salmon on September 2, 2005 (70 FR 52630). It includes all Columbia River estuarine areas and river reaches proceeding upstream to the confluence with the Hood Rivers as well as specific stream reaches in a number of tributary subbasins.

As shown in Figure 8, of the watersheds (HUC 5s) reviewed in NMFS' assessment of critical habitat for the LCR Chinook salmon ESU, 13 subbasins were rated as having a medium conservation value, four were rated as low, and the remaining subbasins (31), were rated as having a high conservation value to LCR Chinook salmon (Table 15). Additionally, four watersheds were given a "possibly high" rating, *i.e.*, they may be essential to conservation of the species but are currently unoccupied.

Table 15. LCR Chinook salmon HUC 5 watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Middle-Columbia/Hood	6	(1)	2	(3)	0	
Lower Columbia/Sandy	7	(1, 3)	1	(3, 1)	1	(3)
Lewis	2	(1, 2, 3)	0		0	
Lower Columbia/Clatskanie	2	(3, 1)	3	(3, 2)	1	(2)
Upper Cowlitz River	5	(3)	0		0	
Lower Cowlitz	4	(3, 1)	4	(3, 1)	0	
Lower Columbia	2	(3, 1)	1		0	
Middle Willamette	0		0		1	(2)
Clackamas	1	(1)	0		1	
Lower Willamette	1	(2)	2	(2)	0	
Lower Columbia Corridor	1	(3)	0		0	
Total	31		13		12	

¹ Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

Timber harvest, agriculture, and urbanization have degraded spawning and rearing PCEs by reducing floodplain connectivity and water quality, and by removing natural cover in several rivers. Hydropower development projects have reduced timing and magnitude of water flows, thereby altering the water quantity needed to form and maintain physical habitat conditions and support juvenile growth and mobility. Adult and juvenile migration PCEs are affected by several dams along the migration route.

Lower Columbia River Chinook ESU
Conservation Value of HUC 5 Watersheds

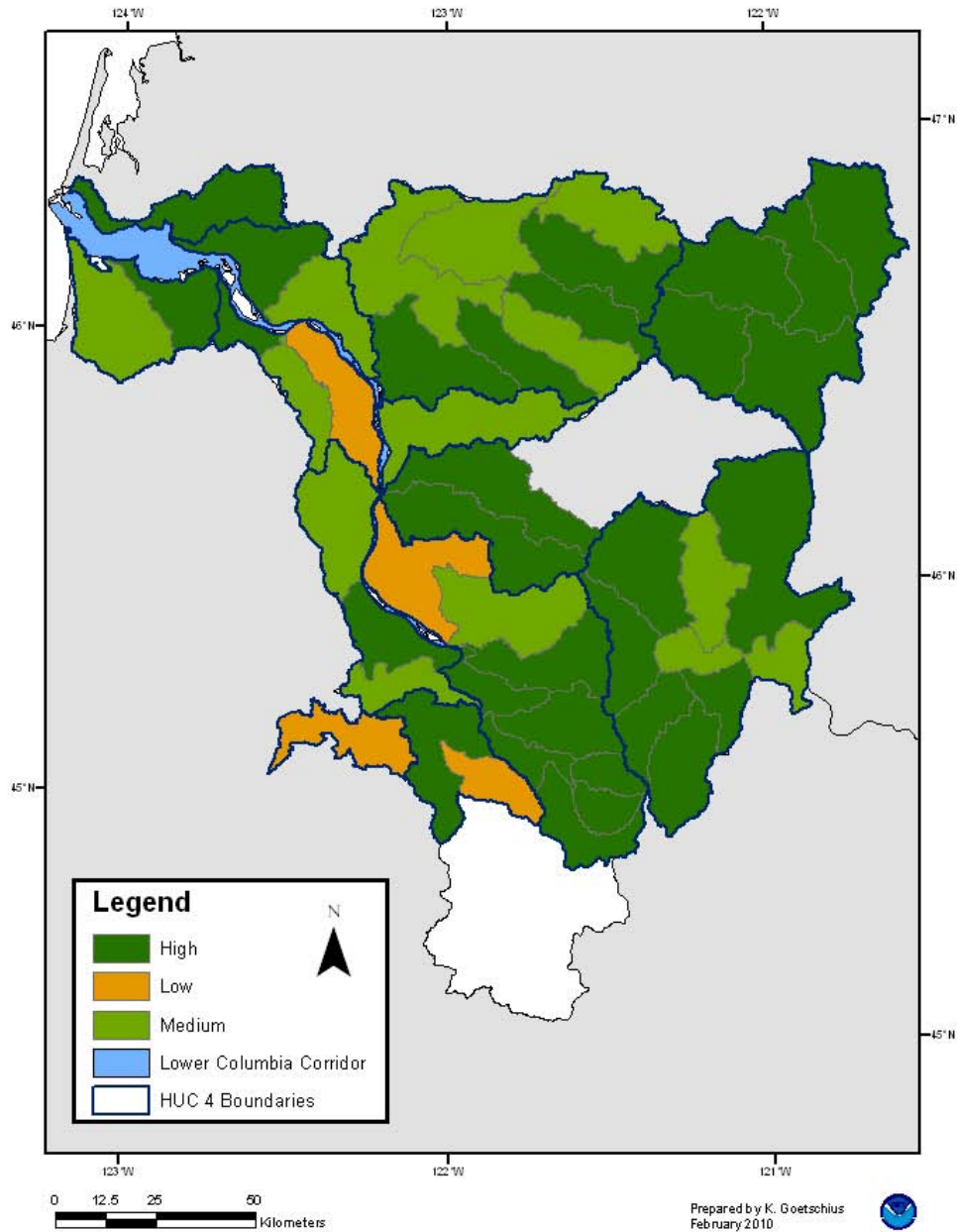


Figure 8. Lower Columbia River Chinook salmon Conservation Values per Sub-Area.

Upper Columbia River Spring-run Chinook Salmon

The Upper Columbia River (UCR) Spring-run Chinook salmon ESU includes all naturally spawned populations of spring-run Chinook salmon in all Columbia River tributaries upstream of the Rock Island Dam and downstream of Chief Joseph Dam in Washington State. Major tributary subbasins with existing runs are the Wenatchee, Entiat, and Methow Rivers (Figure 9).

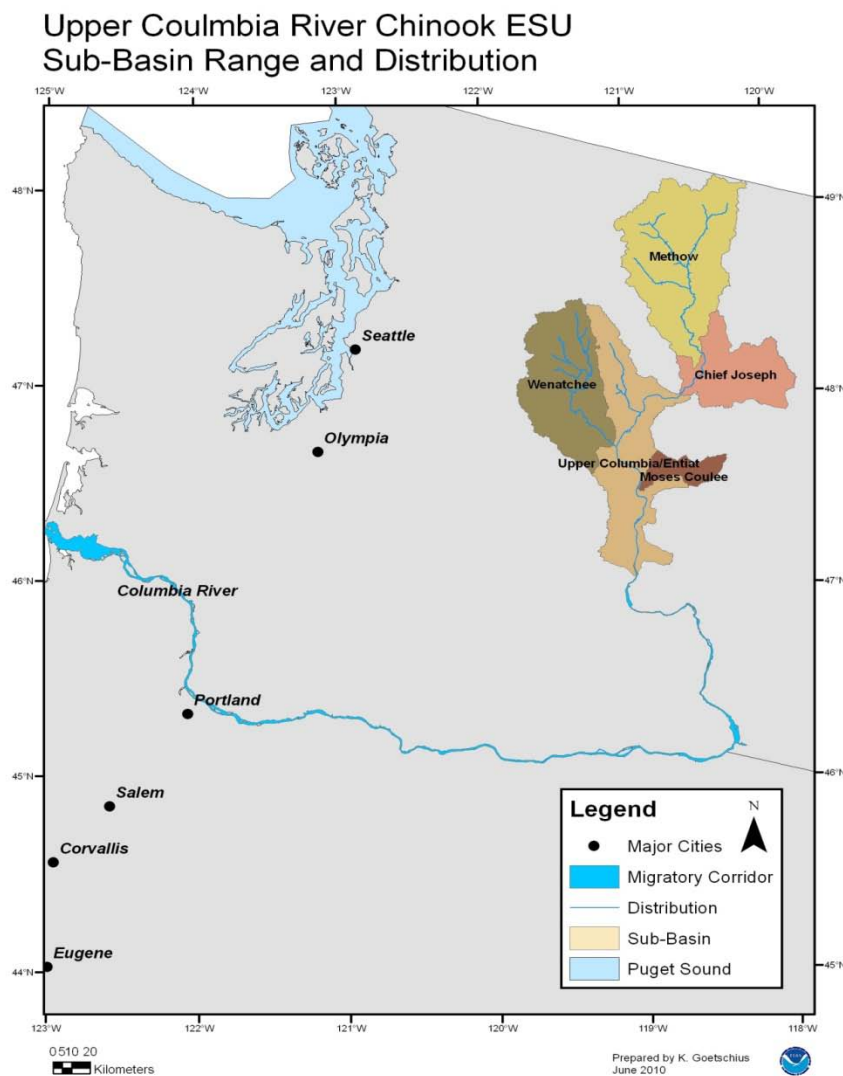


Figure 9. Upper Columbia River Chinook salmon distribution.

Several hatchery populations are also listed (70 FR 37160). These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Life History

UCR Spring-run Chinook salmon begin returning from the ocean in the early spring. They enter the upper Columbia tributaries from April through July, with the run peaking in mid-May. After migration, UCR Spring-run Chinook salmon hold in freshwater tributaries until spawning occurs in the late summer, peaking in mid- to late August. Juvenile spring-run Chinook salmon spend a year in fresh water before emigrating to salt water in the spring of their second year.

Status and Trends

NMFS listed UCR Spring-run Chinook salmon as endangered on March 24, 1999 (64 FR 14308), and reaffirmed their endangered status on June 28, 2005 (70 FR 37160). The ESU consisted of four populations. Of these, one is now extinct and three are extant. The Interior Columbia Basin Technical Review Team (ICBTRT) characterizes the spatial structure risk to UCR Spring-run Chinook populations as “low” or “moderate.” Table 16 identifies populations within the UCR Spring-run Chinook salmon ESU, their abundances, and hatchery input.

Table 16. Upper Columbia River Spring-run Chinook salmon - preliminary population structure, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Mean Number of Spawners (Range) ^a	Hatchery Abundance Contributions
Methow River	~2,100	680 (79-9,9-04)	59%
Twisp River	Unknown	58 redds (10-369)	54%
Chewuch River	Unknown	58 redds (6-1,105)	41%
Lost/Early River	Unknown	12 (3-164)	54%
Entiat River	~380	111 (53-444)	42%
Wenatchee River	~2,400	470 (119 -	42%

		4,446)	
Chiwawa River	Unknown	109 redds (34-1,046)	47%
Nason Creek	Unknown	54 redds (8-374)	39%
Upper Wenatchee River	Unknown	8 redds (0-215)	66%
White River	Unknown	9 redds (1-104)	8%
Little Wenatchee River	Unknown	11 redds (3-74)	21%
Okanogan River	Unknown	Extirpated	NA

^a 5-year geometric mean number of spawners unless otherwise noted; includes hatchery fish. Range denoted in parenthesis. Means calculated from years 1997 to 2001, except Lost/Early Winter creeks did not include 1998 as no data were available. Data reported in Good et al 2006.

For all populations, average abundance over the recent 10-year period is below the average abundance thresholds that the ICBTRT identifies as a minimum for low risk ((ICTRT 2008; ICTRT 2008; ICTRT 2008)). The geometric mean spawning escapements from 1997 to 2001 were 273 for the Wenatchee population, 65 for the Entiat population, and 282 for the Methow population. These numbers represent only 8% to 15% of the minimum abundance thresholds. The five-year geometric mean remained low as of 2003. Recently, the 2007 UCR spring Chinook jack counts, an indicator of future adult returns, have increased to their highest level since 1977.

Based on 1980-2004 returns, the lambda for this ESU is estimated at 0.93 (meaning the population is not replacing itself) (Fisher and Hinrichsen 2006). The long-term trend for abundance and lambda for individual populations indicate a decline for all three populations (Good, Waples et al. 2005). Short-term lambda values indicate an increasing trend for the Methow population, but not for the Wenatchee and Entiat populations ((ICTRT 2008; ICTRT 2008; ICTRT 2008)).

Finally, the ICBTRT characterizes the diversity risk to all UCR Spring-run Chinook populations as “high”. The high risk is a result of reduced genetic diversity from homogenization of populations that occurred under the Grand Coulee Fish Maintenance Project in 1939-1943.

Abundance data showed an increase in spawner returns in 2000 and 2001 (Good, Waples et al. 2005). However, this increase did not manifest itself in subsequent years. Thus, recent available data on population viability suggest that the ESU continues to be at high risk from small population size; all three UCR Spring-run Chinook salmon populations are affected by low abundances and failing recruitment. Should population growth rates continue at the 1980-2004 levels, UCR Spring-run Chinook salmon populations have a high probability of decline within 50 years. The genetic integrity of all populations has been compromised by periods of low effective population size and low proportion of natural-origin fish.

Critical Habitat

NMFS designated critical habitat for UCR Spring-run Chinook salmon on September 2, 2005 (70 FR 52630). It includes all Columbia River estuarine areas and river reaches proceeding upstream to Chief Joseph Dam and several tributary subbasins.

The UCR Spring-run Chinook salmon ESU has 31 watersheds within its range. Five watersheds received a medium rating and 26 received a high rating of conservation value to the ESU (Table 17). The Columbia River rearing/migration corridor downstream of the spawning range was rated as having a high conservation value (Figure 10).

Spawning and rearing PCEs are somewhat degraded in tributary systems by urbanization in lower reaches, grazing in the middle reaches, and irrigation and diversion in the major upper drainages. These activities have resulted in excess erosion of fine sediment and silt that smother spawning gravel; reduction in flow quantity necessary for successful incubation, formation of physical rearing conditions, and juvenile mobility. Moreover siltation further affects critical habitat by reducing water quality through contaminated agricultural runoff; and removing natural cover. Adult and juvenile migration PCEs are heavily degraded by Columbia River Federal dam projects and a number of mid-Columbia River Public Utility District dam projects also obstruct the migration corridor.

Table 17. UCR Spring-run Chinook salmon watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Chief Joseph	1	(3)	0		0	0
Methow	5	(1, <2, <3)	2	(1, 2)	0	
Upper Columbia/Entiat	3	(3, 2 ² , 1 ²)	1	(3)	0	
Wenatchee	3	(1, 2, <3)	2	(2, 1)	0	
Moses Coulee	1	(1, =0.8mi)	0		0	
Upper Columbia/Priest Rapids	3	(3)	0		0	
Middle Columbia/Lake Wallula	5	(3)	0		0	
Middle Columbia/Hood	4	(3)	0		0	
Lower Columbia/Sandy	1	(3)	0		0	
Lower Columbia Corridor	all	(3) ³	0		0	
Total	26		5		0	

1 Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

2 Only one of the three watersheds, Entiat River, had PCEs 1 and 2.

3 The Lower Columbia Corridor includes 46.5 miles of estuarine PCEs.

Upper Columbia River Chinook ESU
Conservation Value of HUC 5 Watersheds

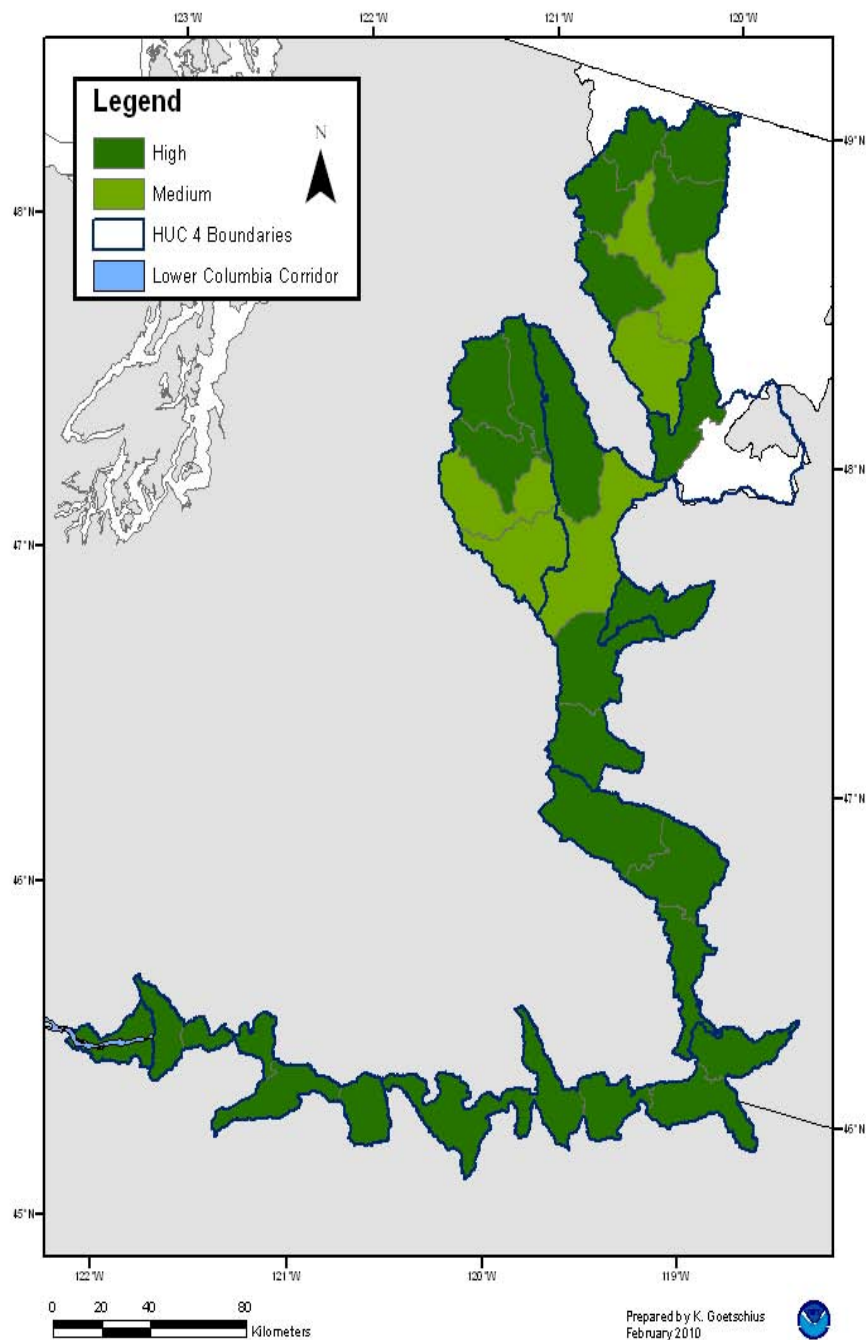


Figure 10. Upper Columbia River Spring-run Chinook salmon Conservation Values per Sub-Area

Snake River Fall-run Chinook Salmon

The Snake River (SR) Fall-run Chinook salmon ESU (Figure 11) includes all naturally spawned populations of fall-run Chinook salmon in the mainstem Snake River below Hells Canyon Dam, and in the Tucannon River, Grande Ronde River, Imnaha River, Salmon River, and Clearwater River subbasins (70 FR 37176,). Four artificial propagation programs are included in the ESU. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

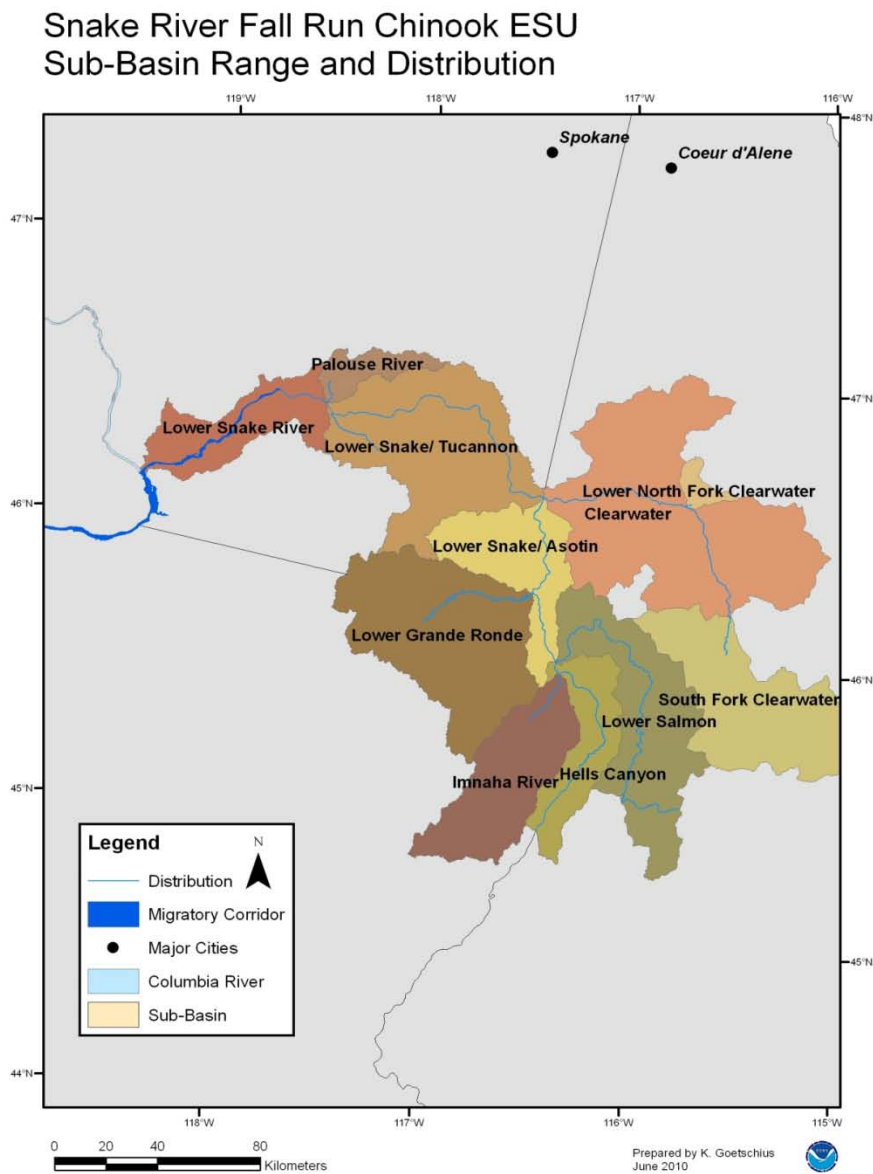


Figure 11. Snake River Fall-run Chinook salmon distribution.

Life History

Adult SR Fall-run Chinook salmon enter the Columbia River in July and migrate into the Snake River from August through October. Fall-run Chinook salmon generally spawn

from October through November. Fry emerge from redds from April through June (Tiffan, Rondorf et al. 2001). Fry rear two months or more in the sandy littoral zone along the river margins above Lower Granite Dam before passing over the dam ((Connor, Burge et al. 2002), (Smith, Muir et al. 2003)). Sub-yearling smolts pass over the Lower Granite Dam from June through October with most migration occurring from July through September (Tiffan, Rondorf et al. 2001). Sub-yearlings increase their rate of seaward movement as they progress downstream (Smith, Muir et al. 2003).

Historically, SR Fall-run Chinook salmon exhibited a largely ocean-type life history. However, as a consequence of dam construction, this ESU now resides in water that is cooler than the historic spawning areas, and alteration of the Lower Snake River by hydroelectric dams has created a series of low-velocity pools in the Snake River. Thus, Fall-run Chinook salmon in the Snake River Basin now exhibit one of two life histories: ocean-type and reservoir-type ((Connor, Sneva et al. 2005), (Tiffan, Rondorf et al. 2001)). The reservoir-type life history is one where juveniles overwinter in the reservoirs created by the dams, prior to migrating out of the Snake River. SR Fall-run Chinook salmon spend one to four years in the Pacific Ocean before beginning their spawning return migration.

Status and Trends

NMFS originally listed SR Fall-run Chinook salmon as endangered in 1992 (57 FR 14653) but reclassified their status as threatened on June 28, 2005 (70 FR 37160). The SR Fall-run Chinook salmon consists of one extant population that is mostly limited to a core spawning area within a 32-km section of the mainstem Snake River (ICTRT 2003). Two populations have been extirpated.

Estimated annual returns for the period 1938 to 1949 were at 72,000 fish. By the 1950s, numbers had declined to an annual average of 29,000 fish (Bjornn and Horner 1980). Numbers of SR Fall-run Chinook salmon continued to decline during the 1960s and 1970s as approximately 80% of their historic habitat were eliminated or severely

degraded by the construction of the Hells Canyon complex (1958 to 1967) and the lower Snake River dams (1961 to 1975). The abundance of natural-origin spawners in the SR Fall-run Chinook ESU for 2001 (2,652 adults) exceeded 1,000 fish for the first time since counts began at the Lower Granite Dam in 1975. The recent five-year mean abundance of 871 naturally produced spawners at the time of the last status review generated concern that despite recent improvements, the abundance level is very low for an entire ESU. On the other hand, during the years from 1975 to 2000, the ESU fluctuated between 500 to 1,000 natural spawners. This suggests a higher degree of stability in growth rate at low population levels than is seen in other salmonid populations. Further, numbers of natural-origin SR Fall-run Chinook salmon have increased over the last few years, with estimates at Lower Granite Dam of 2,652 fish in 2001, 2,095 fish in 2002, and 3,895 fish in 2003.

Long- and short-term trends in natural returns are positive. Productivity is likely sustained largely by a system of small artificial rearing facilities in the lower Snake River Basin. Depending upon the assumptions made regarding the reproductive contribution of hatchery fish, long- and short-term trends in productivity are at or above replacement.

Low abundances in the 1990s combined with a large proportion of hatchery derived spawners likely have reduced genetic diversity from historic levels. Nevertheless, the SR Fall-run Chinook salmon remains genetically distinct from similar fish in other basins.

As the ESU's single population spawning activities are limited to a relatively short reach of the free flowing mainstem Snake River, it is at considerable risk from environmental variability and stochastic events. The 1997 to 2001 geometric mean natural-origin count over Lower Granite Dam approximate 35% of the proposed delisting abundance criteria of 2,500 natural spawners averaged over eight years. Current observed abundances indicate that the ESU is at moderate risk from low abundances.

Critical Habitat

NMFS designated critical habitat for SR Fall-run Chinook salmon on December 28, 1993 (58 FR 68543). It includes the Columbia River reaches presently or historically accessible to listed fall-run Chinook salmon (except river reaches above impassable natural falls, and Dworshak and Hells Canyon Dams) from the estuary upstream to the confluence of the Snake River; all Snake River reaches from the confluence of the Columbia River upstream to Hells Canyon Dam. It also includes the Palouse River from its confluence with the Snake River upstream to Palouse Falls; the Clearwater River from its confluence with the Snake River upstream to its confluence with Lolo Creek; and the North Fork Clearwater River from its confluence with the Clearwater River upstream to Dworshak Dam. Designated areas consist of the water, waterway bottom, and the adjacent riparian zone (defined as an area 300 feet from the normal high water line on each side of the river channel) (58 FR 68543).

Individual watersheds within the ESU have not been evaluated for their conservation value. However, the lower Columbia River corridor is among the areas of high conservation value to the ESU because it connects every population with the ocean and is used by rearing/migrating juveniles and migrating adults. The Columbia River estuary is a unique and essential area for juveniles and adults making the physiological transition between life in freshwater and marine habitats.

Salmon habitat has been altered throughout the ESU through loss of important spawning and rearing habitat and the loss or degradation of migration corridors. The major degraded PCEs within critical habitat designated for SR Fall-run Chinook salmon include: (1) safe passage for juvenile migration which is reduced by the presence of the Snake and Columbia River hydropower system within the lower mainstem; (2) rearing habitat water quality altered by influx of contaminants and changing seasonal temperature regimes caused by water flow management; and (3) spawning/rearing habitat PCE attributes (spawning areas with gravel, water quality, cover/shelter, riparian vegetation, and space to support egg incubation and larval growth and development) that

are reduced in quantity (80% loss) and quality due to the mainstem lower Snake River hydropower system.

Water quality impairments in the designated critical habitat are common within the range of this ESU. Pollutants such as petroleum products, pesticides, fertilizers, and sediment in the form of turbidity enter the surface waters and riverine sediments from the headwaters of the Snake, Salmon, and Clearwater Rivers to the Columbia River estuary; traveling along with contaminated stormwater runoff, aerial drift and deposition, and via point source discharges. Some contaminants such as mercury and pentachlorophenol enter the aquatic food web after reaching water and may be concentrated or even biomagnified in the salmon tissue. This species also requires migration corridors with adequate passage conditions (water quality and quantity available at specific times) to allow access to the various habitats required to complete their life cycle.

Snake River Spring/Summer-Run Chinook Salmon

This ESU includes production areas that are characterized by spring-timed returns, summer-timed returns, and combinations from the two adult timing patterns. The SR Spring/Summer-run Chinook ESU includes all naturally spawned populations of spring/summer-run Chinook salmon in the mainstem Snake River and the Tucannon River, Grande Ronde River, Imnaha River, and Salmon River subbasins (57 FR 23458, Figure 12). Fifteen artificial propagation programs are included in the ESU (70 FR 37176). These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Snake River Spring-Summer Run Chinook Sub-Basin Range and Distribution

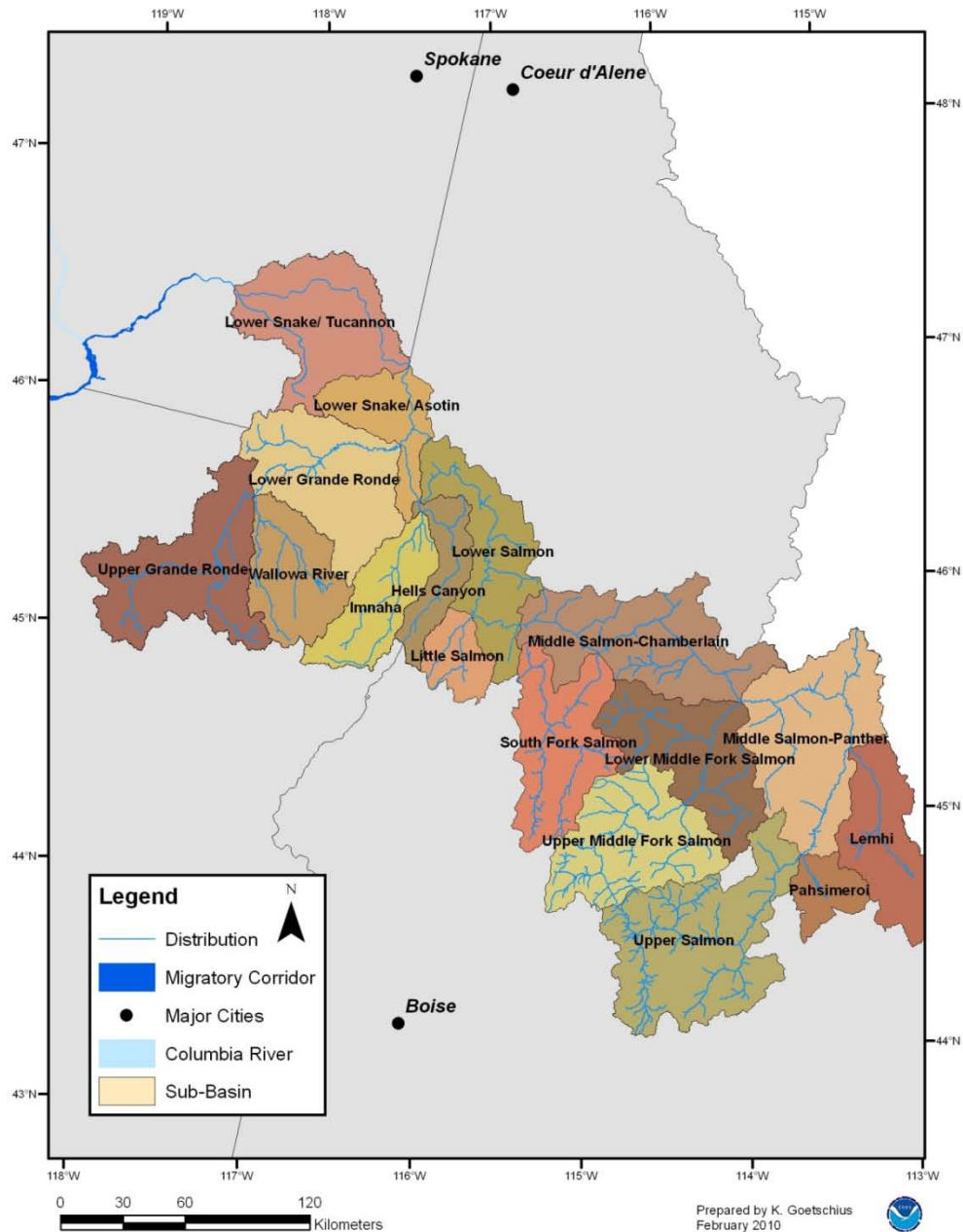


Figure 12. Snake River Spring/Summer-run Chinook salmon distribution.

Life History

Runs classified as spring-run Chinook salmon pass Bonneville Dam beginning in early March to mid-June; runs classified as summer-run Chinook salmon return to the Columbia River from June through August. SR Spring/Summer-run Chinook salmon exhibit a stream-type life history. In general, spring-run type Chinook salmon tend to spawn in higher elevation reaches of major Snake River tributaries while summer-run Chinook salmon tend to spawn lower in the Snake River drainages. However, there is an overlap of summer-run Chinook salmon spawning areas and that of spring-run spawners. Spring-run Chinook salmon spawn in mid- through late August, and summer-run Snake River Chinook salmon spawn approximately one month later than spring-run fish. Eggs incubate over the following winter, and hatch in late winter and early spring of the following year. Juvenile fish mature in fresh water for one year before they migrate to the ocean in the spring of their second year of life. Depending on the tributary and the specific habitat conditions, juveniles may migrate extensively from natal reaches into alternative summer-rearing or overwintering areas. Snake River Spring/Summer-run Chinook salmon return from the ocean to spawn primarily as four and five year-old fish, after two to three years in the ocean.

Status and Trends

NMFS originally listed SR Spring/Summer-run Chinook salmon as threatened on April 22, 1992 (57 FR 14653), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). The ICBTRT has identified 31 historic populations (Table 18). Historic populations above Hells Canyon Dam are considered extinct (ICTRT 2003). Multiple spawning sites are accessible and natural spawning and rearing are well distributed within the ESU. However, many spawning aggregates have also been extirpated, which has increased the spatial separation of some populations. The South Fork and Middle Fork Salmon Rivers currently support the bulk of natural production in the drainage. Table 18 identifies populations within the Snake River Spring/Summer-run Chinook salmon ESU, their abundances, and hatchery input.

Table 18. Snake River Spring/Summer-run Chinook salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005). Note: rpm denotes redds per mile.

Current Populations	Historical Abundance	Mean Number of Spawners (Range)	Hatchery Abundance Contributions
Tucannon River	Unknown	303 (128-1,012)	76%
Wenaha River	Unknown	225 (67-586)	64%
Wallowa River	Unknown	0.57 redds (0-29)	5%
Lostine River	Unknown	34 redds (9-131)	5%
Minam River	Unknown	180 (96-573)	5%
Catherine Creek	Unknown	50 (13-262)	56%
Upper Grande Ronde River	Unknown	46 (3-336)	58%
Imnaha River	Unknown	564 redds (194-3,041)	62%
Big Sheep Creek	Unknown	0.25 redds (0-1)	97%
Little Salmon	Unknown	Unknown	Unknown
South Fork Salmon River	Unknown	496 redds (277-679)	9%
Secesh River	Unknown	144 redds (38-444)	4%
Johnson Creek	Unknown	131 redds (49-444)	0%
Big Creek spring run	Unknown	53 redds (21-296)	0%
Big Creek summer run	Unknown	5 redds (2-58)	Unknown
Loon Creek	Unknown	27 redds (6-255)	0%
Bear Valley/Elk Creek	Unknown	266 (72-712)	0%
Marsh Creek	Unknown	53 (0-164)	0%
North Fork Salmon River	Unknown	5.6 redds (2-19)	Unknown
Lemhi River	Unknown	72 redds (35-216)	0%
Pahsimeroi River	Unknown	161 (72-1,097)	Unknown
East Fork Salmon spring run	Unknown	0.27 rpm (0.2 – 1.41)	Unknown
East Fork Salmon summer run	Unknown	1.22 rpm (0.35 – 5.32)	0%
Yankee Fork spring run	Unknown	0	Unknown
Yankee Fork summer run	Unknown	2.9 redds (1-18)	0%
Valley Creek spring run	Unknown	7.4 redds (2-28)	0%
Valley Creek summer run	Unknown	2.14 rpm (0.71 – 9.29)	Unknown
Upper Salmon spring run	Unknown	69 redds (25-357)	Unknown
Upper Salmon summer run	Unknown	0.24 rpm (0.07 – 0.58)	Unknown
Alturas Lake Creek	Unknown	2.7 redds (0-18)	Unknown
Lick Creek	Unknown	1.44 redds (0-29)	59%
ESU Estimate	~1.5 million	~9,700	

According to Matthews and Waples (Matthews and Waples 1991), total annual SR Spring/Summer-run Chinook salmon production may have exceeded 1.5 million adult

fish in the late 1800s. Total (natural plus hatchery origin) returns fell to roughly 100,000 spawners by the late 1960s (Fulton 1968). Between 1981 and 2000, total returns fluctuated between extremes of 1,800 and 44,000 fish. The 2001 and 2002 total returns increased to over 185,000 and 97,184 adults, respectively.

Abundance of summer run Chinook salmon have increased since the low returns in the mid-1990s (lowest run size was 692 fish in 1995). The 1997 to 2008 geometric mean total return for the summer run component at Lower Granite Dam was slightly more than 8,700 fish, compared to the geometric mean of 3,076 fish for the years 1987 to 1996 (Data from the Columbia Basin Fisheries Agencies and Tribes <http://www.fpc.org/>). However, over 80% of the 2001 return and over 60% of the 2002 return originated from hatcheries (Good, Waples et al. 2005). Good *et al.* (2005) reported that risks to individual populations within the ESU may be greater than the extinction risk for the entire ESU due to low levels of annual abundance of individual populations. Further, despite the increase in abundance during the last ten years, annual abundance continues to be variable and is most pronounced in natural-origin fish. Thus, although the average abundance in the most recent decade is higher than the previous decade, there is no obvious long-term trend (Good, Waples et al. 2005) (Data from the Columbia Basin Fisheries Agencies and Tribes <http://www.fpc.org/>). However, recent trends, buoyed by the last five years, are approaching 1. Additionally, hatchery fish are faring better than wild fish, which comprise roughly 40% of the total returns in the past decade. Overall, most populations are far below their respective interim recovery targets.

There is no evidence of wide-scale genetic introgression by hatchery populations. The high variability in life history traits indicates sufficient genetic variability within the ESU to maintain distinct subpopulations adapted to local environments (Good, Waples et al. 2005).

Critical Habitat

NMFS designated critical habitat for the Snake River (SR) Spring/Summer-run Chinook salmon on October 25, 1999 (64 FR 57399). This critical habitat encompasses the waters, waterway bottoms, and adjacent riparian zones of river reaches of the Columbia, Snake, and Salmon Rivers, and all tributaries of the Snake and Salmon Rivers, that are or were accessible to listed Snake River salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams).

NMFS identified spawning, rearing, and migration as PCEs for the SR Spring/Summer-run Chinook salmon. Spawning and juvenile rearing essential features consist of adequate (1) spawning gravel, (2) water quality, (3) water quantity, (4) water temperature, (5) riparian vegetation, (6) food, (7) cover/shelter, and (8) space. Juvenile and adult migration corridor essential features consist of adequate (1) substrate, (2) water quality, (3) water quantity, (4) water temperature, (5) food (juveniles only), (6) riparian vegetation, and (7) access.

Watersheds within the critical habitat designated for the SR Spring/Summer-run Chinook salmon have not been evaluated for their conservation value. However, the lower Columbia River corridor is among the areas of high conservation value to the ESU because it connects every population with the ocean and is used by rearing/migrating juveniles and migrating adults.

Spawning and juvenile rearing PCEs are regionally degraded by changes in flow quantity, water quality, and loss of cover. Juvenile and adult migrations are obstructed by reduced access that has resulted from altered flow regimes from hydroelectric dams. According to the ICBTRT, the Panther Creek population was extirpated because of legacy and modern mining-related pollutants creating a chemical barrier to fish passage (Chapman and Julius 2005).

Presence of cool water that is relatively free of contaminants is particularly important for the spring/summer run life history as adults hold over the summer and juveniles may rear for a whole year in the river. Water quality impairments are common in the range of the critical habitat designated for this ESU. Pollutants such as petroleum products, pesticides, fertilizers, and sediment in the form of turbidity enter the surface waters and riverine bottom substrate from the headwaters of the Snake, Salmon, and Clearwater Rivers to the Columbia River estuary as contaminated stormwater runoff, aerial drift and deposition, and via point source discharges. Some contaminants such as mercury and pentachlorophenol enter the aquatic food web after reaching water and may be concentrated or even biomagnified in the salmon tissue. This species also requires migration corridors with adequate passage conditions (water quality and quantity available at specific times) to allow access to the various habitats required to complete their life cycle.

Upper Willamette River Chinook Salmon

The Upper Willamette River (UWR) Chinook salmon ESU includes all naturally spawned populations of spring-run Chinook salmon in the Clackamas River and in the Willamette River, and its tributaries, above Willamette Falls, Oregon (64 FR 14208 Figure 13). Seven artificial propagation programs are included in the ESU (70 FR 37160). These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within the ESU.

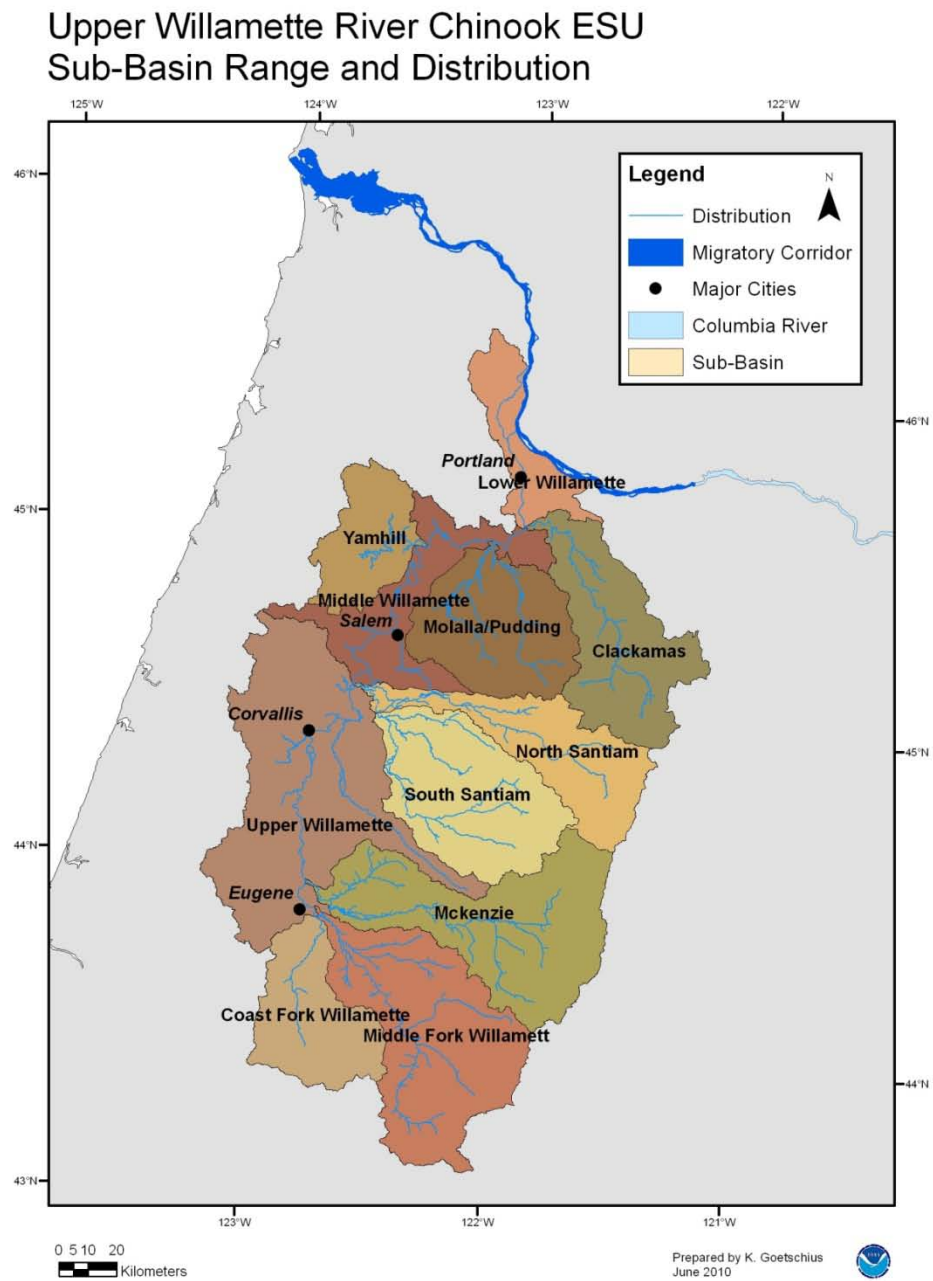


Figure 13. Upper Willamette River Chinook salmon distribution.

Life History

UWR Chinook salmon exhibit an earlier time of entry into the Columbia River than other spring-run Chinook salmon ESUs (Myers, Kope et al. 1998). Adults appear in the lower Willamette River in February, but the majority of the run ascends Willamette Falls in April and May, with a peak in mid- to late May. However, present-day salmon ascend the Willamette Falls via a fish ladder. Consequently, the migration of spring Chinook salmon over Willamette Falls extends into July and August (overlapping with the beginning of the introduced fall-run of Chinook salmon).

The adults hold in deep pools over summer and spawn in late fall or early winter when winter storms augments river flows. Fry may emerge from February to March and sometimes as late as June (Myers, Busack et al. 2006). Juvenile migration varies with three distinct juvenile emigration “runs”: fry migration in late winter and early spring; sub-yearling (0 yr +) migration in fall to early winter; and yearlings (1 yr +) migrating in late winter to spring. Sub-yearlings and yearlings rear in the mainstem Willamette River where they also use floodplain wetlands in the lower Willamette River during the winter-spring floodplain inundation period.

Status and Trends

NMFS originally listed UWR Chinook salmon as threatened on March 24, 1999 (64 FR 14308), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160).

Historically, this ESU included sizable numbers of spawning salmon in the Santiam River, the middle fork of the Willamette River, and the McKenzie River, as well as smaller numbers in the Molalla River, Calapooia River, and Albiqua Creek. Table 19 identifies populations within the UWR Chinook salmon ESU, their abundances, and hatchery input.

The W/LCRTRT identified seven historical independent populations (Myers, Busack et al. 2006) (Table 19). Most natural spring Chinook salmon populations of this ESU are likely extirpated or nearly so. The spring Chinook salmon in the McKenzie River is the only remaining naturally reproducing population in this ESU. Current spatial distribution is reduced by the loss of 30 to 40% of the total historic habitat which has restricted spawning to a few areas below dams.

Table 19. Upper Willamette River Chinook salmon independent populations core (C) and genetic legacy (G) populations, and hatchery contributions (Good, Waples et al. 2005).

Functionally Independent Populations	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Clackamas River	Unknown	2,910	64%
Molalla River	Unknown	52 redds	>93%
North Santiam River	Unknown	~ 7.1 rpm	>95%
South Santiam River	Unknown	982 redds	>84%
Calapooia River	Unknown	16 redds	100%
McKenzie River	Unknown	~2,470	26%
Middle Fork Willamette River	Unknown	235 redds	>39%
Total	>70,000	~9,700	Mostly hatchery

Note: rpm denotes redds per mile

The total abundance of adult spring-run Chinook salmon (hatchery-origin + natural-origin fish) passing Willamette Falls has remained relatively steady over the past 50 years (ranging from approximately 20,000 to 70,000 fish). However, the current abundance is an order of magnitude below the peak abundance levels observed in the 1920s (approximately 300,000 adults). Total number of fish increased during the period from 1996 to 2004 when it peaked at more than 96,000 adult spring-run Chinook salmon passing Willamette Falls. Since then, the run has steadily decreased with only about 14,000 fish counted in 2008, the lowest number since 1960. ESU abundance increased again to about 25,000 adult spring-run Chinook salmon in 2009. Runs consist of a high but uncertain fraction of hatchery-produced fish.

The spring Chinook salmon in the McKenzie River is the only remaining self sustaining naturally reproducing independent population. The other natural-origin populations in

this ESU have very low current abundances, and long- and short-term population trends are negative.

Access of fall-run Chinook salmon to the upper Willamette River and the mixing of hatchery stocks within the ESU have threatened the genetic integrity and diversity of the species. Much of the genetic diversity that existed between populations has been homogenized (Myers, Busack et al. 2006).

Critical Habitat

NMFS designated critical habitat for this species on September 2, 2005 (70 FR 52630). Designated critical habitat includes all Columbia River estuarine areas and river reaches proceeding upstream to the confluence with the Willamette River as well as specific stream reaches in a number of subbasins.

NMFS assessed the conservation value of 59 watersheds within the range of the UWR Chinook salmon (Table 20). Nineteen watersheds received a low rating, 18 received a medium rating, and 22 received a high rating of conservation value to the ESU (NMFS 2005). The lower Willamette/Columbia River rearing/migration corridor downstream of the spawning range is also considered to have a high conservation value and is the only habitat designated in four of the high value watersheds.

The current condition of PCEs of the UWR Chinook salmon critical habitat indicates that migration and rearing PCEs are not currently functioning or are degraded. These conditions impact their ability to serve their intended role for species conservation. The migration PCE is degraded by dams altering migration timing and water management altering the water quantity necessary for mobility and survival. Migration, rearing, and estuary PCEs are also degraded by loss of riparian vegetation and instream cover. Pollutants such as petroleum products, fertilizers, pesticides, and fine sediment enter the stream through runoff, point source discharge, drift during application, and non-point discharge where agricultural and urban development occurs. Degraded water quality in

the lower Willamette River where important floodplain rearing habitat is present affects the ability of this habitat to sustain its role to conserve the species.

Table 20. UWR Chinook salmon watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Middle Fork Willamette	4	(1)	6	(2, 1)	0	
Coastal Fork Willamette	0		0		4	(2, 1)
Upper Willamette	0		3	(2, 1)	3	(2)
McKenzie	5	(1, 2)	2	(2, 1)	0	
North Santiam	2	(1)	1	(2, 1)	0	
South Santiam	3	(1, 2)	3	(2, 1)	0	
Middle Willamette	0		0		4	(2)
Yamhill	0		0		4	(2)
Molalla/Pudding	0		3	(1, 2)	3	(2)
Clackamas	5	(1) ²	0		1	(1)
Lower Willamette	3	(2)	0		0	
Columbia River Corridor	all	(3)	0		0	
Total	22		18		19	

¹ Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

² .Lower Clackamas River provides for 13.4 miles of PCE 2

California Coastal Chinook Salmon

California Coastal (CC) Chinook salmon includes all naturally-spawned coastal Chinook salmon spawning north from Redwood Creek to, and including, the Russian River to the south as shown in Figure 14. Seven artificial propagation programs are part of this ESU. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Life History

CC Chinook salmon are a fall-run, ocean-type fish. Although a spring-run (river-type) component existed historically, it is now considered extinct (Bjorkstedt, Spence et al. 2005). The different populations vary in run timing depending on latitude and hydrological differences between watersheds. Entry of CC Chinook salmon into the Russian River depends on increased flow from fall storms, usually in November to January. Juveniles of this ESU migrate downstream from April through June and may reside in the estuary for an extended period before entering the ocean.

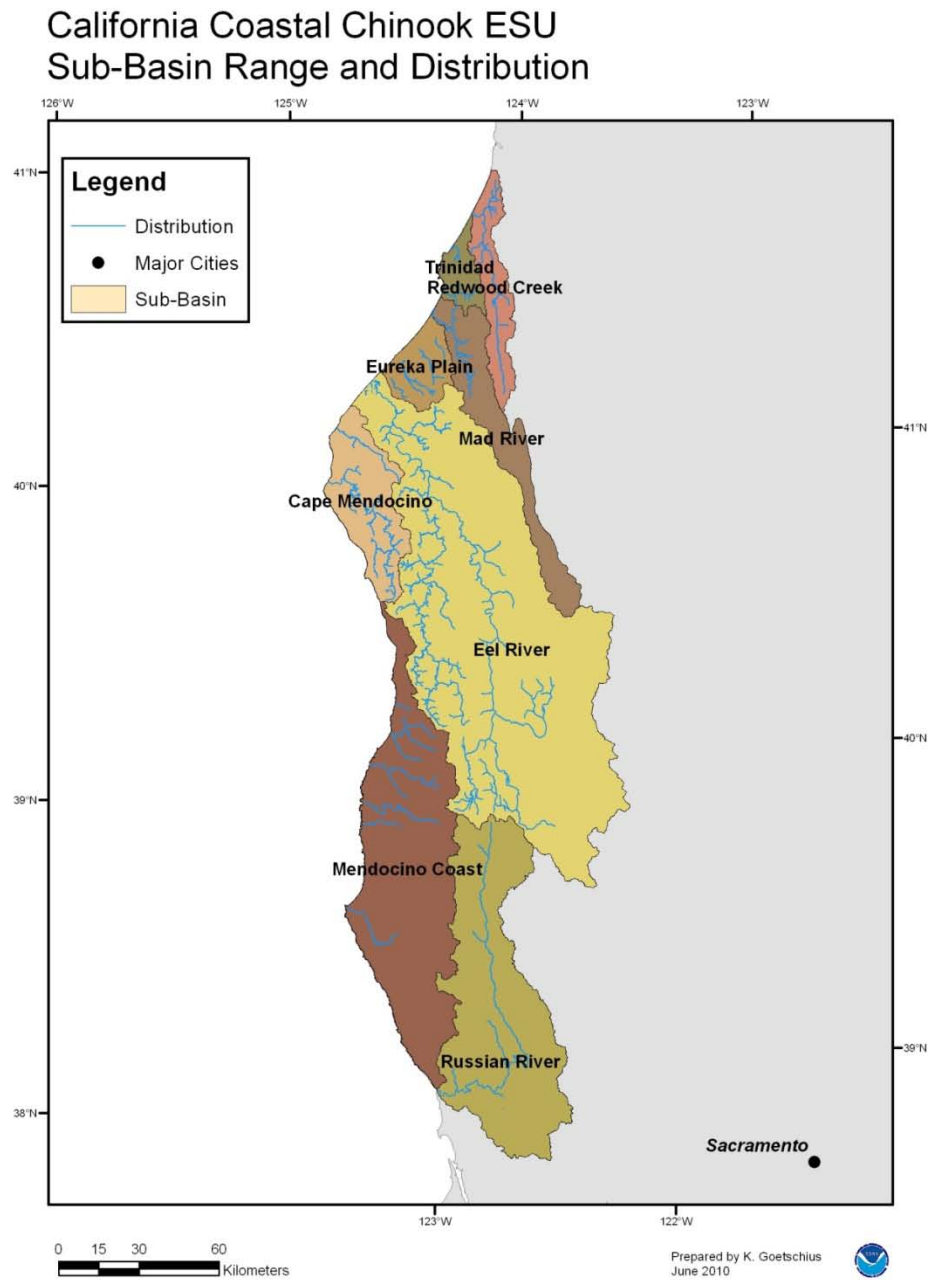


Figure 14. California Coastal Chinook salmon distribution.

Table 21. California Coastal Chinook salmon fall-run populations-preliminary population structure, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historic Spawner Abundance	Mean Number of Spawners	Hatchery Abundance Contributions
Eel River (includes * tributaries below) – 2 populations		156-2,730	~30%
Mainstem Eel River*	13,000	Inc. in Eel River	Unknown
Van Duzen River*	2,500	Inc. in Eel River	Unknown
Middle Fork Eel River*	13,000	Inc. in Eel River	Unknown
South Fork Eel River*	27,000	Inc. in Eel River	Unknown
North Fork Eel River*	Unknown	Inc. in Eel River	Unknown
Upper Eel River*	Unknown	Inc. in Eel River	Unknown
Redwood Creek	1,000-5,000	Unknown	0
Mad River	1,000-5,000	19-103	Unknown
Bear River	100	Unknown	0
Mattole River	1,000-5,000	Unknown	17%
Small Humboldt County rivers	1,500	Unknown	0
Rivers north of Mattole River	600	Unknown	0
Humboldt Bay tributaries	40	120	40 (33%)
Noyo River	50	Unknown	0
Russian River	50-500	>1,383 – >6,103	~0%
Tenmile to Gualala coastal effluents	Unknown	Unknown	0
Total	20,750-72,550	Unknown	

Status and Trends

NMFS listed CC Chinook salmon as threatened on September 16, 1999 (64 FR 50393), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). The CC Chinook ESU historically consisted of 10 functionally independent populations and 5 potentially independent populations (Bjorkstedt, Spence et al. 2005). Seventeen basins may have had Chinook salmon runs that relied on immigration from the larger basins. ESU connectivity is substantially reduced by the near extirpation of all historically independent populations between the Russian River in Sonoma County and Mattole River in Humboldt County ((NMFS 2008), (Spence, Bjorkstedt et al. 2008)). The number of extant populations is uncertain.

Historical estimates of escapement suggest abundance was roughly 73,000 in the early 1960s, with the majority of fish spawning in the Eel River, and about 21,000 in the 1980s

(Good, Waples et al. 2005). Table 21 identifies populations within the CC Chinook salmon ESU, their abundances, and hatchery input.

Comparison of historical and current abundance information indicates that independent populations of Chinook salmon are depressed in many basins ((Bennet 2005), (Good, Waples et al. 2005), (NMFS 2008)). All spring-run populations once occupying the North Mountain Interior are considered extinct or nearly so. Redd counts in Mattole River in the northern portion of the ESU indicate a small but consistent population; the cooler northern climate likely provides for favorable conditions for these populations (Spence, Bjorkstedt et al. 2008). The Eel River interior fall-run populations are severely depressed (Spence, Bjorkstedt et al. 2008). Two functionally independent populations are believed to have existed along the southern coastal portion of the ESU; of these two, only the Russian River currently has a run of any significance (Bjorkstedt, Spence et al. 2005). This is also the only population with abundance time series. The 2000 to 2007 median observed (at Mirabel Dam) Russian River Chinook salmon run size is 2,991 with a maximum of 6,103 (2003) and a minimum of 1,125 (2008) adults ((Cook 2008), (Sonoma County Water Agency (SCWA) 2008)). The number of spawners has steadily decreased since its high returns in 2003 with 1,963 fish observed in 2007 and 1,125 observed by December 22, 2008. The time series is too short to estimate lambda.

The CC Chinook ESU is at considerable risk from population fragmentation and reduced spatial diversity. There is little connectivity between the southern and northern portions of their range. At the southern portion of the ESU, only the Russian River population has had a constant run that exceeded 1,000 adult spawning fish over the last 10 years. This places the ESU at risk from random catastrophic events, chronic stressors, and long-term environmental change. Life history diversity has been significantly reduced by loss of the spring-run race and reduction in coastal populations.

Critical Habitat

NMFS designated critical habitat for the CC Chinook salmon on September 2, 2005 (70 FR 52488). It includes multiple CALWATER hydrological units north from Redwood Creek and south to Russian River (Table 22). The total area of critical habitat includes 1,500 miles of stream habitat and about 25 square miles of estuarine habitat, mostly within Humboldt Bay. A list and maps of watersheds and streams designated as critical habitat for CC Chinook salmon can be found in the Federal Register (70 FR 52488).

There are 45 occupied CALWATER Hydrologic Subarea (HSA) watersheds within the freshwater and estuarine range of this ESU. Eight watersheds received a low rating, 10 received a medium rating, and 27 received a high rating of conservation value to the ESU (70 FR 52488). Two estuarine habitat areas used for rearing and migration (Humboldt Bay and the Eel River Estuary) also received a high conservation value rating (Figure 15).

Table 22. CC Chinook salmon CALWATER HSA watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Redwood Creek	2	(1, 2, 3)	1	(1, 2, 3)	0	
Trinidad	1	(1, 2, 3)	0		1	(1, 2, 3)
Mad River	3	(1, 2, 3)	0		0	
Eureka Plain	1	(1, 2, 3)	0		0	
Eel River	12	(1, 2, 3)	4	(1, 2, 3)	3	(1, 2, 3)
Cape Mendocino	2	(1, 2, 3)	0		0	
Mendocino Coast	2	(1, 2, 3)	3	(1, 2, 3)	2	(1, 2, 3)
Russian River	4	(1, 2, 3)	2	(1, 2, 3)	2	(1, 2, 3)
Total	27		10		8	

¹ Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

Critical habitat in this ESU consists of limited quantity and quality summer and winter rearing habitat, as well as marginal spawning habitat. Compared to historical conditions, there are fewer pools, limited cover, and reduced habitat complexity. The current

condition of PCEs of the CC Chinook salmon critical habitat indicates that PCEs are not currently functioning or are degraded; their conditions are likely to maintain a low population abundance across the ESU. CC Chinook salmon spawning PCE in coastal streams is degraded by years of timber harvest that has produced large amounts of sand and silt in spawning gravel and reduced water quality by increased turbidity. Agriculture and urban areas has impacted rearing and migration PCEs in the Russian River by degrading water quality and by disconnecting the river from its floodplains by the construction of levees. Water management from dams within the Russian and Eel River watersheds maintain high flows and warm water during summer which benefits the introduced predatory Sacramento pikeminnow. This has resulted in excessive predation along migration corridors. Breaches of the sandbar at the mouth of the Russian River result in periodic mixing of salt water. This condition degrades the estuary PCE by altering water quality and salinity conditions that support juvenile physiological transitions between fresh- and salt water.

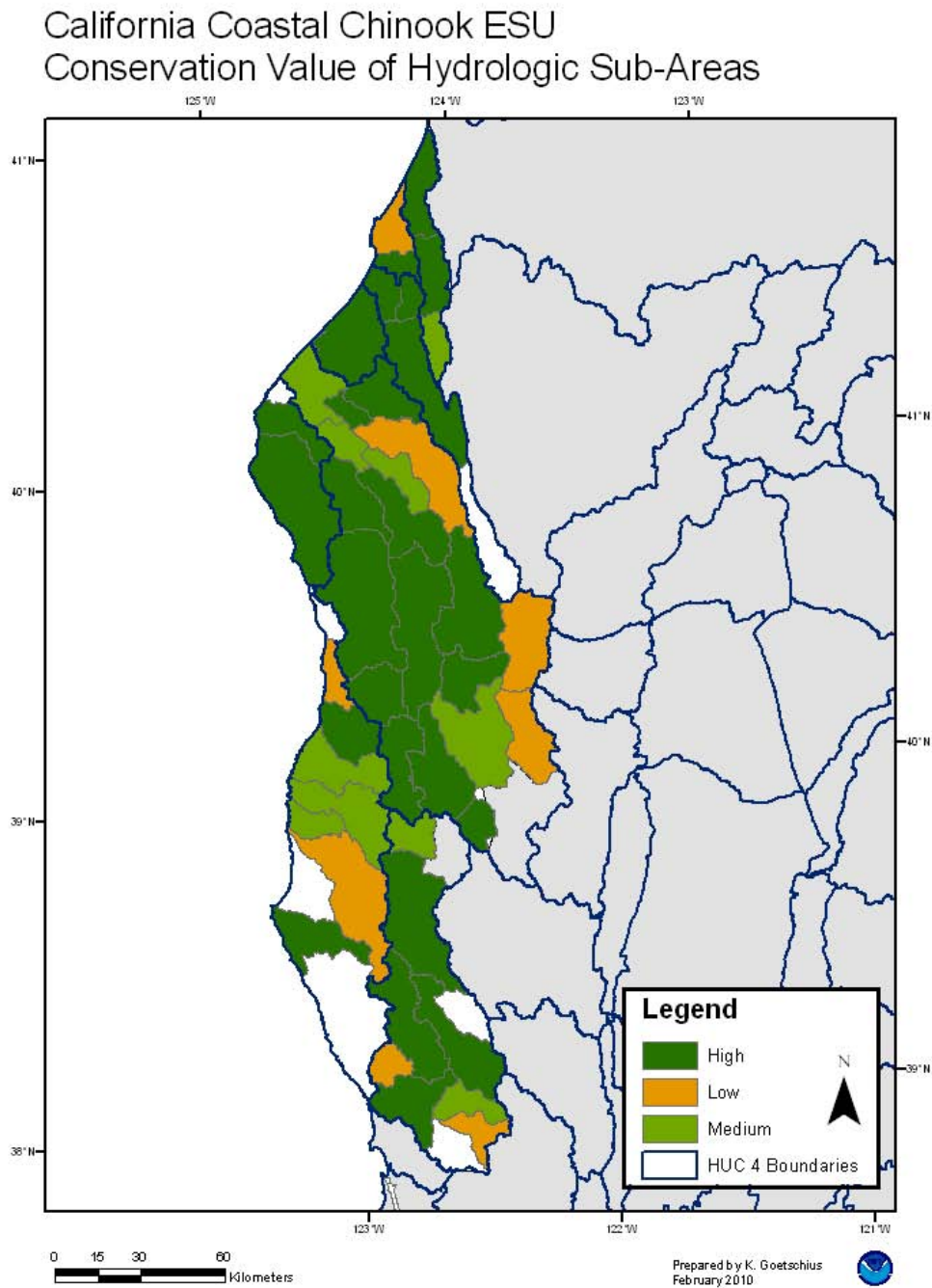


Figure 15. California Coastal Chinook salmon Conservation Values per Sub-Area.

Central Valley Spring-run Chinook Salmon

The Central Valley (CV) Spring-run Chinook salmon includes all naturally spawned populations of spring-run Chinook salmon in the Sacramento River, California, and its tributaries (Figure 16). The Feather River Hatchery spring-run Chinook salmon is included in this ESU. This artificially propagated population is no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU. Table 23 identifies populations within the CV Spring-run Chinook salmon ESU, their abundances, and hatchery input.

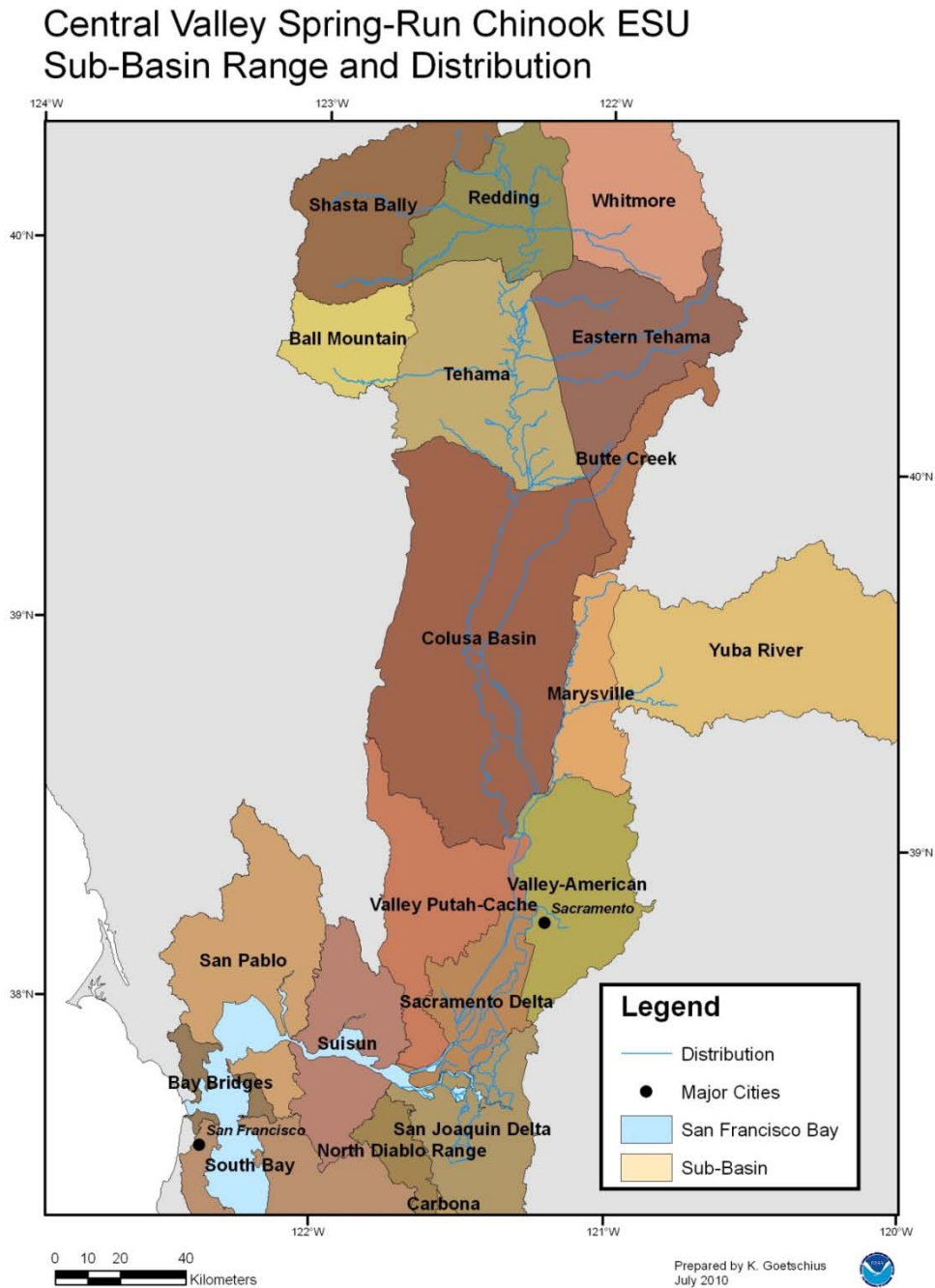


Figure 16. Central Valley Spring-run Chinook salmon distribution.

Life History

CV Spring-run Chinook salmon enter the Sacramento River from March to September and spawn from late August through early October, with a peak in September. Chinook salmon require cool fresh water while they mature over the summer. Adult upstream migration may be blocked by temperatures above 21°C (McCullough 1999). Fry emerge from the gravel November to March. Juvenile spring-run emigration in the Sacramento River is highly variable and they may migrate either as soon as they emerge from the gravel or as yearlings. The majority of spring-run fry emerging in the tributaries migrate downstream from December through February during high flows. Juvenile CV Spring-run Chinook salmon have been observed rearing in the lower reaches of non-natal tributaries and intermittent streams in the Sacramento Valley during the winter months. Peak fry/sub-yearling movements are observed farther downstream in lower Sacramento River (Knights Landing) and the Delta during March and April. Up to 25% of juveniles may remain in the tributaries to rear and outmigrate as yearlings the next fall, normally starting in December.

Status and Trends

NMFS originally listed CV Spring-run Chinook salmon as threatened on September 16, 1999 (64 FR 50393), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). Historically, spring-run Chinook salmon were predominant throughout the Sacramento and San Joaquin River drainages. All runs within the San Joaquin River basin are now extirpated. Naturally spawning populations of CV Spring-run Chinook salmon currently are restricted to accessible reaches of the upper mainstem Sacramento River and its tributaries Butte, Deer, and Mill Creeks. Limited spawning occurs in the basins of smaller tributaries (CDFG 1998).

Table 23. Central Valley Spring-run Chinook salmon--preliminary population structure, historic and most recent natural production, spawner abundance, and hatchery contributions ((Good, Waples et al. 2005), (USFWS and Reclamation 2007)).

Population	Historic Natural Production (1967 – 1991)	Most Recent Natural Production ¹ (2000 – 2006)	Most Recent Spawner Abundance ² (2000- 2006)	Hatchery Abundance Contributions
Butte Creek	1,000	6,516 – 19,809	4,118 – 10,625	Unknown
Deer Creek	3,300	1,387 – 3,461	637 – 2,759	Unknown
Mill Creek	2,200	1,184 – 26,190	544 – 1594	Unknown
Sacramento River	29,000	0 – 1,134	0 – 394	Unknown
Total	Estimated historic abundance: ~700,000 for all populations	11,403 – 26,190	5,370 – 14,044	Unknown

1. Includes catches

2. *i.e.*, escapement

The Central Valley drainage supported spring-run Chinook salmon runs as large as 700,000 fish between the late 1880s and the 1940s (Brown, Moyle et al. 1994). Before construction of Friant Dam, nearly 50,000 adults were counted in the San Joaquin River alone (Fry 1961).

Median natural production of spring-run Chinook salmon from 1970 to 1989 was 30,220 fish. In the 1990s, the population experienced a substantial production failure with an estimated natural production ranging between 3,863 and 7,806 fish (with the exception of 1995 which had a natural production of an estimated 35,640 adults) during the years between 1991 and 1997 (USFWS and Reclamation 2007). Numbers of naturally produced fish increased significantly in 1998 to an estimated 48,755 adults and estimated natural production has remained above 10,000 fish since then (USFWS and Reclamation 2007).

The Sacramento River trends and lambda show a long- and short- term negative trend and negative population growth (Good, Waples et al. 2005). Meanwhile, the median production of Sacramento River tributary populations increased from a low of 4,248 with only one year exceeding 10,000 fish before 1998 to a combined natural production of

more than 10,000 spring-run Chinook in all years after 1998 (data from (USFWS and Reclamation 2007)). Time series data for Mill, Deer, Butte, and Big Chico Creeks spring-run Chinook salmon (updated through 2006) show that all three tributary spring-run Chinook populations have long-and short-term lambdas >1 ; indicating population growth (Good, Waples et al. 2005). Although the populations are small, CV spring-run Chinook salmon have some of the highest population growth rates in the Central Valley.

Critical Habitat

NMFS designated critical habitat for this species on September 2, 2005 (70 FR 52488). The critical habitat boundary includes the Sacramento River and several tributaries from the Big Chico tributary with Sacramento River upstream to Shasta Dam (Table 24).

There are 38 occupied HSA watersheds within the freshwater and estuarine range of this ESU. As shown in Figure 17, seven watersheds received a low rating, 3 received a medium rating, and 27 received a high rating of conservation value to the ESU (NMFS 2005). Four of these HSA watersheds comprise portions of the San Francisco-San Pablo-Suisun Bay estuarine complex which provides rearing and migratory habitat for this ESU.

The current condition of PCEs of the CV Spring-run Chinook salmon critical habitat indicates that PCEs are not currently functioning or are degraded; their conditions are likely to maintain a low population abundance across the ESU. Spawning and rearing PCEs are degraded by high water temperature caused by the loss of access to historic spawning areas in the upper watersheds which maintained cool and clean water throughout the summer. The rearing PCE is degraded by floodplain habitat being disconnected from the mainstem of larger rivers throughout the Sacramento River watershed, thereby reducing effective foraging. Migration PCE is degraded by lack of natural cover along the migration corridors. Juvenile migration is obstructed by water diversions along Sacramento River and by two large state and federal water-export facilities in the Sacramento-San Joaquin Delta.

Table 24. CV Spring-run Chinook salmon CALWATER HSA watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
San Francisco Bay	San Francisco Bay	Estuary PCEs	0	0	1	Estuary PCEs
Suisun Bay	Suisun Bay	1	0	0	0	
Tehama	1	(1, 2, 3)	1	(1, 2, 3)	0	
Whitmore	1	(1, 2, 3)	0		2	(1, 2, 3)
Redding	2	(1, 2, 3)	0		0	
Eastern Tehama	4	(1, 2, 3)	0		0	
Sacramento Delta	1	(2, 3, 1)	0		0	
Valley Putah-Cache	1	(1, 2, 3)	0		0	
Marysville	3	(1, 2, 3)	0		0	
Yuba River	2	(1, 2, 3)	1	(1, 2, 3)	1	(1, 2, 3)
Valley-American	2	(1, 2, 3)	0		0	
Colusa Basin	4	(1, 2, 3)	0		0	
Butte Creek	1	(1, 2, 3)	0		0	
Ball Mountain	0		0		1	(1, 2, 3)
Shasta Bally	3	(1, 2, 3)	0		1	(1, 2, 3)
North Diablo Range	0		1	(1, 2, 3)	0	
San Joaquin Delta	0		0		1	(1, 2, 3)
Total	28		3		7	

¹ Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

Contaminants from agriculture and urban areas have degraded rearing and migration PCEs to the extent that they have lost their functions necessary to serve their intended role to conserve the species. Water quality impairments in the designated critical habitat of this ESU include inputs from fertilizers, insecticides, fungicides, herbicides, surfactants, heavy metals, petroleum products, animal and human sewage, sediment in the form of turbidity, and other anthropogenic pollutants. Pollutants enter the surface waters and riverine sediments as contaminated stormwater runoff, aerial drift and

deposition, and via point source discharges. Some contaminants such as mercury and pentachlorophenol enter the aquatic food web after reaching water and may be concentrated or even biomagnified in salmon tissue.

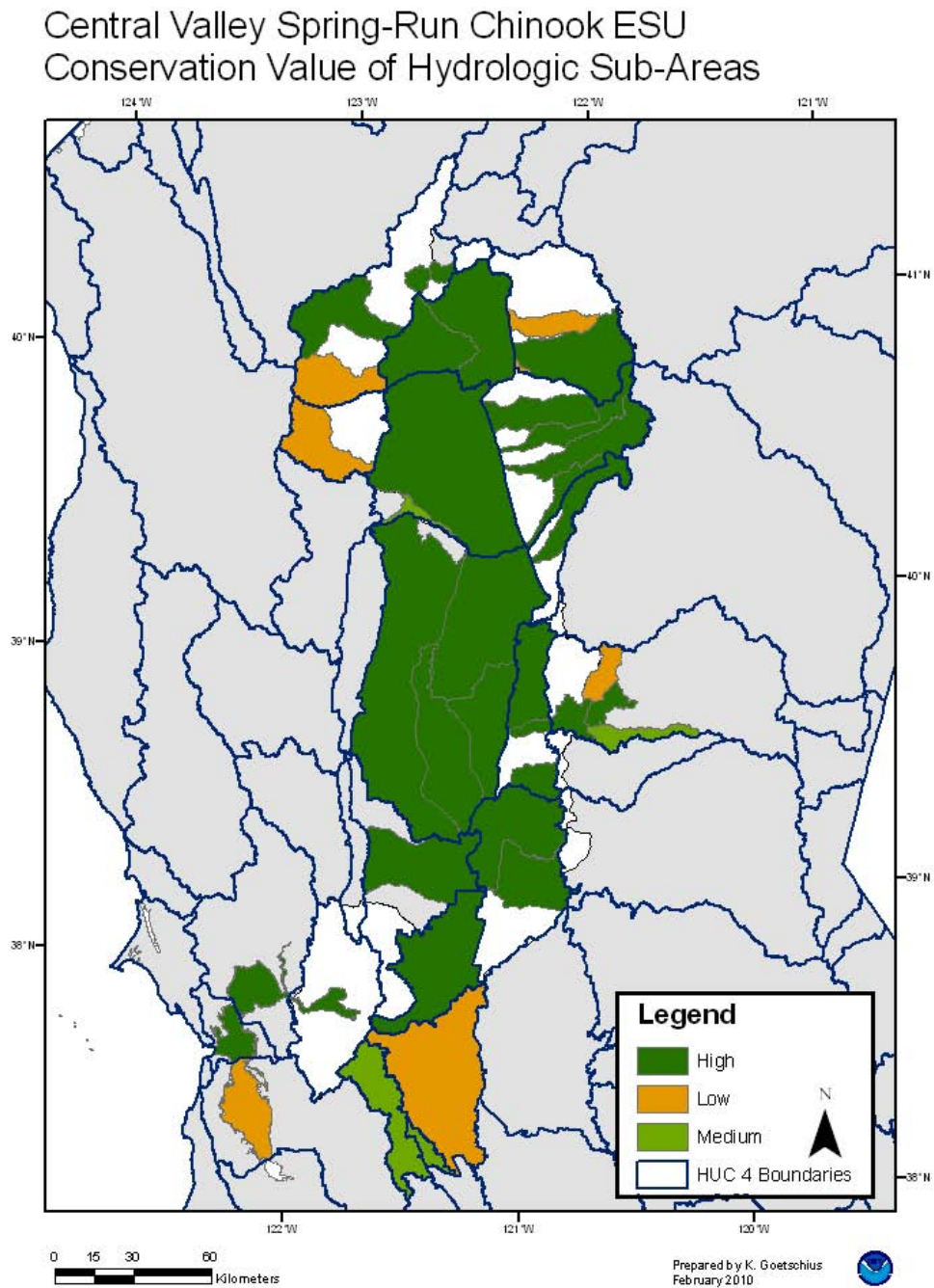


Figure 17. Central Valley Spring-run Chinook salmon Conservation Values per Sub-Area.

Sacramento River Winter-run Chinook Salmon

The ESU includes all winter-run Chinook salmon entering and using the Sacramento River system in the Central Valley, California. The ESU boundary extends from the Carquinez Strait by the City of Vallejo and Benicia upstream the Sacramento River, including all its tributaries, to below Keswick Dam (Figure 18). The ESU now consists of a single spawning population.

Life History

The winter-run Chinook salmon have characteristics of both stream- and ocean-type races (Healey 1991). Adults enter fresh water in winter or early spring but delays spawning until May and June. Fry emerge from the gravel in late June to early July and continue through October (Fisher 1994). Young winter-run Chinook salmon start migrating to sea as early as mid July with a peak movement over the Red Bluff Diversion Dam (RBDD) in September. Some offspring move downstream as fry while other rear in the upper Sacramento River and move down as smolt. Normally fry have passed the RBDD by October while smolts may pass over the RBDD until March. Juvenile winter-runs occur in the Delta primarily from November through early May. Winter-run juveniles remain in the Delta until they are from 5 to 10 months of age, and then begin emigrating to the ocean as early as November and continue through May ((Fisher 1994), (Myers, Kope et al. 1998). The winter-run race matures between two and six years of age with the majority returning as three-year olds.

Sacramento River Winter Run Chinook ESU Sub-Basin Range and Distribution

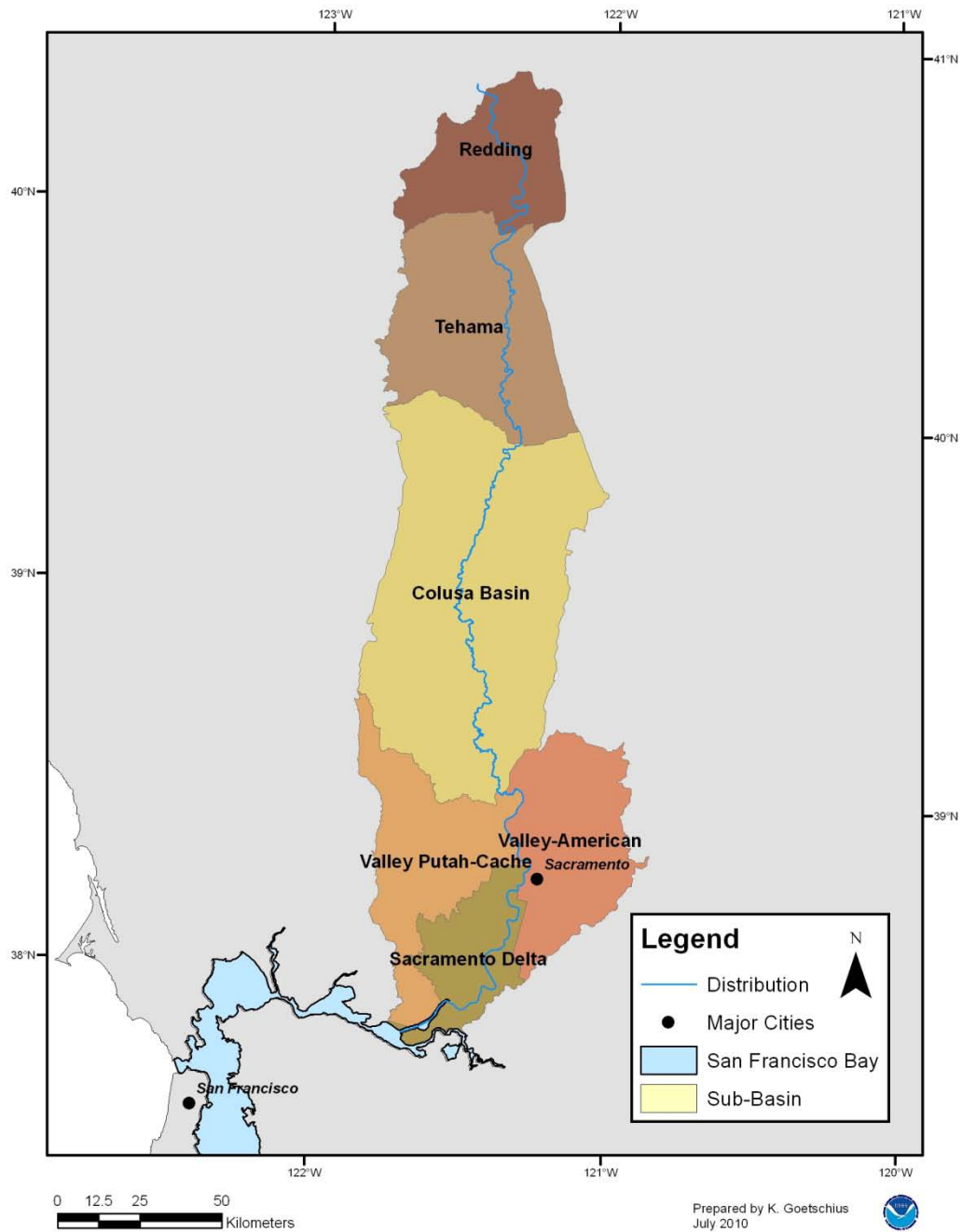


Figure 18. Sacramento River Winter-run Chinook salmon distribution.

Status and Trends

NMFS listed Sacramento River Winter-run Chinook salmon as endangered on January 4, 1994 (59 FR 440), and reaffirmed their endangered status on June 28, 2005 (70 FR 37160). The winter-run Chinook salmon spawned and reared in the upper Sacramento River and its tributaries ((Slater 1963), (Yoshiyama, Gerstung et al. 1998)). Today the Shasta Dam eliminates access to the historic spawning habitat. Cold water releases from the dam have also created conditions suitable for winter-run spawning and rearing in a 60- to 100-mile long portion of the Sacramento River downstream of the dam. As a result, the Sacramento River Winter-run Chinook salmon has been reduced to a single spawning population confined to a portion of the mainstem Sacramento River.

Winter-runs may have been as large as 200,000 fish based upon commercial fishery records from the 1870s (Fisher 1994). During the first three years of operation of the counting facility at the RBDD (From 1967 to 1969), an average of 86,500 winter-run Chinook salmon were counted (CDFG 2008). Critically low levels were reached during the drought of 1987 to 1992 with an absolute bottom of 191 fish counted. The three-year average run size for the period of 1989 to 1991 was 388 fish.

The population grew rapidly from the early 1990s to mid-2005. Mean run size increased from 1,363 before 2000 with all runs estimated to less than 10,000 fish to an average run of 8,470 adults between 2000 and 2006 with two runs estimated to more than 10,000 fish (USFWS and Reclamation 2007). However, the natural produced winter-run Chinook salmon plunged in 2007 and 2008, with 4,461 adults estimated for 2007 and a preliminary estimate between of 2,600-2,950 adults for 2008 ((USFWS 2008), Garwin Yip pers. com).

The Sacramento River Winter-run Chinook salmon is expected to have lost some genetic diversity through bottleneck effects in the late 1980s and early 1990s. Hatchery releases may also have affected population genetics. The loss of natural spawning habitat and

hydrological conditions has further removed the natural evolutionary processes that maintained the unique winter-run life history.

Critical Habitat

NMFS designated critical habitat for this species on June 16, 1993 (58 FR 33212). It includes: the Sacramento River from Keswick Dam, Shasta County (river mile 302) to Chipps Island (river mile 0) at the westward margin of the Sacramento-San Joaquin Delta, and other specified estuarine waters.

NMFS identified specific water temperature criteria, minimum instream flow criteria, and water quality standards as essential physical features (PCEs) of the ESU's habitat for species conservation. In addition, biological features vital for the Sacramento River winter-run Chinook salmon include unimpeded adult upstream migration routes, spawning habitat, egg incubation and fry emergence areas, rearing areas for juveniles, and unimpeded downstream migration routes for juveniles.

This ESU has not been evaluated for the conservation value of individual subbasins or river sections. However, since spawning, rearing, and migration of the winter-run race is restricted to the mainstem of the Sacramento River, the entire Sacramento River is considered of high conservation value. The Delta is similarly considered of high conservation value for rearing and migration.

As there is overlap in designated critical habitat for both the Sacramento River Winter-run Chinook salmon and the spring-run Chinook salmon, the conditions of PCEs for both ESUs are similar. The current condition of PCEs for the Sacramento River Winter-run Chinook salmon indicates that they are not currently functioning or are degraded. Their conditions are likely to maintain low population abundances across the ESU. Spawning and rearing PCEs are especially degraded by high water temperature caused by the loss of access to historic spawning areas in the upper watersheds where water maintain lower temperatures. The rearing PCE is further degraded by floodplain habitat disconnected

from the mainstems of larger rivers throughout the Sacramento River watershed. The migration PCE is also degraded by the lack of natural cover along the migration corridors. Rearing and migration PCEs are further affected by pollutants entering the surface waters and riverine sediments as contaminated stormwater runoff, aerial drift and deposition, and via point source discharges. Juvenile migration is obstructed by water diversions along Sacramento River and by two large state and federal water-export facilities in the Sacramento-San Joaquin Delta.

Chum Salmon

Description of the Species

Chum salmon have the widest natural geographic and spawning distribution of any Pacific salmonid as their range extend farther along the shores of the Arctic Ocean than other salmonids. Chum salmon have been documented to spawn from Korea and the Japanese island of Honshu, east around the rim of the North Pacific Ocean to Monterey Bay, California. Historically, chum salmon were distributed throughout the coastal regions of western Canada and the U.S. Presently, major spawning populations occur as far south as Tillamook Bay on the northern Oregon coast. We discuss the distribution, life history diversity, status, and critical habitat of the two species of threatened chum salmon separately.

Chum salmon are semelparous, spawn primarily in fresh water, and exhibit obligatory anadromy (there are no recorded landlocked or naturalized freshwater populations). Chum salmon spend two to five years in feeding areas in the northeast Pacific Ocean, which is a greater proportion of their life history than other Pacific salmonids. Chum salmon are distributed throughout the North Pacific Ocean and Bering Sea.

North American chum salmon migrate north along the coast in a narrow coastal band that broadens in southeastern Alaska. However, some data suggest that Puget Sound chum, including Hood Canal Summer-run chum, may not migrate into northern British

Columbian and Alaskan waters. Instead, Puget Sound chum salmon travel directly offshore into the North Pacific Ocean.

Chum salmon usually spawn in the lower reaches of rivers. Redds are dug in the mainstem or in side channels of rivers from just above tidal influence to nearly 100 km from the sea. The time to hatching and emergence from the gravel redds are influenced by DO, gravel size, salinity, nutritional conditions, behavior of alevins in the gravel, and incubation temperature (reviewed (Bakkala 1970), (Salo 1991), (Schroder 1977), (Schroder, Koski et al. 1974)). For example, fertilized eggs hatch in about 100-150 days at 4°C, but hatch in only 26-40 days at 15°C. Juveniles outmigrate to sea water almost immediately after emerging from the gravel that covers their redds (Salo 1991). The immature salmon distribute themselves widely over the North Pacific Ocean. The maturing adults return to the home streams at various ages, usually at two through five years, and in some cases up to seven years (Bigler 1985). This ocean-type migratory behavior contrasts with the stream-type behavior of some other species in the genus *Oncorhynchus* (e.g., steelhead, coho, and most types of Chinook and sockeye salmon). Stream-type salmonids usually migrate to sea at a larger size, after months or years of freshwater rearing. Thus, survival and growth for juvenile chum salmon depend less on freshwater conditions than on favorable estuarine conditions. Another behavioral difference between chum salmon and other salmonid species is that chum salmon form schools. Presumably, this behavior reduces predation (Pitcher 1986) especially if fish movements are synchronized to swamp predators (Miller and Brannon 1982).

The duration of estuarine residence for chum salmon juveniles are known for only a few estuaries. Observed residence time ranged from 4 to 32 days, with about 24 days as the most common (Johnson, Grant et al. 1997). Chum salmon juveniles use shallow, low flow habitats for rearing that include inundated mudflats, tidal wetlands and their channels, and sloughs.

Status and Trends

Chum salmon, like the other salmon NMFS has listed, have declined from overharvests, hatcheries, native and non-native exotic species, dams, gravel mining, water diversions, destruction or degradation of riparian habitats, and land use practices (logging, agriculture, and urbanization). Chum salmon are also affected by shifts in climatic conditions that alter patterns and intensity of precipitation.

Hood Canal Summer-run Chum Salmon

The Hood Canal (HC) Summer-run chum salmon ESU (Figure 19) includes all naturally spawned populations in Hood Canal and its tributaries as well as populations in Olympic Peninsula rivers between Hood Canal and Dungeness Bay, Washington (64 FR 14508). Eight artificial propagation programs are included in the ESU: the Quilcene National Fish Hatchery, Hamma Hamma Fish Hatchery, Lilliwaup Creek Fish Hatchery, Union River/Tahuya, Big Beef Creek Fish Hatchery, Salmon Creek Fish Hatchery, Chimacum Creek Fish Hatchery, and the Jimmycomelately Creek Fish Hatchery summer-run chum hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations(s) than what would be expected between closely related natural populations within the species. Table 25 identifies populations within the HC Summer-run chum salmon ESU, their abundances, and hatchery input.

Table 25. Hood Canal Summer-run Chum salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

Historically Independent Populations	Stocks (Streams)	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Strait of Juan de Fuca	Chimacum Creek	Unknown	Extinct	N/A
	Dungeness Creek	Unknown	Unknown	Unknown
	Jimmycomelately Creek	Unknown	~60	Unknown
	Salmon/Snow creeks	Unknown	~2,200	0-69%
Hood Canal	Big/Little Quilcene rivers	Unknown	~4,240	5-51%
	Dosewallips River	Unknown	~900	Unknown
	Duckabush River	Unknown	Unknown	Unknown
	Hamma Hamma River	Unknown	~758	Unknown

Historically Independent Populations	Stocks (Streams)	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
	Lilliwaup Creek	Unknown	~164	Unknown
	Skokomish River	Unknown	Extinct	N/A
	Big Beef Creek*	Unknown	Extinct	100
	Dewetto Creek*	Unknown	Extinct	Unknown
	Anderson Creek*	Unknown	Extinct	N/A
	Mission Creek*	Unknown	Extinct	N/A
	Tahuya River*	Unknown	Extinct	N/A
	Union River*	Unknown	~690	Unknown

* Streams on the east side of Hood Canal.

Hood Canal Summer-Run Chum ESU Sub-Basin Range and Distribution

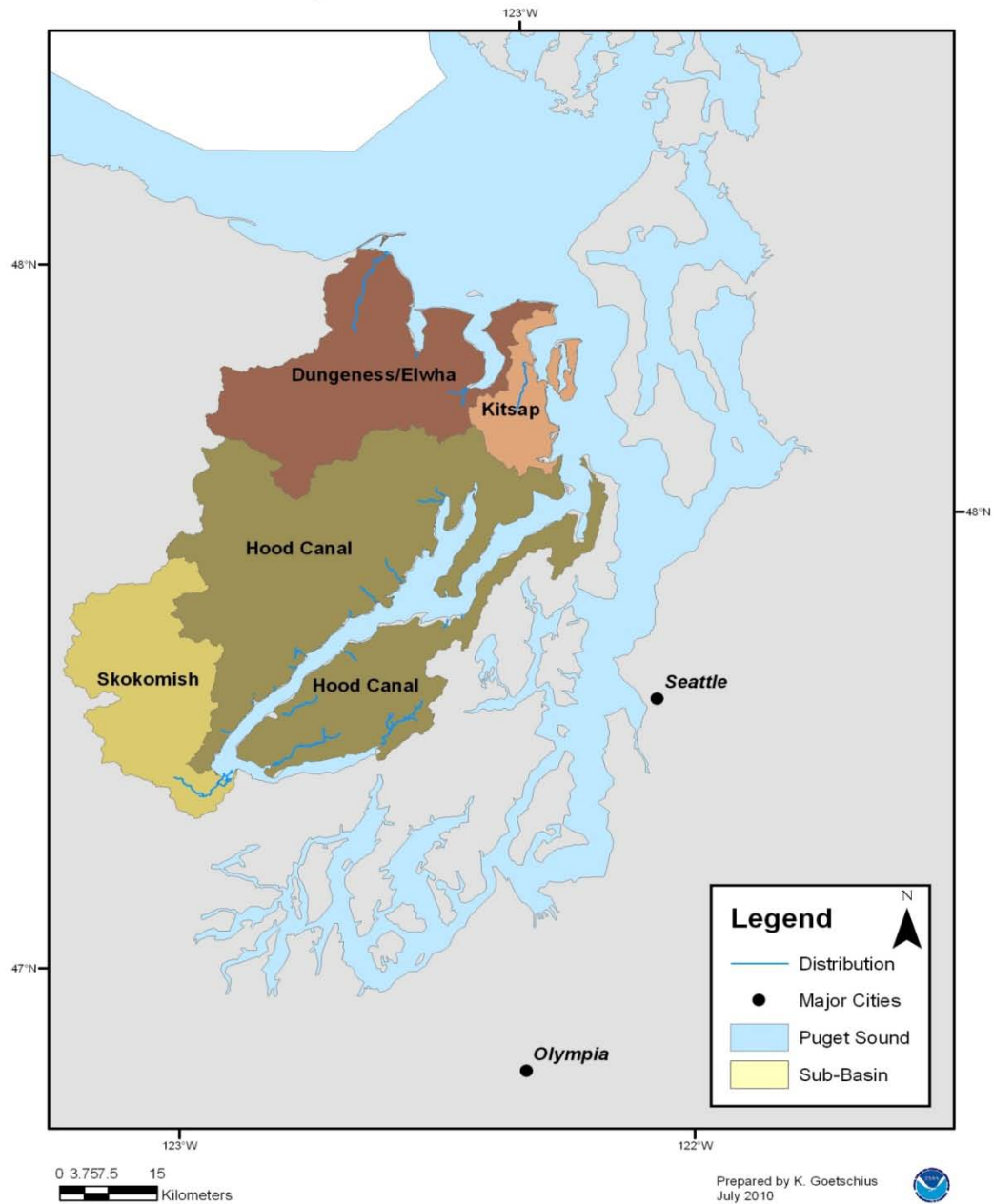


Figure 19. Hood Canal Summer-run Chum salmon distribution.

Life History

Run-timing data from as early as 1913 indicated temporal separation between summer- and fall-run chum salmon in Hood Canal (Johnson, Grant et al. 1997). The HC Summer-run chum salmon enter natal rivers by late August until October (Washington Department of Fish and Wildlife (WDFW) 1993). Spawning occurs from mid-September through mid-October. Adults generally spawn in low gradient, lower mainstem reaches of natal streams, typically in center channel areas due to the low flows encountered in the late summer and early fall. Eggs incubate in redds for five to six months and fry emerge between January and May. After hatching, fry move rapidly downstream to subestuarine habitats. HC Summer-run chum salmon seem to have a longer incubation time than fall-run chum salmon in the same streams. Consequently, offspring of summer-run chum salmon have lower average weight and less lipid content than offspring of fall-run chum salmon. Thus, prey availability during their early life history is important for fry survival.

HC Summer-run chum salmon juveniles quickly migrate up the Hood Canal and into the main body of Puget Sound starting in February/March (Johnson, Grant et al. 1997). The juveniles rear for an average of 23 days in the subestuary deltas which support a diverse array of habitats (tidal channels, mudflats, marshes, and eelgrass meadows). These habitats provide essential rearing and transition environments for this ESU and juveniles rear in these habitats before entering the ocean. Fry in Hood Canal have not been observed to display daily tidal migrations (Bax 1983). Fry movement is associated with prey availability. Juveniles feed primarily on plankton and epibenthic organisms, while subadults feed on similar items as well as larger prey (including fishes and squid).

Fish may emerge from streams over an extended period; some juveniles may remain in Quilcene Bay for several weeks. Most adults return as spawners as three- and four-year old fish.

Status and Trends

NMFS listed HC Summer-run chum salmon as threatened on March 25, 1999 (64 FR 14508), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). The HC extant summer-run chum ESU consists of two historic independent populations (the Strait of Juan de Fuca and Hood Canal populations) that together were constituted of an estimated 16 historic stocks (Sands, Rawson et al. 2007). Of the 16 historic stocks, seven are considered extirpated. With the extirpation of many local stocks, much of the historical spatial structure has been lost on both the population and the ESU level. Most of the extirpated stocks occurred on the eastern side of Hood Canal, which affects the current spatial structure of the ESU. The widespread loss of estuary and lower floodplain habitat continue to impact the ESU's spatial structure and connectivity.

The Strait of Juan de Fuca population includes three extant stocks that spawn in rivers and streams entering the eastern Strait of Juan de Fuca and Admiralty Inlet. The Hood Canal population consists of six extant stocks within the Hood Canal watershed. HC Summer-run chum salmon are part of an extensive rebuilding program developed and implemented in beginning in 1992 by the state and tribal co-managers. The largest supplemental program occurs at the Big Quilcene River fish hatchery. Reintroduction programs occur in Big Beef (Hood Canal population) and Chimacum (Strait of Juan de Fuca population) creeks. All hatchery fish are marked and can be distinguished from naturally produced fish. There is concern that the Quilcene hatchery stock has high rates of straying, and may represent a risk to historical population structure and diversity.

Adult returns for some of the HC Summer-run chum salmon stocks showed modest improvements in 2000, with upward trends continuing in 2001 and 2002. The recent five-year mean abundance is variable among stocks, ranging from one fish to nearly 4,500 fish. Two stocks (Quilcene and Union River) are above the conservation thresholds established by the rebuilding plan. However, most stocks remain depressed. Estimates of the fraction of naturally spawning hatchery fish exceed 60% for some stocks. This indicates that reintroduction programs are supplementing the numbers of

total fish spawning naturally in streams. Both the Strait of Juan de Fuca and the Hood Canal populations have long-term trends above replacement; long-term lambda values range from 0.85 to 1.39 (Good, Waples et al. 2005). Long-term trends in productivity are above replacement only for the Quilcene and Union River stocks.

Critical Habitat

Critical habitat for this species was designated on September 2, 2005 (70 FR 52630). Of 11 watersheds reviewed in NMFS' assessment of critical habitat for the Hood Canal Summer-run chum salmon ESU (Figure 20), nine watersheds were rated as having a high conservation value while three were rated as having a medium value for conservation (Table 26). Five nearshore marine areas were also given a high conservation value rating. None of the watersheds was considered to be of a low conservation value, primarily because approximately half of the historical populations in this ESU have been extirpated, and the remaining populations are limited to only about 60 stream miles. Many of the watersheds have less than four miles of spawning habitat and none of them have more than 8.5 miles.

Table 26. Hood Canal Summer-run chum salmon watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Skokomish	0		1	(1, 3)	0	
Hood Canal	6	(1, 3)	1	(1) ²	0	
Kitsap	1	(1)	0		0	
Dungeness/Elwha	2	(1)	1	(3, 1)	0	
Total	9		3		0	

¹ Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

Spawning PCE is degraded by excessive fine sediment in the gravel. Rearing PCE is degraded by loss of access to sloughs in the estuary and nearshore areas and excessive predation. Low river flows in several rivers also adversely affect most PCEs. In the estuarine areas, both migration and rearing PCEs of juveniles are impaired by loss of

functional floodplain areas necessary for growth and development of juvenile chum salmon. These degraded conditions likely maintain low population abundances across the ESU.

Hood Canal Summer-Run Chum ESU
Conservation Value of HUC 5 Watersheds

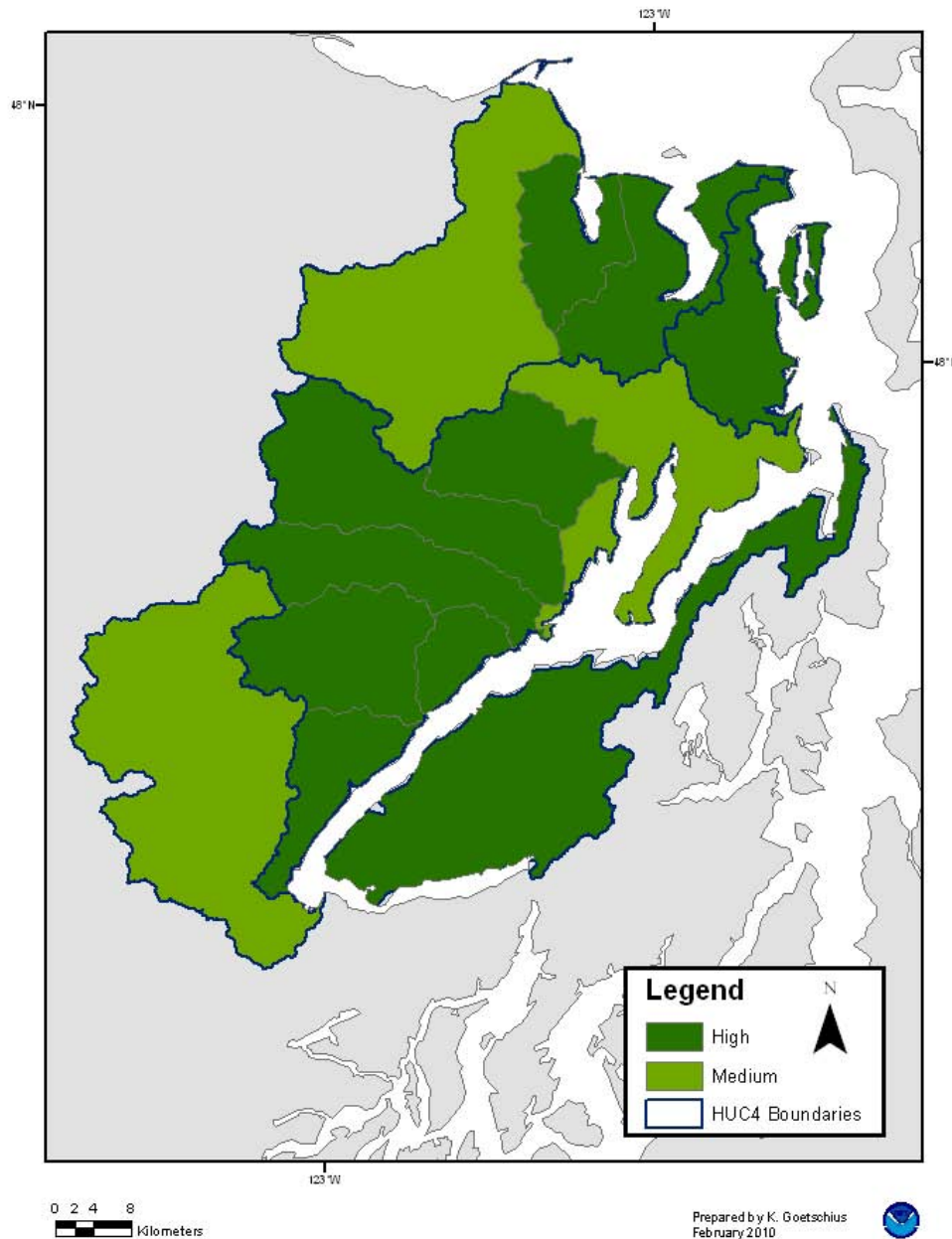


Figure 20. Hood Canal Summer-run Conservation Values per Sub-area.

Columbia River Chum Salmon

Columbia River (CR) chum salmon includes all natural-origin chum salmon in the Columbia River and its tributaries in Oregon and Washington. The species consists of two populations: Grays River and Lower Gorge in Washington State (Figure 21). This ESU also includes three artificial hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

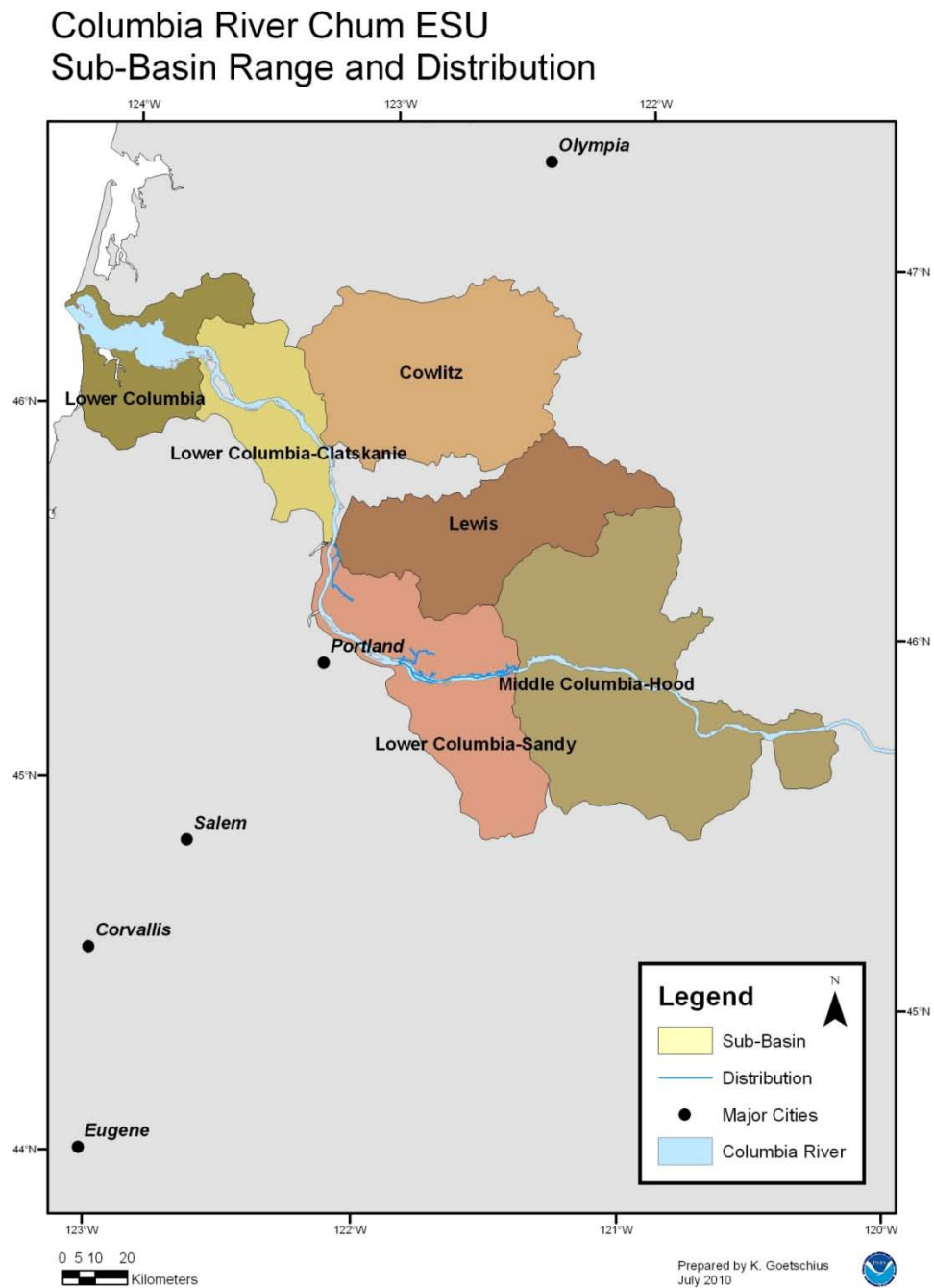


Figure 21. Columbia River Chum salmon distribution.

Table 27. Populations within the Columbia River chum salmon ESU, their abundances, and hatchery input (Good, Waples et al. 2005).

Current Populations	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Youngs Bay	Unknown	Not reported	0
Grays River	7,511	3,832 and 2,720*	Unknown
Big Creek	Unknown	Not reported	0
Elochoman River	Unknown	Not reported	0
Clatskanie River	Unknown	Not reported	0
Mill, Abernathy, and German Creeks	Unknown	Not reported	0
Scappoose Creek	Unknown	Not reported	0
Cowlitz River	141,582	Not reported	0
Kalama River	9,953	Not reported	0
Lewis River	89,671	Not reported	0
Salmon Creek	Unknown	Not reported	0
Clackamas River	Unknown	Not reported	0
Sandy River	Unknown	Not reported	0
Washougal River	15,140	Not reported	0
Lower gorge tributaries	>3,141	425	0
Upper gorge tributaries	>8,912	137 and 223*	0

* Salmon Scape Statistics Query 2009: Estimated total number of natural spawners for the years 2007 and 2008.

Life History

Chum salmon return to the Columbia River in late fall (mid-October to December). They primarily spawn in the lower reaches of rivers, digging redds along the edges of the mainstem and in tributaries or side channels. Some spawning sites are located in areas where geothermally-warmed groundwater or mainstem flow upwells through the gravel.

Chum salmon fry emigrate from March through May shortly after emergence. Juvenile chum salmon reside and feed in estuaries before beginning their long distance oceanic migration. Chum salmon may choose either the upper or lower estuaries depending on the relative productivity of each. The timing of entry of juvenile chum salmon into sea water is correlated with the warming of the nearshore waters and the accompanying plankton blooms (Burgner 1991). The movement offshore generally coincides with the decline of inshore prey resources and when fish have grown to a size that allows them to

feed upon neritic organisms and avoid predators (Burgner 1991). The period of estuarine residence is a critical life history phase and plays a major role in determining the size of the subsequent adult run back to fresh water.

Status and Trends

NMFS listed CR chum salmon as threatened on March 25, 1999, and reaffirmed their threatened status on June 28, 2005 (71 FR 37160). Regarding spatial structure, historically this ESU was highly prolific; CR chum salmon were reported in almost every river in the Lower Columbia River basin. However, few CR chum salmon have been observed in tributaries between the Dalles and Bonneville dams in recent years. Chum salmon were not observed in any of the upper gorge tributaries, including the White Salmon River, during the 2003 and 2004 spawning ground surveys. Surveys of the White Salmon River in 2002 found only one male and one female carcass; the female had not spawned (Ehlke and Keller 2003). However, in the Cascades, chum salmon sampled from each tributary recently appeared as remnants of genetically distinct populations (Greco, Capri et al. 2007).

Historically, the ESU was composed of 17 populations in Oregon and Washington between the mouth of the Columbia River and the Cascade crest (Myers, Busack et al. 2006)

(Table 27). Only two populations with any significant spawning remain today, both on the Washington side (Good, Waples et al. 2005). They are the Grays River and the Lower Gorge (which include Hardy and Hamilton Creeks) populations (Good, Waples et al. 2005). In addition, during the first years after 2000, new (or newly discovered) spawning was observed in the Washougal River mainstem and in the Washington side of the Columbia River mainstem below the mouth of Washougal River (Good, Waples et al. 2005). It is unclear whether this spawning has been maintained. An extensive 2000 survey in Oregon streams supports that chum salmon are extirpated from the Oregon portion of this ESU (Good, Waples et al. 2005).

The CR chum salmon runs have declined substantially from historic levels concurrently with the drastic reduction of spawning populations. In the early 1900s, the ESU numbered in the hundreds of thousands to a million returning adults that supported a large commercial fishery in the first half of this century. However, by the 1950s, most runs had disappeared and fisheries landings in later years rarely exceeded 2,000 chum salmon per year ((Fulton 1970), (Marr 1943), (Rich 1942)). During the 1980s and 1990s, the estimated combined abundance of natural spawners for the Lower Gorge, Washougal, and Grays River populations was below 4,000 adults. However, in 2002, the abundance of natural spawners increased to an estimate of total natural spawners exceeding 20,000 adults. The cause of this dramatic increase in abundance is unknown and was not maintained in the following years.

Current ESU abundance is mostly driven by the Lower Gorge and Grays River populations. The estimated size of the Lower Gorge population is at 400-500 individuals, down from a historical level of greater than 8,900 (Good, Waples et al. 2005). A significant increase in spawner abundance occurred in 2001 and 2002 to around 10,000 adults (Good, Waples et al. 2005). However, spawner surveys indicate that the abundance again decreased to low levels during 2003 through 2008 though the spawner surveys may underestimate abundance since the proportion of tributary and mainstem spawning differ between years and the surveys do not include spawners in the Columbia River mainstem ((Good, Waples et al. 2005), (Washington Department of Fish and Wildlife (WDFW) 2009)). In the 1980s, estimates of the Grays River population ranged from 331 to 812 individuals. However, the population increased in 2002 to as many as 10,000 individuals (Good, Waples et al. 2005). Based on data for number of spawners per river mile, this increase continued through 2003 and 2004. However, fish abundance fell again to less than 5,000 fish during the years 2005 through 2008 (Washington Department of Fish and Wildlife (WDFW) 2009).

Estimates of abundance and trends are available only for the Grays River and Lower Gorge populations. The lambda values indicate a long-term downward trend at 0.954 and

0.984, respectively (Good, Waples et al. 2005). The 10-year trend (up to 2001) was negative for the Grays River population and just over 1.0 for the Lower Gorge. Long- and short-term productivity trends for populations are at or below replacement.

Critical Habitat

Critical habitat was originally designated for the CR chum salmon on February 16, 2000 (65 FR 7764) and was re-designated on September 2, 2005 (70 FR 52630). Sixteen of the 19 subbasins reviewed in NMFS' assessment of critical habitat for the CR chum salmon ESU were rated as having a high conservation value (Table 28). The remaining three subbasins were given a medium conservation value (Figure 22). Washington's federal lands were rated as having high conservation value to the species.

Table 28. CR chum salmon watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Middle Columbia/Hood	3	(3)	0		0	
Lower Columbia/Sandy	3	(3, 1)	0		0	
Lewis	2	(3)	0		0	
Lower Columbia/Clatskanie	3	(3, 2, 1)	0		0	
Cowlitz	3	(3)	3	(3)	0	
Lower Columbia	2	(3, 2, 1)	0		0	
Lower Columbia Corridor	all	(3, 1)	0		0	
Total	16		3		0	

¹ Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

Limited information exists on the quality of essential habitat characteristics for CR chum salmon. However, migration PCE has been significantly impacted by dams obstructing adult migration and access to historic spawning locations. Water quality and cover for estuary and rearing PCEs have decreased in quality to the extent that the PCEs are not likely to maintain their intended function to conserve the species.

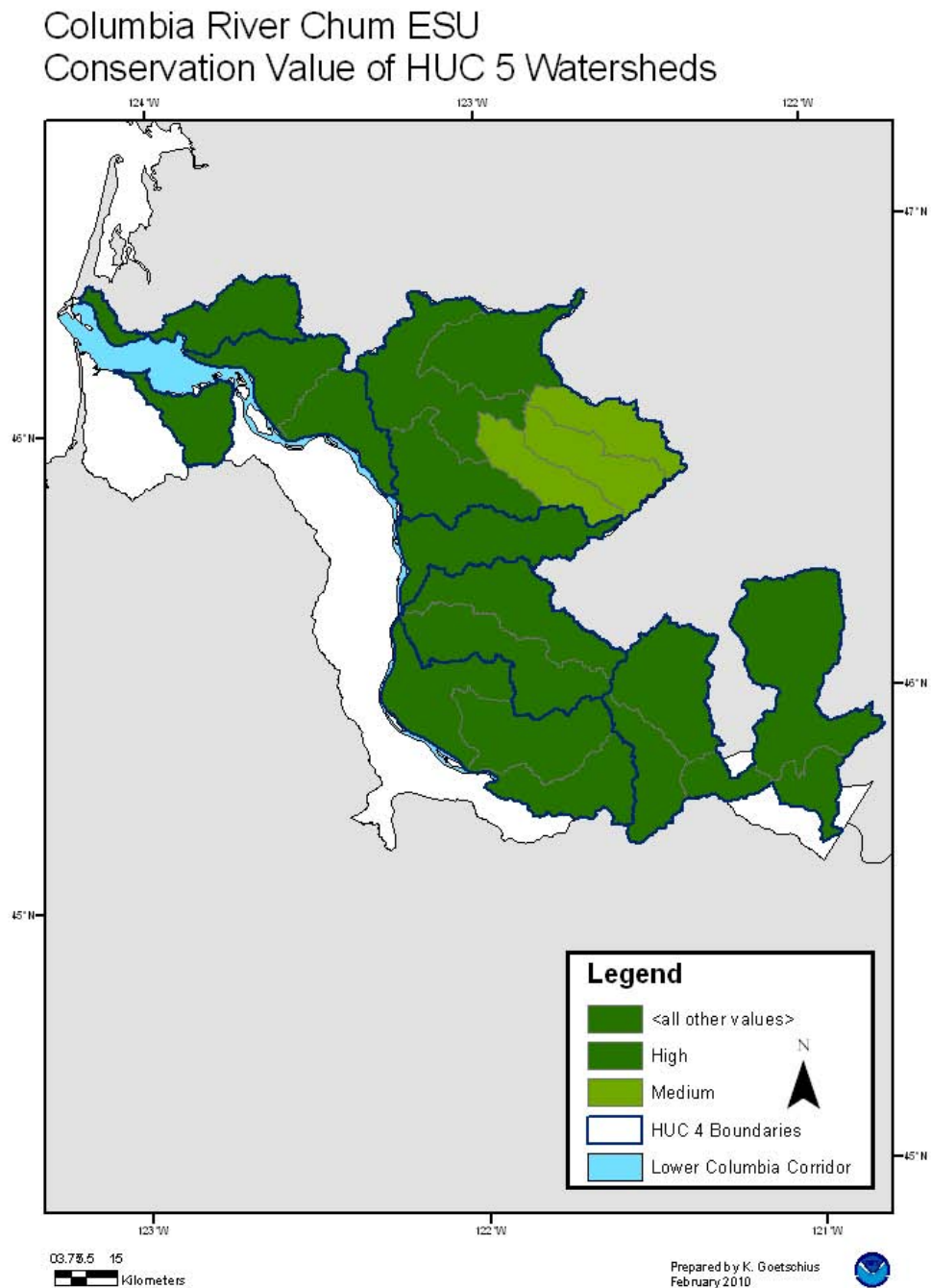


Figure 22. Columbia River Chum salmon Conservation Values per Sub-area.

Coho Salmon

Description of the Species

Coho salmon occur naturally in most major river basins around the North Pacific Ocean from central California to northern Japan (Laufle, Pauley et al. 1986). In this section, we discuss the distribution, life history diversity, status, and critical habitat of the four endangered and threatened coho species separately.

As with other salmon, the coho salmon life cycle consists of a juvenile freshwater phase and a growth phase in the ocean before fish return to rivers to spawn. Along the Oregon/California coast, coho salmon primarily return to rivers to spawn as three-year olds, having spent approximately 18 months rearing in fresh water and 18 months in salt water. In some streams, a smaller proportion of males may return as two-year olds. The presence of two-year old males can allow for substantial genetic exchange between brood years. The relatively fixed three-year life cycle exhibited by female coho salmon limits demographic interactions between brood years. This makes coho salmon more vulnerable to environmental perturbations than other salmonids that exhibit overlapping generations, *i.e.*, the loss of a coho salmon brood year in a stream is less likely than for other Pacific salmon to be reestablished by females from other brood years.

Most coho salmon enter rivers between September and February. In many systems, coho salmon will have to wait to enter until fall rainstorms have provided the river with sufficiently strong flows and depth. Coho salmon spawn from November to January, and occasionally into February and March. Spawning occurs in a few third-order streams. Most spawning activity occurs in fourth- and fifth-order streams. Spawning generally occurs in tributaries with gradients of 3% or less.

Depending on temperature, egg incubation ranges from 35 to 50 days (Sandercock 1991). Hatchlings remain in the gravel as alevins for several weeks while absorbing the yolk sac before emerging from the gravel. In Oregon coastal streams, total average time from egg

deposition to emergence is 110 days (Sandercock 1991). Following emergence, fry move to areas with weak water currents such as backwaters and shallow areas near the stream banks. As the fry grow, they disperse upstream and downstream to establish and defend territories. Territorial behavior limits summer density in streams and subordinate individuals may congregate in pools (Sandercock 1991).

Juvenile coho salmon commonly rear in small streams less than five ft. wide and occasionally in larger ponds and lakes (Pollock, Pess et al. 2004). Juvenile rearing rarely occurs in tributaries exceeding gradients of 3% although they may move to streams with gradients of 4 to 5%. Preferred water quality consists of water with low turbidity, DO levels of 4 to 9 mg/l, and water temperatures ranging from 10° to 15°C ((Bell 1973), (McMahon 1983)). Growth is slowed down considerably at 18°C and ceases at 20°C ((Bell 1973), (Stein, Reimers et al. 1972)). The likelihood of juvenile coho salmon occupying habitat that exceed 16.3°C maximum weekly average temperature declines significantly (Welsh, Hodgson et al. 2001).

During spring and summer, the emphasis is on growth and sustained invertebrate forage production and renewal are necessary. During the growth period, coho salmon fry show low risk averseness and position themselves in open water when sufficient food is available ((Bugert, Bjornn et al. 1991), (Giannico 2000), (Reinhardt 1999)). The main prey are primarily drifting aquatic invertebrates produced in interstices of the gravel substrate and in the leaf litter within pools, and drifting terrestrial insects produced in the riparian canopy (Sandercock 1991). Important food organisms include aquatic insects such as chironomid larvae, mayfly, caddisflies, and stonefly. Coho salmon juveniles also feed opportunistically on non-insects, such as small fish and salmon eggs, and terrestrial insects.

Studies of stream habitat use show that there are a velocity threshold for rearing fry and juveniles. Juveniles prefer focal positions that have water velocity less than 20 cm/s (with a preference of 3 – 6 cm/s) with faster flowing adjacent areas with high food

renewal through drift ((Beecher, Caldwell et al. 2002), (Fausch 1984), (Fausch 1993), (Rosenfeld, Porter et al. 2000), (Shirvell 1990)). High food abundance (*i.e.*, drift) may increase the potential for net energy gain at higher velocities, allowing fish to move into faster waters where fish experience higher growth rate despite the greater swimming costs ((Giannico and Healey 1999), (Rosenfeld, Leiter et al. 2005)). High prey availability also reduces territory size and may increase a stream's rearing capacity ((Dill and Fraser 1984), (Dill, Ydenberg et al. 1981), (Mason 1976)). Reduction in food availability reduces growth by subdominants and less for dominant fish (Rosenfeld, Leiter et al. 2005).

Coho salmon juveniles seek river margins, backwater, and pools during fall and winter; they are rarely found in mid-stream locations of the stream channel during November and February ((Bilby and Bisson 1987), (Bilby and Bisson 2001), (Fausch and Northcote 1992), (Tschaplinski and Hartman 1983)). High densities of juvenile coho salmon also occur in log jams ((Brown 1985), (Tschaplinski and Hartman 1983)). In early fall with the onset of the first seasonal freshets, a large portion of the juvenile population may also migrate to overwinter in off-channel habitat such as larger pools, beaver ponds, off-stream side channels and alcoves, ephemeral swamps, and inundated floodplains ((Brown 1985), (Bustard and Narver 1975), (Nickelson, Rodgers et al. 1992), (Nickelson, Solazzi et al. 1992), (Peterson 1982), (Tschaplinski and Hartman 1983)).

During the winter period, juveniles typically reduce feeding activity and growth rates slow down or stop. In spring, juvenile activity increases. By March of their second spring, the juveniles feed heavily on insects and crustaceans and grow rapidly before smoltification and outmigration (Olegario 2006). Juveniles that overwinter in off-channel habitat, ephemeral streams, and floodplains often experience higher survival and growth than juveniles that overwinter in mainstream channels ((Brown 1985; Dolloff 1987), (Olegario 2006), (Swales, Caron et al. 1988; Quinn and Peterson 1996)).

Availability of suitable overwintering habitat has been suggested to determine smolt production in streams ((Bustard and Narver 1975), (Nickelson, Rodgers et al. 1992)). Adult return or smolt production is related to the area of wetlands, lakes, and ponds within watersheds ((Beechie, Beamer et al. 1994), (Pess, Montgomery et al. 2002), (Sharma and Hilborn 2001)).

Coho salmon juveniles usually migrate to the ocean as smolts in their second spring. Relative to species such as chum salmon, Chinook salmon, and steelhead, coho salmon smolts usually spend a short time in the estuary with little feeding ((Magnusson and Hilborn 2003), (Thorpe 1994)). Estuarine residence times can average one to three days (Miller and Sadro 2003). However, some coho salmon fry may migrate to and rear in the tidally influenced portions of the stream. In one Oregon stream, a portion of the coho salmon fry were observed remaining in the upper estuary to rear after moving into the estuary during their first spring (Miller and Sadro 2003).

After entering the ocean, immature coho salmon initially remain in nearshore waters close to the parent stream. North American coho salmon will migrate north along the coast in a narrow coastal band that broadens in southeastern Alaska. During this migration, juvenile coho salmon tend to occur in both coastal and offshore waters.

Status and Trends

Coho salmon depend on the quantity and quality of the freshwater aquatic systems for spawning, rearing, and on the ocean conditions where they grow to maturity. Coho salmon have declined from overharvests, hatchery supplementation, native and non-native species, dams, gravel mining, water diversions, the destruction or degradation of riparian habitat, and land use practices (logging, agriculture, and urbanization).

Lower Columbia River (LCR) Coho Salmon

The LCR coho salmon include all naturally spawned populations of coho salmon in the Columbia River and its tributaries in Oregon and Washington, from the mouth of the

Columbia up to and including the Big White Salmon and Hood Rivers, Washington, and the Willamette River to Willamette Falls, Oregon (Figure 23). This ESU also includes 25 artificial propagation programs (70FR 37160).

Lower Columbia River Coho ESU Sub-Basin Range And Distribution

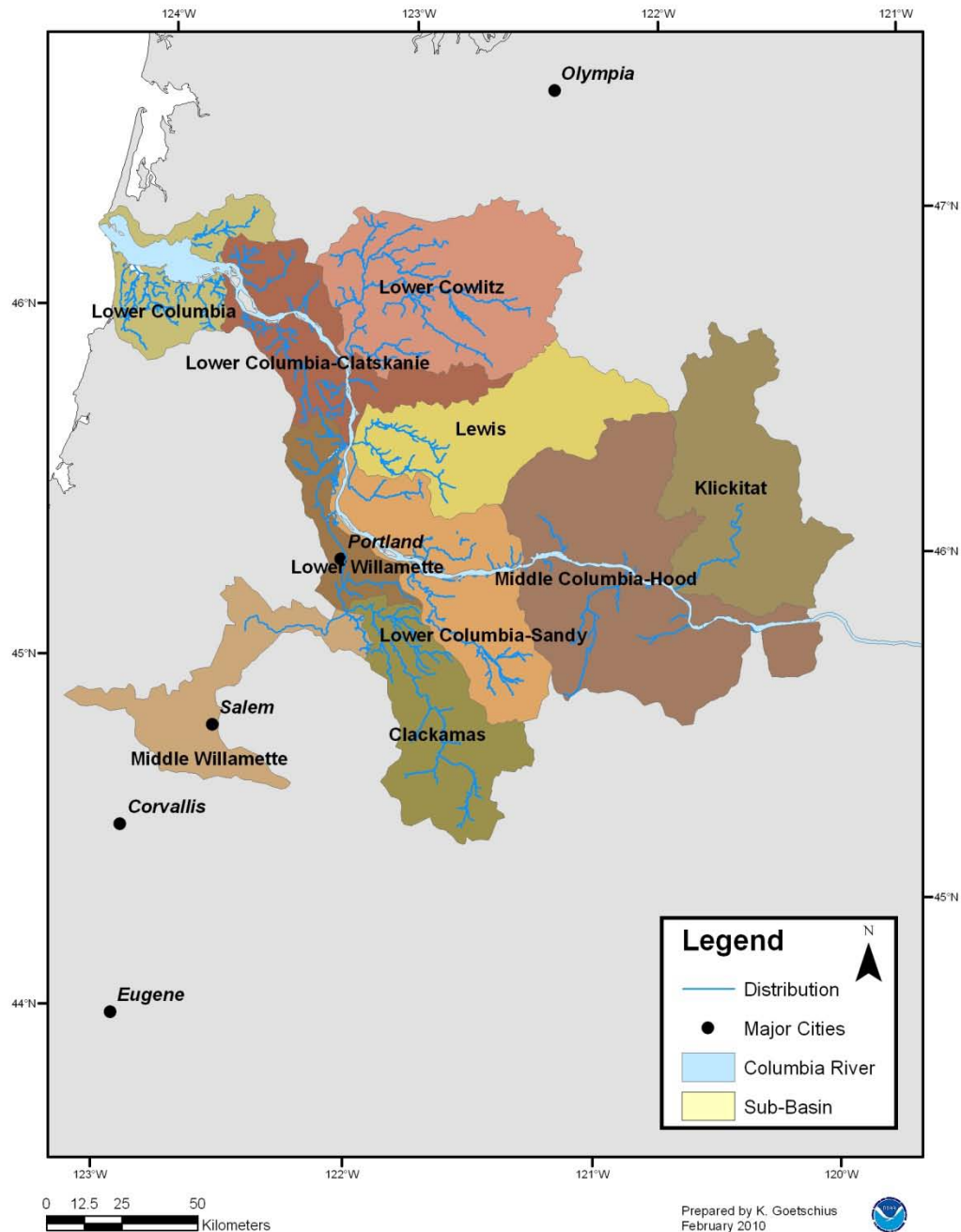


Figure 23. LCR coho salmon distribution.

Life History

The majority of the LCR coho salmon are of hatchery origin. Hatchery runs are currently managed for two distinct runs: early returning (Type S) and late returning (Type N) (Johnson, Flagg et al. 1991). Type S coho salmon return to fresh water in mid-August and to the spawning tributaries in early September. Spawning peaks from mid-October to early November. Type N coho salmon return to the Columbia River from late September through December and enter the tributaries from October through January. Most Type N spawning occurs from November through January.

Analysis of run timing of coho salmon suggests that the Clackamas River population is composed of one later returning population and one early returning population. The late returning population is believed to be descended from the native Clackamas River population. The early returning population is believed to descend from hatchery fish introduced from Columbia River populations outside the Clackamas River basin (Good, Waples et al. 2005). The naturally produced coho salmon return to spawn between December and March (Johnson, Flagg et al. 1991).

Fry emerge from the redds during a three-week period between early March and late July. The juveniles rear in fresh water for a year and smolt outmigration occurs from April through June with a peak in May. Smolts migrate through the Columbia River estuary during dusk and dawn. During movement they are found in mid-river areas of the estuary. However, during mid-morning to late afternoon they reside near the shores of the estuary (Johnson, Flagg et al. 1991).

Status and Trends

NMFS listed the LCR coho salmon as threatened on June 28, 2005 (70 FR 37160). The LCR coho salmon ESU historically consisted of 25 independent populations. The vast majority (over 90%) of these are either extirpated or nearly so (Table 29). Today, only 2 of the 25 populations have any significant natural production in the Sandy and Clackamas Rivers. In addition, wild coho salmon have re-appeared in two additional basins

(Scappoose and Clatskanie) after a 10-year period during the 1980s and 1990s when they were largely absent (McElhany, Chilcote et al. 2007).

Table 29. Lower Columbia River coho salmon populations, estimated natural spawner abundances, and hatchery contributions ((Good, Waples et al. 2005), (McElhany, Chilcote et al. 2007)).

River/Region	Historical Abundance	2002-2004 Spawner Abundance ¹ : Max/Geometric mean	Hatchery Abundance Contributions
Youngs Bay and Big Creek	Unknown	~4,470/200	91%
Grays River	Unknown	Unknown	Unknown
Elochoman River	Unknown	Unknown	Unknown
Clatskanie River	Unknown	~550/286	0-80%
Mill, Germany, and Abernathy creeks	Unknown	Unknown	Unknown
Scappoose Rivers	Unknown	~850/470	0%
Cispus River	Unknown	Unknown	Unknown
Tilton River	Unknown	Unknown	Unknown
Upper Cowlitz River	Unknown	Unknown	Unknown
Lower Cowlitz River	Unknown	Unknown	Unknown
North Fork Toutle River	Unknown	Unknown	Unknown
South Fork Toutle River	Unknown	Unknown	Unknown
Coweeman River	Unknown	Unknown	Unknown
Kalama River	Unknown	Unknown	Unknown
North Fork Lewis River	Unknown	Unknown	Unknown
East Fork Lewis River	Unknown	Unknown	Unknown
Upper Clackamas River	Unknown	~1,770/1,264	12%
Lower Clackamas River	Unknown	~1,180/843	78%
Salmon Creek	Unknown	Unknown	Unknown
Upper Sandy River	Unknown	~1,170/720	0%
Lower Sandy River	Unknown	271/?	97%
Washougal River	Unknown	Unknown	Unknown
Lower Columbia River gorge tributaries	Unknown	Unknown	Unknown
Big White Salmon river	Unknown	Unknown	Unknown
Upper Columbia River gorge tributaries	Unknown	1,317/?	>65%
Hood River	Unknown	~600/~230	Unknown

Prior to 1900, the Columbia River had an estimated annual run of more than 600,000 adults with about 400,000 spawning in the lower Columbia River (Johnson, Flagg et al.

1991). By the 1950s, the estimated number of coho salmon returning to the Columbia River had decreased to 25,000 adults or about 5% of historic levels. Massive hatchery releases since 1960 have increased the Columbia River run size. Between 1980 and 1989, the run varied from 138,000 adults to a historic high of 1,553,000 adults. However, only a small portion of these spawned naturally, and available information indicates that the naturally produced portion has continuously declined since the 1950s. The current number of naturally spawning fish during October and late November ranges from 3,000 to 5,500 fish. The majority of these are of hatchery origin. The 1996 to 1999 geometric mean for the late run in the Clackamas River, the only-run which is considered consisting mainly of native coho salmon, was 35 fish.

Both the long- and short-term trend, and lambda for the natural origin (late-run) portion of the Clackamas River coho salmon are negative but with large confidence intervals (Good, Waples et al. 2005). The short-term trend for the Sandy River population is close to 1, indicating a relatively stable population during the years 1990 to 2002 (Good, Waples et al. 2005). The long-term trend (1977 to 2002) for this same population shows that the population has been decreasing (trend=0.54); there is a 43% probability that the median population growth rate (lambda) was less than one.

Hatchery-origin spawners dominate the majority of populations. However, both the upper Clackamas River and the upper Sandy River spawner populations range from zero to very few hatchery origin spawners. Recent reviews by the W/LCRTRT placed most populations in the high to moderate risk category from eroded diversity ((McElhany, Backman et al. 2004), (McElhany, Busack et al. 2006)).

Critical Habitat

NMFS has not designated critical habitat for Lower Columbia River coho salmon.

Oregon Coast Coho Salmon

The Oregon Coast (OC) coho salmon ESU includes all naturally spawned populations of coho salmon in Oregon coastal streams south of the Columbia River and north of Cape Blanco (63 FR 42587; August 10, 1998, Figure 24). One hatchery stock, the Cow Creek (ODFW stock # 37) hatchery coho, is included in the ESU. This artificially propagated population is no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Life History

The OC coho salmon exhibit the general three year life cycle as described above. Two-year old males commonly occur in some streams and on average make up 20% of spawning males. However, the proportion of two-year old males is highly variable between years and river systems.

There is some variation in run timing between Oregon watersheds but adults generally start to migrate into rivers at the first fall freshet, usually in late October or early November. A delay in rain can delay river entry considerably. Once in the stream, some coho may spend up to two months in fresh water before spawning. Spawning usually occurs from November through January and may continue into February. Juveniles emerge from the gravel in spring and typically spend a summer and winter in fresh water before migrating to the ocean as smolts, usually in April or May, in their second spring. However, the timing varies between years, among river systems, and based on small-scale habitat variability (Lawson, Bjorkstedt et al. 2007). Coastal coho salmon spend little time in estuarine environments during outmigration. Once in coastal waters, the OC coho salmon eventually move northward. By late summer, juveniles are observed distributed off the mouth of Columbia River and the Washington Coast. In fall and winter juvenile coho salmon continue to move northward and have been caught off the coast of Alaska (Lawson, Bjorkstedt et al. 2007). Southward movement starts in winter or early spring with adults starting to home to natal streams by August.

Oregon Coast Coho ESU Sub-Basin Range and Distribution

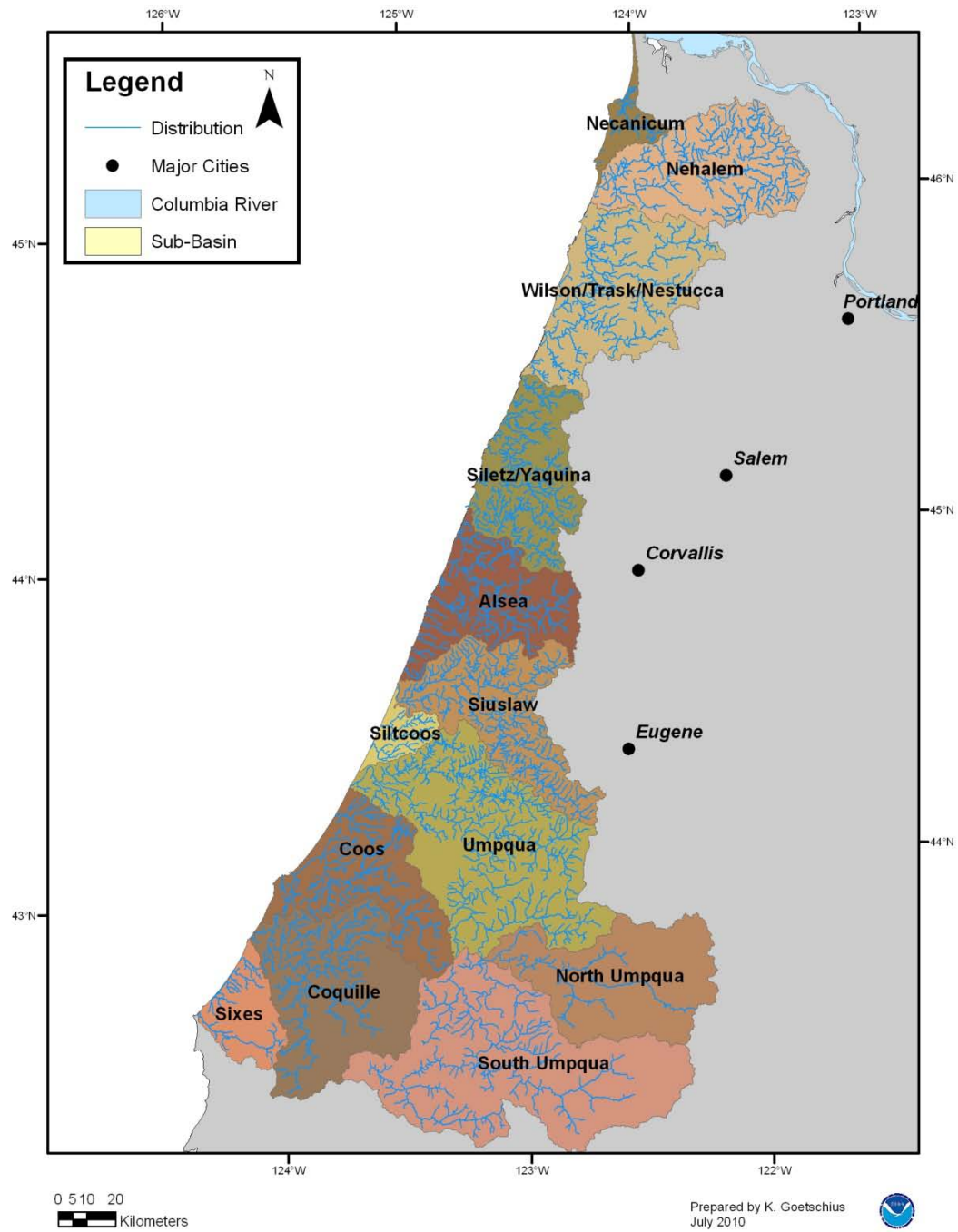


Figure 24. Oregon Coast Coho salmon distribution.

Status and Trends

NMFS listed the OC coho salmon as a threatened species on February 11, 2008 (73 FR 7816). Lawson *et al.* (Lawson, Bjorkstedt et al. 2007) considered the ESU to have historically consisted of 13 functionally independent populations and 8 potentially dependent populations. Current coho salmon coastal distribution has not changed markedly compared to historical distribution (Lawson, Bjorkstedt et al. 2007). However, river alterations and habitat destruction have significantly modified use and distribution within several river basins.

The OC coho salmon historical escapement in the 10 larger basins has been estimated to about 2.4 to 2.9 million spawners (from Table C-1 in (Lawson, Bjorkstedt et al. 2007)). Recent ESU abundances have decreased drastically since then. The estimated median spawning population during the years 1990 to 1999 was 43,183 (min. 21,279, max. 74,021) coho salmon spawners in the ESU (ODFW 2009). After 1999, total ESU abundance increased. A median of 165,324 native OC coho salmon spawners was estimated for the

Table 30. Oregon Coast Coho salmon potential historic and estimated recent spawner abundances, and hatchery contributions ((Lawson, Bjorkstedt et al. 2007), (Good, Waples et al. 2005)).

Basin	Population historic status	Historic Abundance	Recent Spawner Abundance	Hatchery Abundance Contributions
Necanicum	P-I	68,500	1,889	35-40%
Nehalem	F-I	333,000	18,741	40-75%
Tillamook	F-I	329,000	3,949	30-35%
Nestucca	F-I	104,000	3,846	~5%
Siletz	F-I	122,000	2,295	~50%
Yaquina	F-I	122,000	3,665	~25%
Alsea	F-I	163,000	3,621	~40%
Siuslaw	F-I	267,000	16,213	~40%
Umpqua	F-I*	820,000	24,351	<10%
Siltcoos and Tahhenitch	P-I	100,000	15,967**	0%
Tenmile	P-I	53,000	3,251**	0%

Coos	F-I	206,000	20,136	<5%
Coquille	F-I	417,000	8,847	<5%
Total		924,000	107,553	

*The Umpqua River basin is believed to have supported four functionally independent populations.

** Abundance in 2002, ODFW data <http://oregonstate.edu/dept/ODFW/spawn/data.htm>

F-I = Functionally Independent, P-I = Potentially Independent.

period 2000 through 2008 with a range from a low of 66,169 to a high of 260,000

naturally produced spawners. Table 30 identifies independent populations within the OC coho salmon ESU, historic and recent abundances, and hatchery input.

The abundance and productivity of OC coho salmon since the 1997 status review represented some of the best and worst years on record (Good, Waples et al. 2005). Yearly adult returns for this ESU were in excess of 160,000 natural spawners in 2001 and 2002. However, these encouraging increases in spawner abundance in 2000–2002 were preceded by three consecutive brood years (the 1994–1996 brood years returning in 1997–1999, respectively) exhibiting recruitment failure. Recruitment failure is when a given year class of natural spawners fails to replace itself when its offspring return to the spawning grounds three years later. At the time of the 2005 status report, these three years of recruitment failure were the only such instances observed thus far in the entire 55-year abundance time series for OC coho salmon (Good, Waples et al. 2005). The encouraging 2000–2002 increases in natural spawner abundance were primarily observed in populations in the northern portion of the ESU (Good, Waples et al. 2005). Although encouraged by the increase in spawner abundance in 2000–2002, the long-term trends in ESU productivity remained negative due to the low abundances observed during the 1990s (Good, Waples et al. 2005).

Recent data indicate that the total abundance of natural spawners in the OC coho salmon ESU again steadily decreased until 2007 with an estimated spawner abundance of 66,169 fish or approximately 25% of the 2002 peak abundance (260,555 spawners) (ODFW 2009). Thus, recruitment failed during the five years from 2002 through 2007 but abundance increased again in 2008 to 165,324 spawners. There is no apparent weak brood year for the ESU (ODFW 2009).

Critical Habitat

NMFS designated critical habitat for Oregon Coast coho salmon on February 11, 2008 (73 FR 7816). The designation includes 72 of 80 watersheds and total about 6,600 stream miles including all or portions of the Nehalem, Nestucca/Trask, Yaquina, Alsea, Umpqua, and Coquille basins.

There are 80 watersheds within the range of this ESU. Eight watersheds received a low conservation value rating, 27 received a medium rating, and 45 received a high rating to the ESU (Table 31, and Figure 25).

Table 31. OC coho salmon watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Necanicum	0		1	(1, 2)	0	
Nehalem	5	(1, 2)	0		1	(2, 1)
Wilson/Trask/Nestucca	7	(1, 2)	2	(1, 2)	0	
Siletz/Yaquina	3	(1, 2)	5	(1, 2)	0	
Alsea	4	(1, 2)	3	(1, 2)	1	(1, 2=1.5mi)
Siuslaw	6	(1, 2, <3)	2	(1, 2)	0	
Siltcoos	1	(2, 1)	0		0	
North Umpqua	1	(1, <2)	3	(1, 3, <2)	3	(1)
South Umpqua	3	(1, <2, <<3)	8	(1, 2, 3)	1	(1)
Umpqua	6	(1, 3, 2)	1	(1, 3)	1	(1, 2, 3)
Coos	4	(1, 2, <3)	0		0	
Coquille	4	(1, 2, 3))	1	(1, 2)	1	(1, 2)
Sixes	1	(1, 20)	1	(1, 2)		
Total	45		27		8	

¹ Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

The spawning PCE has been impacted in many watersheds from the inclusion of fine sediment into spawning gravel from timber harvest and forestry related activities, agriculture, and grazing. These activities have also diminished the channels' rearing and overwintering capacity by reducing the amount of large woody debris in stream channels,

removing riparian vegetation, disconnecting floodplains from stream channels, and changing the quantity and dynamics of stream flows. The rearing PCE has been degraded by elevated water temperatures in 29 of the 80 HUC 5 watersheds; rearing PCE within the Nehalem, North Umpqua, and the inland watersheds of the Umpqua subbasins have elevated stream temperatures. Water quality is impacted by contaminants from agriculture and urban areas in low laying areas in the Umpqua subbasins, and in coastal watersheds within the Siletz/Yaquina, Siltcoos, and Coos subbasins. Reductions in water quality have been observed in 12 watersheds due to contaminants and excessive nutrition. The migration PCE has been impacted throughout the ESU by culverts and road crossings that restrict passage. As described above the PCEs vary widely throughout the critical habitat area designated for OC coho salmon, with many watersheds heavily impacted with low quality PCEs while habitat in other coho salmon bearing watersheds having sufficient quality for supporting the conservation purpose of designated critical habitat.

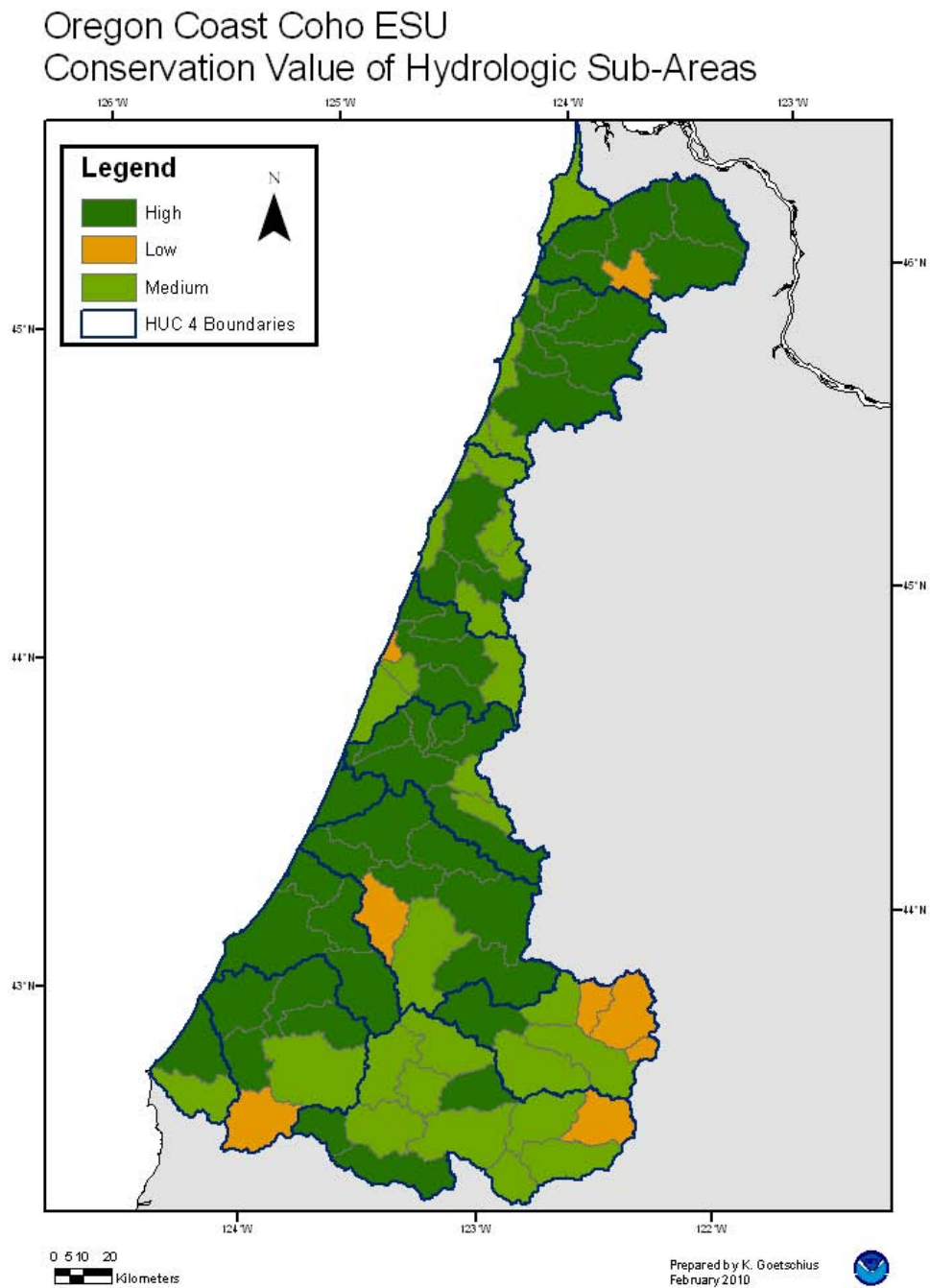


Figure 25. Oregon Coast Coho salmon Conservation Values per Sub-area.

Southern Oregon/Northern California Coast Coho Salmon

The Southern Oregon/Northern California Coast (SONCC) coho salmon ESU consists of all naturally spawning populations of coho salmon that reside below long-term, naturally impassible barriers in streams between Punta Gorda, California and Cape Blanco, Oregon (Figure 26). This ESU also includes three artificial propagation programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Life History

In Oregon, the SONCC coho salmon enter rivers in September or October. River entry is later south of the Klamath River Basin, occurring in November and December, in basins south of the Klamath River to the Mattole River, California. River entry occurs from mid-December to mid-February in rivers farther south. Because coho salmon enter rivers late and spawn late south of the Mattole River, they spend much less time in the river prior to spawning compared to populations farther north. Juveniles emerge from the gravel in spring, and typically spend a summer and winter in fresh water before migrating to the ocean as smolts in their second spring. Coho salmon adults spawn at age three, spending about a year and a half in the ocean.

Southern Oregon Northern California Coho ESU Sub-Basin Range and Distribution

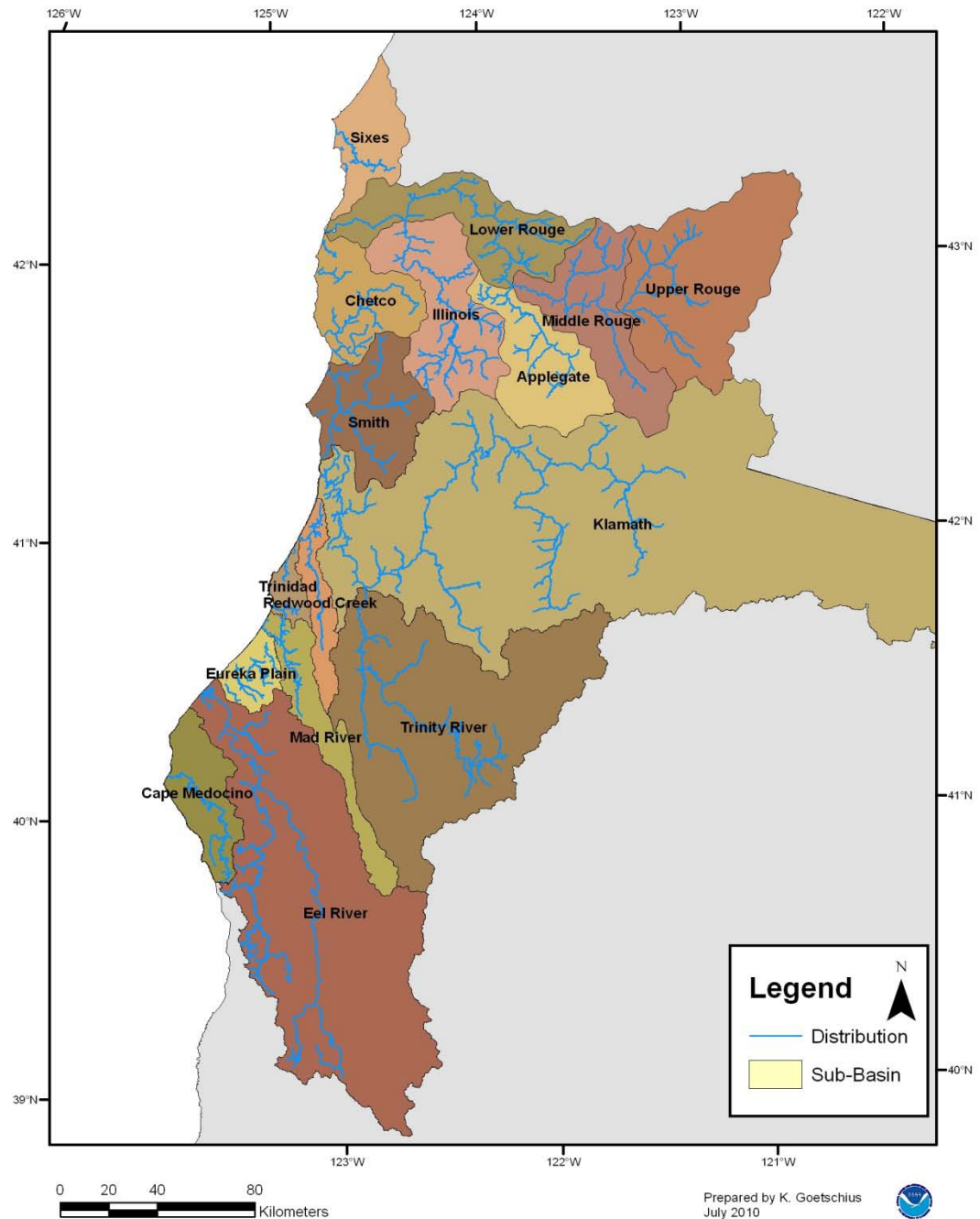


Figure 26. SONCC coho salmon distribution.

Status and Trends

NMFS listed SONCC coho salmon as threatened on May 7, 1997 (62 FR 24588), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160). The ESU consists of three major basins: the Rough (OR), Klamath (OR/CA), and the Eel (CA) Rivers. Three historically independent interior populations have been identified for the Rough River basin, eight for the Klamath River basin, and six in the Eel River basin (Williams, Bjorkstedt et al. 2006). In addition, eight coastal basins within the ESU likely supported functionally independent populations under historical conditions, six basins likely supported potentially independent populations, and 13 supported dependent populations. Presence-absence data indicate a disproportionate loss of southern populations compared to the northern portion of the ESU.

Data on population abundance and trends are limited for this ESU. Historical point estimates of coho salmon abundance for the early 1960s and mid-1980s suggest that California statewide coho spawning escapement in the 1940s ranged between 200,000 and 500,000 fish. Numbers declined to about 100,000 fish by the mid-1960s with about 43% originating from this ESU. Brown *et al.* (Brown, Moyle et al. 1994), estimated that about 7,000 wild and naturalized coho salmon were produced in the California portion of this ESU. Further, presence-absence surveys indicate that the SONCC coho salmon have declined in California compared to past abundances (Good, Waples et al. 2005). Data from surveys in Oregon contrast the California portion of the ESU in that fish presence has been steadily increasing from 1998 through 2007 ((Bennet 2005)), (Good, Waples et al. 2005), (Jepsen and Leader 2008)).

There is no consistent monitoring of any SONCC coho salmon populations. Trend and median population growth for single populations have therefore not been calculated. Information on abundance and production from California streams is limited. However, presence-absence data show that distributions within watersheds have remained

suppressed compared to the historic distribution. Some hatchery releases has occurred but there is not enough information to evaluate the impacts of hatchery on fish diversity.

Critical Habitat

NMFS designated critical habitat for the SONCC coho salmon on May 5, 1999 (64 FR 24049). Species critical habitat encompasses all accessible river reaches between Cape Blanco, Oregon, and Punta Gorda, California and consists of the water, substrate, and river reaches (including off-channel habitats) in specified areas. Accessible reaches are those within the historical range of the ESU that can still be occupied by any life stage of coho salmon. Watersheds within the ESU have not been evaluated for their conservation value.

Critical habitat designated for the SONCC coho salmon is generally of good quality in northern coastal streams. Spawning PCE has been degraded throughout the ESU by logging activities that has increased fines in spawning gravel. Rearing PCE has been considerably degraded in many inland watersheds from the loss of riparian vegetation resulting in unsuitably high water temperatures. Rearing and juvenile migration PCEs have been reduced from the disconnection of floodplains and off-channel habitat in low gradient reaches of streams, consequently reducing winter rearing capacity.

Central California Coast Coho Salmon

The Central California Coast (CCC) coho salmon ESU includes all naturally spawned populations of coho salmon from Punta Gorda in northern California south to and including the San Lorenzo River in central California, as well as populations in tributaries to San Francisco Bay, excluding the Sacramento-San Joaquin River system (Figure 27) The ESU also includes four artificial propagation programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

Life History

In general, coho salmon within California exhibit a three-year life cycle. However, two-year old males commonly occur in some streams. Both run and spawn timing of coho salmon in this region are late (both peaking in January) relative to northern populations, with little time spent in fresh water between river entry and spawning. Spawning runs coincide with the brief peaks of river flow during the fall and winter. Most CCC coho salmon juveniles undergo smoltification and start their seaward migration one year after emergence from the redd. Juveniles spending two winters in fresh water have, however, been observed in at least one coastal stream within the range of the ESU (Bjorkstedt, Spence et al. 2005). Smolt outmigration generally peaks in April and May ((Shapovalov and Taft 1954), (Weitkamp, Wainwright et al. 1995)).

Central California Coastal Coho Sub-Basin Range and Distribution

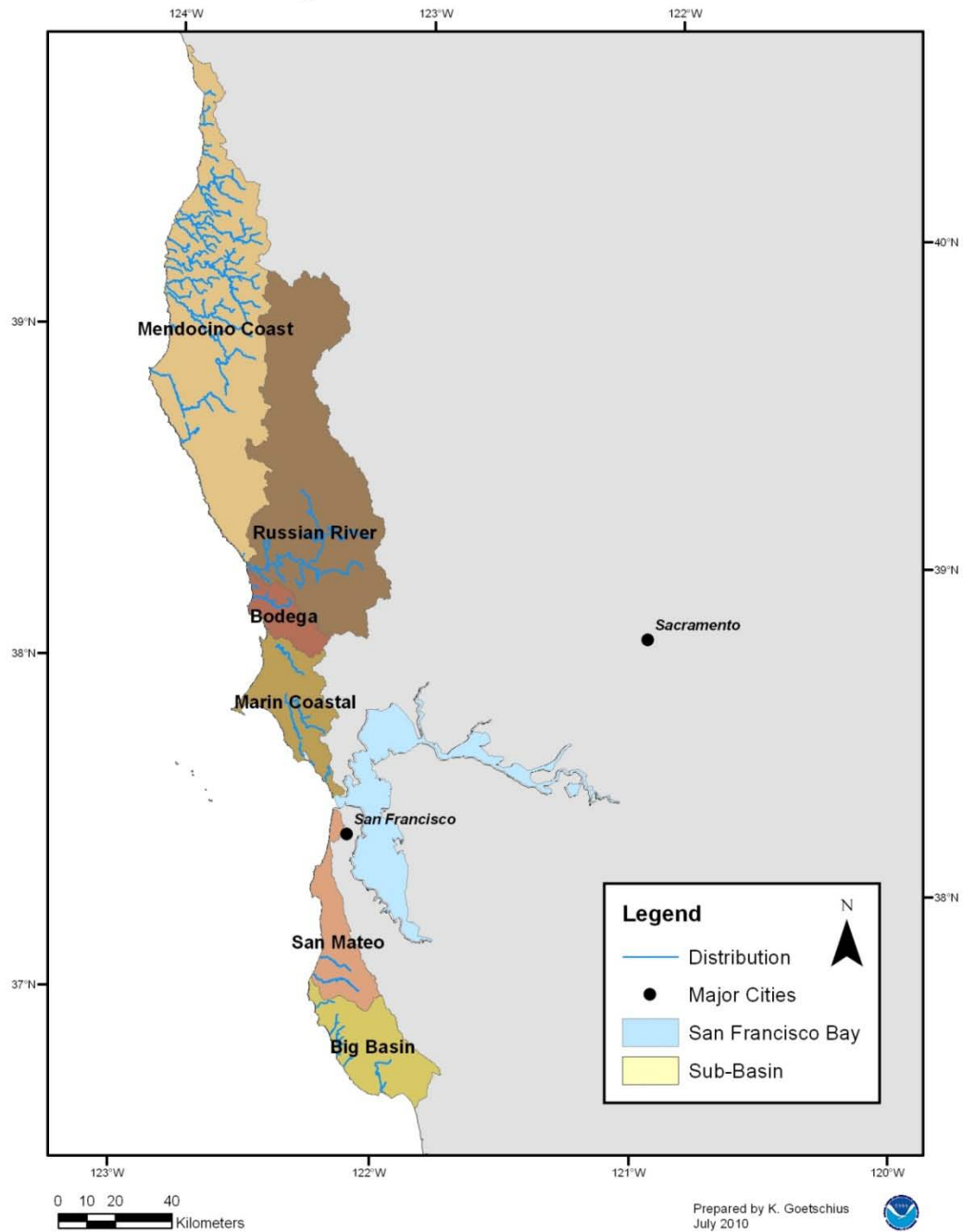


Figure 27. CCC Coho salmon distribution.

Status and Trends

NMFS originally listed the CCC coho salmon as threatened on October 31, 1996 (61 FR 56138), and reclassified their status to endangered on June 28, 2005 (70 FR 37160). The ESU consisted historically of 11 functionally independent populations and a larger number of dependent populations (Spence, Bjorkstedt et al. 2008). ESU spatial structure has been substantially modified due to lack of viable source populations and loss of dependent populations. One of the two historically independent populations in the Santa Cruz mountains (*i.e.*, South of the Golden Gate Bridge) is extirpated ((Good, Waples et al. 2005), (Spence, Bjorkstedt et al. 2008)). Coho salmon are considered effectively extirpated from the San Francisco Bay ((NMFS 2001), (Spence, Bjorkstedt et al. 2008)). The Russian River population, once the largest and most dominant source population in the ESU, is now at high risk of extinction because of low abundance and failed productivity (Spence, Bjorkstedt et al. 2008). The Lost Coast to Navarro Point to the north contains the majority of coho salmon remaining in the ESU.

Limited information exists on abundance of coho salmon within the CCC coho salmon ESU. About 200,000 to 500,000 coho salmon were produced statewide in the 1940s (Good, Waples et al. 2005). This escapement declined to about 99,000 by the 1960s with approximately 56,000 (56%) originating from streams within the CCC coho salmon ESU. The estimated number of coho salmon produced within the ESU in the late 1980s had further declined to 6,160 (46% of the estimated statewide production) (Good, Waples et al. 2005).

Information on the abundance and productivity trends for the naturally spawning component in individual rivers of the CCC coho salmon ESU is extremely limited ((Good, Waples et al. 2005), (Spence, Bjorkstedt et al. 2008)). There are no long-term time series of spawner abundance for individual river systems. Returns increased in 2001 in streams within the northern portion of the ESU (Good, Waples et al. 2005). However,

recent CCC coho salmon returns (2006/07 and 2007/08) have been discouragingly low (McFarlane, Hayes et al. 2008). About 500 fish have returned in 2010 across the entire range. This is the third straight year of abysmal returns for CCC coho salmon. This year's low return suggests that all three year classes are faring poorly across the species' range.

Table 32. Central California Coast Coho salmon populations, abundances, and releases of hatchery raised smolt ((Good, Waples et al. 2005), (Bjorkstedt, Spence et al. 2005)).

River/Region	Historical Escapement (1963)	1987-1991 Escapement Abundance	Hatchery Abundance Contributions*
Ten Mile River	6,000	160	892 – 796,561
Noyo River	6,000	3,740	940,970 – 242,808
Big River	6,000	280	9,988 – 191,310
Navarro River	7,000	300	20,020 – 143,812
Garcia River	2,000	500 (1984-1985)	183,153
Other Mendocino County rivers	10,000	470	Unknown
Gualala River	4,000	200	10,005 – 135,050
Russian River	5,000	255	7,998 – 415,730
Other Sonoma County rivers	1,000	180	Unknown
Marin County	5,000	435	5,760 – 305,421**
San Mateo County	1,000	Unknown	Unknown
San Francisco Bay	Unknown	Extirpated	NA
Santa Cruz County	1,500	50 (1984-1985)	Unknown
San Lorenzo River	1,600	Unknown	17,160 – 145,960
Total	200,000-500,000	6,570 (min)	

*Most coho salmon hatchery contributions have been infrequent and the numbers indicate the range of documented releases. All hatchery data are from Bjorkstedt *et al.* 2005.

**Lagunitas and Walker Creeks

The best data available for the CCC coho salmon are presence-absence surveys and they are used as a proxy for abundance changes (Table 32). At the time of the 1996 listing, coho salmon occurred in about 47% of the streams (62) and were considered extirpated from 53% (71) of the streams that historically harbored coho salmon within the ESU (Brown, Moyle et al. 1994). Later reviews have concluded that the number of occupied streams relative to historic has not changed and may actually have declined ((Good, Waples et al. 2005), (NMFS 2001)).

Hatchery raised smolt have been released infrequently but occasionally in large numbers in rivers throughout the ESU (Bjorkstedt, Spence et al. 2005). Releases have included transfer of stocks within California and between California and other Pacific states as well as smolt raised from eggs collected from native stocks. However, genetic studies show little homogenization of populations, *i.e.*, transfer of stocks between basins have had little effect on the geographic genetic structure of CCC coho salmon (Sonoma County Water Agency (SCWA) 2002). The CCC coho salmon likely has considerable diversity in local adaptations given that the ESU spans a large latitudinal diversity in geology and ecoregions, and include both coastal and inland river basins.

Critical Habitat

Critical habitat for the CCC coho salmon ESU was designated on May 5, 1999 (64 FR 24049). It encompasses accessible reaches of all rivers (including estuarine areas and tributaries) between Punta Gorda and the San Lorenzo River (inclusive) in California. Critical habitat for this species also includes two streams entering San Francisco Bay: Arroyo Corte Madera Del Presidio and Corte Madera Creek. Individual watersheds within the ESU have not been evaluated for their conservation value.

NMFS (NMFS 2008) evaluated the condition of each habitat attribute in terms of its current condition relative to its role and function in the conservation of the species. The assessment of habitat for this species showed a distinct trend of increasing degradation in quality and quantity of all PCEs as the habitat progresses south through the species range, with the area from the Lost Coast to the Navarro Point supporting most of the more favorable habitats and the Santa Cruz Mountains supporting the least. However, all populations are generally degraded regarding spawning and incubation substrate, and juvenile rearing habitat. Elevated water temperatures occur in many streams across the entire ESU.

Sockeye Salmon

Description of the Species

Sockeye salmon occur in the North Pacific and Arctic oceans and associated freshwater systems. This species ranges south as far as the Klamath River in California and northern Hokkaido in Japan, to as far north as Bathurst Inlet in the Canadian Arctic and the Anadyr River in Siberia. We discuss the distribution, life history diversity, status, and critical habitat of the two endangered and threatened sockeye species separately.

Spawning generally occurs in late summer and autumn, but the precise time can vary greatly among populations. Males often arrive earlier than females on the spawning grounds, and will persist longer during the spawning period. Average fecundity ranges from about 2,000 eggs per female to 5,000 eggs, depending upon the population and age of the female.

The vast majority of sockeye salmon spawn in outlet streams of lakes or in the lakes themselves. In lakes, the species commonly spawn along “beaches” where underground seepage creates upwelling that provides eggs and alevins with fresh oxygenated water. Incubation is a function of water temperature, but generally lasts between 100 and roughly 200 days (Burgner 1991). Sockeye salmon fry primarily use lakes as rearing areas with river emerged fry migrating into lakes to rear. Fry emerging in streams emptying into lakes usually move rapidly with the water flow downstream into lakes. Fry emerging from lake-outlet spawning areas migrate upstream into lakes. In these cases, fry hold for a period in the stream and may feed actively before moving upstream into the lake. During upstream migration, they move along the low velocity stream margin. Fry emerging from lakeshore or island spawning grounds distribute along the shoreline of the lake or move offshore into deep water (Burgner 1991). The juvenile sockeye salmon rear in lakes from one to three years after emergence.

Some sockeye spawn in rivers without lake habitat for juvenile rearing. Offspring of these riverine spawners use the lower velocity sections of rivers as juvenile rearing environment for one to two years. Alternatively, juveniles may also migrate to sea in their first year.

Certain populations of *O. nerka* become resident in the lake environment and are called kokanee or little redfish (Burgner 1991). Kokanee and sockeye often co-occur in many interior lakes, where access to the sea is possible but energetically costly. On the other hand, coastal lakes, where the migration to sea is relatively short and energetic costs are minimal, rarely support kokanee populations.

During freshwater rearing, sockeye salmon feeding behavior change as the juvenile transit through stages from emergence to the time of smoltification. As the alevins emerge from gravel, they feed little and depend mostly on the yolk sack, if it is still present, for growth (Burgner 1991). It is therefore critical for the small fry to start feeding as the yolk sack reserves are being depleted; a high mortality is observed when fishes are starved for more than two weeks after yolk absorption (Bilton and Robins 1973). In the earlier fry stage from spring to early summer, juveniles forage exclusively in the warmer littoral (*i.e.*, shoreline) zone where they depend mostly on dipteran insects (mostly chironomidae larvae and pupae) and on cyclopoid copepods and cladocerans. In summer, underyearling sockeye salmon transit from the littoral habitat to a pelagic existence where they feed on larger zooplankton. However, diptera, especially chironomids, can contribute substantially in caloric value. Older and larger fish may also prey on fish larvae. Distribution in lakes and prey preference is, however, a dynamic process that changes diurnally and annually, with water temperature, with the presence and abundance of particular prey species, presence of predators and competitors, and the size of the sockeye salmon juveniles.

Upon smoltification, anadromous sockeye migrate to the ocean. Peak emigration to the ocean occurs in mid-April to early May in southern sockeye populations (<52°N latitude)

and as late as early July in northern populations (62°N latitude) (Burgner 1991). River-type sockeye populations make little use of estuaries during their emigration to the marine environment. Upon entering marine waters, sockeye may reside in the nearshore or coastal environment for several months but are typically distributed offshore by fall (Burgner 1991). Adult sockeye salmon return to their natal lakes to spawn after spending one to four years at sea.

Status and Trends

Sockeye salmon depend on the quantity and quality of aquatic systems. Sockeye salmon, like the other salmon NMFS has listed, have declined from overharvests, hatcheries, native and non-native exotic species; dams, gravel mining, water diversions, destruction or degradation of riparian habitat, and land use practices (logging, agriculture, and urbanization).

Ozette Lake Sockeye Salmon

Distribution

This ESU includes sockeye salmon that migrate into and rear in the Ozette Lake near the northwest tip of the Olympic Peninsula in Olympic National Park, Washington (Figure 28). The Ozette Lake sockeye salmon ESU includes all naturally spawned anadromous populations of sockeye salmon in Ozette Lake, Ozette River, Coal Creek, and other tributaries flowing into Ozette Lake. Composed of only one population, the Ozette Lake sockeye salmon ESU consists of five spawning aggregations or subpopulations which are grouped according to their spawning locations. The five spawning locations are Umbrella and Crooked creeks, Big Rive, and Olsen's and Allen's beaches (Rawson, Sands et al. 2009). Two artificial populations are also considered part of this ESU. These artificially propagated populations are no more divergent relative to the local natural population than would be expected between closely related natural populations (70 FR 37160).

Sockeye salmon stock reared at the Makah Tribe's Umbrella Creek Hatchery were included in the ESU, but were not considered essential for recovery of the ESU. However, once the hatchery fish return and spawn in the wild, their progeny are considered as listed under the ESA.

Ozette Lake Sockeye Watershed Range and Distribution

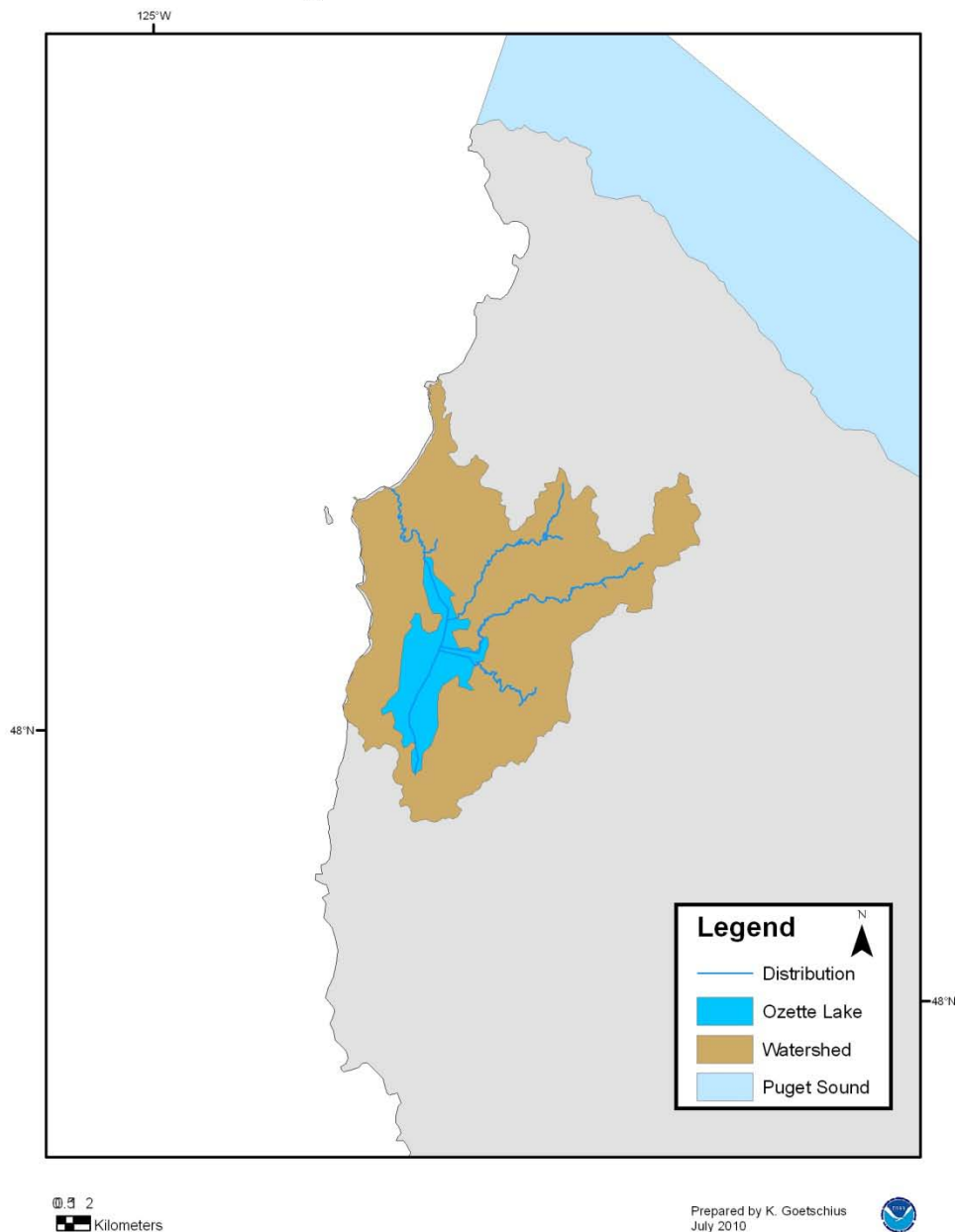


Figure 28. Ozette Lake Sockeye salmon distribution.

Life History

Adult Ozette Lake sockeye salmon enter Ozette Lake through the Ozette River from April to early August. Of these, about 99% are four-year old adults. Adults remain in the lake for an extended period before spawning from late October through February.

Sockeye salmon spawn primarily in lakeshore upwelling areas in Ozette Lake. Minor spawning may occur below Ozette Lake in the Ozette River or in Coal Creek, a tributary of the Ozette River. Native sockeye salmon do not presently spawn in tributary streams to Ozette Lake but they may have spawned there historically. However, a hatchery program has initiated tributary-spawning by hatchery fish in Umbrella Creek and Big River (Good, Waples et al. 2005).

Egg incubation occurs from October through May. Emergence and dispersal in the lake occurs from late-February through May. Fry disperse to the limnetic zone in Ozette Lake, where the fish rear. Tributary fry also migrate to the lake soon after emergence. In their second spring after one year of rearing, Ozette Lake sockeye salmon emigrate seaward as age 1+ smolts. The lake is highly productive and water fleas dominate the diet. Sockeye salmon smolts produced in Ozette Lake are documented as the third largest, averaging 4 ½ to 5 inches in length, among west coast sockeye populations examined for average smolt size. The majority of Ozette Lake sockeye salmon return to spawn after two years in the ocean (NMFS 2008). Ozette Lake also supports a population of kokanee which is not listed under the ESA. There is a large genetic difference between the anadromous and the resident *O. nerka* populations (Crewson, Haggerty et al. 2001).

Status and Trends

NMFS originally listed the Ozette Lake sockeye salmon as a threatened species in 1999 (64 FR 14528), and reaffirmed their threatened status on June 28, 2005 (70 FR 37160).

The Ozette Lake sockeye salmon ESU is composed of one historical population, with substantial substructuring of individuals into multiple spawning aggregations.

Historically at least four beaches in the lake were used for spawning but only two beach spawning locations – Allen’s and Olsen’s beaches – remain today.

The historical abundance of Ozette Lake sockeye salmon is poorly documented, but may have been as high as 50,000 individuals (Blum 1988). Kemmerich (Kemmerich 1945), reported a decline in the run size since the 1920s weir counts and Makah Fisheries Management (Makah Fisheries Management 2000) concluded a substantial decline in the Tribal catch of Ozette Lake sockeye salmon occurred at the beginning of the 1950s. Whether decrease in abundance compared to historic estimates is a result of fewer spawning aggregations, lower abundances at each aggregation, or both, is unknown (Good, Waples et al. 2005).

The most recent (1996-2006) escapement estimates (run size minus broodstock take) range from a low of 1,404 in 1997 to a high of 6,461 in 2004, with a median of approximately 3,800 sockeye per year (geometric mean: 3,353) (Rawson, Sands et al. 2009). No statistical estimation of trends is reported. However, comparing four year averages (to include four brood years in the average since the species primarily spawn as four-year olds) shows an increase during the period 2000 to 2006: For return years 1996 to 1999 the run size averaged 2,460 sockeye salmon, for the years 2000 to 2003 the run size averaged just over 4,420 fish, and for the years 2004 to 2006, the three-year average abundance estimate was 4,167 sockeye (Data from appendix A in (Rawson, Sands et al. 2009)). It is estimated that between 35,500 and 121,000 spawners could be normally carried after full recovery (Hard, Jones et al. 1992).

The supplemental hatchery program began with out-of-basin stocks and make up an average of 10% of the run. The proportion of beach spawners originating from the hatchery is unknown but it is likely that straying is low. Hatchery originated fish is therefore not believed to have had a major effect on the genetics of the naturally spawned population. However, Ozette Lake sockeye has a relatively low allelic diversity at microsatellite DNA loci compared to other *O. nerka* populations examined in

Washington State (Crewson, Haggerty et al. 2001). Genetic differences occur between age cohorts. As different age groups do not spawn with each other, the population may be more vulnerable to significant reductions in population structure due to catastrophic events or unfavorable conditions affecting one year class. Based on this, the Puget Sound TRT's diversity viability criterion is one or more persistent spawning aggregation(s) with each major genetic and life history group being present within the aggregation (Rawson, Sands et al. 2009). Currently this is not the case; both spawning aggregations are at risk from losing year classes.

Critical Habitat

NMFS designated critical habitat for Ozette Lake sockeye salmon on September 2, 2005 (70 FR 52630). It encompasses areas within the Hoh/Quillayute subbasin, Ozette Lake, and the Ozette Lake watershed. The entire occupied habitat for this ESU is within the single watershed for Ozette Lake. This watershed was given a high conservation value rating. Spawning and rearing PCEs are found in the lake and in portions of three lake tributaries. Ozette River also provides rearing and migration PCEs. The river mouth provides estuarine habitat.

Spawning habitat has been affected by loss of tributary spawning areas and exposure of much of the available beach spawning habitat due to low water levels in summer. Further, native and non-native vegetation as well as sediment have reduced the quantity and suitability of beaches for spawning. The rearing PCE is degraded by excessive predation and competition with introduced non-native species, and by loss of tributary rearing habitat. Migration habitat may be adversely affected by high water temperatures and low water flows in summer which causes a thermal block to migration (La Riviere 1991).

Snake River Sockeye Salmon

The Snake River (SR) sockeye salmon ESU includes all anadromous and residual sockeye from the Snake River basin, Idaho, as well as artificially propagated sockeye

salmon from the Redfish Lake Captive Broodstock Program (June 28, 2005, 70 FR 37160). The Redfish Lake is located in the Salmon River basin, a subbasin within the larger Snake River basin (Figure 29).

Life History

SR sockeye salmon are unique compared to other sockeye salmon populations. Sockeye salmon returning to Redfish Lake in Idaho's Stanley Basin travel a greater distance from the sea (approximately 900 miles) to a higher elevation (6,500 ft) than any other sockeye salmon population and are the southern-most population of sockeye salmon in the world (Bjornn et al 1968). Stanley Basin sockeye salmon are separated by 700 or more river miles from two other extant upper Columbia River populations in the Wenatchee River and Okanogan River drainages. These latter populations return to lakes at substantially lower elevations (Wenatchee at 1,870 ft, Okanagon at 912 ft) and occupy different ecoregions.

A resident form of *O. nerka* (kokanee), also occur in the Redfish Lake. The residuals are non-anadromous; they complete their entire life cycle in fresh water. However, studies have shown that some ocean migrating juveniles are progeny of resident females (Rieman, Myers et al. 1994). The residents also spawn at the same time and in the same location as anadromous sockeye salmon.

Snake River Sockeye ESU Sub-Basin Range and Distribution

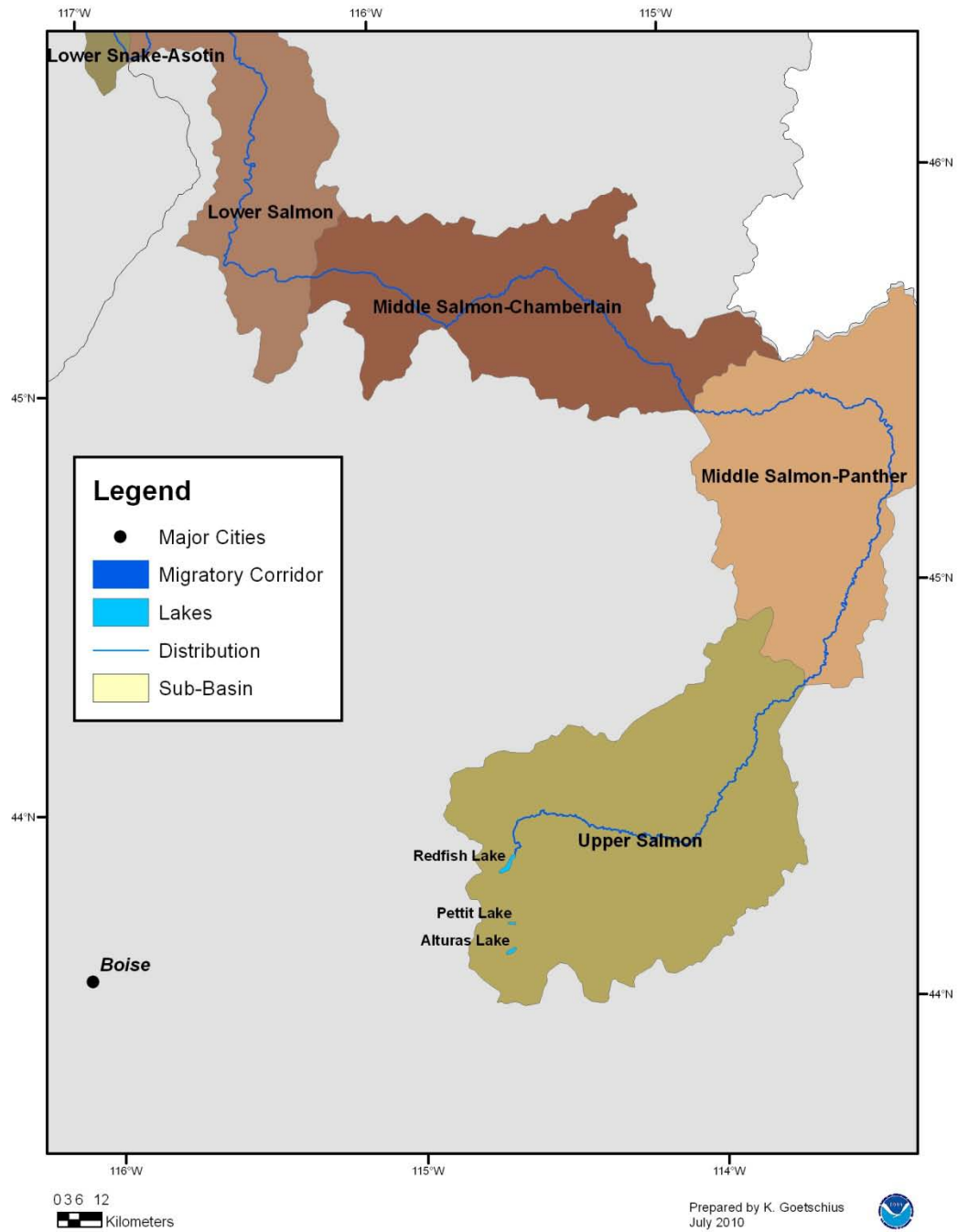


Figure 29. SR Sockeye Salmon distribution.

Historically, sockeye salmon entered the Columbia River system in June and July, and arrived at Redfish Lake between August and September (NMFS 2008). Spawning occurred in lakeshore gravel and generally peaked in October. Fry emerged in the spring (generally April and May) then migrated to open waters of the lake to feed. Juvenile sockeye remained in the lake for one to three years before migrating through the Snake and Columbia Rivers to the ocean. While pre-dam reports indicate that sockeye salmon smolts migrate in May and June, PIT tagged sockeye smolts from Redfish Lake pass Lower Granite Dam from mid-May to mid-July. Adult anadromous sockeye spent two or three years in the open ocean before returning to Redfish Lake to spawn.

Status and Trends

NMFS originally listed SR sockeye salmon as endangered in 1991, and reaffirmed their endangered status on June 28, 2005 (70 FR 37160). Subsequent to the 1991 listing, the residual form of sockeye residing in Redfish Lake was identified. In 1993, NMFS determined that residual sockeye salmon in Redfish Lake was part of the SR sockeye salmon ESU.

The only extant sockeye salmon population in the Snake River basin at the time of listing occurred in Redfish Lake, in the Stanley Basin (upper Salmon River drainage) of Idaho. Other lakes in the Salmon River basin that historically supported sockeye salmon include Alturas Lake above Redfish Lake which was extirpated in the early 1900s as a result of irrigation diversions, although residual sockeye may still exist in the lake (Chapman and Witty 1993). From 1955 to 1965, the Idaho Department of Fish and Game eradicated sockeye salmon from Pettit, Stanley, and Yellowbelly lakes, and built permanent structures on each of the lake outlets that prevented re-entry of anadromous sockeye salmon (Chapman and Witty 1993). Other historic sockeye salmon populations within the Snake River basin include Wallowa Lake (Grande Ronde River drainage, Oregon), Payette Lake (Payette River drainage, Idaho), and Warm Lake (South Fork Salmon River

drainage, Idaho) (Gustafson, Wainwright et al. 1997). These populations are now considered extinct.

Recent annual abundances of natural origin sockeye salmon in the Stanley Basin have been extremely low. No natural origin anadromous adults have returned since 1998 and the abundance of residual sockeye salmon in Redfish Lake is unknown. This species is currently entirely supported by adults produced through the captive propagation program.

Adult returns to Redfish Lake during the period 1954 through 1966 ranged from 11 to 4,361 fish (Bjornn, Craddock et al. 1968). In 1985, 1986, and 1987, 11, 29, and 16 sockeye, respectively, were counted at the Redfish Lake weir (Good, Waples et al. 2005). Only 18 natural origin sockeye salmon have returned to the Stanley Basin since 1987. The first adult returns from the captive brood stock program returned to the Stanley Basin in 1999. From 1999 through 2005, a total of 345 captive brood adults that had migrated to the ocean returned to the Stanley Basin. Recent years have seen an increase in returns to over 600 in 2008 and more than 700 returning adults in 2009. Current smolt-to-adult survival of sockeye originating from the Stanley Basin lakes is rarely greater than 0.3% (Hebdon, Kline et al. 2004).

Critical Habitat

NMFS designated critical habitat for SR sockeye salmon on December 28, 1993 (58 FR 68543). Designated habitat encompass the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed Snake River salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). SR sockeye critical habitat areas include the Columbia River from a straight line connecting the west end of the Clatsop jetty (Oregon side) and the west end of the Peacock jetty (Washington side), all river reaches from the estuary upstream to the confluence of the Snake River, and all Snake River reaches upstream to the confluence of the Salmon River; all Salmon River reaches to Alturas Lake Creek; Stanley, Redfish, Yellow Belly, Pettit, and Alturas Lakes (including their

inlet and outlet creeks); Alturas Lake Creek and that portion of Valley Creek between Stanley Lake Creek; and the Salmon River.

Conservation values of individual watersheds have not been reported (58 FR 68543). However, all areas occupied and used for migration by the SR sockeye salmon should be considered of high conservation value as the species' distribution is limited to a single lake within the Snake River basin.

The quality and quantity of rearing and juvenile migration PCEs have been reduced from activities such as tilling, water withdrawals, timber harvest, grazing, mining, and alteration of floodplains and riparian vegetation. These activities disrupt access to foraging areas, increase the amount of fines in the stream substrate that support production of aquatic insects, and reduce instream cover. Adult and juvenile migration PCE is affected by four dams in the Snake River basin that obstructs migration and increases mortality of downstream migrating juveniles.

Water quality impairments in the designated critical habitat of the SR sockeye salmon include inputs from fertilizers, insecticides, fungicides, herbicides, surfactants, heavy metals, acids, petroleum products, animal and human sewage, dust suppressants (*e.g.*, magnesium chloride), radionuclides, sediment in the form of turbidity, and other anthropogenic pollutants. Pollutants enter the surface waters and riverine sediments from the headwaters of the Salmon River to the Columbia River estuary as contaminated stormwater runoff, aerial drift and deposition, and via point source discharges. Some contaminants such as mercury and pentachlorophenol enter the aquatic food web after reaching water and may be concentrated or even biomagnified in the salmon tissue. Sockeye salmon require migration corridors with adequate passage conditions (water quality and quantity available at specific times) to allow access to the various habitats required to complete their life cycle. Multiple exposures to contaminants occur to all life stages throughout the entire range of the SR sockeye salmon.

Steelhead

Description of the Species

Steelhead are native to Pacific Coast streams extending from Alaska south to northwestern Mexico. We discuss the distribution, life history, status, and critical habitat of the 11 endangered and threatened steelhead species separately.

Steelhead have a protracted run time relative to Pacific salmon and do not tend to travel in large schools. Nevertheless, steelhead can be divided into two basic run-types: the stream-maturing type, or summer steelhead, and the ocean-maturing type, or winter steelhead. The summer steelhead enters fresh water in a sexually immature condition between May and October ((Busby, Wainwright et al. 1996), (Nickelson, Nicholas et al. 1992)). They then hold in cool, deep holding pools during summer and fall before moving to spawning sites as mature adults in January and February ((Barnhart 1986), (Nickelson, Nicholas et al. 1992)). Summer steelhead most commonly occur in streams where snowmelt contributes substantially to the annual hydrograph. The winter steelhead enters fresh water between November and April with well-developed gonads and spawns shortly after river entry ((Busby, Wainwright et al. 1996), (Nickelson, Nicholas et al. 1992)). Variations in migration timing exist between populations. Some adults enter coastal streams in the spring, just before spawning (Meehan and Bjornn 1991).

Steelhead typically spawn in small tributaries rather than large, mainstem rivers; spawning distribution often overlap with coho salmon. However, steelhead tend to prefer higher gradients (generally 2-7%, sometimes up to 12% or more) and their distribution tend to extend farther upstream than for coho salmon. Summer steelhead commonly spawn higher in a watershed than do winter steelhead, sometimes even using ephemeral streams from which juveniles are forced to emigrate as flows diminish.

Unlike Pacific salmon, steelhead are iteroparous, or capable of spawning more than once before death (Busby, Wainwright et al. 1996). Mostly females spawn more than once but

rarely more than twice before dying (Nickelson, Nicholas et al. 1992). Iteroparity is more common among southern steelhead populations than northern populations (Busby, Wainwright et al. 1996).

Juveniles rear in fresh water from one to four years, then smolt and migrate to the ocean in March and April (Barnhart 1986). After two to three weeks, in late spring, and following yolk sac absorption, alevins emerge from the gravel and begin actively feeding. The fry usually inhabit shallow water along banks and stream margins of streams (Nickelson, Nicholas et al. 1992). As they grow, steelhead juveniles commonly occupy faster flowing water such as riffles. Older and larger juveniles are more risk averse; they stay in deeper water and keep close to cover ((Bisson, Nielsen et al. 1982), (Bisson, Sullivan et al. 1988)). Some older juveniles move downstream to rear in larger tributaries and mainstem rivers (Nickelson, Nicholas et al. 1992).

Steelhead juveniles are highly territorial, dominance is based on initial size, and high densities result in increased migration. Juvenile steelhead that have established territories migrate little during their first summer (Bisson, Sullivan et al. 1988). Steelhead fry and parr hold close to the substratum where flows are lower and sometimes counter to the main stream. Here, steelhead foray up into surface currents for drifting food or prey at invertebrates on the stream bottom ((Bisson, Sullivan et al. 1988), (Kalleberg 1958)). Older steelhead commonly uses deeper pools ((Bisson, Nielsen et al. 1982), (Bisson, Sullivan et al. 1988)).

Juvenile steelhead are opportunistic and feed on a wide variety of aquatic and terrestrial insects (Chapman and Bjornn 1969). Prey species varies with season and availability; they utilize higher prey diversity than sympatric coho salmon (Pert 1987). Prey includes common aquatic stream insects such as caddisflies, mayflies, and stoneflies but also other insects (especially chironomid pupae), zooplankton, and benthic organisms ((Merz 2002), (Pert 1987)). Older juveniles sometimes prey on emerging fry, other fish larvae, crayfish, and even small mammals but these are not a major food source (Merz 2002).

All listed salmonids use shallow, low flow habitats at some point in their life cycle. However, steelhead juveniles use such habitat less than coho salmon and prefer faster flowing stream sections. During winter and spring, juveniles often seek protection under rocks and boulders to escape high flows. Contrary to coho salmon, steelhead seem to avoid overwintering in channels that have organic matter or “muck” as bottom substrate. They may move into inundated floodplains to forage during the high flow season.

In Oregon and California, steelhead may enter estuaries where sand bars close off the estuary, thereby creating low salinity lagoons. The migration of juvenile steelhead to lagoons occurs throughout the year, but is concentrated in the late spring/early summer and in the late fall/early winter period ((Shapovalov and Taft 1954), (Zedonis 1992)). In southern California, two discrete groups of juvenile steelhead use different habitat provided by lagoons: steelhead juveniles that use the upper and fresher areas of coastal lagoons for freshwater rearing throughout the year, and smolts that drop down from the watershed and use the lagoon primarily in the spring prior to seawater entry ((Cannata 1998), (Zedonis 1992)).

Immature steelhead migrate directly offshore during their first summer from whatever point they enter the ocean rather than along the coastal belt as salmon do. During the fall and winter, juveniles move southward and eastward ((Hartt and Dell 1986), (Nickelson, Nicholas et al. 1992)). Steelhead typically reside in marine waters for two or three years prior to returning to their natal stream to spawn as four or five-year olds.

Status and Trends

Steelhead survival depends on the quantity and quality of those aquatic systems they occupy. Steelhead have declined from overharvests, hatcheries, native and non-native exotic species, dams, gravel mining, water diversions, destruction or degradation of riparian habitat, and land use practices (logging, agriculture, and urbanization).

Puget Sound Steelhead DPS

This DPS includes all naturally spawned anadromous winter-run and summer-run steelhead in streams in the river basins of the Strait of Juan de Fuca, Puget Sound, and Hood Canal, Washington, bounded to the west by the Elwha River (inclusive) and to the north by the Nooksack River and Dakota Creek (inclusive), as well as the Green River natural and Hamma Hamma winter-run steelhead hatchery stocks (Figure 30). The remaining hatchery programs are not considered part of the DPS because they are more than moderately diverged from the local native populations.

Life History

The Puget Sound steelhead DPS contains both winter-run and summer-run steelhead. Adult winter-run steelhead generally return to Puget Sound tributaries from December to April (NMFS 2005). Spawning occurs from January to mid-June, with peak spawning occurring from mid-April through May. Prior to spawning, maturing adults hold in pools or in side channels to avoid high winter flows. Less information exists for summer-run steelhead as their smaller run size and higher altitude headwater holding areas have not been conducive for monitoring. Based on information from four streams, adult run time occur from mid-April to October with a higher concentration from July through September (NMFS 2005).

The majority of juveniles reside in the river system for two years with a minority migrating to the ocean as one or three-year olds. Smoltification and seaward migration occur from April to mid-May. The ocean growth period for Puget Sound steelhead ranges from one to three years in the ocean (Busby, Wainwright et al. 1996). Juveniles or adults may spend considerable time in the protected marine environment of the fjord-like Puget Sound during migration to the high seas.

Puget Sound Steelhead DPS Sub-Basin Range and Distribution



Figure 30. Puget Sound steelhead distribution.

Status and Trends

NMFS listed Puget Sound steelhead as threatened on May 11, 2007 (72 FR 26722). Fifty-three populations of steelhead have been identified in this DPS, of which 37 are winter-run. Summer-run populations are distributed throughout the DPS but are concentrated in northern Puget Sound and Hood Canal; only the Elwha River and Canyon Creek support summer-run steelhead in the rest of the DPS. The Elwha River run, however, is descended from introduced Skamania Hatchery summer-run steelhead. Historical summer-run steelhead in the Green River and Elwha River were likely extirpated in the early 1900s. Table 33 provides the geometric mean estimates of escapement of natural spawners for Puget Sound steelhead.

In the early 1980s, run size for this DPS was calculated at about 100,000 winter-run fish and 20,000 summer-run fish. By the 1990s, the total run size for four major stocks exceeded 45,000, roughly half of which were natural escapement. The Washington Department of Fish and Wildlife (WDFW) concluded that DPS escapement (excluding the Hamma Hamma population, see below) further declined by 23% during the years from 1999 through 2004 relative to the period from 1994 through 1998 (Washington Department of Fish and Wildlife (WDFW) 2008). Of the 53 known stocks of Puget Sound steelhead, the WDFW 2002 stock assessment categorized five stocks as healthy, 19 as depressed, one as critical, and 27 of unknown status. The WDFW (2008) data show escapement of natural spawners for the period 1980 to 2004 and the period 2000 to 2004 (Washington Department of Fish and Wildlife (WDFW) 2008).

In the 1996 and 2005 status reviews, the Skagit and Snohomish Rivers (North Puget Sound) winter-run steelhead were found to produce the largest escapements ((Busby, Wainwright et al. 1996), (NMFS 2005)). The two rivers still produce the largest wild escapement with a recent (2005 to 2008) four-year geometric mean of 5,468 for the Skagit River and an average 2,944 steelhead in Snohomish River for the two years 2005

and 2006 (Washington Department of Fish and Wildlife (WDFW) 2009). Lake Washington has

Table 33. Geometric mean estimates of escapement of natural spawners for Puget Sound steelhead.

Population	Run type	Long Term	5-Year
Canyon	SSH	N/A	N/A
Skagit	SSH	N/A	N/A
Snohomish	SSH	N/A	N/A
Stillaguamish	SSH	N/A	N/A
Canyon	WSH	N/A	N/A
Dakota	WSH	N/A	N/A
Nooksack	WSH	N/A	N/A
Samish	WSH	501	852
Skagit	WSH	6,994	5,419
Snohomish	WSH	5,283	3,230
Stillaguamish	WSH	1,028	550
Tolt	SSH	129	119
Green	SSH	N/A	N/A
Cedar	WSH	138	37
Green	WSH	1,802	1,620
Lk. Washington	WSH	308	37
Nisqually	WSH	1,116	392
Puyallup	WSH	1,714	907
Dewatto	WSH	24	25
Dosewallips	WSH	71	77
Duckabush	WSH	17	18
Hamma Hamma	WSH	30	52
Quilcene	WSH	17	18
Skokomish	WSH	439	203
Tahuya	WSH	114	117
Union	WSH	55	55
Elwha	SSH	N/A	N/A
Dungeness	WSH	311	174
Elwha	WSH	N/A	N/A
McDonald	WSH	150	96
Morse	WSH	106	103

For each population, estimates are provided for both long term (all yr, ca. 1980-2004 for most populations) and for a recent five year period (5 yr, 2000-2004). SSH, summer steelhead; WSH, winter steelhead. (NMFS 2005 status review updated for Puget Sound steelhead, <http://www.nwr.noaa.gov/ESA-Salmon-Listings/Salmon-Populations/Steelhead/STPUG.cfm>)

the lowest abundances of winter-run steelhead with an escapement of less than 50 fish in each year from 2000 through 2004 (Washington Department of Fish and Wildlife

(WDFW) 2008). The stock is now virtually extirpated with only eight and four returning fish in 2007 and 2008, respectively (Washington Department of Fish and Wildlife (WDFW) 2009). No abundance estimates exist for most of the summer-run populations; all appear to be small, most averaging less than 200 spawners annually.

Long-term trends (1980 to 2004) for the Puget Sound steelhead natural escapement have declined significantly for most populations, especially in southern Puget Sound, and in some populations in northern Puget Sound (Stillaguamish winter-run), Canal (Skokomish winter-run), and along the Strait of Juan de Fuca (Dungeness winter-run) (NMFS 2005). Positive trends were observed in the Samish winter-run (northern Puget Sound) and the Hamma Hamma winter-run (Hood Canal) populations. The increasing trend on the Hamma Hamma River may be due to a captive rearing program rather than to natural escapement (NMFS 2005).

The negative trends in escapement of naturally produced fish resulted from peaks in natural escapement in the early 1980s. Still, the period 1995 through 2004 (short-term) showed strong negative trends for several populations. This is especially evident in southern Puget Sound (Green, Lake Washington, Nisqually, and Puyallup winter-run), Hood Canal (Skokomish winter-run), and the Strait of Juan de Fuca (Dungeness winter-run) (NMFS 2005). As with the long-term trends, positive trends were evident in short-term natural escapement for the Samish and Hamma Hamma winter-run populations, and also in the Snohomish winter-run populations.

Median population growth rates (λ) using 4-year running sums is less than 1, indicating declining population growth, for nearly all populations in the DPS (NMFS 2005). However, some of the populations with declining recent population growth show only slight declines, (*e.g.*, Samish and Skagit winter-run in northern Puget Sound, and Quilcene and Tahuya winter-run in Hood Canal).

Only two hatchery stocks genetically represent native local populations (Hamma Hamma and Green River natural winter-run). The remaining programs, which account for the vast preponderance of production, are either out-of-DPS derived stocks or were within-DPS stocks that have diverged substantially from local populations. The WDFW estimated that 31 of the 53 stocks were of native origin and predominantly natural production (Washington Department of Fish and Wildlife (WDFW) 1993).

Intentional and inadvertent hatchery selection on life history in Chambers Creek winter-run steelhead has resulted in a domesticated strain with a highly modified average run and spawn timing. If interbreeding occurs, such changes can have a detrimental effect on fitness in the wild. However, genetic analyses by Phelps *et al.* (Phelps, Leider et al. 1997), indicated reproductive isolation of and/or poor spawning success by hatchery-origin fish. This was shown in a later study on the Clackamas River in Oregon (Kostow, Marshall et al. 2003). There is, however, some evidence for introgression by hatchery releases into winter-run steelhead populations in tributaries to the Strait of Juan de Fuca. However, this may have been due to the small size of the naturally-spawning populations relative to the hatchery introductions.

Critical Habitat

NMFS has not designated critical habitat for the Puget Sound steelhead.

Lower Columbia River Steelhead

The LCR steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in streams and tributaries to the Columbia River between the Cowlitz and Wind Rivers, Washington (inclusive), and the Willamette and Hood Rivers, Oregon (inclusive) (Figure 31). Two hatchery populations are included in this species, the Cowlitz Trout Hatchery winter-run population and the Clackamas River population but neither was listed as threatened.

Lower Columbia River Steelhead DPS Sub-Basin Range and Distribution

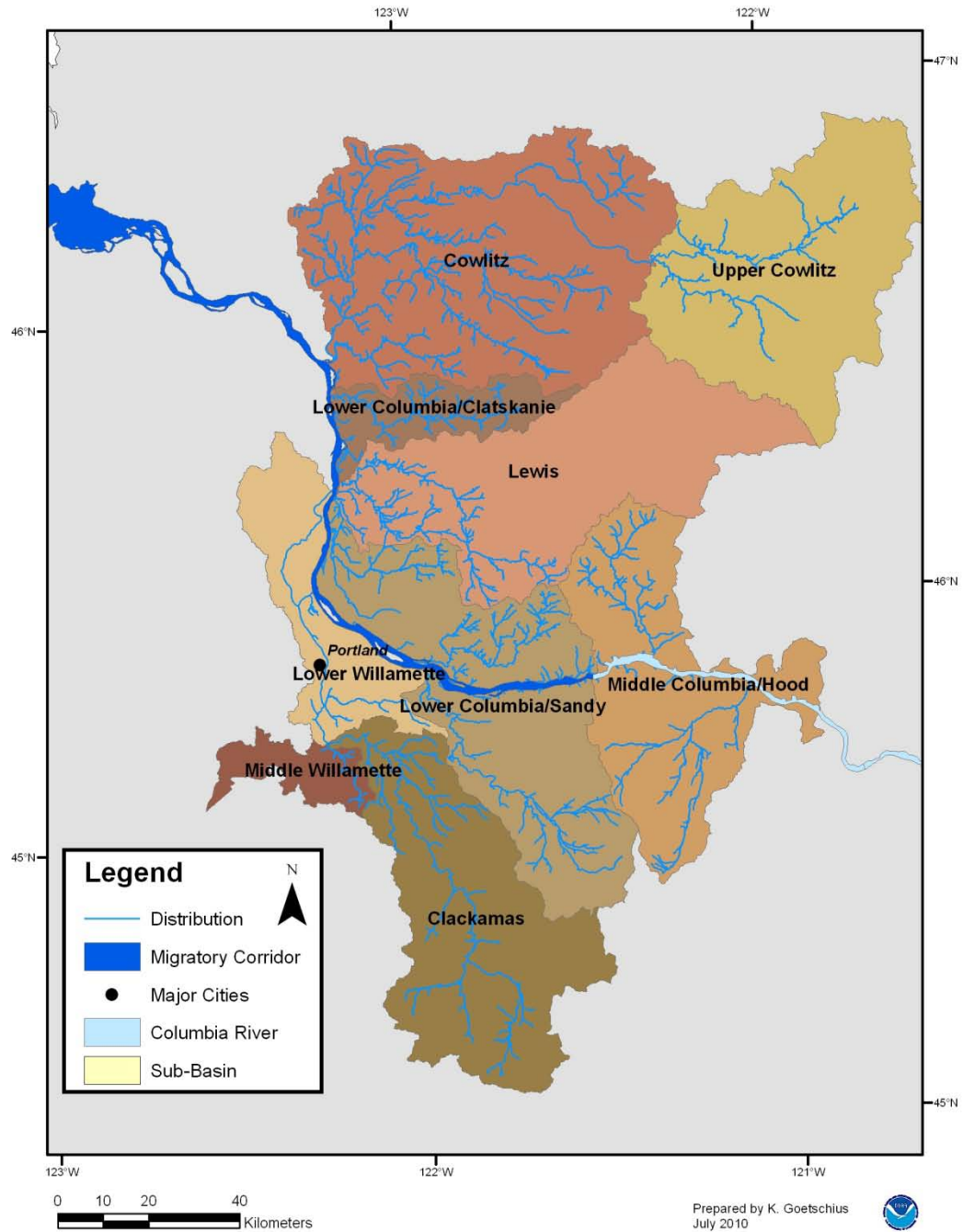


Figure 31. Lower Columbia River steelhead distribution.

Life History

The LCR steelhead DPS includes both summer- and winter-run stocks (Table 34). Summer-run steelhead return sexually immature to the Columbia River from May to November, and spend several months in fresh water prior to spawning. Winter-run steelhead enter fresh water from November to April, are close to sexual maturation during freshwater entry, and spawn shortly after arrival in their natal streams. Where both races spawn in the same stream, summer-run steelhead tend to spawn at higher elevations than the winter-run.

The majority of juvenile LCR steelhead remain for two years in freshwater environments before ocean entry in spring. Both winter- and summer-run adults normally return after two years in the marine environment.

Status and Trends

NMFS listed LCR steelhead as threatened on March 19, 1998 (63 FR 13347), and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The LCR steelhead had 17 historically independent winter steelhead populations and 6 independent summer steelhead populations ((McElhany, Backman et al. 2003); (Myers, Busack et al. 2006)). All historic LCR steelhead populations are considered extant. However, spatial structure within the historically independent populations, especially on the Washington side, has been substantially reduced by the loss of access to the upper portions of some basins due to tributary hydropower development.

All LCR steelhead populations declined from 1980 to 2000, with sharp declines beginning in 1995. Historical counts in some of the larger tributaries (Cowlitz, Kalama, and Sandy Rivers) suggest the population probably exceeded 20,000 fish. During the 1990s, fish abundance dropped to 1,000 to 2,000 fish. Recent abundance estimates of

natural-origin spawners range from completely extirpated for some populations above impassable barriers to over 700 fishes for the Kalama and Sandy winter-run populations. A number of the populations have a substantial fraction of hatchery-origin spawners in

Table 34. LCR Steelhead salmon populations, historic abundances (Good, Waples et al. 2005), 1998 – 2002 and 2004 to 2005 geometric mean abundance (Good, Waples et al. 2005)(Salmon Scape Query 2009), and hatchery contributions ((McElhany, Backman et al. 2003); (Good, Waples et al. 2005)).

Population	Run	Historical Abundance	Recent Geometric Mean Total Abundances	Hatchery Abundance Contributions
Cispus River	Winter	Unknown	Unknown	Unknown
Tilton River		Unknown	2,787/--	~73%
Upper Cowlitz River		Unknown	Unknown	Unknown
Lower Cowlitz River		1,672	Unknown	Unknown
Coweeman River		2,243	466/488	~50%
SF Toutle River		2,627	504/616	~2%
NF Toutle River		3,770	196/169	0%
Kalama River		3,165	726/1440	0%
NF Lewis River		713	Unknown	Unknown
EF Lewis River		3,131	Unknown/514	Unknown
Salmon Creek		Unknown	Unknown	Unknown
Washougal River		2,497	323/528	0%
Clackamas River		Unknown	560/--	41%
Sandy River		Unknown	977/--	42%
Lower tributaries		793	Unknown	Unknown
Upper tributaries		243	Unknown	Unknown
Hood River		Unknown	756/--	~52%
Kalama River	Summer	Unknown	--/384	
NF Lewis River		Unknown	Unknown	Unknown
EF Lewis River		Unknown	--/474	
Washougal River		Unknown	--/668	
Hood River		Unknown	931/--	~83%
Wind River		2,288	--/627	~5%

spawning areas. Many of the long-and short-term trends in abundance of individual populations are negative.

There is a difference in population stability between winter- and summer-run LCR steelhead. The winter-run steelhead in the Cascade region has the highest likelihood of being sustained as it includes a few populations with moderate abundance and positive short-term population growth rates ((McElhany, Chilcote et al. 2007); (Good, Waples et al. 2005)). The Gorge summer-run steelhead is at the highest risk over the long-term as the Hood River population is at high risk of being lost (McElhany, Chilcote et al. 2007).

Critical habitat

Critical habitat was designated for the LCR steelhead on September 2, 2005 (70 FR 52488). Of 41 subbasins listed as critical habitat for the LCR steelhead, 28 subbasins were rated as having a high conservation value. Eleven subbasins were rated as having a medium value and two were rated as having a low value to the conservation of the DPS (Table 35).

Table 35. LCR steelhead watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Middle-Columbia/Hood	4	(1, 3, <2)	1	(3, 1)	1	(3, 1)
Lower Columbia/Sandy	4	(1, 3)	5	(3, 1)	0	
Lewis	2	(3, 1, 2)	0		0	
Lower Columbia/Clatskanie	1	(3, 1)	0		0	
Upper Cowlitz River	5	(3)	0		0	
Cowlitz	3	(3, 1)	5	(3, 1, 2)	0	
Middle Willamette	0		0		1	(1, 2)
Clackamas	6	(1, <2)	0		0	
Lower Willamette	3	(2, 1, 3)	0		0	
Lower Columbia Corridor	all	(3, 2)	0		0	
Total	28		11		2	

¹ Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE

Critical habitat is affected by reduced quality of rearing and juvenile PCBs within the lower portion and alluvial valleys of many watersheds; contaminants from agriculture affect both water quality and food production in these reaches of tributaries and in the mainstem Columbia River. Several dams affect adult migration PCE by obstructing the migration corridor. Watersheds which consist of a large proportion of federal lands such as is the case with the Sandy River watershed, have relatively healthy riparian corridors that support attributes of the rearing PCE such as cover, forage, and suitable water quality (Figure 32).

Lower Columbia River Steelhead DPS
Conservation Value of HUC 5 Watersheds

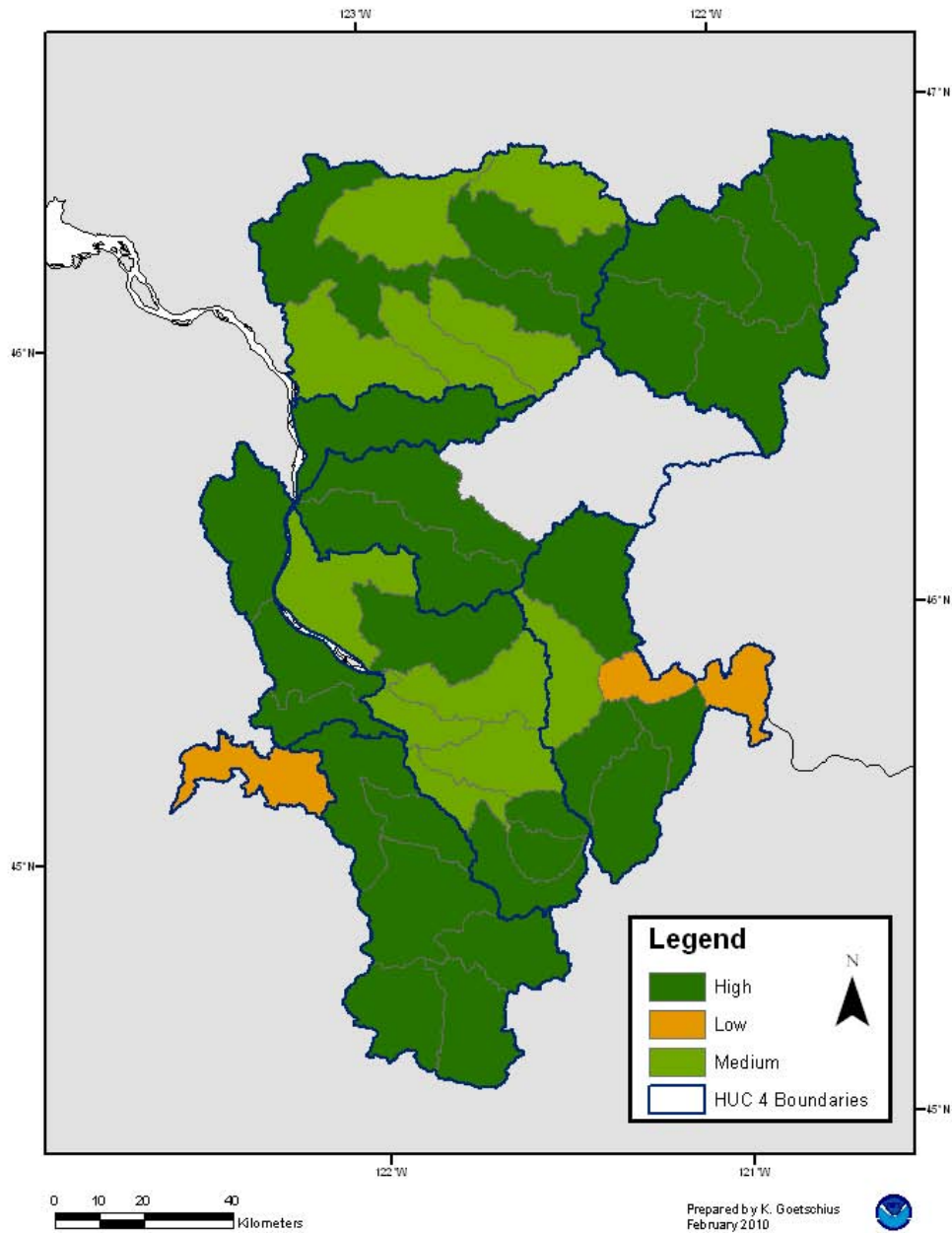


Figure 32. Lower Columbia River Steelhead Conservation Values per Sub-area.

Upper Willamette River Steelhead

The UWR steelhead DPS includes all naturally spawned winter-run steelhead populations below natural and manmade impassable barriers in the Willamette River, Oregon, and its tributaries upstream from Willamette Falls to the Calapooia River (inclusive) (Figure 33). No artificially propagated populations that reside within the historical geographic range of this DPS are included in this listing. Hatchery summer-run steelhead occur in the Willamette Basin but are an out-of-basin population that is not included in this DPS.

Life History

Native steelhead in the Upper Willamette are a late-migrating winter group that enters fresh water in January and February (Howell, Jones et al. 1985). UWR steelhead do not ascend to their spawning areas until late March or April, which is late compared to other West Coast winter steelhead. Spawning occurs from April to June 1. The unusual run timing may be an adaptation for ascending the Willamette Falls, which may have facilitated reproductive isolation of the stock. The smolt migration past Willamette Falls also begins in early April and proceeds into early June, peaking in early- to mid-May (Howell, Jones et al. 1985). Smolts generally migrate through the Columbia via Multnomah Channel rather than the mouth of the Willamette River. As with other coastal steelhead, the majority of juveniles smolt and outmigrate after two years; adults return to their natal rivers to spawn after spending two years in the ocean. Repeat spawners are predominantly female and generally account for less than 10% of the total run size (Busby, Wainwright et al. 1996).

Upper Willamette River Steelhead DPS Sub-Basin Range and Distribution

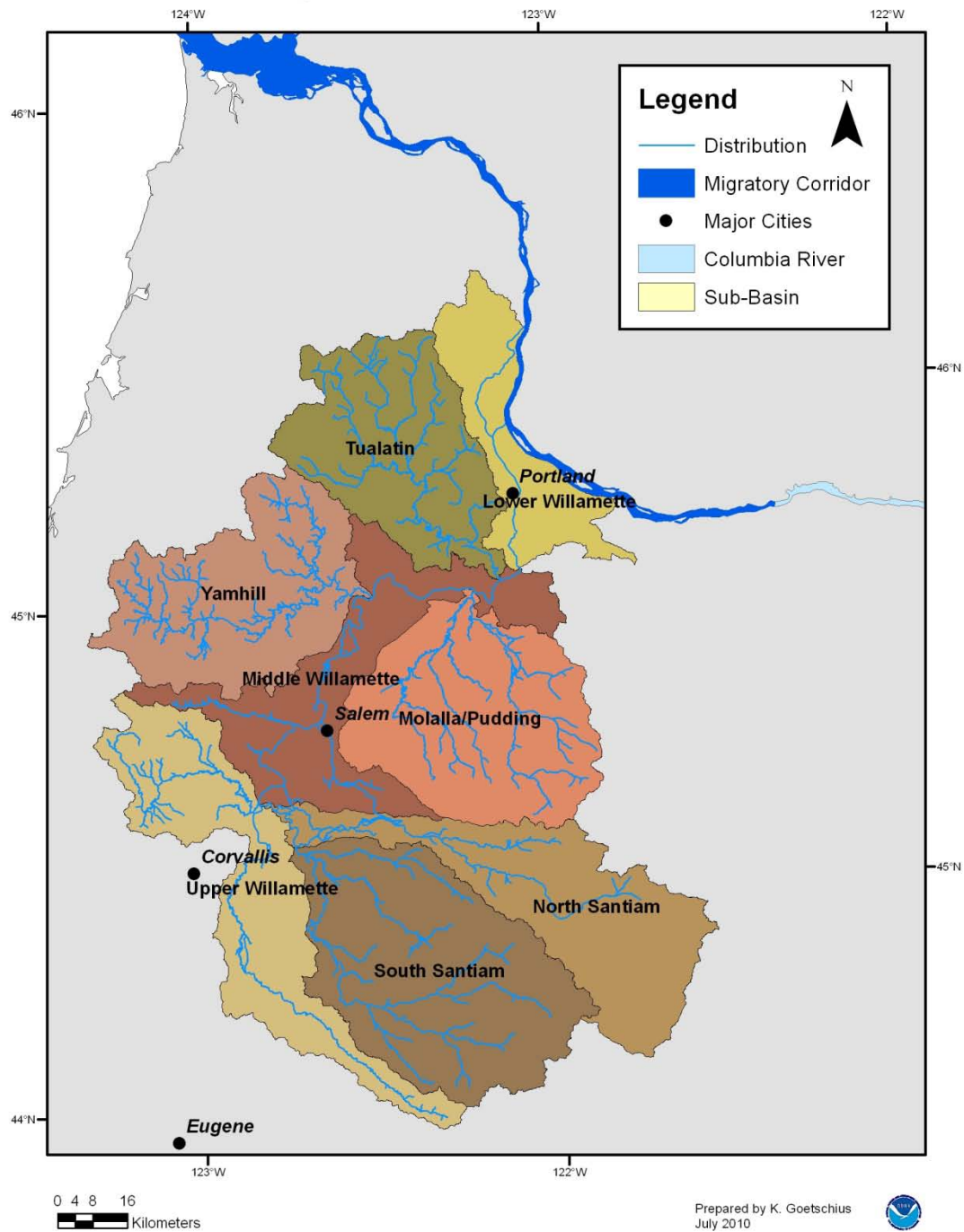


Figure 33. UWR Steelhead distribution.

Status and Trends

NMFS originally listed UWR steelhead as threatened on March 25, 1999 (64 FR 14517), and reaffirmed their threatened status on January 5, 2006 (71 FR 834). Four basins on the east side of the Willamette River historically supported independent populations for the UWR steelhead. All of which remain extant. Data reported in McElhane et al. (2007) indicate that currently the two largest populations within the DPS are the Santiam River populations. Meaner spawner abundance in both the North and South Santiam River is about 2,100 native winter-run steelhead. However, about 30% of all habitat has been lost due to human activities (McElhany, Chilcote et al. 2007). The North Santiam population has been substantially affected by the loss of access to the upper North Santiam basin. The South Santiam subbasin has lost habitat behind non-passable dams in the Quartzville Creek watershed. Notwithstanding the lost spawning habitat, the DPS continues to be spatially well distributed, occupying each of the four major subbasins.

Table 36. Upper Willamette River steelhead salmon populations, core (C) and genetic legacy (G) populations, abundances, and hatchery contributions ((McElhany, Backman et al. 2003), (Good, Waples et al. 2005)).

Historic Independent Populations	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Mollala Rivers	Unknown	0.972 rpm	Unknown
North Santiam River	Unknown	0.963 rpm	Unknown
South Santiam River	Unknown	0.917 rpm	Unknown
Calapooia River	Unknown	1.053 rpm	Unknown
Total	Unknown	5,819	

Note: rpm denotes redds per mile.

UWR steelhead are moderately depressed from historical levels (McElhany, Chilcote et al. 2007). Average number of late-fall steelhead passing Willamette Falls decreased during the 1990s to less than 5,000 fish. The number again increased to over 10,000 fish in 2001 and 2002. The geometric and arithmetic mean number of late-run steelhead passing Willamette Falls for the period 1998 to 2001 were 5,819 and 6,795, respectively.

Population information for individual basins exist as redds per (river) mile. These redd counts show a declining long-term trend for all populations (Good, Waples et al. 2005). One population, the Calapooia, had a positive short-term trend during the years from 1990 to 2001. McElhany *et al.* (2007) however, found that the populations had a low risk of extinction. Two of the populations were considered at moderate risk from failed abundances and recruitment levels and two (North and South Santiam Rivers) were considered at low risk given current abundances and recruitment (McElhany, Chilcote et al. 2007).

Hatchery raised winter-run steelhead were released in the Upper Willamette River up to 1999. These fish were out of basin stocks and had an earlier return timing than the native steelhead. The impact of these releases on the genetic diversity and life history of the native population is unknown (Table 36). Nevertheless, remains of the early run still exist and the release of hatchery fish has been discontinued.

Critical Habitat

NMFS designated critical habitat for this species on September 2, 2005 (70 FR 52488). It includes all Columbia River estuarine areas and river reaches proceeding upstream to the confluence with the Willamette River and specific stream reaches in the following subbasins: Upper Willamette, North Santiam, South Santiam, Middle Willamette, Molalla/Pudding, Yamhill, Tualatin, and Lower Willamette (NMFS 2005).

Table 37. UWR steelhead watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Upper Willamette	1	(1, 2)	2	(2, 1)	0	
North Santiam	3	(1, 2)	0		0	
South Santiam	6	(1, 2)	0		0	
Middle Willamette	0		0		4	(2, 1)
Yamhill	0		1	(1, 2)	6	(2, 1)
Molalla/Pudding	1	(1)	2	(2, 1)	3	(2, 1)
Tualatin	0		1	(1, 2)	4	(1, 2, 3)
Lower Willamette	3	(2)	0		0	
Columbia River	all	(3)	0		0	

Corridor						
Total	14		6		17	

1 Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

Of the subbasins reviewed in NMFS' assessment of critical habitat for the UWR steelhead, 14 subbasins were rated as having a high conservation value, six were rated as having a medium value, and 17 were rated as having a low conservation value (Table 37).

The current condition of critical habitat designated for the UWR steelhead is degraded (Figure 34), and provides a reduced the conservation value necessary for species recovery. Critical habitat is affected by reduced quality of rearing and juvenile PCEs within many watersheds; contaminants from agriculture affect both water quality and food production in several watersheds and in the mainstem Columbia River. Several dams affect adult migration PCE by obstructing the migration corridor.

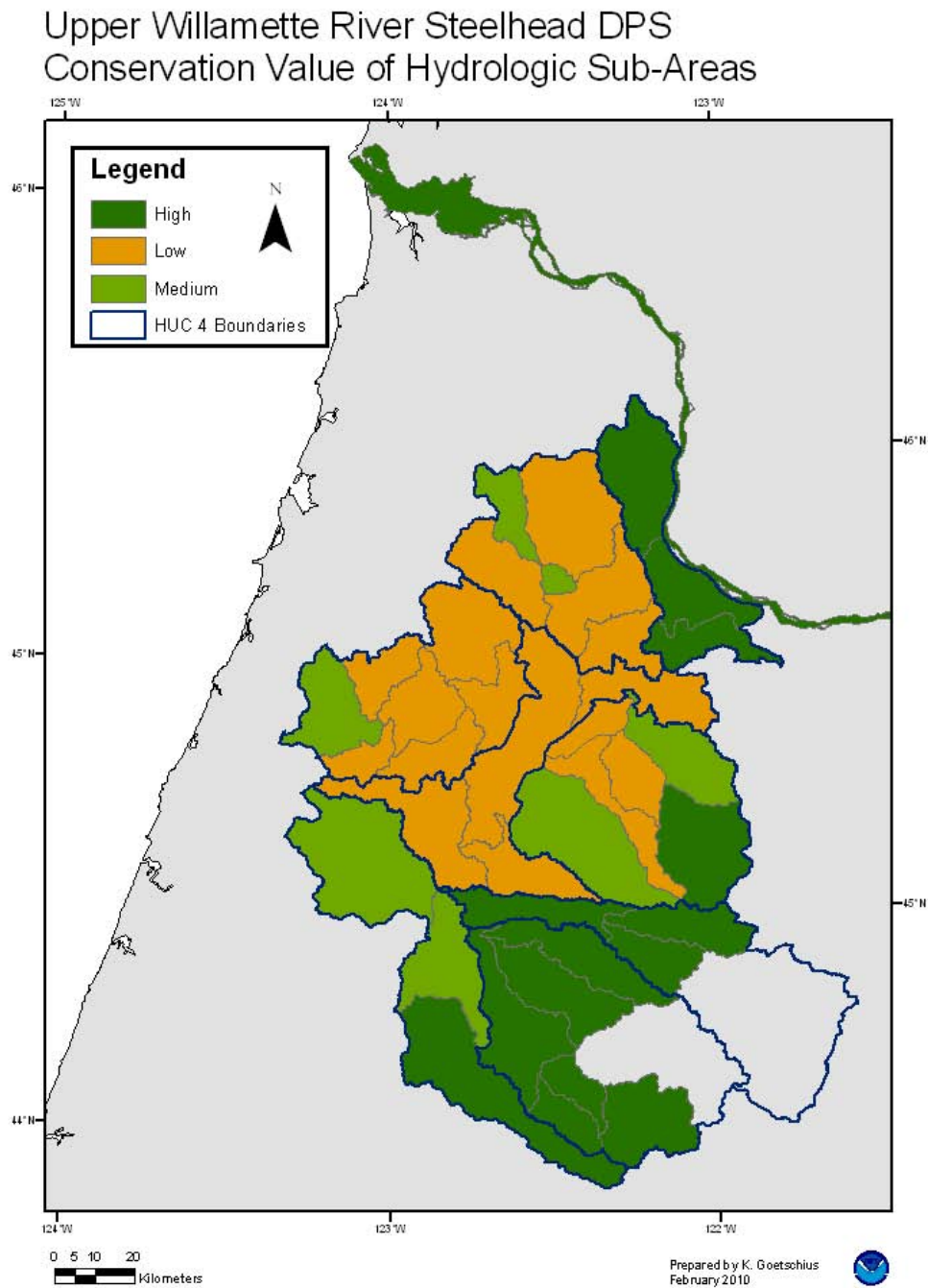


Figure 34. Upper Willamette River Steelhead Conservation Values per Sub-Area.

Middle Columbia River Steelhead

The MCR steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in streams from above the Wind River, Washington, and the Hood River, Oregon (exclusive), upstream to, and including, the Yakima River, Washington, excluding *O. mykiss* from the Snake River Basin. Steelhead from the Snake River basin (described later in this section) are excluded from this DPS. Seven artificial propagation programs are part of this DPS. They include: the Touchet River Endemic, Yakima River Kelt Reconditioning Program (in Satus Creek, Toppenish Creek, Naches River, and Upper Yakima River), Umatilla River, and the Deschutes River steelhead hatchery programs (Figure 35). These artificially propagated populations are considered no more divergent relative to the local natural populations than would be expected between closely related natural populations within the DPS.

According to the ICBTRT (ICTRT 2003), this DPS is composed of 16 populations in four major population groups (Cascade Eastern Slopes Tributaries, John Day River, Walla Walla and Umatilla Rivers, and Yakima River), and one unaffiliated population (Rock Creek).

Middle Columbia River Steelhead DPS Sub-Basin Range and Distribution

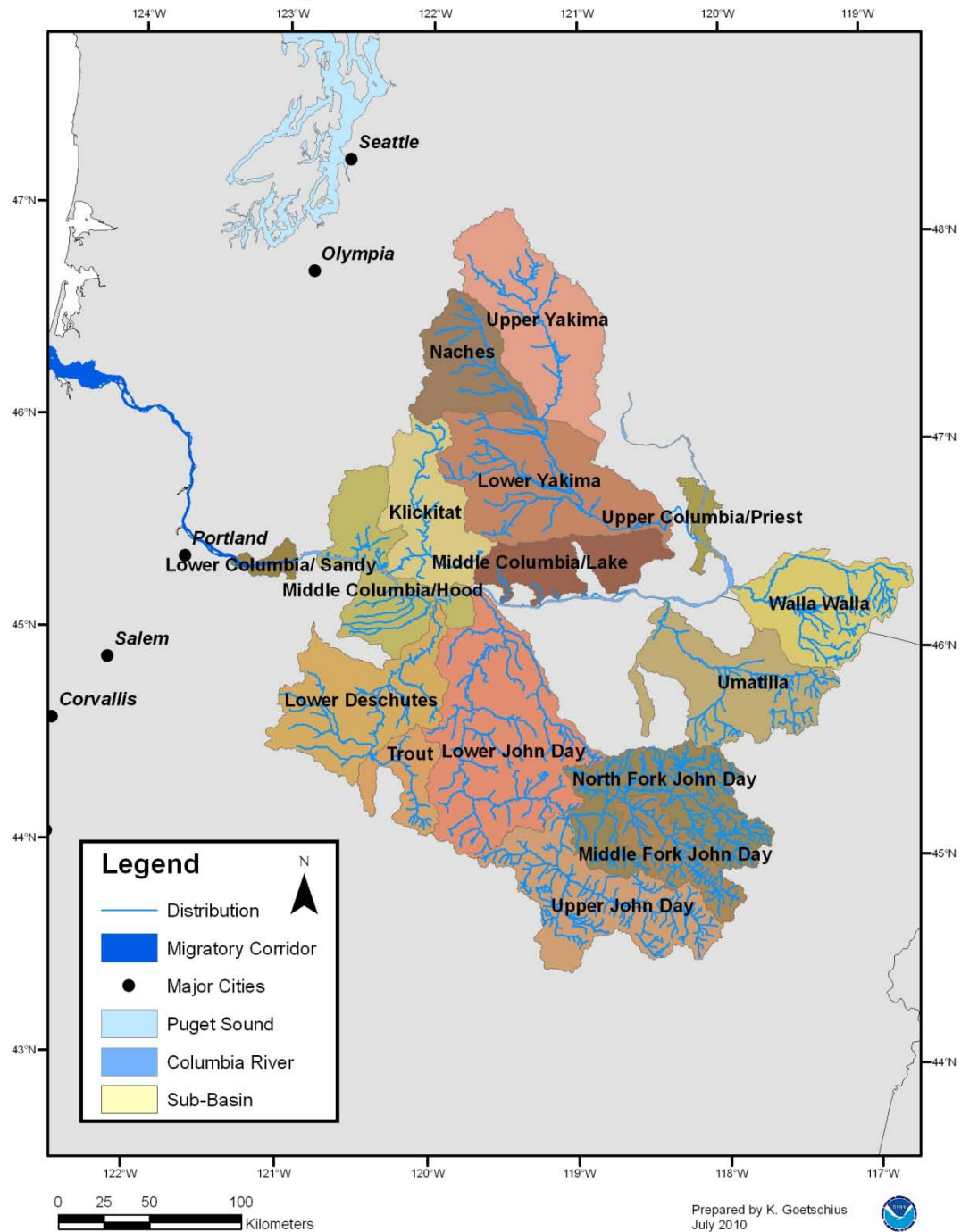


Figure 35. MCR Steelhead distribution.

Life History

MCR steelhead populations are mostly of the summer-run type. Adult steelhead enter fresh water from June through August. The only exceptions are populations of inland winter-run steelhead which occur in the Klickitat River and Fifteenmile Creek (Busby, Wainwright et al. 1996).

The majority of juveniles smolt and outmigrate as two-year olds. Most of the rivers in this region produce about equal or higher numbers of adults having spent one year in the ocean as adults having spent two years. However, summer-run steelhead in Klickitat River have a life cycle more like LCR steelhead whereby the majority of returning adults have spent two years in the ocean (Busby, Wainwright et al. 1996). Adults may hold in the river up to a year before spawning.

Status and Trends

NMFS listed MCR steelhead as threatened on March 25, 1999 (64 FR 14517), and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The ICTRT identified 16 extant populations in four major population groups (Cascades Eastern Slopes Tributaries, John Day River, Walla Walla and Umatilla Rivers, and Yakima River) and one unaffiliated independent population (Rock Creek) (ICTRT 2003). There are two extinct populations in the Cascades Eastern Slope major population group: the White Salmon River and the Deschutes Crooked River above the Pelton/Round Butte Dam complex. Present population structure is delineated largely on geographical proximity, topography, distance, ecological similarities or differences.

Table 38. Middle Columbia River steelhead independent populations, abundances, and hatchery contributions (ICTRT 2003); (Good, Waples et al. 2005)).

Major Basins	Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Cascade Eastern Slope Tributaries	Klickitat River	Unknown	97-261 reds	Unknown
	<i>White Salmon River</i>	<i>Unknown</i>	<i>Extirpated</i>	<i>N/A</i>
	Fifteenmile Creek	Unknown	2.87 rpm	100%
	East and West Deschutes River*	Unknown	10,026-21,457	38%
	<i>Crooked River</i>	<i>Unknown</i>	<i>Extirpated</i>	<i>N/A</i>
John Day	John Day upper main	Unknown	926-4,168	96%
	John Day lower main	Unknown	1.4 rpm	0%
	John Day NF			
	upper NF	Unknown	2.57 rpm	0%
	lower NF	Unknown	.52 rpm	0%
	John Day MF	Unknown	3.7 rpm	0%
	John Day SF	Unknown	2.52 rpm	0%
Walla Walla and Umatilla	Umatilla River	Unknown	1,480-5,157	60%
	Walla Walla River	Unknown	Unknown	Unknown
	Touchet River	Unknown	273-527	Unknown
	<i>Willow Creek</i>	<i>Unknown</i>	<i>Extirpated</i>	<i>N/A</i>
Yakima	Yakima River Basin	Unknown	1,058-4,061	97%
	Satus Creek	Unknown	Unknown	Unknown
	Toppenish Creek	Unknown	Unknown	Unknown
	Naches River	Unknown	Unknown	Unknown
	Upper Yakima	Unknown	Unknown	Unknown

*Deschutes River is divided into two historically independent populations: the Eastside and Westside Tributaries

Historic run estimates for the Yakima River imply that annual species abundance may have exceeded 300,000 returning adults (Busby, Wainwright et al. 1996). The five-year average (geometric mean) return of natural MCR steelhead for 1997 to 2001 was up from previous years' basin estimates. Returns to the Yakima River, the Deschutes River, and sections of the John Day River system were substantially higher compared to 1992 to 1997 (Good, Waples et al. 2005). The five-year average for these basins is 298 and 1,492 fish, respectively (Good, Waples et al. 2005).

Good *et al.* (2005) calculated that the median estimate of long-term trend over 12 indicator data sets was -2.1% per year (-6.9 to 2.9), with 11 of the 12 being negative. Long-term annual population growth rates (λ) were also negative (Good, Waples et al. 2005). The median long-term λ was 0.98, assuming that hatchery spawners do not

contribute to production, and 0.97 assuming that both hatchery- and natural-origin spawners contribute equally.

The median short-term (1990–2001) annual population growth rate assuming no hatchery contribution is estimated to 1.045 (Good, Waples et al. 2005). Of the 12 datasets, 8 indicator trends have a positive growth rate. Assuming that potential hatchery spawners contributed at the same rate as natural-origin spawners resulted in lower estimates of population growth rates. The median short-term λ under the assumption of equal hatchery- and natural-origin spawner effectiveness was 0.967, with 6 of the 12 indicator trends exhibiting positive growth rates.

The Yakima River populations are at a risk from overall depressed abundances and the majority of spawning occurring in only one tributary (Good, Waples et al. 2005). The Cascade populations are at risk by the only population with large runs being dominated by out-of-basin strays (Good, Waples et al. 2005). Returns to sections of the John Day River system increased in the late 1990s and these populations are the only ones with returns consisting mainly of natural spawners (Good, Waples et al. 2005). However, degraded habitat conditions in the John Day River basin (NMFS 1999) may affect the populations' ability to maintain a positive recruitment during less productive ocean conditions (Good, Waples et al. 2005).

Table 38 summarizes MCR steelhead independent populations, abundances and hatchery contributions (ICTRT 2003; Good, Waples et al. 2005). Status reviews in the 1990s noted considerable reduction in abundances in several basins, loss and degraded freshwater habitat, and stray steelhead in Deschutes River. The population experienced a substantial increase in abundance in some basins since these reviews (Good, Waples et al. 2005).

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630).

The CHART assessment for this DPS addressed 15 (HUC4) subbasins containing 106 occupied watersheds (HUC5), as well as the Columbia River rearing/migration corridor (NMFS 2005). Of all the watersheds, 73 were rated as having a high conservation value, 24 as medium value, and 9 as low value (Table 39). The lower Columbia River rearing/migration corridor downstream of the spawning range is also considered to have a high conservation value.

Table 39. MCR steelhead watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Upper Yakima	3	(1, 3, 2)	1	(2, 1)	0	
Naches	3	(1, 3)	0		0	
Lower Yakima	3	(1, 3)	3	(3 ¹ , 2)	0	
Middle Columbia/Lake Wallula	2	(3, <1)	3	(3)	0	
Walla Walla	5	(1, 3, 2)	3	(3, 1, 2)	1	(3)
Umatilla	6	(1, 2)	1	(1, 2)	3	(1, 2)
Middle Columbia/Hood	3	(1, 3)	4	(3, <2)	1	(1)
Klickitat	4	(3, 1)	0		0	
Upper John Day	12	(1, 2, 3)	1	(1, 2)	0	
North Fork John Day	9	(1, 2, 3)	1	(1, 2)	0	
Middle Fork John Day	4	(1, 3)	0		1	(2, 1)
Lower John Day	7	(1, 3)	6	(1, 3, 2)	1	(3, <2)
Lower Deschutes	8 ³	(1, 2)	0		1	(1, =1.9mi)
Trout	2	(1)	1	(1)	1	(1,=1.5mi)
Lower Columbia/Sandy	1	(3)	0		0	
Upper Columbia/Priest Rapids	1	(3)	0		0	
Lower Columbia Corridor	all	(3) ²				
Total	73		24		9	

¹ Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

The current condition of critical habitat designated for the MCR steelhead is moderately degraded (Figure 36). Critical habitat is affected by reduced quality of rearing and juvenile PCEs within many watersheds; contaminants from agriculture affect both water quality and food production in several watersheds and in the mainstem Columbia River. Loss of riparian vegetation to grazing has resulted in high water temperatures in the John Day basin. Reduced quality of the rearing PCEs has diminished its contribution to the conservation value necessary for the recovery of the species. Several dams affect adult migration PCE by obstructing the migration corridor.

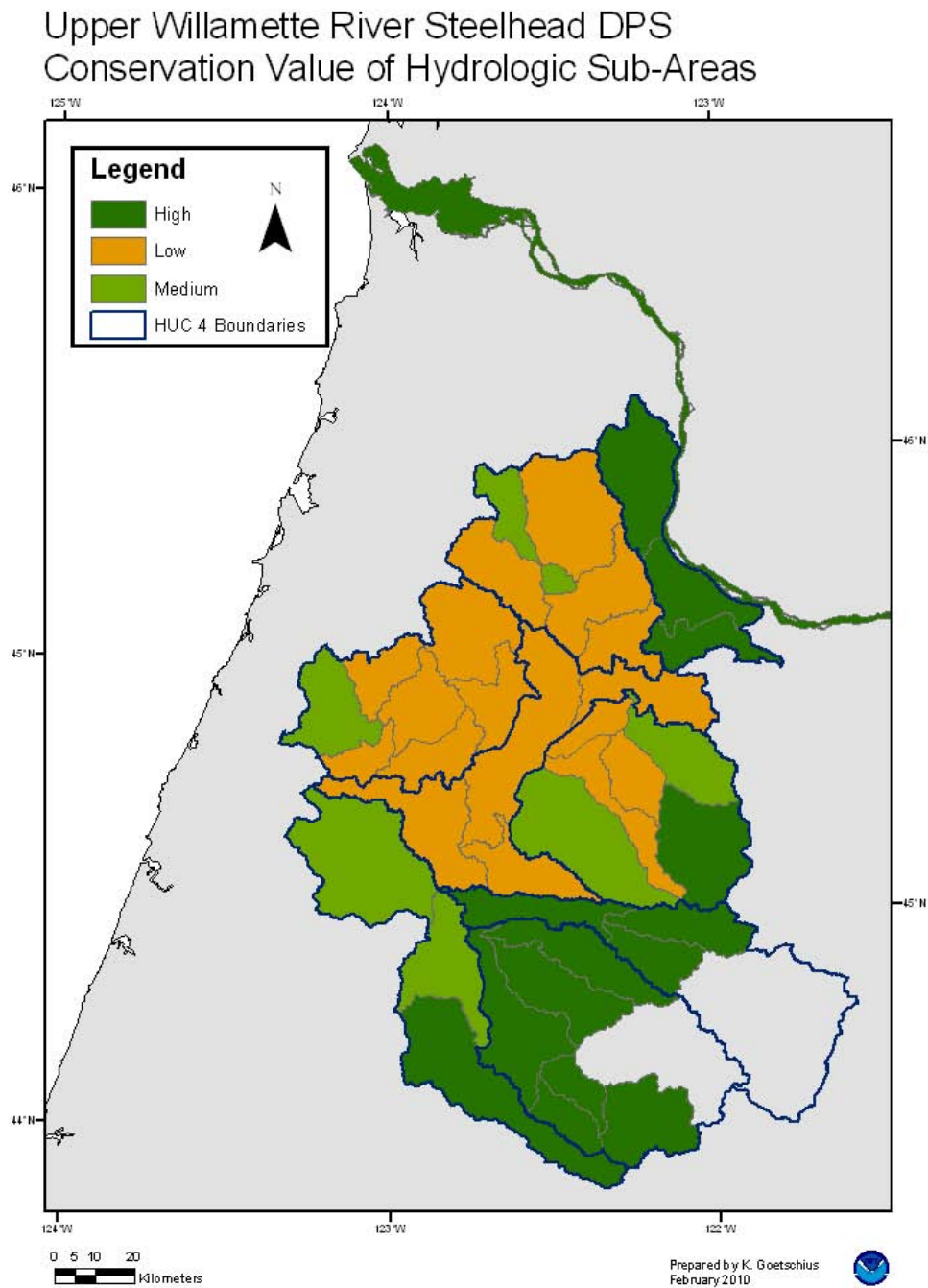


Figure 36. Upper Willamette River Steelhead Conservation Values per Sub-area.

Upper Columbia River Steelhead

The UCR steelhead DPS includes all naturally spawned steelhead populations below natural and man-made impassable barriers in streams in the Columbia River basin upstream from the Yakima River, Washington, to the U.S. - Canada border (Figure 37).

The UCR steelhead DPS also includes six artificial propagation programs: the Wenatchee River, Wells Hatchery (in the Methow and Okanogan Rivers), Winthrop NFH, Omak Creek, and the Ringold steelhead hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this DPS.

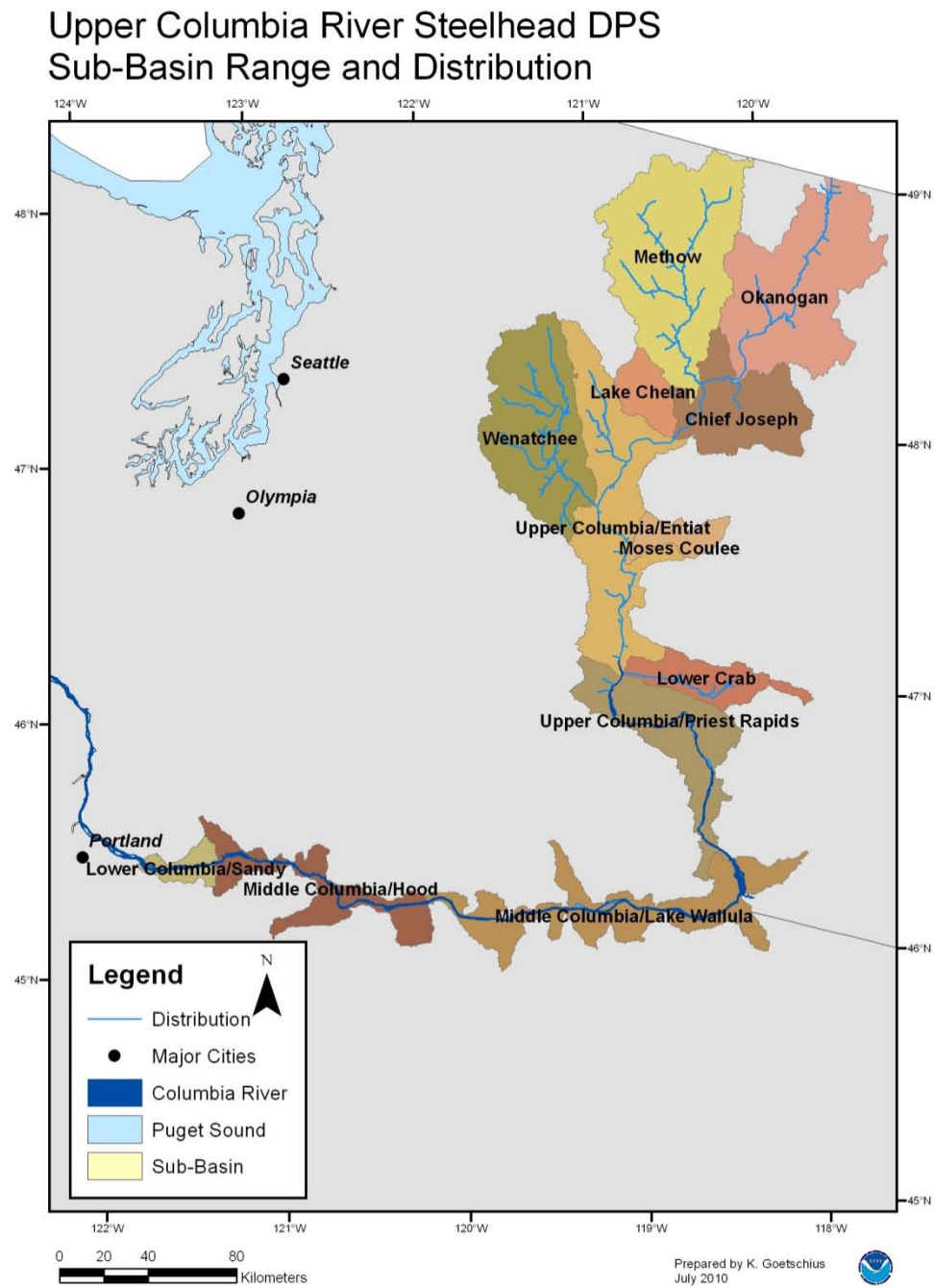


Figure 37. UCR Steelhead distribution.

Life History

All UCR steelhead are summer-run steelhead. Adults return in the late summer and early fall, with most migrating relatively quickly to their natal tributaries. A portion of the returning adult steelhead overwinters in mainstem reservoirs, passing over upper-mid-Columbia dams in April and May of the following year. Spawning occurs in the late spring of the year following river entry. Juvenile steelhead spend one to seven years rearing in fresh water before migrating to sea. Smolt outmigrations are predominantly year class two and three (juveniles), although some of the oldest smolts are reported from this DPS at seven years. Most adult steelhead return to fresh water after sea after one or two years.

Status and Trends

NMFS originally listed UCR steelhead as endangered on August 19, 1997 (62 FR 43937). On June 18, 2009, the species' status was changed to threatened (74 FR 42605). The UCR steelhead consisted of four historical independent populations: the Wenatchee, Entiat, Methow, and Okanogan. All populations are extant. The UCR steelhead must navigate over several dams to access spawning areas. The construction of Grand Coulee Dam in 1939 blocked access to over 50% of the river miles formerly available to UCR steelhead (ICTRT 2003).

Returns of both hatchery and naturally produced steelhead to the upper Columbia River have increased in recent years. The average 1997 to 2001 return counted through the Priest Rapids fish ladder was approximately 12,900 fish. The average for the previous five years (1992 to 1996) was 7,800 fish. Abundance estimates of returning naturally produced UCR steelhead were based on extrapolations from mainstem dam counts and associated sampling information (Good, Waples et al. 2005). The natural component of the annual steelhead run over Priest Rapids Dam increased from an average of 1,040 (1992-1996), representing about 10% of the total adult count, to 2,200 (1997-2001), representing about 17% of the adult count during this period of time (ICTRT 2003).

Table 40. Upper Columbia River Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Wenatchee/Entiat rivers	Unknown	1,899-8,036	71%
Methow/Okanogan rivers	Unknown	1,879-12,801	91%
Total	Unknown	3,778-20,837	

Recent population abundances for the Wenatchee and Entiat aggregate population and the Methow population remain well below the minimum abundance thresholds developed for these populations (ICTRT 2003). A five-year geometric mean (1997 to 2001) of approximately 900 naturally produced steelhead returned to the Wenatchee and Entiat rivers (combined). The abundance is well below the minimum abundance thresholds but it represents an improvement over the past (an increasing trend of 3.4% per year).

Regarding the population growth rate of natural production, on average, over the last 20 full brood year returns (1980/81 through 1999/2000 brood years), including adult returns through 2004-2005, UCR steelhead populations have not replaced themselves. Overall adult returns are dominated by hatchery fish (Table 40), and detailed information is lacking on the productivity of the natural population.

All UCR steelhead populations have reduced genetic diversity from homogenization of populations that occurred during the Grand Coulee Fish Maintenance project from 1939-1943, from 1960, and 1981 (Chapman, Hillman et al. 1994).

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). The CHART assessment for this ESU addressed 10 (HUC4) subbasins containing 41 occupied watersheds (HUC5), as well as the Columbia River rearing/migration corridor. Thirty-one of the watersheds were rated as having a high conservation value, seven as medium value, and three as low value (Table 41). The lower Columbia River

rearing/migration corridor downstream of the spawning range is of high conservation value.

Table 41. UCR Steelhead watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Chief Joseph	1	(3, 2)	0		2	(2)
Okanogan	2	(3, 1)	3	(3)	0	
Similkameen	1	(3)	0		0	
Methow	7	(1, 3)	0		0	
Lake Chelan	0		1	(1, 3)	0	
Upper Columbia/Entiat	3	(3, 1)	1	(3)	0	
Wenatchee	4	(1, 2, 3)	1	(3, 1)	0	
Moses Coulee	0		0		1	(2)
Lower Crab	0		1	(3)	0	
Upper Columbia/Priest Rapids	3	(3)	0		0	
Middle Columbia/Lake Wallula	5	(3)	0		0	
Middle Columbia/Hood	4	(3)	0		0	
Lower Columbia/Sandy	1	(3)	0		0	
Lower Columbia Corridor	all	(3)	0		0	
Total	31		7		3	

¹ Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

The current condition of critical habitat designated for the UCR steelhead is moderately degraded. Habitat quality in tributary streams varies from excellent in wilderness and roadless areas to poor in areas subject to heavy agricultural and urban development (Figure 38). Critical habitat is affected by reduced quality of rearing and juvenile PCEs within many watersheds; contaminants from agriculture affect both water quality and food production in several watersheds and in the mainstem Columbia River. Several dams affect adult migration PCE by obstructing the migration corridor.

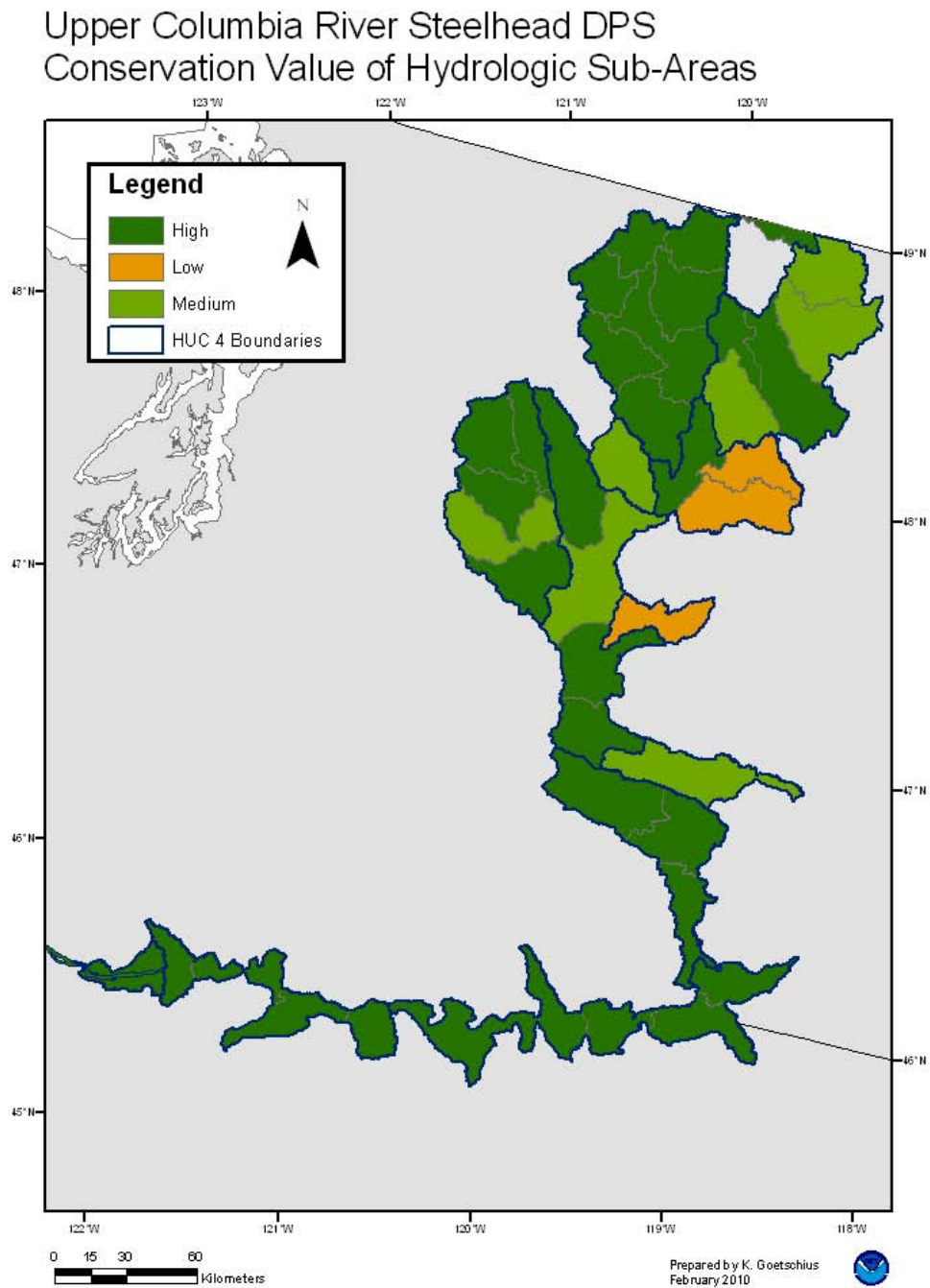


Figure 38. Upper Columbia River Steelhead Conservation Values per Sub-area.

Snake River Steelhead

The Snake River (SR) basin steelhead DPS includes all naturally spawned steelhead populations below natural and man-made impassable barriers in streams in the Columbia River Basin upstream from the Yakima River, Washington, to the U.S. - Canada border (Figure 39). Six artificial propagation programs are also included in the DPS: the Tucannon River, Dworshak National Fish Hatchery, Lolo Creek, North Fork Clearwater, East Fork Salmon River, and the Little Sheep Creek/Imnaha river hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than what would be expected between closely related natural populations within the DPS.

Life History

SR basin steelhead are generally classified as summer-run fish. They enter the Columbia River from late June to October. After remaining in the river through the winter, SR basin steelhead spawn the following spring (March to May). Managers recognize two life history patterns within this DPS primarily based on ocean age and adult size upon return: A-run or B-run. A-run steelhead are typically smaller, have a shorter freshwater and ocean residence (generally one year in the ocean), and begin their up-river migration earlier in the year. B-run steelhead are larger, spend more time in fresh water and the ocean (generally two years in ocean), and appear to start their upstream migration later in the year. SR basin steelhead usually smolt after two or three years.

Snake River Steelhead DPS Sub-Basin Range and Distribution

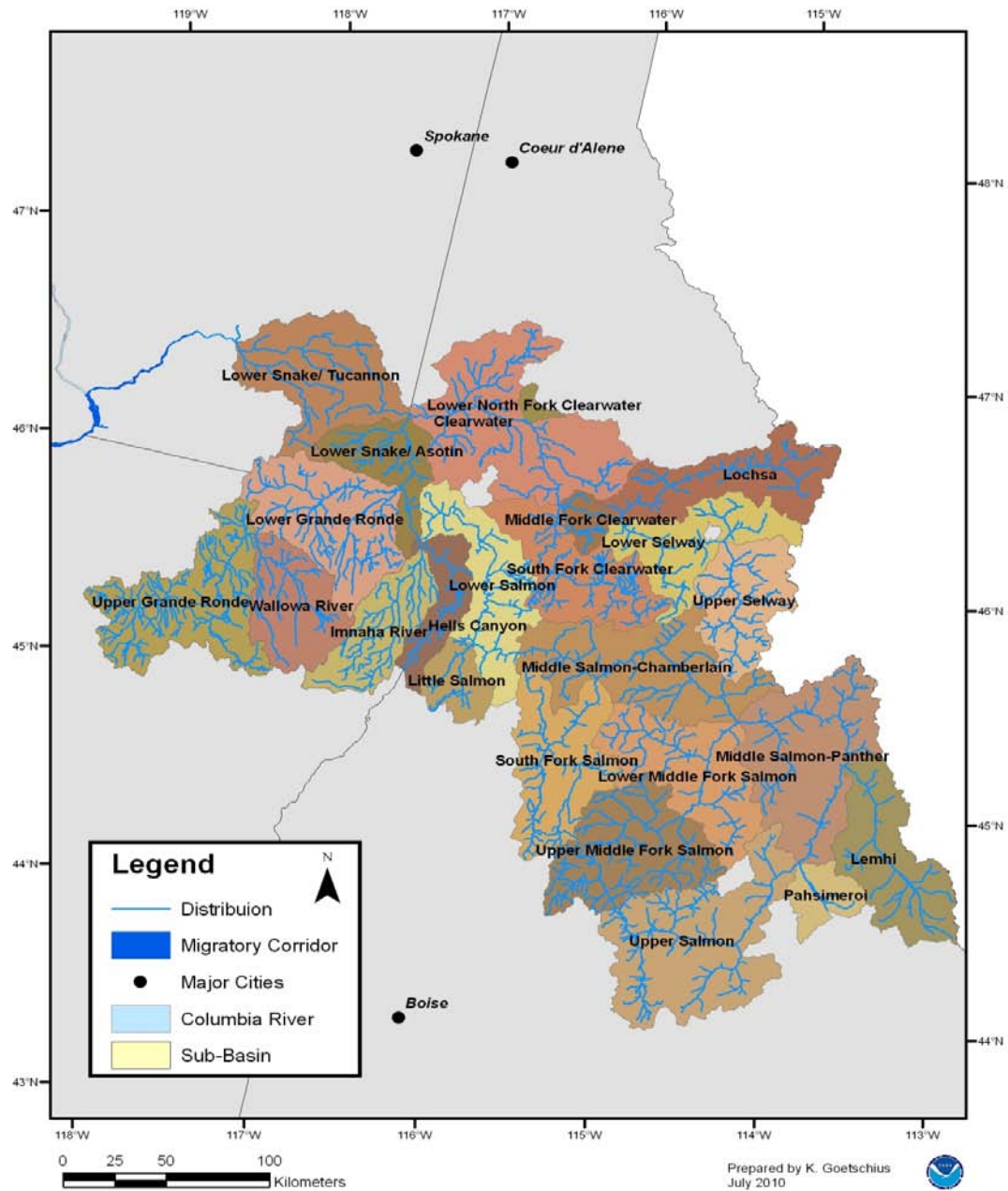


Figure 39. SR Basin Steelhead distribution.

Status and Trends

NMFS listed SR basin steelhead as threatened on August 18, 1997 (62 FR 43937), and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The ICTRT (ICTRT 2003) identified 23 populations. SR basin steelhead remain spatially well distributed in each of the six major geographic areas in the Snake River basin (Good, Waples et al. 2005). The SR basin steelhead B- run populations remain particularly depressed.

Table 42. SR Basin Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

River	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Tucannon River	3,000	257-628	26%
Lower Granite run	Unknown	70,721-259,145	86%
Snake A-run	Unknown	50,974-25,950	85%
Snake B-run	Unknown	9,736-33,195	89%
Asotin Creek	Unknown	0-543 redds	Unknown
Upper Grande Ronde River	15,000	1.54 rpm	23%
Joseph Creek	Unknown	1,077-2,385	0%
Imnaha River	4,000	3.7 rpm	20%
Camp Creek	Unknown	55-307	0%
Total	22,000 (min)	?	

Note: rpm denotes redds per mile.

A quantitative assessment for viability of SR steelhead is difficult given limited data on adult spawning escapement for specific tributary production areas. Annual return estimates are limited to counts of the aggregate return over Lower Granite Dam, and spawner estimates for the Tucannon, Asotin, Grande Ronde, and Imnaha Rivers (Table 42). The 2001 return over Lower Granite Dam was substantially higher relative to the low levels seen in the 1990s; the recent geometric five-year mean abundance (14,768 natural returns) was approximately 28% of the interim recovery target level (52,000 natural spawners). The 10-year average for natural-origin steelhead passing Lower Granite Dam between 1996 and 2005 is 28,303 adults. Parr densities in natural production areas, which are another indicator of population status, have been substantially below estimated capacity for several decades. The Snake River supports approximately 63% of the total natural-origin production of steelhead in the Columbia

River Basin. The current condition of Snake River Basin steelhead (Good, Waples et al. 2005) is summarized below.

There is uncertainty for wild populations given limited data for adult spawners in individual populations. Regarding population growth rate, there are mixed long- and short-term trends in abundance and productivity. Regarding spatial structure, the SR basin steelhead are well distributed with populations remaining in six major areas. However, the core area for B-run steelhead, once located in the North Fork of the Clearwater River, is now inaccessible to steelhead. Finally, genetic diversity is affected by the displacement of natural fish by hatchery fish (declining proportion of natural-origin spawners).

Overall, the abundances remain well below interim recovery criteria. The high proportion of hatchery produced fish in the runs remains a major concern.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). Figure 40 shows the conservation rankings per sub-area. Of the watersheds assessed, 229 were rated as having a high conservation value, 42 as medium value, and 12 as low value (Table 43). The Columbia River migration corridor was also given a high conservation value rating (NMFS 2005a).

The current condition of critical habitat designated for SR basin steelhead is moderately degraded. Critical habitat is affected by reduced quality of rearing and juvenile PCEs within many watersheds; contaminants from agriculture affect both water quality and food production in several watersheds and in the mainstem Columbia River. Loss of riparian vegetation to grazing has resulted in high water temperatures in the John Day

Table 43. SR steelhead watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Hells Canyon	3	(1, 2, 3)	0		0	
Imnaha River	5	(1)	0		0	
Lower Snake/Asotin	3	(1, 2, 3)	0		0	
Upper Grande Ronde	9	(1, 2)	2	(2, 1)	0	
Wallowa River	5	(1)	1	(1)	0	
Lower Grande Ronde	7	(1)	0		0	
Lower Snake/Tucannon	2	(1, 3)	2	(3, 1)	4	(1, 3)
Palouse River	0		1	(3, 1)	0	
Upper Salmon	20	(1)	6	(1)	1	(1)
Pahsimeroi	1	(1)	2	(1)	0	
Middle Salmon-Panther	16	(1, <3)	6	(1)	1	(1)
Lemhi	11	(1) ⁴	1	(1)	0	
Upper Middle Fork Salmon	13	(1)	0		0	
Lower Middle Fork Salmon	17	(1, <2)	0		0	
Middle Salmon-Chamberlain	14	(1, <3)	3	(3, 1)	1	(1)
South Fork Salmon	15	(1)	0		0	
Lower Salmon	12	(1, 3)	5	(1, 3)	0	
Upper Selway	9	(1, 3)	0		0	
Lower Selway	13	(1, 2)	0		0	
Lochsa	14	(1)	0		0	
Middle Fork Clearwater	2	(1)	0		0	
South Fork Clearwater	8	(1, 3)	3	(1)	2	(1, <3)
Clearwater	16	(1)	10	(1, 2, 3)	3	(1)
Lower Snake River	3	(3)	0		0	
Upper Columbia/Priest Rapids	1	(2)	0		0	
Middle Columbia/Lake Wallula	5	(2)	0		0	
Middle Columbia/Hood	4	(2)	0		0	
Lower Columbia/Sandy	1	(2)	0		0	

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Lower Columbia Corridor	all	(3)	0		0	
Total	229		42		12	

1 Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

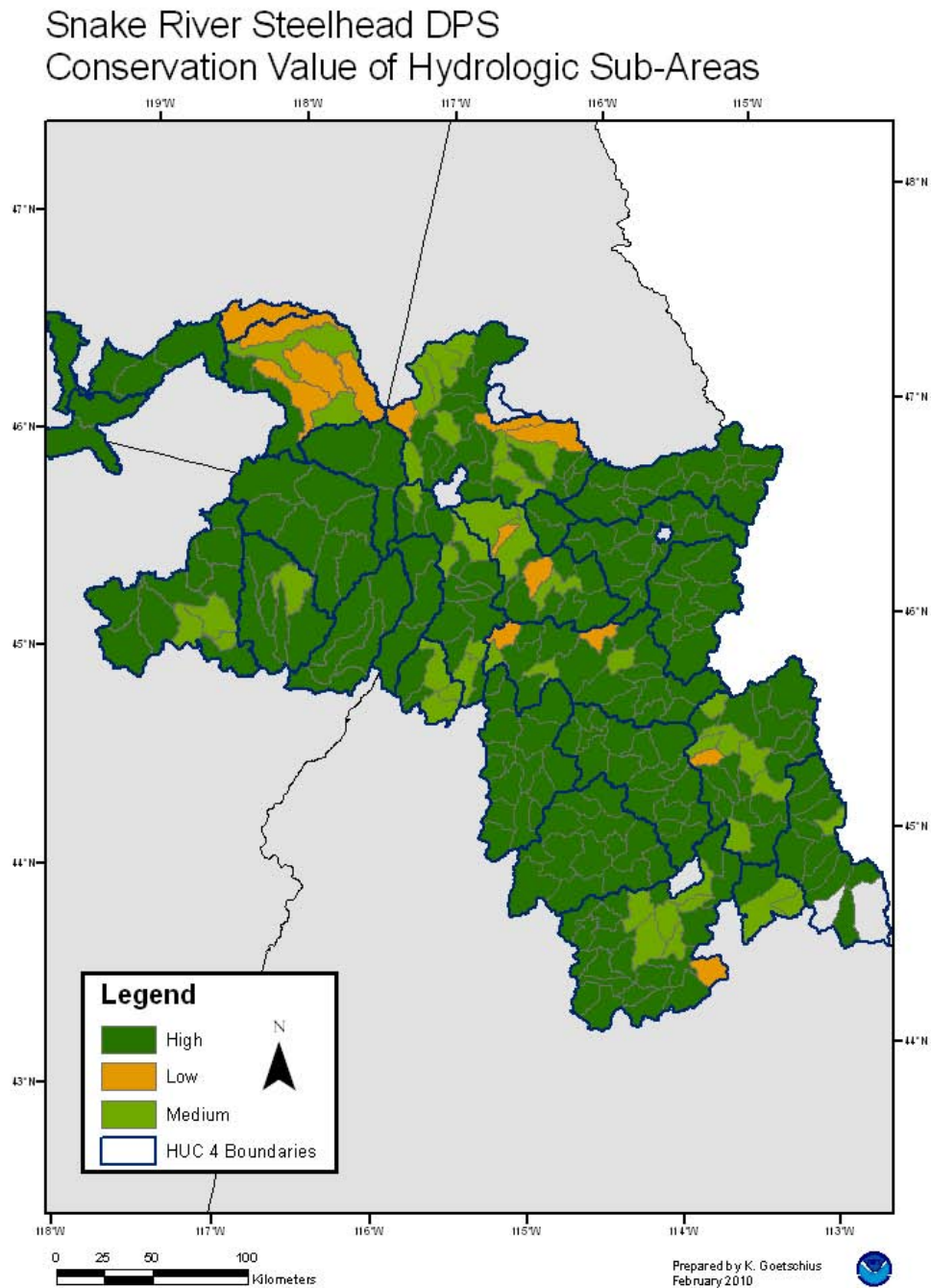


Figure 40. Snake River Steelhead Conservation Values per Sub-area.

basin. These factors have substantially reduced the rearing PCEs contribution to the conservation value necessary for species recovery. Several dams affect adult migration PCE by obstructing the migration corridor.

Northern California Steelhead

The NC steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in California coastal river basins from Redwood Creek southward to, but not including, the Russian River, as well as two artificial propagation programs: the Yeager Creek Hatchery, and North Fork Gualala River Hatchery (Gualala River Steelhead Project) steelhead hatchery programs (Figure 41).

Life History

This DPS includes both winter- and summer –run steelhead. In the Mad and Eel Rivers, immature steelhead may return to fresh water as “half-pounders” after spending only two to four months in the ocean. Generally, a half-pounder will overwinter in fresh water and return to the ocean in the following spring.

Juvenile out-migration appears more closely associated with size than age but generally, throughout their range in California, juveniles spend two years in fresh water (Busby et al 1996). Smolts range from 14-21 cm in length. Juvenile steelhead may migrate to rear in lagoons throughout the year with a peak in the late spring/early summer and in the late fall/early winter period ((Shapovalov and Taft 1954); (Zedonis 1992)).

Steelhead spend anywhere from one to five years in salt water, however, two to three years are most common (Busby, Wainwright et al. 1996). Ocean distribution is not well known but coded wire tag recoveries indicate that most NC steelhead migrate north and south along the continental shelf (Barnhart 1986).

Northern California Steelhead DPS Sub-Basin Range and Distribution

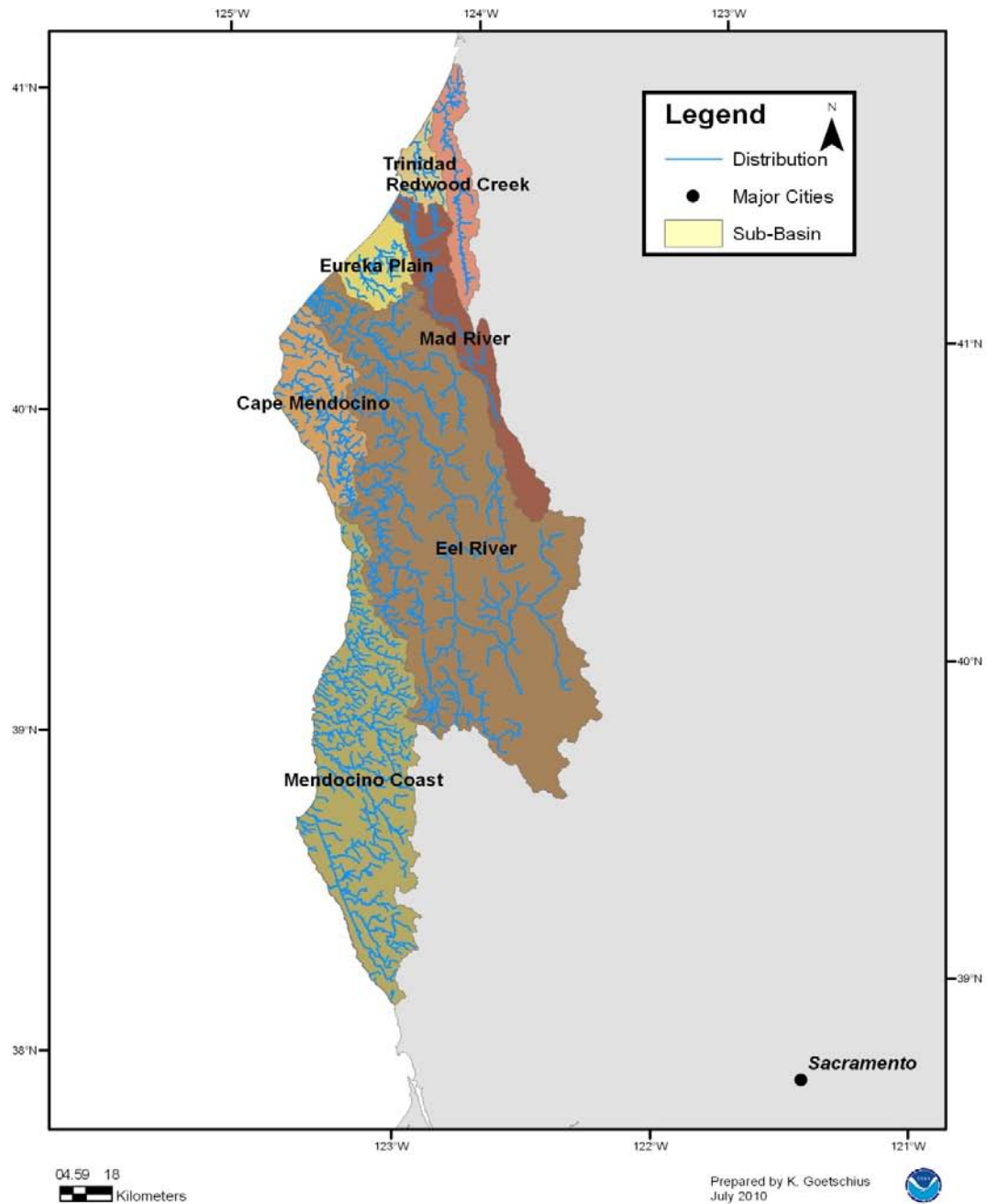


Figure 41. Northern California Steelhead distribution.

Status and Trends

NMFS listed NC steelhead as threatened on June 7, 2000 (65 FR 36074), and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The DPS encompass 15 historic functionally independent populations (and 22 potentially independent populations) of winter steelhead and 10 historic independent populations of summer steelhead (Bjorkstedt, Spence et al. 2005). Although the DPS spatial structure is relatively intact, the spatial structure and distribution within most watersheds have been adversely affected by barriers and high water temperatures. One of the basins, the Upper Mainstem Eel, has lost too much of its habitat to sustain an independent population today (Spence, Bjorkstedt et al. 2008). Production in the Mad River has been substantially reduced by the loss of 36% of its potential steelhead habitat. Large portions of the interior Russian River have been lost to the Coyote Valley Dam on the Russian River and the Warm Springs Hydroelectric Facility on Dry Creek, a major tributary to the Russian River. Spatial distribution in several smaller coastal watersheds has been impacted by constructed barriers blocking access to tributaries and headwaters.

Long-term data sets are limited for the NC steelhead. Before 1960, estimates of abundance specific to this DPS were available from dam counts in the upper Eel River (Cape Horn Dam—annual avg. no. adults was 4,400 in the 1930s), the South Fork Eel River (Benbow Dam—annual avg. no. adults was 19,000 in the 1940s), and the Mad River (Sweasey Dam— annual avg. no. adults was 3,800 in the 1940s). Estimates of steelhead spawning populations for many rivers in this DPS totaled 198,000 by the mid-1960s (Table 44).

Table 44. NC Steelhead salmon historic functionally independent populations and their abundances and hatchery contributions (Good, Waples et al. 2005).

Population	Historical Abundance	Recent Spawner Abundance	Hatchery Abundance Contributions
Mad River (S)	6,000	162-384	2%
MF Eel River (S)	Unknown	384-1,246	0%
NF Eel River (S)	Unknown	Extirpated	N/A
Mattole River (S)	Unknown	9-30*	Unknown
Redwood Creek (S)	Unknown	6*	Unknown

Population	Historical Abundance	Recent Spawner Abundance	Hatchery Abundance Contributions
Van Duzen (W)	10,000	Unknown	Unknown
Mad River (W)	6,000	Unknown	Unknown
SF Eel River (W)	34,000	2743-20,657	Unknown
Mattole River (W)	12,000	Unknown	Unknown
Redwood Creek (W)	10,000	Unknown	Unknown
Humboldt Bay (W)	3,000	Unknown	Unknown
Freshwater Creek (W)		25-32	
Ten Mile River (W)	9,000	Unknown	Unknown
Noyo River (W)	8,000	186-364*	Unknown
Big River (W)	12,000	Unknown	Unknown
Navarro River (W)	16,000	Unknown	Unknown
Garcia River (W)	4,000	Unknown	Unknown
Gualala River (W)	16,000	Unknown	Unknown
Total	198,000	Unknown	

*From Spence et al. (2008). Redwood Creek abundance is mean count over four generations. Mattole River abundances from surveys conducted between 1996 and 2005. Noyo River abundances from surveys conducted since 2000. Summer –run steelhead is noted with a (S) and winter-run steelhead with a (W)

During the first status review on this DPS, adult escapement trends were computed from seven populations. Five of the seven populations exhibited declines while two exhibited increases with a range of almost a 6% annual decline to a 3.5% increase. At that time, little information existed for the actual contribution of hatchery fish to natural spawning, and on present total run sizes for the DPS (Busby, Wainwright et al. 1996).

More recent time series data are from snorkel counts conducted on adult summer-run steelhead in the Middle Fork Eel River. Good *et al.* (2005) estimated lambda at 0.98 with a 95% confidence interval of 0.93 and 1.04. The result is an overall downward trend in both the long- and short- term. Juvenile data were also recently examined. Both upward and downward trends were apparent (Good, Waples et al. 2005).

Reduction of summer-run steelhead populations has significantly reduced current DPS diversity compared to historic conditions. Of the 10 summer-run steelhead populations, only four are extant. Of these, only the Middle Fork Eel River population is at moderate

risk of extinction, the remaining three are at high risk (Spence, Bjorkstedt et al. 2008). Hatchery influence has likely been limited.

Critical Habitat

NMFS designated critical habitat for NC steelhead on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Redwood Creek, Trinidad, Mad River, Eureka Plain, Eel River, Cape Mendocino, and the Mendocino Coast. The total area of critical habitat includes about 3,000 miles of stream habitat and about 25 square miles of estuarine habitat, mostly within Humboldt Bay.

There are 50 occupied CALWATER Hydrologic Subareas (HSA) watersheds within the freshwater and estuarine range of this ESU. Nine watersheds received a low rating, 14 received a medium rating, and 27 received a high rating of conservation value to the ESU (NMFS 2005) (Table 45, and Figure 42). Two estuarine habitat areas used for rearing and migration (Humboldt Bay and the Eel River Estuary) also received a high conservation value rating.

Table 45. NC steelhead CALWATER HSA watersheds with conservation values

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Redwood Creek	2	(1, 2, 3)	1	(1, 2, 3)	0	
Trinidad	1	(1, 2, 3)	0		1	(1, 2, 3)
Mad River	3	(1, 2, 3)	0		1	(1, 2, 3)
Eureka Plain	1	(1, 2, 3)	0		0	
Eel River	10	(1, 2, 3)	9	(1, 2, 3)	0	
Cape Mendocino	1	(1, 2, 3)	0		2	(1, 2, 3)
Mendocino Coast	9	(1, 2, 3)	4	(1, 2, 3)	5	(1, 2, 3)
Total	27		14		9	

¹ Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

The current condition of critical habitat designated for the NC steelhead is moderately degraded. Nevertheless, it does provide some conservation value necessary for species recovery. Within portions of its range, especially the interior Eel River, rearing PCE quality is affected by elevated temperatures by removal of riparian vegetation. Spawning PCE attributes such as the quality of substrate supporting spawning, incubation, and larval development have been generally degraded throughout designated critical habitat by silt and sediment fines in the spawning gravel. Bridges and culverts further restrict access to tributaries in many watersheds, especially in watersheds with forest road construction, thereby reducing the function of adult migration PCE.

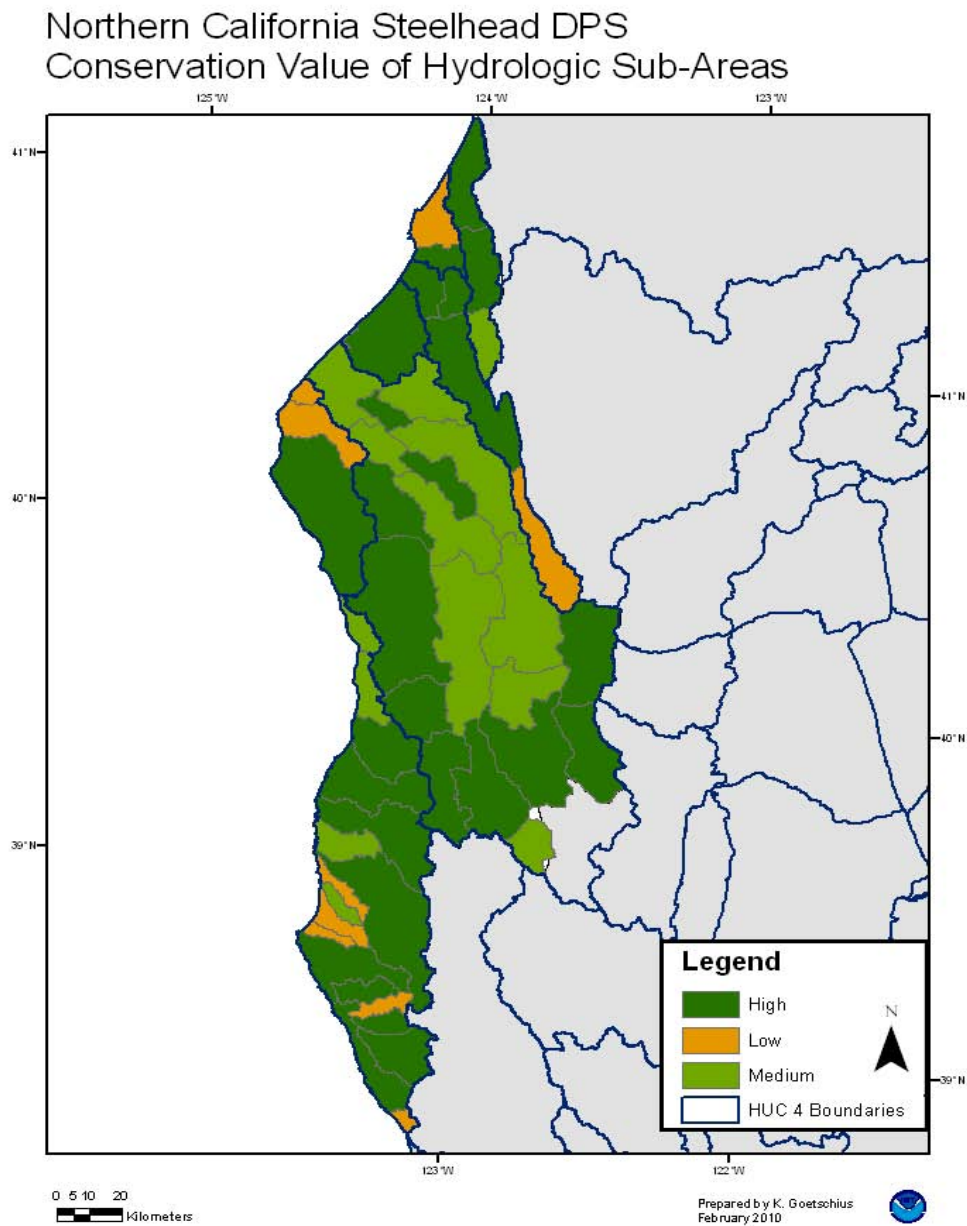


Figure 42. Northern California Steelhead Conservation Values per Sub-area.

Central California Coast Steelhead

The CCC steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in California streams from the Russian River

(inclusive) to Aptos Creek (inclusive), and the drainages of San Francisco, San Pablo, and Suisun Bays eastward to Chipps Island at the confluence of the Sacramento and San Joaquin Rivers (Figure 43).

Life History

The DPS is entirely composed of winter-run fish, as are those DPSs to the south. Adults return to the Russian River and migrate upstream from December – April, and smolts emigrate between March – May) ((Shapovalov and Taft 1954); (Hayes, Bond et al. 2004)). Most spawning takes place from January through April. While age at smoltification typically ranges for one to four years, recent studies indicate that growth rates in Soquel Creek likely prevent juveniles from undergoing smoltification until age two (Sogard, Williams et al. 2009). Survival in fresh water reaches tends to be higher in summer and lower from winter through spring for year classes 0 and 1 (Sogard, Williams et al. 2009). Larger individuals also survive more readily than do smaller fish within year classes (Sogard, Williams et al. 2009). Greater movement of juveniles in fresh water has been observed in winter and spring versus summer and fall time periods. Smaller individuals are more likely to be observed to exceed 0.3 mm per day, and are highest in winter through spring, potentially due to higher water flow rates and greater food availability ((Boughton, Adams et al. 2007); (Sogard, Williams et al. 2009)).

Central California Coast Steelhead DPS Sub-Basin Range and Distribution

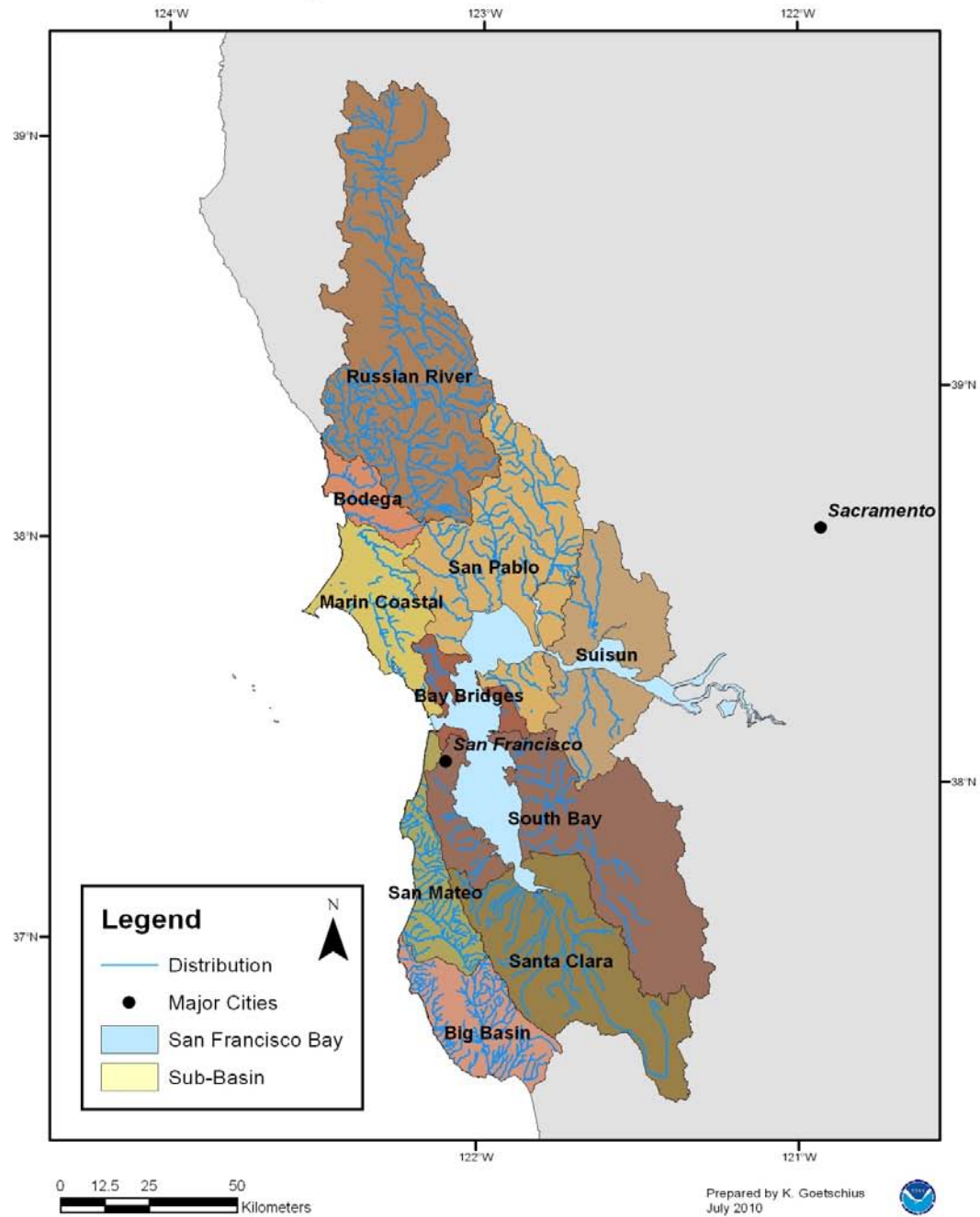


Figure 43. CCC steelhead distribution.

Status and Trends

NMFS listed CCC steelhead as threatened on August 18, 1997 (62 FR 43937), and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The CCC steelhead consisted of nine historic functionally independent populations and 23 potentially independent populations (Bjorkstedt, Spence et al. 2005). Of the historic functionally independent populations, at least two are extirpated while most of the remaining are nearly extirpated. Current runs in the basins that originally contained the two largest steelhead populations for CCC steelhead, the San Lorenzo and the Russian Rivers (Table 46), both have been estimated at less than 15% of their abundances just 30 years earlier (Good, Waples et al. 2005). Steelhead access to significant portions of the upper Russian River has also been blocked (Busby, Wainwright et al. 1996; NMFS 2008).

Table 46. CCC Steelhead populations, historic population type, abundances, and hatchery contributions (Good, Waples et al. 2005; NMFS 2008) .

Basin	Pop. Type	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Upper Russian River	FI	65,000 (1970)	1,750-7,000 (1994)	Unknown
Lagunitas Creek	PI	Unknown	400-500 (1990s)	Unknown
Stemple Creek	PI	Unknown	Extirpated	N/A
Americano Creek	PI	Unknown	Extirpated	N/A
San Gregorio	FI	1,000 (1973)	Unknown	Unknown
Waddell Creek	PI	481	150 (1994)	Unknown
Scott Creek	D	Unknown	<100 (1991)	Unknown
San Vicente Creek	D	150 (1982)	50 (1994)	Unknown
San Lorenzo River	FI	20,000	<150 (1994)	Unknown
Soquel Creek	PI	500-800 (1982)	<100 (1991)	Unknown
Aptos Creek	PI	200 (1982)	50-75 (1994)	Unknown
Guadalupe River	FI	Unknown	Unknown	Unknown
Napa River	FI	Unknown	Unknown	Unknown
San Leandro River	FI	Unknown	Extirpated*	N/A
San Lorenzo River	FI	20,000 pre-1965	<150 (1994)	N/A
Alameda Creek	FI	Unknown	Extirpated	N/A
Total		94,000	2,400-8,125	

*A remnant stray run may still exist (Leidy, Becker et al. 2005)

Population type: FI, historic functionally independent; PI, historic potentially independent.

Historically, the entire CCC steelhead DPS may have consisted of an average runs size of 94,000 adults in the early 1960s (Good, Waples et al. 2005). Information on current CCC

steelhead populations consists of anecdotal, sporadic surveys that are limited to only smaller portions of watersheds. Presence-absence data indicated that most (82%) sampled streams (a subset of all historical steelhead streams) had extant populations of juvenile *O. mykiss* ((Adams 2000); (Good, Waples et al. 2005)). Table YY identifies populations within the CCC steelhead salmon ESU, their abundances, and hatchery input.

Though the information for individual populations is limited, available information strongly suggests that no population is viable. Long-term population sustainability is extremely low for the southern populations in the Santa Cruz mountains and in the San Francisco Bay (NMFS 2008). Declines in juvenile southern populations are consistent with the more general estimates of declining abundance in the region (Good, Waples et al. 2005). The interior Russian River winter-run steelhead has the largest runs with an estimate of an average of over 1,000 spawners; it may be able to be sustained over the long-term but hatchery management has eroded the population's genetic diversity (Bjorkstedt, Spence et al. 2005; NMFS 2008).

Data on abundance trends do not exist for the DPS as a whole or for individual watersheds. Thus, it is not possible to calculate long-term trends or lambda.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). It includes the Russian River watershed, coastal watersheds in Marin County, streams within the San Francisco Bay, and coastal watersheds in the Santa Cruz Mountains down to Apos Creek.

There are 47 occupied HSA watersheds within the freshwater and estuarine range of this ESU. As shown in Figure 44, fourteen watersheds are considered of low conservation value, 13 as having a medium conservation value, and 19 as having a high conservation value to the ESU (NMFS 2005) (Table 47). Five of these HSA watersheds comprise

portions of the San Francisco-San Pablo- Suisun Bay estuarine complex which provides rearing and migratory habitat for this ESU.

Table 47. CCC steelhead CALWATER HSA watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Russian River	7	(1, 2, 3)	2	(1, 2, 3)	1	(1, 2, 3)
Bodega Bay	0		1	(1, 2, 3)	1	(1, 2, 3)
Coastal Marin County	1	(1, 2, 3)	1	(1, 2, 3)	2	(1, 2, 3)
San Mateo	2	(1, 2, 3)	2	(1, 2, 3)	1	(1, 2, 3)
Bay Bridges	1	(estuarine PCEs)	1	(1, 2, 3)	1	(1, 2, 3)
South Bay	1	(estuarine PCEs)	1	(1, 2, 3)	1	(1 mi of 2 and 3)
Santa Clara	1	(estuarine PCEs)	2	(1, 2, 3)	2	(1, 2, 3)
San Pablo	3	(1, 2, 3)	1	(1, 2, 3)	2	(1, 2, 3)
Suisun	0		1	(1, 2, 3)	4	(1, 2, 3)
Big Basin	3	(1, 2, 3)	1	(1, 2, 3)	0	
Total	19		13		15	

¹ Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

Streams throughout the critical habitat have reduced quality of spawning PCEs; sediment fines in spawning gravel have reduced the ability of the substrate attribute to provide well oxygenated and clean water to eggs and alevins. High proportions of fines in bottom substrate also reduce forage by limiting the production of aquatic stream insects adapted to running water. Elevated water temperatures and impaired water quality have further reduced the quality, quantity and function of the rearing PCE within most streams. These impacts have diminished the ability of designated critical habitat to conserve the CCC steelhead.

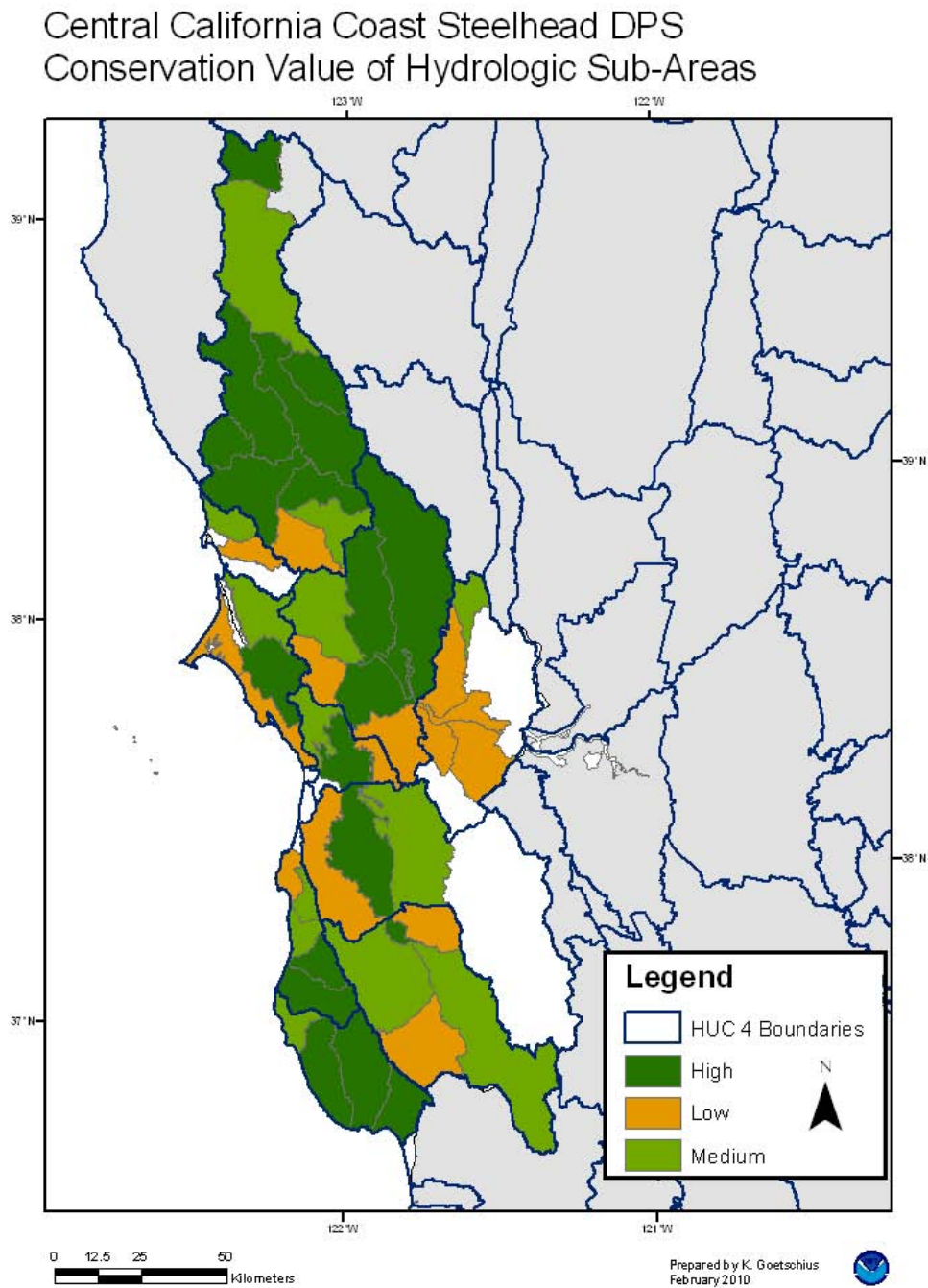


Figure 44. Central California Coast Steelhead Conservation Values per Sub-area.

California Central Valley Steelhead

The California Central Valley (CCV) steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in the Sacramento and San Joaquin Rivers and their tributaries, excluding steelhead from San Francisco and San Pablo Bays and their tributaries, as well as two artificial propagation programs: the Coleman NFH, and Feather River Hatchery steelhead hatchery programs (Figure 45).

Life History

CCV steelhead are considered winter steelhead and have the longest freshwater migration of any population of winter steelhead. CCV steelhead generally leave the ocean from August through April (Busby, Wainwright et al. 1996), and spawn from December through April, with peaks from January through March, in small streams and tributaries where cool, well oxygenated water is available year-round (Hallock, Van Woert et al. 1961; McEwan and Jackson 1996). Most spawning habitat for steelhead in the Central Valley is located in areas directly downstream of dams containing suitable environmental conditions for spawning and incubation.

Newly emerged fry move to the shallow, protected areas associated with the stream margin (McEwan and Jackson 1996). Steelhead rearing during the summer occurs primarily in higher velocity areas in pools, although young of the year also are abundant in glides and riffles. Both spawning areas and migratory corridors comprise rearing habitat for juveniles, which feed and grow before and during their outmigration. Non-natal, intermittent tributaries also may be used for juvenile rearing. Migratory corridors are downstream of the spawning areas and include the lower mainstems of the Sacramento and San Joaquin rivers and the Delta.

Hallock *et al.* (1961) found that juvenile steelhead in the Sacramento River basin migrate downstream during most months of the year, but the peak period of emigration occurred in the spring, with a much smaller peak in the fall. Emigrating CCV steelhead use the

lower reaches of the Sacramento River and the Delta for rearing and as a migration corridor to the ocean. Some juvenile steelhead may use tidal marsh areas, non-tidal freshwater marshes, and other shallow water areas in the Delta as rearing areas for short periods prior to their final emigration to the sea (Hallock, Van Woert et al. 1961).

California Central Valley Steelhead DPS Sub-Basin Range and Distribution

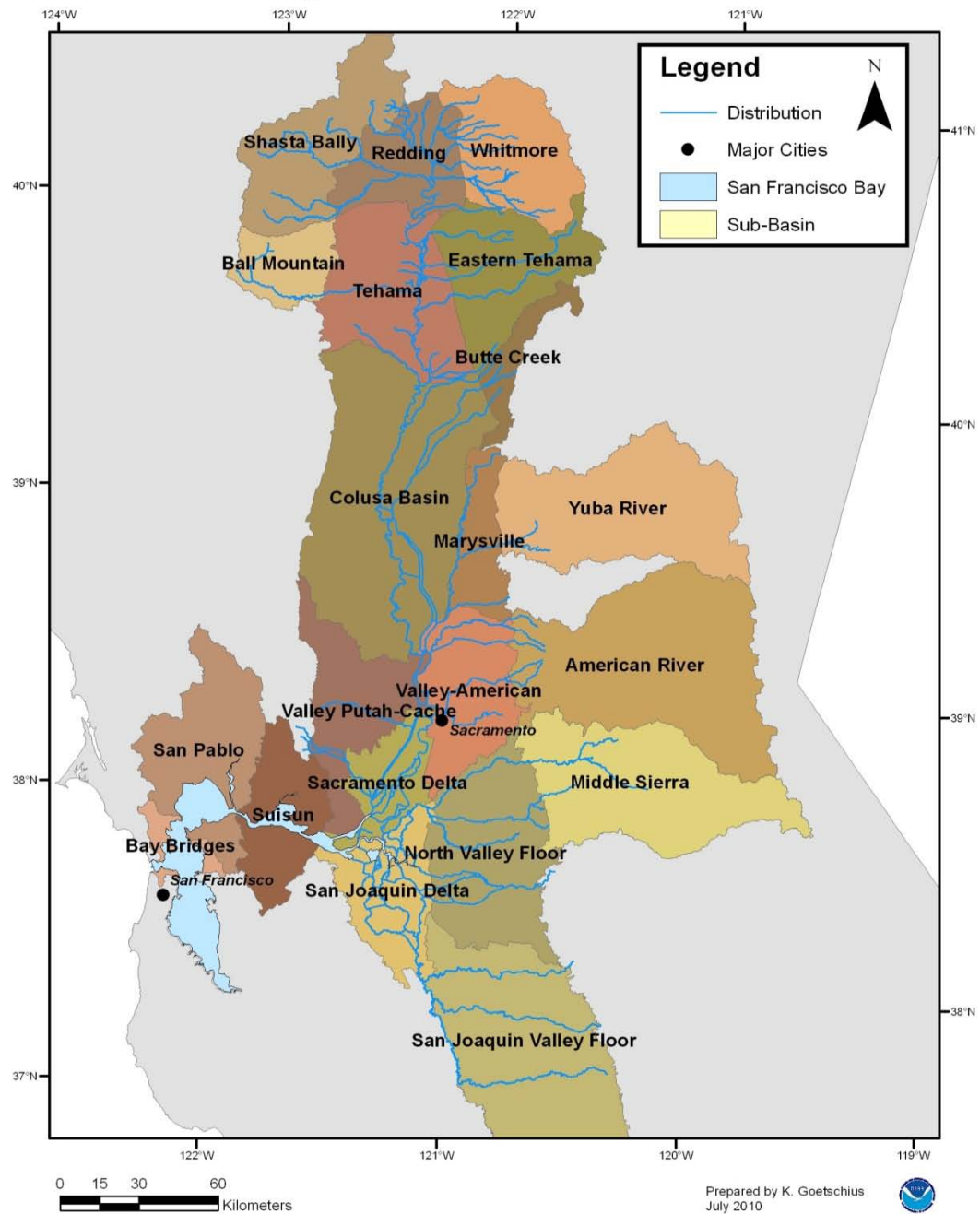


Figure 45. CCV steelhead distribution.

Status and Trends

NMFS originally listed CCV steelhead as threatened on March 19, 1998, and reaffirmed their threatened status on January 5, 2006 (71 FR 834). The CCV steelhead DPS may have consisted of 81 historical and independent populations (Lindley, Schick et al. 2006). Spatial structure and patchiness strongly influenced suitable habitats being isolated due largely to high summer temperatures on the valley floor.

The species' present distribution has been greatly reduced with about 80% of historic habitat lost behind dams and about 38% of habitat patches that supported independent populations are no longer accessible to steelhead (Lindley, Schick et al. 2006). Existing wild steelhead stocks in the Central Valley are mostly confined to the upper Sacramento River and its tributaries, including Antelope, Deer, and Mill Creeks and the Yuba River. Populations may exist in Big Chico and Butte Creeks. A few wild steelhead are produced in the American and Feather Rivers (Good, Waples et al. 2005). Steelhead have also been observed in Clear Creek and Stanislaus River (Demko and Cramer 2000; Good, Waples et al. 2005). Until recently, steelhead were considered extirpated from the San Joaquin River system. Recent monitoring has detected small self-sustaining populations of steelhead in the Stanislaus, Mokelumne, Calaveras, and other streams previously thought to be void of steelhead (Good, Waples et al. 2005). In 2004, a total of 12 steelhead smolts were collected in monitoring trawls at the Mossdale station in the lower San Joaquin River (CDFG unpublished data).

Historic CCV steelhead run size may have approached one to two million adults annually (McEwan 2001). By the early 1960s, the steelhead run size had declined to about 40,000 adults (McEwan 2001). Over the past 30 years, the naturally spawned steelhead populations in the upper Sacramento River have declined substantially. Hallock *et al.* (1961) estimated an average of 20,540 adult steelhead in the Sacramento River, upstream of the Feather River, through the 1960s. Steelhead were counted at the Red Bluff Diversion Dam (RBDD) up until 1993. Counts at the dam declined from an average of 11,187 for the period of 1967 to 1977, to an average of approximately 2,000 through the

early 1990s. An estimated total annual run size for the entire Sacramento-San Joaquin system was no more than 10,000 adults during the early 1990s (McEwan and Jackson 1996; McEwan 2001). Based on catch ratios at Chipps Island in the Delta and using some generous assumptions regarding survival, the average number of CV steelhead females spawning naturally in the entire Central Valley during the years 1980 to 2000 was estimated at about 3,600 (Good, Waples et al. 2005).

CCV steelhead lack annual monitoring data for calculating trends and lambda. However, the RBDD counts and redd counts up to 1993 and later sporadic data show that the DPS has had a significant long-term downward trend in abundance (NMFS 2009).

The CCV steelhead distribution ranged over a wide variety of environmental conditions and likely contained biologically significant amounts of spatially structured genetic diversity (Lindley, Schick et al. 2006). Thus, the loss of populations and reduction in abundances have reduced the large diversity that existed within the DPS. The genetic diversity of the majority of CCV steelhead spawning runs is also compromised by hatchery-origin fish.

Critical Habitat

NMFS designated critical habitat for CCV steelhead on September 2, 2005 (70 FR 52488). Critical habitat includes stream reaches such as those of the Sacramento, Feather, and Yuba Rivers, and Deer, Mill, Battle, and Antelope creeks in the Sacramento River basin; the lower San Joaquin River to the confluence with the Merced River, including its tributaries, and the waterways of the Delta (Figure 46). The total area of critical habitat includes about 2,300 miles of stream habitat and about 250 square miles of estuarine habitat in the San Francisco-San Pablo-Suisan Bay estuarine complex.

There are 67 occupied HAS watersheds within the freshwater and estuarine range of this DPS. Twelve watersheds received a low rating, 18 received a medium rating, and 37 received a high rating of conservation value to the ESU (NMFS 2005). Four of these

HSA watersheds comprise portions of the San Francisco-San Pablo-Suisun Bay estuarine complex which provides rearing and migratory habitat for this ESU.

Table 48. CCV spring-run Chinook salmon CALWATER HSA watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
San Francisco Bay	1	2	0		0	
South Bay	0		0		1	2
San Pablo	1	2	0		0	
Suisun Bay	1	2	0		0	
Tehama	1	1, 2, 3	1	1, 2, 3	0	
Whitmore	3	1, 2, 3	2	1, 2, 3	2	1, 2, 3
Redding	2	1, 2, 3	0		0	
Eastern Tehama	4	1, 2, 3	1	1, 2, 3	1	1, 2, 3
Sacramento Delta	1	1, 2, 3	0		0	
Valley Putah-Cache	0		2	1, 2, 3	0	
American River	0		1	1, 2, 3	0	
Marysville	2	1, 2, 3	1	1, 2, 3	0	
Yuba River	2	1, 2, 3	0		2	1, 2, 3
Valley-American	2	1, 2, 3	0		0	
Colusa Basin	4	1, 2, 3	0		0	
Butte Creek	1	1, 2, 3	1	1, 2, 3	1	1, 2, 3
Ball Mountain	1	1, 2, 3	0		0	
Shasta Bally	2	1, 2, 3	3	1, 2, 3	0	
North Valley Floor	1	1, 2, 3	1	1, 2, 3	1	1, 2, 3
Middle Sierra	0		0		4	1, 2, 3
Upper Calaveras	1	1, 2, 3	0		0	
Stanislaus River	1	1, 2, 3	0		0	
San Joaquin Valley Floor	4	1, 2, 3	3	1, 2, 3	0	
Delta-Mendota Canal	1	1, 2, 3	1	1, 2, 3	0	
North Diablo Range	0		1		0	
San Joaquin Delta	1	1, 2, 3	0		0	
Total	37		18		12	

¹ Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

The current condition of CCV steelhead critical habitat is degraded, and does not provide the conservation value necessary for species recovery (Table 48). In addition, the Sacramento-San Joaquin River Delta, as part of CCV steelhead designated critical

habitat, provides very little function necessary for juvenile CCV steelhead rearing and physiological transition to salt water.

The spawning PCE is subject to variations in flows and temperatures, particularly over the summer months. Some complex, productive habitats with floodplains remain in the system and flood bypasses (*i.e.*, Yolo and Sutter bypasses). However, the rearing PCE is degraded by the channelized, leveed, and riprapped river reaches and sloughs that are common in the Sacramento-San Joaquin system and which typically have low habitat complexity, low abundance of food organisms, and offer little protection from either fish or avian predators. Stream channels commonly have elevated temperatures.

The current conditions of migration corridors are substantially degraded. Both migration and rearing PCEs are affected by dense urbanization and agriculture along the mainstems and in the Delta which contribute to reduced water quality by introducing several contaminants. In the Sacramento River, the migration corridor for both juveniles and adults is obstructed by the RBDD gates which are down from May 15 through September 15. The migration PCE is also obstructed by complex channel configuration making it more difficult for CCV steelhead to migrate successfully to the western Delta and the ocean. In addition, the state and federal government pumps and associated fish facilities change flows in the Delta which impede and obstruct for a functioning migration corridor that enhance migration. The estuarine PCE, which is present in the Delta, is affected by contaminants from agricultural and urban runoff and release of wastewater treatment plants effluent.

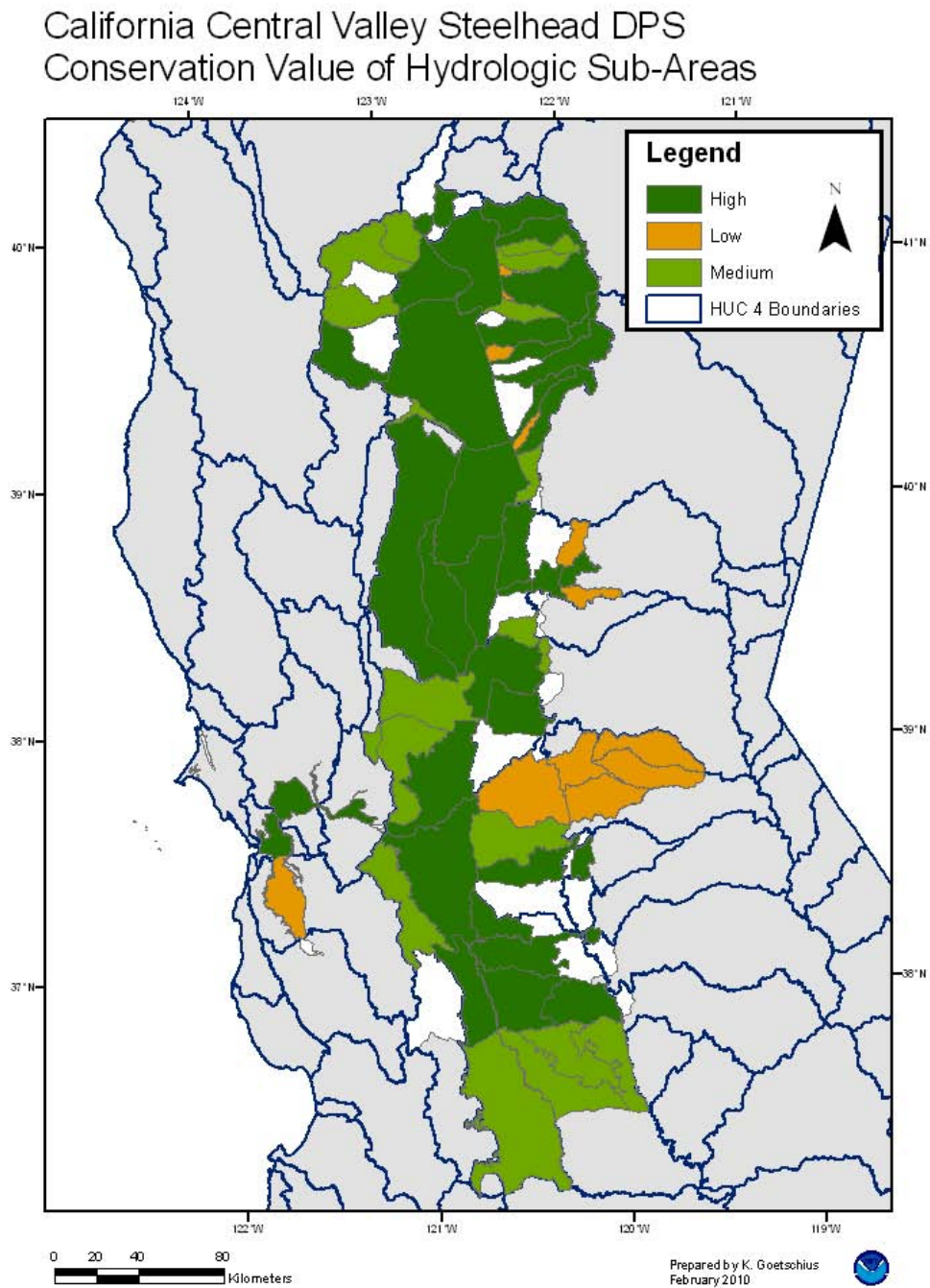


Figure 46. California Central Valley Steelhead Conservation Value per Sub-area.

South-Central California Coast Steelhead

South-Central California Coast (S-CCC) steelhead include all naturally spawned steelhead populations below natural and manmade impassable barriers in streams from the Pajaro River (inclusive) to, but not including the Santa Maria River, California. No artificially propagated steelhead populations that reside within the historical geographic range of this DPS are included in this designation. The two largest basins overlapping within the range of this DPS include the inland basins of the Pajaro River and the Salinas River (Figure 48).

Life History

Only winter steelhead are found in this DPS. Migration and spawn timing are similar to adjacent steelhead populations. There is limited life history information for steelhead in this DPS.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). There are 29 occupied HSA watersheds within the freshwater and estuarine range of this ESU. Figure 47 depicts the conservation values for this DPS. The conservation value of 6 watersheds is low, 11 are of medium conservation value, and 12 are of a high conservation value to the ESU (Table 49)(NMFS 2005). One of these occupied watershed units is Morro Bay, which is used as rearing and migratory habitat for steelhead populations that spawn and rear in tributaries to the Bay .

Table 49. Number of South-Central California Coast steelhead CALWATER HSA watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Pajaro River	2	(2, 3, 1)	3	(2, 3, 1)	0	
Carmel River	1	(1, 2, 3)	0		0	
Santa Lucia	1	(1, 2, 3)	0		0	
Salinas	2	(2, 3, 1)	1	(1, 2)	4	(2, 3, <1)
Estero Bay	6	(2, 1, 3)	7	(1, 2, 3)	2	(1, 2, 3)
Total	12		11		6	

¹ Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

Migration and rearing PCEs are degraded throughout critical habitat by elevated stream temperatures and contaminants from urban and agricultural areas. Estuarine PCE is impacted by most estuaries being breached, removal of structures, and contaminants.

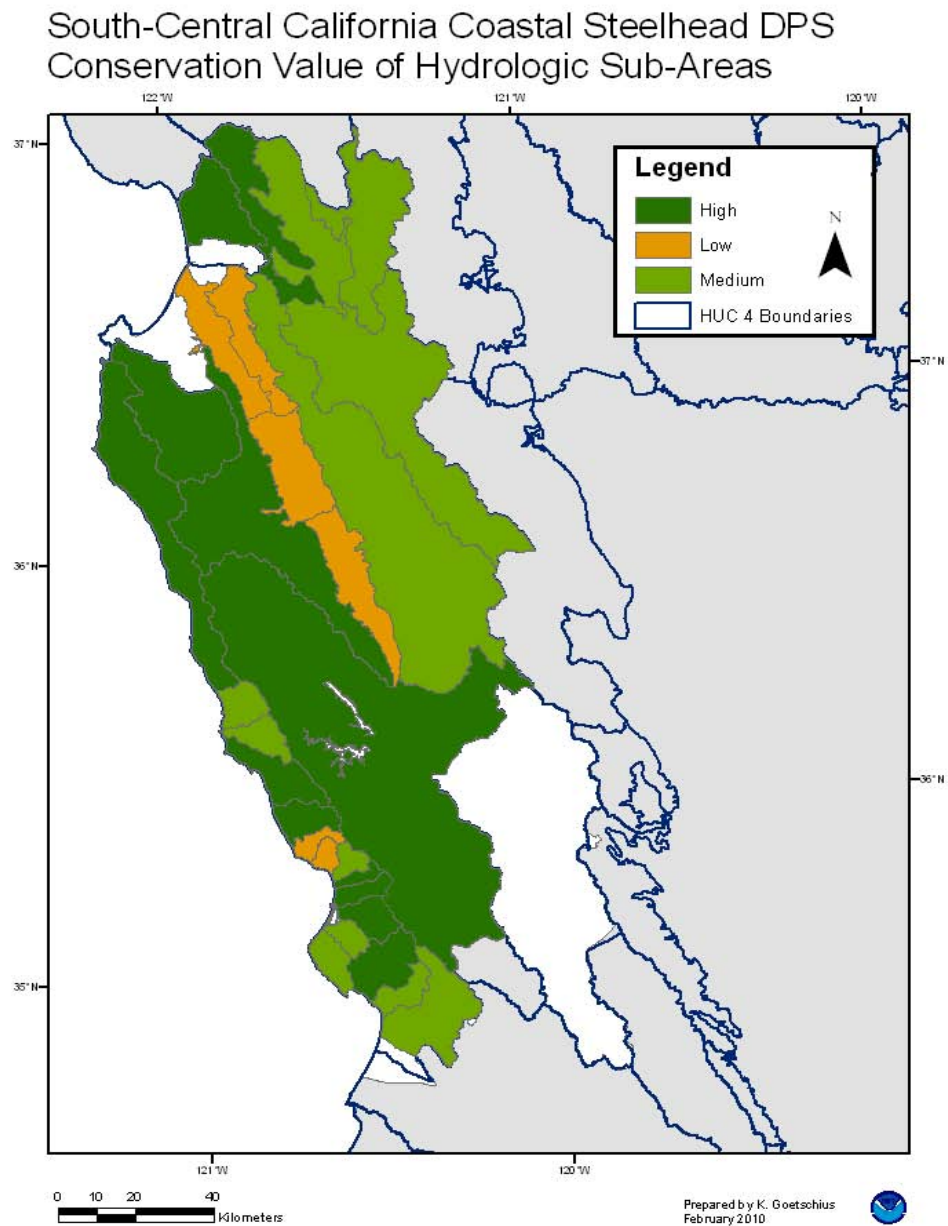


Figure 47. South-Central California Coast Steelhead Conservation Values per Sub-area.

South-Central California Coastal Steelhead DPS Sub-Basin Range and Distribution

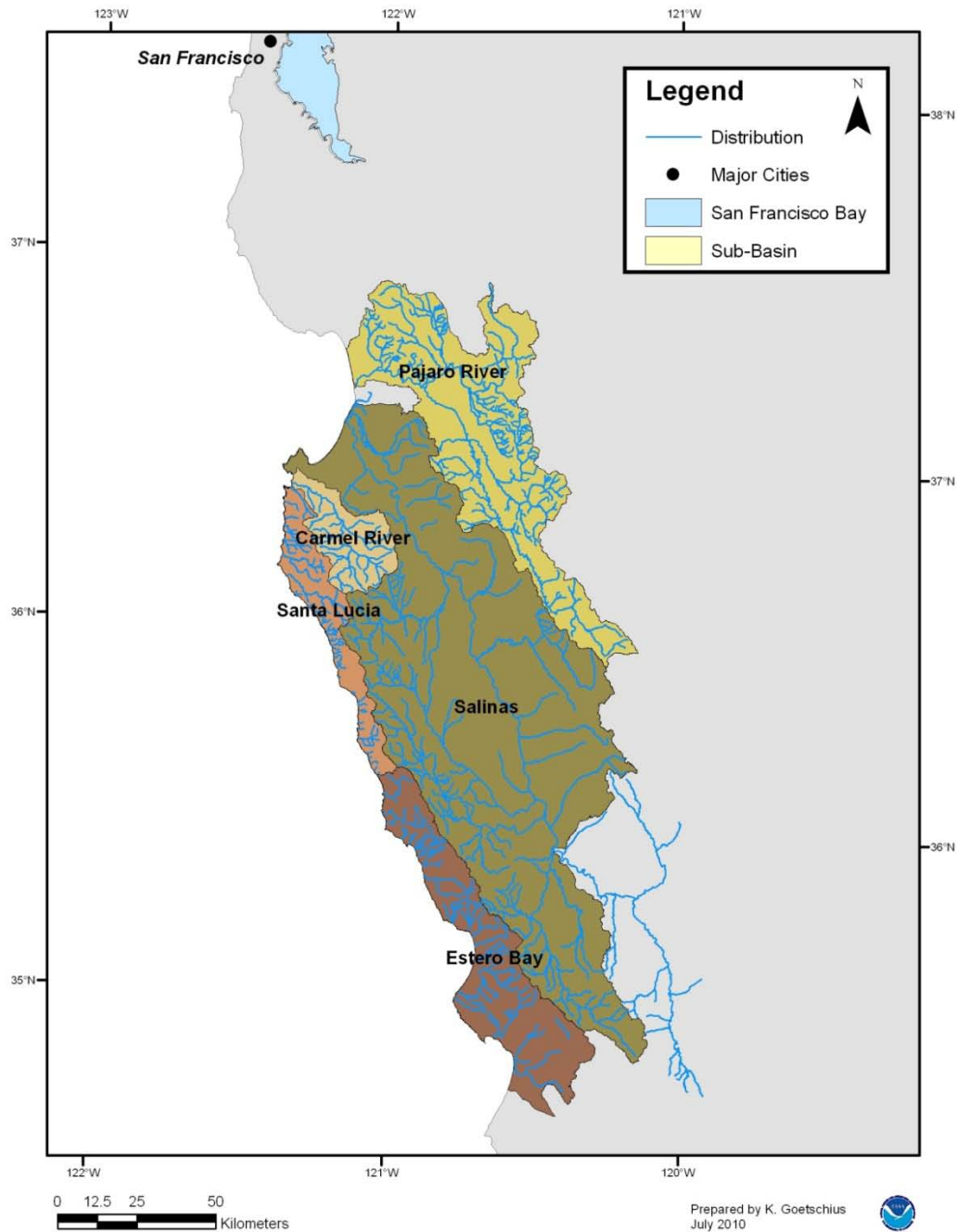


Figure 48. S-CCC steelhead distribution.

Southern California Steelhead

The Southern California (SC) steelhead DPS includes all naturally spawned steelhead populations below natural and manmade impassable barriers in streams from the Santa Maria River, San Luis Obispo County, California, (inclusive) to the U.S. - Mexico Border (Figure 49). Artificially propagated steelhead that reside within the historical geographic range of this DPS are not included in the listing.

Life History

There is limited life history information for SC steelhead. In general, migration and life history patterns of SC steelhead populations are dependent on rainfall and stream flow (Moore 1980). Steelhead within this DPS can withstand higher temperatures compared to populations to the north. The relatively warm and productive waters of the Ventura River have resulted in more rapid growth of juvenile steelhead compared to the more northerly populations (Moore 1980).

Southern California Steelhead DPS Sub-Basin Range and Distribution

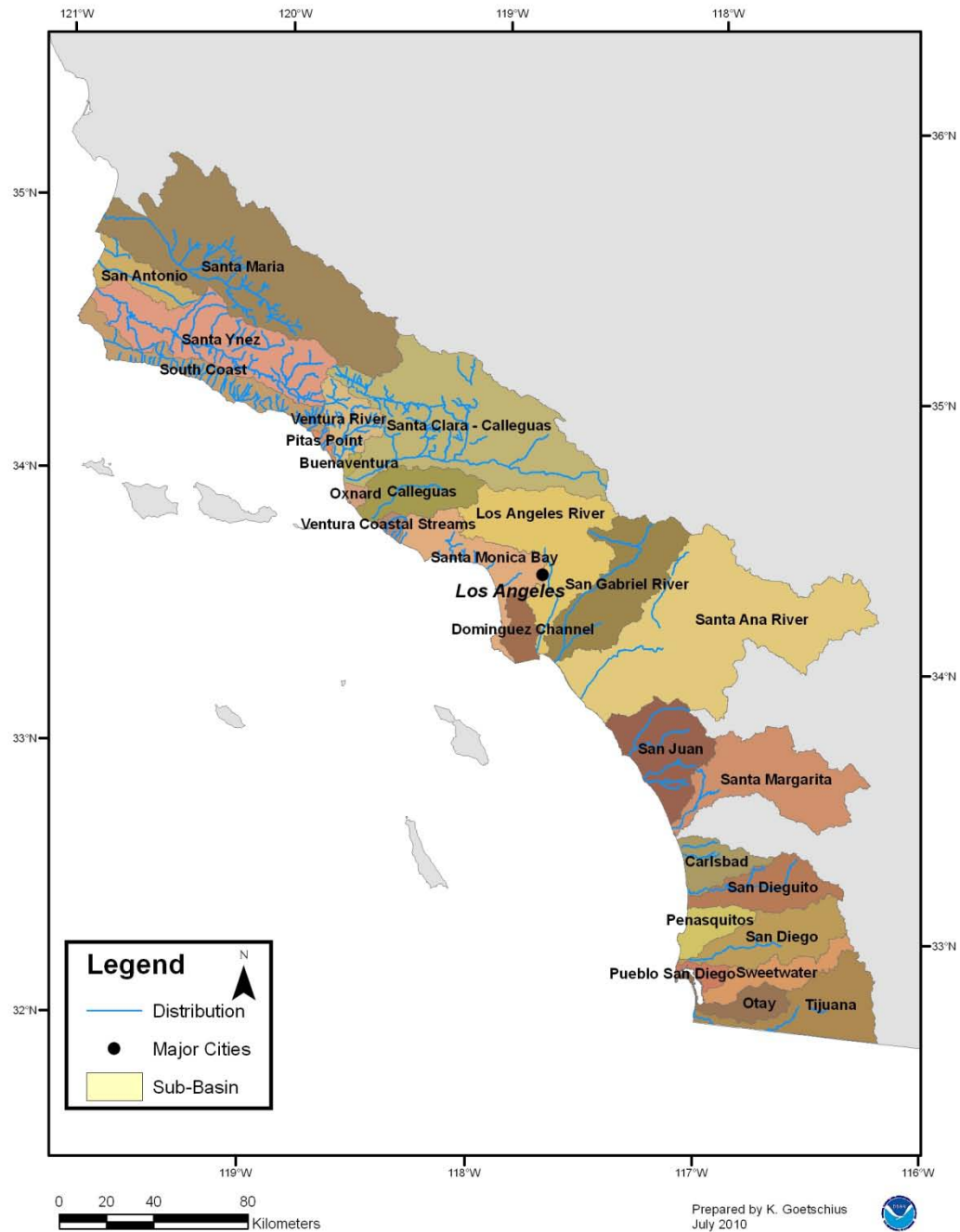


Figure 49. Southern California steelhead distribution.

Status and Trends

NMFS listed the SC steelhead as endangered on August 18, 1997 (62 FR 43937), and reaffirmed their endangered status on January 5, 2006 (71 FR 834). Historic population structure and evaluation of potential stratification of the DPS have not been conducted for this DPS (Table 50).

Table 50. Southern California Steelhead salmon populations, abundances, and hatchery contributions (Good, Waples et al. 2005).

River	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Santa Ynez River	12,995-30,000	Unknown	Unknown
Ventura River	4,000-6,000	Unknown	Unknown
Matilija River	2,000-2,500	Unknown	Unknown
Creek River	Unknown	Unknown	Unknown
Santa Clara River	7,000-9,000	Unknown	Unknown
Total	32,000-46,000	<500	

Construction of dams and corresponding increase in water temperatures have excluded steelhead distribution in many watersheds throughout southern California. Streams in southern California with steelhead present have declined over the last decade with a southward increase in the proportional loss of populations. Consequently, the SC steelhead have experienced a contraction of its southern range limit (Boughton, Fish et al. 2005). This contraction affects the SC steelhead's ability to maintain genetic and life history diversity for adaptation to environmental change

Limited information exists on SC steelhead runs. Based on combined estimates for the Santa Ynez, Ventura, and Santa Clara rivers, and Malibu Creek, an estimated 32,000 to 46,000 adult steelhead occupied this DPS historically. In contrast, less than 500 adults are estimated to occupy the same four waterways presently. The last estimated run size for steelhead in the Ventura River, which has its headwaters in Los Padres National Forest, is 200 adults (Busby, Wainwright et al. 1996). Table 50 identifies populations within the SC Steelhead salmon ESU, their abundances, and hatchery input.

Critical Habitat

Critical habitat was designated for this species on September 2, 2005 (70 FR 52630). There are 29 HSA watersheds within the freshwater and estuarine range of this ESU designated as critical habitat (Table 51). Figure 50 provides conservation values for this DPS per sub-area. Three watersheds received a low, five received a medium, and 21 received a high conservation value rating for the ESU (NMFS 2005).

Table 51. Southern California steelhead CALWATER HSA watersheds with conservation values.

HUC 4 Subbasin	HUC 5 Watershed conservation Value (CV)					
	High CV	PCE(s) ¹	Medium CV	PCE(s) ¹	Low CV	PCE(s) ¹
Santa Maria	1	(1, 2, 3)	0		1	(1, 2, 3)
Santa Ynez	2	(2, 3, 1)	2	(1, 2, 3)	1	(2, 3, 1)
South Coast	5	(2, 3, 1)	0		0	
Ventura River	2	(2, 3, 1)	2	(1, 2, 3)	0	
Santa Clara-Calleguas	5	(2, 3, 1)	1	(2, 3)	0	
Santa Monica Bay	3	(2, 1, 3)	0		0	
Calleguas	0		0		1	(2, 3)
San Juan	3	(2, 3, 1)	0		0	
Total	21		5		3	

¹ Numbers in parenthesis refers to the dominant (in river miles) PCE(s) within the HUC 5 watersheds. PCE 1 is spawning and rearing, 2 is rearing and migration, and 3 is migration and presence. PCEs with < means that the number of river miles of the PCE is much less than river miles of the other PCE.

All PCEs have been affected by degraded water quality by pollutants from densely populated areas and agriculture within the DPS. Elevated water temperatures impact rearing and juvenile migration PCEs in all river basins and estuaries. Rearing and spawning PCEs have also been affected throughout the DPS by management or reduction in water quantity. The spawning PCE has also been affected by the combination of erosive geology and land management activities that have resulted in an excessive amount of fines in the spawning gravel of most rivers.

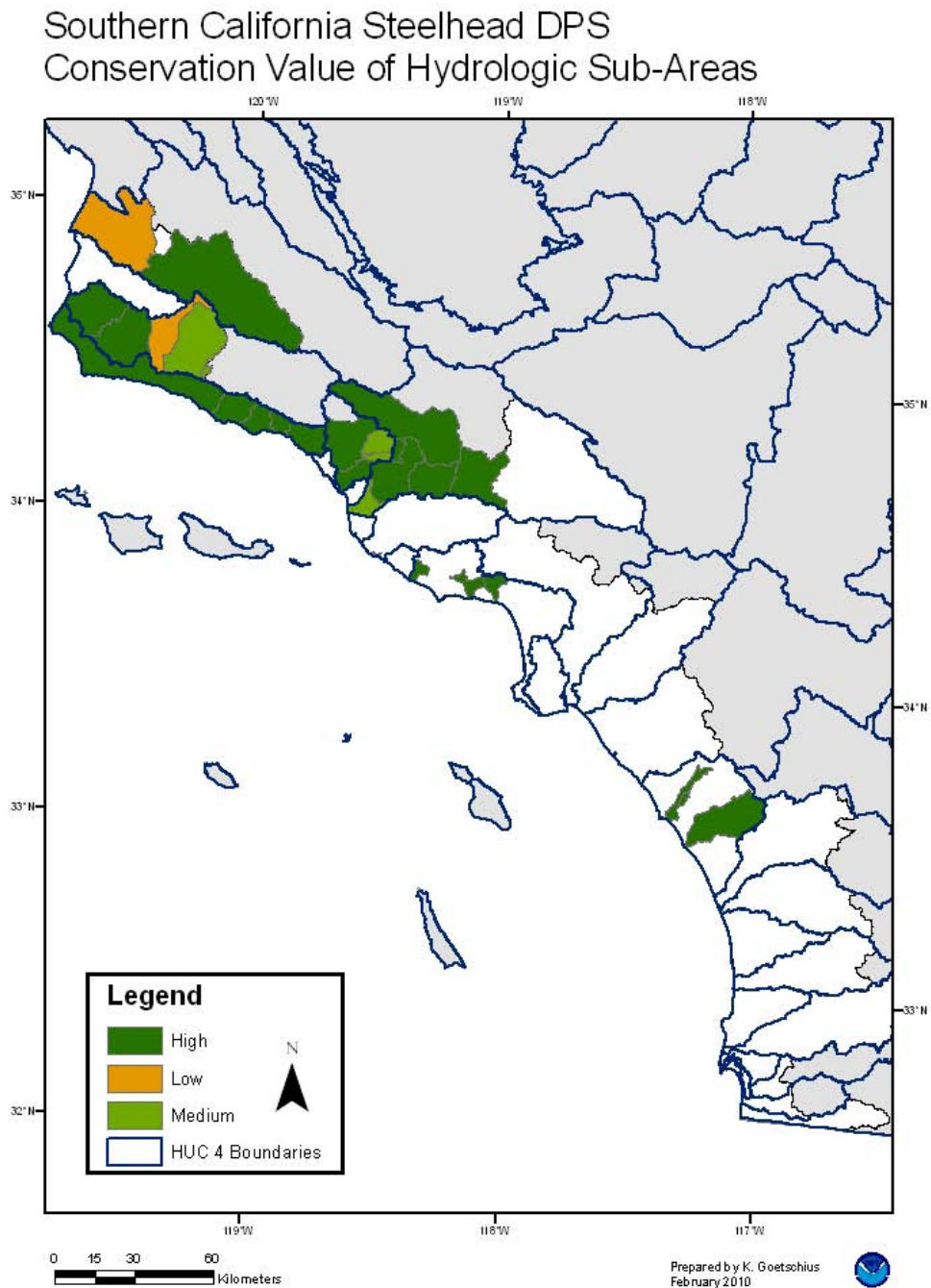


Figure 50. Southern California Steelhead Conservation Values per Sub-area.

Environmental Baseline

By regulation, environmental baselines for Opinions include the past and present impacts of all state, federal or private actions and other human activities in the action area, the anticipated impacts of all proposed federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR §402.02). The environmental baseline for this Opinion includes a general description of the natural and anthropogenic factors influencing the current status of listed Pacific salmonids and the environment within the action area.

Our summary of the environmental baseline complements the information provided in the *Status of Listed Resources* section of this Opinion, and provides the background necessary to understand information presented in the *Effects of the Proposed Action*, and *Cumulative Effects* sections of this Opinion. We then evaluate the consequences of these activities in combination with the environmental baseline to determine the likelihood of jeopardy or adverse modification of designated critical habitat.

The proposed action under consultation is focused geographically on the aquatic ecosystems in the states of California, Idaho, Oregon, and Washington. Accordingly, the environmental baseline for this consultation focuses on the general status and trends of the aquatic ecosystems in these four states and the consequences of that status for listed resources under NMFS' jurisdiction. We describe the principal natural phenomena affecting all listed Pacific salmonids under NMFS jurisdiction in the action area.

We further describe anthropogenic factors through the predominant land and water uses within a region, as land use patterns vary by region. Background information on pesticides in the aquatic environment is also provided. This context illustrates how the physical and chemical health of regional waters and the impact of human activities have contributed to the current status of listed resources in the action area.

Natural Mortality Factors

Available data indicate high natural mortality rates for salmonids, especially in the open ocean/marine environment. According to Bradford (1997), salmonid mortality rates range from 90 to 99%, depending on the species, the size at ocean entry, and the length of time spent in the ocean. Predation, inter- and intraspecific competition, food availability, smolt quality and health, and physical ocean conditions likely influence the survival of salmon in the marine environment (Bradford, Taylor et al. 1997; Brodeur, Fisher et al. 2004). In freshwater rearing habitats, the natural mortality rate averages about 70% for all salmonid species (Bradford, Taylor et al. 1997). Past studies in the Pacific Northwest suggest that the average freshwater survival rate (from egg to smolt) is 2 to 3% throughout the region (Marshall and Britton 1990; Bradford, Taylor et al. 1997). A number of suspected causes contributing to natural mortality include parasites and/or disease, predation, water temperature, low water flow, wildland fire, and oceanographic features and climatic variability.

Parasites and/or Disease

Most young fish are highly susceptible to disease during the first two months of life. The cumulative mortality in young animals can reach 90 to 95%. Although fish disease organisms occur naturally in the water, native fish have co-evolved with them. Fish can carry these diseases at less than lethal levels (Kier Associates 1991; Walker and Foott 1993; Foott, Harmon et al. 2003). However, disease outbreaks may occur when water quality is diminished and fish are stressed from crowding and diminished flows (Spence, Lomnický et al. 1996; Guillen 2003). Young coho salmon or other salmonid species may become stressed and lose their resistance in higher temperatures (Spence, Lomnický et al. 1996). Consequently, diseased fish become more susceptible to predation and are less able to perform essential functions, such as feeding, swimming, and defending territories (McCullough 1999). Examples of parasites and disease for salmonids include whirling disease, infectious hematopoietic necrosis (IHN), sea-lice (*Lepeophtheirus salmonis*), *Henneguya salminicola*, *Ichthyophthirius multifiliis* or Ich, and Columnaris

(*Flavobacterium columnare*).

Whirling disease is a parasitic infection caused by the microscopic parasite *Myxobolus cerebrali*. Infected fish continually swim in circular motions and eventually expire from exhaustion. The disease occurs in the wild and in hatcheries and results in losses to fry and fingerling salmonids, especially rainbow trout. The disease is transmitted by infected fish and fish parts and birds.

IHN is a viral disease in many wild and farmed salmonid stocks in the Pacific Northwest. This disease affects rainbow/steelhead trout, cutthroat trout (*Salmo clarki*), brown trout (*Salmo trutta*), Atlantic salmon (*Salmo salar*), and Pacific salmon including Chinook, sockeye, chum, and coho salmon. The virus is triggered by low water temperatures and is shed in the feces, urine, sexual fluids, and external mucus of salmonids. Transmission is mainly from fish to fish, primarily by direct contact and through the water.

Sea lice also cause deadly infestations of wild and farm-grown salmon. *Henneguya salminicola*, a protozoan parasite, is commonly found in the flesh of salmonids. The fish responds by walling off the parasitic infection into a number of cysts that contain milky fluid. This fluid is an accumulation of a large number of parasites. Fish with the longest freshwater residence time as juveniles have the most noticeable infection. The order of prevalence for infection is coho followed by sockeye, Chinook, chum, and pink salmon.

Additionally, ich (a protozoan) and Columnaris (a bacterium) are two common fish diseases that were implicated in the massive kill of adult salmon in the Lower Klamath River in September 2002 (CDFG 2003; Guillen 2003).

Predation

Salmonids are exposed to high rates of natural predation, during freshwater rearing and migration stages, as well as during ocean migration. Salmon along the U.S. west coast are prey for marine mammals, birds, sharks, and other fishes. Concentrations of juvenile

salmon in the coastal zone experience high rates of predation. In the Pacific Northwest, the increasing size of tern, seal, and sea lion populations may have reduced the survival of some salmon ESUs/DPSs.

Marine Mammal Predation

Marine mammals are known to attack and eat salmonids. Harbor seals (*Phoca vitulina*), California sea lions (*Zalophus californianus*), and killer whales (*Orcinus orca*) prey on juvenile or adult salmon. Killer whales have a strong preference for Chinook salmon (up to 78% of identified prey) during late spring to fall (Hard, Jones et al. 1992; Hanson, Baird et al. 2005; Ford and Ellis 2006). Generally, harbor seals do not feed on salmonids as frequently as California sea lions (Percy 1997). California sea lions from the Ballard Locks in Seattle, Washington have been estimated to consume about 40% of the steelhead runs since 1985/1986 (Gustafson, Wainwright et al. 1997). In the Columbia River, salmonids may contribute substantially to sea lion diet at specific times and locations (Percy 1997). Spring Chinook salmon and steelhead are subject to pinniped predation when they return to the estuary as adults (NMFS 2006). Adult Chinook salmon in the Columbia River immediately downstream of Bonneville Dam have also experienced increased predation by California sea lions. In recent years, sea lion predation of adult Lower Columbia River winter steelhead in the Bonneville tailrace has increased. This prompted ongoing actions to reduce predation effects. They include the exclusion, hazing, and in some cases, lethal take of marine mammals near Bonneville Dam (NMFS 2008).

NOAA Fisheries has granted permits to the states of Idaho, Oregon, and Washington for the lethal removal of individual California sea lions that prey on adult spring-run Chinook salmon in the tailrace of Bonneville Dam under section 120 of the Marine Mammal Protection Act (NMFS 2006). This action may increase the survival of adult Chinook salmon and steelhead. The Humane Society of the U.S. unsuccessfully challenged NOAA Fisheries' issuance of these permits in the U.S. District Court of Oregon (Humane Society of the U.S. v. Gutierrez, 625 F. Supp. 2d 1052 (D. Or. 2008)).

The Ninth Circuit denied the Humane Society's request for a stay of the action pending appeal (Humane Society of the U.S. v. Gutierrez, 5558 F.3d 896 (9th Cir. 2009)). The appeal is currently pending in the Ninth Circuit Court of Appeals.

Avian Predation

Large numbers of fry and juveniles are eaten by birds such as mergansers (*Mergus* spp.), common murre (*Uria aalage*), gulls (*Larus* spp.), and belted kingfishers (*Megaceryle alcyon*). Avian predators of adult salmonids include bald eagles (*Haliaeetus leucocephalus*) and osprey (*Pandion haliaetus*) (Pearcy 1997). Caspian terns (*Sterna caspia*) and cormorants (*Phalacrocorax* spp.) also take significant numbers of juvenile or adult salmon. Stream-type juveniles, especially yearling smolts from spring-run populations, are vulnerable to bird predation in the estuary. This vulnerability is due to salmonid use of the deeper, less turbid water over the channel, which is located near habitat preferred by piscivorous birds (Binelli, Ricciardi et al. 2005). Recent research shows that subyearlings from the LCR Chinook salmon ESU are also subject to tern predation. This may be due to the long estuarine residence time of the LCR Chinook salmon (Ryan, Carper et al. 2006). Caspian terns and cormorants may be responsible for the mortality of up to 6% of the outmigrating stream-type juveniles in the Columbia River basin (Roby, Collis et al. 2006; Collis 2007).

Antolos *et al.* (2005) quantified predation on juvenile salmonids by Caspian terns nesting on Crescent Island in the mid-Columbia reach. Between 1,000 and 1,300 adult terns were associated with the colony during 2000 and 2001, respectively. These birds consumed about 465,000 juvenile salmonids in the first and approximately 679,000 salmonids in the second year. However, caspian tern predation in the estuary was reduced from a total of 13,790,000 smolts to 8,201,000 smolts after relocation of the colony from Rice to East Sand Island in 1999. Based on PIT-tag recoveries at the colony, these were primarily steelhead for Upper Columbia River stocks. Less than 0.1% of the inriver migrating yearling Chinook salmon from the Snake River and less than 1% of the yearling Chinook salmon from the Upper Columbia were consumed. PIT-tagged coho

smolts (originating above Bonneville Dam) were second only to steelhead in predation rates at the East Sand Island colony in 2007 (Roby, Colis et al. 2008). There are few quantitative data on avian predation rates on Snake River sockeye salmon. Based on the above, avian predators are assumed to have a minimal effect on the long-term survival of Pacific salmon (NMFS 2008).

Fish Predation

Pikeminnows (*Ptychocheilus oregonensis*) are significant predators of yearling juvenile migrants (Friesen and Ward 1999). Chinook salmon were 29% of the prey of northern pikeminnows in lower Columbia reservoirs, 49% in the lower Snake River, and 64% downstream of Bonneville Dam. Sockeye smolts comprise a very small fraction of the overall number of migrating smolts (Ferguson 2006) in any given year. The significance of fish predation on juvenile chum is unknown. There is little direct evidence that piscivorous fish in the Columbia River consume juvenile sockeye salmon. The ongoing Northern Pikeminnow Management Program (NPMP) has reduced predation-related juvenile salmonid mortality since 1990. Benefits of recent northern pikeminnow management activities to chum salmon are unknown. However, it may be comparable to those for other salmon species with a sub-yearling juvenile life history (Friesen and Ward 1999).

The primary fish predators in estuaries are probably adult salmonids or juvenile salmonids which emigrate at older and larger sizes than others. They include cutthroat trout (*O. clarki*) or steelhead smolts preying on chum or pink salmon smolts. Outside estuaries, many large non-salmonid populations reside just offshore and may consume large numbers of smolts. These fishes include Pacific hake (*Merluccius productus*), Pacific mackerel (*Scomber japonicus*), lingcod (*Ophiodon elongates*), spiny dogfish (*Squalus acanthias*), various rock fish, and lamprey (Beamish, Thomson et al. 1992; Pearcy 1992; Beamish and Neville 1995).

Wildland Fire

Wildland fires that are allowed to burn naturally in riparian or upland areas may benefit or harm aquatic species, depending on the degree of departure from natural fire regimes. Although most fires are small in size, large size fires increase the chances of adverse effects on aquatic species. Large fires that burn near the shores of streams and rivers can have biologically significant short-term effects. They include increased water temperatures, ash, nutrients, pH, sediment, toxic chemicals, and large woody debris (Buchwalter, Sandahl et al. 2004; Rinne 2004). Nevertheless, fire is also one of the dominant habitat-forming processes in mountain streams (Bisson, Rieman et al. 2003). As a result, many large fires burning near streams can result in fish kills with the survivors actively moving downstream to avoid poor water quality conditions (Greswell 1999; Rinne 2004). The patchy, mosaic pattern burned by fires provides a refuge for those fish and invertebrates that leave a burning area or simply spares some fish that were in a different location at the time of the fire (USFS 2000). Small fires or fires that burn entirely in upland areas also cause ash to enter rivers and increase smoke in the atmosphere, contributing to ammonia concentrations in rivers as the smoke adsorbs into the water (Greswell 1999).

The presence of ash also has indirect effects on aquatic species depending on the amount of ash entry into the water. All ESA-listed salmonids rely on macroinvertebrates as a food source for at least a portion of their life histories. When small amounts of ash enter the water, there are usually no noticeable changes to the macroinvertebrate community or the water quality (Bowman and Minshall 2000). When significant amounts of ash are deposited into rivers, the macroinvertebrate community density and composition may be moderately to drastically reduced for a full year with long-term effects lasting 10 years or more (Buchwalter, Jenkins et al. 2003), (Minshall, Royer et al. 2001; Buchwalter, Sandahl et al. 2004). Larger fires can also indirectly affect fish by altering water quality. Ash and smoke contribute to elevated ammonium, nitrate, phosphorous, potassium, and pH, which can remain elevated for up to four months after forest fires (Buchwalter, Jenkins et al. 2003).

Oceanographic Features, Climatic Variability and Climate Change

Oceanographic features of the action area may influence prey availability and habitat for Pacific salmonids. These features comprise climate regimes which may suffer regime shifts due to climate changes or other unknown influences. The action area includes important spawning and rearing grounds and physical and biological features essential to the conservation of listed Pacific salmonids - *i.e.*, water quality, prey, and passage conditions. These Pacific oceanographic conditions, climatic variability, and climate change may affect salmonids in the action area.

There is evidence that Pacific salmon abundance may have fluctuated for centuries as a consequence of dynamic oceanographic conditions (Beamish and Bouillon 1993; Finney, Gregory-Eaves et al. 2002; Beamish, Sweeting et al. 2009). Sediment cores reconstructed for 2,200-year records have shown that Northeastern Pacific fish stocks have historically been regulated by these climate regimes (Finney, Gregory-Eaves et al. 2002). The long-term pattern of the Aleutian Low pressure system has corresponded to the trends in salmon catch, to copepod production, and to other climate indices, indicating that climate and the marine environment may play an important role in salmon production. Pacific salmon abundance and corresponding worldwide catches tend to be large during naturally-occurring periods of strong Aleutian low pressure causing stormier winters and upwelling, positive Pacific Decadal Oscillation (PDO), and an above average Pacific circulation index (Beamish, Sweeting et al. 2009). A trend of an increasing Aleutian Low pressure indicates high pink and chum salmon production and low production of coho and Chinook salmon (Beamish, Sweeting et al. 2009). The abundance and distribution of salmon and zooplankton also relate to shifts in North Pacific atmosphere and ocean climate (Francis and Hare 1994).

Over the past century, regime shifts have occurred as a result of the North Pacific's natural climate regime. Reversals in the prevailing polarity of the PDO occurred around 1925, 1947, 1977, and 1989 (Mantua, Hare et al. 1997; Hare and Mantua. 2000). The reversals in 1947 and 1977 correspond to dramatic shifts in salmon production regimes in

the North Pacific Ocean (Mantua, Hare et al. 1997). During the pre-1977 climate regime, the productivity of salmon populations from the Snake River exceeded expectations (residuals were positive) when values of the PDO were negative (Levin 2003). During the post-1977 regime when ocean productivity was generally lower (residuals were negative), the PDO was negative (Levin 2003).

A smaller, less pervasive regime shift occurred in 1989 (Hare and Mantua. 2000). Beamish *et al.*(2000) analyzed this shift and found a decrease in marine survival of coho salmon in Puget Sound and off the coast of California to Washington. Trends in coho salmon survival were linked over the southern area of their distribution in the Northeast Pacific to a common climatic event. The Aleutian Low Pressure Index and the April flows from the Fraser River also changed abruptly about this time (Beamish, Noakes et al. 2000).

The Intergovernmental Panel on Climate Change (IPCC) has high confidence that some hydrological systems have been affected through increased runoff and earlier spring peak discharge in glacier- and snow-fed rivers and through effects on thermal structure and water quality of warming rivers and lakes (IPCC 2007). Oceanographic models project a weakening of the thermohaline circulation resulting in a reduction of heat transport into high latitudes of Europe, an increase in the mass of the Antarctic ice sheet, and a decrease in the Greenland ice sheet (IPCC 2001). These changes, coupled with increased acidification of ocean waters, are expected to have substantial effects on marine and hydrological productivity and food webs, including populations of salmon and other salmonid prey (Hard, Jones et al. 1992).

Carbon dioxide emissions are also predicted to have major environmental impacts along the west coast of North America during the 21st century and beyond (IPCC 2001; Climate Impacts Group (CIG) 2004). Eleven of the past 12 years (1995 - 2006) rank among the 12 warmest years in the instrumental record of global surface temperature since 1850 (IPCC 2007). The IPCC predicts that, for the next two decades, a warming of about

0.2°C per decade will occur for a range of predicted carbon dioxide emissions scenarios (IPCC 2007). This warming trend continues in both water and air. Global average sea level has risen since 1961 at an average rate of 1.8 mm/year and since 1993 at 3.1 mm/year, with contributions from thermal expansion, melting glaciers and ice caps, and the polar ice sheets (IPCC 2007).

Poor environmental conditions for salmon survival and growth may be more prevalent with projected warming increases. Increasing climate temperatures can influence smolt development which is limited by time and temperature (McCormick, Lerner et al. 2009). Food availability and water temperature may affect proper maturation and smoltification and feeding behavior (Mangel 1994). Climate change may also have profound effects on seawater entry and marine performance of anadromous fish, including increased salinity intrusion in estuaries due to higher sea levels, as well as a projected decrease of seawater pH (Orr, Fabry et al. 2005). There is evidence that Chinook salmon survival in the Pacific during climate anomalies and El Nino events changes as a result of a shift from predation- to competition-based mortality in response to declines in predator and prey abundances and increases in pink salmon abundance (Ruggerone and Goetz 2004). If climate change leads to an overall decrease in the availability of food, then returning fish will likely be smaller (Mangel 1994). Finally, future climatic warming could lead to alterations of river temperature regimes, which could further reduce available fish habitat (Yates, Galbraith et al. 2008).

Although the impacts of global climate change are less clear in the ocean environment, early modeling efforts suggest that increased temperatures will likely increase ocean stratification. This stratification coincides with relatively poor ocean habitat for most Pacific Northwest salmon populations (IPCC 2001; Climate Impacts Group (CIG) 2004).

We expect changing weather and oceanographic conditions may affect prey availability, temperature and water flow in habitat conditions, and growth for all 28 ESUs/DPSs. Consequently, we expect the long-term survival and reproductive success for listed

salmonids to be greatly affected by global climate change.

Anthropogenic Mortality Factors

In this section we address anthropogenic threats in the geographic regions across the action area. Prior to discussion of each geographic region, three major issues are highlighted: pesticide contamination, elevated water temperature, and loss of habitat/habitat connectivity. These three factors are the most relevant to the current analysis. We provide information on pesticide detections in the aquatic environment and highlight their background levels from past and ongoing anthropogenic activities. This information is pertinent to EPA's proposed registration of 2,4-D, triclopyr, diuron, linuron, captan, and chlorothalonil in the U.S. and its territories. As water temperature plays such a strong role in salmonid distribution, we also provide a general discussion of anthropogenic temperature impacts. Next, we discuss the health of riparian systems and floodplain connectivity, as this habitat is vital to salmonid survival. Finally, we provide a brief overview of the results of section 7 consultations relevant to this analysis.

Baseline Pesticide Detections in Aquatic Environments

In the environmental baseline, we address pesticide detections reported as part of the U.S. Geological Survey (USGS) National Water-Quality Assessment Program's (NAWQA) national assessment (Gilliom, Barbash et al. 2006). We chose this approach because the NAWQA studies present the same level of analysis for each area. Further, given the lack of uniform reporting standards, we are unable to present a comprehensive basin-specific analysis of detections from other sources.

According to Gilliom *et al.* (2006), the distributions of the most prevalent pesticides in streams and ground water correlate with land use patterns and associated present or past pesticide use. When pesticides are released into the environment, they frequently end up as contaminants in aquatic environments. Depending on their physical properties some are rapidly transformed via chemical, photochemical, and biologically mediated reactions

into other compounds, known as degradates. These degradates may become as prevalent as the parent pesticides depending on their rate of formation and their relative persistence.

In the *Exposure* section of the *Effects of the Proposed Action* we present a more comprehensive discussion of available monitoring data from the NAWQA program, state databases maintained by California and Washington, and other targeted monitoring studies.

National Water-Quality Assessment Program

From 1992 - 2001, the USGS sampled water from 186 stream sites within 51 study units; bed-sediment samples from 1,052 stream sites, and fish from 700 stream sites across the continental U.S. Concentrations of pesticides were detected in streams and groundwater within most areas sampled with substantial agricultural or urban land uses. NAWQA results further detected at least one pesticide or degradate more than 90% of the time in water, in more than 80% in fish samples, and greater than 50% of bed-sediment samples from streams in watersheds with agricultural, urban, and mixed land use (Gilliom, Barbash et al. 2006).

Twenty-four pesticides and one degradate were each detected in over 10% of streams in agricultural, urban, or mixed land use areas. These 25 compounds include 11 agriculture-use herbicides and the atrazine degradate deethylatrazine; 7 urban-use herbicides; and 6 insecticides used in both agricultural and urban areas. Two of the herbicides used primarily in urban areas are 2,4-D and diuron. Both herbicides were detected roughly 12% of the time in agricultural streams and between 20% and 25% of the time in urban streams. Five of the insecticides were carbaryl, carbofuran, chlorpyrifos, diazinon, and malathion. NMFS assessed the effects of these five insecticides on listed salmonids in its 2008 and 2009 Opinions (NMFS 2008);(NMFS 2009).

Another dimension of pesticides and their degradates in the aquatic environment is their simultaneous occurrence as mixtures (Gilliom, Barbash et al. 2006). Mixtures result

from the use of different pesticides for multiple purposes within a watershed or groundwater recharge area. Pesticides generally occur more often in natural waterbodies as mixtures than as individual compounds. Mixtures of pesticides were detected more often in streams than in ground water and at relatively similar frequencies in streams draining areas of agricultural, urban, and mixed land use. More than 90% of the time, water from streams in these developed land use settings had detections of two or more pesticides or degradates. About 70% and 20% of the time, streams had five or more and ten or more pesticides or degradates, respectively (Gilliom, Barbash et al. 2006). Fish exposed to multiple pesticides at once may also experience additive and synergistic effects. If the effects on a biological endpoint from concurrent exposure to multiple pesticides can be predicted by adding the potency of the pesticides involved, the effects are said to be additive. If, however, the response to a mixture leads to a greater than expected effect on the endpoint, and the pesticides within the mixture enhance the toxicity of one another, the effects are characterized as synergistic. These effects are of particular concern when the pesticides share a mode of action. NAWQA analysis of all detections indicates that more than 6,000 unique mixtures of 5 pesticides were detected in agricultural streams (Gilliom, Barbash et al. 2006). The number of unique mixtures varied with land use.

More than half of all agricultural streams sampled and more than three-quarters of all urban streams had concentrations of pesticides in water that exceeded one or more benchmarks for aquatic life. Aquatic life criteria are EPA water-quality guidelines for protection of aquatic life. Exceedance of an aquatic life benchmark level indicates a strong probability that aquatic species are being adversely affected. However, aquatic species may also be affected at levels below criteria. In agricultural streams, most concentrations that exceeded an aquatic life benchmark involved chlorpyrifos (21%), azinphos methyl (19%), atrazine (18%), *p,p'*-DDE (16%), and alachlor (15%) (Gilliom, Barbash et al. 2006). Finally, organochlorine pesticides that were discontinued 15 to 30 years ago still exceeded benchmarks for aquatic life and fish-eating wildlife in bed sediment or fish tissue samples from many streams.

National Pollutant Discharge Elimination System

Pollution originating from a discrete location such as a pipe discharge or wastewater treatment outfall is known as a point source. Point sources of pollution require a National Pollutant Discharge Elimination System (NPDES) permit. These permits are issued for aquaculture, concentrated animal feeding operations, industrial wastewater treatment plants, biosolids (sewer/sludge), pre-treatment and stormwater overflows. The EPA administers the NPDES permit program and states certify that NPDES permit holders comply with state water quality standards. Nonpoint source discharges do not originate from discrete points; thus, nonpoint sources are difficult to identify, quantify, and are not regulated. Examples of nonpoint source pollution include, but are not limited to, urban runoff from impervious surfaces, areas of fertilizer and pesticide application, sedimentation, and manure.

According to EPA's database of NPDES permits, about 243 NPDES permits are co-located with listed Pacific salmonids in California. Collectively, the total number of EPA-recorded NPDES permits in Idaho, Oregon, and Washington, that are co-located with listed Pacific salmonids is 1,978. See ESU/DPS maps for NPDES permits co-located within listed salmonid ESUs/DPSs within the states of California, Idaho, Oregon, and Washington in the *Status of Listed Resources* chapter.

On November 27, 2006, EPA issued a final rule which exempted pesticides from the NPDES permit process, provided that application was approved under FIFRA. The NPDES permits, then, do not include any point source application of pesticides to waterways in accordance with FIFRA labels. On January 7, 2009, the Sixth Circuit Court of Appeals vacated this rule (National Cotton Council v. EPA, 553 F.3d 927 (6th Cir. 2009)). The result of the vacature, according to the Sixth Circuit, is that "discharges of pesticide pollutants are subject to the NPDES permitting program" under the CWA. In response, EPA has developed a Pesticide General Permit through the NPDES permitting program to regulate such discharges. The permit is currently undergoing Section 7

consultation.

Baseline Water Temperature - Clean Water Act

Elevated temperature is considered a pollutant in most states with approved Water Quality Standards under the federal Clean Water Act (CWA) of 1972. Under the authority of the CWA, states periodically prepare a list of all surface waters in the state for which beneficial uses - such as drinking, recreation, aquatic habitat, and industrial use – are impaired by pollutants. This process is in accordance with section 303(d) of the CWA. Estuaries, lakes, and streams listed under 303(d) are those that are considered impaired or threatened by pollution. They are water quality limited, do not meet state surface water quality standards, and are not expected to improve within the next two years.

Each state has separate and different 303(d) listing criteria and processes. Generally a water body is listed separately for each standard it exceeds, so it may appear on the list more than once. If a water body is not on the 303(d) list, it is not necessarily contaminant-free; rather it may not have been tested. Therefore, the 303(d) list is a minimum list for the each state regarding polluted water bodies by parameter.

After states develop their lists of impaired waters, they are required to prioritize and submit their lists to EPA for review and approval. Each state establishes a priority ranking for such waters, considering the severity of the pollution and the uses to be made of such waters. States are expected to identify high priority waters targeted for Total Maximum Daily Load (TMDL) development within two years of the 303(d) listing process.

Temperature is significant for the health of aquatic life. Water temperatures affect the distribution, health, and survival of native cold-blooded salmonids in the Pacific Northwest. These fish will experience adverse health effects when exposed to

temperatures outside their optimal range. For listed Pacific salmonids, water temperature tolerance varies between species and life stages. Optimal temperatures for rearing salmonids range from 10°C to 16°C. In general, the increased exposure to stressful water temperatures and the reduction of suitable habitat caused by drought conditions reduce the abundance of salmon. Warm temperatures can reduce fecundity, reduce egg survival, retard growth of fry and smolts, reduce rearing densities, increase susceptibility to disease, decrease the ability of young salmon and trout to compete with other species for food, and to avoid predation (Spence, Lomnický et al. 1996; McCullough 1999). Migrating adult salmonids and upstream migration can be delayed by excessively warm stream temperatures. Excessive stream temperatures may also negatively affect incubating and rearing salmonids (Gregory and Bisson 1997).

Sublethal temperatures (above 24°C) could be detrimental to salmon by increasing susceptibility to disease (Colgrove and Wood 1966) or elevating metabolic demand (Brett 1995). Substantial research demonstrates that many fish diseases become more virulent at temperatures over 15.6°C (McCullough 1999). Due to the sensitivity of salmonids to temperature, states have established lower temperature thresholds for salmonid habitat as part of their water quality standards. A water body is listed for temperature on the 303(d) list if the 7-day average of the daily maximum temperatures

Table 52. Washington State water temperature thresholds for salmonid habitat. These temperatures are representative of limits set by California, Idaho, and Oregon (WSDE 2006).

Category	Highest 7-DADMax
Salmon and Trout Spawning	13°C (55.4°F)
Core Summer Salmonid Habitat	16°C (60.8°F)
Salmonid Spawning, Rearing, and Migration	17.5°C (63.5°F)
Salmonid Rearing and Migration Only	17.5°C (63.5°F)

(7-DADMax) exceeds the temperature threshold (Table 52).

Water bodies that are not designated salmonid habitat are also listed if they have a one-day maximum over a given background temperature. Using publicly available Geographic Information System (GIS) layers, we determined the number of km on the 303(d) list for exceeding temperature thresholds within the boundaries of each ESU/DPS (Table 53). Because the 303(d) list is limited to the subset of rivers tested, the chart values should be regarded as lower-end estimates.

While some ESU/DPS ranges do not contain any 303(d) rivers listed for temperature, others show considerable overlap. These comparisons demonstrate the relative significance of elevated temperature among ESUs/DPSs. Increased water temperature may result in wastewater discharge, decreased water flow, minimal shading by riparian areas, and climatic variation.

Table 53. Number of kilometers of river, stream and estuaries included in state 303(d) lists due to temperature that are located within each salmonid ESU/DPS. Data was taken from the most recent GIS layers available from state water quality assessments reports.*

Species	ESU	California	Oregon	Washington	Idaho	Total
Chinook Salmon	California Coastal	39.3	—	—	—	39.3
	Central Valley Spring - Run	0.0	—	—	—	0.0
	Lower Columbia River	—	56.6	229.8	—	286.4
	Upper Columbia River Spring - Run	—	—	254.6	—	254.6
	Puget Sound	—	—	705.0	—	705.0
	Sacramento River Winter - Run	0.0	—	—	—	0.0
	Snake River Fall - Run	—	610.1	246.6	400.2	1,256.9
	Snake River Spring / Summer - Run	—	809.3	243.2	543.8	1,596.3
	Upper Willamette River	—	2,468.0	—	—	2,468.0
Chum Salmon	Columbia River	—	56.6	225.0	—	281.6
	Hood Canal Summer - Run	—	—	90.1	—	90.1
Coho Salmon	Central California Coast	39.3	—	—	—	39.3
	Lower Columbia River	—	291.9	233.5	—	525.4
	Southern Oregon and Northern California Coast	1,416.2	1,833.0	—	—	3,249.2
	Oregon Coast	—	3,715.8	—	—	3,715.8
Sockeye Salmon	Ozette Lake	—	—	4.8	—	4.8
	Snake River	—	—	—	0.0	0.0
Steelhead	Central California Coast	0.0	—	—	—	0.0
	California Central Valley	0.0	—	—	—	0.0
	Lower Columbia River	—	201.2	169.3	—	370.5
	Middle Columbia River	—	3,518.5	386.2	—	3,904.7
	Northern California	39.3	—	—	—	39.3
	Puget Sound	—	—	704.9	—	704.9
	Snake River	—	990.7	246.6	737.6	1,974.9
	South-Central California Coast	0.0	—	—	—	0.0
	Southern California	0.0	—	—	—	0.0
	Upper Columbia River	—	—	282.3	—	282.3
	Upper Willamette River	—	1,668.0	—	—	1,668.0

*CA 2006, Oregon 2004/2006, Washington 2004, and Idaho 1998. (California EPA TMDL Program 2007b, Oregon Department of Environmental Quality 2007, Washington State Department of Ecology 2005, Idaho Department of Environmental Quality 2001).

Baseline Habitat Condition

Riparian zones are the areas of land adjacent to rivers and streams. These systems serve as the interface between the aquatic and terrestrial environments. Riparian vegetation is

characterized by emergent aquatic plants and species that thrive on close proximity to water, such as willows. This vegetation maintains a healthy river system by reducing erosion, stabilizing main channels, and providing shade. Leaf litter that enters the river becomes an important source of nutrients for invertebrates (Bisson and Bilby 2001). Riparian zones are also the major source of large woody debris (LWD). When trees fall and enter the water, they become an important part of the ecosystem. The LWD alters the flow, creating the pools of slower moving water preferred by salmon (Bilby, Fransen et al. 2001). While not necessary for pool formation, LWD is associated with around 80% of pools in northern California, Washington, and the Idaho pan-handle (Bilby and Bisson 2001).

Bilby and Bisson (2001) discuss several studies that associate increased LWD with increased pools, and both pools and LWD with salmonid productivity. Their review also includes documented decreases in salmonid productivity following the removal of LWD. Other benefits of LWD include deeper pools, increased sediment retention, and channel stabilization.

Floodplains are relatively flat areas adjacent to larger streams and rivers. They allow for the lateral movement of the main channel and provide storage for floodwaters during periods of high flow. Water stored in the floodplain is later released during periods of low flow. This process ensures adequate flows for salmonids during the summer months, and reduces the possibility of high-energy flood events destroying salmonid redds (Smith 2005).

Periodic flooding of these areas creates habitat used by salmonids. Thus, floodplain areas vary in depth and widths and may be intermittent or seasonal. Storms also wash sediment and LWD into the main stem river, often resulting in blockages. These blockages may force the water to take an alternate path and result in the formation of side channels and sloughs (Benda, Miller et al. 2001). Side channels and sloughs are important spawning and rearing habitat for salmonids. The degree to which these off-channel habitats are

linked to the main channel via surface water connections is referred to as connectivity (PNERC 2002). As river height increases with heavier flows, more side channels form and connectivity increases. Juvenile salmonids migrate to and rear in these channels for a certain period of time before swimming out to the open sea.

Healthy riparian habitat and floodplain connectivity are vital for supporting a salmonid population. Chinook salmon and steelhead have life history strategies that rely on floodplains during their juvenile life stages. Chum salmon use adjacent floodplain areas for spawning. Soon after their emergence, chum salmon use the riverine system to rapidly reach the estuary where they mature, rear, and migrate to the ocean. Coho salmon use the floodplain landscape extensively for rearing. Estuarine floodplains can provide value to juveniles of all species once they reach the salt water interface.

Once floodplain areas have been disturbed, it can take decades for their recovery (Smith 2005). Consequently, most land use practices cause some degree of impairment. Development leads to construction of levees and dikes, which isolate the mainstem river from the floodplain. Agricultural development and grazing in riparian areas also significantly change the landscape. Riparian areas managed for logging, or logged in the past, are often impaired by a change in species composition. Most areas in the northwest were historically dominated by conifers. Logging results in recruitment of deciduous trees, decreasing the quality of LWD in the rivers. Deciduous trees have smaller diameters than conifers; they decompose faster and are more likely to be displaced (Smith 2005).

Without a properly functioning riparian zone, salmonids contend with a number of limiting factors. They face reductions in quantity and quality of both off-channel and pool habitats. Also, when seasonal flows are not moderated, both higher and lower flow conditions exist. Higher flows can displace fish and destroy redds, while lower flows cut off access to parts of their habitat. Finally, decreased vegetation limits the available shade and cover, exposing individuals to higher temperatures and increased predation.

Baseline Pesticide Consultations

NMFS has consulted with EPA on the registration of several pesticides. NMFS (NMFS 2008) determined that current use of chlorpyrifos, diazinon, and malathion is likely to jeopardize the continued existence of 27 listed salmonid ESUs/DPSs. NMFS (NMFS 2009) further determined that current use of carbaryl and carbofuran is likely to jeopardize the continued existence of 22 ESUs/DPSs; and the current use of methomyl is likely to jeopardize the continued existence of 18 ESUs/DPSs of listed salmonids. Most recently, NMFS published conclusions regarding the registration of 12 different a.i.s (NMFS 2010). NMFS concluded that pesticide products containing Azinphos methyl, disulfoton, fenamiphos, methamidophos, or methyl parathion are not likely to jeopardize the continuing existence of any listed Pacific Salmon or destroy or adversely modify designated critical habitat. NMFS also concluded that the effects of products containing bensulide, dimethoate, ethoprop, methidathion, naled, phorate, or phosmet are likely to jeopardize the continued existence of some listed Pacific Salmonids and to destroy or adversely modify designated habitat of some listed salmonids.

Additionally, some of the a.i.s discussed in this Opinion have been approved for use in Federal weed control programs. The US Forest Service and BLM both have invasive plant control programs that have gone through the ESA section 7 consultation process (NMFS 2007);(NMFS 2009). These programs include the use of formulations of 2,4-D, triclopyr, and/or diuron within the range of at least one salmon population. Each opinion concluded that the weed control program would not jeopardize the existence of listed salmonids or destroy adversely modify critical habitat because of the limited scope of each project and the management practices included in the action. Each project covered use of the herbicides in a few subbasins, often limiting the number of acres that can be treated. Treatment methods are specified as well, and are frequently limited to spot spraying, wicking, dipping, painting, and injecting. Generally, only 2,4-D products

labeled for aquatic use were permitted to be used within 15 feet of open water. In one case, triclopyr was approved for use within this range, but on only a select number of species and was limited to an area of 0.1 acre per occurrence and 5 acres total per year (NMFS 2009). Most proposals included a minimum distance from water and/or intermittent streams for each product or mixture of products. Among the BMPs included in the action were monitoring weather, using only specified adjuvants, diluting the formulation, marking riparian buffers before application begins, and buffers. As these practices are mandatory, NMFS had to consider them as part of the federal action. Further, in the Incidental Take Statements, NMFS specified compliance monitoring and other types of oversight to be sure these BMPs were being followed. In general, the extent of incidental take for these opinions is defined by the amount of treated land within a given distance of salmon bearing streams. These Opinions recognized the large degree of uncertainty inherent in the effects analysis due to a paucity of information regarding sublethal effects and toxicity of mixtures. As such, the conclusions depended on limited use of the compounds and BMPs to keep products out of the water. If adverse effects were to occur, NMFS Biologists concluded that they would be limited to a specific area to impact the ESU/DPS as a whole.

Geographic Regions

For a more fine scale analysis, we divided the action area into geographic regions: the Southwest Coast Region (California and the southern parts of the State of Oregon) and the Pacific Northwest Region (Idaho, Oregon, and Washington). The Pacific Northwest Region was further subdivided according to ecoregions or other natural features important to NMFS trust resources. Use of these geographic regions is consistent with previous NMFS consultations conducted at the national level (NMFS 2007). We summarize the principal anthropogenic factors occurring in the environment that influence the current status of listed species within each region. Table 54 provides a breakdown of these regions and includes the USGS subregions and accounting units for each region. It also provides a list of ESUs/DPSs found in each accounting unit, as

indicated by Federal Register listing notices.

Table 54. USGS Subregions and accounting units within the Northwest and Southwest Regions, along with ESUs/DPSs present within the area (Seaber, Kapinos et al. 1987).

Region	USGS Subregion	Accounting Unit	State	HUC no.	ESU/DPS
Pacific Northwest: Columbia River Basin	Upper Columbia River Basin	—	WA	170200	Upper Columbia Spring-run Chinook; Upper Columbia Steelhead; Middle Columbia Steelhead
	Yakima River Basin	—	WA	170300	Middle Columbia Steelhead
	Lower Snake River Basin	Lower Snake River Basin	ID, OR, WA	170601	Snake River Steelhead; Snake River Spring/Summer-run Chinook; Snake River Fall-run Chinook; Snake River Sockeye
		Salmon River Basin	ID	170602	Snake River Steelhead; Snake River Spring/Summer - Run Chinook; Snake River Fall - Run Chinook; Snake River Sockeye
		Clearwater River Basin	ID, WA	170603	Snake River Steelhead; Snake River Fall - Run Chinook
	Middle Columbia River Basin	Middle Columbia River Basin	OR, WA	170701	Middle Columbia Steelhead; Lower Columbia Chinook; Columbia Chum; Lower Columbia Coho
		John Day River Basin	OR	170702	Middle Columbia Steelhead
		Deschutes River Basin	OR	170703	Middle Columbia Steelhead
	Lower Columbia River Basin	—	OR, WA	170800	Lower Columbia Chinook; Columbia Chum; Lower Columbia Steelhead; Lower Columbia Coho

Region	USGS Subregion	Accounting Unit	State	HUC no.	ESU/DPS
	Willamette River Basin	—	OR	170900	Upper Willamette Chinook; Upper Willamette Steelhead; Lower Columbia Chinook; Lower Columbia Steelhead; Lower Columbia Coho
Pacific Northwest: Coastal Drainages	Oregon-Washington Coastal Basin	Washington Coastal	WA	171001	Ozette Lake Sockeye
		Northern Oregon Coastal	OR	171002	Oregon Coast Coho
		Southern Oregon Coastal	OR	171003	Oregon Coast Coho; Southern Oregon and Northern California Coast Coho
Pacific Northwest: Puget Sound	Puget Sound	—	WA	171100	Puget Sound Chinook; Hood Canal Summer - Run Chum; Puget Sound Steelhead
Southwest Coast	Klamath-Northern California Coastal	Northern California Coastal	CA	180101	Southern Oregon and Northern California Coast Coho; California Coastal Chinook; Northern California Steelhead; Central California Coast Steelhead; Central California Coast Coho
		Klamath River Basin	CA, OR	180102	Southern Oregon and Northern California Coast Coho
	Sacramento River Basin	Lower Sacramento River Basin	CA	180201	Central Valley Spring-run Chinook; California Central Valley Steelhead; Sacramento River Winter-run Chinook
	San Joaquin River Basin	—	CA	180400	California Central Valley Steelhead
	San Francisco Bay	—	CA	180500	Central California Coast Steelhead; Southern Oregon and Northern California Coast Coho; Central California Coast Coho; Sacramento River Winter-run Chinook

Region	USGS Subregion	Accounting Unit	State	HUC no.	ESU/DPS
	Central California Coastal	—	CA	180600	Central California Coast Steelhead; Southern Oregon and Northern California Coast Coho; South-Central California Coast Steelhead; Southern California Steelhead; Central California Coast Coho; Sacramento River Winter-run Chinook
	Southern California Coastal	Ventura-San Gabriel Coastal	CA	180701	Southern California Steelhead
		Laguna-San Diego Coastal	CA	180703	Southern California Steelhead

Southwest Coast Region

The basins in this section occur in the States of California and the southern parts of Oregon. Ten of the 28 species addressed in the Opinion occur in the Southwest Coast Region. They are the California Coastal Chinook (CC) salmon, Central Valley (CV) Spring-run Chinook salmon, Sacramento River winter-run Chinook salmon, Southern Oregon/Northern California Coast (SONCC) coho salmon, Central California Coast (CCC) coho salmon, Northern California (NC) steelhead, Central California Coast (CCC) steelhead, California Central Valley (CCV) steelhead, South-Central California Coast (S-CCC) steelhead, and Southern California (SC) steelhead (Table 54). Table 55 and Table 56 show land area in km² for each ESU/DPS located in the Southwest Coast Region.

Table 55. Area of land use categories within the range Chinook and Coho Salmon ESUs in km² where bolded numbers are totals for each category. Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (National Land Cover Data 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Land Cover			Chinook Salmon			Coho Salmon	
sub category	code		CA Coastal	Central Valley	Sacramento River	So. Oregon and No. CA	Central CA Coast
Water			128	346	346	208	157
Open Water	11		128	346	346	197	157
Perennial Snow/Ice	12		0	0	0	11	0
Developed Land			1,138	2,588	2,588	1,985	991
Open Space	21		826	1,150	1,150	1,384	629
Low Intensity	22		137	578	578	225	171
Medium Intensity	23		95	567	567	92	138
High Intensity	24		10	135	135	23	30
Barren Land	31		70	158	158	261	23
Undeveloped Land			19,079	15,169	15,169	43,314	9,185
Deciduous Forest	41		850	664	664	1,057	208
Evergreen Forest	42		10,700	3,761	3,761	28,080	4,752
Mixed Forest	43		1,554	479	479	2,426	922
Shrub/Scrub	52		3,801	3,203	3,203	8,864	1,620
Herbaceous	71		2,114	6,317	6,317	2,708	1,646
Woody Wetlands	90		42	191	191	130	25
Emergent Wetlands	95		18	553	553	50	13
Agriculture			395	5,878	5,878	1,189	239
Hay/Pasture	81		183	769	769	736	6
Cultivated Crops	82		212	5,110	5,110	454	233
TOTAL (inc. open water)			20,740	23,982	23,982	46,697	10,572
TOTAL (w/o open water)			20,612	23,636	23,636	46,499	10,415

Table 56. Area of Land Use Categories within the Range of Steelhead Trout DPSs (km²). Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (National Land Cover Data 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Land Cover			Steelhead				
			Northern CA	Central CA Coast	CA Central Valley	South-Central CA coast	Southern CA
Water			106	1,406	409	127	159
Open Water	11		106	1,406	409	127	159
Perennial Snow/Ice	12		0	0	0	0	0
Developed Land			757	3,677	3,252	1,759	7,327
Open Space	21		610	1,224	1,431	1,019	1,952
Low Intensity	22		50	876	693	247	1,787
Medium Intensity	23		32	1,223	744	168	2,726
High Intensity	24		3	327	181	23	767
Barren Land	31		63	26	202	303	95
Undeveloped Land			16,117	11,041	19,216	14,959	13,057
Deciduous Forest	41		763	179	751	1	1
Evergreen Forest	42		9,790	2,506	3,990	1,721	984
Mixed Forest	43		1,159	2,086	598	1,925	1,025
Shrub/Scrub	52		2,878	2,253	3,745	4,952	8,375
Herbaceous	71		1,478	3,588	9,435	6,194	2,539
Woody Wetlands	90		32	36	248	93	83
Emergent Wetlands	95		17	392	450	73	50
Agriculture			193	522	10,724	1,500	1,059
Hay/Pasture	81		179	36	1,671	203	179
Cultivated Crops	82		14	486	9,054	1,297	880
TOTAL (inc. open water)			17,173	16,645	33,601	18,345	21,602
TOTAL (w/o open water)			17,067	15,240	33,193	18,218	21,446

Select watersheds described herein characterize the past, present, and future human activities and their impacts on the area. The Southwest Coast region encompasses all Pacific Coast rivers south of Cape Blanco, Oregon through southern California. NMFS has identified the Cape Blanco area as an ESU biogeographic boundary for Chinook and coho salmon, and steelhead based on strong genetic, life history, ecological and habitat differences north and south of this landmark. Major rivers contained in this grouping of

watersheds are the Sacramento, San Joaquin, Salinas, Klamath, Russian, Santa Ana, and Santa Margarita Rivers (Table 57).

Table 57. Select rivers in the southwest coast region (Carter and Resh 2005).

Watershed	Approx Length (mi)	Basin Size (mi ²)	Physiographic Provinces*	Mean Annual Precipitation (in)	Mean Discharge (cfs)	No. Fish Species (native)	No. Endangered Species
Rogue River	211	5,154	CS, PB	38	10,065	23 (14)	11
Klamath River	287	15,679	PB, B/R, CS	33	17,693	48 (30)	41
Eel River	200	3,651	PB	52	7,416	25 (15)	12
Russian River	110	1,439	PB	41	2,331	41 (20)	43
Sacramento River	400	27,850	PB, CS, B/R	35	23,202	69 (29)	>50 T & E spp.
San Joaquin River	348	83,409	PB, CS	49	4,662	63	>50 T & E spp.
Salinas River	179	4,241	PB	14	448	36 (16)	42 T & E spp.
Santa Ana River	110	2,438	PB	13	60	45 (9)	54
Santa Margarita River	27	1,896	LC, PB	49.5	42	17 (6)	52

* Physiographic Provinces: PB = Pacific Border, CS = Cascades-Sierra Nevada Range, B/R = Basin & Range.

Land Use

Figure 51 displays major land-use categories in California. Within the Southwest Coast Region, forest and vacant land are the dominant land uses. Grass, shrubland, and urban uses are the dominant land uses in the southern basins (Table 58). Overall, the most developed watersheds are the Santa Ana, Russian, and Santa Margarita rivers. The Santa Ana watershed encompasses portions of San Bernardino, Los Angeles, Riverside, and Orange counties. About 50% of the coastal subbasin in the Santa Ana watershed is dominated by urban land uses and the population density is about 1,500 people per square mile. When steep and undevelopable lands are excluded from this area, the population density in the watershed is about 3,000 people per square mile. However, the most densely populated portion of the basin is near the City of Santa Ana. Here, the population density reaches 20,000 people per square mile (Burton, Izbicki et al. 1998; Belitz, Hamlin et al. 2004). The basin is home to nearly 5 million people and this population is projected to increase two-fold in the next 50 years (Burton, Izbicki et al. 1998; Belitz, Hamlin et al. 2004).

Table 58. Land uses and population density in several southwest coast watersheds (Carter and Resh 2005).

Watershed	Land Use Categories (Percent)				Density (people/mi ²)
	Agriculture	Forest	Urban	Other	
Rogue River	6	83	<1	9 grass & shrub	32
Klamath River	6	66	<1	24 grass, shrub, wetland	5
Eel River	2	65	<1	31 grass & shrub	9
Russian River	14	50	3	31 (23 grassland)	162
Sacramento River	15	49	2	30 grass & shrub	61
San Joaquin River	30	27	2	36 grass & shrub	76
Salinas River	13	17	1	65 (49 grassland)	26
Santa Ana River	11	57	32	---	865
Santa Margarita River	12	11	3	71 grass & shrub	135

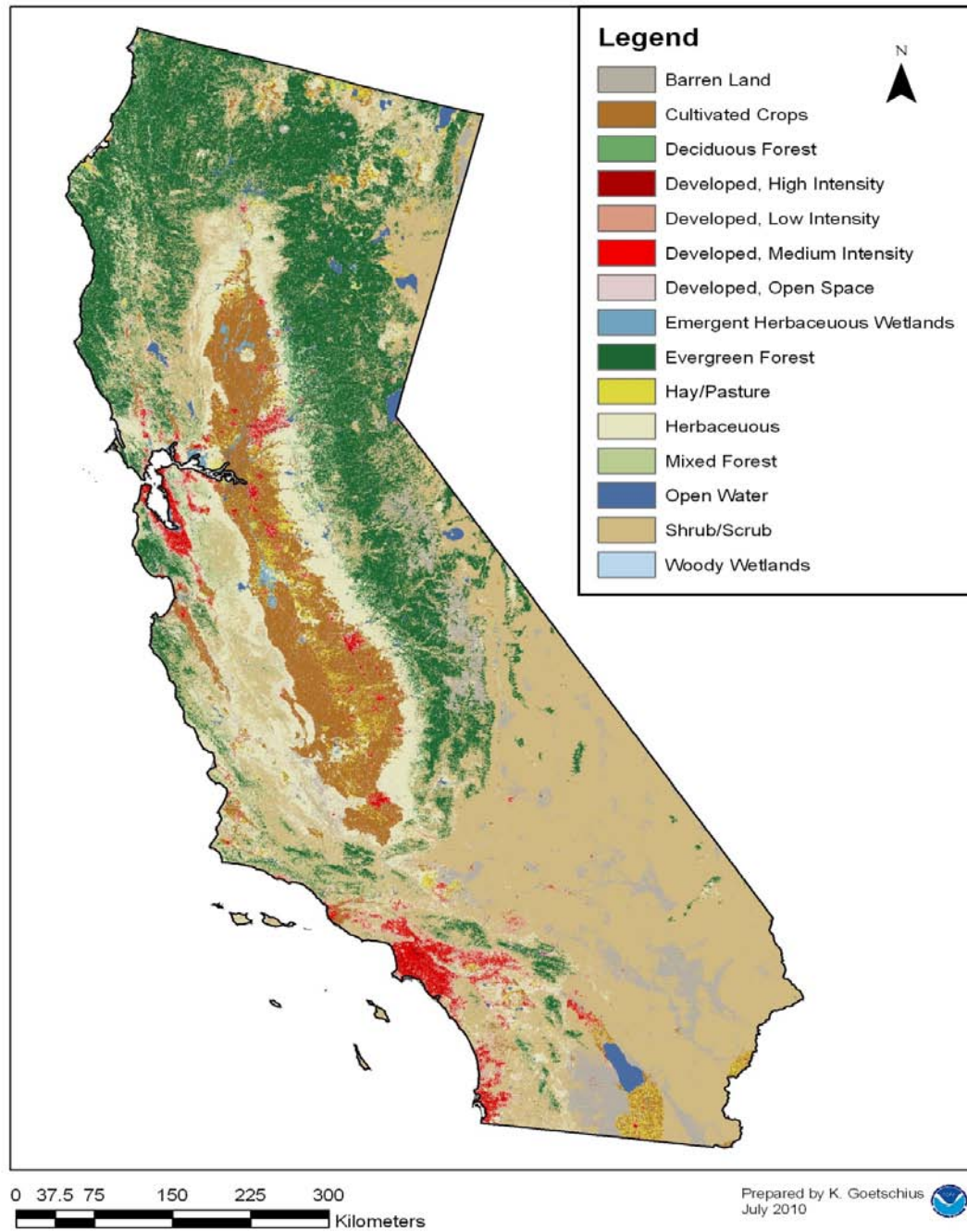


Figure 51. Landuse in Southwest Region.

As a watershed becomes urbanized, population increases and changes occur in stream habitat, water chemistry, and the biota (plants and animals) that live there. The most obvious effect of urbanization is the loss of natural vegetation which results in an increase in impervious cover and dramatic changes to the natural hydrology of urban streams. Urbanization generally results in land clearing, soil compaction, modification and/or loss of riparian buffers, and modifications to natural drainage features (Richter 2002). The increased impervious cover in urban areas leads to increased volumes of runoff, increased peak flows and flow duration, and greater stream velocity during storm events.

Runoff from urban areas also contains all the chemical pollutants from automobile traffic and roads as well as those from industrial sources and residential use. Urban runoff is also typically warmer than receiving waters and can significantly increase temperatures in small urban streams. Warm stream water is detrimental to native aquatic life resident fish and the rearing and spawning needs of anadromous fish. Wastewater treatment plants (WWTP) replace septic systems, resulting in point discharges of nutrients and other contaminants not removed in the processing. Additionally, some cities have combined sewer/stormwater overflows and older systems may discharge untreated sewage following heavy rainstorms. WWTP outfalls often discharge directly into the rivers containing salmonids. These urban nonpoint and point source discharges affect the water quality and quantity in basin surface waters.

In many basins, agriculture is the major water user and the major source of water pollution to surface waters. During general agricultural operations, pesticides are applied on a variety of crops for pest control. These pesticides may contaminate surface water via runoff especially after rain events following application. Agricultural uses of the six a.i.s are described in the *Description of the Proposed Action*. Pesticide detection data for these same a.i.s are reported in the Monitoring subsection of the *Effects of the Proposed Action* chapter.

Pesticide Reduction Programs in the Southwest Coast Region

When using these six a.i.s, growers must adhere to the court-ordered injunctive relief, requiring buffers of 20 yards for ground application and 100 yards for any aerial application. These measures are mandatory in all four states, pending completion of consultation.

California State Code does not include specific limitations on pesticide application aside from human health protections. It only includes statements advising that applicators are required to follow all federal, state, and local regulations. 2,4-D, triclopyr BEE, diuron, and captan all have formulations that are registered as restricted materials in California. This designation means that use of these products is regulated and monitored via licensing and reporting requirements. The degree to which this designation provides additional environmental protections is unclear. Surface water protections exist in draft form for pesticides “that have been determined to have a high potential to contaminate surface water”, including diuron and linuron. While it has undergone public comment and revision, this legislation has not been formally proposed yet.

Additionally, pesticide reduction programs already exist in California to minimize levels of the above a.i.s into the aquatic environment. Monitoring of water resources is handled by the California Environmental Protection Agency’s Regional Water Boards. Each Regional Board makes water quality decisions for its region including setting standards and determining waste discharge requirements. The Central Valley Regional Water Quality Control Board (CVRWQCB) addresses issues in the Sacramento and San Joaquin River Basins. These river basins are characterized by crop land, specifically orchards, which historically rely heavily on organophosphates for pest control.

In 2003, the CVRWQCB adopted the Irrigated Lands Waiver Program (ILWP). Participation was required for all growers with irrigated lands that discharge waste which may degrade water quality. However, the ILWP allowed growers to select one of three methods for regulatory coverage (Markle, Kalman et al. 2005). These options included:

1) join a Coalition Group approved by the CVRWQCB, 2) file for an Individual Discharger Conditional Waiver, and 3) comply with zero discharge regulation (Markle, Kalman et al. 2005). Many growers opted to join a Coalition as the other options were more costly. Coalition Groups were charged with completing two reports – a Watershed Evaluation Report and a Monitoring and Reporting Plan. The Watershed Evaluation Report included information on crop patterns and pesticide/nutrient use, as well as mitigation measures that would prevent orchard runoff from impairing water quality. Similar programs are in development in other agricultural areas of California.

As a part of the Waiver program, the Central Valley Coalitions undertook monitoring of “agriculture dominated waterways”. Some of the monitored waterways are small agricultural streams and sloughs that carry farm drainage to larger waterways. The coalition was also required to develop a management plan to address exceedance of State water quality standards. Currently, the Coalitions monitor toxicity to test organisms, stream parameters (*e.g.*, flow, temperature, etc.), nutrient levels, and pesticides used in the region, including diazinon and chlorpyrifos. Diazinon exceedances within the Sacramento and Feather Rivers resulted in the development of a TMDL. The Coalitions were charged with developing and implementing management and monitoring plans to address the TMDL and reduce diazinon runoff.

The Coalition for Urban/Rural Environmental Stewardship (CURES) is a non-profit organization that was founded in 1997 to support educational efforts for agricultural and urban communities focusing on the proper and judicious use of pest control products. CURES educates growers on methods to decrease diazinon surface water contamination in the Sacramento River Basin. The organization has developed best-practice literature for pesticide use in both urban and agricultural settings (www.curesworks.org). CURES also works with California’s Watershed Coalitions to standardize their Watershed Evaluation Reports and to keep the Coalitions informed. The organization has worked with local organizations, such as the California Dried Plum Board and the Almond Board of California, to address concerns about diazinon, pyrethroids, and sulfur. The CURES

site discusses alternatives to organophosphate dormant spray applications. It lists pyrethroids and carbaryl as alternatives, but cautions that these compounds may impact non-target organisms. The CURES literature does not specifically address the a.i.s discussed in this Opinion.

California also has PURS legislation whereby all agricultural uses of registered pesticides must be reported. In this case “agricultural” use includes applications to parks, golf courses, and most livestock uses.

In 2006, CDPR put limitations on dormant spray application of most insecticides in orchards, in part to adequately protect aquatic life in the Central Valley region. While the legislation was prompted by diazinon and chlorpyrifos exceedences, these limitations also apply to other organophosphates, pyrethroids, and carbamates.

The CDPR publishes voluntary interim measures for mitigating the potential impacts of pesticide usage to listed species. These measures are available online as county bulletins (<http://www.cdpr.ca.gov/docs/endspec/colist.htm>). Measures that apply to 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil use in salmonid habitat are:

- Do not use in currently occupied habitat. (captan and chlorothalonil)
- Do not use in currently occupied habitat except as specified in Habitat Descriptors, in organized habitat recovery programs, or for selective control of exotic plants (2,4 D, triclopyr, diuron and linuron)
- Provide a 20 ft minimum strip of vegetation (on which pesticides should not be applied) along rivers, creeks, streams, wetlands, vernal pools and stock ponds, or on the downhill side of fields where runoff could occur. Prepare land around fields to contain runoff by proper leveling, etc. Contain as much water "on-site" as possible. The planting of legumes or other cover crops for several rows adjacent to off-target water sites is recommended. Mix pesticides in areas not prone to runoff such as concrete mixing/loading pads, disked soil in flat terrain or graveled mix pads, or use a suitable method to contain spills and/or rinsate. Properly empty and triple-rinse pesticide containers at time of use. (captan and chlorothalonil)

- Conduct irrigations efficiently to prevent excessive loss of irrigation waters through runoff. Schedule irrigations and pesticide applications to maximize the interval of time between the pesticide application and the first subsequent irrigation. Allow at least 24 hours between applications of pesticides listed in this bulletin and any irrigation that results in surface runoff into natural waters. Time applications to allow sprays to dry prior to rain or sprinkler irrigations. Do not make aerial applications while irrigation water is on the field unless surface runoff is contained for 72 hours following the application. (diuron, linuron, captan, and chlorothalonil)
- For sprayable or dust formulations: when the air is calm or moving away from habitat, commence applications on the side nearest the habitat and proceed away from the habitat. When air currents are moving toward habitat, do not make applications within 200 yards by air or 40 yards by ground upwind from occupied habitat. The county agricultural commissioner may reduce or waive buffer zones following a site inspection, if there is an adequate hedgerow, windbreak, riparian corridor or other physical barrier that substantially reduces the probability of drift. (all six compounds)
- Do not apply within 30 yards upslope of habitat unless a suitable method is used to contain or divert runoff waters. (diuron and linuron)

Water Diversions for Agriculture in the Southwest Coast Region

Agricultural land use further impacts salmonid aquatic habitats through water diversions or withdrawals from rivers and tributaries. In 1990, nearly 95% of the water diverted from the San Joaquin River was diverted for agriculture. Additionally, 1.5% of the water was diverted for livestock (Carter and Resh 2005). The amount and extent of water withdrawals or diversions for agriculture impact streams and their inhabitants via reduced water flow/velocity and dissolved oxygen levels. For example, adequate water flow is required for migrating salmon along freshwater, estuarine, and marine environments in order to complete their life cycle. Low flow events may delay salmonid migration or lengthen fish presence in a particular water body until favorable flow conditions permit fish migration along the migratory corridor or into the open ocean.

Water diversions may also increase nutrient load, sediments (from bank erosion), and

temperature. Flow management and climate changes have decreased the delivery of suspended particulate matter and fine sediment to the estuary. The conditions of the habitat (shade, woody debris, overhanging vegetation) whereby salmonids are constrained by low flows also may make them more or less vulnerable to predation, elevated temperatures, crowding, and disease. Water flow effects on salmonids may seriously impact adult migration and water quality conditions for spawning and rearing salmonids. High temperature may also result from the loss of vegetation along streams that used to shade the water and from new land uses (buildings and pavement) whereby rainfall picks up heat before it enters into an adjacent stream. Runoff inputs from multiple land use may further pollute receiving waters inhabited by fish or along fish migratory corridors.

Surface and Ground Water Contaminants

Currently, California has over 500 water bodies on its 303(d) list (Wu 2000). The 2006 list includes 779 stream segments, rivers, lakes, and estuaries and 12 pollutant categories (CEPA 2007). Pollutants represented on the list include pesticides, metals, sediments, nutrients or low dissolved oxygen, temperature, bacteria and pathogens, and trash or debris. There are 2,237 water body/pollutant listings; a water body is listed separately for each pollutant detected (CEPA 2007). The 2006 303(d) list identifies water bodies listed due to elevated temperature (Table 59). See species ESU/DPS maps for NPDES permits and 303(d) waters co-located within listed salmonid ESUs/DPSs in California.

Table 59. California's 2006 Section 303(d) List of Water Quality Limited Segments: segments listed for exceeding temperature limits (CEPA 2007).

Pollutant	Estuary Acres Affected	River / Stream Miles Affected	# Water Bodies
Temperature	-	16,907.2	41

Estuary systems of the region are consistently exposed to anthropogenic pressures stemming from high human density sources. For example, the largest west coast estuary is the San Francisco Estuary. This water body provides drinking water to 23 million people, irrigates 4.5 million acres of farmland, and drains roughly 40% of California's

land area. As a result of high use, many environmental measures of the San Francisco Estuary are poor. Water quality suffers from high phosphorus and nitrogen loads, primarily from agricultural, sewage, and storm water runoff. Water clarity is also compromised. Sediments from urban runoff and historical activities contain high levels of contaminants. They include pesticides, polychlorinated biphenyls (PCBs), nickel, selenium, cadmium, mercury, copper, and silver. Specific pesticides include pyrethroids, malathion, carbaryl, and diazinon. Other pollutants include DDT and polynuclear aromatic hydrocarbons (PAHs).

Other wastes are also discharged into San Francisco Bay. Approximately 150 industries discharge wastewater into the bay. Discharge of hot water from power plants and industrial sources may elevate temperatures and negatively affect aquatic life. Additionally, about 60 sewage treatment plants discharge treated effluent into the bay and elevate nutrient loads. However, since 1993, many of the point sources of pollution have been greatly reduced. Pollution from oil spills also occur due to refineries in the bay area. Gold mining has also reduced estuary depths in much of the region, causing drastic changes to habitat. As these stressors persist in the marine environment, the estuary system will likely carry loads for future years, even with strict regulation.

Large urban centers are foci for contaminants. Contaminant levels in surface waters near San Francisco, Oakland, and San Jose are highest. These areas are also where water clarity is at its worst. Some of the most persistent contaminants (PCBs, dioxins, DDT, etc.) are bioaccumulated by aquatic biota and can biomagnify in the food chain. Fish tissues contain high levels of PCB and mercury. Concentrations of PCB were 10 times above human health guidelines for consumption. Birds, some of which are endangered (clapper rail and least tern), have also concentrated these toxins.

As mentioned earlier in this chapter, the distribution of the most prevalent pesticides in streams and ground water correlate with land use patterns and associated past or present pesticide use. The USGS conducted NAWQA analyses for three basins within the Southwest Coast Region. Data for these basins are summarized below:

Santa Ana Basin: NAWQA Analysis

The Santa Ana watershed is the most heavily populated study site out of more than 50 assessment sites studied across the nation by the NAWQA Program. According to Belitz *et al.* (2004), treated wastewater effluent is the primary source of baseflow to the Santa Ana River. Secondary sources that influence peak river flows include stormwater runoff from urban, agricultural, and undeveloped lands (Belitz, Hamlin et al. 2004). Stormwater and agricultural runoff frequently contain pesticides, fertilizers, sediments, nutrients, pathogenic bacteria, and other chemical pollutants to waterways and degrade water quality. The above inputs have resulted in elevated concentrations of nitrates and pesticides in surface waters of the basin. Nitrates and pesticides were more frequently detected here than in other national NAWQA sites (Belitz, Hamlin et al. 2004). Additionally, Belitz *et al.* (2004) found that pesticides and volatile organic compounds (VOCs) were frequently detected in surface and ground water in the Santa Ana Basin.

Of the 103 pesticides and degradates routinely analyzed for in surface and ground water, 58 were detected. Pesticides included diuron, diazinon, carbaryl, chlorpyrifos, lindane, malathion, and chlorothalonil. Diuron was detected in 92% of urban samples – a rate much higher than the national frequency of 25 % (Belitz *et al.* 2004). 2,4-D, triclopyr, and linuron were tested for but not detected. Of the 85 VOCs routinely analyzed for, 49 were detected. VOCs included methyl *tert*-butyl ether (MTBE), chloroform, and trichloroethylene (TCE). Organochlorine compounds were also detected in bed sediment and fish tissue. Organochlorine concentrations were also higher at urban sites than at undeveloped sites in the Santa Ana Basin. Organochlorine compounds include DDT and its breakdown product diphenyl dichloroethylene (DDE), and chlordane. Other contaminants detected at high levels included trace elements such as lead, zinc, and

arsenic. According to Belitz *et al.* (2004), the biological community in the basin is heavily altered as a result from these pollutants.

San Joaquin-Tulare Basin: NAWQA Analysis

A study was conducted by the USGS in the mid-1990s on water quality within the San Joaquin-Tulare basins. Concentrations of dissolved pesticides in this study unit were among the highest of all NAWQA sites nationwide. The USGS detected 49 of the 83 pesticides it tested for in the mainstem and three subbasins. Pesticides were detected in all but one of the 143 samples. The most common detections were of the herbicides simazine, dacthal, metolachlor, and EPTC (Eptam), and the insecticides diazinon and chlorpyrifos. Twenty-two pesticides were detected in over 20% of the samples (Dubrovsky, Kratzer et al. 1998). Further, many samples contained mixtures of at least 7 pesticides, with a maximum of 22 different compounds. Diuron was detected in all three subbasins, despite land use differences. Diuron was detected in roughly 54% of samples, while 2,4-D was found in 12 % . The other two compounds were found much less frequently at 1% for triclopyr and <1% for linuron.

Organochlorine insecticides in bed sediment and tissues of fish or clams were also detected. They include DDT and toxaphene. Levels at some sites were among the highest in the nation. Concentrations of trace elements in bed sediment generally were higher than concentrations found in other NAWQA study units (Dubrovsky, Kratzer et al. 1998).

Sacramento River Basin: NAWQA Analysis

Another study conducted by the USGS from 1996 - 1998 within the Sacramento River Basin compared the pesticides in surface waters at four specific sites – urban, agricultural, and two integration sites (Domagalski 2000). Pesticides included thiobencarb, carbofuran, molinate, simazine, metolachlor, dacthal, chlorpyrifos, carbaryl, and diazinon – as well as the four herbicides assessed in this Opinion. Land use differences between sites are reflected in pesticide detections. Diuron was detected in

66.7 % of agricultural samples, but 85.7% of urban samples (Domagalski 2000). Similarly, 2,4-D and triclopyr both had a detection rate of 19% in agricultural samples, but had higher rates, 28.6% and 32.1% respectively, in urban samples. Linuron detections were lower and more stable, at 4.8% for agricultural and 3.6% for urban samples. Some pesticides were detected at concentrations higher than criteria for the protection of aquatic life in the smaller streams, but were diluted to safer levels in the mainstem river. Intensive agricultural activities also impact water chemistry. In the Salinas River and in areas with intense agriculture use, water hardness, alkalinity, nutrients, and conductivity are also high.

Other Land Uses in the Southwest Coast Region

Habitat Modification

The Central Valley area, including San Francisco Bay and the Sacramento and San Joaquin River Basins, has been drastically changed by development. Salmonid habitat has been reduced to 300 miles from historic estimates of 6,000 miles (CDFG 1993). In the San Joaquin Basin alone, the historic floodplain covered 1.5 million acres with 2 million acres of riparian vegetation (CDFG 1993). Roughly 5% of the Sacramento River Basin's riparian forests remain. Impacts of development include loss of LWD, increased bank erosion and bed scour, changes in sediment loadings, elevated stream temperature, and decreased base flow. Thus, lower quantity and quality of LWD and modified hydrology reduce and degrade salmonid rearing habitat.

The Klamath Basin in Northern California has been heavily modified as well. Water diversions have reduced spring flows to 10% of historical rates in the Shasta River, and dams block access to 22% of historical salmonid habitat. The Scott and Trinity Rivers have similar histories. Agricultural development has reduced riparian cover and diverted water for irrigation (NRC 2003). Riparian habitat has decreased due to extensive logging and grazing. Dams and water diversions are also common. These physical changes resulted in water temperatures too high to sustain salmonid populations. The Salmon

River, however, is comparatively pristine; some reaches are designated as Wild and Scenic Rivers. The main cause of riparian loss in the Salmon River basin is likely wild fires – the effects of which have been exacerbated by salvage logging (NRC 2003).

Mining

Famous for the gold rush of the mid-1800s, California has a long history of mining. Extraction methods such as suction dredging, hydraulic mining, and strip mining may cause water pollution problems. In 2004, California ranked top in the nation for non-fuel mineral production with 8.23% of total production (NMA 2007). Today, gold, silver, and iron ore comprise only 1% of the production value. Primary minerals include construction sand, gravel, cement, boron, and crushed stone. California is the only state to produce boron, rare-earth metals, and asbestos (NMA 2007).

California contains approximately 1,500 abandoned mines. Roughly 1% of these mines are suspected of discharging metal-rich waters into the basins. The Iron Metal Mine in the Sacramento Basin releases more than 1,100 lbs of copper and more than 770 lbs of zinc to the Keswick Reservoir below Shasta Dam. The Iron Metal Mine also released elevated levels of lead (Cain et al. 2000 in Carter and Resh 2005). Metal contamination reduces the biological productivity within a basin. Metal contamination can result in fish kills at high levels or sublethal effects at low levels. Sublethal effects include a reduction in feeding, overall activity levels, and growth. The Sacramento Basin and the San Francisco Bay watershed are two of the most heavily impacted basins within the state from mining activities. The basin drains some of the most productive mineral deposits in the region. Methyl mercury contamination within San Francisco Bay, the result of 19th century mining practices using mercury to amalgamate gold in the Sierra Nevada Mountains, remains a persistent problem today. Based on sediment cores, pre-mining concentrations were about five times lower than concentrations detected within San Francisco Bay today (Conaway, Squire et al. 2003).

Hydromodification Projects

Several of the rivers within California have been modified by dams, water diversions, drainage systems for agriculture and drinking water, and some of the most drastic channelization projects in the nation (see species distribution maps). In all, there are about 1,400 dams within the State of California, more than 5,000 miles of levees, and more than 140 aqueducts (Mount 1995). In general, the southern basins have a warmer and drier climate and the more northern, coastal-influenced basins are cooler and wetter. About 75% of the runoff occurs in basins in the northern half of California, while 80% of the water demand is in the southern half. Two water diversion projects meet these demands—the federal Central Valley Project (CVP) and the California State Water Project (CSWP). The CVP is one of the world’s largest water storage and transport systems. The CVP has more than 20 reservoirs and delivers about 7 million acre-ft per year to southern California. The CSWP has 20 major reservoirs and holds nearly 6 million acre-ft of water. The CSWP delivers about 3 million acre-ft of water for human use. Together, both diversions irrigate about 4 million acres of farmland and deliver drinking water to roughly 22 million residents.

Both the Sacramento and San Joaquin rivers are heavily modified, each with hundreds of dams. The Rogue, Russian, and Santa Ana rivers each have more than 50 dams, and the Eel, Salinas, and the Klamath Rivers have between 14 and 24 dams each. The Santa Margarita is considered one of the last free flowing rivers in coastal southern California with nine dams occurring in its watershed. All major tributaries of the San Joaquin River are impounded at least once and most have multiple dams or diversions. The Stanislaus River, a tributary of the San Joaquin River, has over 40 dams. As a result, the hydrograph of the San Joaquin River is seriously altered from its natural state. Alteration of the temperature and sediment transport regimes had profound influences on the biological community within the basin (Figure 52). These modifications generally result in a reduction of suitable habitat for native species and frequent increases in suitable habitat for non-native species. The Friant Dam on the San Joaquin River is attributed with the extirpation of spring-run Chinook salmon within the basin. A run of the spring-

run Chinook salmon once produced about 300,000 to 500,000 fish (Carter and Resh 2005).

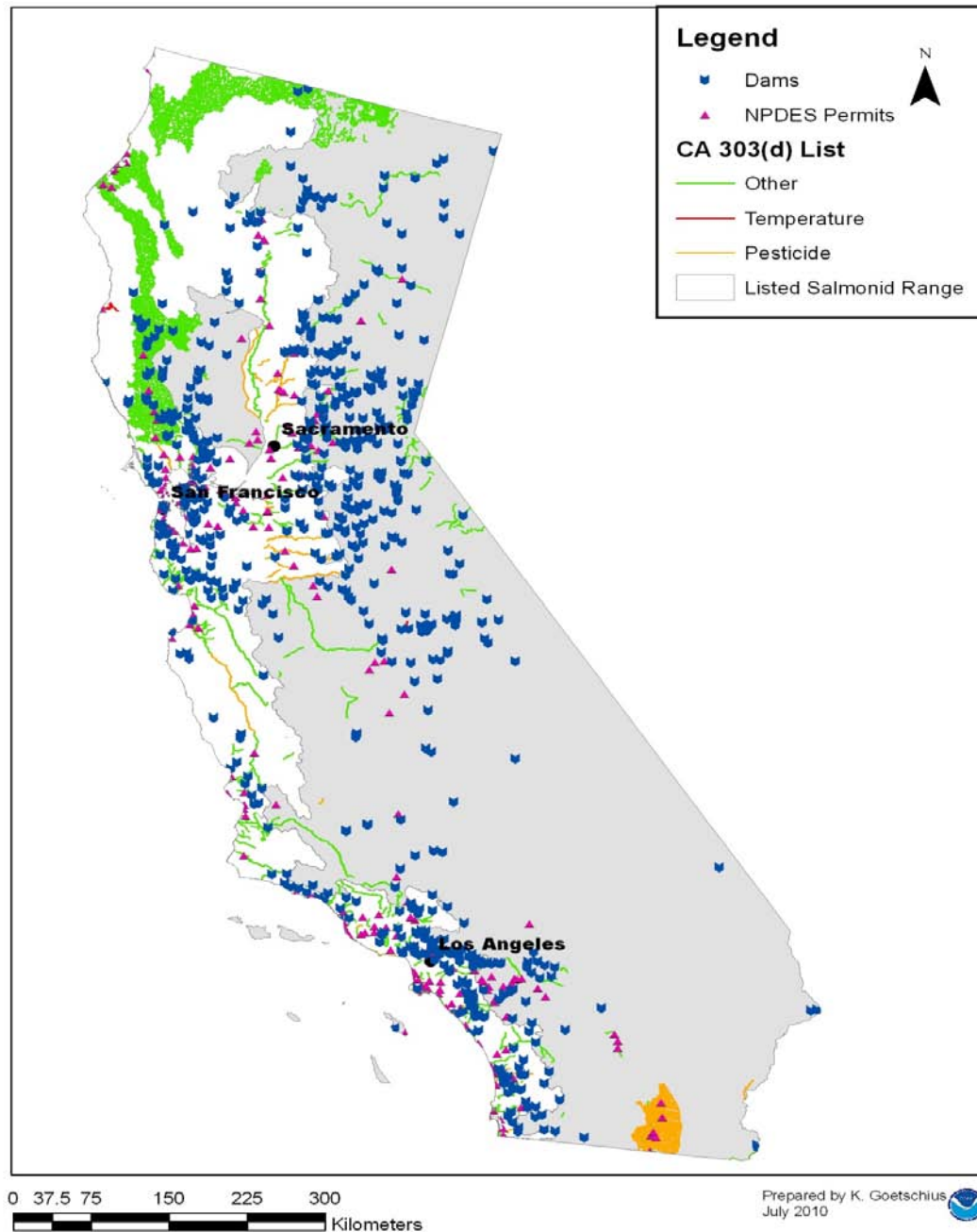


Figure 52 Southwest Coast 303(d) waters, dams, and NPDES permit sites.

Artificial Propagation

Anadromous fish hatcheries have existed in California since establishment of the McCloud River hatchery in 1872. There are nine state hatcheries: the Iron Gate (Klamath River), Mad River, Trinity (Trinity River), Feather (Feather River), Warm Springs (Russian River), Nimbus (American River), Mokelumne (Mokelumne River), and Merced (Merced River). The California Department of Fish and Game (CDFG) also manages artificial production programs on the Noyo and Eel rivers. The Coleman National Fish Hatchery, located on Battle Creek in the upper Sacramento River, is a federal hatchery operated by the USFWS. The USFWS also operates an artificial propagation program for Sacramento River winter run Chinook salmon.

Of these, the Feather River, Nimbus, Mokelumne, and Merced River facilities comprise the Central Valley Hatcheries. Over the last ten years, the Central Valley Hatcheries have released over 30 million young salmon. State and the federal (Coleman) hatcheries work together to meet overall goals. State hatcheries are expected to release 18.6 million smolts in 2008 and Coleman is aiming for more than 12 million. There has been no significant change in hatchery practices over the year that would adversely affect the current year class of fish. A new program marking 25% of the 32 million Sacramento River Fall-run Chinook smolts may provide data on hatchery fish contributions to the fisheries in the near future.

Commercial and Recreational Fishing

The region is home to many commercial fisheries. The largest in terms of total California landings in 2006 were northern anchovy, Pacific sardine, Chinook salmon, sablefish, Dover sole, Pacific whiting, squid, red sea urchin, and Dungeness crab (CDFG 2007). Red abalone is also harvested.

Despite regulated fishing programs for salmonids, listed salmonids are also caught as bycatch. There are several approaches under the ESA to address tribal and state take of ESA-listed species that may occur as a result of harvest activities. Section 10 of the ESA

provides for permits to operate fishery harvest programs. ESA section 4(d) rules provide exemptions from take for resource, harvest, and hatchery management plans.

Management of salmon fisheries in the Southwest Coast Region is a cooperative process involving federal, state, and tribal representatives. The Pacific Fishery Management Council sets annual fisheries in federal waters from three to 200 miles off the coasts of Washington, Oregon, and California. Inland fisheries are those within state boundaries, including those extending out three miles from state coastlines. The states of Oregon, Idaho, California, and Washington issue salmon fishing licenses for inland fisheries. The California Fish and Game Commission (CFGF) establish the salmon seasons and issues permits for all California waters and the Oregon Department of Fish and Game sets the salmon seasons and issues permits for all Oregon waters.

In 2008, there was an unprecedented collapse of the Sacramento River fall-run Chinook salmon that led to complete closure of the commercial and sport Chinook fisheries in California and in Oregon south of Cape Falcon. U.S. Department of Commerce Secretary Gary Locke released a 2008 West Coast salmon disaster declaration for California and Oregon in response to poor salmon returns to the Sacramento River, which led to federal management reducing commercial salmon fishing off southern Oregon and California to near zero. Secretary Locke also released \$53.1 million in disaster funds to aid affected fishing communities.

In 2009, federal fishery managers severely limited commercial salmon fishing in California and Oregon for the second year in a row due to low Sacramento River fall-run Chinook salmon returns. California State sport and commercial ocean salmon seasons were closed by the CFGF through August 28, 2009. There was a 10-day ocean sport fishery in the Klamath Management Zone (Horse Mountain to the California-Oregon border) from August 29 through September 7, 2009. A limited in-river salmon season was considered by the CFGF at its May meeting. The CFGF decided to leave open the Sacramento River between the Highway 113 bridge near Knight's Landing and just below

the Lower Red Bluff (Sycamore) Boat Ramp from November 16 through December 31, 2009. The Klamath-Trinity River Basin had a salmon sport fishing season for Klamath River fall Chinook salmon that began August 15, 2009.

Non-native Species

Plants and animals that are introduced into habitats where they do not naturally occur are called non-native species. They are also known as non-indigenous, exotic, introduced, or invasive species, and have been known to affect ecosystems. Non-native species are introduced through infested stock for aquaculture and fishery enhancement, through ballast water discharge and from the pet and recreational fishing industries (<http://biology.usgs.gov/s+t/noframe/x191.htm>). The Aquatic Nuisance Species Task Force suggests that it is inevitable that cultured species will eventually escape confinement and enter U.S. waterways. Non-native species were cited as a contributing cause in the extinction of 27 species and 13 subspecies of North American fishes over the past 100 years (Miller, Williams et al. 1989). Wilcove, Rothstein *et al.* (1998) note that 25% of ESA-listed fish are threatened by non-native species. By competing with native species for food and habitat as well as preying on them, non-native species can reduce or eliminate populations of native species.

Surveys performed by CDFG state that at least 607 non-native species are found in California coastal waterways (Foss, Ode et al. 2007). The majority of these species are representatives of four phyla: annelids (33%), arthropods (22%), chordates (13%), and mollusks (10%). Non-native chordate species are primarily fish and tunicates which inhabit fresh and brackish water habitats such as the Sacramento-San Joaquin Delta (Foss, Ode et al. 2007). The California Aquatic Invasive Species Management Plan includes goals and strategies for reducing the introduction rate of new invasive species as well as removing those with established populations.

Pesticide Reduction Programs in the Pacific Northwest Region

When using any of the six a.i.s addressed in this Opinion, growers must adhere to the court-ordered injunctive relief, requiring buffers of 20 yards for ground application and 100 yards for any aerial application. These measures are mandatory in all four states, pending completion of consultation. Additionally, pesticide reduction programs exist in Idaho, Oregon, and Washington to minimize levels of pesticides in the aquatic environment. All three states have some limitations on the use of pesticides as a part of their Administrative codes. Most are regulations are focused on chemical use in forestry applications. Table 60 summarizes the existing legislation in the North West.

Table 60. Summary of State-level limitations on pesticide use in Idaho, Oregon, and Washington. All materials are available on organization web-sites.

State	Limitation	Source	Code
Oregon	Forestry Aerial Application: No fungicides applied by aircraft within 300 feet of significant wetlands, type F streams (with salmonids or other game fish), large lakes, areas of other lakes used by fish, or within 60 feet of other perennial streams	Dept. of Forestry	629-620-0400
Oregon	Forestry Aerial: No chemicals by aircraft applied by aircraft within 60 feet of significant wetlands, type F streams , large lakes, and areas of other lakes used by fish	Dept. of Forestry	629-620-0400
Oregon	Forestry: Requires additional state permit for use of isopropyl ester of 2,4-D or any other ester of equal or higher volatility with regard to plant damage	State Forest Laws	634.372
Washington	Rights of Way: 2,4-D amine formulations and Triclopyr Ester cannot be used within 60 feet of water; Diuron cannot be used in western Washington, and not within 60 feet of water in eastern Washington	Washington State Dept. of Transportation	NA

State	Limitation	Source	Code
Washington	Forestry Ground Application: application with power equipment is prohibited within the core and inner zone, channel migration zone of Type S and F Waters; Operators shall maintain a 25 foot no application buffer strip around Type A or B Wetlands and on all sides of all other surface waters.	Washington Administrative Code Forest Practices Board	222-38-020
Washington	Forestry Hand Application: No pesticides may be applied within the core zone, channel migration zone of Type S and F Waters ; Pesticides must be applied to specific targets, such as vegetation, trees, stumps, etc.	Washington Administrative Code Forest Practices Board	222-38-020
Washington	Forestry Aerial Application: mandatory buffers depending on nozzle type, application height, weather conditions, and width of stream; Ranges from 10 to 325 feet	Washington Administrative Code Forest Practices Board	222-38-021 222-38-022
Idaho	Forestry Aerial Application: must leave a buffer of untreated land (minimum off 100 ft) on each side of all Class I streams, flowing Class II streams and other areas of open water.	Idaho Administrative Code, Dept of Lands, Forest Practices Act	20.02.01
Idaho	Forestry Ground Application: must leave a buffer of untreated land (minimum off 100 ft) on each side of all Class I streams, flowing Class II streams and other areas of open water.	Idaho Administrative Code, Dept of Lands, Forest Practices Act	20.02.01
Idaho	Forestry Hand Application: apply only to specific targets	Idaho Administrative Code, Dept of Lands, Forest Practices Act	20.02.01

State	Limitation	Source	Code
Idaho	Home and garden: use of high volatile liquid ester formulations of 2,4-D prohibited - both homeowner use and professional applications to home/garden locations	Idaho Administrative Code, Dept of Agriculture, Pesticide & Chemigation Use & Application Rules	02.03.03 500.01
Idaho	Home and garden: Low volatile liquid ester formulations of 2,4-D; 2,4-DP; MCPA and MCPB shall not be applied around any home or garden between May 1 and October 1 of any year or at any time when air temperature exceeds eighty (80) degrees Fahrenheit.	Idaho Administrative Code, Dept of Agriculture, Pesticide & Chemigation Use & Application Rules	02.03.03 500.02
Idaho	Agriculture Aerial Application: No aircraft pilot shall apply high volatile ester formulations of 2,4-D in Latah, Nez Perce, and Clearwater Counties, or within five miles of a susceptible crop or hazard area in any other county in Idaho	Idaho Administrative Code, Dept of Agriculture, Pesticide & Chemigation Use & Application Rules	02.03.03 550.01
Idaho	Agriculture Aerial Application: No aircraft pilot shall apply low volatile ester formulations of 2,4-D in Latah, Nez Perce, and Clearwater Counties unless ambient air temperatures are not above or expected to exceed eighty-five degrees Fahrenheit within twenty-four hours of the expected application time, or within one miles of a susceptible crop or hazard area in any other county in Idaho	Idaho Administrative Code, Dept of Agriculture, Pesticide & Chemigation Use & Application Rules	02.03.03 550.02

The Idaho State Department of Agriculture has published a BMP guide for pesticide use.

The BMPs include eight “core” voluntary measures that will prevent pesticides from leaching into soil and groundwater. These measures include applying pest-specific controls, being aware of the depth to ground water, and developing an Irrigation Water Management Plan.

Oregon has PURS legislation that requires all agricultural uses of registered pesticides be reported. In this case “agricultural” use includes applications to parks, golf courses, and most livestock uses. Oregon requires reporting if application is part of a business, for a government agency, or in a public place. However, the Governor of Oregon has suspended the PURS program until January 2013 due to budget shortages.

Oregon has also implemented a voluntary program. The Pesticide Stewardship Partnerships (PSP) program began in 1999 through the Oregon Department of Environmental Quality. The PSP’s goal is to involve growers and other stakeholders in water quality management at a local level. Effectiveness monitoring is used to provide feedback on the success of mitigation measures. As of 2006, there were six pilot PSPs planned or in place. Early results from the first PSPs in the Columbia Gorge Hood River and in Mill Creek demonstrate reductions in chlorpyrifos and diazinon levels and detection frequencies. DEQ’s pilot programs suggest that PSPs can help reduce contamination of surface waters.

Oregon is in the process of developing a Pesticide Management Plan for Water Quality Protection, as required under FIFRA. This plan describes how government agencies and stakeholders will collaboratively reduce pesticides in Oregon water supplies. The PSP program is a component of this plan, and will provide information on the effectiveness of mitigation measures.

Washington State has a Surface Water Monitoring Program that looks at pesticide concentrations in some salmonid bearing streams and rivers. The program was initiated in 2003 and now monitors four areas. Three of these were chosen due to high overlap

with agriculture: the Skagit-Samish watershed, the Lower Yakima Watershed, and the Wenatchee and Entiat watersheds. The final area, in the Cedar-Sammamish watershed, is an urban location, intended to look at runoff in a non-agriculture setting. It was chosen due to detection of pesticides coincident with pre-spawning mortality in coho salmon. The Surface Water Monitoring program is relatively new and will continue to add watersheds and testing for additional pesticides over time.

Washington State also has a voluntary program that assists growers in addressing water rights issues within a watershed. Several watersheds have elected to participate, forming Comprehensive Irrigation District Management Plans (CIDMPs). The CIDMP is a collaborative process between government and landowners and growers; the parties determine how they will ensure growers get the necessary volume of water while also guarding water quality. This structure allows for greater flexibility in implementing mitigation measures to comply with both the CWA and the ESA.

The Columbia Gorge Fruit Growers Association is a non-profit organization dedicated to the needs of growers in the mid-Columbia area. The association brings together over 440 growers and 20 shippers of fruit from Oregon and Washington. It has issued a BMP handbook for OPs, including information on alternative methods of pest control. The mid-Columbia area is of particular concern, as many orchards are in close proximity to streams.

Stewardship Partners is a non-profit organization in Washington State that works to build partnerships between landowners, government, and non-profit organizations. In large part, its work focuses on helping landowners to restore fish and wildlife habitat while maintaining the economic viability of their farmland. Projects include restoring riparian areas, reestablishing floodplain connectivity, and removing blocks to fish passage. Another current project is to promote rain gardens as a method of reducing surface water runoff from developed areas. Rain gardens mimic natural hydrology, allowing water to collect and infiltrate the soil.

Stewardship Partners also collaborates with the Oregon-based Salmon-Safe certification program. Salmon-Safe is an independent eco-label recognizing organizations who have adopted conservation practices that help restore native salmon habitat in Pacific Northwest rivers and streams. These practices protect water quality, fish and wildlife habitat, and overall watershed health. While the program began with a focus on agriculture, it has since expanded to include industrial and urban sites as well. The certification process includes pesticide restrictions. Salmon-Safe has produced a list of “high risk” pesticides which, if used, would prevent a site from becoming certified. If a grower wants an exception, they must provide written documentation that demonstrates a clear need for use of the pesticide, that no safer alternatives exist, and that the method of application (such as timing, location, and amount used) represents a negligible risk to water quality and fish habitat. Bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, naled, and phosmet are all on the high risk list. Over 250 farms and businesses currently have the Salmon-Safe certification.

In addition to pesticide usage for agriculture, this land use further affects available salmonid aquatic habitat. The amount and extent of water withdrawals or diversions for agriculture impact streams and their inhabitants via reduced water flow/velocity and dissolved oxygen levels. These impacts are described below.

Pacific Northwest Region

This region encompasses Idaho, Oregon, and Washington and includes parts of Nevada, Montana, Wyoming, and British Columbia. In this section we discuss three major areas that support salmonid populations within the action area. They include the Columbia River Basin and its tributaries, the Puget Sound Region, and the coastal drainages north of the Columbia River (Figure 53).

Eighteen of the 28 ESUs/DPSs addressed in the Opinion occur within the Pacific Northwest Region. They are the Puget Sound Chinook salmon, Lower Columbia River (LCR) Chinook salmon, Upper Columbia River (UCR) Spring-run Chinook salmon, Snake River (SR) Fall-run Chinook salmon, SR Spring/Summer-run Chinook salmon, Upper Willamette River (UWR) Chinook salmon, Hood Canal (HC) Summer-run chum, Columbia River (CR) chum, LCR coho, Oregon Coast (OC) coho, Ozette Lake sockeye, SR sockeye, Puget Sound steelhead, LCR steelhead, UWR steelhead, Middle Columbia River (MCR) steelhead, UCR steelhead, and the SR steelhead (Table 54). Table 61, Table 62, and Table 63 show the types and areas of land use within each salmonid ESU/DPS.

Table 61. Area of land use categories within Chinook Salmon ESUs in km² where bolded numbers are totals of each category. Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (NLCD 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Landcover Type			Chinook Salmon					
			Puget Sound	Lower Columbia River	Upper Columbia River Spring Run	Sneke River Fall Run	Sneke River Spring/Summer Run	Upper Willamette River
Water			6,485	653	203	236	293	130
Open Water	11		6,172	641	188	233	253	124
Perennial Snow/Ice	12		313	12	16	3	40	7
Developed Land			5,271	1,861	847	543	974	2,008
Open Space	21		1,601	649	203	401	328	632
Low Intensity	22		1,694	517	218	79	113	722
Medium Intensity	23		668	290	55	20	30	322
High Intensity	24		266	118	11	2	2	112
Barren Land	31		1,042	287	360	41	500	220
Undeveloped Land			22,481	10,692	16,155	31,231	52,573	14,159
Deciduous Forest	41		999	551	21	30	10	248
Evergreen Forest	42		14,443	6,497	8,138	18,447	27,701	9,531
Mixed Forest	43		2,526	927	7	16	4	1,130
Shrub/Scrub	52		2,415	1,598	6,100	6,315	13,618	1,940
Herbaceous	71		957	520	1,737	6,358	11,053	801
Woody Wetlands	90		648	377	92	35	96	431
Emergent Wetlands	95		492	223	59	30	92	78
Agriculture			1,447	825	964	5,557	4,316	5,972
Hay/Pasture	81		1,188	547	327	59	456	3,617
Cultivated Crops	82		258	278	636	5,497	3,860	2,355
TOTAL (inc. open water)			35,683	14,031	18,168	37,566	58,157	22,269
TOTAL (w/o open water)			29,511	13,390	17,981	37,331	57,904	22,146

Table 62. Area of land use categories within chum and coho ESUs in km². Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (NLCD 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Landcover Type			Chum Salmon		Coho Salmon		Sockeye Salmon	
			Hood Canal Summer Run	Columbia River	Lower Columbia River	Oregon Coast	Ozette Lake	Snake River
Water			755	656	687	200	30	36
Open Water	11		704	655	675	200	30	19
Perennial Snow/Ice	12		51	1	12	0	0	18
Developed Land			403	1,684	1,990	1,807	3	15
Open Space	21		134	605	708	1,107	1	3
Low Intensity	22		77	463	563	163	0	2
Medium Intensity	23		20	258	305	49	0	0
High Intensity	24		6	110	124	20	0	0
Barren Land	31		166	247	290	467	2	9
Undeveloped Land			3,324	8,198	13,254	24,589	195	1,259
Deciduous Forest	41		97	548	575	418	3	0
Evergreen Forest	42		2,477	4,294	8,487	14,943	158	755
Mixed Forest	43		200	892	999	4,126	3	0
Shrub/Scrub	52		299	1,353	1,982	3,134	14	185
Herbaceous	71		61	363	386	263	8	269
Woody Wetlands	90		56	222	225	226	8	16
Emergent Wetlands	95		133	526	600	1,478	1	34
Agriculture			66	746	1,028	925	0	13
Hay/Pasture	81		64	533	680	860	0	12
Cultivated Crops	82		2	213	348	64	0	1
TOTAL (inc. open water)			4,548	11,284	16,959	27,520	228	1,323
TOTAL (w/o open water)			3,843	10,628	16,284	27,320	199	1,304

Table 63. Area of land use categories within sockeye ESUs and steelhead DPSs in km² where bolded numbers are totals for each category. Land cover image data were taken from Multi-Resolution Land Characteristics (MRLC) Consortium, a consortium of nine federal agencies (USGS, EPA, USFS, NOAA, NASA, BLM, NPS, NRCS, and USFWS) (NLCD 2001). Land cover class definitions are available at: http://www.mrlc.gov/nlcd_definitions.php

Landcover			Steelhead					
			Puget Sound	Lower Columbia River	Upper Willamette River	Middle Columbia River	Upper Columbia River	Snake River
sub category	Code							
Water			6,485	262	62	588	375	327
Open Water	11		6,172	250	62	575	359	285
Perennial Snow/Ice	12		313	12	0	13	16	42
Developed Land			5,271	1,601	1,278	2,304	1,092	1,205
Open Space	21		1,601	518	382	1,276	343	515
Low Intensity	22		1,694	506	513	627	294	144
Medium Intensity	23		668	287	231	192	80	40
High Intensity	24		266	116	75	25	13	3
Barren Land	31		1,042	174	77	183	361	504
Undeveloped Land			22,481	10,339	6,942	53,790	19,621	67,839
Deciduous Forest	41		999	382	171	54	25	35
Evergreen Forest	42		14,443	7,023	4,133	18,347	8,223	39,556
Mixed Forest	43		2,526	611	791	41	7	17
Shrub/Scrub	52		2,415	1,589	994	32,089	9,351	15,644
Herbaceous	71		957	398	519	2,752	1,823	12,361
Woody Wetlands	90		648	244	292	217	109	116
Emergent Wetlands	95		492	93	43	291	81	111
Agriculture			1,447	927	4,373	12,771	3,684	6,690
Hay/Pasture	81		1,188	605	2,529	863	448	463
Cultivated Crops	82		258	322	1,844	11,908	3,236	6,227
TOTAL (inc. open water)			35,683	13,128	12,655	69,453	24,771	76,061
TOTAL (w/o open water)			29,511	12,878	12,593	68,878	24,411	75,777

Columbia River Basin

The most notable basin within the Pacific Northwest region is the Columbia River. The Columbia River is the largest river in the Pacific Northwest and the fourth largest river in

terms of average discharge in the U.S. The Columbia River drains over 258,000 square miles, and is the sixth largest in terms of drainage area. Major tributaries include the Snake, Willamette, Salmon, Flathead, and Yakima rivers. Smaller rivers include the Owyhee, Grande Ronde, Clearwater, Spokane, Methow, Cowlitz, and the John Day Rivers (see Table 64 for a description of select Columbia River tributaries). The Snake River is the largest tributary at more than 1,000 miles long. The headwaters of the Snake River originate in Yellowstone National Park, Wyoming. The second largest tributary is the Willamette River in Oregon (Kammerer 1990; Hinck, Schmitt et al. 2004). The Willamette River is also the 19th largest river in the nation in terms of average annual discharge (Kammerer 1990). The basins drain portions of the Rocky Mountains, Bitterroot Range, and the Cascade Range.

Table 64. Select tributaries of the Columbia River (Carter and Resh 2005).

Watershed	Approx Length (mi)	Basin Size (mi ²)	Physiographic Provinces*	Mean Annual Precipitation (in)	Mean Discharge (cfs)	No. Fish Species (native)	No. Endangered Species
Snake/Salmon rivers	870	108,495	CU, NR, MR, B/R	14	55,267	39 (19)	5 fish (4 T, 1 E), 6 (1 T, 5 E) snails, 1 plant (T)
Yakima River	214	6,139	CS, CU	7	3,602	50	2 fish (T)
Willamette River	143	11,478	CS, PB	60	32,384	61 (~31)	5 fish (4 T, 1 E),

* Physiographic Provinces: CU = Columbia-Snake River Plateaus, NR = Northern Rocky Mountains, MR = Middle Rocky Mountains, B/R = Basin & Range, CS = Cascade-Sierra Mountains, PB = Pacific Border

The Columbia River and estuary were once home to more than 200 distinct runs of Pacific salmon and steelhead with unique adaptations to local environments within a tributary (Stanford, Hauer et al. 2005). Salmonids within the basin include Chinook salmon, chum salmon, coho salmon, sockeye salmon, steelhead, redband trout, bull trout, and cutthroat trout.

Land Use in the Columbia River Basin

More than 50% of the U.S. portion of the Columbia River Basin is in federal ownership (most of which occurs in high desert and mountain areas). Approximately 39% is in

private land ownership (most of which occurs in river valleys and plateaus). The remaining 11% is divided among the tribes, state, and local governments (Hinck, Schmitt et al. 2004). See Table 65 for a summary of land uses and population densities in several subbasins within the Columbia River watershed [data from (Stanford, Hauer et al. 2005)].

Table 65. Land use and population density in select tributaries of the Columbia River (Stanford, Hauer et al. 2005).

Watershed	Land Use Categories (Percent)				Density (people/mi ²)
	Agriculture	Forest	Urban	Other	
Snake/Salmon rivers	30	10-15	1	54 scrub/rangeland/barren	39
Yakima River	16	36	1	47 shrub	80
Willamette River	19	68	5	--	171

The interior Columbia Basin has been altered substantially by humans causing dramatic changes and declines in native fish populations. In general, the basin supports a variety of mixed uses. Predominant human uses include logging, agriculture, ranching, hydroelectric power generation, mining, fishing, a variety of recreational activities, and urban uses. The decline of salmon runs in the Columbia River is attributed to loss of habitat, blocked migratory corridors, altered river flows, pollution, overharvest, and competition from hatchery fish. In the Yakima River, 72 stream and river segments are listed as impaired by the Washington State Department of Ecology (DOE) and 83% exceed temperature standards. In the Yakima River, non-native grasses and other plants are commonly found along the lower reaches of the river (Stanford, Hauer et al. 2005). In the Willamette River, riparian vegetation was greatly reduced by land conversion. By 1990, only 37% of the riparian area within 120 m was forested, 30% was agricultural fields, and 16% was urban or suburban lands.

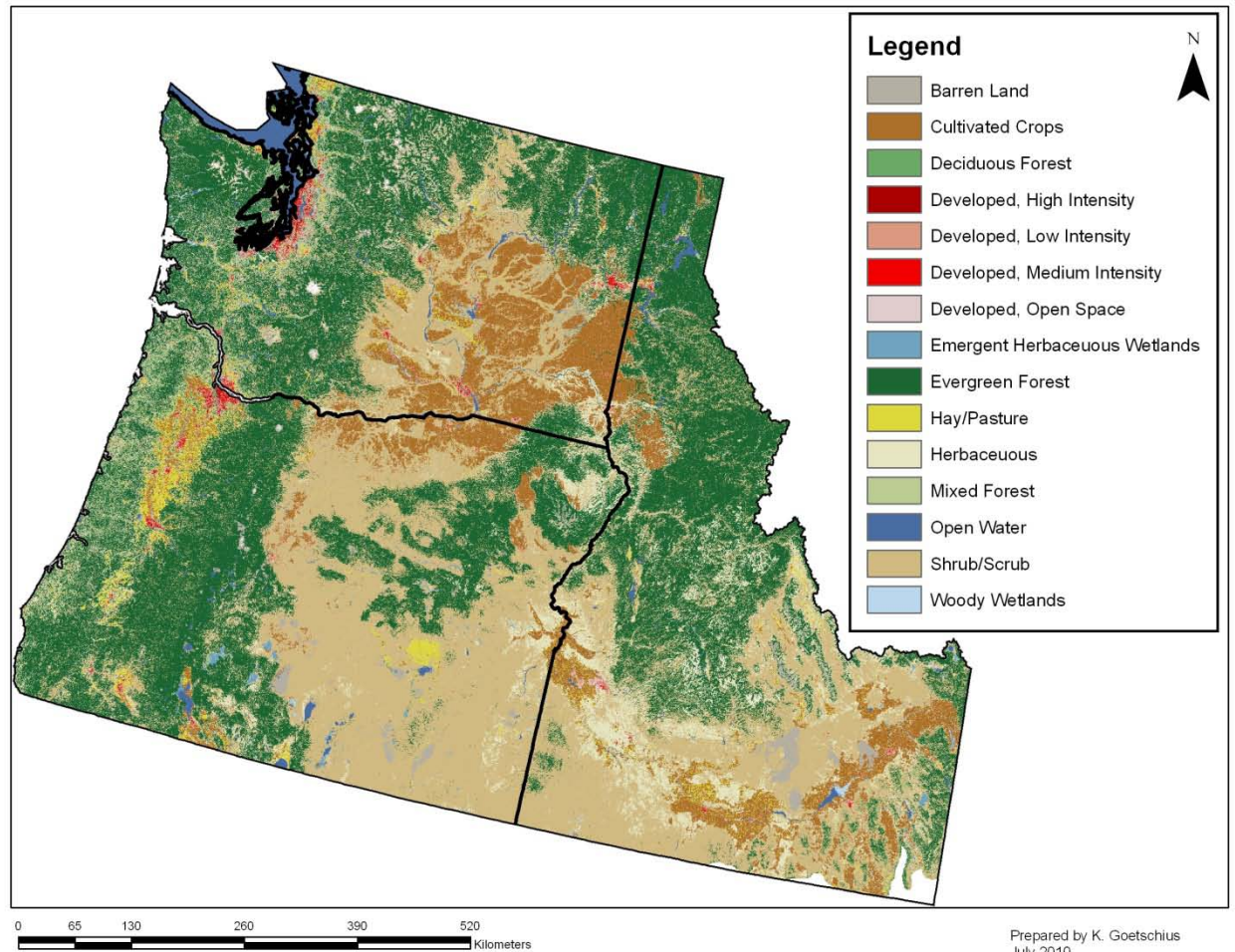


Figure 53. Pacific Northwest Landuse.

Ranching and Agriculture

Ranching, agriculture, and related services in the Pacific Northwest employ more than nine times the national average [19% of the households within the basin (NRC 2004)]. Ranching practices have led to increased soil erosion and sediment loads within adjacent tributaries. The worst of these effects may have occurred in the late 1800s and early 1900s from deliberate burning to increase grass production (NRC 2004). Several measures are currently in place to reduce the impacts of grazing. Measures include restricted grazing in degraded areas, reduced grazing allotments, and lowered stocking rates. Today, the agricultural industry impacts water quality within the basin. Agriculture is second only to the large-scale influences of hydromodification projects regarding power generation and irrigation. Water quality impacts from agricultural activities include alteration of the natural temperature regime, insecticide and herbicide contamination, and increased suspended sediments. During general agricultural operations, pesticides are applied on a variety of crops for pest control. These pesticides may contaminate surface water via runoff especially after rain events following application. Agricultural uses of the a.i.s assessed in this Opinion are discussed in the *Description of the Proposed Action*, while detection data is discussed in the Monitoring subsection of the *Effects of the Proposed Action* chapter.

Water Diversions for Agriculture in the Pacific Northwest Region

Agriculture and ranching increased steadily within the Columbia River basin from the mid- to late-1800s. By the early 1900s, agricultural opportunities began increasing at a much more rapid pace with the creation of more irrigation canals and the passage of the Reclamation Act of 1902 (NRC 2004). Today, agriculture represents the largest water user within the basin (>90%).

Roughly 6% of the annual flow from the Columbia River is diverted for the irrigation of

7.3 million acres of croplands within the basin. The vast majority of these agricultural lands are located along the lower Columbia River, the Willamette, Yakima, Hood, and Snake rivers, and the Columbia Plateau (Hinck, Schmitt et al. 2004).

The impacts of these water diversions include an increase nutrient load, sediments (from bank erosion), and temperature. Flow management and climate changes have further decreased the delivery of suspended particulate matter and fine sediment to the estuary. The conditions of the habitat (shade, woody debris, overhanging vegetation) whereby salmonids are constrained by low flows also may make fish more or less vulnerable to predation, elevated temperatures, crowding, and disease. Water flow effects on salmonids may seriously impact adult migration and water quality conditions for spawning and rearing salmonids. High temperature may also result from the loss of vegetation along streams that used to shade the water and from new land uses (buildings and pavement) whereby rainfall picks up heat before it enters into an adjacent stream. Runoff inputs from multiple land use may further pollute receiving waters inhabited by fish or along fish migratory corridors.

Surface and Ground Water Contaminants

NAWQA analyses were conducted for for five basins within the Pacific Northwest Region. The USGS has a number of fixed water quality sampling sites throughout various tributaries of the Columbia River. Many of the water quality sampling sites have been in place for decades. Water volumes, crop rotation patterns, crop type, and basin location are some of the variables that influence the distribution and frequency of pesticides within a tributary. Detection frequencies for a particular pesticide can vary widely. In addition to current use-chemicals, legacy chemicals continue to pose a serious problem to water quality and fish communities despite their ban in the 1970s and 1980s (Hinck, Schmitt et al. 2004).

Fish and macroinvertebrate communities exhibit an almost linear decline in condition as the level of agriculture intensity increases within a basin (Cuffney, Meador et al. 1997;

Fuhrer, Morace et al. 2004). A study conducted in the late 1990s examined 11 species of fish, including anadromous and resident fish collected throughout the basin, for a suite of 132 contaminants. They included 51 semi-volatile chemicals, 26 pesticides, 18 metals, 7 PCBs, 20 dioxins, and 10 furans. Sampled fish tissues revealed PCBs, metals, chlorinated dioxins and furans (products of wood pulp bleaching operations), and other contaminants.

Yakima River Basin: NAWQA Analysis

The Yakima River Basin is one of the most agriculturally productive areas in the U.S. (Fuhrer, Morace et al. 2004). Croplands within the Yakima Basin account for about 16% of the total basin area of which 77% is irrigated. The extensive irrigation-water delivery and drainage system in the Yakima River Basin greatly controls water quality conditions and aquatic health in agricultural streams, drains, and the Yakima River (Fuhrer, Morace et al. 2004). From 1999 to 2000, the USGS conducted a NAWQA study in the Yakima River Basin. Fuhrer *et al.* (2004) reported that nitrate and orthophosphate were the dominant forms of nitrogen and phosphorus found in the Yakima River and its agricultural tributaries. Arsenic, a known human carcinogen, was also detected in agricultural drains at elevated concentrations.

The USGS also detected 76 pesticide compounds in the Yakima River Basin. They include 38 herbicides (including 2,4-D and diuron), 17 insecticides (such as carbaryl, diazinon, and malathion), 15 breakdown products, and 6 others (Fuhrer et al. 2004). In agricultural drainages, insecticides were detected in 80% of samples and herbicides were present in 91%. They were also detected in mixed land-use streams – 71% and 90 %, respectively. The most frequently detected pesticides were 2,4-D, terbacil, azinphos methyl, atrazine, carbaryl, and deethylatrazine. Generally, compounds were detected in tributaries more often than in the Yakima River itself. Diuron was not detected in samples from the Yakima River, but was found in 23% of tributary samples. The exception to this trend was 2,4-D, which was found in 67% of Yakima River samples, but only 59% of tributary samples.

Ninety-one percent of the samples collected from the small agricultural watersheds contained at least two pesticides or pesticide breakdown products. Samples contained a median of 8 and a maximum of 26 chemicals (Fuhrer, Morace et al. 2004). The herbicide 2,4-D, occurred most often in the mixtures, along with azinphos methyl, the most heavily applied pesticide, and atrazine, one of the most aquatic mobile pesticides (Fuhrer, Morace et al. 2004). 2,4-D was detected in over 80% of samples, and diuron was detected in over 30% of samples. Additionally, roughly 30% of samples contained both a.i.s. Linuron was screened for but not detected. The most frequently detected pesticides in the Yakima River Basin are total DDTs, dichloro-diphenyl-dichloroethane (DDD), and dieldrin (Johnson and Newman 1983; Joy 2002; Joy and Madrone 2002; Fuhrer, Morace et al. 2004). Nevertheless, concentrations of total DDT in water have decreased since 1991. These reductions are attributed to erosion-controlling best management practices (BMPs).

Another study conducted by the USGS between May 1999 and January 2000 in the surface waters of Yakima Basin detected 25 pesticide compounds (Ebbert and Embry 2001). Atrazine was the most widely detected herbicide and azinphos methyl was the most widely detected insecticide. Other detected compounds include simazine, terbacil, trifluralin; deethylatrazine, carbaryl, diazinon, malathion, and DDE. Linuron was the only chemical monitored for in this study that was assessed in this Opinion; it was only detected once.

Central Columbia Plateau: NAWQA Analysis

The Central Columbia Plateau is a prominent apple growing region. The USGS sampled 31 surface-water sites representing agricultural land use, with different crops, irrigation methods, and other agricultural practices for pesticides in Idaho and Washington from 1992 - 1995 (Williamson, Munn et al. 1998). Pesticides were detected in samples from all sites, except for the Palouse River at Laird Park (a headwaters site in a forested area). Many pesticides were detected in surface water at very low concentrations. Concentrations of six pesticides exceeded freshwater-chronic criteria for the protection of

aquatic life in one or more surface-water samples. They include the herbicide triallate and five insecticides (azinphos methyl, chlorpyrifos, diazinon, *gamma*-HCH, and parathion). All four herbicides addressed in this Opinion were detected in samples from this region, though at different frequencies (Williamson, Munn et al. 1998). 2,4-D was detected in 27% of samples, diuron in 20%, linuron in 3%, and triclopyr <1%.

Detections at four sites were high, ranging from 12 to 45 pesticides. The two sites with the highest detection frequencies are in the Quincy-Pasco subunit, where irrigation and high chemical use combine to increase transport of pesticides to surface waters. Pesticide detection frequencies at sites in the dryland farming (non-irrigated) areas of the North-Central and Palouse subunits are below the national median for NAWQA sites. All four sites had at least one pesticide concentration that exceeded a water-quality standard or guideline.

Concentrations of organochlorine pesticides and PCBs are higher than the national median (50th percentile) at seven of 11 sites; four sites were in the upper 25% of all NAWQA sites. Although most of these compounds have been banned, they still persist in the environment. Elevated concentrations were observed in dryland farming areas and irrigated areas.

Willamette Basin: NAWQA Analysis

From 1991 to 1995, the USGS also sampled surface waters in the Willamette Basin, Oregon. Wentz *et al.* (1998) reported that 50 pesticides and pesticide degradates of the 86 were detected in streams. Atrazine, simazine, metolachlor, deethylatrazine, diuron, and diazinon were detected in more than one-half of stream samples (Wentz, Bonn et al. 1998). Diuron was found in 53% of samples with a maximum concentration of 14 µg/L. The other herbicides assessed in this Opinion were detected less frequently: 2,4-D in 12%, triclopyr in 9%, and linuron in 1% of samples. The highest pesticide concentrations generally occurred in streams draining predominately agricultural land. Forty-nine pesticides were detected in streams draining predominantly agricultural land. About 25

pesticides were detected in streams draining mostly urban areas.

Lower Clackamas River Basin: NAWQA Analysis

Carpenter *et al.* (Carpenter, Sobieszczyk et al. 2008) summarized four different studies that monitored pesticide levels in the lower Clackamas River from 2000 to 2005. Water samples were collected from sites in the lower mainstem Clackamas River, its tributaries, and in pre- and post-treatment drinking-water. In all, 63 pesticide compounds (33 herbicides, 15 insecticides, 6 fungicides, and 9 degradates) were detected in samples collected during storm and nonstorm conditions. Fifty-seven pesticides or degradates were detected in the tributaries (mostly during storms), whereas fewer compounds (26) were detected in samples of source water from the lower mainstem Clackamas River, with fewest (15) occurring in drinking water. The two most commonly detected pesticides were the triazine herbicide simazine and atrazine, which occurred in about one-half of samples. The a.i. in common household herbicides RoundUP (glyphosate) and Cross bow (triclopyr and 2,4-D) were frequently detected together. All four herbicides addressed in this Opinion were detected in samples throughout the study area (Table 66).

Table 66. Summarized detection information from (Carpenter, Sobieszczyk et al. 2008). Note that percentages aren't comparable because results were pooled from multiple sources.

Pesticide	Percent Detection	Tributary Detections	Clackamas River Detections
2,4-D	35	28	4
Triclopyr	22	20	1
Diuron	44	22	15
Linuron	2	1	0

Upper Snake River Basin: NAWQA Analysis

The USGS conducted a water quality study from 1992 - 1995 in the upper Snake River basin, Idaho and Wyoming (Clark, Maret et al. 1998). This basin does not overlap with any of the 28 ESU/DPSs, though it does feed into the migratory corridor of all Snake River species, and eventually into the Columbia River. In basin wide stream sampling in

May and June 1994, Eptam, atrazine (and desethylatrazine), metolachlor, and alachlor were the most commonly detected pesticides. These compounds accounted for 75% of all detections. Seventeen different pesticides were detected downstream from American Falls Reservoir. 2,4-D was present in 14% of samples, while diuron was found in 1%. Triclopyr and linuron were screened for but not detected (Clark, Maret et al. 1998).

Hood River Basin

The Hood River Basin ranks fourth in the state of Oregon in total agricultural pesticide usage (Jenkins, Jepson et al. 2004). The land in Hood River basin is used to grow five crops: alfalfa, apples, cherries, grapes, and pears. About 61 a.i.s, totaling 1.1 million lbs, are applied annually to roughly 21,000 acres. Of the top nine, three are carbamates and three are organophosphate insecticides (Table 67).

Table 67. Summarized detection information from (Carpenter, Sobieszczyk et al. 2008). Note that percentages aren't comparable because results were pooled from multiple sources.

Active Ingredient	Class	Lbs applied
Oil	-	624,392
Lime Sulfur	-	121,703
Mancozeb	Carbamate	86,872
Sulfur	-	60,552
Ziram	Carbamate	45,965
Azinphos methyl	Organo-phosphate	22,294
Metam-Sodium	Carbamate	17,114
Phosmet	Organo-phosphate	15,919
Chlorpyrifos	Organo-phosphate	14,833

The Hood River basin contains approximately 400 miles of perennial stream channel, of which an estimated 100 miles is accessible to anadromous fish. These channels are important rearing and spawning habitat for salmonids, making pesticide drift a major concern for the area.

Other Land Use in the Pacific Northwest Region

Urban and Industrial Development

The largest urban area in the basin is the greater Portland metropolitan area, located at the mouth of the Willamette River. Portland's population exceeds 500,000 (Hinck, Schmitt et al. 2004). Although the basin's land cover is about 8% of the U.S. total land mass, its human population is one-third the national average (about 1.2% of the U.S. population) (Hinck, Schmitt et al. 2004).

Discharges from sewage treatment plants, paper manufacturing, and chemical and metal production represent the top three permitted sources of contaminants within the lower basin according to discharge volumes and concentrations (Rosetta and Borys 1996). Rosetta and Borys (1996) review of 1993 data indicate that 52% of the point source waste water discharge volume is from sewage treatment plants, 39% from paper and allied products, 5% from chemical and allied products, and 3% from primary metals. However, the paper and allied products industry are the primary sources of the suspended sediment load (71%). Additionally, 26% of the point source waste water discharge volume comes from sewage treatment plants and 1% is from the chemical and allied products industry. Nonpoint source discharges (urban stormwater runoff) account for significant pollutant loading to the lower basin, including most organics and over half of the metals. Although rural nonpoint sources contributions were not calculated, Rosetta and Borys (1996) surmised that in some areas and for some contaminants, rural areas may contribute a large portion of the nonpoint source discharge. This is particularly true for pesticide contamination in the upper river basin where agriculture is the predominant land use.

Water quality has been reduced by phosphorus loads and decreased water clarity, primarily along the lower and middle sections of the Columbia River Estuary. Although sediment quality is generally very good, benthic indices have not been established within the estuary. Fish tissue contaminant loads (PCBs, DDT, DDD, DDE, and mercury) are high and present a persistent and long lasting effect on estuary biology. Health advisories

have been recently issued for people eating fish in the area that contain high levels of dioxins, PCBs, and pesticides.

Habitat Modification

This section briefly describes how anthropogenic land use has altered aquatic habitat conditions for salmonids in the Pacific Northwest Region. Basin wide, critical ecological connectivity (mainstem to tributaries and riparian floodplains) has been disconnected by dams and associated activities such as floodplain deforestation and urbanization. Dams have flooded historical spawning and rearing habitat with the creation of massive water storage reservoirs. More than 55% of the Columbia River Basin that was accessible to salmon and steelhead before 1939 has been blocked by large dams (NWPPC 1986). Construction of the Grand Coulee Dam blocked 1,000 miles (1,609 km) of habitat from migrating salmon and steelhead (Wydoski and Whitney 1979). Similarly, over one third (2,000 km) of coho salmon habitat is no longer accessible (Good, Waples et al. 2005). The mainstem habitats of the lower Columbia and Willamette rivers have been reduced primarily to a single channel. As a result, floodplain area is reduced, off-channel habitat features have been eliminated or disconnected from the main channel, and the amount of LWD in the mainstem has been reduced. Remaining areas are affected by flow fluctuations associated with reservoir management for power generation, flood control, and irrigation. Overbank flow events, important to habitat diversity, have become rare as a result of controlling peak flows and associated revetments. Portions of the basin are also subject to impacts from cattle grazing and irrigation withdrawals. Consequently, estuary dynamics have changed substantially.

Habitat loss has fragmented habitat and human density increase has created additional loads of pollutants and contaminants within the Columbia River Estuary (Anderson, Dugger et al. 2007). About 77% of swamps, 57% of marshes, and over 20% of tree cover have been lost to development and industry. Twenty four threatened and endangered species occur in the estuary, some of which are recovering while others (*i.e.*, Chinook salmon) are not.

Stream habitat degradation in Columbia Central Plateau is relatively high (Williamson, Munn et al. 1998). In the most recent NAWQA survey, a total of 16 sites were evaluated - all of which showed signs of degradation (Williamson, Munn et al. 1998). Streams in this area have an average of 20% canopy cover and 70% bank erosion. These factors have severely affected the quality of habitat available to salmonids. The Palouse subunit of the Lower Snake River exceeds temperature levels for the protection of aquatic life (Williamson, Munn et al. 1998).

The Willamette Basin Valley has been dramatically changed by modern settlement. The complexity of the mainstem river and extent of riparian forest have both been reduced by 80% (PNERC 2002). About 75% of what was formerly prairie and 60% of what was wetland have been converted to agricultural purposes. These actions, combined with urban development, extensive (96 miles) bank stabilization, and in-river and nearshore gravel mining, have resulted in a loss of floodplain connectivity and off-channel habitat (PNERC 2002).

Habitat Restoration

Since 2000, land management practices included improving access by replacing culverts and fish habitat restoration activities at Federal Energy Regulatory Commission (FERC)-licensed dams. Habitat restoration in the upper (reducing excess sediment loads) and lower Grays River watersheds may benefit the Grays River chum salmon population as it has a sub-yearling juvenile life history type and rears in such habitats. Short-term daily flow fluctuations at Bonneville Dam sometimes create a barrier (*i.e.*, entrapment on shallow sand flats) for fry moving into the mainstem rearing and migration corridor. Some chum fry have been stranded on shallow water flats on Pierce Island from daily flow fluctuations. Coho salmon are likely to be affected by flow and sediment delivery changes in the Columbia River plume. Steelhead may be affected by flow and sediment delivery changes in the plume (Casillas 1999).

In 2006, NOAA Fisheries completed consultation on issuance of a 50-year incidental take permit to the State of Washington for its Washington State Forest Practices Habitat Conservation Plan (HCP). The HCP is expected to improve habitat conditions on state forest lands within the action area. Improvements include removing barriers to migration, restoring hydrologic processes, increasing the number of large trees in riparian zones, improving stream bank integrity, and reducing fine sediment inputs (NMFS 2008).

Mining

Most of the mining in the basin is focused on minerals such as phosphate, limestone, dolomite, perlite, or metals such as gold, silver, copper, iron, and zinc. Mining in the region is conducted in a variety of methods and places within the basin. Alluvial or glacial deposits are often mined for gold or aggregate. Ores are often excavated from the hard bedrocks of the Idaho batholiths. Eleven percent of the nation's output of gold has come from mining operations in Washington, Montana, and Idaho. More than half of the nation's silver output has come from a few select silver deposits.

Many of the streams and river reaches in the basin are impaired from mining. Several abandoned and former mining sites are also designated as superfund cleanup areas (Stanford, Hauer et al. 2005; Anderson, Dugger et al. 2007). According to the U.S. Bureau of Mines, there are about 14,000 inactive or abandoned mines within the Columbia River Basin. Of these, nearly 200 pose a potential hazard to the environment [Quigley, 1997 *in* (Hinck, Schmitt et al. 2004)]. Contaminants detected in the water include lead and other trace metals.

Hydromodification Projects

More than 400 dams exist in the basin, ranging from mega dams that store large amounts of water to small diversion dams for irrigation (Figure 54). Every major tributary of the Columbia River except the Salmon River is totally or partially regulated by dams and diversions. More than 150 dams are major hydroelectric projects. Of these, 18 dams are

located on the mainstem Columbia River and its major tributary, the Snake River. The FCRPS encompasses the operations of 14 major dams and reservoirs on the Columbia and Snake rivers. These dams and reservoirs operate as a coordinated system. The Corps operates 9 of 10 major federal projects on the Columbia and Snake rivers, and the Dworshak, Libby and Albeni Falls dams. The BOR operates the Grand Coulee and Hungry Horse dams. These federal projects are a major source of power in the region. These same projects provide flood control, navigation, recreation, fish and wildlife, municipal and industrial water supply, and irrigation benefits.

BOR has operated irrigation projects within the basin since 1904. The irrigation system delivers water to about 2.9 million acres of agricultural lands. About 1.1 million acres of land are irrigated using water delivered by two structures, the Columbia River Project (Grand Coulee Dam) and the Yakima Project. The Grand Coulee Dam delivers water for the irrigation of over 670,000 acres of croplands and the Yakima Project delivers water to nearly 500,000 acres of croplands (Bouldin, Farris et al. 2007).

The Bonneville Power Administration (BPA), an agency of the U.S. Department of Energy, wholesales electric power produced at 31 federal dams (67% of its production) and non-hydropower facilities in the Columbia-Snake Basin. The BPA sells about half the electric power consumed in the Pacific Northwest. The federal dams were developed over a 37-year period starting in 1938 with Bonneville Dam and Grand Coulee in 1941, and ending with construction of Libby Dam in 1973 and Lower Granite Dam in 1975.

Development of the Pacific Northwest regional hydroelectric power system, dating to the early 20th century, has had profound effects on the ecosystems of the Columbia River Basin (ISG 1996). These effects have been especially adverse to the survival of anadromous salmonids. The construction of the FCRPS modified migratory habitat of adult and juvenile salmonids. In many cases, the FCRPS presented a complete barrier to habitat access for salmonids. Approximately 80% of historical spawning and rearing habitat of Snake River fall-run Chinook salmon is now inaccessible due to dams. The

Snake River spring/summer run has been limited to the Salmon, Grande Ronde, Imnaha, and Tuscanon rivers. Damming has cut off access to the majority of Snake River Chinook salmon spawning habitat. The Sunbeam Dam on the Salmon River is believed to have limited the range of Snake River sockeye salmon as well.

Both upstream and downstream migrating fish are impeded by the dams. Additionally, a substantial number of juvenile salmonids are killed and injured during downstream migrations. Physical injury and direct mortality occurs as juveniles pass through turbines, bypasses, and spillways. Indirect effects of passage through all routes may include disorientation, stress, delay in passage, exposure to high concentrations of dissolved gases, warm water, and increased predation. Non-federal hydropower facilities on Columbia River tributaries have also partially or completely blocked higher elevation spawning.

Qualitatively, several hydromodification projects have improved the productivity of naturally produced SR Fall-run Chinook salmon. Improvements include flow augmentation to enhance water flows through the lower Snake and Columbia Rivers [USBR 1998 *in* (NMFS 2008)]; providing stable outflows at Hells Canyon Dam during the fall Chinook salmon spawning season and maintaining these flows as minimums throughout the incubation period to enhance survival of incubating fall-run Chinook salmon; and reduced summer temperatures and enhanced summer flow in the lower Snake River [see (Corps, BPA et al. 2007), *Appendix 1 in* (NMFS 2008)]. Providing suitable water temperatures for over-summer rearing within the Snake River reservoirs allows the expression of productive “yearling” life history strategy that was previously unavailable to SR Fall-run Chinook salmon.

The mainstem FCRPS corridor has also improved safe passage through the hydrosystem for juvenile steelhead and yearling Chinook salmon with the construction and operation of surface bypass routes at Lower Granite, Ice Harbor, and Bonneville dams and other configuration improvements (Corps, BPA et al. 2007).

For salmon, with a stream-type juvenile life history, projects that have protected or restored riparian areas and breached or lowered dikes and levees in the tidally influenced zone of the estuary have improved the function of the juvenile migration corridor. The FCRPS action agencies recently implemented 18 estuary habitat projects that removed passage barriers. These activities provide fish access to good quality habitat.

The Corps *et al.* (2007) estimated that hydropower configuration and operational improvements implemented from 2000 to 2006 have resulted in an 11.3% increase in survival for yearling juvenile LCR Chinook salmon from populations that pass Bonneville Dam. Improvements during this period included the installation of a corner collector at Powerhouse II (PH2) and the partial installation of minimum gap runners at Powerhouse 1 (PH1) and of structures that improve fish guidance efficiency at PH2. Spill operations have been improved and PH2 is used as the first priority powerhouse for power production because bypass survival is higher than at PH1. Additionally, drawing water towards PH2 moves fish toward the corner collector. The bypass system screen was removed from PH1 because tests showed that turbine survival was higher than through the bypass system at that location.

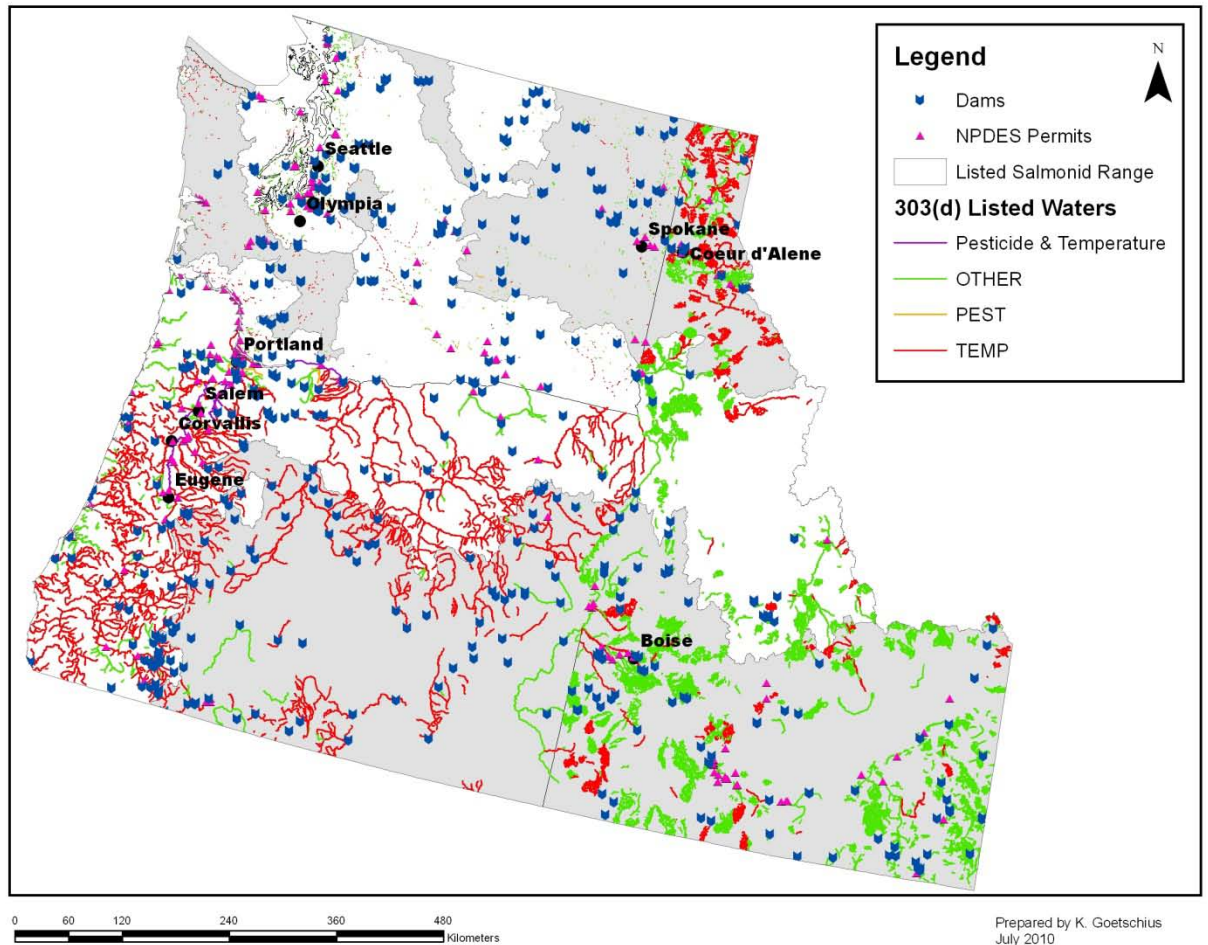


Figure 54. Pacific Northwest 303(d) waters, dams, and NPDES permit sites.

Artificial Propagation

There are several artificial propagation programs for salmon production within the Columbia River Basin. These programs were instituted under federal law to lessen the effects of lost natural salmon production within the basin from the dams. Federal, state, and tribal managers operate the hatcheries. For more than 100 years, hatcheries in the Pacific Northwest have been used to produce fish for harvest and replace natural production lost to dam construction. Hatcheries have only minimally been used to protect and rebuild naturally produced salmonid populations (*e.g.*, Redfish Lake sockeye salmon). In 1987, 95% of the coho salmon, 70% of the spring Chinook salmon, 80% of the summer Chinook salmon, 50% of the fall-run Chinook salmon, and 70% of the steelhead returning to the Columbia River Basin originated in hatcheries (CBFWA 1990). More recent estimates suggest that almost half of the total number of smolts produced in the basin come from hatcheries (Beechie, Liermann et al. 2005).

The impact of artificial propagation on the total production of Pacific salmon and steelhead has been extensive (Hard, Jones et al. 1992). Hatchery practices, among other factors, are a contributing factor to the 90% reduction in natural coho salmon runs in the lower Columbia River over the past 30 years (Flagg, Waknitz et al. 1995). Past hatchery and stocking practices have resulted in the transplantation of salmon and steelhead from non-native basins. The impacts of these hatchery practices are largely unknown. Adverse effects of these practices likely included: loss of genetic variability within and among populations (Busack 1990; Riggs 1990; Hard, Jones et al. 1992; Reisenbichler 1997), disease transfer, increased competition for food, habitat, or mates, increased predation, altered migration, and the displacement of natural fish (Steward and Bjornn 1990; Hard, Jones et al. 1992; Fresh 1997). Species with extended freshwater residence may face higher risk of domestication, predation, or altered migration than species that spend only a brief time in freshwater (Hard, Jones et al. 1992). Nonetheless, artificial propagation may also contribute to the conservation of listed salmon and steelhead.

However, it is unclear whether or how much artificial propagation during the recovery process will compromise the distinctiveness of natural populations (Hard, Jones et al. 1992).

The states of Oregon and Washington and other fisheries co-managers are engaged in a substantial review of hatchery management practices through the Hatchery Scientific Review Group (HSRG). The HSRG was established and funded by Congress to provide an independent review of current hatchery program in the Columbia River Basin. The HSRG has completed its work on Lower Columbia River populations and provided its recommendations. A general conclusion is that the current production programs are inconsistent with practices that reduce impacts on naturally-spawning populations, and will have to be modified to reduce adverse effects on key natural populations identified in the Interim Recovery Plan. The adverse effects are caused by hatchery-origin adults spawning with natural-origin fish or competing with natural-origin fish for spawning sites (NMFS 2008). Oregon and Washington initiated a comprehensive program of hatchery and associated harvest reforms (Washington Department of Fish and Wildlife (WDFW) 2005; ODFW 2007). The program is designed to achieve HSRG objectives related to controlling the number of hatchery-origin fish on the spawning grounds and in the hatchery broodstock.

Coho salmon hatchery programs in the lower Columbia have been tasked to compensate for impacts of fisheries. However, hatchery programs in the LCR have not operated specifically to conserve LCR coho salmon. These programs threaten the viability of natural populations. The long-term domestication of hatchery fish has eroded the fitness of these fish in the wild and has reduced the productivity of wild stocks where significant numbers of hatchery fish spawn with wild fish. Large numbers of hatchery fish have also contributed to more intensive mixed stock fisheries. These programs largely overexploited wild populations weakened by habitat degradation. Most LCR coho salmon populations have been heavily influenced by hatchery production over the years.

Commercial, Recreational, and Subsistence Fishing

Despite regulated fishing programs for salmonids, listed salmonids are also caught as bycatch. There are several approaches under the ESA to address tribal and state take of ESA-listed species that may occur as a result of harvest activities. Section 10 of the ESA provides for permits to operate fishery harvest programs. ESA section 4(d) rules provide exemptions from take for resource, harvest, and hatchery management plans.

Furthermore, there are several treaties that have reserved the right of fishing to tribes in the North West Region.

Management of salmon fisheries in the Columbia River Basin is a cooperative process involving federal, state, and tribal representatives. The Pacific Fishery Management Council sets annual fisheries in federal waters from three to 200 miles off the coasts of Washington, Oregon, and California. Salmon and steelhead fisheries in the Columbia River and its tributaries are co-managed by the states of Washington, Oregon, Idaho, four treaty tribes, and other tribes that traditionally have fished in those waters. A federal court oversees Columbia River harvest management through the U.S. v. Oregon proceedings. Inland fisheries are those in waters within state boundaries, including those extending out three miles from the coasts. The states of Oregon, Idaho, and Washington issue salmon fishing licenses for these areas.

Fisheries in the Columbia River basin are managed within the winter/spring, summer, and fall seasons. There are Treaty Indian and non-Treaty fisheries which are managed subject to state and tribal regulation, consistent with provisions of a U.S. v. Oregon 2008 agreement. The winter/spring season extends from January 1 to June 15. Commercial, recreational, and ceremonial subsistence fisheries target primarily upriver spring Chinook stocks and spring Chinook salmon that return to the Willamette and lower Columbia River tributaries. Some steelhead are also caught incidentally in these fisheries. The summer season extends from June 16 to July 31. Commercial, recreational, and ceremonial and subsistence fisheries are managed primarily to provide harvest opportunity directed at unlisted UCR summer Chinook salmon. Summer fisheries are

constrained primarily by the available opportunity for UCR summer Chinook salmon, and by specific harvest rate limits for SR sockeye salmon and harvest rate limits on steelhead in non-Treaty fisheries. Fall season fisheries begin on August 1 and end on December 31. Commercial, recreational, and ceremonial and subsistence fisheries target primarily harvestable hatchery and natural origin fall Chinook and coho salmon. Fall season fisheries are constrained by specific ESA related harvest rate limits for listed SR fall Chinook salmon, and SR steelhead.

Treaty Indian fisheries are managed subject to the regulation of the Columbia River Treaty Tribes. They include all mainstem Columbia River fisheries between Bonneville Dam and McNary Dam, and any fishery impacts from tribal fishing that occurs below Bonneville Dam. Tribal fisheries within specified tributaries to the Columbia River are included.

Non-Treaty fisheries are managed under the jurisdiction of the states. These include mainstem Columbia River commercial and recreational salmonid fisheries at the river mouth of Bonneville Dam, designated off channel Select Area fisheries, mainstem recreational fisheries between Bonneville Dam and McNary Dam, recreational fisheries between McNary Dam and Highway 305 Bridge in Pasco, Washington, recreational and Wanapum tribal spring Chinook fisheries from McNary Dam to Priest Rapids Dam, and recreational spring Chinook fisheries in the Snake River upstream to Lower Granite Dam.

Archeological records indicate that indigenous people caught salmon in the Columbia River more than 7,000 years ago. One of the most well known tribal fishing sites within the basin was located near Celilo Falls, an area in the lower river that has been occupied by Dalles Dam since 1957. Salmon fishing increased with better fishing methods and preservation techniques, such as drying and smoking. Salmon harvest substantially increased in the mid-1800s with canning techniques. Harvest techniques also changed over time, from early use of hand-held spears and dip nets, to riverboats using seines and gill nets. Harvest techniques eventually transitioned to large ocean-going vessels with

trolling gear and nets and the harvest of Columbia River salmon and steelhead from California to Alaska (Beechie, Liermann et al. 2005).

During the mid-1800s, an estimated 10 to 16 million adult salmon of all species entered the Columbia River each year. Large annual harvests of returning adult salmon during the late 1800s ranging from 20 million to 40 million lbs of salmon and steelhead significantly reduced population productivity (Beechie, Liermann et al. 2005). The largest known harvest of Chinook salmon occurred in 1883 when Columbia River canneries processed 43 million lbs of salmon (Lichatowich 1999). Commercial landings declined steadily from the 1920s to a low in 1993. At that time, just over one million lbs of Chinook salmon were harvested (Beechie, Liermann et al. 2005).

Harvested and spawning adults reached 2.8 million in the early 2000s, of which almost half are hatchery produced (Beechie, Liermann et al. 2005). Most of the fish caught in the river are steelhead and spring/summer run Chinook salmon. Ocean harvest consists largely of coho and fall-run Chinook salmon. Most ocean catches are made north of Cape Falcon, Oregon. Over the past five years, the number of spring and fall salmon commercially harvested in tribal fisheries has averaged between 25,000 and 110,000 fish (Beechie, Liermann et al. 2005). Recreational catch in both ocean and in-river fisheries varies from 140,000 to 150,000 individuals (Beechie, Liermann et al. 2005).

Non-Indian fisheries in the lower Columbia River are limited to a harvest rate of 1%. Treaty Indian fisheries are limited to a harvest rate of 5 to 7%, depending on the run size of upriver Snake River sockeye stocks. Actual harvest rates over the last 10 years have ranged from 0 to 0.9%, and 2.8 to 6.1%, respectively [see TAC 2008, Table 15 *in* (NMFS 2008)].

Columbia River chum salmon are not caught incidentally in tribal fisheries above Bonneville Dam. However, Columbia River chum salmon are incidentally caught occasionally in non-Indian fall season fisheries below Bonneville Dam. There are no

fisheries in the Columbia River that target hatchery or natural-origin chum salmon. The species' later fall return timing make them vulnerable to relatively little potential harvest in fisheries that target Chinook salmon and coho salmon. CR chum salmon rarely take the sport gear used to target other species. Incidental catch of chum amounts to a few tens of fish per year (TAC 2008). The harvest rate of CR chum salmon in proposed state fisheries in the lower river is estimated to be 1.6% per year and is less than 5%.

LCR coho salmon are harvested in the ocean and in the Columbia River and tributary freshwater fisheries of Oregon and Washington. Incidental take of coho salmon prior to the 1990s fluctuated from approximately 60 to 90%. However, this number has been reduced since its listing to 15 to 25% (LCFRB 2004). The exploitation of hatchery coho salmon has remained approximately 50% through the use of selective fisheries.

LCR steelhead are harvested in Columbia River and tributary freshwater fisheries of Oregon and Washington. Fishery impacts of LCR steelhead have been limited to less than 10% since implementation of mark-selective fisheries during the 1980s. Recent harvest rates on UCR steelhead in non-Treaty and treaty Indian fisheries ranged from 1% to 2%, and 4.1% to 12.4%, respectively (NMFS 2008).

Non-native Species

Many non-native species have been introduced to the Columbia River Basin since the 1880s. At least 81 non-native species have currently been identified, composing one-fifth of all species in some areas. New non-native species are discovered in the basin regularly; a new aquatic invertebrate is discovered approximately every 5 months (Sytsma, Cordell et al. 2004). It is clear that the introduction of non-native species has changed the environment, though whether these changes will impact salmonid populations is uncertain (Sytsma, Cordell et al. 2004).

Puget Sound Region

Puget Sound is the second largest estuary in the U.S. It has about 1,330 miles of

shoreline and extends from the mouth of the Strait of Juan de Fuca east. Puget Sound includes the San Juan Islands and south to Olympia, and is fed by more than 10,000 rivers and streams.

Puget Sound is generally divided into four major geographic marine basins: Hood Canal, South Sound, Whidbey Basin, and the Main Basin. The Main Basin has been further subdivided into two subbasins: Admiralty Inlet and Central Basin. About 43% of the Puget Sound's tideland is located in the Whidbey Island Basin. This reflects the large influence of the Skagit River, which is the largest river in the Puget Sound system and whose sediments are responsible for the extensive mudflats and tidelands of Skagit Bay.

Habitat types that occur within the nearshore environment include eelgrass meadows, kelp forest, mud flats, tidal marshes, sub-estuaries (tidally influenced portions of river and stream mouths), sand spits, beaches and backshore, banks and bluffs, and marine riparian vegetation. These habitats provide critical functions such as primary food production and support habitat for invertebrates, fish, birds, and other wildlife.

Major rivers draining to Puget Sound from the Cascade Mountains include the Skagit, Snohomish, Nooksack, Puyallup, and Green rivers, as well as the Lake Washington/Cedar River watershed. Major rivers from the Olympic Mountains include the Hamma Hamma, the Duckabush, the Quilcene, and the Skokomish rivers. Numerous other smaller rivers drain to the Sound, many of which are significant salmonid production areas despite their small size.

The Puget Sound basin is home to more than 200 fish and 140 mammalian species. Salmonids within the region include coho, Chinook, sockeye, chum, and pink salmon, kokanee, steelhead, rainbow, cutthroat, and bull trout (Wydoski and Whitney 1979; Kruckeberg 1991). Important commercial fishes include the five Pacific salmon and several rockfish species. A number of introduced species occur within the region, including brown and brook trout, Atlantic salmon, bass, tunicates (sea squirts), and a

saltmarsh grass (*Spartina* spp.). Estimates suggest that over 90 species have been intentionally or accidentally introduced in the region (Ruckelshaus and McClure 2007). At present, over 40 species in the region are listed as threatened and endangered under the ESA.

Puget Sound is unique among the nation's estuaries as it is a deep fjord-like structure that contains many urban areas within its drainage basin (Collier, O'Neill et al. 2006).

Because several sills limit entry of oceanic water into Puget Sound, it is relatively poorly flushed compared to other urbanized estuaries of North America. Thus, toxic chemicals that enter Puget Sound have longer residence times within the system. This entrainment of toxics can result in biota exposure to increased levels of contaminant for a given input, compared to other large estuaries. This hydrologic isolation puts the Puget Sound ecosystem at higher risk from other types of populations that enter the system, such as nutrients and pathogens.

Because Puget Sound is a deep, almost oceanic habitat, the tendency of a number of species to migrate outside of Puget Sound is limited relative to similar species in other large urban estuaries. This high degree of residency for many marine species, combined with the poor flushing of Puget Sound, results in a more protracted exposure to contaminants. The combination of hydrologic and biological isolation makes the Puget Sound ecosystem highly susceptible to inputs of toxic chemicals compared to other major estuarine ecosystems (Collier, O'Neill et al. 2006).

An indication of this sensitivity occurs in Pacific herring, one of Puget Sound's keystone forage fish species (Collier, O'Neill et al. 2006). These fish spend almost all of their lives in pelagic waters and feed at the lower end of the food chain. Pacific herring should be among the least contaminated of fish species. However, monitoring has shown that herring from the main basins of Puget Sound have higher body burdens of persistent chemicals (*e.g.*, PCBs) compared to herring from the severely contaminated Baltic Sea. Thus, the pelagic food web of Puget Sound appears to be more seriously contaminated

than previously anticipated.

Chinook salmon that are resident in Puget Sound (a result of hatchery practices and natural migration patterns) are several times more contaminated with persistent bioaccumulative contaminants than other salmon populations along the West Coast (Collier, O'Neill et al. 2006). Because of associated human health concerns, fish consumption guidelines for Puget Sound salmon are under review by the Washington State Department of Health.

Extremely high levels of chemical contaminants are also found in Puget Sound's top predators, including harbor seals and ESA-listed southern resident killer whales (Collier, O'Neill et al. 2006). In addition to carrying elevated loads of toxic chemicals in their tissues, Puget Sound's biota also show a wide range of adverse health outcomes associated with exposure to chemical contaminants. They include widespread cancer and reproductive impairment in bottom fish, increased susceptibility to disease in juvenile salmon, acute die-offs of adult salmon returning to spawn in urban watersheds, and egg and larval mortality in a variety of fish. Given current regional projections for population growth and coastal development, the loadings of chemical contaminants into Puget Sound will increase dramatically in future years.

Land Use

The Puget Sound Lowland contains the most densely populated area of Washington. The regional population in 2003 was an estimated 3.8 million people, with 86% residing in King, Pierce, and Snohomish counties (Snohomish, Cedar-Sammamish Basin, Green-Duwamish, and Puyallup River watersheds). The area is expected to attract 4 to 6 million new human residents in the next 20 years (Ruckelshaus and McClure 2007). The Snohomish River watershed, one of the fastest growing watersheds in the region, increased about 16% in the same period.

Land use in the Puget Sound lowland is composed of agricultural areas (including forests for timber production), urban areas (industrial and residential use), and rural areas (low density residential with some agricultural activity). Pesticides are regularly applied to agricultural and non-agricultural lands and are found virtually in every land use area. Pesticides and other contaminants drain into ditches in agricultural areas and eventually to stream systems. Roads bring surface water runoff to stream systems from industrial, residential, and landscaped areas in the urban environment. Pesticides are also typically found in the right-of-ways of infrastructure that connect the major landscape types. Right-of-ways are associated with roads, railways, utility lines, and pipelines.

In the 1930s, all of western Washington contained about 15.5 million acres of “harvestable” forestland. By 2004, the total acreage was nearly half that originally surveyed (PSAT 2007). Forest cover in Puget Sound alone was about 5.4 million acres in the early 1990s. About a decade later, the region had lost another 200,000 acres of forest cover with some watersheds losing more than half the total forested acreage. The most intensive loss of forest cover occurred in the Urban Growth Boundary, which encompasses specific parts of the Puget Lowland. In this area, forest cover declined by 11% between 1991 and 1999 (Ruckelshaus and McClure 2007). Projected land cover changes indicate that trends are likely to continue over the next several decades with population changes (Ruckelshaus and McClure 2007). Coniferous forests are also projected to decline at an alarming rate as urban uses increase.

According to the 2001 State of the Sound report (PSAT 2007), impervious surfaces covered 3.3% of the region, with 7.3% of lowland areas (below 1,000 ft elevation) covered by impervious surfaces. From 1991 to 2001, the amount of impervious surfaces increased 10.4% region wide. Consequently, changes in rainfall delivery to streams alter stream flow regimes. Peak flows are increased and subsequent base flows are decreased and alter in-stream habitat. Stream channels are widened and deepened and riparian vegetation is typically removed which can cause increases in water temperature and will reduce the amounts of woody debris and organic matter to the stream system.

Pollutants carried into streams from urban runoff include pesticides, heavy metals, PCBs, polybrominated diphenyl ethers (PBDEs) compounds, PAHs, nutrients (phosphorus and nitrogen), and sediment (Table 68). Other ions generally elevated in urban streams include calcium, sodium, potassium, magnesium, and chloride ions where sodium chloride is used as the principal road deicing salt (Paul and Meyer 2001). The combined effect of increased concentrations of ions in streams is the elevated conductivity observed in most urban streams.

Table 68. Examples of Water Quality Contaminants in Residential and Urban Areas.

Contaminant groups	Select constituents	Select example(s)	Source and Use Information
Fertilizers	Nutrients	Phosphorus Nitrogen	lawns, golf courses, urban landscaping
Heavy Metals	Pb, Zn, Cr, Cu, Cd, Ni, Hg, Mg	Cu	brake pad dust, highway and parking lot runoff, rooftops
Pesticides including- Insecticides (I) Herbicides (H) Fungicides (F) Wood Treatment chemicals (WT) Legacy Pesticides (LP) Other ingredients in pesticide formulations (OI)	Organophosphates (I) Carbamates (I) Organochlorines (I) Pyrethroids (I) Triazines (H) Chloroacetanilides (H) Chlorophenoxy acids (H) Triazoles (F) Copper containing fungicides (F) Organochlorines (LP) Surfactants/adjuvants (OI)	Chlorpyrifos (I) Diazinon (I) Carbaryl (I) Atrazine (H) Esfenvalerate (I) Creosote (WT) DDT (LP) Copper sulfate (F) Metalaxyl (F) Nonylphenol (OI)	golf courses, right of ways, lawn and plant care products, pilings, bulkheads, fences
Pharmaceuticals and personal care products	Natural and synthetic hormones soaps and detergents	Ethinyl estradiol Nonylphenol	hospitals, dental facilities, residences, municipal and industrial waste water discharges
Polyaromatic hydrocarbons (PAHs)	Tricyclic PAHs	Phenanthrene	fossil fuel combustion, oil and gasoline leaks, highway runoff, creosote-treated wood
Industrial chemicals	PCBs PBDEs Dioxins	Penta-PBDE	utility infrastructure, flame retardants, electronic equipment

Many other metals have been found in elevated concentrations in urban stream sediments including arsenic, iron, boron, cobalt, silver, strontium, rubidium, antimony, scandium, molybdenum, lithium, and tin (Wheeler, Angermeier et al. 2005). The concentration, storage, and transport of metals in urban streams are connected to particulate organic matter content and sediment characteristics. Organic matter has a high binding capacity for metals and both bed and suspended sediments with high organic matter content frequently exhibit 50 - 7,500 times higher concentrations of zinc, lead, chromium, copper, mercury, and cadmium than sediments with lower organic matter content.

Although urban areas occupy only 2% of the Pacific Northwest land base, the impacts of urbanization on aquatic ecosystems are severe and long lasting (Spence, Lomnický et al. 1996). O'Neill *et al.* (2006) found that Chinook salmon returning to Puget Sound had significantly higher concentrations of PCBs and PBDEs compared to other Pacific coast

salmon populations. Furthermore, Chinook salmon that resided in Puget Sound in the winter rather than migrate to the Pacific Ocean (residents) had the highest concentrations of persistent organic pollutants (POPs), followed by Puget Sound fish populations believed to be more ocean-reared. Fall-run Chinook salmon from Puget Sound have a more localized marine distribution in Puget Sound and the Georgia Basin than other populations of Chinook salmon from the west coast of North America. This ESU is more contaminated with PCBs (2 to 6 times) and PBDEs (5 to 17 times). O'Neill *et al.* (2006) concluded that regional body burdens of contaminants in Pacific salmon, and Chinook salmon in particular, could contribute to the higher levels of contaminants in federally-listed endangered southern resident killer whales.

Endocrine disrupting compounds are chemicals that mimic natural hormones, inhibit the action of hormones and/or alter normal regulatory functions of the immune, nervous and endocrine systems and can be discharged with treated effluent (King County 2002). Endocrine disruption has been attributed to DDT and other organochlorine pesticides, dioxins, PAHs, alkylphenolic compounds, phthalate plasticizers, naturally occurring compounds, synthetic hormones and metals. Natural mammalian hormones such as 17 β -estradiol, are also classified as endocrine disruptors. Both natural and synthetic mammalian hormones are excreted through the urine and are known to be present in wastewater discharges.

Jobling *et al.* (1995) reported that ten chemicals known to occur in sewage effluent interacted with the fish estrogen receptor by reducing binding of 17 β -estradiol to its receptor, stimulating transcriptional activity of the estrogen receptor or inhibiting transcription activity. Binding of the ten chemicals with the fish endocrine receptor indicates that the chemicals could be endocrine disruptors and forms the basis of concern about WWTP effluent and fish endocrine disruption.

Fish communities are impacted by urbanization (Wheeler, Angermeier et al. 2005). Urban stream fish communities have lower overall abundance, diversity, taxa richness

and are dominated by pollution tolerant species. Lead content in fish tissue is higher in urban areas. Furthermore, the proximity of urban streams to humans increases the risk of non-native species introduction and establishment. Thirty-nine non-native species were collected in Puget Sound during the 1998 Puget Sound Expedition Rapid Assessment Survey (Brennan, Higgins et al. 2004). Lake Washington, located within a highly urban area, has 15 non-native species identified (Ajawani 1956).

PAH compounds also have distinct and specific effects on fish at early life history stages (Incardona, Collier et al. 2004). PAHs tend to adsorb to organic or inorganic matter in sediments, where they can be trapped in long-term reservoirs (Johnson, Collier et al. 2002). Only a portion of sediment-adsorbed PAHs are readily bioavailable to marine organisms, but there is substantial uptake of these compounds by resident benthic fish through the diet, through exposure to contaminated water in the benthic boundary layer, and through direct contact with sediment. Benthic invertebrate prey are a particularly important source of PAH exposure for marine fishes, as PAHs are bioaccumulated in many invertebrate species (Varanasi, Stein et al. 1989; Varanasi, Stein et al. 1992; Meador, Stein et al. 1995).

PAHs and their metabolites in invertebrate prey can be passed on to consuming fish species, PAHs are metabolized extensively in vertebrates, including fishes (Johnson, Collier et al. 2002). Although PAHs do not bioaccumulate in vertebrate tissues, PAHs cause a variety of deleterious effects in exposed animals. Some PAHs are known to be immunotoxic and to have adverse effects on reproduction and development. Studies show that PAHs exhibit many of the same toxic effects in fish as they do in mammals (Johnson, Collier et al. 2002).

Habitat Modification

Much of the estuarine wetlands in Puget Sound have been heavily modified, primarily from agricultural land conversion and urban development (NRC 1996). Although most estuarine wetland losses result from conversions to agricultural land by ditching,

draining, or diking, these wetlands also experience increasing effects from industrial and urban causes. By 1980, an estimated 27,180 acres of intertidal or shore wetlands had been lost at 11 deltas in Puget Sound (Bortleson, Chrzastowski et al. 1980). Tidal wetlands in Puget Sound amount to roughly 18% of their historical extent (Collins and Sheikh 2005). Coastal marshes close to seaports and population centers have been especially vulnerable to conversion with losses of 50 - 90%. By 1980, an estimated 27,180 acres of intertidal or shore wetlands had been lost at eleven deltas in Puget Sound (Bortleson, Chrzastowski et al. 1980). More recently, tidal wetlands in Puget Sound amount to about 17 - 19% of their historical extent (Collins and Sheikh 2005). Coastal marshes close to seaports and population centers have been especially vulnerable to conversion with losses of 50 - 90% common for individual estuaries. Salmon use freshwater and estuarine wetlands for physiological transition to and from salt water and rearing habitat. The land conversions and losses of Pacific Northwest wetlands constitute a major impact. Salmon use marine nearshore areas for rearing and migration, with juveniles using shallow shoreline habitats (Brennan, Higgins et al. 2004).

About 800 miles of Puget Sound's shorelines are hardened or dredged (PSAT 2004; Ruckelshaus and McClure 2007). The area most intensely modified is the urban corridor (eastern shores of Puget Sound from Mukilteo to Tacoma). Here, nearly 80% of the shoreline has been altered, mostly from shoreline armoring associated with the Burlington Northern Railroad tracks (Ruckelshaus and McClure 2007). Levee development within the rivers and their deltas has isolated significant portions of former floodplain habitat that was historically used by salmon and trout during rising flood waters.

Urbanization has caused direct loss of riparian vegetation and soils and has significantly altered hydrologic and erosion rates. Watershed development and associated urbanization throughout the Puget Sound, Hood Canal, and Strait of Juan de Fuca regions have increased sedimentation, raised water temperatures, decreased LWD recruitment, decreased gravel recruitment, reduced river pools and spawning areas, and dredged and

filled estuarine rearing areas (Bishop and Morgan 1996 in (NMFS 2008)). Large areas of the lower rivers have been channelized and diked for flood control and to protect agricultural, industrial, and residential development.

The principal factor for decline of Puget Sound steelhead is the destruction, modification, and curtailment of its habitat and range. Barriers to fish passage and adverse effects on water quality and quantity resulting from dams, the loss of wetland and riparian habitats, and agricultural and urban development activities have contributed and continue to contribute to the loss and degradation of steelhead habitats in Puget Sound (NMFS 2008).

Industrial Development

More than 100 years of industrial pollution and urban development have affected water quality and sediments in Puget Sound. Many different kinds of activities and substances release contamination into Puget Sound and the contributing waters. According to the State of the Sound Report (PSAT 2007) in 2004, more than 1,400 fresh and marine waters in the region were listed as “impaired.” Almost two-thirds of these water bodies were listed as impaired due to contaminants, such as toxics, pathogens, and low dissolved oxygen or high temperatures, and less than one-third had established cleanup plans. More than 5,000 acres of submerged lands (primarily in urban areas; 1% of the study area) are contaminated with high levels of toxic substances, including polybrominated diphenyl ethers (PBDEs; flame retardants), and roughly one-third (180,000 acres) of submerged lands within Puget Sound are considered moderately contaminated. In 2005 the Puget Sound Action Team (PSAT) identified the primary pollutants of concern in Puget Sound and their sources listed below in Table 69.

Table 69. Pollutants of Concern in Puget Sound (PSAT 2005).

Pollutant	Sources
Heavy Metals: Pb, Hg, Cu, and others	vehicles, batteries, paints, dyes, stormwater runoff, spills, pipes.
Organic Compounds: Polycyclic aromatic hydrocarbons (PAHs)	Burning of petroleum, coal, oil spills, leaking underground fuel tanks, creosote, asphalt.
Polychlorinated biphenyls (PCBs)	Solvents electrical coolants and lubricants, pesticides, herbicides, treated wood.
Dioxins, Furans	Byproducts of industrial processes.
Dichloro-diphenyl-trichloroethane (DDTs)	Chlorinated pesticides.
Phthalates	Plastic materials, soaps, and other personal care products. Many of these compounds are in wastewater from sewage treatment plants.
Polybrominated diphenyl ethers (PBDEs)	PBDEs are added to a wide range of textiles and plastics as a flame retardant. They easily leach from these materials and have been found throughout the environment and in human breast milk.

Puget Sound Basin: NAWQA Analysis

The USGS sampled waters in the Puget Sound Basin between 1996 and 1998. (Ebbert, Embrey et al. 2000) reported that 26 of 47 analyzed pesticides were detected. A total of 74 manmade organic chemicals were detected in streams and rivers, with different mixtures of chemicals linked to agricultural and urban settings. 2,4-D, triclopyr, diuron, and linuron were all detected in Puget sound samples (Ebbert, 2000). NAWQA results reported that the herbicides atrazine, prometon, simazine and tebuthiuron were the most frequently detected herbicides in surface and ground water (Bortleson and Ebbert 2000). Herbicides were the most common type of pesticide found in an agricultural stream (Fishtrap Creek) and the only type of pesticide found in shallow ground water underlying agricultural land (Bortleson and Ebbert 2000). The most commonly detected VOC in the agricultural land use study area was associated with the application of fumigants to soils prior to planting (Bortleson and Ebbert 2000). One or more fumigant-related compounds (1,2-dichloropropane, 1,2,2-trichloropropane, and 1,2,3-trichloropropane) were detected in over half of the samples. Insecticides, in addition to herbicides, were detected frequently in urban streams (Bortleson and Ebbert 2000). Sampled urban streams showed the highest detection rate for the three insecticides: carbaryl, diazinon, and malathion. No insecticides were found in shallow ground water below urban residential land (Bortleson and Ebbert 2000).

Habitat Restoration

Positive changes in water quality in the region are evident. One of the most notable improvements was the elimination of sewage effluent to Lake Washington in the mid-1960s. This significantly reduced problems within the lake from phosphorus pollution and triggered a concomitant reduction in cyanobacteria (Ruckelshaus and McClure 2007). Even so, as the population and industry has risen in the region a number of new and legacy pollutants are of concern.

Mining

Mining has a long history in Washington. In 2004, the state was ranked 13th nationally in total nonfuel mineral production value and 17th in coal production (Palmisano, Ellis et al. 1993; NMA 2007). Metal mining for all metals (zinc, copper, lead, silver, and gold) peaked between 1940 and 1970 (Palmisano, Ellis et al. 1993). Today, construction sand and gravel, Portland cement, and crushed stone are the predominant materials mined. Where sand and gravel is mined from riverbeds (gravel bars and floodplains) it may result in changes in channel elevations and patterns, instream sediment loads, and seriously alter instream habitat. In some cases, instream or floodplain mining has resulted in large scale river avulsions. The effect of mining in a stream or reach depends upon the rate of harvest and the natural rate of replenishment, as well as flood and precipitation conditions during or after the mining operations.

Artificial Propagation

The artificial propagation of late-returning Chinook salmon is widespread throughout Puget Sound (Good, Waples et al. 2005). Summer/fall Chinook salmon transfers between watersheds within and outside the region have been commonplace throughout this century. Therefore, the purity of naturally spawning stocks varies from river to river. Nearly 2 billion Chinook salmon have been released into Puget Sound tributaries since the 1950s. The vast majority of these have been derived from local late-returning adults.

Returns to hatcheries have accounted for 57% of the total spawning escapement. However, the hatchery contribution to spawner escapement is probably much higher than that due to hatchery-derived strays on the spawning grounds. The genetic similarity between Green River late-returning Chinook salmon and several other late-returning Chinook salmon in Puget Sound suggests that there may have been a significant and lasting effect from some hatchery transplants (Marshall, Smith et al. 1995).

Overall, the use of Green River stock throughout much of the extensive hatchery network in this ESU may reduce the genetic diversity and fitness of naturally spawning

populations (Good, Waples et al. 2005).

Hydromodification Projects

More than 20 dams occur within the region's rivers and overlap with the distribution of salmonids. A number of basins contain water withdrawal projects or small impoundments that can impede migrating salmon. The resultant impact of these and land use changes (forest cover loss and impervious surface increases) has been a significant modification in the seasonal flow patterns of area rivers and streams, and the volume and quality of water delivered to Puget Sound waters. Several rivers have been modified by other means including levees and revetments, bank hardening for erosion control, and agriculture uses. Since the first dike on the Skagit River delta was built in 1863 for agricultural development (Ruckelshaus and McClure 2007), other basins like the Snohomish River are diked and have active drainage systems to drain water after high flows that top the dikes. Dams were also built on the Cedar, Nisqually, White, Elwha, Skokomish, Skagit, and several other rivers in the early 1900s to supply urban areas with water, prevent downstream flooding, allow for floodplain activities (like agriculture or development), and to power local timber mills (Ruckelshaus and McClure 2007).

Over the next few years, however, a highly publicized and long discussed dam removal project is expected to begin in the Elwha River. The removal of two dams in the Elwha River, a short but formerly very productive salmon river, is expected to open up more than 70 miles of high quality salmon habitat (Wunderlich, Winter et al. 1994; Ruckelshaus and McClure 2007). Estimates suggest that nearly 400,000 salmon could begin using the basin within 30 years after the dams are removed (PSAT 2007).

In 1990, only one-third of the water withdrawn in the Pacific Northwest was returned to the streams and lakes (NRC 1996). Water that returns to a stream from an agricultural irrigation is often substantially degraded. Problems associated with return flows include increased water temperature, which can alter patterns of adult and smolt migration; increased toxicant concentrations associated with pesticides and fertilizers; increased

salinity; increased pathogen populations; decreased dissolved oxygen concentration; and increased sedimentation (NRC 1996). Water-level fluctuations and flow alterations due to water storage and withdrawal can affect substrate availability and quality, temperature, and other habitat requirements of salmon. Indirect effects include reduction of food sources; loss of spawning, rearing, and adult habitat; increased susceptibility of juveniles to predation; delay in adult spawning migration; increased egg and alevin mortalities; stranding of fry; and delays in downstream migration of smolts (NRC 1996).

Commercial and Recreational Fishing

Despite regulated fishing programs for salmonids, listed salmonids are also caught as bycatch. There are several approaches under the ESA to address tribal and state take of ESA-listed species that may occur as a result of harvest activities. Section 10 of the ESA provides for permits to operate fishery harvest programs. ESA section 4(d) rules provide exemptions from take for resource, harvest, and hatchery management plans.

Furthermore, there are several treaties that have reserved the right of fishing to tribes in the North West Region.

Management of salmon fisheries in the Puget Sound Region is a cooperative process involving federal, state, tribal, and Canadian representatives. The Pacific Fishery Management Council sets annual fisheries in federal waters from three to 200 miles off the coasts of Washington, Oregon, and California. The annual North of Falcon process sets salmon fishing seasons in waters such as Puget Sound, Willapa Bay, Grays Harbor, and Washington State rivers. Inland fisheries are those in waters within state boundaries, including those extending out three miles from the coasts. The states of Oregon, Idaho, and Washington issue salmon fishing licenses for these areas. Adult salmon returning to Washington migrate through both U.S. and Canadian waters and are harvested by fishermen from both countries. The 1985 Pacific Salmon Treaty helps fulfill conservation goals for all members and is implemented by the eight-member bilateral Pacific Salmon Commission. The Commission does not regulate salmon fisheries, but provides regulatory advice.

Most of the commercial landings in the region are groundfish, Dungeness crab, shrimp, and salmon. Many of the same species are sought by Tribal fisheries and by charter and recreational anglers. Nets and trolling are used in commercial and Tribal fisheries. Recreational anglers typically use hook and line, and may fish from boat, river bank, or docks. Entanglement of marine mammals in fishing gear is not uncommon and can lead to mortality or serious injury.

Harvest impacts on Puget Sound Chinook salmon populations average 75% in the earliest five years of data availability and have dropped to an average of 44% in the most recent five-year period (Good, Waples et al. 2005). Populations in Puget Sound have not experienced the strong increases in numbers seen in the late 1990s in many other ESUs. Although more populations have increased than decreased since the last BRT assessment, after adjusting for changes in harvest rates, trends in productivity are less favorable. Most populations are relatively small, and recent abundance within the ESU is only a small fraction of estimated historic run size.

Oregon-Washington-Northern California Coastal Drainages

This region encompasses drainages originating in the Klamath Mountains, the Oregon Coast Mountains, and the Olympic Mountains. More than 15 watersheds drain the region's steep slopes including the Umpqua, Alsea, Yaquina, Nehalem, Chehalis, Quillayute, Queets, and Hoh rivers. Numerous other small to moderately sized streams dot the coastline. Many of the basins in this region are relatively small. The Umpqua River drains a basin of 4,685 square miles and is slightly over 110 miles long. The Nehalem River drains a basin of 855 square miles and is almost 120 miles long. However, systems here represent some of the most biologically diverse basins in the Pacific Northwest (Kagan, Hak et al. 1999; Belitz, Hamlin et al. 2004; Carter and Resh 2005).

Land Use

The rugged topography of the western Olympic Peninsula and the Oregon Coastal Range has limited the development of dense population centers. For instance, the Nehalem River and the Umpqua River basins consist of less than 1% urban land uses. Most basins in this region have long been exploited for timber production, and are still dominated by forest lands. In Washington State, roughly 90% of the coastal region is forested (Palmisano, Ellis et al. 1993). Roughly 80% of the Oregon Coastal Range is forested as well (Gregory 2000). Approximately 92% of the Nehalem River basin is forested, with only 4% considered agricultural (Belitz, Hamlin et al. 2004). Similarly, in the Umpqua River basin, about 86% is forested land, 5% agriculture, and 0.5% is considered urban lands. Roughly half the basin is under federal management (Carter and Resh 2005).

Habitat Modification

While much of the coastal region is forested, it has still been impacted by land use practices. Less than 3% of the Oregon coastal forest is old growth conifers (Gregory 2000). The lack of mature conifers indicates high levels of habitat modification. As such, overall salmonid habitat quality is poor, though it varies by watershed. The amount of remaining high quality habitat ranges from 0% in the Sixes to 74% in the Siltcoos (ODFW 2005). Approximately 14% of freshwater winter habitat available to juvenile coho is of high quality. Much of the winter habitat is unsuitable due to high temperatures. For example, 77% of coho salmon habitat in the Umpqua basin exceeds temperature standards.

Reduction in stream complexity is the most significant limiting factor in the Oregon coastal region. An analysis of the Oregon coastal range determined the primary and secondary life cycle bottlenecks for the 21 populations of coastal coho salmon (Nicholas, McIntosh et al. 2005). Nicholas *et al.* (2005) determined that stream complexity is either the primary (13) or secondary (7) bottleneck for every population. Stream complexity has been reduced through past practices such as splash damming, removing riparian

vegetation, removing LWD, diking tidelands, filling floodplains, and channelizing rivers.

Habitat loss through wetland fills is also a significant factor. Table 70 summarizes the change in area of tidal wetlands for several Oregon estuaries (Good 2000).

Table 70. Change in total area (acres²) of tidal wetlands in Oregon (tidal marshes and swamps) due to filling and diking between 1870 and 1970 (Good 2000).

Estuary	Diked or Filled Tidal Wetland	Percent of 1870 Habitat Lost
Necanicum	15	10
Nehalem	1,571	75
Tillamook	3,274	79
Netarts	16	7
Sand Lake	9	2
Nestucca	2,160	91
Salmon	313	57
Siletz	401	59
Yaquina	1,493	71
Alsea	665	59
Siuslaw	1,256	63
Umpqua	1,218	50
Coos Bay	3,360	66
Coquille	4,600	94
Rogue	30	41
Chetco	5	56
Total	20,386	72%

The only listed salmonid population in coastal Washington is the Ozette Lake sockeye. The range of this ESU is small, including only one lake (31 km²) and 71 km of stream. Like the Oregon Coastal drainages, the Ozette Lake area has been heavily managed for logging. Logging resulted in road building and the removal of LWD, which affected the nearshore ecosystem (NMFS Salmon Recovery Division 2008). LWD along the shore offered both shelter from predators and a barrier to encroaching vegetation (NMFS Salmon Recovery Division 2008). Aerial photograph analysis shows near-shore vegetation has increased significantly over the past 50 years (Ritchie 2005). Further, there is strong evidence that water levels in Ozette Lake have dropped between 1.5 and 3.3 ft from historic levels [Herrera 2005 *in* (NMFS Salmon Recovery Division 2008)]. The impact of this water level drop is unknown. Possible effects include increased desiccation of sockeye redds and loss of spawning habitat. Loss of LWD has also contributed to an increase in silt deposition, which impairs the quality and quantity of

spawning habitat. Very little is known about the relative health of the Ozette Lake tributaries and their impact on the sockeye salmon population.

Mining

Oregon is ranked 35th nationally in total nonfuel mineral production value in 2004. In that same year, Washington was ranked 13th nationally in total nonfuel mineral production value and 17th in coal production (Palmisano, Ellis et al. 1993; NMA 2007). Metal mining for all metals (*e.g.*, zinc, copper, lead, silver, and gold) peaked in Washington between 1940 and 1970 (Palmisano, Ellis et al. 1993). Today, construction sand, gravel, Portland cement, and crushed stone are the predominant materials mined in both Oregon and Washington. Where sand and gravel is mined from riverbeds (gravel bars and floodplains) changes in channel elevations and patterns, and also changes in instream sediment loads, may result and alter instream habitat. In some cases, instream or floodplain mining has resulted in large scale river avulsions. The effect of mining in a stream or reach depends upon the rate of harvest and the natural rate of replenishment. Additionally, the severity of the effects is influenced by flood and precipitation conditions during or after the mining operations.

Hydromodification Projects

Compared to other areas in the greater Northwest Region, the coastal region has fewer dams and several rivers remain free flowing (*e.g.*, Clearwater River). The Umpqua River is fragmented by 64 dams, the fewest number of dams on any large river basin in Oregon (Carter and Resh 2005). According to Palmisano *et al.* (1993) dams in the coastal streams of Washington permanently block only about 30 miles of salmon habitat (Figure 54). In the past, temporary splash dams were constructed throughout the region to transport logs out of mountainous reaches. The general practice involved building a temporary dam in the creek adjacent to the area being logged, and filling the pond with logs. When the dam broke the floodwater would carry the logs to downstream reaches where they could be rafted and moved to market or downstream mills. Thousands of splash dams were constructed across the Northwest in the late 1800s and early 1900s.

While the dams typically only temporarily blocked salmon habitat, in some cases dams remained long enough to wipe out entire salmon runs. The effects of the channel scouring and loss of channel complexity resulted in the long-term loss of salmon habitat (NRC 1996).

Commercial and Recreational Fishing

Despite regulated fishing programs for salmonids, listed salmonids are also caught as bycatch. There are several approaches under the ESA to address tribal and state take of ESA-listed species that may occur as a result of harvest activities. Section 10 of the ESA provides for permits to operate fishery harvest programs. ESA section 4(d) rules provide exemptions from take for resource, harvest, and hatchery management plans.

Management of salmon fisheries in the Washington-Oregon-Northern California drainage is a cooperative process involving federal, state, and tribal representatives. The Pacific Fishery Management Council sets annual fisheries in federal waters from three to 200 miles off the coasts of Washington, Oregon, and California. Inland fisheries are those within state boundaries, including those extending out three miles from state coastlines. The states of Oregon, Idaho, California and Washington issue salmon fishing licenses for these areas.

Most commercial landings in the region are groundfish, Dungeness crab, shrimp, and salmon. Many of the same species are sought by Tribal fisheries, as well as by charter, and recreational anglers. Nets and trolling are used in commercial and Tribal fisheries. Recreational anglers typically use hook and line and may fish from boat, river bank, or docks.

Integration of the Environmental Baseline on Listed Resources

Collectively, the components of the environmental baseline for the action area include sources of natural mortality as well as influences from natural oceanographic and climatic features in the action area. Climatic variability may affect the growth, reproductive

success, and survival of listed Pacific salmonids in the action area. Temperature and water level changes may lead to: (1) Reduced summer and fall stream flow, leading to loss of spawning habitat and difficulty reaching spawning beds; (2) increased winter flooding and disturbance of eggs; (3) changes in peak stream flow timing affecting juvenile migration; and (4) rising water temperature may exceed the upper temperature limit for salmonids at 64°F (18°C) (JISAO 2007). Additional indirect impacts include changes in the distribution and abundance of the prey and the distribution and abundance of competitors or predators for salmonids. These conditions will influence the population structure and abundance for all listed Pacific salmonids.

The baseline also includes human activities resulting in disturbance, injury, or mortality of individual salmon. These activities include hydropower, hatcheries, harvest, and habitat degradation, including poor water quality and reduced availability of spawning and rearing habitat for all 28 ESUs/DPSs. As such, these activities degrade salmonid habitat, including all designated critical habitat and their PCEs. While each area is affected by a unique combination of stressors, the two major impacts to listed Pacific salmonid critical habitat are habitat loss and decreased prey abundance. Although habitat restoration and hydropower modification measures are ongoing, the long-term beneficial effects of these actions on Pacific salmonids, although anticipated, remain to be realized. Thus, we are unable to quantify these potential beneficial effects at this time.

Listed Pacific salmonids and designated critical habitat may be adversely affected by the proposed registration of 2,4-D, triclopyr BEE, diuron, linuron, captan and chlorothalonil in California, Idaho, Oregon, and Washington. These salmonids are and have been exposed to the components of the environmental baseline for decades. The activities discussed above have some level of effect on all 28 ESUs/DPSs in the proposed action area. They have also eroded the quality and quantity of salmonid habitat – including designated critical habitat. We expect the combined consequences of those effects, including impaired water quality, temperature, and reduced prey abundance, may

increase the vulnerability and susceptibility of overall fish health to disease, predation, and competition for available suitable habitat and prey items. The continued trend of anthropogenic impairment of water quality and quantity on Pacific salmonids and their habitats may further compound the declining status and trends of listed salmonids, unless measures are implemented to reverse this trend.

Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids

The analysis includes three primary components: exposure, response, and risk characterization. We analyze exposure and response, and integrate the two in the risk characterization phase where we address support for risk hypotheses. These risk hypotheses are predicated on effects to salmonids. Designated critical habitat is analyzed separately (see *Effects of the Proposed Action to Designated Critical Habitat* and *Integration and Synthesis for Designated Critical Habitat*).

Exposure Analysis

In this section, we identify and evaluate potential exposure of salmonids to the stressors of the action (Figure 55). We begin by presenting general life history information of vulnerable life stages of Pacific salmon and steelhead. Next, we present a general discussion of the physical and chemical properties of the six a.i.s and their degradation products that influence exposure of listed species and designated critical habitat to these stressors of the action. We then evaluate co-occurrence of the salmon habitat with the stressors of the action by comparing the distribution of sites authorized for pesticide use by product labeling to the distribution of each species and their designated critical habitat. To further characterize exposure where co-occurrence exists, we summarize EPA exposure estimates presented in the six BEs, present additional exposure estimates for shallow floodplain habitats utilized by salmonids, and summarize the available waterquality monitoring data. Finally, we conclude with a summary of anticipated ranges

of exposure when pesticide use is proximate to salmon habitats and characterize the uncertainty contained in this analysis. Because the ESA section 7 consultation process is intended to insure that the agency action is not likely to jeopardize listed species or destroy or adversely modify critical habitat, NMFS considers a variety of scenarios in addition to those presented in EPA's BEs. These scenarios provide estimates for the range of habitats used by listed salmonids.

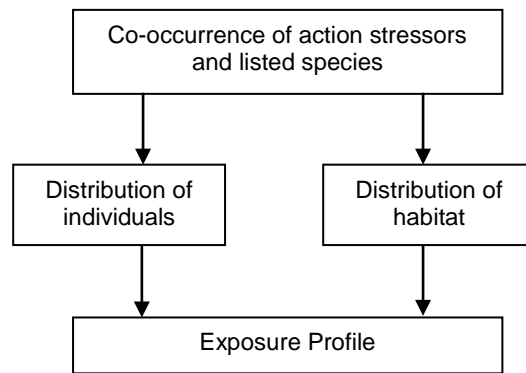


Figure 55. Exposure analysis.

Threatened and Endangered Pacific Salmonids use of Aquatic Habitats

Within the Status Section we discussed salmonid lifecycles, life histories, and the use and significance of aquatic habitats. Listed salmonids occupy a variety of aquatic habitats that range from shallow, low-flow freshwaters to open reaches of the Pacific Ocean. All listed Pacific salmonid species use freshwater, estuarine, and marine habitats at some point during their life. The temporal and spatial use of habitats by salmonids depends on the species and the individuals' life history and life stage. General life history descriptions describing use of aquatic habitats is provided below in Table 71. Additionally information on timing of presence of the ESUs/DPSs is presented in *Appendix 6*. Many species migrate hundreds or thousands of miles during their lifetime, increasing the likelihood that they will come in contact with aquatic habitats contaminated with pesticides.

Table 71. General life-histories of Pacific salmonids.

Species	General Life History Descriptions		
(number of listed ESUs or DPSs)	Spawning Migration	Spawning Habitat	Juvenile Rearing and Migration
Chinook (9)	Mature adults (usually four to five years old) enter rivers (spring through fall, depending on run). Adults migrate and spawn in river reaches extending from above the tidewater to as far as 1,200 miles from the sea. Chinook salmon migrate and spawn in four distinct runs (spring, fall, summer, and winter). Chinook salmon are semelparous ¹ .	Generally spawn in the middle and upper reaches of main stem rivers and larger tributary streams.	The alevin life stage primarily resides just below the gravel surface until they approach or reach the fry stage. Immediately after leaving the gravel, fry distribute to habitats that provide refuge from fast currents and predators. Juveniles exhibit two general life history types: Ocean-type fish migrate to sea in their first year, usually within six months of hatching. Ocean-type juveniles may rear in the estuary for extended periods. Stream-type fish migrate to the sea in the spring of their second year.
Coho (4)	Mature adults (usually two to four years old) enter the rivers in the fall. The timing varies depending on location and other variables. Coho salmon are semelparous.	Spawn throughout smaller coastal tributaries, usually penetrating to the upper reaches to spawn. Spawning takes place from October to March.	Following emergence, fry move to shallow areas near stream banks. As fry grow they distribute up and downstream and establish territories in small streams, lakes, and off-channel ponds. Here they rear for 12-18 months. In the spring of their second year juveniles rapidly migrate to sea. Initially, they remain in nearshore waters of the estuary close to the natal stream following downstream migration.
Chum (2)	Mature adults (usually three to four years old) enter rivers as early as July, with arrival on the spawning grounds occurring from September to January. Chum salmon are semelparous.	Generally spawn from just above tidewater in the lower reaches of mainstem rivers, tributary stream, or side channels to 100 km upstream.	The alevin life stage primarily resides just below the gravel surface until they approach or reach the fry stage. Immediately after leaving the gravel, swim-up fry migrate downstream to estuarine areas. They reside in estuaries near the shoreline for one or more weeks before migrating for extended distances, usually in a narrow band along the Pacific Ocean's coast.

Species (number of listed ESUs or DPSs)	General Life History Descriptions		
	Spawning Migration	Spawning Habitat	Juvenile Rearing and Migration
Sockeye (2)	Mature adults (usually four to five years old) begin entering rivers from May to October. Sockeye are semelparous.	Spawn along lakeshores where springs occur and in outlet or inlet streams to lakes.	The alevin life stage primarily resides just below the gravel surface until they approach or reach the fry stage. Immediately after leaving the gravel, swim-up fry migrate to nursery lakes or intermediate feeding areas along the banks of rivers. Populations that migrate directly to nursery lakes typically occupy shallow beach areas of the lake's littoral zone; a few cm in depth. As they grow larger they disperse into deeper habitats. Juveniles usually reside in the lakes for one to three years before migrating to off shore habitats in the ocean. Some are residual, and complete their entire lifecycle in freshwater.
Steelhead (11)	Mature adults (typically three to five years old) may enter rivers any month of the year, and spawn in late winter or spring. Migration in the Columbia River system extends up to 900 miles from the ocean in the Snake River. Steelhead are iteroparous ² .	Usually spawn in fine gravel in a riffle above a pool.	The alevin life stage primarily resides just below the gravel surface until they approach or reach the fry stage. Immediately after leaving the gravel, swim-up fry usually inhabit shallow water along banks of stream or aquatic habitats on streams margins. Steelhead rear in a wide variety of freshwater habitats, generally for two to three years, but up to six or seven years is possible. They smolt and migrate to sea in the spring.

1 spawn only once

2 spawn more than once

Freshwater, estuarine, and marine near-shore habitats are areas subject to pesticide loading from runoff and drift given their proximity to pesticide application sites. Small streams and many floodplain habitats are more susceptible to higher pesticide concentrations than other aquatic habitats used by salmon because their physical characteristics provide less dilution and dissipation. Examples of floodplain habitats

include alcoves, channel edge sloughs, overflow channels, backwaters, terrace tributaries, off-channel dredge ponds, off-channel ponds, and braids (Swift III 1979; Anderson 1999; Beechie and Bolton 1999). The transition from yolk sac fry to exogenous feeding is a critical life stage for all salmon species requiring availability of prey. Diverse, abundant communities of invertebrates (many of which are salmonid prey items), also populate floodplain habitats and, in part, are responsible for juvenile salmonids' reliance on them. Juvenile coho salmon, stream-type Chinook salmon, and steelhead use floodplain habitats for extended durations (several months). Although these habitats typically vary in surface area, volume, and flow, they are frequently shallow, low to no-flow systems protected from a river's or a stream's primary flow. Thus, rearing and migrating juvenile salmonids use these habitats extensively (Caffrey 1996; Beechie and Bolton 1999; Montgomery 1999; Roni 2002; Opperman and Merenlender 2004; Beechie, Liermann et al. 2005; Morley, Garcia et al. 2005; Henning, Gresswell et al. 2006).

Exposure Pathways to Salmonids Habitats

Aquatic habitats can be contaminated by pesticides applied to terrestrial target sites through several alternative pathways. For example, spray drift or primary drift refers to the off-target deposition of droplets from spray-applied pesticides at the time of application. The likelihood of spray drift to an aquatic habitat is determined by the application method, the proximity to the habitat, and meteorological conditions at the time of application. Some pesticides are applied directly to surface water for control of plants, mosquitoes, and other aquatic pests. Other pathways of surface water contamination are influenced primarily by the environmental fate properties of the chemical. For example, secondary drift or vapor drift is dependent on a chemical's volatility and refers to the redistribution of pesticides from plant and soil surfaces through volatilization and subsequent atmospheric deposition. Runoff and leaching, the horizontal and vertical movement of pesticides with rainwater or irrigation water, are influenced by chemical-specific properties that determine the compound's persistence and mobility in soil and water. Standardized tests are typically used to characterize

mobility (*e.g.* solubility, K_d and K_{oc}) and persistence under different environmental conditions (*e.g.* hydrolysis, photolysis, and metabolism half-lives in aerobic and anaerobic environments). Below we present environmental fate properties of the six a.i.s to characterize the relative importance of these exposure pathways in terms of the potential for the active ingredients and their toxic degradates to contaminate salmonid bearing habitats and designated critical habitats.

Summary of Chemical Fate of the Six Active Ingredients

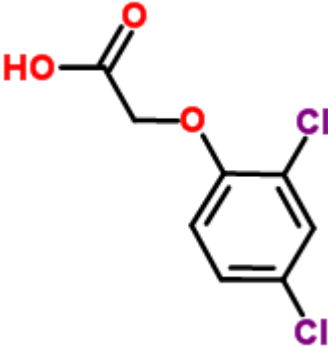
Pesticides can contaminate surface waters via runoff, erosion, leaching, spray drift from application at terrestrial sites or direct application to aquatic habitats, and atmospheric deposition. The six a.i.s are primarily registered for use in terrestrial habitats although some 2,4-D products are registered for use as aquatic herbicides and may be directly applied to a variety of habitats utilized by listed salmonids including ponds, lakes, reservoirs, marshes, ditches, canals, and slow moving rivers and streams. Fish are most likely exposed to the six a.i.s from the water column where the chemicals enter the fish during respiration, (*i.e.*, across the gills), or where fish sensory systems come in direct contact with contaminated water (*i.e.*, olfactory sensory neurons). Other secondary routes may contribute to overall exposure including incidental ingestion of the chemical in sediment or ingestion of the chemical in food items. Below we summarize chemical fate properties of the six a.i.s reported by EPA in the salmon BEs and red-legged frog BEs. Where discrepancies existed between the two documents, we deferred to the more recent document.

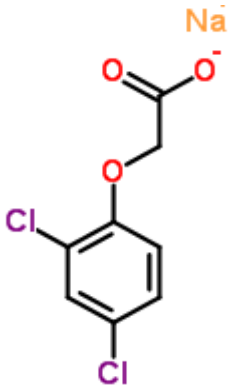
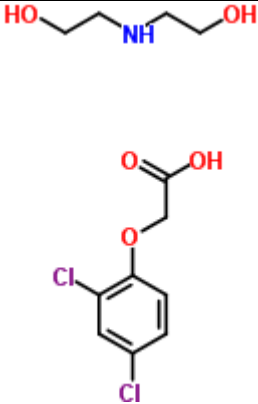
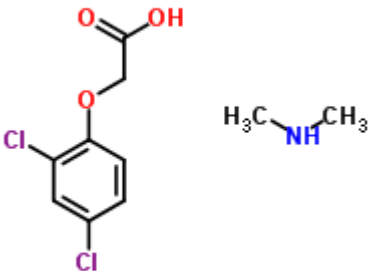
2,4-D

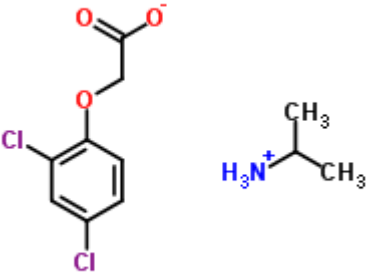
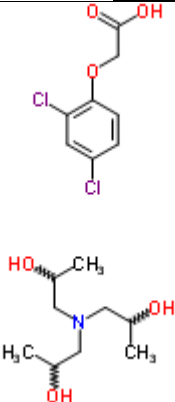
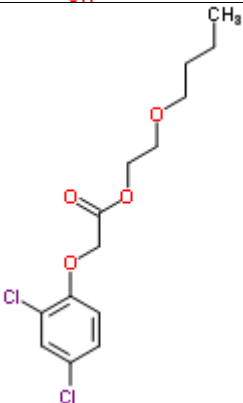
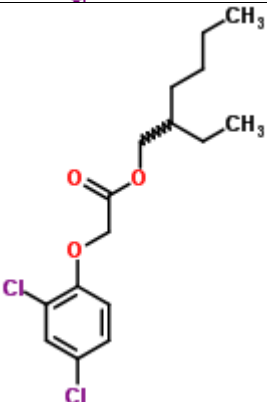
EPA developed an environmental fate bridging strategy to account for the differences among the different chemical forms of 2,4-D (Table 72)(Section 2.4.1 in (EPA 2009)). The strategy is based on the rapid degradation, under most environmental conditions, of 2,4-D esters and 2,4-D amine salts to 2,4-D acid. In most aquatic and terrestrial environments 2,4-D amine salts dissociation to form 2,4-D acid instantaneously. The conversion of 2,4-D esters to 2,4-D acid is also rapid (<2.9 d) in agricultural soils and

water (EPA 2009). Physical and chemical fate parameters of 2,4-D acid are provided in Table 73. 2,4-D has a relatively low potential for volatilization from soil and water suggesting secondary drift is not a likely pathway of high exposure. 2,4-D is frequently detected in rainfall although concentrations are generally < 1 ug/L. The low octanol water partition coefficient also suggest bioaccumulation in fish would be relatively low. The degradation of 2,4-D acid is dependent on oxidative microbial-mediated mineralization and photodegradation in water. In buffered solution, the photodegradation half-life was 12.98 days. In soil, the photodegradation half-life is 68 days. Thirty terrestrial field studies found dissipation half-lives of 2,4-D acid that ranged from 1 – 43 d, and had a median half-life of 6 d. However, these half-lives did not account for residues below the surface layer of soil (0-6 inches), and leaching can be a significant route of dissipation for 2,4-D. Aquatic field studies suggest that persistence of 2,4-D in the water column is dependent on chemical form and site specific conditions (e.g. pH). 2,4-D half-lives ranged in the water column ranged from 1 – 40 days. 2,4-D acid has low binding affinity in mineral soils and it's mobility is characterized as intermediate to very mobile depending on soil type. These properties suggest that drift and runoff are the most likely pathways of deposition of 2,4-D into aquatic habitats (EPA 2009).

Table 72. Chemical structures and the molecular weight ratios of various chemical forms of 2,4-D¹.

Chemical Name	Chemical Structure	Molecular Weight Ratio relative to 2,4-D acid
2,4-D acid		1.00

2,4-D sodium salt		1.10
2,4-D diethanolamine salt (DEA)		1.48
2,4-D dimethylamine salt (DMA)		1.20

2,4-D isopropylamine salt (IPA)		1.27
2,4-D triisopropanolamine salt (TIPA)		1.87
2,4-D butoxyethyl ester (BEE)		1.45
2,4-D ethylhexyl ester (EHE)		1.51

2,4-D isopropyl ester (IPE)		1.19
-----------------------------	--	------

1 (EPA 2009)

Table 73. Environmental fate characteristics of 2,4-D acid¹.

Parameter	Value
Water solubility	569 mg/L at 20° C
Vapor pressure	1.47×10^{-7} mm Hg at 25° C
Henry's law constant	4.74×10^{-10} atm m ³ mol ⁻¹
Octanol/Water partition coefficient	Log K _{ow} = 2.81
Hydrolysis (t _{1/2}) pH 5, pH 7, & pH 9	stable
Aqueous photolysis (t _{1/2})	12.98 d
Soil photolysis (t _{1/2})	68 d
Aerobic soil metabolism (t _{1/2})	1.44 to 12.4 d
Anaerobic soil metabolism (t _{1/2})	Not Specified
Aerobic aquatic metabolism (t _{1/2})	15 d
Anaerobic aquatic metabolism (t _{1/2})	41 to 333 d
Soil partition coefficient	K _d = 0.17 – 0.36; K _{oc} = 56 – 117 L/kg _{soil}

1 (EPA 2009)

Triclopyr BEE

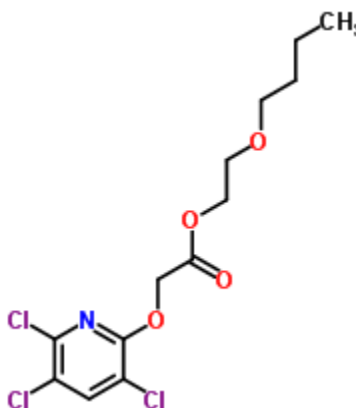


Figure 56. Chemical structure of triclopyr BEE.

Triclopyr BEE (Figure 56) is non-volatile and has low solubility (6.8 mg/L). It quickly hydrolyzes in the environment to form triclopyr acid/anion and butoxyethanol (0.5 d at pH 6.7). The rate of hydrolysis occurs more rapidly at higher PHs. Butoxyethanol is rapidly dissipated by microbial degradation. The predominant moiety present in the environment is triclopyr anion when either triclopyr BEE or triclopyr TEA are applied (EPA 2009). The environmental fate parameters for triclopyr acid are presented below in Table 74. Triclopyr acid/anion is relatively persistent and mobile in the environment. Triclopyr acid primarily degrades through photodegradation in water and through microbial processes in the soil. Microbial degradation triclopyr in soil produces the major metabolite TCP (>10% of applied parent), which is likely to be transported to surface waters because it is both persistent and mobile (EPA 2009). Triclopyr BEE products are applied in broadcast applications by ground and aerial application methods to several use sites suggesting transport to surface waters via primary drift is likely. Its persistence and mobility suggest runoff is a likely pathway of exposure to aquatic habitats, and its relatively low volatility suggest secondary drift and long range transport is a pathway of less concern.

Table 74. Environmental fate characteristics of triclopyr acid¹.

Parameter	Value
Water solubility	440 mg/L at 25° C
Vapor pressure	1.26×10^{-6} mm Hg
Henry's law constant	9.66×10^{-7} atm m ³ mol ⁻¹
Octanol/Water partition coefficient	Not specified
Hydrolysis ($t_{1/2}$)	Stable at pH 5,7,9
Aqueous photolysis ($t_{1/2}$)	0.3 – 1.7 d
Soil photolysis ($t_{1/2}$)	Not Specified
Aerobic soil metabolism ($t_{1/2}$)	8 – 18 d
Anaerobic soil metabolism ($t_{1/2}$)	Stable (1300 d)
Aerobic aquatic metabolism ($t_{1/2}$)	142 d
Anaerobic aquatic metabolism ($t_{1/2}$)	Not Specified
Soil partition coefficient	$K_d = 0.165-0.975$; $K_{oc} = 25-134$ L/kg _{oc}

1- (EPA 2009)

Diuron

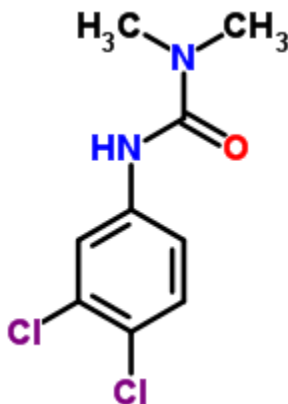


Figure 57. Chemical structure of diuron

Environmental fate studies indicate diuron (Figure 57) is moderately to highly persistent (Table 75). Field residue studies indicate highly variable half-lives of diuron in the soil that range from 1 month to more than 1 year. EPA considers the average field dissipation half-life of diuron to be 115 d. It is primarily degraded through microbial processes and to a lesser degree from photodegradation (EPA 2009). In water, microbial breakdown is the primary degradation pathway. Diuron is stable to hydrolysis at the pH range of 5 - 9 and has a photolysis half-life of 43 d. In aquatic metabolism studies, half-lives of 5 d and 33 d were determined for anaerobic and aerobic conditions, respectively. Diuron tends to

not sorb well to soil, and its mobility is inversely correlated with soil organic matter. The metabolites are less mobile than the parent. Diuron is prone to surface water runoff and leaching given its persistence and mobility in soils. Studies have found relatively high peak concentrations of diuron in runoff under simulated rainfall conditions (600-1700 µg/L) and in runoff monitoring in fields months after application (200-890 µg/L). Primary spray drift is also a likely pathway of exposure to aquatic organisms given broadcast ground and aerial application methods. Secondary drift is less likely considering its relatively low volatility (EPA 2009). Information to characterize the potential for accumulation in aquatic organisms is lacking.

Table 75. Environmental fate characteristics of diuron¹.

Parameter	Value
Water solubility	42 mg/L at 20° C
Vapor pressure	6.9×10^{-8}
Henry's law constant	$5.10 \times 10^{-10} \text{ atm m}^3 \text{ mol}^{-1}$
Octanol/Water partition coefficient	Not Specified
Hydrolysis ($t_{1/2}$) pH 5, pH 7, & pH 9	Stable
Aqueous photolysis ($t_{1/2}$)	43 d
Soil photolysis ($t_{1/2}$)	173 d
Aerobic soil metabolism ($t_{1/2}$)	372 d
Anaerobic soil metabolism ($t_{1/2}$)	1,000 d
Aerobic aquatic metabolism ($t_{1/2}$)	33 d
Anaerobic aquatic metabolism ($t_{1/2}$)	5 d
Soil partition coefficient	$K_{oc} = 468-1666$; $K_d = 14 \text{ L/kg}_{\text{soil}}$

¹ (EPA 2009)

Linuron

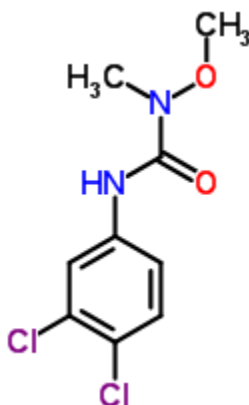


Figure 58. Chemical structure of linuron.

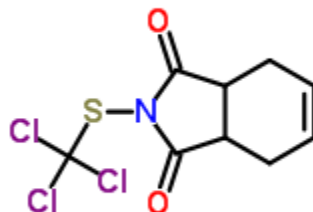
Data are not adequate to fully characterize the terrestrial dissipation of linuron (Figure 58) in the field. However, environmental fate studies suggest half-lives of < 60d in soil (Table 76). Linuron dissipates principally through microbial degradation in soils. In water, linuron is stable to hydrolysis and can be degraded through metabolism and secondarily through photolysis (EPA 2008). Its half-life in an anaerobic metabolism study was less than 21 d. However, persistence may increase under conditions of low microbial activity. Significant fractions of linuron may exist in the water column given its low to intermediate tendency to partition to sediment (EPA 2008). Linuron can be transported to surface waters through primary drift from ground spray applications. Secondary drift is less likely given low potential to volatilize based on its physico-chemical properties. Linuron is only slightly mobile in high organic content soils. It has greater mobility in permeable soils and soils with low organic matter. Transport to linuron to surface water is likely from linuron dissolved in surface runoff and suspended sediment. Bioconcentration factors (40-240x) indicate some accumulation in aquatic organism may occur (EPA 2008).

Table 76. Environmental fate characteristics of linuron¹.

Parameter	Value
Water solubility	810 mg/L (estimate)
Vapor pressure	1.5×10^{-5} mm Hg at 26° C
Henry's law constant	2.6×10^{-6} atm m ³ mol ⁻¹
Octanol/Water partition coefficient	Not Specified
Hydrolysis (t _{1/2})	stable
Aqueous photolysis (t _{1/2})	49 d
Soil photolysis (t _{1/2})	Not Specified; 79% remaining at 15 d
Aerobic soil metabolism (t _{1/2})	49 d
Anaerobic soil metabolism (t _{1/2})	Not Specified
Aerobic aquatic metabolism (t _{1/2})	48 d
Anaerobic aquatic metabolism (t _{1/2})	21 d
Soil partition coefficient ²	K _d = 2.7-7.2 L/kg; K _{oc} = 370 L/kg

1 (EPA 2008)

2 (EPA 1995)

Captan**Figure 59. Chemical structure of captan.**

The environmental fate characteristics of captan suggest it is quickly degraded by hydrolysis and aerobic metabolism (Table 77). Photodegradation on soil also occurs, but is secondary to hydrolysis and metabolism. Captan (Figure 59) has a relatively short half-life, generally less than 10 d in both soils and water. Terrestrial field dissipation studies found captan dissipated with half lives of 2.5 – 24 d and was relatively immobile to slightly mobile at six sites. Transport of captan to aquatic habitats may occur from primary drift associated with ground and aerial-spray pesticide applications. Secondary drift of captan is less likely given relatively low volatility. Captan is expected to be

moderately mobile in the soil based on its soil adsorption properties although field studies found it to be only slightly mobile ($K_d = 3\text{-}8\text{ L/Kg}$). These data suggest surface water runoff of dissolved or suspended material is also a likely transport mechanism for captan in aquatic habitats (EPA 2007).

Table 77. Environmental fate characteristics of captan¹.

Parameter	Value
Water solubility	3.3 mg/L
Vapor pressure	8×10^{-8} mm Hg at 26° C
Henry's law constant	9.6×10^{-10} atm m ³ mol ⁻¹
Octanol/Water partition coefficient	2.79
Hydrolysis pH 5, 7, and 9($t_{1/2}$)	0.8, 0.25, 0.006 d
Aqueous photolysis ($t_{1/2}$)	0.42 d
Soil photolysis ($t_{1/2}$)	Not Specified
Aerobic soil metabolism ($t_{1/2}$)	<1 d
Anaerobic soil metabolism ($t_{1/2}$)	1.85 d
Aerobic aquatic metabolism ($t_{1/2}$)	<1 d
Anaerobic aquatic metabolism ($t_{1/2}$)	Not Specified
Soil partition coefficient	$K_d = 3\text{-}8\text{ L/kg}$

¹ (EPA 2007)

Chlorothalonil

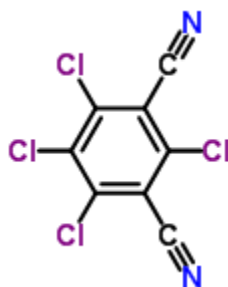


Figure 60. Chemical structure of chlorothalonil.

Chlorothalonil degrades rapidly in clear, shallow water through photolysis (Table 78).

Aqueous photolysis is limited to aquatic habitats exposed to direct sunlight.

Chlorothalonil (Figure 60) is stable to hydrolysis at pHs below 9. At pH of 9 and greater chlorothalonil is degraded by hydrolyses with a half-life of 40-60 d at. The main route of dissipation for chlorothalonil in the environment is though biotic metabolism

(EPA 2007). Bioaccumulation factors are moderately high for bivalves (2600) and fish (2700). Chlorothalonil has low mobility in loamy soils and low to moderate mobility in sandy soils. Its moderate persistence suggests the potential of transport to surface waters through runoff. Its major metabolite 4-hydroxy-2,5,6-trichloro-1,3-dicyanobenzene (SDS-3701) has a greater potential for runoff given greater persistence and mobility in soil (EPA 2007). Transport to aquatic habitats from primary spray drift is likely given ground and aerial broadcast spray applications. Its physical properties suggest chlorothalonil can volatilize from soil and water and therefore transport from secondary drift may also occur. Air monitoring has detected trace amounts of chlorothalonil up to a mile away from pesticide application sites (EPA 2007). However, concentrations were not considered high enough to suggest that long range transport through secondary drift would result in a substantial increase in risk to nontarget species (EPA 2007).

Table 78. Environmental fate characteristics of chlorothalonil¹.

Parameter	Value
Water solubility	0.8 mg/L at 20° C
Vapor pressure	5.7×10^{-7} mm Hg
Henry's law constant	2.6×10^{-7} atm m ³ mol ⁻¹
Octanol/Water partition coefficient	Log K _{ow} = 3.8
Hydrolysis (t _{1/2}) pH 5, pH 7, & pH 9	Stable, stable, 40-60 d
Aqueous photolysis (t _{1/2})	0.42 d
Soil photolysis (t _{1/2})	Not Specified
Aerobic soil metabolism (t _{1/2})	22-68 d
Anaerobic soil metabolism (t _{1/2})	Not Specified
Aerobic aquatic metabolism (t _{1/2})	7-16 d
Anaerobic aquatic metabolism (t _{1/2})	21-29 d
Soil partition coefficient	K _d = 3-29

¹ (EPA 2007)

Degradates of the 6 active ingredients

The molecular structure of a pesticide may be modified by biotic (*e.g.* microbial metabolism) or abiotic processes (*e.g.* photolysis and hydrolysis). The products of these processes typically have different toxicities, environmental fate characteristics, and risks compared to the parent pesticide. Several degradates of parent 2,4-D have been identified in laboratory environmental fate studies suggesting possible exposure to salmonids and their habitats. 1,2,4-benzenetriol is a product of photodegradation. However, it may be

less likely to occur in many environments as degradation appears to be dependent primarily on oxidative microbial-mediated mineralization. Metabolic products (Table 79) that may result in exposure in the aquatic environment include 2,4-DCP; 2,4-DCA; 4-chlorophenol; 2-chlorophenol; 4-chlorophenoxyacetic acid; and chlorohydroquinone (EPA 2009).

Table 79. Metabolites and degradates of 2,4-D.

Laboratory study	Degradates/metabolites of 2,4-D acid
Direct aqueous photolysis	1,2,4-benzenetriol (37% of applied)
Aerobic soil metabolism	2,4-DCP (3.5% of applied) 2,4-DCA (2.8% of applied)
Anaerobic aquatic metabolism	2,4-DCP, 4-chlorophenol, and 2-chlorophenol
Aerobic aquatic metabolism	2,4-DCP, 4-chlorophenol, 4-chlorophenoxyacetic acid, and chlorohydroquinone

(EPA 2009)

Triclopyr BEE may be quickly degraded to triclopyr acid through hydrolysis (Table 80). Conversion to the acid through hydrolysis is pH-dependent with a half-life of 7 hrs at pH of 9 and a half-life of 84 days at pH 5 (EPA 2004). Laboratory studies indicate triclopyr acid is somewhat persistent and mobile. In aquatic environments, photodegradation products of triclopyr acid include TCP and oxamic acid. In soils, TCP and TMP are formed through biotic metabolism. TCP is both mobile and persistent suggesting likely transport to aquatic habitats (EPA 2009).

Table 80. Metabolites and degradates of triclopyr BEE.

Laboratory study	Degradates/metabolites of triclopyr BEE
Hydrolysis	Triclopyr acid
Aqueous photolysis	Oxamic acid (48% of applied)
Aerobic soil metabolism	TCP (26.4% of applied) TMP (7.8%)
Anaerobic aquatic metabolism	TCP (26% of applied)
Aerobic aquatic metabolism	TCP (<5% of applied)

(EPA 2009)

Although relatively resistant to abiotic degradation, diuron can be broken down somewhat through hydrolysis and photolysis (Table 81). It is also metabolized in the soil

and aquatic environments which may lead to exposure to several metabolic products including DCPMU, MCPDMU, and PDMU. Other degradates which may be deposited or form in the aquatic environment include DCPU, 3,4-DCA, CPMU, and TCAB (EPA 2009). TCAB and TCAOB are ‘dioxin-like’ substances that are also impurities in the diuron manufacturing process. TCAB levels between 0.15 and 28 ppm have been found in diuron samples tested. TCAOB is present at lower levels (EPA 2009).

Table 81. Metabolites and degradates of diuron.

Laboratory study	Degradates/metabolites of diuron
Hydrolysis	3,4-DCA (~2% of applied)
Soil photolysis	DCPMU (3.6% of applied) DCMU ($\leq 0.7\%$ of applied) 3,4-DCA (0.370 ppm) TCAP (0.038 ppm)
Aerobic soil metabolism	DCPMU (22.5% of applied) DCPU (3.4% of applied) CO ₂ (3.36% of applied)
Anaerobic soil metabolism	DCPMU (10.3% of applied)
Anaerobic aquatic metabolism	MCPDMU (83% of applied) PDMU (13% of applied) MCPMU (23% of applied)
Aerobic aquatic metabolism	MCPDMU (25% of applied) DCPMU (9.2% of applied) CPMU (8% of applied)

(EPA 2009)

Linuron and diuron have similar chemical structures and they degrade into several common metabolites (Table 82). In the soil, linuron degrades to 3,4- DCA, DCPMU, DML, and DCPU. In aquatic environments, linuron degrades to desmethoxy linuron, desmethoxy monolinuron, and several minor metabolites (EPA 2008).

Table 82. Metabolites and degradates of linuron.

Laboratory study	Degradates/metabolites of linuron
Hydrolysis	3,4-DCA (~1% of applied) DCPMU (~1% of applied) DML (~1% of applied) DCPU (~1% of applied)
Direct aqueous photolysis	3-(3-chloro-4-hydroxyphenyl)-1-methoxy-1-methylurea, 3,4-ichlorophenylurea, and 3-(3,4-dichlorophenyl)-1-methylurea

Soil photolysis	Norlinuron (<8.4% of applied) Desmethyl linuron (<8.4% of applied) 3,4-DCA (<8.4% of applied)
Aerobic soil metabolism	Desmethoxy linuron (3% of applied) Desmethyl linuron (2.1% of applied) Norlinuron (1.9% of applied)
Anaerobic aquatic metabolism	Desmethoxy linuron (46.7% of applied) Desmethoxy monolinuron (78% of applied) Desmethyl linuron (<5% of applied) Norlinuron (<5% of applied) 3,4-DCA (<5% of applied)
Aerobic aquatic metabolism	Desmethoxy linuron (<10% of applied) Desmethyl linuron (<10% of applied) Norlinuron (<10% of applied) 3,4-DCA (<10% of applied)

(EPA 2008)

Captan degrades rapidly in soil and water. The major routes of degradation are believed to be hydrolysis and aerobic metabolism. Degradation products formed that may result in exposure in aquatic environments include THPI, TCMT, THPAm, THCY, inorganic sulfur and chlorine, and thiophosgene (Table 83). Captan is slightly mobile, however THPI and THPAm have significantly greater mobility and are more likely to be transported to aquatic habitats through runoff (EPA 2003).

Table 83. Metabolites and degradates of captan.

Laboratory study	Degradates/metabolites of captan
Hydrolysis	TCMT THPI
Soil photolysis	THPI (21.3% of applied) THCY (9.4% of applied) CO ₂ (41.7% of applied)
Aerobic soil metabolism	THPI THPAm THCY CO ₂ Thiophosgene Inorganic sulfur Chlorine
Anaerobic soil metabolism	THPI THPAm THCY
Aquatic fate study	THPI (81.2% of applied) THPAm (27% of applied) THPI epoxide (9.4% of applied) THPI (81.2% of applied)

(EPA 2003)

Laboratory fate studies with chlorothalonil reveal a number of degradates and metabolites that may be formed in, or transported to the aquatic environments (Table 84). The major degradate of chlorothalonil in the soil under aerobic conditions is SDS-3701. This degradate appears to be more persistent and mobile than chlorothalonil based on both groundwater detections and estimates using a structural analysis model (EPA 1999; EPA 2007). Substantial amounts of SDS-3701 could be available for runoff for longer periods than chlorothalonil (EPA 1999). SDS-3701 can also persist in the aquatic environment. Anaerobic aquatic metabolism studies showed SDS-3701 reached maximum concentrations 1-2 months after introduction and remained at a near constant levels (30-40% of applied) until the end of a 4-month study in silt loam soil (EPA 1999). An aerobic metabolism study conducted under nonstandard test conditions, which included continuous agitation and aeration and high concentrations of suspended sediment resulted in different metabolite profiles and degradation rates than seen in other laboratory environmental fate studies. These conditions may not reflect quiescent bodies of water with lower amounts of suspended sediment and would give higher rates of reaction than would be expected under natural conditions (EPA 1999).

Table 84. Metabolites and degradates of chlorothalonil.

Laboratory study	Degradates/metabolites of captan
Hydrolysis	SDS-19221 (~50% of applied) ² SDS-3701 (~20% of applied) ²
Aqueous photolysis	SDS-3701 (4-10% of applied) ^{1,2}
Aerobic soil metabolism	SDS-3701 (32% of applied) ¹ SDS-19221 ² SDS-46851 ² SDS-47523/ SDS-47524 ² SDS-47425 ²
Anaerobic soil metabolism	SDS-3701 (43% of applied) ¹
Anaerobic aquatic metabolism	SDS-3701 (30-40% of applied) ² SDS-19221 (7% of applied) ² SDS-46851 (3% of applied) ² SDS-47523/ SDS-47524 (9% of applied) ² SDS-47425 (4% of applied) ²
Aerobic aquatic metabolism	SDS-67042 (25-30% of applied) ² SDS-67042 sulfoxide (15% of applied) ² SDS-3701(5-10% of applied) ² SDS-66432 ² SDS-66382 ² SDS-13353 ²

- 1- (EPA 2007)
- 2- (EPA 1999)

Exposure of salmonid habitats to the stressors of the action

Co-occurrence associated with pesticide uses.

Rights-of-way uses are authorized for 2,4-D, triclopyr BEE, diuron, and linuron. These use sites are the most difficult to analyze as they are not tied to a particular land-use class. EPA classifies three specific kinds of rights-of-way: highway, railroad, and utility (including pipeline) (EPA 2003). By definition, they are tied to the transportation of goods and services, which cross urban, agricultural, and wilderness areas alike. Highways and utilities are ubiquitous and rights-of-way applications are likely to occur during the freshwater residence of all of the listed Pacific salmonids (*Appendix 6*). As such, we make the reasonable assumption that they are present in all ESUs to varying degrees. These uses likely have less of an impact in less populated areas and for species that spend less time in freshwater habitats. This approach is consistent with EPA's Biological assessments: *Non-agricultural usages cannot be broken down by state or county as can agricultural usages. However, because they are major uses of 2,4-D, we must assume that they are likely to occur in salmonid Evolutionarily Significant Units (ESUs) in the Pacific Northwest and California* (EPA 2004).

We evaluated co-occurrence of listed salmonids with other uses of the six pesticides by comparing the spatial and temporal distribution of salmon (*Appendix 5 and 6*) with potential use of pesticides based on label specifications. To evaluate areal extent of application sites near salmon-bearing waters, NMFS used a GIS overlay containing landuse classifications and salmon distributions to determine overlap. Because cropping patterns and registered use sites may change over time, landuse classifications (agricultural, forestry, urban/developed) are used rather than specific crops. Details of the GIS analysis and the maps are provided in *Appendix 5*. A summary of our findings is presented in Table 85. "NA" denotes uses that are not applicable because they are not

authorized through labeling. “Y” indicates both spatial and temporal overlap of potential pesticide use with species presence. “N” denotes labeled uses are authorized but spatial or temporal overlap with the species is lacking. Most species are present in the freshwater year-round in some lifestage. The only exceptions include the two Chum ESUs and California Coastal Chinook salmon; these species occur in freshwater 9 – 11 months of the year. Additionally, all of the ESUs/DPSs contained pesticides use sites within the watersheds where the species spawn and rear. Considering that all listed Pacific salmonid ESUs/DPSs use watersheds where the use of the six a.i.s are authorized and that these pesticides are permitted for use in close proximity to salmonid habitats, we expect all listed Pacific salmonid ESUs/DPSs and their designated critical habitats may be exposed to the stressors from one or more of these authorized uses.

Table 85. Co-occurrence of listed Pacific salmonids with potential application of pesticides to use sites within the salmon's freshwater distribution.

		Temporal overlap of ESU and labeled use of pesticide					
Pesticide Use Site	Spatial coverage within species spawning and rearing habitat	2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chloro-thalonil
Puget Sound Chinook Salmon							
Aquatic	17.3%	Y	NA	NA	NA	NA	NA
Cropland	0.7%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	3.3%	Y	Y	NA	NA	NA	NA
Forest	50.4 %	Y	Y	NA	NA	NA	Y
Wetland	3.2%	Y	NA	NA	NA	NA	NA
Residential/Industrial	14.8%	Y	Y	Y	NA	Y	Y
Lower Columbia River Chinook Salmon							
Aquatic	4.6%	Y	NA	NA	NA	NA	NA
Cropland	2.0%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	3.9%	Y	Y	NA	NA	NA	NA
Forest	56.8%	Y	Y	NA	NA	NA	Y
Wetland	4.3%	Y	NA	NA	NA	NA	NA
Residential/Industrial	13.3%	Y	Y	Y	NA	Y	Y
Upper Columbia River Spring Run Chinook Salmon							
Aquatic	1.0%	Y	NA	NA	NA	NA	NA

Pesticide Use Site	Spatial coverage within species spawning and rearing habitat	Temporal overlap of ESU and labeled use of pesticide					
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chloro-thalonil
Cropland	3.5%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	1.8%	Y	Y	NA	NA	NA	NA
Forest	44.9%	Y	Y	NA	NA	NA	Y
Wetland	0.8%	Y	NA	NA	NA	NA	NA
Residential/Industrial	4.7%	Y	Y	Y	NA	Y	Y
Snake River Fall-Run Chinook Salmon							
Aquatic	0.6%	Y	NA	NA	NA	NA	NA
Cropland	14.6%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	0.2%	Y	Y	NA	NA	NA	NA
Forest	49.2%	Y	Y	NA	NA	NA	Y
Wetland	0.2%	Y	NA	NA	NA	NA	NA
Residential/Industrial	1.4%	Y	Y	Y	NA	Y	Y
Snake River Spring-Summer Run Chinook Salmon							
Aquatic	0.4%	Y	NA	NA	NA	NA	NA
Cropland	6.6%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	0.8%	Y	Y	NA	NA	NA	NA
Forest	47.7%	Y	Y	NA	NA	NA	Y
Wetland	0.3%	Y	NA	NA	NA	NA	NA
Residential/Industrial	1.7%	Y	Y	Y	NA	Y	Y
Upper Willamette River Chinook Salmon							
Aquatic	0.6%	Y	NA	NA	NA	NA	NA
Cropland	10.6%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	16.2%	Y	Y	NA	NA	NA	NA
Forest	49.0%	Y	Y	NA	NA	NA	Y
Wetland	2.3%	Y	NA	NA	NA	NA	NA
Residential/Industrial	9.0%	Y	Y	Y	NA	Y	Y
California Coastal Chinook Salmon							
Aquatic	0.6%	Y	NA	NA	NA	NA	NA
Cropland	1.0%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	0.9%	Y	Y	NA	NA	NA	NA
Forest	63.2%	Y	Y	NA	NA	NA	Y
Wetland	0.3 %	Y	NA	NA	NA	NA	NA
Residential/Industrial	5.5 %	Y	Y	Y	NA	Y	Y

Pesticide Use Site	Spatial coverage within species spawning and rearing habitat	Temporal overlap of ESU and labeled use of pesticide					
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chloro-thalonil
Industrial							
Central Valley Spring-Run Chinook Salmon							
Aquatic	1.4%						
Cropland	21.3%	Y	NA	NA	NA	NA	NA
Pasture/Rangeland	3.2%	Y	Y	Y	Y	Y	Y
Forest	20.4%	Y	Y	NA	NA	NA	NA
Wetland	3.1%	Y	Y	NA	NA	NA	Y
Residential/Industrial	10.8%	Y	NA	NA	NA	NA	NA
Sacramento River Winter-run Chinook							
Aquatic	1.4%	Y	NA	NA	NA	NA	NA
Cropland	21.3%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	3.2%	Y	Y	NA	NA	NA	NA
Forest	20.4%	Y	Y	NA	NA	NA	Y
Wetland	3.1%	Y	NA	NA	NA	NA	NA
Residential/Industrial	10.8%	Y	Y	Y	NA	Y	Y
Columbia River Chum Salmon							
Aquatic	5.8%	Y	NA	NA	NA	NA	NA
Cropland	1.9%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	4.7%	Y	Y	NA	NA	NA	NA
Forest	50.8%	Y	Y	NA	NA	NA	Y
Wetland	2.5%	Y	NA	NA	NA	NA	NA
Residential/Industrial	14.9%	Y	Y	Y	NA	Y	Y
Hood Canal Chum Salmon							
Aquatic	15.5%	Y	NA	NA	NA	NA	NA
Cropland	0.0%	N	N	N	N	N	N
Pasture/Rangeland	1.4%	Y	Y	NA	NA	NA	NA
Forest	61.0%	Y	Y	NA	NA	NA	Y
Wetland	2.6%	Y	NA	NA	NA	NA	NA
Residential/Industrial	8.9%	Y	Y	Y	NA	Y	Y
Lower Columbia River Coho Salmon							
Aquatic	4.0%	Y	NA	NA	NA	NA	NA
Cropland	2.1%	Y	Y	Y	Y	Y	Y
Pasture/	4.0%	Y	Y	NA	NA	NA	NA

Pesticide Use Site	Spatial coverage within species spawning and rearing habitat	Temporal overlap of ESU and labeled use of pesticide					
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chloro-thalonil
Rangeland							
Forest	59.3%	Y	Y	NA	NA	NA	Y
Wetland	3.6%	Y	NA	NA	NA	NA	NA
Residential/Industrial	11.7%	Y	Y	Y	NA	Y	Y
Oregon Coast Coho Salmon							
Aquatic	0.7%	Y	NA	NA	NA	NA	NA
Cropland	0.2%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	3.1%	Y	Y	NA	NA	NA	NA
Forest	70.8%	Y	Y	NA	NA	NA	Y
Wetland	1.8%	Y	NA	NA	NA	NA	NA
Residential/Industrial	6.6%	Y	Y	Y	NA	Y	Y
Southern Oregon and Northern California Coast Coho Salmon							
Aquatic	0.4%	Y	NA	NA	NA	NA	NA
Cropland	1.0%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	1.6%	Y	Y	NA	NA	NA	NA
Forest	67.6%	Y	Y	NA	NA	NA	Y
Wetland	0.4%	Y	NA	NA	NA	NA	NA
Residential/Industrial	4.2%	Y	Y	Y	NA	Y	Y
Central California Coast Coho Salmon							
Aquatic	1.5%	Y	NA	NA	NA	NA	NA
Cropland	2.2%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	0.1%	Y	Y	NA	NA	NA	NA
Forest	55.6%	Y	Y	NA	NA	NA	Y
Wetland	0.4%	Y	NA	NA	NA	NA	NA
Residential/Industrial	9.4%	Y	Y	Y	NA	Y	Y
Ozette Lake Sockeye Salmon							
Aquatic	10.3%	Y	NA	NA	NA	NA	NA

Pesticide Use Site	Spatial coverage within species spawning and rearing habitat	Temporal overlap of ESU and labeled use of pesticide					
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chloro-thalonil
Cropland	0.0%	N	N	N	N	N	N
Pasture/ Rangeland	0.0%	N	N	NA	NA	NA	NA
Forest	56.8%	Y	Y	NA	NA	NA	Y
Wetland	3.3%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	1.1%	Y	Y	Y	NA	Y	Y
Snake River Sockeye Salmon							
Aquatic	1.4%	Y	NA	NA	NA	NA	NA
Cropland	0.0%	Y ¹	Y ¹	Y ¹	Y ¹	Y ¹	Y ¹
Pasture/ Rangeland	0.9%	Y	Y	NA	NA	NA	NA
Forest	57.2%	Y	Y	NA	NA	NA	Y
Wetland	3.8%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	1.1%	Y	Y	Y	NA	Y	Y
Puges Sound Steelhead							
Aquatic	17.3%	Y	NA	NA	NA	NA	NA
Cropland	0.7%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	3.3%	Y	Y	NA	NA	NA	NA
Forest	50.4%	Y	Y	NA	NA	NA	Y
Wetland	4.5%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	14.8%	Y	Y	Y	NA	Y	Y
Lower Columbia River Steelhead							
Aquatic	1.9%	Y	NA	NA	NA	NA	NA
Cropland	2.5%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	4.6%	Y	Y	NA	NA	NA	NA
Forest	61.1%	Y	Y	NA	NA	NA	Y
Wetland	2.6%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	12.2%	Y	Y	Y	NA	Y	Y
Upper Willamette River Steelhead							

Pesticide Use Site	Spatial coverage within species spawning and rearing habitat	Temporal overlap of ESU and labeled use of pesticide					
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chloro-thalonil
Aquatic	0.5%	Y	NA	NA	NA	NA	NA
Cropland	14.6%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	20.0%	Y	Y	NA	NA	NA	NA
Forest	40.3%	Y	Y	NA	NA	NA	Y
Wetland	2.6%	Y	NA	NA	NA	NA	NA
Residential/Industrial	10.1%	Y	Y	Y	NA	Y	Y
Middle Columbia River Steelhead							
Aquatic	0.8%	Y	NA	NA	NA	NA	NA
Cropland	17.1%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	1.2%	Y	Y	NA	NA	NA	NA
Forest	26.6%	Y	Y	NA	NA	NA	Y
Wetland	0.7%	Y	NA	NA	NA	NA	NA
Residential/Industrial	3.3%	Y	Y	Y	NA	Y	Y
Upper Columbia River Steelhead							
Aquatic	1.5%	Y	NA	NA	NA	NA	NA
Cropland	13.1%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	1.8%	Y	Y	NA	NA	NA	NA
Forest	33.3%	Y	Y	NA	NA	NA	Y
Wetland	0.8%	Y	NA	NA	NA	NA	NA
Residential/Industrial	4.4%	Y	Y	Y	NA	Y	Y
Snake River Basin Steelhead							
Aquatic	0.4%	Y	NA	NA	NA	NA	NA
Cropland	8.2%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	0.6%	Y	Y	NA	NA	NA	NA
Forest	52.1%	Y	Y	NA	NA	NA	Y
Wetland	0.3%	Y	NA	NA	NA	NA	NA
Residential/Industrial	1.6%	Y	Y	Y	NA	Y	Y

Pesticide Use Site	Spatial coverage within species spawning and rearing habitat	Temporal overlap of ESU and labeled use of pesticide					
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chloro-thalonil
Industrial							
Northern California Steelhead							
Aquatic	0.6%	Y	NA	NA	NA	NA	NA
Cropland	0.1%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	1.0%	Y	Y	NA	NA	NA	NA
Forest	68.2%	Y	Y	NA	NA	NA	Y
Wetland	0.3%	Y	NA	NA	NA	NA	NA
Residential/Industrial	4.4%	Y	Y	Y	NA	Y	Y
Central California Coast Steelhead							
Aquatic	8.4%	Y	NA	NA	NA	NA	NA
Cropland	2.9%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	0.2%	Y	Y	NA	NA	NA	NA
Forest	28.7%	Y	Y	NA	NA	NA	Y
Wetland	2.6%	Y	NA	NA	NA	NA	NA
Residential/Industrial	22.1%	Y	Y	Y	NA	Y	Y
California Central Valley Steelhead							
Aquatic	1.2%	Y	NA	NA	NA	NA	NA
Cropland	26.9%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	5.0%	Y	Y	NA	NA	NA	NA
Forest	15.9%	Y	Y	NA	NA	NA	Y
Wetland	2.1%	Y	NA	NA	NA	NA	NA
Residential/Industrial	9.7%	Y	Y	Y	NA	Y	Y
South-Central California Coast Steelhead							
Aquatic	0.7%	Y	NA	NA	NA	NA	NA
Cropland	7.1%	Y	Y	Y	Y	Y	Y
Pasture/Rangeland	1.1%	Y	Y	NA	NA	NA	NA
Forest	19.9%	Y	Y	NA	NA	NA	Y

Pesticide Use Site	Spatial coverage within species spawning and rearing habitat	Temporal overlap of ESU and labeled use of pesticide					
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chloro-thalonil
Wetland	0.9%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	9.6%	Y	Y	Y	NA	Y	Y
Southern California Steelhead							
Aquatic	0.7%	Y	NA	NA	NA	NA	NA
Cropland	4.1%	Y	Y	Y	Y	Y	Y
Pasture/ Rangeland	0.8%	Y	Y	NA	NA	NA	NA
Forest	9.3%	Y	Y	NA	NA	NA	Y
Wetland	0.6%	Y	NA	NA	NA	NA	NA
Residential/ Industrial	33.9%	Y	Y	Y	NA	Y	Y

1-Although cropland does not occur within spawning and rearing habitat of this species, exposure from agricultural uses is expected given authorized uses within the species migration corridor.

Modeling: Estimates of Exposure to the six a.i.s

EPA exposure estimates

EPA's BEs indicate that pesticides containing the six a.i.s are approved for a variety of uses (Table 86). All are approved for use on agricultural crops. Some are also approved for use on other sites such as forestry, rights-of-way, golf courses, nurseries, parks, residential areas, noncrop agricultural lands, aquatic habitats, and rangeland.

Table 86. Summary of use sites approved on active labels.

Active Ingredient	Aquatic	Cropland	Pasture/ Rangeland	Forestry	Residential/ Industrial	Rights of Way
2,4-D	Yes	Yes	Yes	Yes	Yes	Yes
Triclopyr BEE	No	Yes ¹	Yes	Yes	Yes	Yes
Diuron	No ²	Yes	No ³	No	Yes	Yes
Linuron	No	Yes	No	No	No	Yes ⁴
Captan	No	Yes	No	No	Yes ⁵	No
Chlorothalonil	No	Yes	No	Yes	Yes ⁶	No
1- Commercially grown turf, ornamentals, and abandoned orchards 2- Approved use sites include ornamental ponds and dry irrigation ditches 3- Approved use sites include hay and fallow land, alfalfa, and general weed control in noncrop and nontimber areas 4- Approved use sites include roadsides and fence rows 5- Approved use sites include golf course, ornamentals, and paints 6- Forestry uses approved for Washington state						

The BEs for the six a.i.s evaluated some, but not all registered uses of the compounds (Table 87). In general, the BEs provided few estimates of exposure given the number and variety of uses currently authorized.

Table 87. Examples of current registered uses of the six a.i.s and the exposure method used by EPA in salmonid BEs.

Active Ingredient	Examples of Registered Uses	Exposure Methods Applied in BEs
2,4-D	Crops: cereal grains, field and pop corn, sweet corn, sorghum, soybeans, sugarcane, rice, apples, pears, cherries, peaches, plums, apricots, nut orchards, pistachios, filberts, potatoes, asparagus, hops, strawberries, blueberries, grapes, cranberries, citrus, clover, cottonwood and poplar trees grown for pulp,	PRZM-EXAMS for apples, corn, filberts, wheat RICE Model estimates for rice
	Other use sites: forestry, pastures, rangeland, fallow land and crop stubble, grass grown for seed or sod, irrigation ditch banks, abandoned orchards, grasslands not in agricultural production, ornamental turf, tree and brush control, non-cropland such as fencerows, hedgerows, roadsides, ditches, rights-of-way, utility power lines, railroads, airports, industrial sites, and other non-crop areas, and aquatic herbicide uses	PRZM-EXAMS estimates for pastures and turf; direct application for aquatic herbicide uses
Triclopyr BEE	Crops: sod/turf farms	GENEEC Ornamental lawns and turf

Active Ingredient	Examples of Registered Uses	Exposure Methods Applied in BEs
	Other use sites: range and pastures; golf course and residential turf; rights-of-way; industrial areas; noncrop agricultural areas such as abandoned orchards, around farm buildings, fence rows, roads, and ditch banks; forestry;	GENEEC: agricultural and nonagricultural use sites; DIRECT APPLICATION: assumed for forestry applications
Diuron	Crops: alfalfa, apple, artichoke, asparagus, barley, blueberry, caneberry, gooseberry, blackberry, boysenberry, dewberry, loganberry, raspberry, citrus, corn, cotton, filberts, grape, grass seed, oats, olives, papaya, peas, peach, pear, pecan, peppermint, spearmint, red clover, sorghum, tree planting, walnut, wheat, lilly bulbs, triticale.	GENEEC: grape, citrus, alfalfa, peaches, walnuts, apples, pears, pecans, cotton PRZM-EXAMS: grass seed, walnuts, apple, alfalfa, citrus
	Other use sites: irrigation and drainage systems when water is not present, impervious surfaces, fence lines, rights-of-way (pipelines, powerlines, railway lines, roads), footpaths, in timber yards and storage areas, around commercial, industrial and farm buildings, electrical substations, and petroleum storage tanks. It has some use as an algicide in ornamental ponds, fountains, and aquaria, but not natural water bodies. It may be used as a mildewicide in paints used on buildings and structures.	GENEEC: rights-of-way, irrigation and drainage ditches
Linuron	Crops: asparagus, bulbs, carrots, celery, corn, kenaf, marigolds seed, parsley seed, parsnips, sorghum, soybeans, wheat, post-harvest crop stubble and fallow lands	Carrots
	Other use sites: Non-crop areas such as roadsides and fence rows	None reported
Captan	Crops: alfalfa, clover, lespedeza, trefoil, almond, apple, apricot, artichoke, azalea, barley, beans, beets, begonias, blackberry, blueberry, blue grass, bassica, cabbage, canola, cauliflower, camellias, cantaloupe, cucumber, carnation, carrot, cherry, chrysanthemum, cilantro, cole crops, cauliflower, conifers, cotton, collard, corn, cowpeas, crucifer, cucurbits, dewberries, dichondra, eggplant, flax, gladiolus, ginseng, honeydew, kale, lentils, lettuce, milo, mustard, nectarines, oats, onion, okra, peach, plum, prune, peanut, pea, pepper, potato, radish, raspberry, rye, rutabaga, strawberry, Swiss chard, soybean, spinach, squash, watermelon, pumpkin, muskmelon, safflower, sunflower, sesame, sorghum, sugar beets, tomatillo, tomato, turnip, wheat	PRZM-EXAMS for almond, apple, peaches, prune, cherry, and blueberry
	Other use sites: ornamental grasses, golf course turf, paint additive, roses, greenhouse soil	GENEEC: turf grass
Chlorothalonil	Crops: almonds, apricot, asparagus, beans, blueberry, broccoli, brussels sprouts, cabbage, cucumber, cantaloupe, carrot, celery, chayote,	PRZM-EXAMS/GENEEC: cucurbits, peanuts,

Active Ingredient	Examples of Registered Uses	Exposure Methods Applied in BEs
	cherry, chickpeas, chinese waxgourd, corn, cranberry, eggplant, filberts, flowering bulbs, ginseng, grass grown for seed, groundcherry, honeydew melon, horseradish, lentil, lupine, mango, mint, muskmelon, okra, onion, papaya, parsnips, passionfruit, peach, peanut, peppers, pepino, persimmon, pistachio, plum, potato, prune, rhubarb, soybean, squash, strawberry, sugar beets, tomato, yam, watermelon	potatoes, tomatoes, cherries, papaya, cranberries
	Other use sites: golf course, forestry, lawns around commercial and industrial buildings, athletic fields, landscaping in residential, public, parks, and recreational areas, ornamentals, wood preservative (stains, paints)	PRZM-EXAMS: turf

EPA's BEs provided EECs for the six a.i.s in surface water. These EECs were generated using the PRZM-EXAMS model and used as expected concentrations of the six a.i.s for all aquatic habitats where listed salmonids and their prey reside (Table 88). However, no exposure estimates were provide for other identified stressors of the action including inert/other ingredients, other active ingredients with formulations, and for all of the toxic degradates identified. These missing estimates introduce substantial uncertainty into the exposure analysis. The PRZM-EXAMS model generates pesticide concentrations for a generic "farm pond". The pond is assumed to represent all aquatic habitats including rivers, streams, floodplain habitats, estuaries, and near shore ocean environments. EPA's BEs indicate that the PRZM-EXAMS scenarios provide "worst-case" estimates of salmonid exposure and EPA "believes that the EECs from the farm pond model do represent first order streams, such as those in headwaters areas" used by listed salmon. However, listed salmonids use aquatic habitats with physical characteristics that would be expected to yield higher pesticide concentrations than would be predicted with the "farm pond" based model. Juvenile salmonids rely upon a variety of floodplain habitats that are critical to rearing. All listed salmonids use shallow, low flow habitats at some point in their life cycle (Table 71). Below, we discuss the utility of the EECs for the current consultation. NMFS presents information that indicates the EECs do not represent worst-case environmental concentrations to which listed Pacific salmonids may be exposed.

Finally, NMFS provides additional modeling estimates to evaluate potential exposure in floodplain habitats used by salmonids.

Table 88. PRZM-EXAMS exposure estimates from EPA's salmonid BEs.

Use Site Scenario: crop, state (surrogate scenario)	Application: rate (lbs a.i./A)/ method/ number of applications	Acute EEC peak (µg/L)	Chronic EEC 60-d average (µg/L)
2,4-D			
Pasture, CA (CA alfalfa)	2.88 ¹ /aerial/4 ⁷	23 ¹	18.5 ¹
Pasture, OR (OR wheat)	2.88 ¹ /aerial/2 ⁷	23.9 ¹	19.2 ¹
Turf, CA (OR grass seed)	2.85 ¹ /aerial/2 ⁷	14.2 ¹	9.4 ¹
Turf, OR (OR grass seed)	2.85 ¹ /aerial/2 ⁷	19.3 ¹	15.6 ¹
Wheat, CA (CA alfalfa)	2.88 ¹ /aerial/1 ^{7, 8}	3.7 ¹	3.0 ¹
Wheat, OR	1.25 ¹ /NR/1 ⁸	9.0 ¹	7.5 ¹
Corn, CA	1.00 ¹ /NR/3 ⁸	9.7 ¹	8.2 ¹
Filberts, OR	1.00 ¹ /NR/4	8.8 ¹	7.4 ¹
Apples, OR	2.00 ¹ /NR/2	12.2 ¹	9.9 ¹
Rice ²	1.5 ¹ /NR/1 ⁸	1431 ¹	NR ³
Triclopyr BEE			
Pastures and Rangeland (GENEEC) ⁴	1/ground/1; 1/aerial/1 ⁸	19;20	NR ³
Turf and ornamental lawns (GENEEC) ⁴	2/ground spray/1 ⁸	38	NR ³
Nonagricultural areas (GENEEC) ⁴	8/ground/1; 8/aerial/1	152; 152	NR ³
Forests (direct application) ⁵	2-8/direct application/1 ^{7, 8}	1,468-5,872	NR ³
Drainages system, channeled water (direct application) ⁵	1/direct application/1; 8/direct application/1	734; 5,872	NR ³
Diuron			
Grape, citrus, apples, pears, pecan (GENEEC) ⁴	3.2/ ground/1 ⁸	110	62
Alfalfa (GENEEC) ⁴	3.2/ ground/1; 3.2/aerial/1 ⁷	110; 116	62; 66
Peaches, walnuts (GENEEC) ⁴	4/ground/1 ^{7, 8}	137	78
Cotton	2/ ground/1; 2/aerial/1 ⁷	69; 73	39; 41

Use Site Scenario: crop, state (surrogate scenario)	Application: rate (lbs a.i./A)/ method/ number of applications	Acute EEC peak (µg/L)	Chronic EEC 60-d average (µg/L)
(GENEEC) ⁴			
Rights-of-way, irrigation drainage ditches (GENEEC) ⁴	12/ ground/1; 12/aerial for railroad applications/1	412; 437	234; 248
Grass seed, OR	3.2/ground/1 ⁷	16	NR ³
Apples, CA	3.2/ground/1 ⁸	10	NR ³
Apples, OR	3.2/ground/1 ⁸	43	NR ³
Apples, NY	3.2/ground/1 ⁸	42	41
Alfalfa, CA	3.2/aerial/1 ⁷	22	NR ³
Citrus, CA	3.2/ground/1 ⁸	3	NR
Citrus, FL	3.2/ground/1 ⁸	69	65
Grape, CA	3.2/ground/1 ⁸	13	12
Walnuts, CA (Almonds, CA)	4.0/ground/1 ⁷	3	NR ³
Walnuts, OR	4.0/ground/1 ⁷	21	NR ³
<i>Linuron</i>			
Carrots (Index Reservoir)	1.0/ground/1 ⁸	31.3	NR
<i>Captan</i>			
Almonds, CA	4.5/airblast/5 ⁷	19.8	3.3
Apples, NY	4.0/airblast/8	10.6	2.9
Peaches, CA	4.0/airblast/8	19.5	6.0
Prunes, CA	3.0/airblast/9 ⁷	13.1	3.5
Cherries, CA	2.0/airblast/7	1.1	0.97
Blueberries, MI	2.5/airblast/14	1.7	1.6
Turf (GENEEC)	4.0/ground/8 ⁷	43.4	0.8
<i>Chlorothalonil</i>			
Cucurbits (state NR)	1.75-6.25/NR/1-8 ^{7,8}	18 - 33	0.6 – 3.6
Peanuts (state NR)	1.125/NR/6-9	18 - 26	0.7 – 2.8
Potatoes (state NR)	1.125/NR/6-10	6 - 9	0.4 – 1.5
Tomatoes (state NR)	1.75-2.25/NR/5-8	26 - 44	0.7 – 3.5
Cherries (GENEEC)	3.6-4.1/NR/4-6 ⁷	106 - 122	2 - 8
Papaya (GENEEC)	3/NR/5 ⁷	83-88	2 - 6
Cranberries ⁶ (GENEEC)	5.3/NR/3	82	1 - 5
Turf (GENEEC)	4 – 22.7/NR/1-10 ^{7,8}	48 - 363	4 - 26

- 1- 2,4-D values reported in acid equivalents (a.e.) rather than a.i.
- 2- This estimate derived with EPA RICE Model, not PRZM-EXAMS
- 3- Not reported
- 4- EPA standard scenario, Generic Estimated Environmental Concentration
- 5- Assumed direct application to 6 inches of water (EEC in µg/L = lbs ae applied * 734). However, products containing triclopyr BEE cannot be applied to wetlands, marshes, drainage ditches or other aquatic habitats when surface water is present.
- 6- Estimated concentration in discharge from bog.
- 7- Exceeds current label maximum in number of applications or application rate

8- Less than current label maximum in number of applications or application rate

Very few non-crop uses of pesticides were evaluated in the salmonid BEs. However, NMFS also reviewed aquatic exposure estimates developed by EPA within the red legged frog BEs. Although these estimates were specific to registered uses in California only, they provided surface water estimates for pesticides authorized for non-crop uses that were not included in the BEs for listed salmon (Table 89).

Table 89. Estimates of pesticide concentrations in surface waters from California red legged frog BEs.

a.i.	PRZM-EXAMS exposure estimates for surface water ($\mu\text{g/L}$) ¹		
		Non-Crop	Crops
2,4-D acid/salt	peak	Aquatic weeds: 4,000 Ornamentals, forestry, rights-of-way: 6 - 47	0.08 – 33 (rice:1431)
	60-d avg	Aquatic weeds: 2,610 Ornamentals, forestry, rights-of-way: 5 - 39	0.07 - 27
2,4-D Esters	peak	Pastures, forestry, ornamentals, other: 1.3 - 13	0.55 - 5.5
	60-d avg	NR ²	NR ²
Triclopyr BEE ³	peak	Noncrop agricultural: 65 - 990 ³ Pastures: 33 - 395 ³ Ornamentals: 5 - 35 ³ Golf course turf: 270 ³ Residential lawns: 75 - 1499 ³	21 - 165 ³
	60-d avg	Noncrop agricultural: 50 - 794 ³ Pastures: 25 - 322 ³ Ornamentals: 4 - 29 ³ Golf course turf: 220 ³ Residential lawns: 61 - 1172 ³	16 - 134 ³
Diuron	peak	Noncrop agriculture, industrial areas, irrigation and drainage systems, ornamentals, rights-of-way, paved areas: 37- 4911	5 - 140
	60-d avg	Noncrop agriculture, industrial areas, irrigation and drainage systems, ornamentals, rights-of-way, paved areas: 24 - 3428	3 - 103
Linuron	peak	Rights-of-way, impervious surfaces: 60 - 337	2.6 - 41
	60-d avg	Rights-of-way, impervious surfaces: 39 - 211	1.8 - 31
Captan	peak	Golf course turf, ornamental grasses: 3.6 - 29	<0.001 - 21.6
	60-d avg	Golf course turf, ornamental grasses: 0.08 – 1.09	<0.001 – 0.06
Chlorothalonil	peak	Conifers, golf course and general turf, ornamentals: 19 - 274	3 - 69
	60-d avg	Conifers, golf course and general turf, ornamentals: 12 - 146	2 - 43

1- 2,4-D and triclopyr values expressed in a.e./L

2- NR = not reported

3- These values are based on the application rates approved on active labels of triclopyr BEE and triclopyr TEA on October 19, 2009. Some of these labels did not conform to the

1998 triclopyr RED. Additionally, the maximum use rate of triclopyr TEA on some use sites is 9 lbs a.e./A versus 8 lbs a.e./A for triclopyr BEE.

Utility of EPA-derived EECs for defining exposure to Pacific salmonid habitats

As described in the *Approach to the Assessment* section, our exposure analysis begins at the organism (individual) level of biological organization. We consider the life stage and life histories of the individuals likely to be exposed. This scale of assessment is essential as adverse effects to individuals may result in population-level consequences, particularly for populations of extremely low abundance, (*i.e.* threatened and endangered species). Characterization of impacts to an individual's fitness is necessary to assess potential impacts to populations, and ultimately to the species. To assess risk to individuals, we must consider the highest concentrations to which any individuals of the population may be exposed. Several lines of evidence discussed below suggest that EECs in the BEs underestimate exposure of some listed salmonids and designated critical habitat.

Although EPA characterized these exposure estimates as “worst case” in the BEs, it has also acknowledged that measured concentrations in the environment sometimes exceed PRZM-EXAMS EECs (EPA 2007). EPA has subsequently clarified that rather than providing worst case estimates, PRZM-EXAMS estimates are high end estimates for the vast majority of applications and aquatic habitats (EPA 2007). NMFS agrees that the model is designed to produce upper bound exposure estimates for some, but not all aquatic habitats.

Recent formal consultation and reviews of EPA informal consultations by the Services found that concentrations measured in surface water sometimes exceed peak concentrations predicted with PRZM/EXAMS modeling (NMFS 2007; NMFS 2008; USFWS 2008; NMFS 2009). These findings demonstrate that the EECs generated using PRZM-EXAMS can underestimate peak concentrations of active ingredients that occur in salmonid habitats. Consequently, underestimation of exposure and subsequent risk to

species is likely. Below, we discuss the primary reasons why EPA's exposure estimates do not represent worst case exposures to salmonid habitats.

Model outputs are 90th percentile time-weighted averages and are not worst case exposures for all salmonid habitats. It is important to recognize that the PRZM-EXAMS model predicts concentrations based on site-specific assumptions (*e.g.*, rainfall) and that environmental concentrations provided for the estimate do not represent the highest aquatic concentrations predicted given the assumptions. The exposure estimates provided in the BEs are reported as peak concentrations or time-weighted average concentrations (*e.g.*, 21 d and 60 d). Peak concentrations do not represent the maximum concentration predicted by the model. Rather, they represent the 90th percentile of the estimates derived for the given scenario (Lin, Hetrick et al. 1998). Although NMFS agrees this is a relatively protective approach for evaluating exposure in some aquatic habitats, it does not represent the possible "worst case" exposure in many salmonid habitats.

Model inputs did not use maximum application rates, maximum number of applications, and minimum application intervals. We compiled maximum use rates (single and seasonal), maximum number of applications, and minimum application intervals from current labels in the *Description of the Proposed Action*. Several of the PRZM-EXAMS inputs within the BEs (Table 88) were not consistent with the maximum application rates and maximum number of applications allowed.

Few application scenarios were assessed relative to the number of approved uses. The salmonid BEs provide exposure estimates for only a portion of the labeled uses (Table 87). For example, chlorothalonil is approved for use on more than 50 agricultural commodities yet the BE provides estimates for only 6 agricultural crops. Estimates are provided for turf but not for several other types of uses such as applications to ornamentals and the use of chlorothalonil as a wood protectant.

Crop scenarios provided are not representative of the entire action area. The regional scale that the modeled scenarios are intended to represent is unclear. Some of the scenarios were conducted for states outside the distribution of listed salmonids. Others did not provide information on geographic locations simulated (e.g., county, state, region, etc.). The amount of rainfall and other site-specific input assumptions can have large impacts on predicted exposure. For example, assumed site-specific inputs resulted in differences in predicted aquatic concentrations that exceed 20-fold, even when the same application rates and methods of application were evaluated (Table 88). Large differences in site-specific variables that influence transport of pesticides to aquatic habitats exist across the states of California, Idaho, Oregon, and Washington. The relatively small numbers of simulations are not likely representative of all of the pesticide use sites throughout these states. For example, the 2,4-D BE assessed scenarios developed as representative for California and Oregon while no simulations were provided for Idaho or Washington. NMFS also questions whether input values were adequate to represent the geographic variability throughout the action area. Site-specific meteorological and soil conditions vary greatly throughout the four states where listed salmonids are distributed and crops are grown. The BEs did not indicate how sites were selected and how well scenarios represented the range of conditions throughout the four states. Without a description of EPA's scenario selection, it is difficult to determine the representativeness of scenario estimates for the complete range of crop uses.

Crop scenarios do not consider application of more than one pesticide. All six a.i.s are formulated with other a.i.s to produce end-use pesticide products. More than 50 of the pesticide formulations involved in this consultation contain multiple pesticidal a.i. For example, a single 2,4-D product may contain as many as four a.i.s., and several captan formulations also contain malathion and/or carbaryl, neurotoxic compounds that both inhibit cholinesterase and whose use has previously been determined to jeopardize or adversely affect listed salmonids (NMFS 2008; NMFS 2009). Additionally, a majority of the labels provide recommendations for co-application (tank mixtures) of more than on

pesticide to improve product efficacy. Yet, the BEs did not provide exposure estimates for multiple a.i.s associated with either product mixtures or tank mixtures.

NMFS exposure estimates for floodplain habitats

Model inputs used in BEs are not representative of most vulnerable salmonid habitats.

The EECs within EPA's BEs were derived primarily using the PRZM-EXAMS model. The EPA "farm pond" scenario is likely a poor surrogate of many habitats used by listed salmonids that are more susceptible to higher pesticide concentrations given their physical characteristics. Small streams and some floodplain habitats represent examples of habitats used by salmonids that can have a lower capacity to dilute pesticide inputs than the farm pond. The PRZM-EXAM estimates assume that a 10-hectare (approximately 25 acres) drainage area is treated and the aquatic habitat is assumed to be static (no inflow or outflow). Pesticide treatment areas of 10-hectares and larger occur frequently in agricultural crops, particularly under pest eradication programs. Additionally, aquatic habitats used by salmon vary in volume and recharge rates and consequently have different dilution capacities to spray drift and runoff events. The assumed drainage area to water volume ratio ($100,000 \text{ m}^2$: $20,000 \text{ m}^3$) is easily exceeded for small water bodies. For example, a one-acre pond with an average depth of 1 m would exceed this ratio for treated drainage areas of approximately five acres in size and larger. The assumed aquatic habitat and size of the treated area for the PRZM-EXAMS scenarios suggest that exposure is underestimated for listed salmonids that use relatively small aquatic habitats with low dilution capacities.

Direct over-spray of pesticides to aquatic habitats

To estimate potential exposure of salmon to pesticides in floodplain and other shallow-water habitats, we first determined the initial average concentrations that will result from a direct overspray of shallow surface water (Table 90). When pesticides are applied directly to aquatic habitats the resulting initial concentration is a function of the amount applied and the volume of the water body. The active labels for the six a.i.s do not

authorize direct application of pesticides to surface water. The only exceptions include the use of 2,4-D products on aquatic habitats for weed control and applications to rice. Labels recommend treatment rates for aquatic weed control target surface water concentrations of up to 4,000µg a.e./L. Considering first order degradation of 2,4-D, this would result in a 60-d average concentration of 2610 µg a.e./L for the farm pond scenario (EPA 2004). EPA provided EECs resulting from use on rice. EPAs model assumes uniform application of the pesticide to a rice paddy and partitioning of the pesticide between water and the upper 1 cm of sediment (EPA 2009). Modeling the maximum seasonal application rate of 1.5 pounds a.e./A results in an estimated concentration in the paddy of 1431 µg a.e./L (EPA 2004). The resulting concentrations in salmon habitat resulting from discharged water from treated paddys will be influenced by the volume of discharge and the physical properties of the habitats (e.g. volume and flow).

Table 90. Average initial concentration of any a.i. in surface water resulting from a direct overspray of aquatic habitat.

Application Rate (lbs a.i. / acre)	Water Depth (meters)	a.i. Concentration in Surface Water (µg/L)
0.25	2	14
0.5	2	28
1	2	56
3	2	168
10	2	560
0.25	1	28
0.5	1	56
1	1	112
3	1	336
10	1	1,121
0.25	0.5	56
0.5	0.5	112
1	0.5	224
3	0.5	673
10	0.5	2,242
0.25	0.3	93
0.5	0.3	187
1	0.3	374
3	0.3	1,121
10	0.3	3,736
0.25	0.1	280

Application Rate (lbs a.i. / acre)	Water Depth (meters)	a.i. Concentration in Surface Water (µg/L)
0.5	0.1	560
1	0.1	1,121
3	0.1	3,363
10	0.1	11,208

Application of pesticides to adjacent terrestrial habitat

Some chlorothalonil products specify a no-application buffer of up to 150 feet to marine and estuarine habitats (EPA Registration No. 66222-149). However, all of the a.i.s may be applied at the immediate edge of freshwater habitats utilized by the listed salmonids. Primary drift is a likely transport mechanism for pesticides applications that occur immediately adjacent to aquatic habitats including shallow floodplain habitats where juvenile salmonids rear and shelter. We derived exposure estimates for floodplain habitats that incorporated label-specified application requirements (Table 91). These estimates were derived using the AgDrift model and estimate downwind deposition from pesticide drift (Teske 2001). This method does not incorporate additional contributions that may occur through the runoff pathway. The drift estimates derived represent average projected drift. Although AgDrift reasonably predicts drift, drift is highly variable and is influenced by site-specific conditions and application equipment (Bird, Perry et al. 2002). Our simulations assumed an aquatic habitat that was 0.1 m deep and 2 m wide. These dimensions are consistent with some of the smaller, and potentially more vulnerable floodplain habitats used by salmonids.

Table 91 Estimated average initial pesticide concentrations in a floodplain habitat that is 2m wide and 0.1m deep using AgDrift.

a.i.	Use Site	Simulation: Rate in lbs a.i./A	Buffer (feet)	Average Initial Concentration in Surface Water (µg/L)
2,4-D amines, acides, salts	Cropland	Ground ¹ : 0.07-2 Air ² : 0.07-2	0	13-368 33-956
	Pasture/Rangeland	Ground ¹ : 1-2 Air ² : 1-2	0	184-368 478-956

a.i.	Use Site	Simulation: Rate in lbs a.i./A	Buffer (feet)	Average Initial Concentration in Surface Water (µg/L)
	Forest	Ground ¹ : 4 Air ² : 4	0	736 1,912
	Residential/Industrial	Ground ¹ : 1.5	0	276
	Rights-of-way	Ground ¹ : 2-4 Air ² : 2-4	0	368-736 956-1,912
2,4-D esters	Cropland	Ground ¹ : 0.05-2 Air ² : 0.05-2	0	9-368 24-956
	Pasture/Rangeland	Ground ¹ : 1-2 Air ² : 1-2	0	184-368 478-956
	Forest	Ground ¹ : 4 Air ² : 4	0	736 1,912
	Residential/Industrial	Ground ¹ : 1.5	0	276
	Rights-of-way	Ground ¹ : 2-4 Air ² : 2-4	0	368-736 956-1,912
Triclopyr BEE ³	Cropland	Ground ¹ : 2-8 ⁴ Air ² : 2-8	0	368 – 1,471
	Pasture/Rangeland	Ground ¹ : 2 Air ² : 2	0	368 956
	Forest	Ground ¹ : 6 Air ² : 6	0	1,103 2,868
	Residential/Industrial	Ground ¹ : 1-8	0	184 – 1,471 478 – 3,824
	Rights-of-way	Ground ¹ : 8 Air ² : 8	0	1,471 3,824
Diuron	Cropland	Ground ¹ : 1.2 - 4 Air ² : 1.2 – 2.4	0	221 – 736 574 – 1,147
	Pasture/Rangeland	NA	NA	NA
	Forest	NA	NA	NA
	Residential/Industrial	Ground ¹ : 12	0	2,207
	Rights-of-way	Ground ¹ : 12	0	2,207
Linuron	Cropland	Ground ¹ : 1 - 4	0	184 - 736
	Pasture/Rangeland	NA	NA	NA
	Forest	NA	NA	NA
	Residential/Industrial	NA	NA	NA
	Rights-of-way	Ground ¹ : 3	0	552
Captan	Cropland	Ground ¹ : 2– 4.5 Air ² : 2– 4.5	0	368 – 828 956 – 2,151
	Pasture/Rangeland	NA	NA	NA
	Forest	NA	NA	NA
	Residential/Industrial	Ground ¹ : 1	0	184

a.i.	Use Site	Simulation: Rate in lbs a.i./A	Buffer (feet)	Average Initial Concentration in Surface Water (µg/L)
	Rights-of-way	NA	NA	NA
Chlorothalonil	Cropland	Ground ¹ : 1– 7.3 Ground ² : 41.25 Air ² : 1– 7.3	0	184 – 1,343 7,586 478 – 3,490
	Pasture/Rangeland	NA	NA	NA
	Forest	Ground ¹ : 4.1 Air ² : 4.1	0	754 1,960
	Residential/Industrial	Ground ¹ : 11.3	0	2,078
	Rights-of-way	NA	NA	NA

1 – Low ground boom spray, ASAE fine to medium/course distribution, 50th percentile estimate

2 – Tier 1 aerial spray, ASAE medium to course droplet distribution

3 – Applying excess dip material as spary to flowering bulbs (Tier 1 aerial spray, ASAE medium to course droplet distribution)

NA – Spray drift calculation is not applicable because use on site not authorized

NMFS exposure estimates for pesticide mixtures

All six of the a.i.s are formulated into pesticide products that contain other a.i.s. More than 50 of the pesticides subject to this consultation contain multiple a.i.s. As an example of potential exposure to pesticide mixtures, we evaluated the use of one of these products, EPA Reg. No. 228-654. This pesticide contains 62.22% diuron and 7.78% imazapyr, another photosynthetic-inhibiting herbicide. It is approved for use on utility and pipeline rights-of-way, highway rights-of-way, railroads, fence-rows, non-irrigation ditchbanks, farmyards, non-crop areas around farm buildings, and other industrial non-crop areas. It can be applied by aerial and ground application methods on these sites at single application rates up to 19 lbs of product per acre (12 lbs diuron and 1.5 lbs imazapyr/acre). The label also recommends that the product be tank mixed with nonionic surfactants, methylated seed oils, silicone-based surfactants, or fertilizer/surfactant blends. Additionally, the label provides specifications for simultaneously applying this product with other herbicides including Roundup®, Oust®, Garlon® Finale®, MSMA, Banvel®, Plateau®, and Arsenal®.

Simulations using the AgDrift model were run according to label specifications to account for potential drift of the 2 a.i.s in the formulation to aquatic habitats. A 2004 Court order for injunctive relief requires implementation of no-spray buffers to certain water containing listed salmon in California, Oregon, and Washington. Buffers of 60 feet for ground applications and 300 feet for aerial applications are in effect until EPA completes its consultation. *Washington Toxics Coalition v. EPA*, C01-132C (W.D. Wash. 1/22/2004). Consequently, we evaluated potential loading via drift with and without the injunctive relief buffers. Results are presented below in Table 92. To simulate a linear right-of-way application, a single aerial swath with a coarse droplet size was assumed. Other input values are reported in *Appendix 8*.

Table 92. AgDrift estimated concentrations of pesticides in surface water adjacent to aerial application at the maximum labeled use rate for EPA Reg. No. 228-654 (12 lbs diuron/A and 1.5 lb imazapyr/A).

Chemical	Buffer	Average initial concentration (µg/L)	
	Ft	EPA-defined pond	NMFS-defined floodplain habitat
With injunctive relief buffers			
diuron	300	1.002	30.8
imazapyr	300	0.125	3.85
Without aquatic habitat buffers			
diuron	0	25.0	2,789
Imazapyr	0	3.12	349
Label prohibits use of product in California			

Monitoring Data: Measured Concentrations of Parent Compounds and Degradates in Surface Waters

We reviewed two types of pesticide monitoring data: 1) ambient data that measure concentrations of pesticides and other contaminants in surface waters, but are not targeted at any specific pesticide application or land cover type, and 2) data from more targeted studies (frequently found in published scientific literature and gray literature), which may be collected in waters near or downstream of agricultural or other pesticide uses. We evaluated data from three central sources: USGS' NAWQA database, state databases maintained by California and Washington and targeted monitoring studies which may not be included in monitoring databases. Neither Oregon nor Idaho currently maintains a state database. Data from Washington include studies conducted by the Washington State Departments of Ecology and Agriculture. The NAWQA data typically are general monitoring data, and sampling stations are distributed across a range of land uses, although some data may be from investigations into specific uses. The California and Washington databases contain data from studies that fall into both categories.

Diuron and linuron may persist in the aquatic environment for several weeks depending on site-specific characteristics that contribute to dissipation. 2,4-D, trichlopyr, captan, and chlorothalonil have relatively short aquatic half-lives in the range of a few days to a

few weeks, and therefore the detection of these compounds is less likely, and more greatly influenced by sample design (e.g. timing of sampling relative to the timing of application or runoff events). In the following section we describe study design considerations for assessing the utility of monitoring data for evaluating exposure of pesticides to salmon.

Monitoring data considerations

Surface water monitoring can provide useful information regarding real-time exposure and the occurrence of environmental mixtures. A primary consideration in evaluating monitoring data is whether the study design is sufficient to address exposure in a qualitative, quantitative, or probabilistic manner. The available monitoring studies were conducted under a variety of protocols and for varying purposes. General water quality monitoring conducted in larger streams and rivers frequently does not capture “peak” concentrations because it is not correlated with applications and/or storm events following those applications and not all habitats types are sampled. This is one of the reasons NMFS did not use available monitoring data for probabilistic modeling (*i.e.*, it likely does not contain the complete range of possible concentrations). Additionally, the monitoring sampling designs and sites do not represent many salmonid species’ ranges (see Figure 61).

Of the monitoring programs discussed, only the Washington State Department of Ecology program was specifically designed to evaluate potential exposure to listed salmonids. This study monitored selected urban and agricultural areas that do overlap with some listed Pacific salmonid habitats in Washington state. This sampling program was intended to evaluate pesticide occurrence in a limited number of salmonid-bearing streams during the pesticide application seasons (Johnson and Cowles 2003). The study design included sampling during the pesticide application season but did not target specific applications of pesticides nor did it target salmonid habitats that would be expected to produce the highest concentrations of pesticides (*e.g.*, shallow off-channel habitat in close proximity to pesticide applications). Sampling was generally conducted

on a weekly basis, so it is likely peak concentrations associated with drift and runoff events were not captured. This monitoring program is discussed in more detail in *Monitoring Data from Washington State*.

Other available monitoring data are also applicable to assessing exposure in listed salmon, but to varying degrees. Common aspects that limit the utility of the available monitoring data as accurate depictions of exposure within listed salmonid habitats include: 1) protocols were not designed to capture peak concentrations or durations of exposure in habitats occupied by listed species; 2) limited utility as a surrogate for other non-sampled surface waters; 3) lack of representativeness of current and future pesticide uses and conditions; and 4) lack of information on actual pesticide use to correlate with observed surface water concentrations.

Protocols not designed to capture peak exposure. The NAWQA monitoring studies contain the largest data set evaluated. However, these studies were designed to evaluate trends in water quality and were not designed to characterize exposure of pesticides to listed salmonids (Hirsch 1988). The NAWQA design does not result in an unbiased representation of surface waters, which limits the ability to make statistical extrapolations to unsampled waters. Also, some agricultural activities and related pesticide uses that may be very important in a particular region may not be represented in the locations sampled. Sampling from the NAWQA studies and other studies reviewed was typically not conducted in coordination with specific applications of the six a.i.s addressed in this Opinion. Similarly, sampling was not designed with consideration to salmon distribution or to target the salmonid habitats most likely to contain the greatest concentrations of pesticides. Given the relatively rapid dissipation of these pesticides in flowing water habitats, it is not surprising that pesticide concentrations from these datasets were generally much lower than those predicted by modeling efforts.

Limited applicability to other locations. Pesticide runoff and drift are influenced by a variety of site-specific variables such as meteorological conditions, soil type, slope, and

physical barriers to runoff and drift. Additionally, surface water variables such as volume, flow, and pH influence both initial concentrations and persistence of pesticides in aquatic habitats. Finally, cropping patterns and pesticide use have high spatial variability. Given these and other site-specific factors, caution should be used when extrapolating monitoring data to other sites.

Representativeness of current and future uses. Pesticide use varies annually depending on regulatory changes, market forces, cropping patterns, and pest pressure. The use of the six a.i.s in California over the recent decade has either shown a general decrease or remained relatively stable. However, pesticide use patterns change annually and may result in either increases or decreases in use of pesticide products for specific uses. There is considerable uncertainty regarding the representativeness of monitoring conditions to forecast future use of products containing these a.i.s.

Lack of information on actual use to correlate with observed concentrations. A common constraint in the monitoring data was lack of information on actual use of pesticides containing the six a.i.s. For example, the ability to relate surface water monitoring data to the proposed action was severely hampered because information on application rates, setbacks/buffers, and applications methods associated with the monitoring were generally not reported. In most cases, the temporal and spatial aspect of pesticide use relative to sampling was not reported, further limiting the utility of the information.

Data Described in USEPA's Biological Evaluations

EPA summarized monitoring data in the BEs, derived mostly from the same sources we have considered. As we considered information from these databases, including the more recent data, we do not reiterate the BE summaries herein.

USGS NAWQA Data for California, Idaho, Oregon, and Washington

We obtained updated data from the USGS NAWQA database to evaluate the occurrence of 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil in surface waters monitored in California, Idaho, Oregon, and Washington. Triclopyr BEE (which rapidly converts to the triclopyr acid/anion) and captan are not on the USGS list of analytes. However, we obtained the available data for some of the identified degradates including the triclopyr acid, a degradate of 2,4-D (2,4-DCP) and two degradates of diuron (3,4-DCA and CPMU). No information was available for captan or its degradates. Land uses associated with the sampling stations included agriculture, forest, rangeland, urban, and mixed use. The database query resulted in approximately 5,400 samples in which one or more of the a.i.s or degradates was an analyte. Approximately 360 unique sampling locations were represented, with sample sites located in 11 NAWQA basins distributed throughout California, Idaho, Oregon and Washington (Figure 61). Some waterbodies and/or basins in this dataset do not contain listed salmonids and several of the species have had no sampling within their freshwater and coastal habitats (Table 93). Most notable are those ESUs/DPSs along the coasts of Oregon and California as well as listed salmonid habitats within Idaho. Available data included samples collected from 1991-2010. More than one third of the stations were sampled only once during the span of 19 years, and a relatively small number of sites accounted for the majority of the data; approximately 75% of the data was collected from 36 sites. The temporal and spatial distribution of sampling is inconsistent with temporal and spatial aspects of salmonid distribution. Consequently, we do not expect the data set to be representative of exposure distributions for listed salmonids.

Table 93. Number of NAWQA sample sites within the distribution of listed Pacific salmonids.

Species	ESU	Kilometers of Stream Inhabited	Sites in Spawning and Rearing Habitat	Sites in Migratory Corridor
Chinook	Puget Sound	3,639.65	39	NA
	Lower Columbia River	2,443.29	15	NA
	Upper Columbia River Spring - Run	1,646.75	0	5
	Ssnake River Fall - Run	1,370.44	0	3
	Ssnake River Spring/Summer - Run	5,288.23	0	3
	Upper Willamette River	3,013.85	43	3
	California Coastal	2,422.44	0	NA
	Central Valley Spring - Run	2,212.94	6	0
	Sacramento River Winter - Run	546.84	6	0
Chum	Hood Canal Summer - Run	141.89	2	NA
	Columbia River	1,162.18	14	NA
Coho	Lower Columbia River	3,307.78	18	NA
	Oregon Coast	10,220.00	0	NA
	Southern Oregon and Northern California Coast	5,619.58	0	NA
	Central California Coast	1,287.78	0	NA
Sockeye	Ozette Lake	70.98	0	0
	Snake River	1,493.94	0	3
Steelhead	Puget Sound	3,849.64	39	NA
	Lower Columbia River	4,302.03	17	1
	Upper Willamette River	3,063.07	26	3
	Middle Columbia River	10,196.80	84	2
	Upper Columbia River	2,143.15	8	2
	Snake River	13,423.40	0	3
	Northern California	5,324.31	0	NA
	Central California Coast	4,620.72	0	NA
	California Central Valley	4,273.66	49	0
	South-Central California Coast	5,104.56	0	NA
	Southern California	3,015.86	2	NA

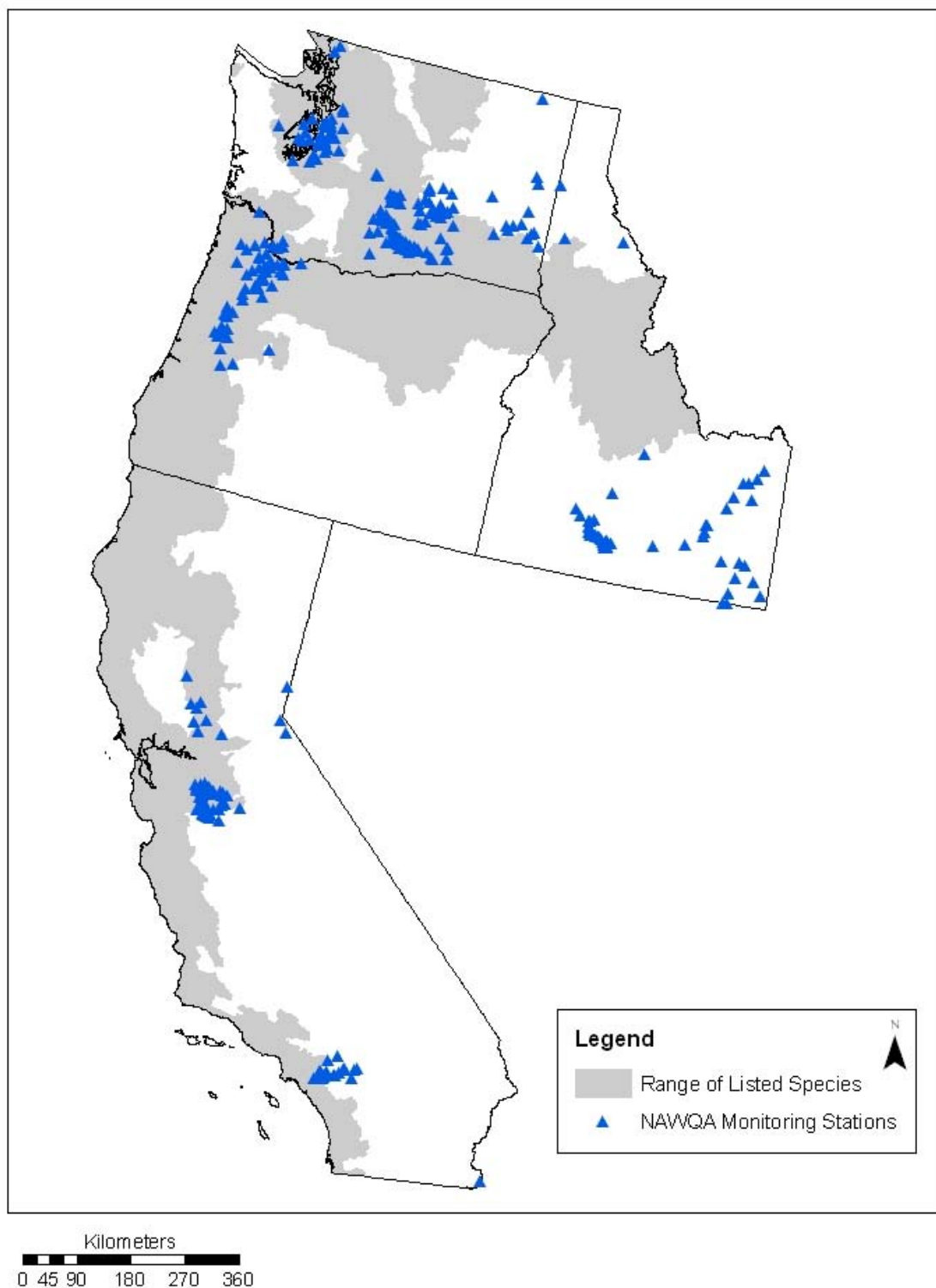


Figure 61. Distribution of NAWQA monitoring sites and listed Pacific salmonids.

The frequency of detection is a combination of the actual occurrence of pesticides in the water and the sampling intensity. NAWQA surface water detections represent the dissolved phase, as the water sample is filtered through a 0.7 micron glass fiber filter. Chemicals transported primarily in the particulate phase would be underreported in this data set. No sediment or tissue data were available from USGS for the six a.i.s. This is also a recognized uncertainty and compromises our ability to determine toxicity of contaminated sediments. Because the USGS monitoring program does not generally coordinate sampling efforts with specific pesticide applications or runoff events, detected concentrations are likely to be lower than actual peak concentrations adjacent to pesticide application sites that occur immediately following drift or a runoff event.

Summary information for quantifiable concentrations of the pesticides addressed in this Opinion (Table 94) and their degradates (

Table 95) are presented below. In the USGS database, non-detects are reported as less than (“<”) the laboratory reporting level (LRL) for that sample. Other than total number of samples (n), summary statistics were calculated on samples not designated as (“<”). The LRL ranges reported were estimated based on “<”-qualified data. Many of the concentrations that could be quantified were designated as “E,” meaning the concentrations were estimated. These data are included in the summary statistics.

Sampling intensity varied considerably among the available analytes and ranged from 5 – 4327 samples collected during the 19 year sampling period. The most commonly occurring were diuron (46%, range 0.002-23.3 µg/L, median 0.111 µg/L), the 3,4-DCA degradate of diuron (41%, range 0.01-0.7 µg/L, median 0.013 µg/L), and 2,4-D (27%, range 0.007-7.66 µg/L, median 0.085 µg/L). Triclopyr (8.7%, range 0.09-3.35 µg/L, median 0.119 µg/L) and CPMU (4.9%, range 0.0037-0.1701 µg/L, median 0.0231 µg/L) were also detected on a frequent basis, while linuron (0.92%, range 0.007-0.68 µg/L, median 0.022 µg/L) and chlorothalonil were not (0.08%, 0.29 µg/L).

Table 94. Concentrations of Parent Pesticides in NAWQA Water Samples for California, Idaho, Oregon, and Washington.

Statistic	2,4-D	Diuron	Linuron	Chlorothalonil
Samples	1274	1364	4327	1188
Percent detections	27.00	45.60	0.92	0.08
LRL range (µg/L)	0.0218 - 0.27	0.009 - 1.25	0.014 - 1.1	0.035 - 0.48
Minimum concentration (µg/L)	0.007	0.002	0.007	0.29
Maximum concentration (µg/L)	7.660	23.3	0.68	0.29
Median concentration	0.085	0.111	0.022	0.29

Table 95. Concentrations of Degradates in NAWQA Water Samples for California, Idaho, Oregon, and Washington.

Statistic	2,4-DCP	Triclopyr Acid	3,4-DCA	CPMU
Samples	5	1366	1309	526
Percent Detections	0	8.71	41.33	4.94
LRL range (µg/L)	5	0.05 – 0.68	0.004 – 0.023	0.0242 – 0.0915
Minimum concentration (µg/L)	ND	0.09	0.01	0.0037
Maximum concentration (µg/L)	ND	3.35	0.7	0.1703
Median concentration	ND	0.119	0.013	0.0231

Monitoring Data from California Department of Pesticide Regulation

We evaluated monitoring data available from the CDPR, which maintains a public database of pesticide monitoring data for surface waters in California. Entries in the database (www.cdpr.ca.gov/docs/emon/surfwtr/surfddata.htm) are from multiple sources, including monitoring conducted by CDPR, USGS (data from the NAWQA program, as well as other studies), state, city, and county water resource agencies; and some non-governmental or inter-governmental groups such as Deltakeeper. The CDPR requires a formal QA/QC protocol for data submitted or does a separate QA/QC review, thus only data subject to appropriate QA/QC procedures are included in the surface water database. Unlike the USGS NAWQA data set, the CDPR database may contain whole water samples as well as filtered samples. If whole water concentrations are reported for

compounds that sorb significantly to the particulate phase, concentrations would appear higher than in a filtered sample, which represents only the dissolved phase. The majority of the studies, which are described in metadata available from CDPR, are not targeted at correlating water concentrations with specific application practices. The database, last updated in June 2008, consists of approximately 270,000 data records. Each record reports a specific sampling site, date, and analyte. The number of records associated with a particular compound is indicative of monitoring intensity rather than actual occurrence in surface waters. In this database, detections below the LOQ are reported as 0 µg/L. Summary statistics were calculated on samples with values above the LOQ.

Some data were available for all a.i.s considered in this opinion, although the number of samples was variable among the analytes. Summary information is provided below in Table 96. Diuron, triclopyr, and 2,4-D were the most frequently detected compounds (14-47% frequency of detection). Linuron was sampled the most frequently, and was detected in about 1% of the samples. Captan and chlorothalonil were not detected in any of the samples, but sampling for these compounds was relatively limited compared to the other analytes.

Table 96. Surface Water Concentrations of Pesticides in CDPR Database.

Statistic	2,4-D	Triclopyr	Diuron	Linuron	Captan	Chlorothalonil
Samples	809	583	1971	2805	220	400
Percent Detections	14.0	17.5	46.6	0.93	0	0
LOQ range (µg/L)	0.0218-0.27	0.0224-0.68	0.009-1.25	0.002-5	0.1	0.035-0.578
Minimum concentration (µg/L)	0.0244	0.0251	0.0035	0.0073	0	0
Maximum concentration (µg/L)	4.42	14.5	160	1.6	0	0
Median concentration	0.16	0.7	0.275	0.275	0	0
Dates	1991-2006	1993-2006	1992-2006	1992-2006	1994-2006	1993-2005
# of Studies	9	7	21	12	2	3

Monitoring Data from Washington State

Data from monitoring studies conducted in the state of Washington are included in Department of Ecology's Environmental Information Management (EIM) database (<http://www.ecy.wa.gov/eim/>). Data in the database are from multiple sources, including state agencies, and may contain whole water samples as well as filtered samples. The EIM requires a formal QA/QC protocol for data submitted or does a separate QA/QC review, thus only data subject to appropriate QA/QC procedures are included. Some of the studies contained in this database may be targeted with respect to specific pesticide uses, while others are more generalized water quality surveys. Some data for all pesticides considered in this Opinion were available, and are shown in Table 99.

The procedure for reporting in the EIM database includes reporting non-detects as the reporting limit for that particular sample, and adding a "U" data qualifier. The reporting limit was not specified in the data accessed by NMFS, thus LOQ ranges were estimated based on "U"-qualified data. Summary statistics were calculated on samples with values above the LOQ (*i.e.*, not qualified with a "U"). Data with a "REJ" qualifier did not meet quality control standards and were not considered. Statistics include data qualified with a "J" (analyte positively identified, resulting value an estimate) and data qualified with an "NJ" (analyte tentatively identified, resulting value an estimate).

In the complete dataset, sample sets consisted of 1282-2370 samples for each of the analytes of interest. 2,4-D acid (48%) , diuron (22%), and triclopyr acid (16%) were the most frequently detected pesticides. Maximum concentrations for these three compounds were 6.57 µg/L, 4.1 µg/L, and 1.3 µg/L, respectively. Linuron, captan, and chlorothalonil were detected 1-3% of the time with maximum concentrations ranging from 0.12 – 0.6 µg/L .

Table 97. Concentrations of Pesticides in Washington EIM Database.

Statistic	2,4D acid	Triclopyr acid	Diuron	Linuron	Captan	Chlorothalonil
Samples	2040	1984	2370	1282	1802	2080
Percent Detections	47.55	16.13	21.90	3.28	1.28	1.88
LRL range (µg/L)	0.03 - 1	0.063-0.25	0.0045 - 2.9	0.0054-5.433	0.033-2.1	0.0056 – 19
Minimum concentration (µg/L)	0.001	0.0028	0.0022	0.03	0.6	0.0027
Maximum concentration (µg/L)	6.57	1.3	4.1	0.12	0.6	0.36
Median concentration (µg/L)	0.063	0.0415	0.043	0.12	0.6	0.12
Dates	1992-2009	1992-2009	1992-2009	1992-2009	1992-2009	1992-2009
# of Studies	31	25	29	8	22	26

Included in the EIM database is a subset of recent monitoring efforts conducted by the Washington State Departments of Ecology and Agriculture in some of Washington's salmon-bearing streams. Final reports for 2003-2008 seasons are publically available on the internet (<http://agr.wa.gov/PestFert/natresources/SWM/default.htm>). Monitoring was conducted in 2009, and is also provided in the database. A separate summary of data from those investigations is provided below. Water samples are not filtered, and thus concentrations reported include pesticides in both dissolved and particulate phases, although the sampling protocol specifies an attempt to avoid collection of excessive particulates (Johnson and Cowles 2003). Whole water concentrations for compounds that sorb significantly to the particulate phase will appear higher than those for a filtered sample, which represents only the dissolved phase.

The Washington program sampled between 0 and 16 sample stations for the six analytes from 2003 - 2009 (Table 98). Sampling stations were located primarily in agricultural-dominated watersheds (Figure 62). A single watershed, the Cedar-Sammamish (Thornton Creek) represented the urban sites. Three sites were sampled in Thornton Creek in 2003, and 2 sites from 2004-2008. Agricultural sites were distributed in four

watersheds (Lower Yakima, Skagit/Samish, Wenatchee and Entiat), but only the Lower Yakima sites have been sampled since 2003. Sites in the Skagit/Sammish watershed were added in 2006 and sites in the Wenatchee and Entiat were added in 2007. Sampling favored the detection of multiple pesticides, rather than peak concentrations in some habitats used by listed salmonids as sampling was not coordinated with pesticide applications at the field scale. Generally, samples were taken weekly between March and September at the various sites, but the specific sampling design has changed somewhat over the years. The limited number and spatial distribution of samples sites does not reflect the distribution of listed ESUs/DPSs in the state and are not expected to represent the full range of habitats and potential exposure of listed salmonids to pesticides.

Table 98. Number of stations sampled for pesticide in Washington Department of Ecology Study.

Sample collection year	2,4D acid	Triclopyr acid	Diuron	Linuron	Captan	Chlorothalonil
2003	9	9	9	0	9	9
2004	6	6	6	0	5	6
2005	6	6	6	0	6	6
2006	11	11	11	11	11	11
2007	16	16	16	16	16	16
2008	16	16	16	16	16	16
2009	4	4	4	4	4	4

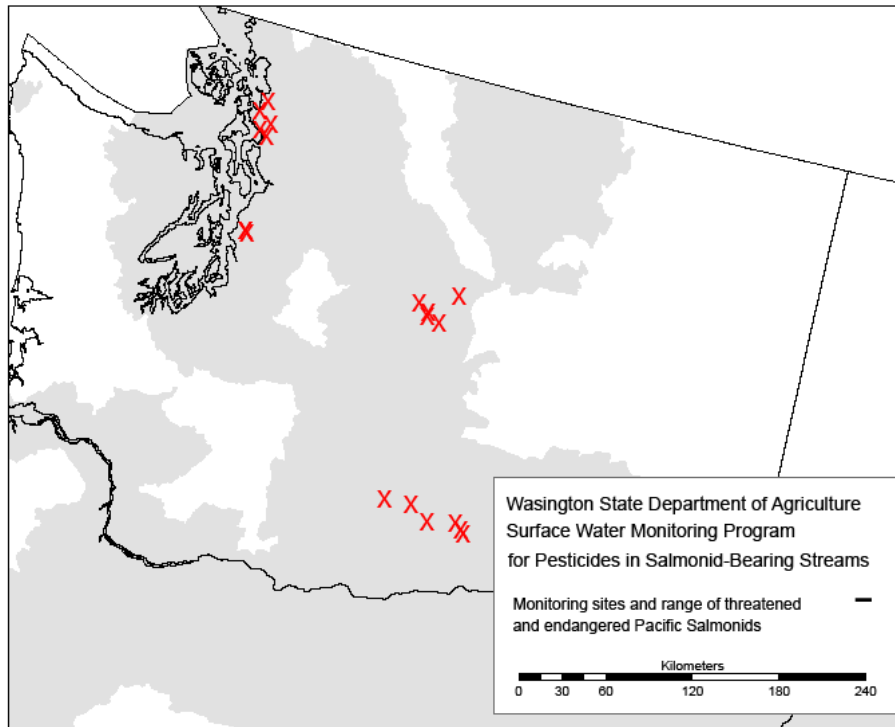


Figure 62. Distribution of Washington Department of Ecology sample stations compared to the distribution of listed salmon ESUs/DPSs.

Table 99. Washington Department of Ecology sample sites within the distribution of listed Pacific salmonids.

Species	ESU	Kilometers of Stream Inhabited	Sites in Spawning and Rearing Habitat	Sites in Migratory Corridor
Chinook	California Coastal	2,422.44	NA	NA
	Central Valley Spring - Run	2,212.94	NA	NA
	Lower Columbia River	2,443.29	0	0
	Upper Columbia River Spring - Run	1,646.75	5	0
	Puget Sound	3,639.65	8	0
	Sacramento River Winter - Run	546.84	NA	NA
	Snake River Fall - Run	1,370.44	NA	0
	Snake River Spring/Summer - Run	5,288.23	0	0
	Upper Willamette River	3,013.85	NA	0
Chum	Columbia River	1,162.18	0	0
	Hood Canal Summer - Run	141.89	0	0
Coho	Central California Coast	1,287.78	NA	NA
	Lower Columbia River	3,307.78	0	0
	Southern Oregon and Northern California Coast	5,619.58	NA	NA
	Oregon Coast	10,220.00	NA	NA
Sockeye	Ozette Lake	70.98	0	0
	Snake River	1,493.94	NA	0
Steelhead	Central California Coast	4,620.72	NA	NA
	California Central Valley	4,273.66	NA	NA
	Lower Columbia River	4,302.03	0	0
	Middle Columbia River	10,196.80	6	0
	Northern California	5,324.31	NA	NA
	Puget Sound	3,849.64	8	0
	Snake River	13,423.40	NA	0
	South-Central California Coast	5,104.56	NA	NA
	Southern California	3,015.86	NA	NA
	Upper Columbia River	2,143.15	5	0
	Upper Willamette River	3,063.07	NA	0

The subset of data provided by the Washington State Departments of Ecology and Agriculture was largely reflective of the results of the larger EIM dataset as it accounted

for more than 84-96% of the data for each of the 6 analytes (Table 100). 2,4-D acid (44%), diuron (22%), and triclopyr acid (12%) were the most frequently detected pesticides. Maximum concentrations for these three compounds were 6.57 µg/L, 4.1 µg/L, and 1.3 µg/L, respectively. Chlorothalonil and linuron were detected <1% of the time at maximum concentrations of 0.03 and 0.5, respectively. Captan was not detected.

Table 100. Concentrations of pesticides detected in recent studies by Washington Department of Ecology (2003-2009)¹.

Statistic	2,4-D acid	Triclopyr acid	Diuron	Linuron	Captan	Chlorothalonil
Samples	1708	1708	2011	1229	1526	1738
Percent Detections	43.74	12.30	21.73	0.16	0	0.35
LRL range (µg/L)	0.076-0.17	0.079-0.15	0.19 - 1	0.0054-0.5	0.015-0.44	0.0056 – 0.41
Minimum concentration (µg/L)	0.0047	0.0028	0.0022	0.03	0	0.0071
Maximum concentration (µg/L)	6.57	1.3	4.1	0.5	0	0.03
Median concentration (µg/L)	0.058	0.0325	0.04	0.042	0	0.0144

¹ Data in this table are a subset of data used to create Table 97.

Targeted Monitoring Studies

In some cases, EPA documents including the salmonid BEs, California Red-legged Frog (RLF) BEs, and EPA RED documents reported targeted monitoring studies where water concentrations or the percentage of runoff is associated with particular application rates and/or methods. We describe those in this section, along with other information available on targeted monitoring within the open literature.

2,4-D

The salmonid BE, RLF BE, and EPA RED documents do not identify any targeted surface water monitoring data for 2,4-D. However, a recent study reports 2,4-D and chlorothalonil concentration in water discharged from a golf course (King and Balogh 2010). Inflow and outflow concentrations of the two chemicals were measured in April through November from 2003 to 2008. Information on chlorothalonil is discussed in the corresponding section below. Total annual 2,4-D used on the golf course ranged from

0.4 – 1.1 lbs/A versus a labeled maximum rate of 4 lbs/A. The median 2,4-D outflow concentration of 2,4-D (0.85 µg/L) was significantly greater than the inflow concentration (0.31 µg/L). The mean annual loading of 2,4-D was 4.9 g per hectare or 0.5% of applied. The maximum 2,4-D concentration observed was 67.1 µg/L.

Triclopyr BEE

The salmonid BE (EPA 2004) reports a single targeted study that resulted in a maximum surface water concentration of 350 µg/L. Triclopyr BEE is not permitted for direct application to aquatic habitats. However, Thompson et al. (1991) investigated the environmental fate of triclopyr BEE in a stream to characterize potential aquatic exposure in the event of an accidental overspray or drift from an aerial forest application. A nominal application rate of 3.4 lbs a.i./A was assessed versus a maximum 6 lbs a.i./A single application rate, which is permitted on labels. The stream depth ranged from 0.5 – 2 m with a velocity of 16 cm/s. Concentrations of triclopyr BEE, triclopyr acid, and pyridinol metabolite (PYR) were evaluated. Peak concentrations of tryclopyr BEE measured were 230 and 350 µg/L in the treatment area and downstream sample sites, respectively. The pH level in the stream (7 – 7.5) was conducive to hydrolysis of triclopyr BEE and dissipation of triclopyr BEE was relatively rapid. The time-weighted average concentrations of triclopyr BEE during the first 12-14 hrs post application were 50 µg/L in samples from the treated area and 110 µg/L in samples from a downstream sample site. Average concentrations of triclopyr BEE declined to less than 1 µg/L 72 h post-application. The maximum concentration of triclopyr acid detected was 140 µg/L. PYR residues did not exceed the level of quantification, which was relatively high (50 µg/L). A spike in total triclopyr residues of 90 µg/L was measured in the stream following a rain event that occurred 2 days post treatment. Sampling following additional storms at 4 and 31 days post treatment did not reveal quantifiable concentrations of total triclopyr (> 50 µg/L). Although triclopyr cannot be directly applied to surface water, the information in this report is pertinent as accidental oversprays of small water bodies are expected to occur with aerial applications in forested environments. Additionally, initial concentrations resulting from the application

rate studied (3.4 lbs per acre) are comparable to concentrations that are expected in shallow water bodies from aerial applications that lack buffers (Thompson, Stanik et al. 1991). No other targeted monitoring studies were reported in the EPA documents or found in the open literature.

Diuron

The diuron BEs report four studies that investigated concentrations in surface water runoff, although the maximum labeled application rate of 12 lbs a.i./A was not evaluated (Table 101). All studies reported relatively high concentrations of diuron present in runoff water. The highest concentrations detected were those associated with runoff from rights-of-way treatments at approximately 3 lbs diuron/A (3.59 Kg/Ha). Powell et al. (1996) treated 2.4 meter wide strips and measured diuron concentrations in surface water runoff. Detected diuron concentrations in runoff ranged from 144 – 1770 µg/L following a simulated rainfall and 46 – 2849 µg/L following natural rain. The largest amount removed during any sample period was 8.4% of the applied material in a 28-hr period (Powell, Neal et al. 1996). Assuming this material is deposited in an adjacent waterway that runs the length of the treatment area and is 2 meters wide and 0.5 meters deep would result in a diuron concentration of approximately 72 µg/L.

Table 101. Concentrations of diuron detected in surface water runoff.

Use Site	Application Rate lbs diuron/A	Concentrations Detected µg/L	Source
Vineyards	1.8	<0.3 - 467	(Andrieux, Lennartz et al. 1997)
Citrus	Variable:1-4	3.1 - 891	(Braun and Hawkins 1991)
Citrus	1.6	600 – 1700	(Spurlock, Garretson et al. 1997)
Rights-of-way	3.2	46 – 2849	(Powell, Neal et al. 1996)

Chlorothalonil

The BEs indicate that surface water monitoring for chlorothalonil that is coordinated with specific use patterns of the pesticide were not available. However, a recent study discussed above, reports 2,4-D and chlorothalonil concentration in water discharged from

a golf course (King and Balogh 2010). Total annual chlorothalonil used on the golf course ranged from 2003-2007 ranged from 2.5 – 3.3 lbs/A versus a labeled maximum rate single application rate of 11 lbs/A and annual application rates of up to 26 lbs/A on fairways and 73 lbs/A on greens. The median outflow concentration of chlorothalonil (0.58 µg/L) was significantly greater than the inflow concentration, which was below the detection limit (0.07 µg/L). The mean annual loading of chlorothalonil was 10.5 g per hectare or 0.3% of applied. The maximum chlorothalonil concentration observed was 48.1 µg/L. During the months of April – October 90th percentile concentrations for chlorothalonil ranged from 2 – 4 µg/L, but reached approximately 11 µg/L during November.

Linuron and Captan

No targeted monitoring studies were described in either the BE, RED chapter or the RLF effect determinations. Additionally, we found no targeted studies in the open literature that evaluated surface water concentrations of these two pesticides related to field level use of pesticides.

Summary of Monitoring Data

NMFS did not locate many edge-of-field studies for the compounds addressed in this Opinion. However, when targeted monitoring studies were available maximum concentrations detected exceeded maximum concentrations in the monitoring databases reviewed (NWQA, CDPR, and EIM; Table 102). The open literature evaluated and the general state of knowledge regarding field runoff from pesticide applications leads us to anticipate the following:

- edge-of-field runoff concentrations will be higher than concentrations measured in waterbodies with substantial diluting volume,
- low-flow or runoff-dominated systems likely contain the highest concentrations (approaching or exceeding modeled concentrations), and
- measured concentrations in general monitoring programs are likely to be lower than peak runoff concentrations, as sampling may not coincide with initial application and/or runoff events.

Table 102. Monitoring Data Concentrations.

Chemical Parent compounds	From Databases ¹				From Targeted Studies	
	Min Conc. µg/L	Monitoring Database	Max Conc. µg/L	Monitoring Database	Max Conc. µg/L	Study Reference
2,4-D	0.11	EIM	7.66	NWQA	67.1	(King and Balogh 2010)
Triclopyr BEE	NE	NA	NE	NA	350	(Thompson, Stanik et al. 1991) ²
Diuron	0.002	NWQA	160	CDPR	2849	(Powell, Neal et al. 1996) ³
Linuron	0.003	EIM	1.6	CDPR	NE	NA
Captan	0.6	EIM	0.6	EIM	NE	NA
Chlorothalonil	0.0027	EIM	0.36	EIM	48.1	(King and Balogh 2010)
<i>Degradates</i>						
2,4-DCP	ND	NWQA	ND	NA	NE	NA
Triclopyr acid	0.0028	EIM	14.5	CDPR	140	(Thompson, Stanik et al. 1991) ²
3,4-DCA	0.01	NWQA	0.7	NWQA	NE	NA
CPMU	0.0037	NWQA	0.1703	NWQA	NE	NA

¹Minimum and maximum based on detected values²Monitoring from direct overspray of forest stream³Concentration in surface runoff water

ND Not detected

NE Not evaluated

NA Not applicable

Exposure to Other Action Stressors

Stressors of the action also include the metabolites and degradates of the a.i.s, other active and inert ingredients included in their product formulations, and tank mixtures and adjuvants authorized on their product labels. Below we summarize information presented in the BEs and provide additional information to characterize exposure to these stressors.

Metabolites and degradates of the six a.i.s

EPA documents identified major degradates and degradates of toxicological concern of the six a.i.s (see *Summary of the Chemical Fate of the Six Active Ingredients*).

However, estimates quantifying potential exposure of listed salmonids and their habitat to these transformation products were not provided and remain a considerable source of

uncertainty. In general, failure to consider exposure to these breakdown products increases the likelihood that risk is underestimated.

Other ingredients in formulated products

Registered pesticide products containing the six a.i.s include other ingredients such as carriers, surfactants, and synergists. NMFS reviewed many of the active labels of the six a.i.s and found pesticide products contain multiple a.i.s were common. Examples of some of the formulations that contain multiple a.i.s are presented in Table 103. Several other products are also formulated with petroleum distillates and other solvents, but did not indicate the concentration in the formulation as needed to estimate potential exposure in aquatic environments. Other ingredients in the formulation were frequently not specified.

Table 103. Examples of pesticide product ingredients.

EPA Product Registration Number	Active Ingredients %	Other Ingredients %
62719-260, 228-552	2,4-D 34.4, triclopyr BEE 16.5	49.1
228-317	MCPA 56.14, triclopyr BEE 5.00, dicamba 3.60	35.26
2217-920	triclopyr BEE 8.4, sulfentrazone 0.73, 2,4-D 31.82, dicamba 2.43	56.62
62719-477	triclopyr BEE 25, fluroxypyr 8.6	66.4
352-505	diuron 40, bromacil 40	20
352-618	diuron 46.8, hexazinone 13.2	40
228-386	diuron 4, bromacil 2	94
228-654	diuron 62.22, imazapyr 7.78	30
228-678	diuron 6, thidiazuron 12	82
352-660	linuron 20.3, diuron 20	59.7
264-949	captan 46, PCNB 15, carboxin 10	29
264-998	captan 50, trifloxystrobin 2, thiophanate-methyl 13.6, metalaxyl 0.8	33.6
400-561	captan 25, carboxin 12.5, metalaxyl 3.75	58.75
4-122	captan 12, malathion 0.246, carbaryl 0.30	81.7
400-568	captan 20, carboxin 20, imidacloprid 4	56
66330-48	captan 4.8, fenhexamid 14.3	30.9
100-1347	propiconazole 2.9, chlorothalonil 38.5	58.6
100-1315	chlorothalonil 45.0, azoxystrobin 3.0	52
100-1231	chlorothalonil 29.9, propiconazole 4.7, fludioxonil 1.2	64.2
100-800	chlorothalonil 72, mefenoxam 4.5	23.5

Nonylphenol (NP) and nonylphenol polyethoxylates are “other ingredients” that may be part of a pesticide product formulation and are common adjuvant ingredients added during pesticide applications. NP and nonylphenol polyethoxylates are also ingredients in detergents, cosmetics, and other industrial products and are a common wastewater contaminant from industrial and municipal sources. NP has been linked to endocrine disrupting effects in aquatic systems (Koplin, Furlong et al. 2002). A national survey of streams found that NP was among the most ubiquitous organic wastewater contaminants in the U.S., detected in more than 50% of the samples tested. The median concentration of NP in streams surveyed was 0.8 µg/L and the maximum concentration detected was 40.0 µg/L (Table 104). Related compounds were also detected at a relatively high frequency (Kolpin, Furlong et al. 2002).

Table 104. Detection and concentrations of nonionic detergent degradates in streams of the U.S. (Koplin et al 2002).

Chemical	Frequency Detected	Maximum (µg/L)	Median (µg/L)
4-nonylphenol	50.6	40	0.8
4-nonylphenol monoethoxylate	45.9	20	1
4-nonylphenol diethoxylate	36.5	9	1
4-octylphenol monoethoxylate	43.5	2	0.2
4-octylphenol diethoxylate	23.5	1	0.1

We are uncertain to what degree NP and NP-ethoxylates may or may not occur in pesticide products that contain the six a.i.s and/or are added prior to application. Inert ingredients are often not specified on product labels. Additionally, NP and NP-ethoxylates represent a very small portion of the more than 4,000 inert ingredients that EPA permits for use in pesticide formulations (Koplin et al 2002). Many of these inerts are known to be hazardous in their own right (*e.g.*, xylene is a neurotoxin and coal tar is a known carcinogen). Several permitted inerts are also registered a.i.s (*e.g.*, copper, zinc, chloropictrine, chlorothalonil). Inerts can be more than 50% of the mass of pesticide products, and millions of pounds of these products are applied to the landscape each year (Koplin et al 2002). This equates to large contaminant loads of inerts that may adversely

affect salmon or their habitat. Uncertainty regarding exposure to these ingredients will be qualitatively incorporated into our analysis.

Tank Mixtures

Several pesticide labels authorize the co-application of other pesticide products and other materials in tank mixes, thereby increasing the likelihood of exposure to multiple chemical stressors (Table 105). In some cases specific tank mixtures with other pesticide products or adjuvants are recommended. In all cases tank mixtures are authorized unless specifically prohibited on the product label. These ingredients and the other inert ingredients in these products are considered part of the action because they are authorized by EPA's approval of the FIFRA label. Exposure to and risk associated with potential ingredients in tank mixtures were not addressed in EPA's BEs and remain a significant source of uncertainty.

Table 105. Examples of label recommended tank mixtures.

Pesticide Products containing	Tank mixture recommendation
2,4-D	Mix with other herbicide products, including Accord® (glyphosate) to increase weed control. Mix with surfactants (e.g. LI700) and drift redardants (e.g. Compadre ®) to increase efficacy and reduce drift.
Triclopyr BEE	Mix with other herbicides, including 2,4-D products, and liquid fertilizers. Mix product with agricultural surfactants and drift control agents.
Diuron	Mix with other suitable herbicides (e.g. OUST XP®, 2,4-D, hexazinone, Sinbar®, glyphosate, trifluralin, ammonium sulfate, DSMA, MSMA, bromoxanil, Arsenal®, and others) and adjuvants (e.g. non-ionic surfactants and crop oil concentrates).
Linuron	Mix with other herbicides (e.g. metribuzin DF®, alachlor, metolachlor, pendamethalin, paraquat, glyphosate), surfactants, and fertilizers.
Captan	Combine with other fungicides or insecticides at recommended rates.
Chlorothalonil	Compatable with many commonly used insecticides, fungicides, and spray adjuvants.

Environmental Mixtures

As described in the *Approach to the Assessment*, we analyze the status of listed species, in conjunction with the *Environmental Baseline* in evaluating the likelihood that action

stressors will reduce the viability of populations of listed salmonids. This involves considering interactions between the stressors of the action and the *Environmental Baseline*. For example, we consider that listed salmonids may be exposed to the wide array of chemical stressors that occur in the various marine, estuarine, and freshwater habitats they occupy throughout their life cycle. Exposure to multiple pesticide ingredients most likely occurs in freshwater habitats and nearshore environments adjacent to areas where pesticides are used. As of 1997, about 900 a.i.s were registered in the U.S. for use in more than 20,000 different pesticide products (Aspelin and Grube 1999). Typically 10 to 20 new a.i.s are registered each year (Aspelin and Grube 1999). In a typical year in the U.S., pesticides are applied at a rate of approximately five billion pounds of a.i. per year ((Kiely, Donaldson et al. 2004)). Pesticide contamination in the nation's freshwater habitats is ubiquitous and pesticides usually occur in the environment as mixtures (Gilliom, Barbash et al. 2006) "More than 90% of the time, water from streams with agricultural, urban, or mixed-land-use watersheds had detections of two or more pesticides or degradates, and about 20% of the time they had detections of 10 or more," (Gilliom et al 2006). The likelihood of exposure to multiple pesticides throughout a listed salmonids' lifetime is great, considering their migration routes and habitats occupied for spawning and rearing. In a three-year monitoring study conducted by the Washington DOE, pesticide mixtures were found to be common in both urban and agricultural watersheds (Burke, Anderson et al. 2006). An average of three pesticides was found in each sample collected on urban sampling sites, with as many as nine pesticides found in a single sample. Agricultural sites averaged three to five pesticides per sample, with as many as 14 pesticides being detected in a single sample (Burke, Anderson et al. 2006). Mixtures of chemicals that share a common mode or mechanism of action are of particular concern to NMFS.

Gilliom and others (2006) suggested that assessment of pesticide mixture toxicity to aquatic life is needed given the widespread and common occurrence of pesticide mixtures, particularly in streams, because the total combined toxicity of pesticides in water is often greater than that of any single pesticide compound. Exposure to multiple

pesticide ingredients can result in additive and synergistic responses as described in the *Risk Characterization* section. It is reasonable to conclude that compounds sharing a common mode of action cause additive effects and in some cases synergistic effects. Exposure to these compounds and other baseline stressors (*e.g.*, thermal stress) was not a consideration in the BEs, which only considered effects from single a.i.s. Therefore, risk to listed species may be underestimated in EPA's assessments.

Exposure Conclusions

Pacific salmon and steelhead use a wide range of freshwater, estuarine, and marine habitats and many migrate hundreds of miles to complete their life cycle. Many of the a.i.s and degradates addressed in this Opinion, especially 2,4-D, triclopyr, diuron and its degradates, are detected frequently in freshwater habitats within the four western states where listed Pacific salmonids are distributed. Because the action of registering the six a.i.s for the next 15 years authorizes a number of the same uses, they will continue to be present in the action area. Additionally, all of the a.i.s are used when listed species are present in freshwater habitats. Therefore, we expect some individuals within all the listed Pacific salmon and steelhead ESUs/DPSs will be exposed to these chemicals and other stressors of the action. Given variable use of these pesticides across the landscape, and variable temporal and spatial distributions of listed salmonids, we expect exposure is also highly variable among individuals and populations of listed salmon. However, defining exposure and distributions of exposure among differing life stages of each independent population is complicated by several factors. Paramount among these is the uncertainty associated with the use of pesticide products containing these a.i.s. More specifically:

- EPA-authorized labels contain language that frequently does not provide clear distinctions on product use (*e.g.*, many labels do not specify the maximum number of applications, application interval, or maximum annual application rate);
- Product labels authorize the application of chemical mixtures that are not specified or not clearly defined (*e.g.*, the ingredients of pesticide formulations are not fully disclosed, labels recommend tank mixture applications with other

pesticides and adjuvants, and tank mixtures with other pesticides are permitted unless specifically stated otherwise);

- Defining actual use of these products is highly uncertain. Historical use information is limited and may not reflect future use.

A major limitation of these assessments is that the majority of monitoring data used was not designed to determine exposure to listed salmonids. Studies conducted by Washington state departments of ecology and agriculture were an exception, but those studies were not designed to evaluate peak exposure or exposure distributions in listed salmon. Therefore, caution should be exercised in using these data for that purpose especially when conducting probabilistic assessments.

Additionally, the assessments lack uncertainty analyses of the monitoring and toxicity data used, which limit the confidence in the given estimates (Warren-Hicks and Moore 1998). Given the complexity and scale of this action, we are unable to accurately define exposure distributions for the chemical stressors. We assume the highest probability of exposure occurs in freshwater, and nearshore estuarine/marine environments in close proximity to areas where pesticide products containing 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil are applied. We considered several sources of information to define the range of potential exposure to action stressors. These sources are summarized in (Table 106). Ranges of concentrations for the monitoring data, EECs generated by EPA in the salmonid BEs and California Red-legged Frog (RLF) BEs, and NMFS generated spray drift estimates for characterizing initial concentrations in floodplain habitats are given (Table 106). Typically, the estimates for the floodplain habitat are higher than or near the high end of the range of EECs generated by EPA's PRZM-EXAMS modeling. Estimates for the pertinent degradates were generally not evaluated. The highest concentrations detected in surface waters were those associated with applications directly to aquatic habitats and from targeted monitoring studies that sampled surface water immediately adjacent to pesticide application sites.

Table 106. Chemical Concentration Ranges in Monitoring Data and Modeling.

Chemical	Monitoring Data		EPA Estimates		NMFS Estimates
Parent compounds	Range ¹ µg/L (Database)	Max Conc. µg/L (Targeted)	Salmonid BE Conc. Range µg/L	RLF BE Conc. Range µg/L	Spray Drift µg/L
2,4-D	0.11-7.66	67.1	3.7 -1431	0.08 – 4,000 ⁴	13 - 1912
Triclopyr BEE	NE	350 ²	20 – 5,872	21 - 1499	184 - 3824
Diuron	0.002-160	2849 ³	3 - 437	5 - 4911	221 - 1147
Linuron	0.003-1.6	NE	31.3	2.6 - 337	552
Captan	0.6-0.6	NE	1.1 – 43.4	0.001- 29	184
Chlorothalonil	0.0027- 0.36	48.1	6 - 363	3 - 274	754 - 2078
Degradates					
2,4-D					
2,4-DCP	ND	NE	NE	NE	NE
1,2,4- benzenetriol; 2,4-DCA; 4- chlorophenol; 2- chlorophenol; chlorophenoxy acetic acid; chlorohydroquin one	NE	NE	NE	NE	NE
triclopyr BEE					
Triclopyr acid	0.0028- 14.5	140 ²	NE	NE	NE
TCP; TMP; oxamic acid	NE	NE	NE	NE	NE
diuron					
3,4-DCA	0.01-0.7	NE	NE	NE	NE
CPMU	0.0037- 0.1703	NE			NE
DCPMU; MCPDMU DCMU; DCPU; TCAP;	NE	NE	NE	NE	NE
linuron					
3,4-DCA	0.01-0.7	NE	NE	NE	NE
DCPMU; DML; DCPU; 3-(3- chloro-4- hydroxyphenyl)- 1-methoxy-1- methylurea; 3,4- ichlorophenylure	NE	NE	NE	NE	NE

Chemical	Monitoring Data		EPA Estimates		NMFS Estimates
Parent compounds	Range ¹ µg/L (Database)	Max Conc. µg/L (Targeted)	Salmonid BE Conc. Range µg/L	RLF BE Conc. Range µg/L	Spray Drift µg/L
a; 3-(3,4-dichlorophenyl)-1-methylurea; norlinuron; desmethyl linuron; desmethoxy linuron					
captan					
TCMT; THPI; THCY; THPAm; Thiophosgene; inorganic sulfur;	NE	NE	NE	NE	NE
chlorothalonil					
SDS-19221; SDS-3701; SDS-46851; SDS-47523/ SDS-47524; SDS-47425; SDS-67042; SDS-66432; SDS-66382; SDS-13353	NE	NE	NE	NE	NE

NE – Not estimated

1- Minimum and maximum based on detected values in NWQA, CDPR, and EIM databases

2- Direct overspray of forest stream

3- Surface water runoff

4- Target concentration for control of aquatic weeds

Inherent in the modeling estimates is the assumption that the pesticide is applied in a location next to or draining into salmon-bearing waters. Monitoring data may reflect pesticide applications proximate to the waterbody (i.e. values derived from targeted monitoring), or resulting from more distant uses in the watershed or airshed. We assume that the exposure estimates provided by EPA in the BEs and additional modeling and monitoring information provided above represent realistic exposure levels for some

individuals of the listed species. Further, we assume the distribution within the range of exposures is a function of pesticide use and the duration of time listed salmonids spend in these habitats. All listed Pacific salmon and steelhead occupy habitats that could contain high concentrations of these pesticides at one or more life stages. However, the time spent in these habitats varies among species. Adult salmon and steelhead spend weeks to several months in freshwater habitats during their migration and spawning activities. Immediately after emerging from the gravel substrate and transitioning from alevins to fry, salmonids move to habitats where they can swim freely and forage. At this point in their development most salmon occupy freshwater habitats. Chum salmon are an exception. They immediately migrate downstream following emergence to nearshore environments in estuaries near the mouth of their natal stream. Upon arrival in the estuary the chum salmon fry inhabit nearshore areas at a preferred depth of 1.5-5 m. In Puget Sound, WA, surveys indicate chum salmon fry are distributed extremely close to the shoreline and concentrated in the top 15 cm of water. Therefore, chum salmon fry are less likely to be exposed to high concentrations of pesticides than other salmonids given they quickly migrate to larger estuaries with greater dilution potential. They may reside immediately next to the shore in estuaries for as little as one or two weeks before moving offshore or into deeper-water habitats within the nearshore environment. Sockeye salmon fry most frequently rear in lakes, where they distribute in the littoral zones. They initially occupy shoreline habitats of only a few centimeters in depth before moving further off-shore and taking on a more pelagic existence. Coho salmon, Chinook salmon, and steelhead fry typically select the stream's nearshore zone and floodplain habitats associated with their natal rivers and streams. These species are most likely to experience higher pesticide exposures given their use of shallow freshwater habitats for juvenile rearing. Coho salmon and steelhead have a greater preference for the shallow habitats and rear in freshwater for more than a year. Coho salmon fry rear in lower gradient river channels and often rear in pools of the river channels. They may also rear in ponds and lakes. Steelhead juveniles use riffles and faster flowing waters more than coho salmon, and are often found in steeper gradient channels. Coho salmon juveniles may make extensive migrations in fall to overwinter in floodplain habitats such as ponds,

sloughs, oxbows, flooded wetlands, and other seasonally connected and inundated habitat. Spring foraging in these habitats often provides substantial growth before smoltification and juveniles in these habitats can grow significantly larger than mainstem overwintering coho salmon juveniles. Steelhead do not use channels with organic bottom substrate for overwintering and often seek refuge under larger stones in the flowing river as protection from strong winter flows. Chinook salmon commonly spawn and rear in larger rivers and tributaries than the other *Oncorhynchus* species. Juvenile Chinook salmon in California, with the exception of the Central Valley spring-run Chinook salmon, spend less than six months in freshwater and out-migrate as fry or sub-yearlings (ocean type). Juveniles in the Columbia River basin and in Puget Sound may out-migrate as fry/sub-yearlings (ocean type) or as yearlings (river type), depending on race and the river basin of origin. Fry of the ocean type life history typically rear in estuarine shallow waters, tidal wetlands, and sloughs for days to weeks before entering the ocean while yearling or older juveniles spend less time and use deeper water in the estuary.

Substantial data gaps in EPA's exposure characterization include exposure estimates associated with product uses on many crops and non-crop uses. Additionally, exposure estimates for other chemical stressors including other ingredients in pesticide formulations, other pesticide products authorized for co-application, adjuvants, degradates, and metabolites were not provided in BEs. Although NMFS is unable to comprehensively quantify exposure to these chemical stressors, we are aware that exposure to these stressors is likely. We assume these chemical stressors may pose additional risk to listed Pacific salmonids. However, in order to ensure that EPA's action is not likely to jeopardize listed species or destroy or adversely modify critical habitat, NMFS analyzes potential exposure based on all stressors that could result from all uses authorized by EPA's action.

Response Analysis

In this section, we identify and evaluate toxicity information for the stressors of the action and organize the information under assessment endpoints relating to both individual and habitat responses (Figure 63). The assessment endpoints are biological attributes that, when adversely affected, may reduce fitness of individual salmonids or degrade PCEs (*e.g.*, prey abundance, water quality, and suitability of habitat).

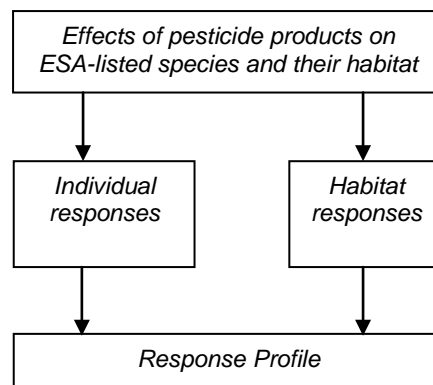


Figure 63. Response analysis.

We begin the response analysis by describing the toxic mode and mechanism of action of the a.i.s, then summarize information associated with relevant assessment endpoints. Toxicity information is derived from EPA documents (salmonid BEs, REDS, IREDs, California red legged frog BEs, and EFED Science Chapters) for the six a.i.s, and also from open literature studies. We have not reproduced data tables from the EPA documents, although we have included ecological effects information summarized by EPA in the appendices of the California red-legged frog as our *Appendix 7*. In some cases (noted in relevant text) we went back to the original source documents referenced by EPA to confirm values. The information provided by EPA addressed aspects of survival, growth, and reproduction of aquatic species (freshwater and saltwater), and provided some discussion on other information found in the open literature, such as results from some field experiments and experiments that evaluated sublethal effects.

Under Section 7 of the ESA and its implementing regulations, NMFS evaluates all direct and indirect effects of an action. We therefore evaluate all aspects of an action that may reduce fitness of individuals or reduce PCEs of designated critical habitat. The evaluation includes information that EPA provided on survival, growth, or reproduction, but also encompasses a broader range of endpoints including behaviors, endocrine disruption, and other physiological alterations such as impairment of olfactory-mediated behaviors.

The information we evaluated is derived from published, peer-reviewed scientific journals, government agency reports (federal, state and local), theses and dissertations, books, information and data provided by the registrants identified as applicants, and independent reports. NMFS scientists evaluate the quality and applicability of all documents used, although unlike EPA, we do not develop formal data evaluation reports (DERs) for the sources we review. Typically, the most relevant study results are those which directly measure effects to on identified assessment endpoint and are derived from experiments with salmonids. Studies with listed Pacific salmonids or hatchery surrogates are preferable, but we also include data from closely-related species in our “salmonid” endpoint summaries. We present data from other fish species as well. Often, there is not a complete suite of information relating to effects on fish, especially for some of the sublethal endpoints. Where appropriate, we include information from studies on other taxa, recognizing and noting where there may be significant interspecies extrapolation. Likewise, we consider information from studies on chemicals that are structurally similar to the a.i.s addressed in this Opinion.

EPA’s ecological risk assessments and BEs primarily summarize acute and chronic toxicity data from “standardized toxicity tests” submitted by pesticide registrants during the registration process, or tests from government laboratories available in EPA databases, or from published, peer-reviewed scientific publications (books and journals). The assessment endpoints from these tests for an individual organism generally include

only survival (death), reproduction, and growth measured in laboratory dose-response experiments conducted on a single a.i.

Survival is typically measured in both acute (48-96 h) and chronic (21-60 d) tests. Fish reproduction and growth are generally measured using chronic tests (21-60 d).

Population-level endpoints and analyses were generally absent in the BEs, other than a few measurements of fish and aquatic invertebrate reproduction. Adverse effects to organisms were not translated into consequences to populations. For this Opinion, NMFS translates effects to individual salmonids into potential population-level consequences as explained in the *Risk Characterization* portion of the *Effects of the Proposed Action* section, and ultimately draws a conclusion on the likely risk to listed salmonids based on exposure and anticipated individual and population-level effects.

Survival Endpoints

Survival of individual fish is typically measured by incidences of death following 96 h exposures to the a.i. (acute test). Survival data may also include incidences of death following longer exposures (21 or more days, known as chronic tests) which are intended to evaluate effects on growth and reproductive endpoints. Tests are conducted on a subset of freshwater and marine fish species reared in laboratories under controlled conditions (temperature, pH, light, salinity, dissolved oxygen, *etc.*,) (EPA 2004).

Lethality of the pesticide (a technical product or formulated product) is usually reported as the median lethal concentration (LC₅₀), the statistically-derived concentration sufficient to kill 50% of the test population. For aquatic invertebrates it may be reported as a median effective concentration (EC₅₀), because death of these organisms may be too difficult to confirm and immobilization is considered a terminal endpoint. An LC₅₀ is derived from the number of surviving individuals at each concentration tested following a 96 h exposure and is typically estimated by probit or logit analysis and recently by statistical curve fitting techniques. In FIFRA guideline tests, LC₅₀s are typically calculated by probit analysis. If the data are not normally distributed for a probit analysis, then either a moving average or binomial is used, resulting in no slope being

reported. Ideally, to maximize the utility of a given LC₅₀ study, a slope, variability around the LC₅₀, and a description of the experimental design, such as experimental concentrations tested, number of treatments and replicates used, solvent controls, *etc.*, are reported. The slope of the observed dose-response relationship is particularly useful in estimating the magnitude of death at concentrations below or above an estimated LC₅₀. The variability of an LC₅₀ is usually depicted by a confidence interval (95% CI) or standard deviation/error and is illustrative of the degree of confidence associated with a given LC₅₀ estimate *i.e.*, the smaller the range of uncertainty, the higher the confidence in the estimate. Without an estimate of variability, it is difficult to infer the precision of the estimate. Furthermore, survival experiments are of most utility when conducted with the most sensitive life stage of the listed species or a representative surrogate. In the case of ESA-listed Pacific salmonids, there are several surrogates that are available for toxicity testing including hatchery reared coho salmon, Chinook salmon, steelhead, and chum salmon, as well as rainbow trout³. The available toxicity data include a variety of salmonids. Unfortunately, slopes, estimates of variability for an LC₅₀, and experimental concentrations frequently are not reported. In our review of the salmonid BEs, we did not locate any reported slopes of dose-response curves, although some of this information was presented in some of the corresponding Science Chapters and the CRLF BEs. Consequently, we must err on the side of the species in the face of these uncertainties and select LC_{50s} from the lower range of available salmonid studies. We evaluate concentrations that are expected to kill fish and apply qualitative and quantitative methods to infer population-level responses of ESA-listed salmonids within the *Risk Characterization* section.

³ Rainbow trout and steelhead are the same genus species (*Oncorhynchus mykiss*), with the key differentiation that steelhead migrate to the ocean while rainbow trout remain in freshwaters. Rainbow trout are therefore good toxicological surrogates for freshwater life stages of steelhead, but are less useful as surrogates for life stages that use estuarine and ocean environments.

Growth Endpoints

Growth of individual organisms is an assessment endpoint derived from standard chronic fish and invertebrate toxicity tests summarized in the BEs. It is difficult to translate the significance of reduced growth derived from a guideline study on fish growth in aquatic ecosystems. The health of the fish, availability and abundance of prey items, and the ability of the fish to adequately feed are not assessed in standard chronic fish tests. These are important factors affecting the survival of wild fish. What is generally assessed is size or weight of fish measured at several times during an experiment. The test fish are usually fed twice daily, *ad libitum*, (i.e., an over abundance of food is available to the fish). Therefore, any reductions in size are a result of fish being affected to such an extent that they are not feeding or are unable to metabolize food even when presented with an abundance of food. Subtle changes in feeding behaviors or availability of food would not be detected from these types of experiments. If growth is affected in these experiments, it is highly probable that growth of fish in natural aquatic systems would be severely affected. If effects to growth are likely, we assess salmonid population-level consequences based on reductions in juvenile growth and subsequent reduction in size prior to ocean entry.

Reproduction Endpoints

Reproduction, at the scale of an individual, can be measured by the number of offspring per female (fecundity), and at the scale of a population by measuring the number of offspring per females in a population over multiple generations. The BEs summarized reproductive endpoints at the individual scale from chronic freshwater fish experiments where hatchability and larval-juvenile survival is measured. NMFS also considers many other assessment measures of reproduction, including egg size, spawning success, sperm and egg viability, gonadal development, reproductive behaviors, and hormone levels, as these endpoints could have considerable effect on wild populations. These endpoints are not generally measured in standardized toxicity assays used in pesticide registration.

Sublethal Endpoints

Sometimes qualitative observations of sublethal effects are summarized from 96 h lethality dose-response bioassays in EPA's risk assessments. These observations generally were limited in the BEs, and when noted, pertained to unusual swimming behaviors. None of these behaviors were rigorously measured and therefore are of limited value in assessing the effects of these pesticides on Pacific salmonids. We do, however note a few of the observations when they pertained to a relevant assessment endpoint, such as impaired swimming. Some BEs presented toxicity information on degradates, metabolites, and formulations. However, toxicity information on other or "inert" ingredients found in pesticide formulations was usually not presented.

Multi-species (Micro- and Mesocosm) Studies

Results from multiple species tests, called microcosm and mesocosm studies, were also discussed in the BEs to a varying degree. These types of experiments are likely closer approximations of potential ecosystem-level responses such as interactions among species (predator-prey dynamics), recovery of species, and indirect effects of pesticides on fish. However, the interpretation of results is complicated by how well the results represent natural aquatic ecosystems and how well the studies apply to salmonid-specific assessment endpoints and risk hypotheses. These studies typically measured individual responses of aquatic organisms to contaminants in the presence of other species. Some are applicable to questions of trophic effects and invertebrate recovery, as well as providing pesticide fate information. The most useful mesocosm study results for this Opinion are those that directly pertain to identified assessment endpoints and risk hypotheses. We discuss study results in the context of salmonid prey responses, emphasizing the capacity of prey taxa to rebound following death of individuals as well as shifts community structure. For herbicides, we also consider modifications in the plant communities in and around the waterbody. One of the notable limitations of most micro- and mesocosm studies is they do not typically represent real world aquatic ecosystems which are degraded from various stressors.

Results from aquatic field studies were generally not discussed in great detail within the BEs. We discuss field studies that evaluated assessment endpoints, particularly those which address salmonid prey responses in systems with ESA-listed salmonids.

Potential effects of herbicides on juvenile salmonids and their critical habitats

Importance of plants and other photosynthetic organisms in fueling secondary production within salmonid habitats

Secondary production within aquatic systems - including production of juvenile salmonids - is ultimately fueled by plants and other photosynthetic organisms (e.g., green algae, diatoms, cyanobacteria). In salmonid freshwater and estuarine habitats, this energy comes from two sources: 1) primary production within aquatic habitats (autochthonous inputs) as well as 2) inputs of organic matter from adjacent terrestrial ecosystems (allochthonous inputs) (Allan and Castillo 2008). Plants and other photosynthetic organisms are primary producers, and they can be consumed (by “consumers”) as living tissue that is grazed from the benthos (periphyton), as living tissue collected from the water column (phytoplankton), or as dead tissue that is consumed after being colonized by microbial communities (detritus). Invertebrates and fish are specific with regard to their ability to feed on these various food resources, and these distinctions help define functional feeding groups that include grazers, shredders, and predators, among others. Therefore, although there is great diversity in the pathways this energy takes in an aquatic system, much of the energy that fuels production in aquatic habitats derives ultimately from plants and other photosynthetic organisms.

Fish can consume a very high proportion of the invertebrate secondary production in aquatic habitats ((Huryn 1996);(Huryn 1998)). Juvenile salmonids, often at high densities and growing quickly, are predators and consume a wide range of invertebrates, including those from all functional feeding groups. Changes in the production of any of these

groups could change prey availability for these fish; for example, a reduction in periphyton production on rocks in a stream could reduce invertebrate grazer production. Likewise, a change in the quantity or quality of terrestrial leaf litter falling in from a riparian buffer could alter the production of invertebrate shredders downstream. In addition to being the ultimate source of food for much of the invertebrate community, plants also provide habitat for invertebrates and fish, including but not limited to substrate for them to shelter on and under (e.g. macrophytes, root wads). Plants and other photosynthetic organisms within and adjacent to salmonid freshwater and estuarine habitats are therefore essential components of productive salmonid habitats. Actions that affect the diversity, biomass and/or the production of primary producers in and around salmonid habitats may limit or alter secondary production within those systems (Figure 64).

As food resources, living plants and other living photosynthetic organisms are especially nutritious for grazing invertebrates and herbivorous fish (Torres-Ruiz, Wehr et al. 2007), and they often contribute more to overall secondary production within a system than would be expected simply by their standing stock at any one point in time (Allan and Castillo 2008). Because of this high nutritional value, the autochthonous production of plants and other photosynthetic organisms can be limited by grazers, though abiotic factors such as light, nutrients and water velocity are also often limiting (e.g., (Blanchet, Loot et al. 2008); (Rosemond, Mulholland et al. 2000); (Sanderson, Barnas et al. 2009)). The relative importance of these biotic and abiotic factors in limiting primary production varies by system and can change seasonally (e.g., (Huryn 1998); (Sanderson, Barnas et al. 2009)).

When primary production is limited or low, consumer production can be limited. This has been demonstrated primarily by amending a limiting resource to the point at which it is no longer limiting. For example, when nutrients are added to nutrient-limited systems, primary production and consequently secondary production can increase (e.g., (Harvey, Peterson et al. 1998); (Mundie, Simpson et al. 1991)). Fewer studies have examined

explicitly how *reductions* in primary producers (or primary production) affect fish and invertebrates, as would potentially occur when sensitive photosynthetic organisms at the base of an aquatic food web are exposed to herbicides. In some cases in which algal biomass is reduced by disturbances, invertebrate grazer growth and abundance decline. Higher trophic levels can be affected by these bottom-up effects, as Perry et al. (Perry, Bradford et al. 2003) observed in juvenile Chinook salmon. In small tributaries of the Yukon River, a fire and flood reduced the proportion of high quality autochthonously derived energy that salmon consumed, suggesting there may be direct and indirect effects of disturbances on energy transfer among trophic levels including salmon (Perry, Bradford et al. 2003).

The loss or reduction of inputs of organic matter (including leaf litter, woody debris, and terrestrial insects) from adjacent terrestrial ecosystems can also significantly reduce invertebrate secondary production and potentially fish production ((Wallace, Eggert et al. 1999); (Allan, Wipfli et al. 2003)). This was demonstrated by Wallace et al. (Wallace, Eggert et al. 1999) when they excluded terrestrial leaf litter from a forest stream in the southeast for four years and found that invertebrate production in the affected habitats declined by 78%. Although there were no fish in these systems, they did observe reductions in the top invertebrate predators, illustrating that bottom-up effects of this exclusion of plant material permeated throughout the food web ((Wallace, Eggert et al. 1999). Similarly, Fischer et al. (Fischer, Quist et al. 2010) suggested that differences in food availability that were associated with the presence or absence of riparian buffers likely affected the differences in fish growth they observed. In systems where allochthonous inputs sustain secondary production (including shaded, forested streams that provide rearing habitat for some salmonids), a reduction in allochthonous inputs could reduce secondary production, and consequently affect fish production. In addition to organic inputs, riparian vegetation provides shade for aquatic habitats, increases bank stability, helps buffer aquatic habitats from contaminants present upland, and helps maintain natural flow dynamics of water, nutrients and sediment (Richardson, Taylor et al. 2010).

Numerous studies illustrate the trophic linkages among plants and other photosynthetic organisms and the secondary production of fish and their prey. While it is logical that reductions in autochthonous and/or allochthonous food resources could limit consumers and predators, including juvenile salmonids, there are often a number of factors that affect the magnitude and even the direction of change within complex aquatic food webs. These relationships may be directly or indirectly affected by herbicides. The following sections briefly review some of these impacts and discuss the challenges faced in predicting how herbicides may affect salmonids and their critical habitats.

Effects of herbicides on non-target aquatic communities

Potential effects of herbicides on aquatic and riparian communities are illustrated in Figure 64 and in Figure 65. The range of effects includes direct effects (primarily negative) on photosynthetic organisms and water quality parameters as well as indirect effects (positive and negative) on multiple trophic levels and water quality. Generally, if an herbicide exposure is great enough to reduce primary production within or adjacent to aquatic habitats, there may be effects on multiple higher trophic levels, including fish. There are a number of factors that determine the magnitude of the effects as well as the direction of effects, but it is often difficult to predict those patterns from one system to the next.

Numerous studies using standard toxicity tests have demonstrated that herbicides reduce the growth and biomass of non-targeted as well as targeted photosynthetic organisms. As expected, plants and photosynthetic organisms are more sensitive to herbicides than invertebrates and fish because of the herbicides' various mechanisms of action. For example, Brock and others (Brock, Crum et al. 2004) determined HC5s (hazardous concentrations for 5% of the species) for two herbicides (metribuzin and metamitron) on a variety of taxa, and found not surprisingly that the algae and macrophytes were >100 to

>1000x more sensitive than invertebrates and fish. Similarly, Van den Brink et al. (Van den Brink, Blake et al. 2006) found that herbicides varied in their toxicity, but the relative sensitivities (based on short-term toxicity tests) of the taxonomic groups included were as follows: algae≥macrophytes>invertebrates>vertebrates. For some herbicides, algae and macrophytes were similar in their sensitivities (e.g., for atrazine and diquat, (Van den Brink, Blake et al. 2006)), but for others, such as 2,4-D (an auxin simulator), macrophytes were significantly more sensitive than all of the algae taxa included in the analyses (Van den Brink, Blake et al. 2006). In their extensive review of herbicides, Brock et al. (Brock, Lahr et al. 2000) also concluded that auxin simulators like 2,4-D were generally more toxic than other photosynthesis inhibitors to macrophytes.

The direct effects of herbicides on diverse communities of aquatic primary producers can be highly variable. In some cases, few if any effects are found. For instance, Gruessner and Watzin (Gruessner and Watzin 1996) exposed stream communities in microcosms to a low concentration of atrazine (5 µg/L) for 14 days, but found no effect on algal biomass. In other studies, the species composition of primary producers changes after exposure while abundance may increase or decrease. Wendt-Rasch et al. (Wendt-Rasch, Pirzadeh et al. 2003) found that even though macrophyte root growth in mesocosms declined following exposure to metsulfuron methyl, the biomass of periphytic algae on those macrophytes actually increased. In addition, the algal species composition was significantly different in the mesocosm exposed to the highest dose (Wendt-Rasch, Pirzadeh et al. 2003). Hartgers et al. (Hartgers, Aalderink et al. 1998) observed an initial decline in the abundance of some phytoplankton taxa following exposure to a mixture of herbicides (atrazine, diuron and metolachor), but by 14 days post-application several phytoplankton taxa had actually increased in abundance.

In addition to direct effects on primary producers, there may be direct effects of herbicides on microbial communities. The processing of organic matter by microbial communities – which includes in part making leaf litter palatable to some invertebrates – is a critical energy pathway within aquatic food webs. Despite their importance, there are

relatively few studies examining the effects of pesticides in general on microbial communities. Of the few studies regarding herbicides, it appears there may be some direct and indirect effects at relatively low concentrations. For example, DeLorenzo et al. (DeLorenzo, Lauth et al. 1999) found that microbial communities were altered following exposure to various concentrations of atrazine, with some taxa becoming more abundant and productive while others declined. In another study, the herbicide diuron limited algal growth in mesocosms, but because of this the abundance, diversity, and activity of the associated microbial community was also limited (Pesce, Fajon et al. 2006). These authors suggest that the diuron exposure ultimately decreased the capacity of the microbial community to recover when favorable conditions were provided (as was the case in the control mesocosms), and this reduced the efficiency of the microbial food web (Pesce, Fajon et al. 2006). Although it is difficult to extrapolate short-term mesocosm studies to potential longer-term effects in the natural environment, these studies suggest that exposure to herbicides can directly affect the structure as well as function of the diverse communities that are the base of aquatic food webs.

The effects of herbicides, either by reducing primary producers or changing the processes and paths through which energy flows, can have significant effects on higher trophic levels. For example, herbicides are commonly found to reduce the abundance (or biomass or growth rates) of consumers. Interestingly, these indirect effects of herbicides are often reported at concentrations well below those found to have direct effects on those consumers. The population growth rate of an aquatic oligochaete *Lumbriculus variegatus* was reduced by 50% after being exposed to only 6 µg/L of the herbicide terbutryn (Brust, Licht et al. 2001). This effect was attributed to the reduction of the food source of the oligochaete by the herbicide at a concentration three orders of magnitude lower than the concentration that caused acute toxicity to the oligochaete itself. Similarly, Dewey (Dewey 1986) found that multiple trophic levels within experimental ponds were impacted by atrazine, though effects on higher trophic levels were likely due to indirect effects (reduction in food resources). These effects throughout the food web were found at concentrations one order of magnitude lower than acute toxicity values for a common

midge (Dewey 1986). Brock et al. (Brock, Crum et al. 2004) observed long-term (lasting >8 weeks) changes in the macroinvertebrate communities within mesocosms treated with metribuzin at concentrations 20x lower than the HC5s for aquatic invertebrates. In a similar study, predatory ciliates were relatively more affected by the reduction of their prey (phototrophic flagellates) due to exposure to the herbicide prometryn than by the direct toxicity (Liebig, Schmidt et al. 2008). Finally, a number of studies have documented declines in zooplankton densities due to reductions in their phytoplankton food sources following exposure to herbicides ((DeNoyelles, Kettle et al. 1982);(Juttner, Peither et al. 1995);(Kasai and Hanazato 1995)).

These examples have illustrated that reduced primary production due to herbicide exposure can have bottom-up effects. Alternatively, if an herbicide is directly toxic to consumers, primary production may actually increase as grazing pressure declines (Rohr and Crumrine 2005). In addition, sublethal effects of herbicides on invertebrates have also been found at environmentally relevant concentrations, and this may also have effects throughout the food web. For example, Cook and Moore (Cook 2008) found the herbicide metolachlor (at an environmentally relevant concentration of 80 ug/L) altered agonistic behavior in crayfish.

Effects on water quality are also often reported. These changes are due in part to changes in community metabolism (Brock, Lahr et al. 2000). For example if photosynthetic efficiency declines, it is expected and often found that oxygen concentrations and pH decrease ((Hartgers, Aalderink et al. 1998);(Brock, Lahr et al. 2000)). These effects have been shown to be dose-dependent; for instance, Pratt et al. (Pratt, Melendez et al. 1997) found oxygen levels decreased most significantly in microcosms exposed to the highest doses of the herbicide diquat. These changes in water quality, especially significant declines in dissolved oxygen, may affect sensitive taxa but it is unclear how often this may occur in salmonid habitats.

Brock et al. (Brock, Lahr et al. 2000) concluded in their review of herbicides that indirect effects of photosynthetic inhibitors on consumers and predators occur at concentrations around the EC₅₀ for standard algae taxa; these impacts on consumers and predators are likely due to reduced availability of food resources and the effects may be delayed relative to the exposure event. Other effects on the ecosystem (e.g. blooms of insensitive algae) can occur at lower concentrations (e.g. 0.1 of the EC₅₀ of standard algae), and these effects may also be delayed. When macrophytes are impacted, organisms using those macrophytes as habitat are immediately impacted. Some studies published after the Brock et al. (Brock, Lahr et al. 2000) review note indirect effects at surprisingly low concentrations, but generally papers published since their review corroborate their findings.

Challenges in scaling up effects and making predictions across salmonid habitats

The current literature describes a wide range of effects of herbicides. While it is difficult to generalize across these studies, it is clear that many studies illustrate that herbicides can have direct and indirect effects on multiple trophic levels within aquatic food webs and often these effects occur at concentrations well below concentrations expected based on single-species acute toxicity tests. That said, it is difficult to predict the magnitude, duration, and direction that these effects may have on juvenile salmonids and their habitat because multiple factors influence these effects. These factors include but are not limited to the composition and relative abundances of taxa at the time of exposure (e.g., (Relyea 2009)), the functional redundancy among taxa within the system, and the resilience of the various communities within the system (Brock, Lahr et al. 2000). In addition, the abiotic conditions, the presence of other stressors, and the properties of the herbicides themselves (e.g. mode of action, persistence) as well as the exposure can affect the magnitude, duration and direction of effects.

Juvenile salmonids are generally opportunistic drift-feeders, and are therefore sensitive to factors that influence the general quantity and quality of invertebrate prey items. If for instance there were reductions in the production of invertebrate grazers or the inputs of invertebrate prey from riparian vegetation, salmonids may be forced to alter their foraging behavior (e.g. take more risks, select less energy-rich prey) (as shown in Figure 64 and in Figure 65). Alternatively, if there were shifts rather than reductions in the abundances and composition of the prey community within riparian and aquatic habitats, indirect impacts on salmonids may be minimal if foraging behaviors were not altered. Whether or not production of prey decreases or shifts (or increases) after exposure to herbicides will depend in part on the composition of the community (structure and function) and the relative sensitivities of those taxa. Multiple experiments conducted in mesocosms have demonstrated that the particular composition of the community at the time of exposure influences the magnitude of the impact as well as the trajectory of the recovery ((Jenkins and Buikema 1998);(Pesce, Fajon et al. 2006);(Relyea 2009);(Rohr and Crumrine 2005)), and this would likely be the case as well in salmonid habitats.

Abiotic conditions may also affect how herbicides directly and indirectly affect salmonids and their habitats. For instance, herbicides can affect water quality parameters that may indirectly affect aquatic communities. Austin et al. (Austin, Harris et al. 1991) suggest that increased algal production in oligotrophic systems after exposure to glyphosate may be due to the addition of phosphorous (in the glyphosate), and they suggest this could lead to eutrophication of salmonid habitats. Likewise, total phosphorous increased by eight-fold in earthen mesocosms treated with glyphosate (Perez, Torremorell et al. 2007). In forested watersheds in the southeastern United States, nitrogen concentrations were elevated in streams for two years after herbicides were applied (Neary, Bush et al. 1993). This effect was likely due to the increased leaching from the terrestrial environment and/or reduction in uptake within the stream. Regardless of how nutrients become elevated (from the herbicide itself or from changes in biogeochemical cycles within the watershed) elevated nitrogen and phosphorous concentrations can stimulate periphyton growth in nutrient-limited systems and

consequently affect higher trophic levels. Indirect effects from herbicides may also include an increase in stressful water temperatures due to reduced shading and long-term reductions in woody debris used for cover by salmonids from loss of riparian vegetation. If herbicides were used to reduce plant growth over a large area within a watershed, instream flow dynamics may be impacted enough to affect salmonids and their habitats (e.g., (Likens, Bormann et al. 1970)). Finally, changes such as increased turbidity (due to reduced bank stability) or decrease dissolved oxygen could have impacts on primary producers as well as consumers within salmonid habitats (Figure 64 and Figure 65).

In addition to the uncertainties associated with variable and diverse communities and the range of sensitivities they have to various abiotic conditions, there are uncertainties about how herbicides may affect aquatic systems that are impacted by other stressors. When experiments are used to examine multiple stressors, the results are often variable and again (like simpler experiments) often depend on the abiotic and biotic conditions at the time. In a series of experiments, Rohr et al. (Rohr, Elskus et al. 2004) found few interactions among food availability, drying conditions and atrazine (at 4 concentrations) on a streamside salamander, but they did find that the lethality of atrazine varied by year and may be condition dependent. These types of experiments reveal that effects may be significant, even if hard to predict. For instance, if side channel habitat in an agricultural region is already impacted by high temperatures and nutrient enrichment, the additional effects of low concentrations of herbicides may be negligible or may be synergistic. Figure 64 and Figure 65 illustrate the direct and indirect effects stemming from herbicide exposure, but they do not attempt to capture the complex web of interactions that may arise when multiple stressors affect a system.

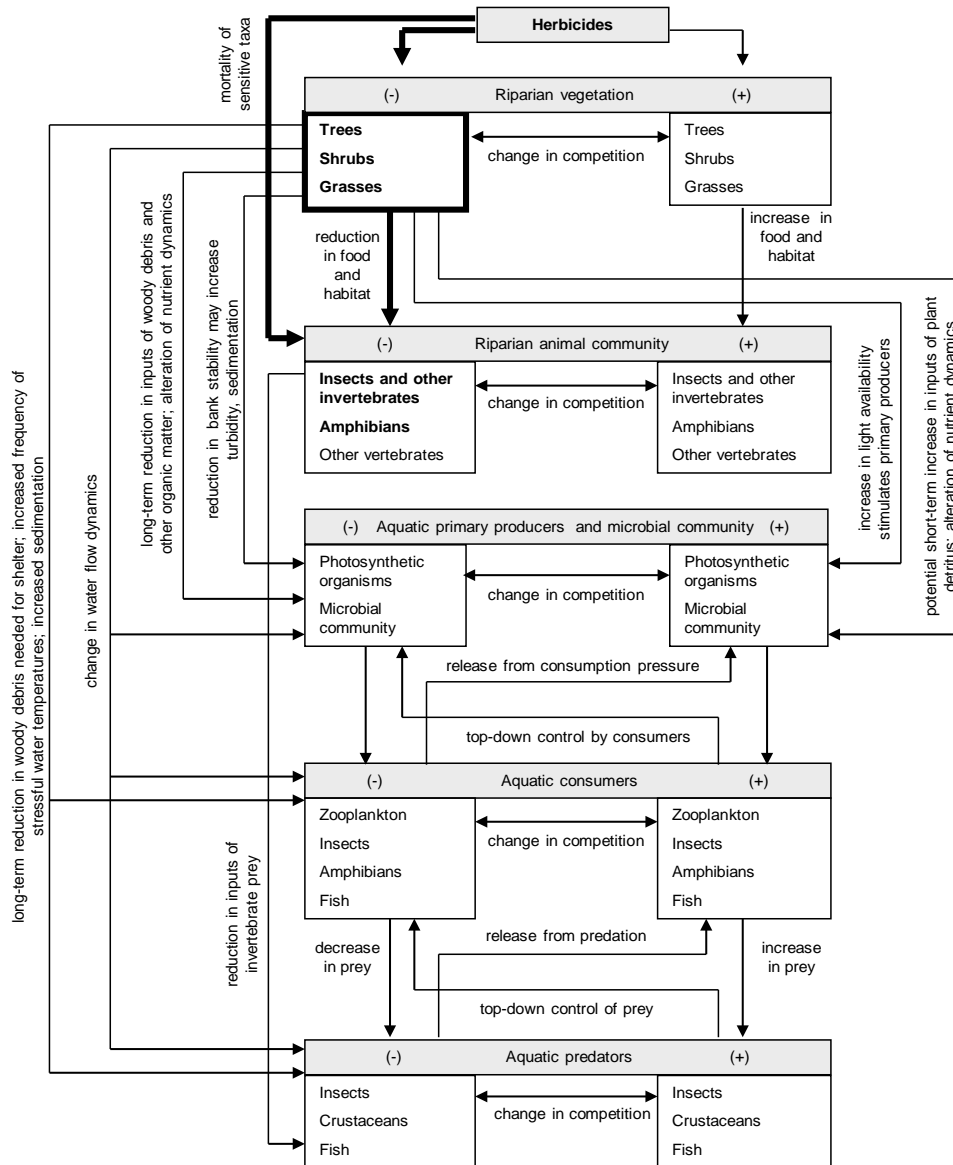


Figure 64. Part I of a conceptual model of potential effects of herbicides aquatic communities. This figure focuses on potential effects of herbicides applied to riparian areas adjacent to salmonid habitats. Bolded arrows and text note those effects that are most likely to occur based on the frequency that they are reported in the literature.

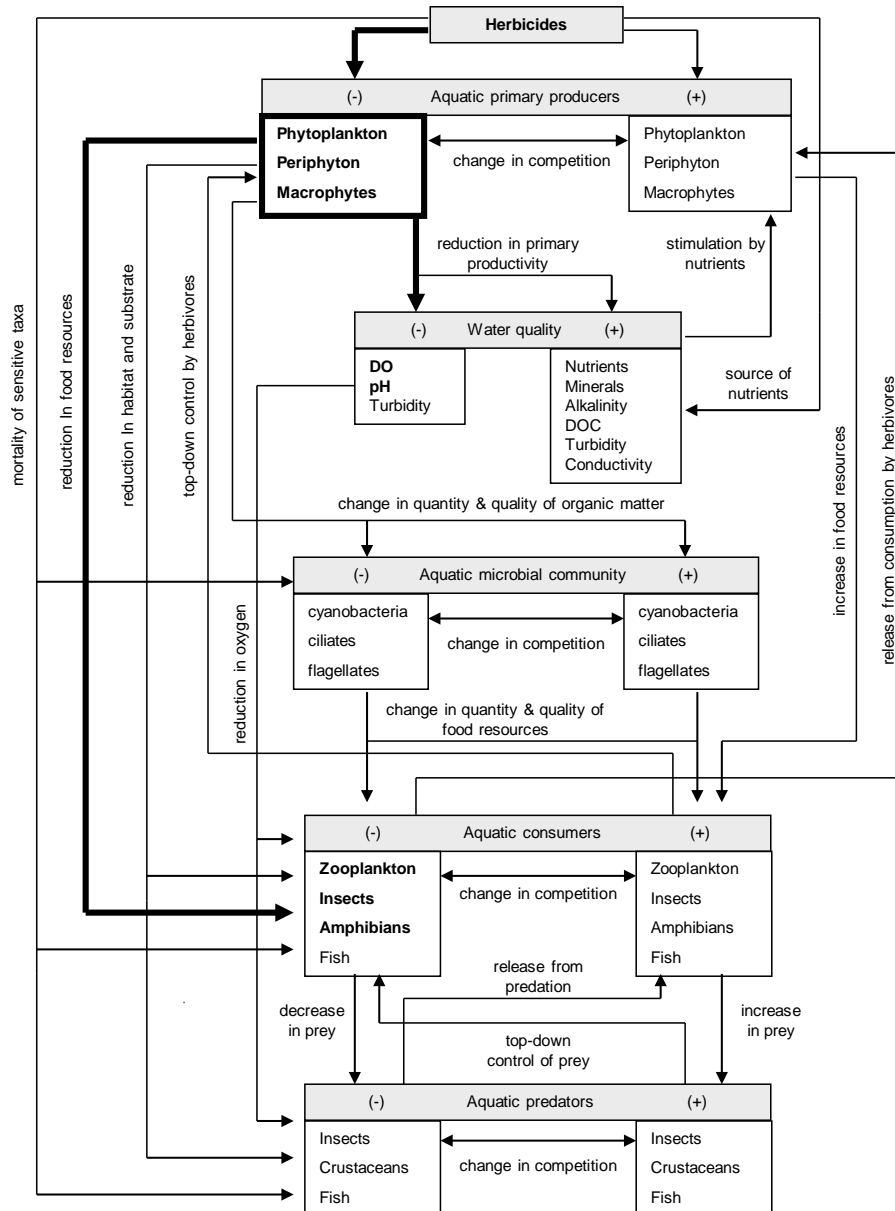


Figure 65. Part II of a conceptual model of potential effects of herbicides aquatic communities. This figure focuses on potential effects of herbicides that are applied to or otherwise reach salmonid habitats. Bolded arrows and text note those effects that are most likely to occur based on the frequency that they are reported in the literature.

Mixtures of pesticides present a particular challenge. Most of the experiments described above were conducted in mesocosms with a single exposure of a single herbicide. In field surveys in the United States as well as throughout Europe, herbicides are often among the most concentrated pesticides detected, but they are almost always found in mixtures with insecticides and fungicides ((Gilliom 2007);(Schafer, Caquet et al. 2007)). Although it is becoming more apparent that herbicides are often found in mixtures, the toxicity of the herbicides within those mixtures may depend on the composition of the mixture itself. When Van den Brink et al. (Van den Brink, Crum et al. 2009) examined the effects of a simple herbicide-insecticide mixture on mesocosm communities, they found the herbicide (atrazine) had fewer effects than expected, and they suggest this may have been due to the reduced grazing pressure that resulted directly from the reduction in invertebrates caused by the insecticide (lindane). In a series of experiments comparing effects of single herbicides, single insecticides and mixtures of these, Relyea (Relyea 2009) found that a mixture of five herbicides had relatively few effects on mesocosm communities compared to several individual insecticides, a mixture of 5 insecticides and a mixture of all 10 pesticides. One effect he did find was that chlorophyll a concentrations in phytoplankton were similarly reduced after 16 days in both the acetochlor-alone treatment as well as the 5-herbicide mixture treatment. This suggests acetochlor alone, and not the other four herbicides, likely contributed to the overall toxicity of the mixture for this response variable. It is unclear, however, how other communities exposed to the numerous possible combinations of mixtures would respond. Finally, in addition to the composition of the mixture, the dose of the mixture may also be important in determining the direction of effect. In a study on eelgrass, low concentrations of a mixture of three herbicides (glyphosate, benazone, and MCPA) were synergistic but high concentrations had an antagonistic effect (Nielsen and Dahllof 2007).

A final consideration and uncertainty in how herbicides may impact salmonids and their habitats is the question of how resilient are these aquatic ecosystems. The recovery of primary and secondary production – to rates observed prior to exposure – depends on the

communities themselves and the exposure. For instance, if herbicides persist in the landscape, exposures may occur repeatedly (or continuously) depending on application rate, precipitation, and conditions in the watershed. Michael et al. (Michael, Batzer et al. 2006) found exposures of sulfometuron occurred repeatedly, due to wash off from the upstream forest, after a single application (see also (Michael, Webber et al. 1999);(Michael 2003)). The persistence of an herbicide can affect the recovery of a community, as seen when the herbicide 3,4-dichloroaniline was added to mesocosms (Maund, Biggs et al. 2009). This herbicide was initially added at a concentration equal to the median LC₅₀ value of taxa in the mesocosms, but it persisted several months (median dissipation time was estimated as 30 days). The lack of recovery of populations within the mesocosms by 10 months and the delay of recovery even when colonists were added following exposure was attributed to the persistent toxicity (Maund, Biggs et al. 2009). Generally, photosynthesis has been found to resume rapidly once exposure stops, while indirect effects on longer-lived taxa can persist much longer ((Brock, Lahr et al. 2000);(Brock, Crum et al. 2004)). This difference can lead to dynamics in trophic interactions (e.g. alterations between top-down and bottom-up control). These fluctuations have been found to stabilize in mesocosms within weeks to months, but for juvenile salmonids that require reliable food resources daily, this time period of recovery may be too long.

These uncertainties make it difficult to predict how herbicides will affect salmonids and their critical habitats, but they do not preclude that there may be an impact.

2,4-D and Triclopyr (Synthetic Auxins)

Mode of Action

2,4-D, a phenoxyacetic herbicide, is an auxin inhibitor (Cremlyn 1991). Auxin (also known as indole-3-acetic acid, or IAA) is a plant hormone which regulates cell elongation. The phenoxyacetic herbicides produce the same reaction in the plants, but are not broken down by the plant and cause it to grow in an unregulated manner

eventually resulting in death (Cremlyn 1991). The phenoxyacetic herbicides are also known as phenoxyalkanoic acids.

2, 4-D contains an acid group (-COOH), and is marketed in a variety of forms, including a sodium salt, various amines, and a number of esters. The acid structure and examples of an amine and ester are shown in Figure 66. The salt, amines, and ester forms have different environmental fate and toxicity properties. The salt and amines rapidly break down to the acid form, especially in water, and toxicity of these forms is more similar to the acid. The esters break down more slowly, and their toxicity is greater (sometimes orders of magnitude) than the acid or salts. Greater toxicity of the esters of 2,4-D and similar compounds appears to be related to faster uptake by the organisms (Barron, Mayers et al. 1990). Auxins are specific to plants, and toxic effects in fish appear to be a general narcosis effect (Barron, Mayers et al. 1990).

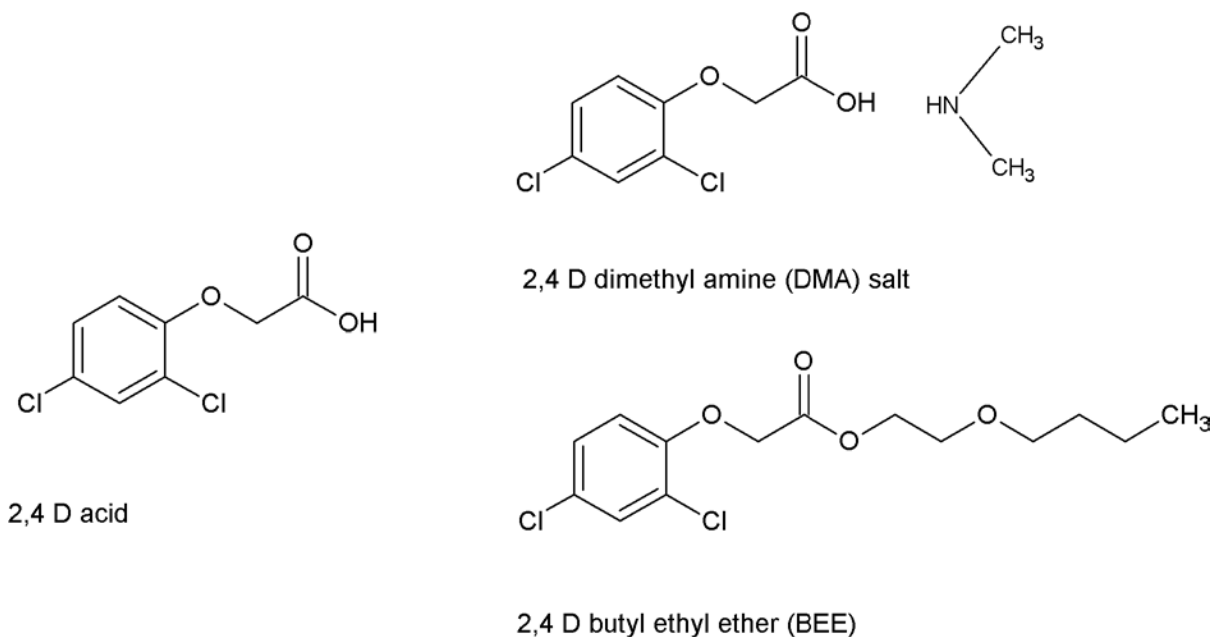


Figure 66. 2,4-D acid, and example amine and ether forms.

Triclopyr is a pyridine herbicide, and is also an auxin inhibitor. In the basic form, it is an acid (-COOH), and like 2,4-D is marketed as both an amine (triclopyr TEA) and an ester (triclopyr BEE). However, it is available in less forms than 2,4-D, and by definition of the settlement agreement, we are only addressing the butoxy ester ester (BEE) form in

this Opinion. The triclopyr acid and triclopyr BEE structures are shown in Figure 67. Similar to 2,4-D, the ester form is more persistent in the environment and more toxic than the amine form.

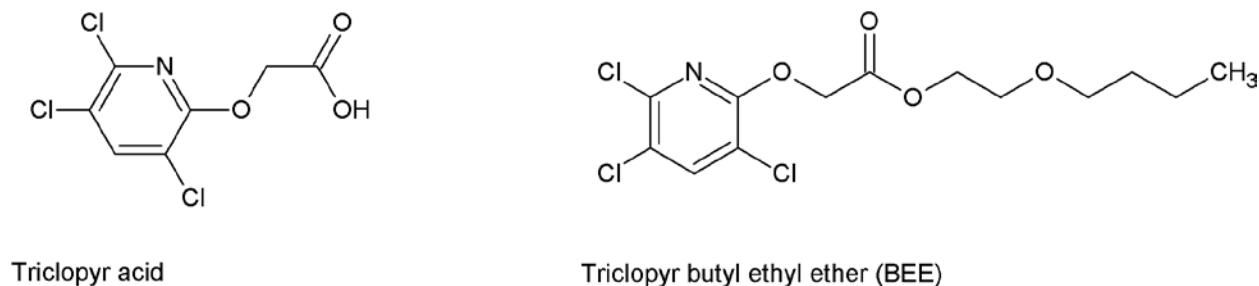


Figure 67. Triclopyr acid and triclopyr BEE.

Evaluating Toxicity: Using acid versus ester data

We present environmental concentrations (in the *Exposure Section*) and toxicity data in terms of acid equivalents (a.e.s). We present toxicity values separately for the acid, amine, and ester forms. Given the short half-life of the triclopyr ester form, the registrant for triclopyr BEE has suggested evaluating the toxicity of the ester in the environment is not relevant, and that toxicity values for the acid should be used (Dow 2010). In the triclopyr salmonid BE, EPA estimated acute EECs for the BEE form and used the toxicity data for the BEE form (EPA 2004). They did not evaluate chronic effects. Concentrations were expressed as mg or μg a.i./L, and not converted to a.e. In the more recent California red-legged frog assessment, EECs were estimated based on the fate properties of the acid form, and other than the chronic fish endpoint, toxicity values for the BEE were used (EPA 2009). Water concentrations were expressed in a.e.s.

Based on a pharmacokinetic study done in coho salmon, the BEE form of triclopyr was taken up by the fish extremely rapidly (Barron, Mayers et al. 1990). Essentially all uptake for the BEE occurred in the first 12 hours of exposure, and the majority occurred within the first 6 hours. The uptake rate for the acid form is much slower. This is consistent with what might be predicted based on the log K_{ow} s for the two forms. The

fish quickly metabolized the ester to the acid form. Barron et al (1990) postulated the apparent difference in toxicity between the two forms of triclopyr was associated with accumulation of the a.i. (*i.e.* differences in effective dose). We concur with this assessment. Thus, for short-term (acute) exposures, we use the toxicity values from the BEE form, and for longer-term (chronic) exposures we use toxicity values for the acid (preferable) or amine (if acid value is not available) form.

Although we have not reviewed a pharmacokinetic study on 2,4-D esters, given the differences in log K_{ow} s for these compounds (IDE 3.81, BEE 4.35, and EHE 5.78 (EPA 2009) as compared to the log K_{ow} for the acid (2.81(EPA 2009))), we believe it is reasonable to assume the differences in toxicity between the 2,4-D amines and esters are also related to the uptake by organisms. Hughes (1973) tested a 78% butyl ester of 2,4-D, with results reported in terms of a.i. Both striped bass larvae and fingerlings were tested, and author reports LC_0 , LC_{50} , and LC_{100} for both age classes (Hughes 1973). At 96 h those values were 100 $\mu\text{g/L}$, 150 $\mu\text{g/L}$, and 250 $\mu\text{g/L}$, respectively for the larvae, and 2,000 $\mu\text{g/L}$, 3000 $\mu\text{g/L}$, and 4,000 $\mu\text{g/L}$ for the fingerlings. Authors also provide all three endpoints at 24 h, 48, h and 72 h. For both larvae and fingerlings, LC_{xx} values at 24 h and all other time periods were the same as at 96 h, confirming the rapid uptake. Thus, as with triclopyr BEE, for the ester forms we use the ester toxicity values to evaluate short-term (acute) exposures, and acid (preferable) or amine (if acid value is not available) form to evaluate longer-term (chronic) exposures.

Temperature and toxicity

We located no information indicating temperature specifically affects the toxicity of 2,4-D or triclopyr. However, we do note higher water temperatures can affect salmonids in two ways, regardless of specific chemical effects. Higher water temperatures will increase the metabolic rate for fish, thus increasing the rate at which they process the toxicant. Depending on the chemical, this may be either beneficial or detrimental. Water

temperatures higher than optimum increase general physiological stress for salmonids, making them more susceptible to other stressors.

pH and toxicity

We located no information indicating pH specifically affects the toxicity of 2,4-D or triclopyr. We do anticipate pH will affect the availability of the a.i.s

Toxicity of 2,4-D and Triclopyr (Assessment Endpoints)

Direct Effects to Salmonids

We evaluated potential effects to salmonids based on toxicity information included in the salmonid BEs ((EPA 2004), (EPA 2004)), the more recent California red-legged frog assessments ((EPA 2009; EPA 2009)), and the REDs ((EPA 2005), (EPA 1998)). These data are presented in the tables, separated by acid, amine, and ether forms. We also searched open literature for endpoints not reported in EPA documentation, and data available since their publication. These are discussed qualitatively.

Survival

Survival is evaluated primarily based on guideline tests used to determine the LC₅₀ of a population of test fish (Table 107).

Table 107. Acute toxicity data for fish.

Species		96-h LC ₅₀ Median Concentration ¹ (mg a.e./L)	Range (mg a.e./L)	n	Source
2,4-D					
Salmonids					
Rainbow trout	<i>Oncorhynchus mykiss</i>	Acid 358 Amines 1,203 Esters 1.05	NA 162-2,244 0.45-14.5	1 2 7	(EPA 2009) (Appendix F)
Other fish					
Bluegill	<i>Lepomis macrochirus</i>	Acid 292 Amines 234 Esters 0.69	263-320 101-1,722 0.26-11.9	2	(EPA 2009) (Appendix F)
Fathead minnow	<i>Pimepheles promelas</i>				
Triclopyr					
Rainbow trout	<i>Oncorhynchus mykiss</i>	Acid 117 TEA 79.2 BEE 0.47	NA	1	(EPA 2009) (Appendix A)
Bluegill sunfish	<i>Lepomis macrochirus</i>	Acid 148 TEA 155.4 BEE 0.26	NA	1	(EPA 2009) (Appendix A)

¹ 96 hr test data on 95% technical only. Most data from static tests, although one flow-through test is included in cutthroat trout data set, and one in lake trout data set.

² 96 hr test endpoint calculated by authors

Reproduction and Growth

Data presented in Table 108 below are based on guideline study information included in the EPA BEs. For triclopyr, we also located an open literature study on the effects of triclopyr TEA to fathead minnow (Mayes, Dill et al. 1984). It describes both acute tests, and a 28-d embryo-larval test. For the acute tests, they note most toxicity occurs within the first 24 h. Chronic exposure concentrations were 13 to 112 mg ai/L, selected based on acute toxicity values. Authors noted in this test that they saw little evidence of effects on hatchability, development, and growth.

Table 108. Chronic toxicity data for fish.

Species		NOAEC LOAEC (mg a.e/L)	Endpoint Affected (Exposure duration)	Source
<i>2,4-D</i>				
Fathead minnow	<i>Pimephales promelas</i>	Acid 63.4 Amines 17 (n=2) Esters 0.0792 Acid 102 Amines 45.1 (n=2) Esters 0.1452	Reduction in growth Reduction in larval survival	(EPA 2009) (Appendix F)
<i>Triclopyr</i>				
Rainbow trout	<i>Oncorhynchus mykiss</i>	Acid ND TEA ND BEE 0.019 Acid ND TEA ND BEE 0.034	Reduction in growth (Not given)	(EPA 2009) (Appendix A)
Fathead minnow	<i>Pimephales promelas</i>	Acid ND TEA >32.2 BEE ND Acid ND TEA <50.2 BEE ND	Reduction in growth (Not given)	(EPA 2009) (Appendix A)

Swimming

During a 12 d exposure to 400 mg/L 2,4-D acid, (Gomez, Masot et al. 1998) noted irregular swimming behavior, hemorrhaging, and pale gills. Hemorrhagic fluid appeared within 24 h of exposure. We located no other reports specifically of impaired swimming behavior.

We located no data regarding effects of triclopyr on swimming.

Olfaction

A study tested serine-evoked electro-olfactogram (EOG) responses in coho salmon parr (*O.kisutch*) exposed to 1, 10, and 100 mg/L of 2,4-D acid. No response was noted for the two lower concentrations, but exposure to 100 mg/L quickly (within 2 minutes)

eliminated the EOG response (Tierney, Ross et al. 2006). EOG response partially recovered after 30 minutes of exposure, but not to the level of control fish. Authors cite some other research indicating salmon may have an avoidance response for 2,4-D (Tierney, Ross et al. 2006).

We located no data regarding effects of triclopyr on olfaction.

Cellular-level Effects and Carcinogenicity

The phenoxy herbicides, including 2,4-D, have been identified as peroxisome proliferators (Ackers, Johnston et al. 2000) Peroxisome proliferation causes oxidative damage in cells, especially liver hepatocytes (Rakitsky, Koblyakov et al. 2000). This damage may induce cancer by causing uncontrolled growth, rather than via modifications in genes (*i.e.*, an epigenetic mechanism rather than genotoxicity.) Epigenetic mechanisms are considered threshold-based, and are frequently regulated for human health risk based on No Observable Effect Levels (NOELs) and safety factors (Rakitsky, Koblyakov et al. 2000). Currently, OPP considers no cancer-based endpoints for ecological risk.

Ackers et al (2000) evaluated peroxisome proliferation in mummichog (*Fundulus heteroclitus*) with *in vivo* exposures. Authors concluded 2,4-D does induce peroxisome proliferation in mummichog at a concentration of 1 mg/L 2,4-D acid. The exposure duration was 21 days, but increases in the protein they measured (PMP70) were noted after 7 days of exposure. Some increase was seen in the 0.01 mg/L exposure group, but it did not occur in a dose-response pattern. In addition to potential tumor induction, authors cite other research indicating peroxisome proliferation is associated with modified sex-steroid metabolism, lipid metabolism, and development of young.

Gomez et al (1998) evaluated 2,4-D induced lesions in tench (*Tinca tinca*) exposed *in vivo* to a concentration of 400 mg/L for 12 days. Symptoms appeared within the first 24 h of exposure, and included hemorrhage, enlargement of the kidney parenchyma, and

fluid in the cranial cavity. Microscopic observations of the kidney showed increased presence of vacuoles, increased phagocytic activity, and necrosis (Gomez, Masot et al. 1998). Their work indicates even short exposures may have marked sublethal effects, depending on exposure concentrations. Authors selected test concentration based on a tench 96-h LC₅₀ of 800 mg/L (Gomez, Masot et al. 1998).

Cope et al (1970) conducted a long-term (5 months) outdoor study of bluegill (*L. macrochirus*) exposed to the propylene glycol butyl ether ester of 2,4-D. Nominal exposure concentrations were 0.1, 0.5, 1.0, 5.0 and 10.0 mg/L. There was some mortality in the 5 and 10 mg/L treatment groups, and these groups also were delayed in spawning by two weeks. Number of nests and offspring produced in the ponds were similar, even with the delayed spawning in the higher treatment groups. Note that predators and competitors were excluded from the ponds. Some residues of 2,4-D were detected in the pond for six weeks, but concentrations declined markedly after the first two weeks. Liver lesions were present in fish exposed to concentrations of ≥ 1 mg/L. The number of lesions peaked at about 2 weeks, and then declined in most groups. They were still measurable in the 5 and 10 mg/L groups at 4 weeks, and could be measured in the 10 mg/L group at 9 weeks. Based on this study, it appears that liver effects are closely correlated with the delayed spawn in the 5 mg/L and 10 mg/L groups. As concentrations of 2,4-D decline (either by partitioning to the sediment, or possibly conversion to the acid form, which was not measured), some of the groups appeared to recover liver function (Cope, Wood et al. 1970).

McBride et al (1981) noted physical changes in the gill, liver, and interrenal tissue of sockeye fry (*O. nerka*) exposed to 0.7 and 1.0 mg/L of the butoxyethanol ester (BEE) of 2,4-D. Damage to the interrenal occurred within 48 h of exposure. Following 96-h of exposure, damage was noted at 0.3 mg/L. Sockeye smolts were slightly more resistant, displaying damage only to the interrenal, and at slightly higher concentrations (0.7 – 1.0 mg/L). Following exposure to uncontaminated water, the damage at the higher concentration was reversed (McBride, Dye et al. 1981). Authors cite a preexisting study

(Rogers and Stalling 1972), which indicates uptake of the BEE form of 2,4-D is rapid, and peak tissue accumulation occurs within the first 6 h of exposure.

We located no data regarding effects of triclopyr on cellular damage.

Endocrine Disrupting Effects

2,4-D is on the EPA list to be evaluated for endocrine disrupting effects. We did not locate any studies evaluating 2,4-D specifically in fish, but did locate a study regarding anti-androgen effects in alligators (Crain, Guilette Jr. et al. 1997). Authors considered the endpoints of plasma hormone, gonadal-adrenal mesonephros (GAM) aromatase activity, and gonadal histopathology. Test procedures were effective, as they did detect differences for their positive control and some of the other chemicals tested, but 2,4-D did not affect any of the parameters tested.

Triclopyr is not on the EPA list to be evaluated for endocrine disrupting effects. We did not locate any studies evaluating such effects for fish.

Indirect Effects to Salmonids (Prey and Habitat Modifications)

Indirect effects on salmon include reductions in prey base (aquatic invertebrates), disruptions in primary productivity in the stream (phytoplankton and macrophytes), and effects on riparian vegetation.

Aquatic Invertebrates (Acute and Chronic Toxicity)

Values presented in the tables below (Table 109 and

Table 110) summarize aquatic invertebrate data tabulated in the appendices of the California red-legged frog BEs ((EPA 2009; EPA 2009))

Table 109. Acute toxicity data for aquatic invertebrates.

Species		48 h ¹ or 96-h ² EC ₅₀ Median Concentration ¹ (mg a.e./L)	Range (mg a.e./L)	n	Source
<i>2,4-D</i>					
Waterflea (Cladoceran)	<i>Daphnia magna</i>	Acid 4.97 Amines ND Esters 3.4	NA ND 2.2-11.9	1 0 3	EPA RLF 2009 (Appendix F)
<i>Triclopyr</i>					
Waterflea (Cladoceran)	<i>Daphnia magna</i>	Acid 132.9 TEA 346 BEE 0.25	NA	1	EPA RLF 2009 (Appendix A)

Table 110. Chronic toxicity data for aquatic invertebrates.

Species		NOAEC LOAEC (µg/L)	Endpoint Affected	Source
<i>2,4-D</i>				
Waterflea (Cladoceran)	<i>Daphnia magna</i>	Acid 79 Amines 16.05 Esters 0.2 Acid 151 Amines 25.64 Esters 0.483	Not discussed in RLF or Salmon BEs	EPA RLF 2009 (Appendix F)
<i>Triclopyr</i>				
Waterflea (Cladoceran)	<i>Daphnia magna</i>	Acid ND TEA 25 BEE ND Acid ND TEA 46.2 BEE ND	Reduction in young and total brood size	EPA RLF 2009 (Appendix A)

Aquatic Plants (Phytoplankton and Vascular Plants)

Given that 2,4-D and triclopyr are both herbicides, we anticipate the most sensitive receptors in salmon habitat will be plants. Instream plants include various types of algal species and vascular plants. Generally the phytoplankton provide an energy source for

the stream and the macrophytes are a structural component, providing attachment sites for other organisms and refugia for juvenile fishes. Reductions in primary productivity or modifications in community structure via removal of sensitive species can result in “bottom-up” trophic cascades which may adversely affect salmonids. Loss of structure provided by macrophytes may result in decreased population of aquatic invertebrates or increased predation on juvenile salmonids. Table 111 below summarizes toxicity data for aquatic plants.

Table 111. Toxicity data for aquatic plants.

Species		EC ₅₀ Median Concentration (mg a.e./L)	Range (mg a.e./L)	n	Source
2,4-D					
Green algae	<i>Selenastrum capricornutum</i>	Acid ND Amines 41.69 Esters 1.28 NOAEC Acid ND Amines 16 Esters 0.59	NA 3.88-156.5 0.066-17.14	0 10 7	EPA RLF 2009 (Appendix F)
Blue-green algae	<i>Anabaena flos-aquae</i>				
FW diatom	<i>Navicula pelliculosa</i>				
SW diatom	<i>Skeletonema costatum</i>				
Vascular plant	<i>Lemna gibba</i>	Acid 0.695 Amines 0.3896 Esters 0.3637	NA 0.2992-0.48 0.33-0.3974	1 2 2	EPA RLF 2009 (Appendix F)
		NOAEC Acid 0.0581 Amines 0.1388 Esters 0.1015	NA 0.0476-0.23 0.062-0.141	1 2 2	
Triclopyr					
Green algae	<i>Selenastrum capricornutum</i>	Acid 29.8 TEA 12.1 BEE 2.5	NA	1	EPA RLF 2009 (Appendix A)
FW Diatom	<i>Navicula pelliculosa</i>	Acid ND TEA 10.6 BEE 0.07	NA	1	EPA RLF 2009 (Appendix A)
Blue-green algae	<i>Anabaena flos-aquae</i>	Acid ND TEA 4.1 BEE 1.42	NA	1	EPA RLF 2009 (Appendix A)
Duckweed	<i>Lemna gibba</i>	Acid ND TEA 6.1 BEE 0.86	NA	1	EPA RLF 2009 (Appendix A)

2,4-D and Triclopyr Degradate Toxicity

In this assessment, we have primarily focused on the acid forms of both 2,4-D and triclopyr as these are the forms most commonly found in the aquatic environment. EPA indicates they have no ecological risk concerns for 2,4-D degradates, and provide no data for any (EPA 2009). We have located no information that would cause us to draw a different conclusion. The 2,4-D assessment also discusses potential concerns related to dioxin (2,3,7,8-TCDD) impurities which may occur in 2,4-D and concludes there are no

ecological risk concerns (EPA RLF 2009, pg 101). In soil, triclopyr also breaks down to 3,5,6-trichloro-2-pyridinol (TCP) and 2,5,6-trichloro-2-methoxypyridine. Limited data were available, but values presented in the California red-legged frog assessment show TCP to be less toxic than the BEE on an acute basis, but more toxic than the TEA (EPA 2009). EPA did not evaluate the TCP, stating “toxicity data for the degradate (Table 112) indicates that when converted to acid equivalent TCP is less toxic than the most sensitive endpoint for triclopyr” ((EPA 2009), p 24).

Table 112. Degradate toxicity data for Triclopyr.

Species		LC ₅₀ or EC ₅₀ Concentration (mg a.e./L)	Range (mg a.e./L)	n	Source
<i>Triclopyr</i>					
Rainbow trout	<i>Oncorhynchus mykiss</i>	1.9	NA	1	EPA RLF 2009 (Appendix A)
Bluegill sunfish	<i>Lepomis macrochirus</i>	16.1	NA	1	EPA RLF 2009 (Appendix A)
Aquatic invertebrate	<i>Daphnia magna</i>	13.4	NA	1	EPA RLF 2009 (Appendix A)
Green algae	<i>Selenastrum capricornutum</i>	2.3	NA	1	EPA RLF 2009 (Appendix A)
Blue-green algae	<i>Anabaena flos-aquae</i>	2.3	NA	1	EPA RLF 2009 (Appendix A)

Microbial Community Effects (Sediment, Soil, and Water Column)

Given the nature of 2,4-D and triclopyr and their specificity as auxin inhibitors, we do not anticipate adverse effects on the microbial community.

Riparian Vegetation

Riparian vegetation is important for providing shade to the stream, stabilizing the stream banks, reducing sedimentation, and providing allochthonous input, both in terms of plant material and terrestrial insects. Generally there is not good data regarding the effects of herbicides on wild plants, other than weed species, but EPA requires submission of crop effects data as part of the registration process. We believe this provides a reasonable basis for evaluating effects on herbs and forbs. Based on typical uses, for 2,4-D we

expect that woody shrubs and trees are likely to be more resistant. Triclopyr is known to be effective on woody shrubs and trees. Although we did not locate specific toxicity data for those types of plants, we make the conservative assumption that sensitivity of these plants is similar to the tested crops. Guideline studies determine EC₂₅s of end-use products on the endpoints of vegetative vigor and seedling emergence. We present the most sensitive endpoints for each plant type (monocots and dicots) in Table 113 based on data summarized in EPA California red-legged frog assessments ((EPA 2009; EPA 2009)).

Table 113. Terrestrial plant data.

Test	Monocot EC ₂₅ ^{1,2} (lb a.e./A)Test	Dicot EC ₂₅ (lb a.e./A)	Source
<i>2,4-D</i> ¹			
Vegetative vigor	Acid <0.0075 Amines 0.04 Esters 0.2016	Acid 0.0075 Amines 0.003 Esters 0.02	EPA RLF 2009 (Appendix F)
Seedling emergence	Acid 2.1 Amines 0.203 Esters 0.218	Acid 0.033 Amines 0.273 Esters 0.037	EPA RLF 2009 (Appendix F)
<i>Triclopyr</i> ²			
Vegetative vigor	Acid ND TEA 0.0114 BEE 0.063	Acid ND TEA 0.005 BEE 0.006	EPA RLF 2009 (Appendix A)
Seedling emergence	Acid ND TEA >0.23 BEE 0.053	Acid ND TEA ND BEE 0.045	EPA RLF 2009 (Appendix A)

¹Median of values in Appendix if more than one available, ²Most sensitive values in Appendix.

Summary of Toxicity Data

Assessment endpoints and associated concentrations are summarized in Table 114 for 2,4-D and in Table 115 for triclopyr.

Table 114. Assessment Endpoints and Measures for 2,4-D.

Note: Units in this table are µg ae/L, other tables are mg ae/L

Assessment Endpoint		Assessment Measure	Median Concentration ^{1,2} (µg ae/L)	Range (µg ae/L)	n			
Direct Effects on Salmonids	Survival	<i>Ester</i> Salmonid Acute LC ₅₀ ¹ Other Fish Acute LC ₅₀ ¹	1,050 690	450-14,500 260-11,900	7 2			
	Growth	<i>Acid</i> Chronic NOAEC ¹ Chronic LOAEC ¹	634 102,000	NA	1			
	Reproduction							
	Swimming	We located no data regarding effects of 2,4-D on swimming.						
	Olfaction	Modifications in EOG response at 100 mg/L (Tierney, Ross et al. 2006).						
	Endocrine Disruption	We located no data regarding effects of 2,4-D on endocrine disruption in fish. One study of 2,4-D did not cause endocrine disruption in alligators (Crain, Guilette Jr. et al. 1997).						
	Cellular Damage, Carcinogenicity	<i>Acid</i> Peroxisome Proliferation (Ackers, Johnston et al. 2000)	1,000	NA	1			
<i>Ester</i> Interrenal damage (McBride, Dye et al. 1981)		300	1					
Effects on Prey (Aquatic Invertebrates)	Survival	<i>Ester</i> Acute Invert EC ₅₀ ¹	3,400	2,200-11,900				
	Growth	<i>Acid</i> Chronic NOAEC ¹ Chronic LOAEC ¹	79,000 151,000	NA	1			
	Reproduction							
Effects on Primary Productivity,	Biomass & Abundance	<i>Ester</i> Algal EC ₅₀ ¹	1,280	66-17,140	7			
Effects on Submerged and Emergent Vegetation	Biomass & Abundance	<i>Ester</i> Vascular Plant EC ₅₀ ¹	364	330-397	2			
Effects on Ecosystem Functioning	Community Metabolism	No data located						
Effects on Riparian Vegetation	Biomass & Abundance	<i>Amine & Ester</i> Vegetative Vigor ^{1,2}	Monocot Amine 0.40 Ester 0.202	NA 0.190-0.218	1 3			
			Dicot Amine 0.003 Ester 0.020	NA 0.004-0.020	1 3			

¹If more than one value was available. If only one value was available, the actual number is given.² Terrestrial plant endpoints given in lb ae/A

NA Not applicable, only one value available

Table 115. Assessment Endpoints and Measures for Triclopyr.

Note: Units in this table are µg ae/L, other tables are mg ae/L

Assessment Endpoint		Assessment Measure	Median Concentration ^{1,2} (µg ae/L)	Range (µg ae/L)	n
Direct Effects on Salmonids	Survival	<i>BEE</i> Salmonid Acute LC ₅₀ ¹ Other Fish Acute LC ₅₀ ¹	470 260	NA NA	1 1
	Growth	TEA Chronic NOAEC ¹ Chronic LOAEC ¹	>32,200 <50,200	NA	1
	Reproduction	TEA Chronic NOAEC ¹ Chronic LOAEC ¹	No data available		
	Swimming	We located no data regarding effects of triclopyr on swimming.			
	Olfaction	We located no data regarding effects of triclopyr on olfaction.			
	Cellular Damage	No specific studies on cellular damage located			
Effects on Prey (Aquatic Invertebrates)	Survival	<i>BEE</i> Acute Invert EC ₅₀ ¹	3,400	2,200-11,900	3
	Growth	TEA Chronic NOAEC ¹ Chronic LOAEC ¹	No data available		
	Reproduction	TEA Chronic NOAEC ¹ Chronic LOAEC ¹	25,500 46,200	NA	1
Effects on Primary Productivity, Submerged and Emergent Vegetation	Biomass & Abundance	Algal and Vascular Plant EC ₅₀ ¹	1,140	70-2,500	4
Effects on Ecosystem Functioning	Community Metabolism	No data located			
Effects on Riparian Vegetation	Biomass & Abundance	<i>BEE</i> Vegetative Vigor EC ₂₅ ^{2,3} Seedling Emergence EC ₂₅ ^{2,3}	Monocot 0.053 Dicot 0.006	NA	1

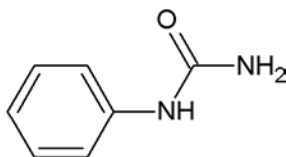
¹If more than one value was available. If only one value was available, the actual number is given.² Terrestrial plant endpoints given in lb ae/A³ Most sensitive plant values as given in RLF assessment

NA Not applicable, only one value available

Diuron and Linuron (Photosystem II Inhibiting Herbicides)

Mode of Action

Diuron and linuron are members of a class of pesticides known as phenyl ureas (Figure 68). The basic structure for this class includes an aromatic ring (-phenyl) attached to the urea molecule (-NH₂CONH₂). One or more positions on the phenyl ring are often substituted, as are one or more -R groups on the terminal nitrogen atom (Kamrin 1997). Diuron and linuron are very similar in structure, with two chlorine atoms substituted on the phenyl ring. The difference between the two is that diuron has two methyls substituted on the terminal nitrogen, whereas linuron has one methyl group and one methoxy group. Other phenyl ureas currently registered in the U.S. include fluometuron, siduron, tebuthiuron, and thiadizaunon. It appears monouron used to be registered in the U.S. but was cancelled during the re-registration process (www.epa.gov/pesticides/reregistration/status.htm).

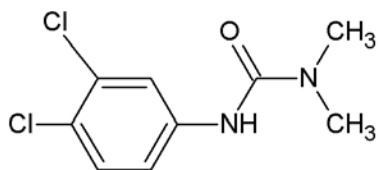


Phenylurea

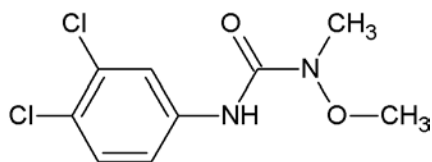
Figure 68. Basic phenylurea structure.

Diuron and linuron (

Figure 69) are systemic photosynthesis inhibitors, and are often used for controlling both annual and perennial grasses (Kamrin 1997). Phenyl ureas bind to active sites in the plant chloroplasts, interfering with the photosystem II pathway that typically fixes CO₂ and produces energy. An additional effect of the inhibition is production of reactive peroxides, which break down cellular structures in the plant, including the cell membrane. Visible manifestations of the inhibition and cellular breakdown include spotting and browning of the leaves.



Diuron



Linuron

Figure 69. Structures of diuron and linuron.

Temperature and toxicity

We located no information indicating temperature specifically affects the toxicity of diuron or linuron. However, we do note higher water temperatures can affect salmonids in two ways, regardless of specific chemical effects. Higher water temperatures will increase the metabolic rate for fish, thus increasing the rate at which they process the toxicant. Depending on the chemical, this may be either beneficial or detrimental. Water temperatures higher than optimum increase general physiological stress for salmonids, making them more susceptible to other stressors.

pH and toxicity

We located no information indicating pH specifically affects the toxicity of diuron or linuron.

Toxicity of Diuron and Linuron (Assessment Endpoints)

Direct Effects to Salmonids

We evaluated potential effects to salmonids based on toxicity information included in the salmonid BEs ((EPA 2003),(EPA 2004), the more recent California red-legged frog assessments ((EPA 2008),(EPA 2009)), and the REDs ((EPA 1995);(EPA 2003). We also searched open literature for endpoints not reported in EPA documentation, and data available since their publication.

Toxicity data for diuron was reported differently in the salmonid BE (EPA 2003) and the California red-legged frog assessment (EPA 2009) and we had difficulty reconciling the two, even after reference to the RED (EPA 2003). It appears much of the toxicity data is derived either from Mayer and Ellersieck 1986, a compilation of toxicity test data from a U.S. FWS laboratory (Mayer and Ellersieck 1986), or open source publications rather than registrant-submitted guideline tests. Sometimes the sources are referred to by MRID number, and sometimes by author. In some cases, neither MRID number nor author is listed in the bibliography of either the main document or appendices. Thus, wherever possible, we obtained original sources, and in some cases, recalculated genus or species medians. Sources we used or reference sources as cited in EPA documentation are noted at appropriate locations in the data tables. As with the other a.i.s addressed in this Opinion, the ecological effects appendices from the California red-legged frog assessments, which are generally EPA's most recent and comprehensive listing of toxicity data, are included in *Appendix 7*.

Based on what was presented in the EPA documents, and what we located in literature, there are less data available for linuron than diuron, especially for chronic (reproduction and growth) endpoints. As with diuron, we used original sources whenever possible.

Both diuron (<http://www.epa.gov/endo/pubs/prioritysetting/draftlist2.htm>) and linuron (<http://www.epa.gov/endo/pubs/prioritysetting/finallist.html>) have been placed on EPA's list for evaluation endocrine-disrupting compounds (EDCs). According to EPA's website, screening tests for initial a.i.s are due in February 2011, so we do not anticipate any data from those tests will be available for this consultation. We did locate some open literature studies on endocrine-disrupting properties of linuron. Information from those studies are included in the toxicity data. Other chemicals on the evaluation list for endocrine effects are 2,4-D, captan, and chlorothalonil (<http://www.epa.gov/endo/pubs/prioritysetting/finallist.html>).

Survival

Survival is evaluated primarily based on guideline tests used to determine the LC₅₀ of a population of test fish.

Diuron

Mayer and Ellersieck (1986) report a range of tests on diuron for several salmonid species, including cutthroat trout and rainbow trout, which are of the same genus (*Oncorhynchus*) as the salmonids considered in this Opinion, and lake trout, which are also salmonids, but of a different genus (*Salvelinus*). They also report a number of tests on the bluegill sunfish, a warm-water fish. Values below were derived from the 96 h tests with 95% technical a.i. (Mayer and Ellersieck 1986). Raw data downloaded from the USGS website are contained in *Appendix 7*. Medians for each species were calculated using an Excel spreadsheet. Overall, salmonids (96 h LC₅₀ range 700-7,700 µg/L) appear slightly more sensitive than most of the other fish tested (96 h LC₅₀ range 500-14,200 µg/L). On EPA's qualitative scale, diuron is moderately toxic to fish.

The single exception to that is the LC₅₀ reported for the striped bass. In one EPA document it was reported as 400 µg/L (EPA 2009), and in another it was reported as 500 µg/L (EPA 2003). In the salmonid BE (EPA 2003) the 500 µg/L value is associated with a test of an 80% a.i. formulated product, and appears to have been conducted on larvae. Another reported data point for fingerlings is 6,000 µg/L also with the 80% a.i. formulated product, which is more consistent with other standardized test data. In the original source (Hughes 1973) the test material is described as an 80% a.i. product Karmex. The 96 h LC₅₀ for larvae is 500 µg/L, and for fingerlings it is 6,000 µg/L. Thus, it appears striped bass larvae are more sensitive than fingerlings, and this likely applies to all fish species.

Linuron

Less data were available for linuron. Mayer and Ellersieck (1986) reported one test with a 95% technical a.i. on channel catfish (96 h LC₅₀ 2,900 µg/L). Tests for rainbow trout (96 h LC₅₀ 3,000 µg/L, MRID 40445501) and bluegill (96 h LC₅₀ 9,600 µg/L, MRID 40354201) were reported in all three EPA documents ((EPA 1995), (EPA 2004), and (EPA 2008)) and summarized in Table 116. Complete references for these MRIDs were not provided in any of the documents, but we assume they are registrant-submitted guideline studies. On EPA's qualitative scale, linuron is moderately toxic to fish.

Table 116. Acute toxicity data for fish.

Species		96-h LC ₅₀ Median Concentration ¹ (µg/L)	Range (µg/L)	n	Source
<i>Diuron</i>					
Salmonids					
Cutthroat trout	<i>Oncorhynchus clarkii</i>	1,800 ¹	710-2,200	10	Mayer & Ellersieck 1986
Lake trout	<i>Salvelinus namaycush</i>	2,200 ¹	1,100-2,700	12	Mayer & Ellersieck 1986
Rainbow trout	<i>Oncorhynchus mykiss</i>	6,700 ¹	4,900-7,700	5	Mayer & Ellersieck 1986
Other fish					
Striped bass	<i>Morone saxatilis</i>	Larvae 500 Fingerlings 6,000 Karmex formulation (80% a.i.)	NR	NR	Hughes 1973
Bluegill	<i>Lepomis macrochirus</i>	8,500 ¹	2,800-10,400	12	Mayer & Ellersieck 1986
Fathead minnow	<i>Pimephales promelas</i>	14,200 ²	NA	1	Call et al 1987
<i>Linuron</i>					
Rainbow trout	<i>Oncorhynchus mykiss</i>	3,000	NR	NR	MRID 40445501 as cited in RLF
Channel catfish	<i>Ictalurus punctatus</i>	2,900	NA	1	Mayer & Ellersieck 1986
Bluegill sunfish	<i>Lepomis macrochirus</i>	9,600	NR	NR	MRID 40354201 as cited in RLF

¹ 96 hr test data on 95% technical only. Most data from static tests, although one flow-through test is included in cutthroat trout data set, and one in lake trout data set.

² 96 hr test endpoint calculated by authors

Reproduction and Growth

Reproduction and growth endpoints are typically evaluated in guideline tests that expose fish to the a.i. and then measure effect on a number of growth and reproductive parameters. Neither diuron nor linuron appeared to have these guideline studies, although other work appeared to have been conducted under contract to EPA, and is likely a guideline test or very similar protocol (Call, Brooke et al. 1983). We have not reviewed the original source of Call et al (1983), although we have reviewed the original source of a later publication which appears to be based on the same work (Call, Brooke et al. 1987). We have no specific explanation for the discrepancies between the two sets of values. In addition to the data reported in EPA documentation, we also located another work, which reports a reproductive effect not typically considered in the standard test (Gagnon and Rawson 2009), and is based on a shorter exposure duration at the egg stage (Table 117). The most sensitive endpoint and the shortest exposure is for the pink snapper (NOAEC 5 µg/l, LOAEC 50 µg/l). The range between the NOAEC and LOAEC in this test overlaps with the endpoints derived from the two Call et al (1983 and 1987) studies. The Nebeker and Schytéma (1998) work, which was also with fathead minnow, resulted in NOAECs that are two orders of magnitude higher than the other three tests (Nebeker and Schuytema 1998). We have no particular explanation for this discrepancy, other than perhaps the growth endpoints they measured are less sensitive than the reproduction endpoints measured in the other studies.

There appears to be virtually no chronic fish data available for linuron. The salmonid BE (EPA 2004) gives an NOAEC of <42 µg/L, and notes an additional test has been requested since effects were observed at the lowest concentration tested and a NOAEC could not be established. No MRID is given for the study. Neither the RED (EPA 1995) nor the California red-legged frog assessment (EPA 2008) give any data. The California red-legged frog assessment used an acute-to-chronic ratio (ACR) calculation. We did not locate any chronic data in the open literature.

Table 117. Chronic toxicity data for fish.

Species		NOAEC LOAEC (µg/L)	Endpoint Affected (Exposure duration)	Source
<i>Diuron</i>				
Pink snapper	<i>Pagrus auratus</i>	5 50	Increase in spinal deformities (36 h exposure in egg stage)	(Gagnon and Rawson 2009)
Fathead minnow	<i>Pimephales promelas</i>	26 62	Number of survivors (Not given)	MRID 00141636, as cited in RLF, Call et al 1983
Fathead minnow	<i>Pimephales promelas</i>	33.4 78.0	Increase in dead or abnormal fry post hatch (64 d exposure)	(Call, Brooke et al. 1987)
Fathead minnow	<i>Pimephales promelas</i>	<3,400 3,400	Decrease in juvenile growth (10 d exposure)	(Nebeker and Schuytema 1998)
Fathead minnow	<i>Pimephales promelas</i>	4,200 8,300	Decrease in embryo-larval growth (10 d exposure)	(Nebeker and Schuytema 1998)
<i>Linuron</i>				
No data given			Calculated as ACR for RLF	
Rainbow trout	<i>Oncorhynchus mykiss</i>	<42 42	Decreased growth (Not given)	As given in salmonid BE, no MRID referenced

Endocrine Disrupting Effects

Linuron has been identified by EPA as an endocrine disrupting compound, functioning as “competitive androgen receptor agonist ((EPA 2008), Appendix J). Diuron has not been specifically identified as an endocrine disruptor by EPA, although it is on the list of potential endocrine disrupting chemicals, and will be subject to further evaluation. Given structural similarities between the two a.i.s, and lack of available data regarding diuron, we believe it is reasonable to assume diuron acts in a fashion similar to linuron, acknowledging there may be differences in potency. Thus, in absence of data showing diuron not to be an androgen agonist, or to be a weaker androgen agonist than linuron, we have opted to use available data on linuron to evaluate endocrine effects for both chemicals.

Lambright et al (2000) examined the effects of linuron both *in vitro* and *in vivo*, and concluded that *in vitro* linuron binds both human and rat androgen receptors and *in vivo*, affects sexual differentiation in rats via this mechanism (Lambright, Ostby et al. 2000). It is difficult to extrapolate dietary data from rats to aquatic organisms, given differences in exposure routes (dietary versus gill uptake from water) and metabolism (homeotherms versus poikilotherms), but we did locate two studies evaluating androgenic effects of linuron in aquatic species. Jolly et al (2009) evaluated effects of linuron and several other contaminants both *in vitro* and *in vivo* using the three-spined stickleback (*Gasterosteus aculeatus*). Using production of an androgen-induced protein in kidney cells as a measure, they found concentration dependent effects in both *in vitro* and *in vivo* tests. *In vivo*, linuron caused a significant decrease in protein production at a water concentration of 100 µg/L after a 21-day exposure. Effects were not significant at 25µg/L (Jolly, Katsiadaki et al. 2009). Kashian and Dodson (2002) evaluated effects on the sex determination, fecundity, and growth rate for *D. magna* following a 6 d exposure. There were no discernable effects on any of these endpoints at the highest exposure concentration of 100 µg/L (Kashian and Dodson 2001).

Indirect Effects to Salmonids (Prey and Habitat Modifications)

Indirect effects on salmon include reductions in prey base (aquatic invertebrates), disruptions in primary productivity in the stream (phytoplankton and macrophytes), and effects on riparian vegetation.

Aquatic Invertebrates (Acute and Chronic Toxicity)

Survival - Diuron

Mayer and Ellersieck (1986) report acute EC₅₀ data on diuron for 5 separate species (Table 118). We located additional references that reported data on three more species ((Hernando, Ejerhoon et al. 2003),(Neuwoehner, Zilberman et al. 2010),(Nebeker and Schuytema 1998), and (Martins, Saker et al. 2007)). Although the dataset is weighted heavily towards cladocerans (5 of the 9 values), it does include other functional groups,

such as a stonefly, amphipods, and an isopod. EC₅₀ values range from 160-19,400 µg/L, with an overall median of 8,600 µg/L. Based on EPA's qualitative scale, diuron is highly toxic to moderately toxic to aquatic invertebrates.

Survival - Linuron

Based on available data, linuron appears to be more toxic to aquatic invertebrates than diuron (Table 118). Three data points for *D. magna* were available, ranging from 120 µg/L (MRID 0014932) to 1,900 µg/L (MRID 43996501). TKI contends the study producing the 120 µg/L values was flawed, and recommends the use of the newer 1,900 µg/L based on a better documented study, and better agreement with chronic data (MRID42153401). The median of the available values for *D. magna* is 270 µg/L. Mayer and Ellersieck (1986) also report an acute EC₅₀ for midge fly larvae (*Chironomid sp*) of 2,900 µg/L. Median value for an invertebrate data is 1,085 µg/L. Using EPA's qualitative scale, linuron is highly toxic to moderately toxic to aquatic invertebrates.

Table 118. Acute toxicity data for aquatic invertebrates.

Species		48 h ¹ or 96-h ² EC ₅₀ Median Concentration ¹ (µg/L)	Range (µg/L)	n	Source
<i>Diuron</i>					
Amphipod	<i>Gammarus fasciatus</i>	160 ²	NA	1	(Mayer and Ellersieck 1986)
Stonefly	<i>Pteronarcys californica</i>	1,200 ²	NA	1	(Mayer and Ellersieck 1986)
Waterflea (Cladoceran)	<i>Daphnia pulex</i>	1,400 ¹	NA	1	(Mayer and Ellersieck 1986)
Waterflea (Cladoceran)	<i>Simocephalus serrulatus</i>	2,000 ¹	NA	1	(Mayer and Ellersieck 1986)
Isopod	<i>Asellus brevicaudus</i>	15,500 ²	NA	1	(Mayer and Ellersieck 1986)
Waterflea (Cladoceran)	<i>Daphnia magna</i>	8,600 ¹	NA	1	(Hernando, Ejerhoon et al. 2003)
Waterflea (Cladoceran)	<i>Daphnia magna</i>	11,655	NA	1	(Neuwoehner, Zilberman et al. 2010)
Waterflea (Cladoceran)	<i>Daphnia pulex</i>	17,900	NA	1	(Nebeker and Schuytema 1998)
Amphipod	<i>Hyalella azteca</i>	19,400	NA	1	(Nebeker and Schuytema 1998)
Waterflea (Cladoceran)	<i>Daphnia magna</i>	3,000 (Reduced O ₂ consumption, 15 min exposure)	NA	1	(Martins, Saker et al. 2007)
<i>Linuron</i>					
Waterflea (Cladoceran)	<i>Daphnia magna</i>	120	NA	1	MRID 0014932 as cited in RLF
Waterflea (Cladoceran)	<i>Daphnia magna</i>	270	NA	1	(Mayer and Ellersieck 1986)
Waterflea (Cladoceran)	<i>Daphnia longispina</i>	360	NA	1	Stephenson & Kane 1984 as cited in (Cuppen, Van den Brink et al. 1997) NOT CONFIRMED, NOT IN MEDIAN CALCULATION
Waterflea (Cladoceran)	<i>Daphnia magna</i>	1,900	NA	1	MRID 43996501 as cited in Pyxis 2010
Midge larvae	<i>Chironomus sp</i>	2,900	NA	1	(Mayer and Ellersieck 1986)

Reproduction and Growth - Diuron

There are little chronic data for aquatic invertebrates for either diuron or linuron, although one study by Nebeker & Schuytema (1998) did evaluate the chronic effects of diuron on a range of aquatic species (Table 119). In their study (7-10 d exposure, depending on species) NOAECs ranged from 1,800-13,400 µg/L, with a median of 4,000 µg/L (Nebeker and Schuytema 1998). In some cases, they noted mortality at the LOAEC in addition to effects on reproduction and growth. A single study, cited in the salmonid BE as TN2418, gives an NOAEC of 200 µg/L for *D. magna* (EPA 2004). However, we were unable to find a complete reference for TN2418 in any of the EPA documents, so we cannot confirm this endpoint from the original source. The BE did not report endpoints affected or a LOAEC.

Reproduction and Growth - Linuron

For linuron there was one chronic study on *D. magna*, which gives the NOAEC as 130 µg/L and a LOAEC of 240 µg/L. This value was reported in both the salmonid BE and the RED (MRID 42153401, (EPA 1995), (EPA 2004)). Neither reports the endpoints affected. We have not reviewed the original source or DER. Given the fact that the current chronic endpoint is actually greater than the lowest available acute endpoint, EPA opted to calculate an ACR for the California red-legged frog assessment (EPA 2008). The current registrant, TKI, contends the study producing the 120 µg/L values was flawed, and recommends the use of the newer 1,900 µg/L based on a better documented study, and better agreement with the chronic data (MRID 42153401).

Table 119. Chronic toxicity data for aquatic invertebrates.

Species		NOAEC LOAEC (µg/L)	Endpoint Affected	Source
<i>Diuron</i>				
Waterflea (Cladoceran)	<i>Daphnia magna</i>	200 Not given	Not given	TN2418 as cited in RLF
Waterflea (Cladoceran)	<i>Daphnia pulex</i>	4,000 7,700	Mortality, reproduction (7 d exposure)	(Nebeker and Schuytema 1998)
Amphipod	<i>Hyalella azteca</i>	7,900 15,700	Mortality, reduced growth (10 d exposure)	(Nebeker and Schuytema 1998)
Midge larvae	<i>Chironomus tentans</i>	3,400 7,100	Mortality, growth (10 d exposure)	(Nebeker and Schuytema 1998)
Annelid worm	<i>Lubriculus varigatus</i>	1,800 3,500	Reduced growth (10 d exposure)	(Nebeker and Schuytema 1998)
Snail	<i>Physa gyrina</i>	13,400 22,800	Reduced growth (10 d exposure)	(Nebeker and Schuytema 1998)
<i>Linuron</i>				
Waterflea (Cladoceran)	<i>Daphnia magna</i>	130 240	Based on existing data, chronic effects appeared to have occurred at same concentrations as acute effects Calculated as ACR for RLF	MRID 42153401 as cited in RLF Also given in salmonid BE

Aquatic Plants (Phytoplankton and Vascular Plants)

Given that diuron and linuron are both herbicides, we anticipate the most sensitive receptors in salmon habitat will be plants. Table 120 summarizes toxicity data for aquatic plants exposed to diuron or linuron. Instream plants include various types of algal species and vascular plants. Generally the phytoplankton provide an energy source for the stream and the macrophytes are a structural component, providing attachment sites for other organisms and refugia for juvenile fishes. Reductions in primary productivity or modifications in community structure via removal of sensitive species can result in “bottom-up” trophic cascades which may adversely affect salmonids. Loss of structure provided by macrophytes may result in decreased population of aquatic invertebrates or increased predation on juvenile salmonids.

EPA documents for diuron reported EC₅₀s for a number of algal species and we located some additional data in the open literature. Within the source, these were grouped by generic classification (*e.g.*, green algae, diatoms) and if multiple values were available, we calculated median values using Excel. Based on the data available, all aquatic plants appear much more sensitive to diuron than either fish or invertebrates. EC₅₀s for green algae range from 2.4-51 µg/L. One test result for a red alga was reported. With an EC₅₀ of 24 µg/L, it falls within the range of the green algae. Diatoms appear only slightly more tolerant of diuron, with a range of EC₅₀s from 18-95 µg/L. Calculation of a single median value based on all phytoplankton tests resulted in a phytoplankton EC₅₀ of 28 µg/L. The one test we located for vascular plants gave a 7-day EC₅₀ of 15 µg/L for duckweed.

For linuron, we located only three species specific endpoints for plant species, all of which were reported in the California red-legged frog assessment. We have not been able to confirm these data from the original source. These data include one endpoint each for a green alga (67 µg/L), a diatom (13.7 µg/L), and a vascular plant (2.5µg/L). Median value for all plant data is 13.7 µg/L.

However, we did locate a body of work examining the effect of linuron in various types of microcosms ((Cuppen, Van den Brink et al. 1997), (Daam and Van den Brink 2007), (Daam, Rodrigues et al. 2009), (Daam, Van den Brink et al. 2009)). These studies evaluated a range of ecological effects including changes in functional parameters (*e.g.*, DO and pH), changes in phytoplankton communities, changes in macrophytes, changes in the zooplankton community, and sometimes changes in macroinvertebrates. Based on changes in these parameters, authors derived an NOAEC for ecosystem functioning. Separate studies, Cuppen et al (1997) and Daam and Van den Brink (2007), based on very similar experiments on indoor microcosms, concluded the ecosystem NOAEC for linuron was 0.5 µg/L ((Daam and Van den Brink 2007),(Cuppen, Van den Brink et al. 1997)). The most sensitive parameters were DO and pH. In both cases, LOAEC was 5 0

µg/L. Daam et al (2009) followed up on this concept with an outdoor microcosm experiment and a review of various linuron microcosm

Table 120. Toxicity data for aquatic plants.

Species		72 h ¹ or 96-h ² EC ₅₀ Median Concentration (µg/L)	Range (µg/L)	n	Source
<i>Diuron</i>					
Green algae	<i>Selenastrum capricornutum</i>	2.4 ²	NA	1	MRID 42218401, as cited in EPA RLF and RED, MRID reference not located
Green algae	<i>Psuedokirchneriella subcapitata</i>	EC ₅₄ =5 ³	NA	1	(Knauer, Sobek et al. 2007)
Green algae	<i>Scendesmus vacuolatus</i>	12 ⁴			(Neuwoehner, Zilberman et al. 2010)
Green algae	<i>Chlorella</i> sp <i>Chlorococcum</i> sp <i>Chlamydomonas</i> sp <i>Platymonas</i> sp <i>Neochloris</i> sp	19 ¹	10-28		MRID 40228401, 1986 as cited in EPA RLF appendix, reference not located
Green algae	<i>Psuedokirchneriella subcapitata</i>	51 ⁴	NA	1	(Neuwoehner, Zilberman et al. 2010)
Red algae	<i>Porphyridium cruentum</i>	24 ¹	NA	1	MRID 40228401, 1986 as cited in EPA RLF appendix, reference not located
Diatoms	<i>Nitzschia closterium</i> <i>Amphora exigua</i> <i>Stauronoides amphoroides</i> <i>Achnanthes brevipes</i> <i>Cyclotella nana</i> <i>Monochrysis lutheri</i> <i>Thalassiosira fluviatilis</i> <i>Navicula incerta</i>	35 ¹	18-95	8	MRID 40228401, 1986 as cited in EPA RLF appendix, reference not located
Duckweed	<i>Lemna perpusilla</i>	15 (7 day)	UNK	UNK	ECOTOX 8628 as cited in RLF
Ecosystem	Community metabolism	NOEAC 0.5 LOAEC 5.0	NA	NA	(Daam, Rodrigues et al. 2009)

Species		72 h ¹ or 96-h ² EC ₅₀ Median Concentration (µg/L)	Range (µg/L)	n	Source
<i>Linuron</i>					
Green algae	<i>Selenastrum capricornutum</i>	67	NA	1	MRID 42086801 as cited in RLF
Diatom	<i>Navicula pelliculosa</i>	13.7	NA	1	MRID 43992302 as cited in RLF
Western waterweed or Anacharis	<i>Elodea nuttali</i>	2.5	NA	1	ECOTOX 18629 as cited in RLF
Ecosystem	Community metabolism	NOEAC 0.3 LOAEC 3.0	NA	NA	(Daam, Rodrigues et al. 2009)

³ Author did not calculate EC₅₀, and data were only presented graphically ⁴ 24 h tests

experiments under different exposure regimes (single-peak, pulsed, and constant) (Daam, Rodrigues et al. 2009). Two of the single peak exposure studies produced higher NOAECs based on community metabolism (15 µg/L and <20 µg/L) but other studies corroborated the 0.5 µg/L NOAEC. They then applied a toxic unit (TU) approach suggested another body of work (Brock, Lahr et al. 2000) to calculate ecosystem threshold values for a range of other photosynthesis inhibitors, including diuron (Daam, Rodrigues et al. 2009). They concluded an ecosystem function threshold for diuron was 0.2 TU, which we calculated as 3 µg/L based on TU data for diuron presented in the paper.

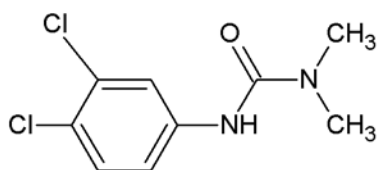
Diuron and Linuron Degradate Toxicity

Both diuron and linuron are relatively resistant to abiotic degradation (hydrolysis and photolysis) in the environment and are primarily broken down by microbial action in either soil or water ((EPA 1995), (EPA 2003), (Tixier, Sancelme et al. 2001)). Neither is broken down by hydrolysis, and when photolytic reactions occur, they primarily result in a hydroxyl group substitution for one of the chlorines on the phenyl ring. In contrast, the microbial-mediated reactions occur at the terminal nitrogen on the urea moiety.

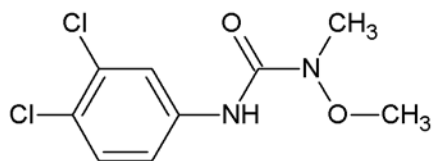
Although there are slight differences between the a.i.s as to how the degradation occurs, it generally follows the pathway of removal of the methyl and/or methoxy groups,

followed by eventual cleavage of the molecule at the carbonyl atom to produce 3,4-dichloroaniline (DCA) (Figure 70). Metabolic studies on mammals indicate a similar pathway occurs in vertebrates, with initial removal of the methyl groups attached to the urea, and eventually complete processing to DCA. In linuron, the methoxy group appears to be removed preferentially, followed by the methyl group. Metabolites are excreted primarily in urine ((EPA 1995), Geldmacher et al 1971, as cited in (Call, Brooke et al. 1987), (Menzie 1969), (Gosselin, Smith et al. 1984)). Call et al (1987) evaluated metabolism of diuron in rainbow trout and concluded metabolic products produced by the fish were DCA and two demethylated compounds, likely DCPMU and DCPU.

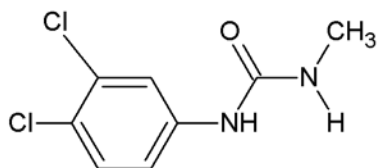
Thus, aquatic organisms may be exposed to the following degradates of diuron and linuron (Table 121, Figure 70).



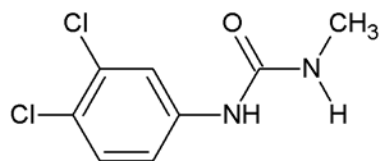
Diuron



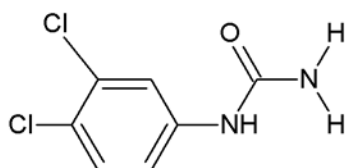
Linuron



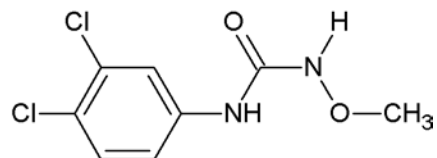
DCPMU



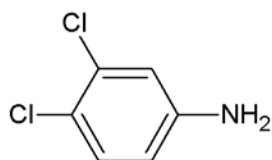
Desmethoxy linuron



DCPU



Desmethyl linuron



DCA (3,4 dichloroaniline)

Figure 70. Toxic degradates of diuron and linuron.

Table 121. Degradates of Diuron and Linuron.

Degradate	Chemical Name	Parent
DCPMU Desmethoxy-linuron	<i>1-(3,4,dichlorophenyl)-1-methylurea</i>	Diuron Linuron
DCPU Norlinuron	<i>3,4-dichlorophenylurea</i>	Diuron Linuron
DCA	<i>3,4-dichloroaniline</i>	Diuron Linuron
Desmethyl-linuron	<i>1-(3,4,dichlorophenyl)-1-methoxyurea</i>	Linuron

We did not locate a great deal of information on the toxicity of the degradates, although we did locate one highly relevant study that considered a number of the degradates, and their effects on both algae and aquatic invertebrates (*D. magna*)(Neuwoehner, Zilberman et al. 2010). Another study also did a toxicity analysis, but they used a bacterium, *Vibrio fischeri* in a Microtox® test (Tixier, Sancelme et al. 2001). We are uncertain how to extrapolate those results to our organisms of concern, thus we do not present those data.

The Neuwoehner et (2010) data is presented in Table 122. Although they considered the degradates of diuron, linuron also has several of the same degradates (Table 121), so we consider this information for both a.i.s. Based on their data, they concluded that loss of one or both methyl groups made the degradates less toxic to the algal species, and more toxic to *D. magna*. The final biotransformation product, DCA, was approximately 2 orders of magnitude more toxic to *D. magna* than parent diuron. However, total removal of the urea moiety⁴ resulted in it being approximately 3 orders of magnitude less toxic to the algal species. The less commonly occurring phototransformation products, hydroxyl group substitutions on the phenyl ring, were less toxic than the parent to both algal species and *D. magna*.

⁴ Moiety means molecular fragment, in this case urea $-(\text{NH}_2)_2\text{CO}$

Neither the linuron salmonid BE (EPA 2004) nor the linuron California red-legged frog assessment (EPA 2008) included any degradate toxicity data for any species. No degradate toxicity data for any species is presented in either of the diuron assessments ((EPA 2003), (EPA 2008)). The diuron salmonid assessments does note DCA as a degradate, but states it was not included in the risk analysis as it is anticipated to be less than 1% of applied a.i. (EPA 2003).

Table 122. Diuron degradate toxicity data for *D. magna* and algae.

Species		24 h ¹ or 48 h ² EC ₅₀ Concentration ¹ (µg/L)	Source
Diuron			
Waterflea	<i>Daphnia magna</i>	11,655 ²	(Neuwoehner, Zilberman et al. 2010)
Green algae	<i>Psuedokirchneriella subcapitata</i>	51 ¹	(Neuwoehner, Zilberman et al. 2010)
Green algae	<i>Scendesmus vacuolatus</i>	12 ¹	(Neuwoehner, Zilberman et al. 2010)
1-(3,4,dichlorophenyl)-1-methylurea			
Waterflea	<i>Daphnia magna</i>	10,734 ²	(Neuwoehner, Zilberman et al. 2010)
Green algae	<i>Psuedokirchneriella subcapitata</i>	70 ¹	(Neuwoehner, Zilberman et al. 2010)
Green algae	<i>Scendesmus vacuolatus</i>	15 ¹	(Neuwoehner, Zilberman et al. 2010)
3,4,dichlorophenylurea			
Waterflea	<i>Daphnia magna</i>	2,871 ²	(Neuwoehner, Zilberman et al. 2010)
Green algae	<i>Psuedokirchneriella subcapitata</i>	11,277 ¹	Neuwoehner et al 2010
Green algae	<i>Scendesmus vacuolatus</i>	3,286 ¹	(Neuwoehner, Zilberman et al. 2010)
3,4-dichloroaniline			
Waterflea	<i>Daphnia magna</i>	227 ²	(Neuwoehner, Zilberman et al. 2010)
Green algae	<i>Psuedokirchneriella subcapitata</i>	24,303 ¹	(Neuwoehner, Zilberman et al. 2010)
Green algae	<i>Scendesmus vacuolatus</i>	13,286 ¹	(Neuwoehner, Zilberman et al. 2010)
1-(3-chloro,4-hydroxyphenyl)-1,1-dimethylurea and 1-(3-hydroxy,4-chlorophenyl)-1,1-dimethylurea (phototransformation products)			

Species		24 h ¹ or 48 h ² EC ₅₀ Concentration ¹ (µg/L)	Source
Waterflea	<i>Daphnia magna</i>	77,474 ²	(Neuwoehner, Zilberman et al. 2010)
Green algae	<i>Psuedokirchneriella subcapitata</i>	278 ¹	(Neuwoehner, Zilberman et al. 2010)
Green algae	<i>Scendesmus vacuolatus</i>	103 ¹	(Neuwoehner, Zilberman et al. 2010)

Microbial Community Effects (Sediment, Soil, and Water Column)

Given the nature of diuron and linuron and their specificity for photosystem II, we do not anticipate adverse effects on the microbial community. We did locate one study (Mukherjee, Alam et al. 2006) evaluating the effects of linuron on soil microorganisms in a cropped field. The authors concluded application of linuron did not affect soil microfauna.

Riparian Vegetation

Riparian vegetation is important for providing shade to the stream, stabilizing the stream banks, reducing sedimentation, and providing allochthonous input, both in terms of plant material and terrestrial insects. Generally there are not good data regarding the effects of herbicides on wild plants, other than weed species, but EPA requires submission of crop effects data as part of the registration process. We believe this provides a reasonable basis for evaluating effects on herbs and forbs, and based on the mode of action for diuron and linuron, expect that woody shrubs and trees are likely to be more resistant. Guideline studies determine EC₂₅s of end-use products on the endpoints of vegetative vigor and seedling emergence. We calculated medians for the range of crop data presented the EPA assessments, which is presented in Table 123.

Table 123. Terrestrial plant data.

Plants	Test	EC ₂₅ Median (lb ai/A)	Range (lb ai/A)	n	Source
<i>Diuron</i>					
Rapeseed Sorghum Soybean Sugarbeet Tomato Wheat	Vegetative vigor	0.0164	0.00017-0.0753	6	EPA RLF 2009 (Appendix L)
Sorghum Sugarbeet Tomato	Seedling emergence	0.092 ¹		3	EPA RLF 2009 (Appendix L)
<i>Linuron</i>					
No guideline study data available					EPA RLF 2008

¹ Some studies had indeterminate endpoints, which were not included in the calculation.

Summary of Toxicity Data for Diuron and Linuron

Assessment endpoints and associated concentrations are summarized in Table 124 for diuron and in Table 125 for linuron.

Table 124. Assessment Endpoints and Measures for Diuron.

Assessment Endpoint		Assessment Measure	Median Concentration ¹ (µg/L)	Range	n
Direct Effects on Salmonids	Survival	Salmonid Acute LC ₅₀ ² Other Fish Acute LC ₅₀ ²	2,325 10,450	710-23,800 6,700-14,200	8 2
	Growth	Chronic NOAEC ² Chronic LOAEC ²	No data available		
	Reproduction	Chronic NOAEC ² Chronic LOAEC ²	30 70	5-4,200 62-8,200	4
	Swimming	We located no data regarding effects of diuron on swimming.			
	Olfaction	Tierney 2007 no effect 1mg/L (awaiting paper)			
	Endocrine Disruption	Androgen Protein NOAEC Androgen Protein LOAEC (Jolly et 2009)	25 100	NA	1
Effects on Prey (Aquatic Invertebrates)	Survival	Acute Invert EC ₅₀ ²	8,600	160-19,400	9
	Growth	Chronic NOAEC ² Chronic LOAEC ²	5,650 11,400	3,400-13,400 3,500-22,800	4
	Reproduction	Chronic NOAEC ² Chronic LOAEC ²	4,000 7,700	NA	1
Effects on Primary Productivity, Submerged and Emergent Vegetation	Biomass & Abundance	Algal and Vascular Plant EC ₅₀ ²	28	10-95	17
Effects on Ecosystem Functioning	Community Metabolism	Community NOAEC Community LOAEC Damm et al 2009a	0.5 5.0	NA	1
Effects on Riparian Vegetation	Biomass & Abundance	Vegetative Vigor	0.0164 lb ai/A	0.00017- 0.0753lb ai/A	6

¹If more than one value was available. If only one value was available, the actual number is given.

NA Not applicable, only one value available

Table 125. Assessment Endpoints and Measures for Linuron.

Assessment Endpoint		Assessment Measure Check footnotes!	Median Concentration ¹ (µg/L)	Range	n
Direct Effects on Salmonids	Survival	Salmonid Acute LC ₅₀ ² Other Fish Acute LC ₅₀ ²	3,000 6,250	NA 2,400-9,600	1 2
	Growth	Chronic NOAEC ² Chronic LOAEC ²	<42 42	NA	1
	Reproduction	Chronic NOAEC ² Chronic LOAEC ²	No data available		
	Swimming	We located no data regarding effects of linuron on swimming.			
	Olfaction	Tierney 2007 no effect 1mg/L (awaiting paper)			
	Endocrine Disruption	Androgen Protein NOAEC Androgen Protein LOAEC (Jolly et 2009)	25 100	NA	1
Effects on Prey (Aquatic Invertebrates)	Survival	Acute Invert EC ₅₀ ²	1,085	120-2,900	4
	Growth	Chronic NOAEC ² Chronic LOAEC ²	No data available		
	Reproduction	Chronic NOAEC ² Chronic LOAEC ²	No data available		
Effects on Primary Productivity, Submerged and Emergent Vegetation	Biomass & Abundance	Algal and Vascular Plant EC ₅₀ ²	13.7	2.5-67	3
Effects on Ecosystem Functioning	Community Metabolism	Community NOAEC Community LOAEC Damm et al 2009a	0.3 3.0	NA	1
Effects on Riparian Vegetation	Biomass & Abundance	Vegetative Vigor	No data available		

¹If more than one value was available. If only one value was available, the actual number is given.

NA Not applicable, only one value available

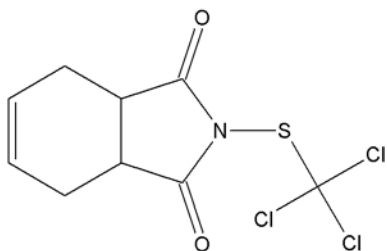
Fungicides (Captan and Chlorothalonil)

Captan Mode of Action

Captan is a non systemic, phthalimide fungicide which inhibits the normal cell division of microorganisms and fungi ((EPA 2007), (Cremlyn 1991), (Krieger and Krieger 2001)).

The parent captan molecule (Figure 71) cleaves at the bond between the nitrogen and sulfur atoms, and the trichloromethylthio (TCMT) reacts with thiol groups in various

enzymes and cellular structures (Figure 72). The reaction of TCMT with the thiol groups releases thiophosgene, which also reacts with thiol groups ((Cremlyn 1991), (Lukens 1966), (Krieger and Krieger 2001), (Bernard and Gordon 2000)).



Captan

Figure 71. Structure of Captan



Where R' is a thiol group

Figure 72. Captan Mode of Action

There are two other phthalimide fungicides with similar chemical structures, folpet and captafol (Figure 73). U.S. registrations of captafol were cancelled during re-registration (http://www.epa.gov/pesticides/reregistration/status_page_c.htm, Case #0116). Folpet is still an active registration, but appears to be registered in the U.S. only for use on avocados (EPA 1999). At the time the RED was written, the avocado use appeared to be limited to Florida (EPA 1999). Although we searched the NPRS and PPLS systems (11/29/10 pdd), we were unable to locate a label for folpet to confirm this information. Folpan Folpet Technical (Reg. No. 11678-18) is still listed as registered, with a label date of 08/02/04.

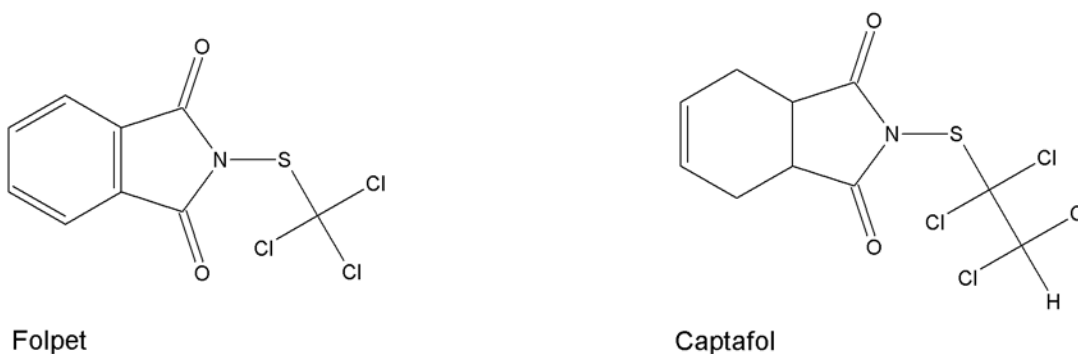


Figure 73. Structures of Folpet (left) and Captafol (right)

Bernard and Gordon (2000) evaluated both captan and folpet, concluding they had a common mechanism of action (Bernard and Gordon 2000). Although they did not consider captafol, we believe it is reasonable to assume it also has the same mechanism, given structural similarities. Thus, in cases where folpet may be used in salmonid habitat, we assume additivity for captan and folpet. We also believe it is reasonable to consider data and information regarding the toxicological effects of both folpet and captafol in this assessment, while recognizing that the severity of effects will vary dependent on the specific a.i.

Parent captan produces two primary degradates, TCMT (the toxic moiety), and tetrahydrophthalimide (THPI), the phthalimide ring structure. Data presented by EPA in various assessments show THPI to be orders of magnitude less toxic than parent captan for all taxa, which is not unexpected given the loss of the thiochloromethyl group. We do not consider THPI further in this assessment, although we have noted toxicity data provided by EPA. However, the EPA documents ((EPA 2003), (EPA 2004), (EPA 2007)) do not provide toxicity data or environmental half-lives for either TCMT or thiophosgene, other than noting:

“TCMT moiety degrades moderately rapidly to rapidly by aerobic soil metabolism to CO₂, thiophosgen, and organic sulfur and chlorine. Thiophosgene dissipation is expected to be dependent on volatilization (est. vapor pressure 29.7 mm Hg and estimated Henry’s Law Constant 0.00586 atm-m₃mole⁻¹).” (pg 38, (EPA 2004)).

We presume TCMT to be as toxic as parent captan, given it mediates the reaction but are unable to determine how long it may be present in the aquatic environment compared to the parent given the information provided. Thiophosgene is generally recognized to be toxic, primarily as a gas. We located no aquatic toxicity data for TCMT or thiophosgene.

Temperature and toxicity

We located no information indicating temperature specifically affects the toxicity of captan. However, we do note higher water temperatures can affect salmonids in two ways, regardless of specific chemical effects. Higher water temperatures will increase the metabolic rate for fish, thus increasing the rate at which they process the toxicant. Depending on the chemical, this may be either beneficial or detrimental. Water temperatures higher than optimum increase general physiological stress for salmonids, making them more susceptible to other stressors.

pH and toxicity

We located no information indicating pH specifically affects the toxicity of captan, although the pH does affect the hydrolysis rate. The hydrolysis rates increases with increasing pH.

Toxicity of Captan (Assessment Endpoints)

Although captan has been registered and used in commerce since 1951, it does not appear to have been heavily researched from a wildlife perspective, and we did not locate a large number of studies addressing aquatic effects. Both captan and folpet have been identified as probable human carcinogens (EPA 2004), although the captan classification was reduced to “not likely” in 2004 (EPA 2004). The metabolite and degradate thiophosgene has been identified as a probable common mechanism of action for the two a.i.s. There is a substantial body of literature evaluating carcinogenic and mutagenic effects of folpet and captan in small mammals, thus we find it reasonable to believe similar effects could occur in aquatic organisms.

Direct Effects to Salmonids

Survival

Twenty acute LC₅₀ values for fish were presented in the RLF assessment (EPA 2007). This was the most complete set of data, and included the information presented in the salmonid BE (EPA 2003) and the captan RED (EPA 2004). Data tables detailing each end point are located in *Appendix 7*.

We divided the data into salmonids and other fish species. The salmonids group (n=11) includes 5 data points from species in the genus *Onchorhynchus*, the same genus as all of the listed salmonids addressed in this Opinion. Another 5 data points are from fish in the genus *Salvelinus*, and 1 is from a fish in the genus *Salmo*. Most are standard 96 h tests, although some were conducted in static systems, and others were conducted in flow-through systems. For the same species tested in both static and flow-through systems, the LC_{50s} from the flow-through tests are generally lower. Most data are compiled from two studies, which are compendiums of aquatic toxicity data rather than registrant-submitted studies ((Johnson and Finley 1980), (Mayer and Ellersieck 1986)). No sublethal effects are described in any of the EPA documents. For salmonids, the range of LC_{50s}⁵ was 26.2-137 µg/L, and the median was 56.5 µg/L. Under the qualitative toxicity classification scale used by EPA, captan is very highly toxic to salmonids on an acute basis. No probit slope data was available for any of these tests.

LC₅₀ values were available for other fish (n=9). Three were for the bluegill sunfish (genus *Lepomis*), three were for fathead minnow (genus *Pimophelus*), and there was one each for channel catfish (*Ictalurus punctatus*), yellow perch (*Perca flavescens*), and the harlequin fish (*Rasbora heteromorpha*). For these fish the range of LC_{50s}⁶ was 65-310 µg/L, and the median was 134 µg/L. Under the qualitative toxicity classification scale

⁵ Based on LC50 value, does not include 95% confidence interval.

⁶ Based on LC50 value, does not include 95% confidence interval.

used by EPA, captan is highly toxic to very highly toxic to these fish on an acute basis. One test for bluegill sunfish resulted in a probit slope of 1.17.

Standard 96 hr LC₅₀ endpoints were available for the degradates THPI and THPAm, tested with rainbow trout (*Onchorhynchus mykiss*). Both endpoints were non-determinate⁷. The LC₅₀ for THPI is >120,000 µg/L and the LC₅₀ for THPAm is >126,000 µg/L. Under the qualitative toxicity classification scale used by EPA, these degradates of captan are practically non-toxic to fish on an acute basis. We do not carry them through the remainder of the analysis.

No toxicity data for formulations was presented. In the toxicity to freshwater fish section, the RED notes:

“The Agency waived the acute formulated product testing with a 50% wettable powder (WP) formulation since the confidential statement of composition for captan 50WP and 80WP end use products showed that the major and minor inert are not likely to enhance the toxicity of captan (EPA 2004).”

Reproduction and Growth

Reproduction and growth endpoints were evaluated in an fish early-life stage toxicity test. Although not specified in the EPA documents, the recommended duration for fathead minnow, the test species used, is 28 d post-hatch (guideline OPPTS 850.1400). In this test, effects noted included reductions in adult and larval survival, growth, and overall larval-juvenile development. Other effects included a reduction in the number of eggs laid, and an inability for the juveniles to reproduce. The NOAEC established by this test was 16.5 µg/L and the LOAEC was 39.5 µg/L (MRID 0057846). There was no additional discussion in any of the EPA documents regarding reproduction or growth endpoints for fish.

⁷ Only a > value is available, not a specific endpoint.

Carcinogenicity and Mutagenicity

EPA routinely requires carcinogenicity studies in rats and mice to support their human health assessments. There is no correlative assessment of cancer or mutagenicity in wildlife species, although some of the endpoints evaluated in the chronic wildlife assessments might arguably be considered to be expressions of such cellular-level changes (*e.g.*, failure to hatch, or reduced viability of young). In the small mammal studies, captan caused intestinal tumors in mice, and caused renal neoplasms and uterine sarcomas in rats (EPA 2004). This was attributed primarily to the thiophosgene metabolite.

One study reviewed evaluated genotoxic effects of captan on the South African clawed frog (*Xenopus laevis*) and the Spanish newt (*Pleurodeles waltl*) (Mouchet, Gauchet et al. 2006). The larval organisms were exposed to captan in water, so the exposure pathway is more relevant to salmonids than dietary tests with mammals would be. Physiologically, the amphibians are also more similar to fish. Genotoxicity was evaluated via the micronucleus test (MNT) and the comet test (CA), both of which have been used previously to demonstrate DNA damage in both fish and amphibians exposed to xenobiotics (Mouchet, Gauchet et al. 2006). Captan concentrations tested ranged from 15.6 – 2,000 µg/L in both a mineral water (MW, low ionic strength) and reconstituted water (RW, with minerals added, higher ionic strength). The mineral water is similar to water used in most OPPTS guideline tests. Concentrations provided are nominal. Exposure duration was 12 days.

At captan concentrations of ≥ 500 µg/L, mortality of both test species in both MW and RW solutions was 50% or higher. The EPA reviewer calculated LC_{50s} based on data in the paper (EPA 2007). LC_{50s} were 119.4 µg/L(MW) and 353.6 µg/L (RW) for the frog, and 311.1 µg/L (MW) and 500 µg/L (RW) for the newt. These ranges are similar to fish LC_{50s}.

Cardial hepatocytes from the larvae exposed to 15.6 µg/L, 31.25 µg/L, 62.5 µg/L and 125 µg/L of captan in MW were sampled at 1, 2, 4, 8, and 12 days and assayed using MNT and CA (Mouchet, Gauchet et al. 2006). MNT demonstrated a genotoxic response in the frog at 62.5 µg/L, but at not the lower concentrations. Based on the MNT there was no response at any concentration for the newt. MNT detects cytogenetic damage. For the frog, CA showed genotoxic effects at concentrations ≥ 15.6 µg/L. For the newt, CA showed genotoxic effects at 62.5 µg/L and 125µg/L. Effects appeared as early as 1 day of exposure, although extent of effects varied through the course of the experiment. CA indicates primary DNA damage, and authors note that some DNA lesions could “give rise to fixed mutations.”

Based on this study, we conclude that even short-term exposure to captan in aquatic systems may cause cellular and/or DNA damage to salmonids. We note that some of this damage may be repaired by normal cellular damage repair mechanisms, but also that there is potential for some of these changes to cause heritable mutations. We also recognize effects may occur at different (higher or lower) concentrations given phylogenetic differences between the test organisms and salmon

Indirect Effects to Salmonids (Prey and Habitat Modifications)

Aquatic Invertebrates (Acute and Chronic Toxicity)

EPA documents provided standard toxicity test results (both acute and chronic) for *Daphnia magna*, a water flea. *D. magna* is a standard test organism, and for many chemicals, is one of the more sensitive aquatic invertebrates tested. However, when tests are available for high-quality salmonid prey like mayflies or stoneflies, they are frequently more sensitive than *D. magna*. Based on the guideline tests, the EC₅₀ for *D. magna* is 8,400 µg/L (95% CI, 7,060-9,960 µg/L, slope 1.19) (MRID GS0120041, (EPA 2007)). MRID GS0120041 is a 1979 study conducted by the Laboratory of Terrestrial and Aquatic Biology in Beltsville, MD, and may not have been conducted in accordance with current guideline test protocols. Based on these test results, captan is considered

moderately toxic to aquatic invertebrates on an acute basis using EPA's qualitative scale. The salmonid BE (EPA 2003) and the RED (EPA 2004), report some additional tests for *D. magna*. Two of these tests did not determine the EC₅₀, as it was greater than the concentrations tested. A third reported a 1,300 µg/L EC₅₀ (MRID 00002875). It was not reported in the RLF assessment, presumably because the test duration was only 26 h (EPA 2004).

Effects noted in the chronic test included reductions in parental and juvenile growth, decreased survival for both parents and juveniles, and a reduction in neonates produced. Based on this test, the NOAEC for *D. magna* is 560 µg/L and the LOAEC is 1,000 µg/L (MRID 44148801, (EPA 2007)). The assessment notes risk may be underestimated based on these values, because concentrations used were nominal, not measured and the test material was reported as "unstable in water". NMFS concurs with this evaluation. The assessment also provided an EC₅₀ of > 113,000 µg/L for THPI with *D. magna* as the test organism.

Aquatic Plants (Phytoplankton and Vascular Plants)

Standard toxicity test data for 3 species of phytoplankton and one species of aquatic vascular plants were provided in the most recent EPA assessment (EPA 2007). The RLF document does not report specifically which endpoints were affected in the plant tests, nor does the RED. Aquatic plant tests are not reported in the salmonid BE. The 3 phytoplankton species tested were *Scenedesmus subspicatus* (a green alga), *Selenastrum capricornutum* (a green alga), and *Anabaena flos-aque* (a blue-green alga). EC₅₀s for these phytoplankton were, 320 µg/L, 1,770 µg/L, and 1,200 µg/L, respectively,. All tests were conducted with ≥90% a.i. The vascular plant tested was *Lemna gibba* (duckweed), and EC₅₀ for this test was >12,700 µg/L. The RLF document notes the data for *Scenedesmus subspicatus* may underestimate risk, as concentrations were nominal, not measured, and the EC₅₀ may in fact be lower. *Selenastrum capricornutum* was also tested with the degradate THPI, and the EC₅₀ was >180,000 µg/L. Based on a statement

in the RLF assessment (Appendix A, footnote, (EPA 2007)) marine/estuarine algal species appear to have been tested (MRID 40228401) but that information was not provided in any of the documents.

NMFS has not located any additional literature documenting aquatic plant endpoints.

Microbial Community Effects (Sediment, Soil, and Water Column)

Although EPA does not typically evaluate the effects of registered a.i.s on microbial processes in water, soil, or sediment, NMFS believes these are a relevant endpoint to consider for fungicides, which are targeted at organisms such as mycorrhizae, which serve important functions in soil.

We located one study evaluating the effects of captan on microbial community respiration, microbial community biomass, bacterial activity, and denitrification (Widenfalk, Svensson et al. 2004). The experiments were conducted in microcosms. Captan was tested at 0.013 µg/kg dwt sed, 1.3 µg/kg dwt sed, and 130 µg/kg dwt sed. Test concentrations were derived from a Maximum Permissible Concentration (MPC) standard of 0.11 µg/L (Crommentuijn, Sijm et al. 2000) using equilibrium partitioning theory. Of the 4 parameters tested, captan affected only bacterial activity. The response was not dose-dependent, as a significant difference from controls was only noted for the 1.3 µg/kg dwt sed test concentration. This particular test evaluated only an 8 h interval, although some of the other parameters were tested for longer periods of time. Based on the data provided, it cannot be determined how long the effects might last past the test period. We note that given the reducing nature and presence of sulfur compounds in most natural sediment, it seems unlikely that captan's sulfur-containing degradates would remain active for very long.

We located no studies evaluating effects of captan on the water column microbial community

Riparian Vegetation

No terrestrial plant guideline data are available to evaluate effects on terrestrial plants, as EPA waived those studies (EPA 2004). In the RLF assessment, EPA notes a literature search resulted in some data regarding captan use as a seed treatment on germination of the test species. We do not believe this type of application is particularly relevant for riparian vegetation, and do not repeat it here. They also noted most papers they located describing effects of captan used as a foliar spray did not describe “any lasting phytotoxic effects” on the plants. One study did have an lb a.i./A rate associated with temporary phytotoxic effects. This study, on highbush blueberries, found an application of captan end-use products (Captan 80WP and Captec 4L) at a rate of 2.4 lb a.i./A, caused mild phytotoxicity, although bushes recovered by harvest (EPA 2007). Assessment endpoints and associated concentrations for captan are summarized in Table 126.

Table 126. Assessment Endpoints and Measures for Captan

Assessment Endpoint		Assessment Measure	Median Concentration ¹ (µg/L)	Range	n
Direct Effects on Salmonids	Survival	Salmonid Acute LC ₅₀ ² Other Fish Acute LC ₅₀ ²	56.5 134	26.2-137 65-310	11 9
	Growth	Chronic NOAEC ² Chronic LOAEC ²	16.5 39.5	NA	1
	Reproduction	Chronic NOAEC ² Chronic LOAEC ²	16.5 39.5	NA	1
	Swimming	We located no data regarding effects of captan on swimming or olfaction.			
	Olfaction				
	Genotoxicity	DNA Change NOAEC DNA Change LOAEC (Mouchet et al 2006) ³	23.4 46.9	15.6-31.3 31.3-46.9	2
Effects on Prey (Aquatic Invertebrates)	Survival	Acute Invert EC ₅₀ ²	8,400	NA	1
	Growth	Chronic NOAEC ² Chronic LOAEC ²	560 1,000	NA	1
	Reproduction	Chronic NOAEC ² Chronic LOAEC ²	560 1,000	NA	1
	Abundance (?)	Meso/microcosm Data, if available and applicable	None located as of 12/6/10		
Effects on Primary Productivity	Biomass & Abundance	Algal EC ₅₀ ²	1,200	320-1,770	3

Assessment Endpoint		Assessment Measure	Median Concentration ¹ (µg/L)	Range	n
Effects on Submerged and Emergent Vegetation	Biomass & Abundance	Vascular Plant EC ₅₀ ²	>12,700	NA	1
Effects on Riparian Vegetation	Biomass & Abundance	Terrestrial Plant Phytotoxicity (Polavarapu 2000)	2.4 lb a.i./A	NA	NA

¹If more than one value was available. If only one value was available, the actual number is given.

²From guideline/standardized test data presented in EPA documents (primary source most recent BE on RLF)

³Data from amphibian species, water exposure, NOAEC and LOAEC based on 1 d exposure
NA Not applicable, only one value available

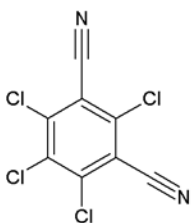
Chlorothalonil Mode of Action

Chlorothalonil is classed as an aromatic fungicide. There are a number of other chemicals in this class, including some that also have chloro- and/or nitro- sidechains on the benzene ring, but chlorothalonil is unique in having the triple-bonded nitrogen atoms on the benzene (Figure 74).

Chlorothalonil interacts with sulfhydryls via substitution of the (R-S-) group at the chlorine atom in the 4 or 6 position on the ring (Figure 75) (Long and Siegel 1975). “The fungitoxic action of chlorothalonil has been attributed to its ability to deplete cellular glutathione reserves (Tillman et al, 1973 as cited in (Davies 1985)).” Following depletion of those reserves, it can then interact with sulfhydryl groups in other proteins or enzymes, interfering with cellular processes. In fish, Davies and White (1985) found chlorothalonil interacted with glutathione in a range of tissues, including internal organs, gills, muscle and blood, but that the primary site of accumulation was in the gall bladder and bile (Davies and White 1985).

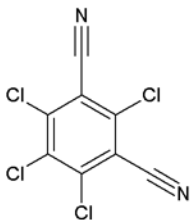
Glutathione, a tripeptide consisting of glutamic acid, cysteine, and glycine, is present at high concentrations in all animal tissues ((Lehninger 1975), pg 795). It is involved in the amino acid transport cycle ((Lehninger 1975), pg 795), and is also involved in phase II

xenobiotic transformation in the liver ((Parkinson 1996), pg 177). Glutathione protects cells by binding electrophiles, and reducing the effects of oxidizing agents on other cellular components ((Parkinson 1996), pg 189). Glutathione conjugations may occur enzymatically (catalyzed by glutathione-*S*-transferase, or nonenzymatically ((Parkinson 1996), pg 177). Glutathione-*S*-transferase is inducible ((Parkinson 1996), pg 177, (Davies 1985)). It was noted there was induction of hepatic glutathione and glutathione-*S*-transferase at sublethal concentrations (10 µg/L) in rainbow trout, but not at lethal (30 µg/L) concentrations (Davies 1985). Dröge and Breitkreutz (2000) discuss the importance of cellular glutathione in immune response, noting that levels too low or too high are associated with autoimmune diseases like human immunodeficiency virus (HIV)(Dröge and Breitkreutz 2000). Also, it was clearly shown that chlorothalonil reacts first with cellular glutathione, and then reacts with glyceraldehyde 3-phosphate dehydrogenase (GADPH) (Tillman, Siegel et al. 1973). GAPDH is an important component of the glycolysis cycle, and also protects cells against oxidative stress. Thus chlorothalonil's depletion of glutathione and GAPDH contributes to suppression of the organism's immune response.

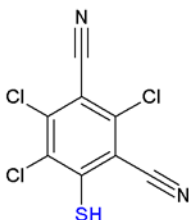


Chlorothalonil

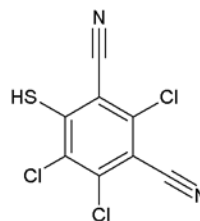
Figure 74. Structure of Chlorothalonil



Chlorothalonil



Substituted at 4 position



Substituted at 6 position

Figure 75. Chlorothalonil substitutions at 4 and 6 position (-SH represents any -S R group)

Comparison of Captan and Chlorothalonil Modes of Action

Long and Siegal (1975) compare the modes of action for chlorothalonil (isophthalonitrile) with that of captan and folpet (trichloromethyl sulfenyls) and note in their discussion (p 392-393):

“Both the isophthalonitrile and trichloromethyl sulfenyl fungicides are considered to act as general or non-specific toxicants. These fungicides have certain other similar reaction characteristics. They react with low and high molecular weight cellular thiols with toxicity ultimately residing with the inactivation of thiol containing proteins. . . There are, however, major differences in reactivity between these groups of fungicides. They differ in their rate of reaction, complexity of reaction, and the type of binding sites involved in the reaction. . . While the mode of action of chlorothalonil resembles that of the trichloromethyl sulfenyl fungicides in certain general characteristics, pronounced differences can be noted in their chemical mechanism of reaction with cellular constituents (Long and Siegal 1975).”

Based on their analysis, although captan and chlorothalonil both interact with sulfhydryl groups, we believe it is inappropriate to consider it additivity. We do note, however, that based on mode of action for both of these a.i.s, it is likely organisms exposed to one or both will be more susceptible to some types of other toxicants, especially those which cause oxidizing reactions.

Temperature and toxicity

Davies and White (1985) varied the water temperature from 10°C -16°C in the studies they performed on rainbow trout. In these experiments, LC_{50s} did decrease as temperature increased (Davies and White 1985). Three trials at 10°C, 14°C, and 16°C

produced 96-hr LC_{50s} of 18.0 µg/L, 17.1 µg/L, and 10.5 µg/L, respectively. However, the test conducted at 16°C also had lower DO. Additionally, two of the tests (at 14°C and 16°C) were conducted as flow-through, and the 10°C test was static, which can cause variability in results between the tests. No measure of variability (95% CI) was provided for the LC_{50s} in the paper, so we cannot determine if these temperature-toxicity differences are significant.

We also note higher water temperatures can affect salmonids in two ways, regardless of specific chemical effects. Higher water temperatures will increase the metabolic rate for fish, thus increasing the rate at which they process the toxicant. Depending on the chemical, this may be either beneficial or detrimental. Water temperatures higher than optimum increase general physiological stress for salmonids, making them more susceptible to other stressors.

pH and toxicity

We located no information indicating pH specifically affects the toxicity of chlorothalonil.

Toxicity of Chlorothalonil (Assessment Endpoints)

A number of LC₅₀ values for a range of fish species were available for chlorothalonil. Additionally, studies by Davies ((Davies 1985; Davies 1985; Davies 1985) and Davies and White (Davies and White 1985) provide significant insight into the biochemistry and metabolism of chlorothalonil in fish including rainbow trout. Other researchers (Baier-Anderson and Anderson 1998) have also investigated effects at a cellular level. Tierney (Tierney, Ross et al. 2006), considered the effects of chlorothalonil on salmon olfaction.

Direct Effects to Salmonids

Survival

In Davies and White (Davies and White 1985), authors made behavioral observations in addition to determining LC_{50s}. Dependent on temperature and dissolve oxygen (DO)

concentrations, 96 hr LC_{50s} for *Onchorhynchus mykiss* (referred to by its older latin name of *Salmo gairdneri*), ranged from 10.5 µg/L to 18.0 µg/L. All temperature and DO variations were in acceptable (although not optimal) ranges for salmonids. They noted all fish were lethargic when exposed to chlorothalonil. At concentrations below 8.7 µg/L *O. mykiss* exhibited normal startle reactions. However, at higher concentrations they observed the following.

“In *S. gairdneri*, loss of startle reaction was followed by a reduction of activity and station holding at the surface or bottom. A gradual bronzing of skin colour occurred especially in facial patches and around the lateral-line. Reddening at fin bases was also observed. Dead fish were a pronounced bronze colour. No fin collapse occurred [although it did for *Galaxias* species tested].”

To develop the LC₅₀ median and range for fish we took the data from the EPA RLF assessment (EPA 2007), which contains the most recent comprehensive analysis of aquatic chlorothalonil toxicity data, and cross-referenced it with the salmonid BE (EPA 2003) and the review document provided by the applicant (Brain, Perkins et al. 2010).

There was a range of test durations, beginning at 24 h and extending to 10 d. In several cases, data sets from individual studies included measurements at multiple time intervals. In most cases, longer exposure times produced lower LC_{50s}. The 10 d LC_{50s} were indeterminate, so it is difficult to tell precisely how they relate to the rest of the tests, which are typically 24-96 h. To ensure comparability across the data, we used the 96 h tests to calculate median and range for salmonids and other fish.

We divided the data into salmonids and other fish species. The salmonids group (n=5) includes only the genus *Oncorhynchus*, the same genus as all of the listed salmonids addressed in this Opinion. All are for *Oncorhynchus mykiss*, the rainbow trout. For salmonids, the range of LC_{50s} was 18-42 µg/L, and the median was 18 µg/L. Under the

qualitative toxicity classification scale used by EPA, chlorothalonil is very highly toxic to salmonids on an acute basis. No probit slope data was available for any of these tests.

LC₅₀ values were available for other fish (n=9). Three were for fish of the genus *Galaxias*, a trout species common in Australia and New Zealand. Although not closely related to the Pacific salmonids considered in this opinion, it is a cool-water fish, and occupies a similar ecological niche. Some of the rainbow trout data is from the same study (Davies and White 1985). For *Galaxias* species, the range of LC_{50s} was 16.3-29.2 µg/L, and the median was 18.9 µg/L, indicating they are similar in sensitivity to salmonids in regards to chlorothalonil.

Additionally, there were three 96 h LC_{50s} for the bluegill sunfish (genus *Lepomis*), and one each for fathead minnow (*Pimophyes promelas*), channel catfish (*Ictalurus punctatus*), and the threespine stickleback (*Gasterosteus aculeatus*). For these fish the range of LC_{50s} was 23-84 µg/L, and the median was 55.5 µg/L. Under the qualitative toxicity classification scale used by EPA, captan is highly toxic to very highly toxic to these fish on an acute basis.

Degradates

EPA documentation (EPA 2007) included toxicity data for the 4-hydroxy chlorothalonil degradate, although only acute LC₅₀ data for bluegill sunfish. These two studies (MRID 0029415 and MRID 00030393) resulted in values of 45,000 µg/L and 15,000 µg/L, respectively (median 30,000 µg/L). Based on results from the a.i. and the general observation that salmonids are typically more sensitive to pesticides, we estimated an LC₅₀ for salmonids based on the available data. Using median values of available data for bluegill only (30,000 µg/L for degradate (n=2) and 60 µg/L for a.i. (n=3)), the degradate appears to be 500x less toxic than the a.i. Applying the 500x factor to the median LC₅₀ for salmonids (18 µg/L, n=5) results in an estimated LC₅₀ of 9,000 µg/L for salmonids. Acute toxicity data (EC_{50s}) were also available for the aquatic invertebrate *D. magna* for both the a.i. and the degradate. Two guideline studies were available for the

a.i., MRID 00068754 and MRID 45710221, with EC₅₀s of 68 µg/L and 54 µg/L respectively, resulting in a median of 61 µg/L. Comparing this to the 26,000 µg/L EC₅₀ of the degradate (MRID 0030394) results in a similar ratio of 426. Using EPA's qualitative scale, the 4-hydroxy degradate is classified as slightly toxic to fish and aquatic invertebrates based on available data.

Based on available data, the 4-hydroxy degradate seems less toxic to aquatic organisms on an acute basis. There is, however, little data available to evaluate acute effects, and we located no data to evaluate chronic effects, either in EPA documentation or the open literature. Thus we considered the mechanism of the glutathione interaction with the chlorothalonil molecule to provide additional insight into potential toxicity of 4-hydroxy chlorothalonil. Based on the work of Long and Siegal (Long and Siegel 1975), chlorothalonil interaction with glutathione produces 2 derivatives in fungal cells. The sulfur atom of the cysteine subunit of glutathione substitutes at the 4- or 6- position on the benzene ring of chlorothalonil. Davies (Davies 1985) further characterized the conjugates in rainbow trout, noting that these conjugates were mono-, or di-conjugates, with the first substitution occurring at the 4- site, and the second substitution at the 6- site. He also identified a less commonly occurring tri-conjugate, with a substitution at the 2- site. Long and Siegal (Long and Siegel 1975) noted the glutathione reaction proceeded quickly in fungal cells, indicating parent chlorothalonil is highly reactive, especially at the 4- site. Hydroxyl (-OH groups) are strong electron donors to the ring structure, and chlorines (-Cl) are strong electron withdrawing groups from the ring. Thus the substitution of the -OH at the 4- site would also change the reactivity of the secondary conjugation site. Based on this analysis, the available data is likely a good predictor of the toxicity of the 4-hydroxy degradate compared to the parent.

No data were located regarding the toxicity of the diamide chlorothalonil degradate or the 1-amide 4-hydroxy chlorothalonil degradate. Based on the structural analysis, the 1-amide 4-hydroxy degradate is likely similar in toxicity to the 4-hydroxy degradate, as the

more reactive 4- site has again been substituted. The diamide substitution would behave in a fashion similar to parent chlorothalonil, although perhaps less strongly.

No data were available regarding potential chronic effects of the 4- hydroxyl or other degradates, however we elected to make an estimate using an acute-to-chronic ratio (ACR) for fathead minnow, and applying the degrade factor derived from the bluegill. Based on the LC₅₀ of 23 µg/L and the NOAEC of 3 µg/L for the fathead minnow (EPA 2007), the calculated ACR is 7.67 (unitless). Applying this to the salmonid median LC₅₀ results in a estimated chronic NOAEC of 2.4 µg/L for salmonids. Then multiplying it by the fish degrade factor of 500x, the anticipated chronic toxicity of the 4-hydroxy degrade would be 1,174 µg/L.

Thus, although fate properties indicate the 4-hydroxy degrade is more persistent than parent chlorthalonil, we anticipate it would have to occur in concentrations of ~1,000 µg/L to manifest effects. Based on data we located, we are unable to estimate toxic concentrations of the diamide degrade, although a conservative assumption is toxicity equivalent to the parent.

Formulations

EPA included some test data for formulations of Bravo in the both the salmonid assessment (EPA 2003) and the RLF assessment (EPA 2007). Because these data are more clearly delineated in the RLF assessment, we have opted to use them from that source. Only acute toxicity data were provided, and only the Bravo formulations were tested. Some studies are from the early 1970s, and tested a 75% a.i. product listed as Bravo TM W-75 (MRIDs 0087304, 0087303, and 00087258). Only two are 96 h tests, and we do not present data from the 48 h test (MRID 0087304). For rainbow trout, the formulation LC₅₀ was 103 µg/L (MRID 0087303). When expressed in µg/L a.i., it is given as 77.2 µg/L a.i. For bluegill, the formulation LC₅₀ was 167 µg/L (MRID 0087258). When expressed in µg/L a.i., it is given as 125 µg/L a.i. More recent studies (~1990s) on the formulation Bravo-720 were also available, and test species included

rainbow trout (MRID 43302101), bluegill (MRID 42433804), and *D. magna* (MRID 42433806). For rainbow trout, the formulation LC₅₀ was 61 µg/L and when expressed in µg/L a.i., it is given as 33.2 µg/L a.i. For bluegill, the formulation LC₅₀ was 49 µg/L and when expressed in µg/L a.i., it is given as 26.2 µg/L a.i. For *D. magna*, the formulation LC₅₀ was 180 µg/L and when expressed in µg/L a.i., it is given as 97 µg/L a.i. No chronic tests were reported.

Based on this data, using the toxicity values from TGAI tests would be protective for these formulations. We note that no data is available for other formulations, and that formulations may change over time and for specific uses.

Reproduction and Growth

Other than a somatic growth measurement in Davies et al. (Davies, Cook et al. 1994), we located no information on reproduction or growth except the guideline studies reported in EPA documentation. Davies et al. (Davies, Cook et al. 1994) do not report any values for growth, thus we assume there was no significant effect compared to the controls. One early life-stage test, on fathead minnow, was submitted ((EPA 2007), MRID 00030391). Endpoints affected were hatching success, with reduction of ~10% compared to controls at 6.5 µg/L, and survivability of F₀ generation, with a reduction of ~85% compared to controls at 16 µg/L. At 16 µg/L, hatching success was reduced by ~44%. Based on this test, the NOAEC was 3.0 µg/L, and the LOAEC was 6.5 µg/L.

Olfaction

Tierney et al. (Tierney, Ross et al. 2006), evaluated the effects chlorothalonil, along with endosulfan, glyphosate acid, iodocarb (IPBC), trifluralin, and 2,4-D on juvenile coho salmon electro-olfactograms (EOGs). Chlorothalonil was tested at 1 mg/L (1,000 µg/L), which is actually above the water solubility of the compound. Authors provide a water solubility value of 0.6 mg/L, but EPA documentation lists it as 0.8 mg/L (EPA 2007). Acetone was used as a solvent to achieve the test concentration. Reported concentrations

are nominal, but authors note solutions were prepared daily and report no precipitation. At the test concentration of 1 mg/L, chlorothalonil did not modify the EOG amplitude. Authors state test concentration is “well above the 96 h LC₅₀ range of 14.3 to 17.1 µg/L, so chronic effects or other sublethal effects are likely to develop before olfactory toxicity.”

Although the specific mechanism of salmon olfactory inhibition has not been well established, authors do state:

Most pesticides known to inhibit salmonid olfaction such as the organophosphorus insecticide diazinon, atrazine, and the carbamate fungicide IPBC have acetylcholinesterase-inhibiting properties. . . Considering that elevated peripheral AChE may lead to increased mucous production, it is not unreasonable to hypothesize that olfaction could be diminished by an anticholinesterase mucous plug, which is exactly what Jarrad, et al. (Jarrad, Delaney et al. 2004) hypothesized.”

Biochemical and Immunological Effects

Although EPA routinely requires carcinogenicity and developmental studies in rats and mice to support their human health assessments, they do not regularly consider immunological effects for either human or wildlife risk assessments. Generally, for wildlife, the guideline chronic tests evaluating reproduction (a sensitive endpoint) are thought to provide reliable estimates of no effect concentrations for other endpoints, such as immunological effects. To our knowledge, this assumption has not been empirically tested, but may be appropriate in some instances for some chemicals. However, it may not be appropriate for other cases, especially where effects appear only after prolonged exposure or during a recovery period. Perturbation of metabolic cascades can have far-reaching consequences, including lowered resistance to pathogens, increased susceptibility to other chemical toxicants, or simply a diverting of energetic resources that might otherwise be used for growth or reproduction. Linkage of such effects at a cellular and biochemical level to effects on whole organisms is difficult, and is still an

evolving science. Some evidence of such effects from chlorothalonil exists in open literature, and we present it here, although at this point we are unable to quantitatively link it to organism or population level effects.

Several different research groups have considered sublethal effects. Some of these studies evaluated the mechanism *in vitro*, but others exposed whole organisms to chlorothalonil. Baier-Anderson and Anderson (1998) used an exposure of striped bass (*Morone saxatilis*) phagocytes to chlorothalonil to investigate effects on cell viability, phagocytosis, production of reactive oxygen species (ROS) and production of NADPH. Impairment of these cellular and metabolic processes increases susceptibility to infections (Baier-Anderson and Anderson 1998). They concluded chlorothalonil impairs ROS and NADPH production in the cells at $\geq 250 \mu\text{g/L}$. As this is an exposure, we note there is no clear correlation to actual water concentrations that might lead to this level of cellular exposure.

Davies et al. (Davies, Cook et al. 1994) exposed live fish (*O. mykiss*, *G. maculatus*, and *Pseudaphritis urvilli*) to concentrations of chlorothalonil ranging from 0.3-8.2 $\mu\text{g/L}$ for 10 d. Authors measured a number of parameters. Oxygen consumption (an indicator of stress) for *P. urvilli* increased significantly at 0.3 $\mu\text{g/L}$ and for *O. mykiss* at 1.4 $\mu\text{g/L}$. Increases in liver glutathione and glutathione-*S*-transferase were noted for *O. mykiss* at 1.4 $\mu\text{g/L}$ and for *P. urvilli* at 8.2 $\mu\text{g/L}$. DO consumption and glutathione increases occurred in a dose-dependent manner. *P. urvilli* also showed a significant decrease in leucocrit at 4.4 $\mu\text{g/L}$, but this effect did not appear to occur in a dose-dependent manner. Based on their evaluation, they considered “consistent” indicators to show a NOAEC of 0.8 $\mu\text{g/L}$, and a LOAEC of 1.4 $\mu\text{g/L}$ for sublethal effects. Effects on glutathione and glutathione-*S*-transferase liver enzymes would likely occur prior to effects on immunological endpoints.

Shelley et al. (Shelley, Balfry et al. 2009), evaluated effects of chlorothalonil on the immune responses of rainbow trout. They included a 28 d exposure period and a 14 d

recovery period in their experiment, and exposed the fish at concentrations of 0.1 µg/L, 0.2 µg/L, 0.5 µg/L, and 1.0 µg/L. In addition to measuring biochemical immune response parameters, they also challenged the exposed fish with a marine pathogen, *Listonella anguillarum* during the recovery period. Evaluations of the fish during exposure showed no significant difference in parameters measured (lymphocyte and granulocyte counts, respiratory burst, number of phagocytic cells, and phagocytic capacity during the exposure period. However, they did see significant effects on respiratory burst (0.1 µg/L) and percent phagocytic cells (0.1 µg/L, 0.2 µg/L) during the recovery period. Chlorothalonil appeared to have no significant effect on susceptibility to the pathogen. The authors concluded “Our demonstration of immunomodulation following exposure to environmentally relevant concentrations of chlorothalonil and pentachlorophenol suggests the potential sensitivity of the immune system to the effects of these CUPs [current use pesticides].” Additionally, they illustrated the concern for lingering effects on the organism after the stressor is removed (Shelley, Balfry et al. 2009).

Overall, evidence for immunocompromise is inconclusive, although we do note that sublethal effects have been documented in *Onchorhyncus* species at concentrations below the chronic NOAEC (3.0 µg/L) for fish based on guideline studies submitted to EPA by the registrants.

Indirect Effects to Salmonids (Prey and Habitat Modifications)

Aquatic Invertebrates (Acute and Chronic Toxicity)

EPA described guideline test data for aquatic invertebrates in the both the salmonid assessment (EPA 2003) and the RLF assessment (EPA 2007). Because these data are more clearly delineated in the RLF assessment, we have opted to use them from that source. Two studies evaluated the acute EC₅₀s for *D. magna*. EC₅₀ values were 68 µg/L (MRID 00068754) and 54 µg/L (MRID 45710221), resulting in a median of EC₅₀ of 61µg/L. Two chronic studies were also conducted, with one reporting a NOAEC of 0.6

µg/L and a LOAEC of 1.8 µg/L (MRID 45710222). Another reported a NOAEC of 39 µg/L and a LOAEC of 79 µg/L (MRID 45710222). The NOAEC of 0.6 µg/L and LOAEC of 1.8 µg/L (MRID 45710222) are more consistent with the fish data (NOAEC of 3.0 µg/L, MRID 00030391), and the fact that *D. magna* are generally more sensitive to aquatic toxicants than fish. Although EPA noted (EPA 2007) that there was some uncertainty in the measured concentrations of chlorothalonil in the study, values derived from it were used as endpoints in their risk assessment. Given that the LOAEC (79 µg/L) established in the study producing the higher value (MRID 45710222) is actually greater than the EC₅₀ of 61 µg/L established by the acute tests, we have opted to disregard the higher values in evaluating chronic effects to aquatic invertebrates.

Davies et al. (Davies, Cook et al. 1994), evaluated mortality four species of Australian aquatic invertebrates: the giant Tasmanian freshwater crayfish (*Astacopsis gouldi*), the freshwater aytid shrimp (*Parataya australiensis*), an amphipod (*Neoniphargus sp. A*), and an isopod (*Colubotelson chiltoni minor*). Authors calculated EC₅₀s for all species following exposure durations of 4 d and 7 d, which is longer than the typical 48 h (2 d) acute tests for *D. magna*, but not as long as the chronic (21 d) tests. They were unable to calculate a determinate EC₅₀ for the isopod and amphipod, giving it only as >40 µg/L for both species. However, they were able to determine a 4 d and 7 d for the crayfish (16.0 µg/L and 10.9 µg/L, respectively) and the shrimp (12.0 µg/L and 3.6 µg/L), respectively. Authors also measured sublethal endpoints, such as whole body glutathione and glutathione-S-transferase concentrations/activities and AChE inhibitions. Significant changes in glutathione were observed at 0.3 µg/L for the crayfish, and in both glutathione and glutathione-S-transferase at 1.8 3 µg/L for the crayfish. Thus the NOAEC for glutathione (the most sensitive endpoint) for the crayfish was not established, but is something < 3 µg/L.

In the Davies et al. study (Davies, Cook et al. 1994), the authors also considered AChE inhibition, likely because they were also evaluating organophosphates. They did see

significant inhibition in tail muscle of the crayfish at a chlorothalonil concentration of 6.7 µg/L. The declining inhibition trend continued in the next highest test concentration (17.5 µg/L) and there was mortality at the highest concentration. We have located no other discussion in literature regarding effects of chlorothalonil on AChE.

Aquatic Plants (Phytoplankton and Vascular Plants)

EPA reported aquatic plant data from both guideline studies and open literature in the both the salmonid assessment (EPA 2003) and the RLF assessment (EPA 2007). Because these data are more clearly delineated in the RLF assessment, we have opted to use them from that source.

Guideline tests for freshwater phytoplankton included one for *Selenastrum capricornutum*, a green alga, and *Navicula pelliculosa*, a freshwater diatom. The EC₅₀ and NOAEC for *S. capricornutum* were 190 µg/L and 50 µg/L, respectively. Typically, the green alga is the most sensitive of the phytoplankton species tested, but for chlorothalonil, the diatom was more sensitive. The EC₅₀ and NOAEC for *N. pelliculosa* were 14 µg/L and 3.9 µg/L, respectively. The only vascular plant data reported is for duckweed (*Lemna gibba*). The NOAEC was 290 µg/L and the LOAEC was 630 µg/L based on a reduction in biomass (dry weight).

Microbial Community Effects (Sediment, Soil, and Water Column)

Although EPA does not typically evaluate the effects of registered a.i.s on microbial processes in water, soil, or sediment, NMFS believes these are a relevant endpoint to consider for fungicides, which are targeted at organisms such as mycorrhizae, which serve important functions in soil. We located no relevant studies on these endpoints

Riparian Vegetation

EPA reported terrestrial plant data from both guideline studies and open literature in the RLF assessment (EPA 2007). They located no additional applicable plant information in the open literature. The seed germination/seedling emergence NOAEL and vegetative

vigor NOAEL are both 16 lb a.i./A. Assessment endpoints and associated concentrations for chlorothalonil are summarized in Table 127.

Table 127. Assessment Endpoints and Measures for Chlorothalonil

Assessment Endpoint		Assessment Measure	Median Concentration ¹ (µg/L)	Range	n
Direct Effects on Salmonids	Survival	Salmonid Acute LC ₅₀ ² Galaxias Acute LC ₅₀ ² Other Fish Acute LC ₅₀ ²	18 18.9 55.5	10.5-42.3 16.3-29.2 23-84	5 3 6
	Growth	Chronic NOAEC ² Chronic LOAEC ²	No values reported		
	Reproduction	Chronic NOAEC ² Chronic LOAEC ²	3.0 6.5	NA	1
	Swimming	We located no data regarding effects of chlorothalonil on swimming.			
	Olfaction	EOG NOAEC (Tierney et al 2006)	>1,000	NA	1
	Immune Response	GSH Inhibition NOAEC GSH Inhibition LOAEC (Davies et al 1994)	0.8 1.4	NA	NA
Effects on Prey (Aquatic Invertebrates)	Survival	Acute Invert EC ₅₀ ²	61	54-68	2
	Growth	Chronic NOAEC ² Chronic LOAEC ²	No values reported		
	Reproduction	Chronic NOAEC ² Chronic LOAEC ²	0.3 1.8	NA	1 ³
	Abundance (?)	Meso/microcosm Data, if available and applicable			
Effects on Primary Productivity	Biomass & Abundance	Algal EC ₅₀ ²	14	6.8-190	3
Effects on Submerged and Emergent Vegetation	Biomass & Abundance	Vascular Plant NOAEC ² Vascular Plant EC ₅₀ ²	290 630	NA	1
Effects on Riparian Vegetation	Biomass & Abundance	Terrestrial Plant NOAEL	16 lb ai/A	NA	NA

¹If more than one value was available. If only one value was available, the actual number is given.

²From guideline/standardized test data presented in EPA documents (primary source most recent BE on RLF)

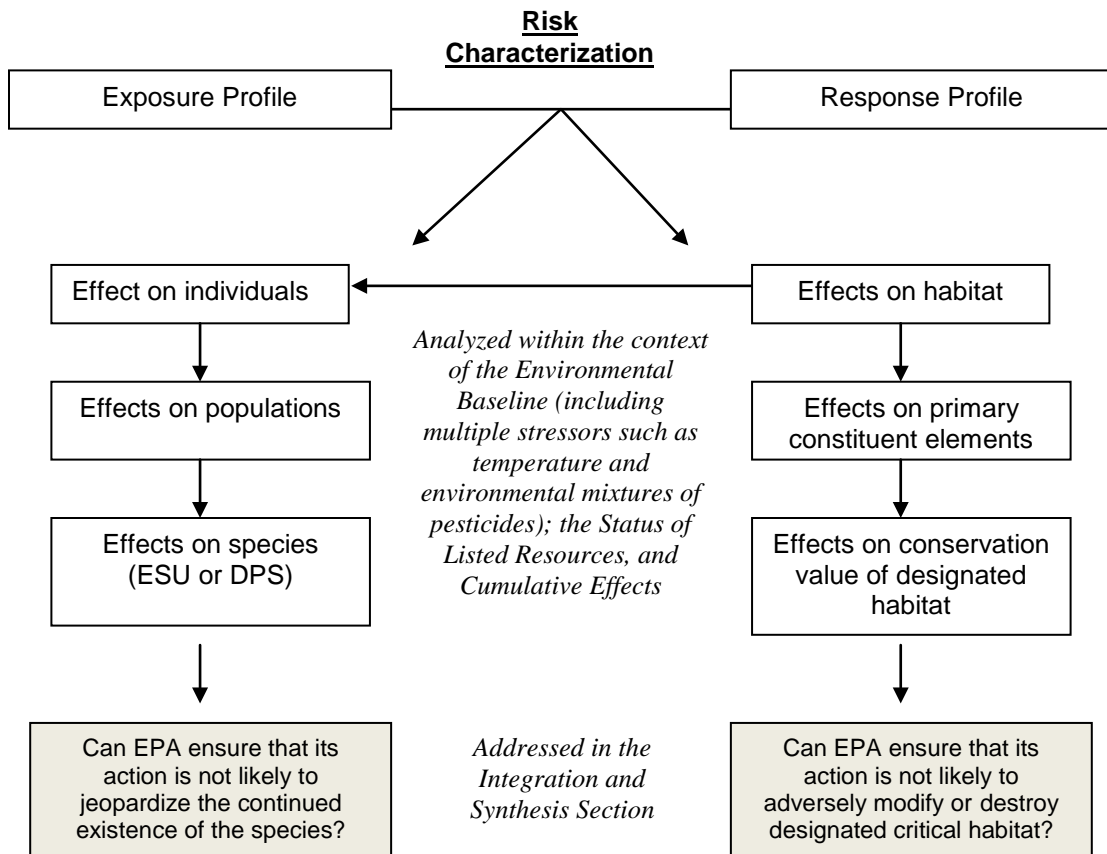
³Two studies available, but only one used. See text

NA Not applicable, only one value available

Risk Characterization

In this section we integrate our exposure and response analyses to evaluate the likelihood of adverse effects to individuals and populations (Figure 76). We combined the exposure analysis with the response analysis to: 1) determine the likelihood of salmonid and habitat effects occurring from the stressors of the action; 2) evaluate the evidence presented in the exposure and response analyses to support or refute risk hypotheses; and 3) translate fitness level consequences of individual salmonids to population-level effects. The risk characterization section concludes with a general summary of species responses from population-level effects. We then evaluate the effects to specific ESUs *i.e.*, species, in the *Integration and Synthesis* section.

Figure 76. Schematic of the Risk Characterization Phase.



Integration of Exposure and Response

In Table 128 through Table 133, we show the overlap between exposure estimates for the 6 a.i.s and concentrations which affect assessment endpoints. This portion of the analysis is based on the a.i., and does not take into account other stressors of the action that may contribute to toxicity, and/or that other a.i.s may be present, creating additive or synergistic toxicity. The tables show the exposure concentration ranges (minimum – maximum values) gleaned from the three sources of exposure data we analyzed: EPA’s estimates presented in the BEs, NMFS’ modeling estimates for floodplain habitats; and surface water monitoring data from ambient monitoring programs and targeted monitoring. In addition to the salmonid BEs submitted to NMFS, we also considered the exposure estimates developed by EPA in the BEs for the California red-legged frog. Some, although not all, of those BEs considered non-crop uses and those estimates have been included in our summary. We have grouped uses into generic categories to better compare the overlap of use sites with landuse information for the various ESUs. The effect concentrations are based on the toxicity data reviewed in the *Response Analysis Section*. For the survival assessment endpoint, effect concentrations are LC_{50s} , however, death of individuals occurring at concentrations below the LC_{50} are not represented by this metric. Consequently, when LC_{50} effect concentrations are not exceeded by the exposure estimates, it does not mean there are no incidences of mortality. We cannot accurately predict at what concentrations death first occurs because dose-response slope information was generally not provided for acute lethality studies. Where overlap occurs between exposure concentrations and effect concentrations, NMFS discusses the potential for adverse effects. If data suggest exposure exceeds adverse effects thresholds, we discuss the likelihood and expected frequency of effects based on species information and results of the exposure and response analyses. This analysis considers both spatial and temporal overlap of the fish with pesticide applications at the scale of the a.i.s themselves, but not at a crop or use category scale. Based on a run-timing analysis (*Appendix 6*) and an evaluation of landuse within the

ESUs (*Appendix 5*) some sensitive life-stage may be present when the a.i.s are applied. This holds true for all ESUs and all a.i.s, although not necessarily all use sites. The analysis is also predicated primarily on standard toxicity endpoints as we located only a few studies with ecologically relevant sublethal information, and that information was not available for all a.i.s. This analysis does allow NMFS to systematically address which assessment endpoints are likely to be affected by exposure to the 6 a.i.s. Where significant uncertainty arises, NMFS highlights the information and discusses its influence on our inferences and conclusions.

2,4-D

Of the a.i.s addressed in this Opinion, 2,4-D has the widest range of uses, and is the only one registered for use as an aquatic herbicide (Table 128). As a direct water application, it is often used to control nuisance weeds, including invasive species such as European watermilfoil. It also has residential and homeowner uses. Given the wide-range of uses, and the persistence of the acid form in water, we believe it is reasonable to assume all populations in most ESUs are exposed to 2,4-D during their lifetimes. 2,4-D is one of the most commonly detected a.i.s considered in this Opinion, with 14 - 48% detections. For 2,4-D we primarily used the EECs from the California red-legged frog assessment (EPA RLF 2009), as it separated the estimates for the ester and acid forms. The ester breaks down relatively quickly in the environment, but uptake of it by fish is rapid. Thus we used data and estimates for the ester to evaluate acute toxicity, and used data and estimates for the acid and amine forms to evaluate chronic toxicity.

While there is a possibility of mortality caused by 2,4-D, as the LC_{50s} for the ester forms do overlap with estimates for the floodplain habitats and for direct applications of ester formulations to water, in general we believe lethal effects to be unlikely. Peak estimates for the floodplain habitats are particularly relevant for the esters, which are taken up by the fish quickly. Endpoints for cellular damage overlap with concentrations in the floodplain habitats, and from the direct water applications. Overall we believe it is possible but unlikely that 2,4-D will kill fish, and possible that it may cause cellular

damage. Use of the ester forms near shallow, low-flow environments creates a particularly vulnerable situation. Based on what data we were able to locate, effects on olfaction appear unlikely.

There is no overlap of prey survival endpoints except in the case of direct water applications, and no overlap with any estimate for any form for the growth and reproduction endpoint. Plants, especially vascular plants, are sensitive to 2, 4-D, and plants may suffer mortality, particularly in direct water applications or in the floodplain habitats. Based on the toxicity profile for 2,4-D, we anticipate the most likely effects to be reduction in biomass, abundance, and diversity of riparian vegetation or vascular plants in-stream.

Table 128. Summary of estimated and measured environmental concentrations, co-occurrence, and effect concentrations for 2,4-D.

Use categories		2,4-D Ester (µg ae/L)	2,4-D Acid, salt, amine (µg ae/L)	All forms, using acid, amine, salt endpoints	Co-occurrence analysis and overlap between endpoints and concentrations
		Peak	Peak	Chronic	
Ag (crop and pasture)	Farm pond ^a	0.55-5.5	0.08-33	0.07-27	All ESUs except Ozette Lake sockeye and Snake River sockeye have some cropland. The Snake River sockeye ESU does have pasture/rangeland which may be treated. Snake River Chinook species do have cropland. The majority have some agricultural areas. Some ESUs have agricultural area that comprise over 20% of land area, such as CA Central Valley species.
	Floodplain ^b	9-956	13-956	NA	
Forest	Farm pond ^a	1.3-13	6-47	5-39	All ESUs have some forested areas, and for many forested area is ≥50% of land area. Generally in forested areas, 2,4-D is used for control of invasive species
	Floodplain ^b	736-1,912	736-1,912	NA	
ROW	Farm pond ^a	NE	6-47	5-39	Rights-of-way uses occur in all ESUs, and are difficult to pinpoint geographically.
	Floodplain ^b	368-1,912	368-1,912	NA	
Urban	Farm pond ^a	NE	NE	NE	Urban and residential use is expected to be a significant source for 2,4-D, as it is in a number of homeowner products
	Floodplain ^b	276	276	NA	
Direct water	Farm pond ^a	1,431-4,000	1,431-4,000	2,610	2,4-D is commonly used to control nuisance plants. The only ester that is approved for direct water application is the butoxyethanol ester
Monitoring data		Databases: 0.007-7.660 (14-48% detections); targeted monitoring maximum: 67.1 µg ae/L			
Assessment Endpoints					Overlap between endpoints and concentrations
Salmonid survival		450-14,500	162,000-2,244,000	162,000-2,244,000	Yes, but only for ester form in floodplain habitats, and direct water applications. Other EECs generally ≥ 10X lower than lowest endpoint. Lethality appears unlikely.
Fish reproduction and growth (NOAEC)		17,000-63,400	17,000-63,400	17,000-63,400	Yes, in floodplain habitats, and direct water. Concentrations not expected to remain high for extended durations.
Olfactory-mediated behaviors		100,000	100,000	100,000	No overlap
Cellular damage		300	1,000	1,000	Overlap for all forms in floodplain habitats. Ester anticipated to be taken up quickly, and cause effects. Available information indicates effects may be reversible if fish are in clean water
Endocrine disruption		ND	ND	ND	Based on available data, appears unlikely
Swimming		ND	ND	ND	

Use categories	2,4-D Ester ($\mu\text{g ae/L}$)	2,4-D Acid, salt, amine ($\mu\text{g ae/L}$)	All forms, using acid, amine, salt endpoints	Co-occurrence analysis and overlap between endpoints and concentrations
	Peak	Peak	Chronic	
Prey survival	3,400	4,970	4,970	No overlap except in direct water applications
Prey reproduction and growth (NOAEC)	79,000	79,000	79,000	No overlap
Primary productivity Algae EC_{50} Vascular plant EC_{50}	≥ 364	390	390	Vascular plants more sensitive than algae. No overlap except in floodplain habitats and direct water applications.
Riparian vegetation	Monocot 0.20 lb ai/A Dicot 0.020 lb ai/A	Monocot 0.40 lb ai/A Dicot 0.003 lb ai/A	NA	Effects likely if applied near waterbody.
Ecosystem functioning $\text{NOEC}_{\text{ecco}}$ $\text{LOEC}_{\text{ecco}}$	10^c 100^c	10^c 100^c	NA	Overlap with all approved use categories. Macrophytes much more sensitive than algae.

ND – No data, NE – Not estimated, NA – Not applicable

^a EPA estimate using PRZM-EXAMS, farm pond from California red-legged frog BE.

^b NMFS estimate in floodplain habitat using AgDrift.

^c (Brock, Lahr et al. 2000)

Triclopyr BEE

Triclopyr is not used generally on most agricultural crops, although it is authorized for use on sod farms, which may be located in agricultural areas, and non-crop agricultural areas. One of its primary use is in forestry, and it is also registered for use on rights-of-way and in industrial areas. Similar to 2,4-D, the ester form breaks down relatively quickly in the environment, but uptake of it by fish is rapid. We use the toxicity data for the BEE to evaluate acute toxicity, and the toxicity data for the TEA to evaluated effects from longer exposures. The salmonid BE (EPA slamonid 2004) did not provide chronic EECs or evaluate these effects, and the California red-legged frog BE appears to have used acute and chronic values both based on fate parameters of the ester form. We expect chronic concentrations estimated based on the persistence of the acid or amine forms might be higher. While the best type of estimate for the evaluation might be one that was developed using a total toxic residues (TTR) approach, none of the EPA documents had done a TTR. We used the estimates provided, along with NMFS-generated floodplain estimates. Floodplain estimates address only short-term (acute) concentrations.

There was little toxicity data available for triclopyr, either for the BEE or the TEA form, thus there is significant uncertainty associated with our conclusions. However, based on what data was available there is a significant overlap of the mortality endpoint for the fish and all estimated concentrations (Table 129). The TEA is much less toxic, and there was no overlap with the reproduction and growth endpoints for fish. The acid form is relatively persistent in water, and percent detections ranged from 8.7-17.5%, depending on database. There was a slight overlap in prey survival endpoints with both some of the farm pond estimates and some of the floodplain estimates. There is overlap the aquatic plant endpoints and narly all estimates. Riparian vegetation is also likely to be affected. We anticipate there could be fish mortality if triclopyr BEE is used adjacent to a waterbody, and that waterbody receives input of triclopyr BEE or its degradates in the ester form. Given the overlap of assessment endpoints and concentrations for aquatic

plants, we expect some reduction in plant biomass, diversity, and abundance, both in-stream and in the riparian zone.

Table 129. Summary of estimated and measured environmental concentrations, co-occurrence, and effect concentrations for triclopyr BEE.

Use categories		Triclopyr BEE (µg ae/L)	Triclopyr TEA (µg ae/L)	Co-occurrence analysis and overlap between endpoints and concentrations
		Peak	Peak	
Ag (crop and pasture)	Farm pond ^a	21-990	16-794	All ESUs except Ozette Lake sockeye and Snake River sockeye have some cropland. The Snake River sockeye ESU does have pasture/rangeland which may be treated. Snake River Chinook species do have cropland. The majority have some agricultural areas. Some ESUs have agricultural area that comprise over 20% of land area, such as CA Central Valley species. Triclopyr BEE use in agricultural areas is typically comerically grown turf, ornamentals, and abandoned orchards.
	Floodplain ^b	368-1,471	NA	
Forest	Farm pond ^a	NE	NE	All ESUs have some forested areas,and for many forested area is ≥50%of land area. Triclopyr commonly used in forested areas.
	Floodplain ^b	1,103-2,868	NA	
ROW	Farm pond ^a	NE	NE	Rights-of-way uses occur in all ESUs, and are difficult to pinpoint geographically. Triclopyr commonly used in ROW applications.
	Floodplain ^b	1,147-3,824	NA	
Urban	Farm pond ^a	75-1,499	61-1,172	Urban and residential use is expected.
	Floodplain ^b	184-3,824	NA	
Monitoring data		In databases as triclopyr or triclopyr acid, not triclopyr BEE 0.025-14.5 (8.7-17.5% detections)		
Assessment Endpoints				Overlap between endpoints and concentrations
Salmonid survival		470	NA	Ranges overlap for all other use categories.
Fish reproduction and growth (NOAEC)		NA	>32,200	No overlap
Olfactory-mediated behaviors		ND	ND	Not evaluated
Cellular damage		ND	ND	Not evaluated
Endocrine disruption		ND	ND	Not evaluated
Swimming		ND	ND	Not evaluated
Prey survival		250	346,000	Ranges overlap for all use categories.
Prey reproduction and growth (NOAEC)		NA	25,500	No overlap for any uses or estimates
Primary productivity (Vasc plant EC ₅₀)		70-2,500	ND	Overlap for all uses and estimates

Use categories	Triclopyr BEE (µg ae/L)	Triclopyr TEA (µg ae/L)	Co-occurrence analysis and overlap between endpoints and concentrations
	Peak	Peak	
Riparian vegetation	Monocot 0.053 lb ai/A Dicot 0.006 lb ai/A	NA	Effects likely if applied near waterbody.
Ecosystem functioning NOEC _{ecco} LOEC _{ecco}	320 ^c 3,200 ^c	ND	Ranges overlap for all use categories. Review of pulse exposure in flowing system (Kreutzweiser and Capell 1992). Sensitive endpoint was impact to invertebrates.

ND – No data, NE – Not estimated, NA – Not applicable

^a EPA estimate using PRZM-EXAMs, farm pond from California red-legged frog BE.

^b NMFS estimate in floodplain habitat using AgDrift

^c (Brock, Lahr et al. 2000)

Diuron

There is overlap with fish survival endpoints for all estimated concentrations except the farm pond estimates for agricultural uses (Table 130). Diuron is heavily used as right-of-way herbicide, so these higher EECs are of concern. It is also relatively persistent in water, so there may be extended exposure in lower-flow environments such as ponds and lakes, and/or an extended chemical plume in faster-flowing waters such as streams and rivers. Measured concentrations are lower, but it is fairly frequently detected in monitoring (29-47% detections), with more frequent detections in California. We are uncertain of the specific reason for greater detection frequency, but likely it is related to use. There is significant overlap in the estimated concentrations and the lower end of the fish reproduction and growth assessment endpoints. Diuron has not been identified as an endocrine disruptor, but linuron has, and given structural similarities, we believe it is reasonable to use the linuron endocrine disruption endpoint for diuron. There is significant overlap of this concentration and estimates from all uses. It is also within the range of detected values. We expect that diuron will sometimes kill salmonids, but that more often, there will be effects on growth, reproduction, and perhaps endocrine systems.

There is some overlap of the prey survival endpoints with all estimated concentrations except the farm pond estimates for agricultural uses, but no overlap with reproduction and growth estimates. We note that if diuron does have endocrine disrupting properties, a single-generation test on *Daphnia* may not be sensitive enough to detect these types of effects. Thus, risk to aquatic invertebrates may be underestimated by available data. We anticipate there may be some reduction in abundance and/or diversity of prey species.

There is significant overlap of primary productivity endpoints (algal and vascular plants) with all uses and estimates, and these endpoints are also within the range of measured concentrations. Estimates for all uses and measured concentrations are also much higher than the most sensitive endpoint of ecosystem functioning (NOAEC 0.5 µg/L). We believe the in-stream plant community of any waterbody receiving diuron will be affected

by it. Effects may include reductions in plant biomass, loss of diversity, and shifts in the plant community to more tolerant species. Based on use rates and available data, we anticipate effects on riparian vegetation if diuron is applied near the waterbody. Effects may include reductions in plant biomass, loss of diversity, and shifts in the plant community to more tolerant species.

Table 130. Summary of estimated and measured environmental concentrations, co-occurrence, and effect concentrations for diuron.

Use categories		Diuron (µg ai/L)	Diuron (µg ai/L)	Co-occurrence analysis and overlap between endpoints and concentrations
		Peak	Chronic	
Ag (crop and pasture)	Farm pond ^a	5-140	3-103	All ESUs except Ozette Lake sockeye and Snake River sockeye have some cropland. The Snake River sockeye ESU does have pasture/rangeland which may be treated. Snake River Chinook species do have cropland. The majority have some agricultural areas. Some ESUs have agricultural area that comprise over 20% of land area, such as CA Central Valley species.
	Floodplain ^b	221-1,147	NA	
Forest	Farm pond ^a	NA	NA	Diuron is not approved for forestry uses.
	Floodplain ^b	NA	NA	
ROW	Farm pond ^a	37-4,911	24-3,428	Rights-of-way uses occur in all ESUs, and are difficult to pinpoint geographically. ROW important use site for diuron.
	Floodplain ^b	2,207	NA	
Urban	Farm pond ^a	37-4,911	24-3,428	Urban and industrial use is expected to be a source of diuron
	Floodplain ^b	2,207	NA	
Monitoring data		0.002-160 (29-47% detections) More detections and higher concentrations in CA databases than WA databases; maximum concentrations detected in target surface water runoff: 467 µg ai/L vinyards, 1,700 µg ai/L citrus, 2,849 µg ai/L rights-of-way.		
Assessment Endpoints				Overlap between endpoints and concentrations
Salmonid survival		710-23,800	NA	Ranges overlap for all use categories except farm pond Ag uses.
Fish reproduction and growth (NOAEC)		NA	5-4,200	Ranges overlap for all use categories. Commonly detected in monitoring data, although at much lower concentrations than models.
Olfactory-mediated behaviors		10 ^c	10 ^c	Ranges overlap for all use categories.
Cellular damage		ND	ND	Not evaluated
Endocrine disruption		25	25	Ranges overlap for all use categories.
Swimming		ND	ND	Not evaluated.
Prey survival		160-19,400	NA	Ranges overlap for all use categories.
Prey reproduction and growth (NOAEC)		NA	4,000	No overlap with any estimates

Use categories	Diuron ($\mu\text{g ai/L}$)	Diuron ($\mu\text{g ai/L}$)	Co-occurrence analysis and overlap between endpoints and concentrations
	Peak	Chronic	
Primary productivity (Vasc plant EC_{50})	10-95	NA	Significant overlap for all uses and estimates. Commonly detected in monitoring data, although at much lower concentrations than models. Some detections overlap with effects range.
Riparian vegetation	Monocot 0.0164 lb ai/A Dicot lb ai/A	Monocot lb ai/A Dicot lb ai/A	Effects likely if applied near waterbody.
Ecosystem functioning $\text{NOEC}_{\text{ecco}}$ $\text{LOEC}_{\text{ecco}}$	0.5 2.9 ^d 28.5 ^d	0.5	Ranges overlap for all use categories.

ND – No data, NE – Not estimated, NA – Not applicable

^a EPA estimate using PRZM-EXAMs, farm pond from California red-legged frog BE.

^b NMFS estimate in floodplain habitat using AgDrift

^cNot determined for diuron, assume similar response as determined for linuron.

^d(Brock, Lahr et al. 2000).

Linuron

Linuron is primarily a specialty crop herbicide, with uses such as carrots. It does have some rights-of-way uses, but is not commonly selected for these uses due to cost, according the representatives for the applicant (expressed verbally at applicant meeting 10/08/2010). Data for some of the standard endpoints was not available for linuron, but based on what data are available, toxicity appears similar to diuron. Also, like diuron, it is persistent in aquatic systems. However, it has a much lower rate of detection than diuron (0.007-1.6% as opposed to 29-47%). This is likely a reflection of differences in usage.

There is no overlap of fish assessment endpoints with estimated or measured uses other than the NOAEC for endocrine disruption. The endocrine disruption endpoint overlaps with all estimates from all uses, although it is higher than measured concentrations (Table 131). Endocrine disruption effects would likely be expressed in lowered reproductive rates, and given linuron's persistence in water, could occur in waterbodies receiving direct runoff. Slow-moving waters such as the floodplain habitats would be most at risk for endocrine effects.

Estimated and measured concentrations do not overlap with prey survival endpoints. Although there were no data on aquatic invertebrate reproduction and growth, based on comparisons with diuron data and structural similarities between the two a.i.s, we do not anticipate estimated and measured concentrations would overlap with them either.

There is significant overlap of primary productivity endpoints (algal and vascular plants) with all uses and estimates, and these endpoints are also within the range of measured concentrations. Estimates for all uses and measured concentrations are also much higher than the most sensitive endpoint of ecosystem functioning (NOAEC 0.3 µg/L). We believe the in-stream plant community of any waterbody receiving linuron will be affected by it. Effects may include reductions in plant biomass, loss of diversity, and

shifts in the plant community to more tolerant species. No data were available for terrestrial plant endpoints. However, based on linuron's mode of action, and recommended application rates, we anticipate there will be effects on riparian vegetation if it is applied near the waterbody. Effects may include reductions in plant biomass, loss of diversity, and shifts in the plant community to more tolerant species.

Table 131. Summary of estimated and measured environmental concentrations, co-occurrence, and effect concentrations for linuron.

Use categories		Linuron (µg ai/L)	Linuron (µg ai/L)	Co-occurrence analysis and overlap between endpoints and concentrations
		Peak	Chronic	
Ag (crop and pasture)	Farm pond ^a	2.6-41 ¹	1.8-31 ¹	All ESUs except Ozette Lake sockeye and Snake River sockeye have some cropland. The Snake River sockeye ESU does have pasture/rangeland which may be treated. Snake River Chinook species do have cropland. The majority have some agricultural areas. Some ESUs have agricultural area that comprise over 20% of land area, such as CA Central Valley species. Linuron is a specialty crop chemical, especially for carrots
	Floodplain ^b	184-736	NA	
Forest	Farm pond ^a	NA	NA	No forest uses
	Floodplain ^b	NA	NA	
ROW	Farm pond ^a	60-337 ¹	39-211 ¹	Rights-of-way uses occur in all ESUs, and are difficult to pinpoint geographically. ROW uses for linuron are approved, but it is not used extensively for this purpose.
	Floodplain ^b	552	NA	
Urban	Farm pond ^a	NA	NA	No urban uses
	Floodplain ^b	NA	NA	
Monitoring data		0.007-1.6 (0.16-3.2% detections)		
Assessment Endpoints				Overlap between endpoints and concentrations
Salmonid survival		3,000	NA	No overlap with any estimates
Fish reproduction and growth (NOAEC)		NA	<42	Uncertain endpoint, may overlap with Ag estimate. Overlaps with ROW estimate
Olfactory-mediated behaviors		10	10	Ranges overlap for all use categories.
Cellular damage		ND	ND	Not evaluated
Endocrine disruption		25	25	Significant overlap for all uses and estimates
Swimming		ND	ND	Not evaluated
Prey survival		270-2,900	130	Overlap with ROW and Ag floodplain estimates
Prey reproduction and growth (NOAEC)		ND	ND	Not evaluated
Primary productivity (Vasc plant EC ₅₀)		2.5-67	2.5-67	Significant overlap for all uses and estimates
Riparian vegetation		ND	ND	Not evaluated

Use categories	Linuron (µg ai/L)	Linuron (µg ai/L)	Co-occurrence analysis and overlap between endpoints and concentrations
	Peak	Chronic	
Ecosystem functioning NOEC _{ecco} LOEC _{ecco}	0.3 0.5 ^c 5 ^c	0.3	Ranges overlap for all use categories.

ND – No data, NE – Not estimated, NA – Not applicable

^a EPA estimate using PRZM-EXAMs, farm pond from California red-legged frog BE.

^b NMFS estimate in floodplain habitat using AgDrift

^c(Brock, Lahr et al. 2000).

Captan

Captan dissociates rapidly in aquatic systems, so exposure to the parent molecule is expected to be limited. Parent captan breaks down into a ring structure (THPI) and the toxic moiety, a trichloromethylthio molecule (TCMT). EPA documentation provided virtually no data on TCMT, either from a fate or toxicity perspective, and we did not locate any data regarding it in our literature review. The discussion below is relevant to effects of the parent.

There is no overlap of fish survival endpoints with farm pond estimates from agricultural uses, but there is overlap with farm pond estimates for turf uses. Reproduction and growth endpoints, and genotoxicity endpoints overlap with all peak exposure estimates for all uses (Table 132). Information to assess the duration of exposure necessary to elicit these responses was not available for captan. However, the reproduction, growth, and genotoxicity effects may require longer exposure durations to the parent molecule than is likely to occur in a natural environment. We do not anticipate captan will kill fish. It may cause sublethal effects.

There is no overlap of prey survival, reproduction, or growth endpoints with any of the chronic exposure estimates for any uses. There is no overlap of primary productivity endpoints, either algal or vascular plants, with any estimates for any uses. Based on what data are available, there could be effects on riparian vegetation if captan is applied near the waterbody. We do not anticipate captan will reduce prey abundance or diversity. We do not anticipate captan will modify primary productivity or riparian vegetation.

Table 132. Summary of estimated and measured environmental concentrations, co-occurrence, and effect concentrations for captan.

Use categories		Captan (µg ai/L)	Captan (µg ai/L)	Co-occurrence analysis and overlap between endpoints and concentrations
		Peak	Chronic	
Ag (crop and pasture)	Farm pond ^a	<0.001-21.6	<0.001-0.06	All ESUs except Ozette Lake sockeye and Snake River sockeye have some cropland. The Snake River sockeye ESU does have pasture/rangeland which may be treated. Snake River Chinook species do have cropland. The majority have some agricultural areas. Some ESUs have agricultural area that comprise over 20% of land area, such as CA Central Valley species.
	Floodplain ^b	368-2,151	NA	
Forest	Farm pond ^a	NA	NA	Captan is not approved for forestry uses.
	Floodplain ^b	NA	NA	
ROW	Farm pond ^a	NA	NA	No ROW uses
	Floodplain ^b	NA	NA	
Urban	Farm pond ^a	3.6-29	0.08-1.1	Urban uses are primarily turf.
	Floodplain ^b	184		
Monitoring data		Infrequently detected, not included in NAWQA monitoring (0-1.3 % detections)		
Assessment Endpoints				Overlap between endpoints and concentrations
Salmonid survival		26.2-137	NA	Overlap with farm pond estimates for turf uses. Overlap with floodplain estimates for ag and turf uses)
Fish reproduction and growth (NOAEC)		NA	16.5	No overlap with any estimates
Olfactory-mediated behaviors		ND	ND	Not evaluated
Cellular damage		15.6-31.3	15.6-31.3	Significant overlap for all uses and estimates
Endocrine disruption		ND	ND	Not evaluated
Swimming		ND	ND	Not evaluated
Prey survival		1,200-8,400	NA	Overlap would include only Ag Floodplain habitats that receive aerial application at higher application rates.
Prey reproduction and growth (NOAEC)		NA	560	No overlap with any estimates

Use categories	Captan (µg ai/L)	Captan (µg ai/L)	Co-occurrence analysis and overlap between endpoints and concentrations
	Peak	Chronic	
Primary productivity (Vasc plant EC ₅₀)	1,200 >12,700	1,200 >12,700	No overlap with any estimates
Riparian vegetation	Dicot 2.4 lb ai/A	Dicot 2.4 lb ai/A	Effects possible if applied near waterbody
Ecosystem functioning	ND	ND	Not evaluated

ND – No data, NE – Not estimated, NA – Not applicable

^a EPA estimate using PRZM-EXAMs, farm pond from California red-legged frog BE.

^b NMFS estimate in floodplain habitat using AgDrift

Chlorothalonil

For chlorothalonil, there is a significant overlap for fish survival, reproduction and growth, and immune response endpoints with all estimates for all uses (Table 133). There is overlap of the immune response endpoint with measured concentrations. However, chlorothalonil degrades rapidly in aquatic ecosystems via photolytic processes. It is detected infrequently in ambient monitoring programs (0-1.88% detections). When it is detected, it is typically at low levels in comparison to the assessment endpoints (0.003-0.36 µg/L). However, a recent targeted monitoring study on managed turf found stream concentrations up to 48 µg/L, exceeding many of the assessment endpoints (King and Balogh 2010). We expect chlorothalonil may sometime kill fish if it enters a waterbody via overspray or in a runoff event shortly following application.

There is also a significant overlap in prey endpoints for survival, reproduction, and growth. There is no overlap in endpoints and estimated concentrations for vascular plants, but there is for algal plants. Based on the same fate properties discussed above, We expect chlorothalonil may sometime kill aquatic invertebrates or phytoplankton if it enters a waterbody via overspray or in a runoff event shortly following application. Based on available data, we believe it is unlikely that chlorothalonil will adversely affect riparian vegetation.

Table 133. Summary of estimated and measured environmental concentrations, co-occurrence, and effect concentrations for Chlorothalonil.

Use categories		Chlorothalonil (µg ai/L)	Chlorothalonil (µg ai/L)	Co-occurrence analysis and overlap between endpoints and concentrations
		Peak	Chronic	
Ag (crop and pasture)	Farm pond ^a	3-69	2-43	All ESUs except Ozette Lake sockeye and Snake River sockeye have some cropland. The Snake River sockeye ESU does have pasture/rangeland which may be treated. Snake River Chinook species do have cropland. The majority have some agricultural areas. Some ESUs have agricultural area that comprise over 20% of land area, such as CA Central Valley species.
	Floodplain ^b	184-3,490	NA	
Forest	Farm pond ^a	19-274	12-146	All ESUs have some forested areas,and for many forested area is ≥50%of land area. Forestry use is approved in WA, but not elsewhere.
	Floodplain ^b	1,960	NA	
ROW	Farm pond ^a	NA	NA	Chlorthalonil is not approved for ROW uses.
	Floodplain ^b	NA	NA	
Urban	Farm pond ^a	19-274	12-146	Urban and residential use are primarily turf.
	Floodplain ^b	2,078	NA	
Monitoring data		0-1.88% detections; targeted monitoring maximum 48.1 µg ai/L.		
Assessment Endpoints				Overlap between endpoints and concentrations
Salmonid survival		10.5-42.3	NA	Significant overlap for all uses and estimates
Fish reproduction and growth (NOAEC)		NA	3.0	Significant overlap for all uses and estimates
Olfactory-mediated behaviors		>1,000	>1,000	Uncertain endpoints. No overlap with farm pond estimates. Overlap with floodplain estimates for all uses
Cellular damage		0.8	0.8	Significant overlap for all uses and estimates
Endocrine disruption		ND	ND	Not evaluated
Swimming		ND	ND	Not evaluated
Prey survival		54-68	NA	Significant overlap for all uses and estimates
Prey reproduction and growth (NOAEC)		NA	0.3	Significant overlap for all uses and estimates
Primary productivity (Vasc plant EC ₅₀)		14 630	14 630	Significant overlap for all uses and estimates for algal plants. Significant overlap with floodplain estimates for vascular plants.

Use categories	Chlorothalonil (µg ai/L)	Chlorothalonil (µg ai/L)	Co-occurrence analysis and overlap between endpoints and concentrations
	Peak	Chronic	
Riparian vegetation	16 lb ai/A	16 lb ai/A	Effects unlikely
Ecosystem functioning	ND	ND	Not evaluated

ND – No data, NE – Not estimated, NA – Not applicable

^a EPA estimate using PRZM-EXAMs, farm pond from California red-legged frog BE.

^b NMFS estimate in floodplain habitat using AgDrift.

Mixture Analysis

More than 50 pesticides products are currently registered that contain one of the six a.i.s and a least one other pesticide. It is not feasible for us to assess the unique risk of each of these individual products. As an example of potential risk of pesticide product mixtures, we more closely evaluate the risk associated with one of these products formulated with two active ingredients, diuron (62.22% of the formulation by weight) and imazapyr (7.78% of the formulation by weight). Both a.i.s are herbicides so we focus on potential changes to the plant communities in surface water and the riparian zone. This product is approved for use on a variety of rights-of-way and non-crop agricultural lands. It can be applied at rates of 19 lbs of product per acre. To simulate a rights-of-way application we assumed a single aerial swath was applied by air. We used AgDrift to estimate the initial average concentration in a floodplain habitat, with and without a 300 foot buffer to the aquatic habitat (Table 134). This buffer was chosen because a 2004 order for injunctive relief requires implementation of no-spray buffers to certain water containing listed salmon in California, Oregon, and Washington. Buffers of 60 feet for ground applications and 300 feet for aerial applications are in effect until EPA completes its consultation on diuron.

Table 134. AgDrift estimated concentrations of pesticides in surface water adjacent to aerial application at the maximum labeled use rate for EPA Reg. No. 228-654 (12 lbs diuron/A and 1.5 lb imazapyr/A).

Chemical	Buffer	Average initial concentration (µg/L)	
		EPA-defined pond	NMFS-defined floodplain habitat
With injunctive relief buffers			
diuron	300	1.002	30.8
imazapyr	300	0.125	3.85
Without aquatic habitat buffers			
diuron	0	25.0	2,789
imazapyr	0	3.12	349
Label prohibits use of product in California			

We also used AgDrfit to estimate deposition of the two active ingredients on riparian zone habitat at various distances downwind from the targeted treatment site (Table 135).

Table 135. AgDrift estimated point deposition of diuron and imazapyr at various distances downwind from application site at the maximum labeled use rate for EPA Reg. No. 228-654 (12 lbs diuron/A and 1.5 lb imazapyr/A).

Buffer in Feet	Rate (lbs/A)	
distance from edge of treatment area	Diuron	Imazapyr
10	1.74	0.217
100	0.250	0.031
300	0.028	0.004

Next we compared the concentrations estimated for the floodplain and riparian habitats to toxicity values for aquatic and terrestrial plants. We did not do a thorough search of the toxicity data available for imazapyr but considered the toxicity values used by EPA in their 2006 registration eligibility decision for imazapyr (Table 136). The values for aquatic plants suggest imazapyr is comparable to diuron in its toxicity to vascular plants but is much less toxic to nonvascular plants. However, imazapyr has much greater toxicity to both terrestrial monocots and dicots compared to diuron.

Table 136. Toxicity of diuron and imazapyr to nontarget plants

Assessment Endpoint	Concentration (µg/L)	
	Diuron ¹	Imazapyr ²
<i>Aquatic concentration (µg/L)</i>		
Aquatic vascular plant EC ₅₀	15	18
Aquatic nonvascular plant EC ₅₀	2.4	11,500
<i>Terrestrial rate (lbs/A)</i>		
Terrestrial monocot EC ₂₅		
Emergence	0.099	0.0046
Vegetative vigor	0.021	0.012
Terrestrial dicot EC ₂₅		
Emergence	0.075	0.0024
Vegetative vigor	0.002	0.0009
1 – (EPA 2009)		
2 – (EPA 2006)		

These data suggests impacts to aquatic plants are likely for diuron in some aquatic habitats; estimated concentrations exceeded median toxicity thresholds for both vascular and nonvascular plants, with and without the buffer. Concentrations of imazapyr were greater than the median toxicity threshold for vascular plants when there was not a buffer to the aquatic habitat; imazapyr concentrations did not exceed the median toxicity threshold for nonvascular plants in either estimate, or vascular plants when a 300 foot buffer was simulated. The data also suggest impacts to nontarget terrestrial vegetation are also likely for diuron and imazapyr. Downwind deposition of diuron exceeded EC₂₅ values for plant emergence and vegetative in both monocots and dicots at distances of 100 feet and less; there was also some overlap with these endpoints at the 300 foot buffer distance suggesting plant impacts throughout the buffer would be likely. The data suggest even greater risks associated with imazapyr to terrestrial vegetation; all EC₂₅ values for plant emergence and vegetative were exceeded in both monocot and dicot plants.

When the toxicity values of the individual a.i.s are considered independently of one another they suggest reduction of primary producers in surface waters and riparian habitats are likely, particularly if the buffers to those habitats are reduced. We expect that the combined exposure to these two active ingredients will amplify these potentially harmful responses. Herbicides are frequently combined into pesticide product mixtures because such mixtures improve the spectrum of weeds controlled and/or increase their effectiveness against target weeds. We expect the same is true of nontarget plants in the riparian area and aquatic habitats; exposure to a mixture of diuron and imazapyr is more likely to cause a reduction in the abundance of primary producers than exposure to diuron alone. It should also be noted that this particular product authorized tank mixtures with a number of other herbicides that increase the likelihood of exposure of salmonids and their habitat to mixtures will occur, and adverse responses may be amplified. For example, the label advises “for faster burndown or brown-out of target weeds, tank mix this product with Roundup® or Finale®.” It also authorizes tank mixtures with several other herbicides including Oust®, Garlon®, MSMA, Banvel®, Plateau®, and Arsenal®. We

assume co-application of these products increase the risk of adverse responses to the riparian zone and aquatic habitats. Further evaluation of the effects of herbicide mixtures and herbicide-induced modifications to riparian and aquatic habitats are discussed in the *Risk Hypotheses* section below.

Evaluation of Risk Hypotheses:

In this phase of our analysis we examine the weight of evidence from the scientific and commercial data to determine whether it supports or refutes a given risk hypothesis. This is not a statistical analysis, but rather a qualitative weighing of the available lines of evidence. We also highlight general uncertainties and data gaps associated with the data. In some instances there may be no information specifically related to a given hypothesis. In some cases, if information on a similar endpoint or chemical is available, and it is reasonable to do so, we extrapolate from the available data to fill gaps, recognizing that this may introduce additional uncertainty in the analysis. If the evidence supports the hypothesis we determine whether it warrants an assessment at the population level. Although six a.i. are addressed in this Opinion, we recognize the modes of action and toxicities of these compounds vary widely, and have considered them separately through the analysis. In some cases, a group of compounds may be discussed together in this section if toxicities are in a similar range, or the toxicity/exposure profiles are similar.

The available information to characterize pesticide exposure included surface water monitoring data and estimates from pesticide transport models. We combine this information with the distribution and life-history characteristics of listed Pacific salmonids. As discussed in the *Exposure Analysis* section above, each source of information has inherent limitations and uncertainties. For example, the pesticide monitoring data were generally not designed to quantify peak exposure concentrations or distributions of exposure in listed Pacific salmonid habitats. Consequently, models were used to supplement monitoring data and together the information was used to describe the potential range of pesticide concentrations in salmonid habitats. The NMFS AgDrift model runs provided estimates for concentrations resulting from drift to a shallow and

narrow body of water, such as those found in floodplain habitats used by listed Pacific salmonids. Small streams and many floodplain habitats are more susceptible to higher pesticide concentrations than larger, high flow systems as their physical characteristics provide less dilution.

We recognize that pesticide concentrations will vary greatly among habitats used by salmonids, and exposure durations will be reduced in flowing water systems where higher velocities occur. There is uncertainty as to what the magnitude of response of fish and salmonid prey will be under different environmental dissipation patterns.

Standardized toxicity tests for pesticide registration are poor predictors of real world aquatic ecosystems as fish and other test organisms are exposed to relatively constant pesticide concentrations for arbitrary durations (*e.g.* acute, 96 h and chronic, 21 d) that may poorly reflect field exposures, which tend to be repeated pulses. The response of fish and their prey to different durations of exposure, and exposure mimicking different environmentally relevant dissipation patterns of the six a.i.s is a prominent data gap. **We therefore did not average exposure concentrations over time, so called time-weighted averages, because adverse responses to short term exposures such as pulses would likely be masked.**

Large spatial and temporal variability exists in the use of aquatic habitats by listed Pacific salmonids. These differences occur at multiple scales of biological organization (*i.e.*, individual, population, and species). Both an individual's lifestage and its life history are important considerations in its use of aquatic habitats. This natural variation is overlaid with the inherent variation of environmental factors including climate (*e.g.*, precipitation patterns), habitat stressors, and land use. Given this biological and environmental variability, it is difficult to predict the precise exposure to the stressors of the action for any one individual let alone for a population or species.

Consequently, we used general life history information to evaluate potential exposure in the myriad aquatic habitats. For example, all listed Pacific salmon and steelhead occupy

habitats that could contain high concentrations of these pesticides at one or more life stages. That said, populations show temporal variation in use of those habitats. Most species use shallow floodplain habitats and/or small streams during their freshwater and estuarine rearing period. These periods of development and growth can differ significantly between species and populations (Refer to *Appendix 6*). Coho, steelhead, sockeye, and stream-type Chinook spend much longer in freshwater systems prior to migrating to the ocean, while ocean-type Chinook and chum spend less time rearing in freshwater. Ocean-type Chinook migrate from their natal stream within 2-6 months of hatching and spend several months rearing in floodplain, estuary, nearshore habitats before continuing on to the open ocean. Chum spawn in side channels, tributary streams, and mainstem rivers. The egg and alevin life stages reside at these sites until they approach or reach the fry stage. Swim-up fry immediately migrate downstream to estuarine areas, where they typically reside near the shoreline for one or more weeks. Thus, a chum fry's freshwater residency period is only a few days, compared with more than a year for other species such as steelhead.

To account for the temporal and spatial variation of aquatic habitats across individuals, populations, and species, we evaluated the potential for individual fitness consequences, (*i.e.*, assessment endpoints) by comparing the range in expected exposure concentrations with adverse effect levels in the context of aquatic habitat utilization. We divided salmonid habitats into two basic groups.

The first group is composed of spawning and rearing habitats. These freshwater aquatic habitats range from first order streams to large mainstem rivers as well as lakes. They are essential to successful reproduction and to the development and growth of young fish.

The second habitat group is composed of migratory corridors, estuaries, and nearshore marine areas. Most salmonid species use some of these habitats to migrate and rear (feed, develop, shelter), prior to moving into open ocean areas. In general, pesticide exposure will likely be less intense in these areas compared to freshwater systems given

their size, flow, and use by salmonids. Exceptions include estuaries and nearshore marine environments where juveniles are rearing for extended periods (weeks-months) proximate to high pesticide use areas such as right of ways, agricultural operations near tidal areas and stormwater runoff from dense urban centers.

Although we recognize this as a simplification of the diversity in life histories as well as aquatic habitats used by listed Pacific salmonids, the framework allows us to evaluate risk hypotheses based on differences in habitats and their use by salmonids. We explicitly address species differences in the *Integration and Synthesis* section by evaluating the potential for the stressors of the action to jeopardize the continued existence of the species; or for the potential for stressors to adversely modify their designated critical habitat. Ultimately, for each of the risk hypotheses we make a determination of whether fitness of individuals is compromised to warrant an analysis at the population level.

Risk Hypotheses

Here we evaluate the available evidence to determine whether each risk hypothesis is supported. If the available information supports a hypothesis, we analyze the effects at the population scale. If the available information does not support a hypothesis, we do not conduct population level analyses.

1. Exposure to 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil is sufficient to:

A. Kill salmonids from direct, acute exposure

Species' life history information indicates that listed salmonids are at the greatest risk of exposure to acutely toxic concentrations of the six a.i.s during freshwater occupancy. Salmonids that rear in small streams and floodplain habitats are particularly vulnerable to the highest expected concentrations. We found limited survival data comparing the various salmonid lifestage (*i.e.*, eggs, fry, smolts, returning jacks, and returning adults) for any of the six a.i.s. We also located no survival data for estuarine or marine salmonid

life stages. The vast majority of lethality data is based on standard toxicity laboratory tests conducted with juvenile salmonids (predominantly rainbow trout) that determine the LC₅₀. These data show the six a.i.s have a wide range of LC₅₀s, and the salmonid species tended to be among the most sensitive of the freshwater fish species tested. We relied on these data to evaluate whether expected concentrations of the six a.i.s are sufficient to kill individual salmonids.

Of the chemicals assessed, the two fungicides are the most toxic based on salmonid survival data. They are classified as very highly toxic⁸ based on LC₅₀ ranges of 26.2-137 µg/L and 10.5-42.7 µg/L for captan and chlorothalonil, respectively. We expect concentrations of these fungicides will reach lethal levels based on the range of toxicity and exposure values derived from monitoring data, EPA's modeling estimates, and NMFS modeling estimates. Model estimates indicate concentrations of captan may reach lethal levels in some aquatic habitats associated with turf, and to a lesser degree agricultural uses. Chlorothalonil is expected to reach lethal concentrations for various agricultural, forest, and residential/urban. Although infrequently detected with ambient monitoring, targeted monitoring show chlorothalonil concentrations as high as 48 µg/L in surface water (King and Balogh 2010).

Of the herbicides assessed, triclopyr BEE (LC₅₀ 470 µg/L) is highly toxic to salmonids¹, while 2,4-D esters (LC₅₀ 450-14,500 µg/L) and diuron (LC₅₀ 710-23,800 µg/L) show variable results that range from highly toxic to slightly toxic. We expect concentrations of triclopyr BEE and diuron to reach lethal levels in some habitats based on estimated concentrations associated with most registered uses. 2,4-D esters may also reach lethal concentration in shallow floodplain habitats when pesticides are applied in close proximity and when applied directly to water for aquatic weed control (2,4-D Master

⁸ EPA uses a descriptive scale for acute aquatic effects: very highly toxic (LC₅₀ <100 µg/L), highly toxic (LC₅₀ 100-1,000 µg/L), moderately toxic (LC₅₀ >1,000-10,000 µg/L), slightly toxic (LC₅₀ >10,000-100,000 µg/L), and practically non-toxic (LC₅₀ >100,000 µg/L), as published in Kamrin 1997.

Label indicates a target treatment rate of 4,000 µg/L for butoxyethanol ester products). However, non-ester forms of 2,4-D are not expected to cause acute lethality in fish as LC_{50s} are >162,000 µg/L. The evidence supports evaluating population level consequences from reductions in salmonid survival for triclopyr BEE, 2,4-D and diuron.

Linuron is moderately toxic¹ to salmonids (3,000 µg/L). However, neither exposure estimates nor monitoring data suggest linuron will reach concentrations that are lethal to salmonids.

We expect concentrations of some of the a.i.s in salmonid floodplain habitats will reach lethal levels based on exposure concentrations derived from monitoring data, EPA's modeling estimates, and NMFS modeling estimates (See *Exposure Analysis*). The youngest swimming salmonids appear to be the most likely to die from short-term, acutely toxic exposures in these habitats. It is less likely that adults would be killed by acute concentrations in most freshwater aquatic habitats compared to juveniles. However, if adults are present in smaller floodplain habitats during spray applications or severe runoff events pre-spawn mortality is possible. Additionally, direct applications of 2,4-D butoxyethanol ester to aquatic habitats may also result in sufficient concentrations to kill adult fish. However, we recognize some states have aquatic weed control programs in place requiring spray timing when there is a lower risk of salmon co-occurrence that reduces the potential for this outcome. Other formulations and uses of 2,4-D pose less risk. The available monitoring data, if representative of salmonid habitats, indicated that concentrations rarely achieve LC₅₀ values for most of compounds in freshwaters. However, the monitoring data do not reflect treatment rates for aquatic weed control and it is unlikely that they reflect peak concentrations for other registered uses. As described in the *Exposure Analysis*, monitoring data are limited when compared to the range of habitats used by salmonids. Few data were found that targeted applications and subsequent concentrations in edge of field habitats which typically show much higher concentrations than weekly, monthly, or seasonal monitoring efforts.

In conclusion, the available information on measured and expected concentrations of the a.i.s supports this risk hypothesis for 2,4-D, triclopyr BEE, diuron, captan, and chlorothalonil, but not for linuron.

B. Reduce salmonid survival through impacts to growth or development.

Fish growth can be affected by pesticides in two ways: by a reduction in somatic processes and by behavior modifications that reduce foraging (NMFS 2008; NMFS 2009; NMFS 2010). Salmonids are at the greatest risk of reduced growth from pesticide exposure during their fry to smolt life stage where rapid growth is needed in order to survive. The longer salmonids remain in freshwater the greater the probability for pesticide exposure. Juveniles rearing in estuaries and nearshore environments are also susceptible to growth impacts. For most of the listed salmonid species, but especially stream-type Chinook and coho, extended periods of growth occur in shallow, low-flow habitats, including floodplain habitats and small streams. Time to first feeding is a critical lifestage transition period for all salmonids. Following the adsorption of the yolk sac, fry need adequate prey upon which to feed and the ability to capture them before the onset of starvation.

We did not identify any studies conducted with the six a.i.s that provided a quantitative relationship between growth and fish survival in the lab or field. However, there is abundant ecological literature showing smaller salmonids have reduced first year survival (discussed in *Appendix I*).

Information to identify a specific threshold for growth impacts of diuron and linuron in fish were not provided or found in the open literature and are consequently highly uncertain. However, the estimated chronic thresholds for standardized toxicity studies that include growth and other endpoints fall well within the expected concentration ranges for these two compounds. Additionally, diuron and linuron are relatively persistent in the aquatic environment. Therefore, we assume impacts to growth may occur. Laboratory studies with 2,4-D, triclopyr, captan, and chlorothalonil either show

no significant effects to growth, or impacts at concentrations that are not expected to occur or persist in aquatic habitats occupied by listed salmonids. Therefore impacts to somatic growth from exposure to 2,4-D, triclopyr, captan, and chlorothalonil are less likely.

One mechanism of reproductive and developmental impairment is through endocrine disruption. However, among the six a.i.s this mechanism has only been evaluated for linuron. As discussed in the *Response* section above, linuron acts as an androgen receptor agonist. Additionally, we expect that salmonid exposure to linuron will reach concentrations that have been shown to cause endocrine-related effects in fish. We did not find any studies evaluating the reproductive and developmental endpoints of linuron in fish. However, the information presented by EPA suggests that these endpoints may be impaired at expected environmental concentrations. Endocrine disruption has not been evaluated in diuron but it may result in similar responses to linuron given structural similarities. Laboratory studies show that early life stages of fish exposed to environmentally relevant concentrations of diuron suffer increases in developmental abnormalities and death. A number of reproductive or developmental effects have been seen with 2,4-D, captan, and chlorothalonil. Chlorothalonil exposure to fish has been shown to reduce hatching success and survival at concentrations that are expected and have been measured in surface water. Developmental effects observed with 2,4-D (decreased larval survival) and captan (growth, larval and juvenile development) in fish are considered to be less likely given estimated and measured environmental concentrations.

The weight of evidence supports the conclusion that fitness level consequences from reduced size and/or impaired development is likely in rearing salmonids exposed to diuron, linuron, and chlorothalonil. Growth and developmental effects are less likely to occur from the uses of 2,4-D, triclopyr BEE, and captan.

C. Reduce salmonid growth through impacts on the availability and quantity of salmonid prey

This hypothesis focuses on rearing juveniles and the amount of prey available to ensure adequate growth and ultimately, size. As mentioned previously, habitats most vulnerable to pesticide contamination are shallow, low flow habitats where salmonids congregate to feed on a variety of terrestrial and aquatic invertebrates. Other aquatic habitats used by rearing salmonids are also vulnerable to reductions in prey, including channel edges along larger streams, rivers, estuaries, and nearshore marine areas.

We address several lines of evidence to determine the likelihood of reduced salmonid growth from impacts to aquatic invertebrate prey. The first line of evidence we evaluated is whether salmonid prey items are sensitive to acute and chronic exposures from expected concentrations of the six a.i.s. This primarily involved evaluating laboratory experimental results reporting on acute toxicity of the pesticides to aquatic invertebrates. Survival estimates were available for all of the active ingredients but we found primarily data with standard cladoceran test species for most of the a.i.s. Diuron and chlorothalonil were exceptions to this pattern. We reviewed diuron toxicity data for a range of relevant taxa including cladocerans, an amphipod, isopod, and a stonefly. We expect initial concentrations of diuron will be sufficient to kill the more sensitive species in some habitats. Additionally, surface water concentrations may remain toxic to these species for extended durations where flow or recharge does not play a significant role in dissipation. We expect a similar pattern for chlorothalonil. The expected peak concentrations of chlorothalonil are sufficient to kill aquatic invertebrates based on the available standardized acute toxicity information with *Daphnia magna*. Tests with other taxa suggest predicted chronic exposures to chlorothalonil may be sufficient to kill other taxa evaluated including an isopod, amphipod, crayfish, and a shrimp with EC₅₀s ranging from 3.6 µg/L to >40 µg/L for 4 – 7 d exposure durations. We also expect initial concentrations of triclopyr BEE and linuron to be sufficient to kill aquatic invertebrates in some habitats. The environmental fate parameters of triclopyr suggest receiving waters will not remain toxic to invertebrates for extended durations. Linuron is more persistent although it can

be applied to relatively limited number of use sites. Although only moderately toxic to aquatic invertebrates, use of 2,4-D butoxyethanol ester at the treatment rate of 4,000 µg/L to control aquatic weeds is a concentration that exceeds aquatic invertebrate EC₅₀s and is thus expected to reduce the availability of aquatic invertebrates. However, we do not expect other uses and other forms of 2,4-D to impact the availability of salmonid prey through direct toxicity. Impact to salmonid prey from captan exposure also appear less likely given the available toxicity and exposure information.

The second line of evidence evaluated is whether field-level reductions in aquatic invertebrates correlate to use of the six pesticides addressed in this Opinion. Data to assess this line of evidence were lacking. Although we located few microcosm or community studies for the pesticides evaluated in this Opinion, data from other pesticides show shifts in benthic communities - from sensitive mayfly, stonefly and caddisfly taxa, the preferred prey of salmonids, to worms and midges, occur in areas with degraded water quality (Cuffney, Meador et al. 1997; Hall, Killen et al. 2006). Recovery of salmonid prey communities following acute and chronic exposures to pesticides depends on the organisms' sensitivity, life stage, and length of life cycle, among other characteristics. Univoltine species will take longer than multivoltine species to recover (Liess and Schulz 1999). Recovery of high quality salmonid prey items such as caddisflies, stoneflies, and mayflies will be slow, given they have long life cycles and infrequent reproduction.

The third line of evidence we evaluated was whether salmonids showed reduced growth in areas of low prey availability. An evaluation of this line is complicated by multiple factors affecting habitat quality (*e.g.*, water quantity, quality, temperature, riparian zone condition, *etc.*), which in turn affects prey items and salmonids. We were unable to locate information attributing reduced growth in salmonids to prey reduction caused by specific exposure to the six pesticides, as most studies focused on measuring direct effects on salmonids or direct effects on invertebrates. However, there are multiple field experiments and studies that demonstrate reduced fish growth resulting from reduced

prey availability (Brazner and Kline 1990; Metcalfe, Fraser et al. 1999; Baxter, Fresh et al. 2007) or document fish growth rates below maximal potential growth rates when prey are limited (Dineen, Harrison et al. 2007).

Collectively, the lines of evidence support the overall hypothesis that 2,4-D, triclopyr BEE, diuron, linuron, and chlorothalonil can reduce salmonid growth through impacts to the availability of prey. The evidence does not support this hypothesis for uses of captan.

D. Reduce survival, migration, and reproduction through impacts to olfactory-mediated behaviors.

Pacific salmonids rely on olfaction to sense environmental cues that facilitate success in mating, locating food, migration, homing, and avoiding predation. Several classes of pesticides, including herbicides and fungicides, are known to impair olfaction in fish and several studies have shown that pesticides and other contaminants can disrupt olfactory processes that are important for survival and reproduction (Tierney, Baldwin et al. 2010). We located studies that evaluated Pacific salmon olfactory response to 2,4-D, chlorothalonil and linuron. Reduced fitness from olfactory-mediated behaviors associated with exposure to 2,4-D and chlorothalonil appear unlikely given the available toxicity and exposure estimates (Tierney, Ross et al. 2006). However, we expect concentrations of linuron will reach levels sufficient to impair olfactory responses in salmon. Linuron reduced salmonid response to the odorant L-serine 50-80% following a 30 minute exposure, and in more sensitive species response was significantly inhibited within 15 minutes (Tierney, Ross et al. 2007). Reduction in L-serine responses suggest reduced fitness through diminished predator avoidance, food location, and imprinting (Tierney, Baldwin et al. 2010). Additionally, environmentally realistic concentrations of a mixture of linuron and other pesticides compromised juvenile steelhead's ability to detect changes in odorant concentrations (Tierney, Sampson et al. 2008). Considering that data are unavailable for diuron while data with linuron support this hypothesis, and giving the benefit of the doubt to the species, NMFS assumes that these similar responses will also occur when salmonids are exposed to diuron.

We located no studies directly measuring olfactory-mediated behavioral responses of fish following exposures to triclopyr or captan. This recognized data gap introduces uncertainty as to whether these a.i.s impair olfaction and, if so, at what concentrations effects might occur.

The weight of evidence supports olfactory-mediated behaviors will be impaired from exposures to linuron and diuron and will reduce fitness for individuals. Impairment of olfactory response is less likely or more uncertain for 2,4-D, triclopyr BEE, captan, and chlorothalonil.

2. Exposure to 2,4-D, triclopyr BEE, diuron, and linuron is sufficient to:

A. reduce aquatic primary producers thereby affecting salmonid prey communities and salmon

Plants and other primary producers are an essential component of productive salmonid habitats because they provide food resources to aquatic invertebrates and provide shelter for invertebrates and fish. Ecosystem studies show that herbicides have variable effects following reductions in primary producers, however impacts to fish through trophic level interactions can occur, particularly in systems that are dominated by sensitive plants. We expect 2,4-D, triclopyr BEE, diuron, and linuron will reduce primary producers and have fitness consequences for some salmon given the toxicity information available on ecological responses and the estimated and measured concentrations of these herbicides. However, the response will depend on site-specific conditions and are difficult to predict. In part, the response is dependent on the relative tolerance of the existing primary producers and the corresponding shifts that can occur to more tolerant and potentially less desirable prey species. Ecosystem recovery may occur relatively rapidly where there is functional redundancy with more tolerant species. However, recovery of invertebrates and fish may be prolonged in situations where the dominant vascular plants are sensitive. Diuron and linuron are photosynthetic inhibitors and broad spectrum herbicides that are toxic to broad range of primary producers. 2,4-D and triclopyr BEE are auxin-simulators

and vascular plants and other macrophytes tend to be more sensitive than other primary producers like algae. Collectively, the available lines of evidence support the overall hypothesis that the four herbicides cause adverse affects to salmonids and their habitat through reductions in primary production, and subsequent reductions in the availability of prey, particularly for the uses of diuron, linuron, and triclopyr BEE. For 2,4-D, direct applications to macrophyte-dominated systems are most likely to cause changes to the abundance and composition of primary producers, and consequently changes in the quality and quantity of salmonid prey. But as discussed above, whether or not these responses are detrimental will depend on site-specific conditions. This hypothesis is also supported for some terrestrial applications of 2,4-D immediately adjacent to shallow habitats. Other uses of 2,4-D are less likely to reduce primary producers in salmonid habitats.

B. reduce riparian vegetation to such an extent that stream temperatures are elevated, erosion increases, and reduction in inputs of woody debris and other organic matter occurs.

This risk hypothesis considers aquatic habitat changes due to potential herbicide impacts to riparian vegetation. Possible changes to salmonid habitat associated with modifications to the riparian zone include alterations in terrestrial input of organic matter (including leaf litter, woody debris, and terrestrial insects); increased input of contaminants due to decreased vegetative filtering; reduced maintenance of natural flow dynamics; decreased bank stability and associated increased erosion and sedimentation; and decreased shading and increased stream temperatures. We are not aware of any studies that specifically evaluated aquatic habitat responses that may correspond with changes to the riparian habitat from these four herbicides. However, we expect 2,4-D, triclopyr, diuron, and linuron will alter riparian vegetation either from direct application to riparian habitats or, to a lesser extent, from transport of the herbicides to riparian habitats. 2,4-D, triclopyr BEE, diuron, and linuron are all systemic herbicides that can be taken up by the roots or leaves of plants and can be spray applied. All four herbicides may be transported to riparian habitats through drift and runoff. 2,4-D and triclopyr BEE are selective herbicides expected to reduce woody vegetation and broadleaf plants.

Diuron and linuron are non-selective (broad spectrum) herbicides for control of broadleaf plants, grasses, algae, and moss. Given their broad spectrum action on plants, riparian zone impacts associated with applications of diuron and linuron to sites adjacent to the riparian habitat are expected to be more severe than those of 2,4-D and triclopyr BEE. However, riparian zone impacts are most likely to occur from direct applications of herbicides to those habitats and neither diuron or linuron are permitted for riparian habitat use. 2,4-D and triclopyr BEE will modify riparian habitats when these chemicals are applied directly to riparian zones. However, changes to the riparian zone habitat may or may not result in negative responses to salmonid habitat. Removal of woody vegetation can reduce shading and consequently increase stream temperatures, it can also decrease bank stability which may result in increased erosion and sedimentation detrimental to salmonids, and reduce inputs of organic material that are beneficial to the aquatic community. Alternatively, removal of woody vegetation could increase growth of understory vegetation and improve the contaminant filter capacity of the riparian zone which may be beneficial. Whether or not riparian zone applications of 2,4-D and triclopyr BEE are beneficial or detrimental to the aquatic habitat and salmonid will depend on site specific conditions. Overall, the weight of evidence suggests that 2,4-D, triclopyr, diuron, and linuron can cause alterations to riparian zones that may result in adverse effects to aquatic communities that impact the individual fitness of salmon.

3. Exposure to mixtures of the diuron and linuron can act in combination to increase adverse effects to salmonids and salmonid habitat.

We are not aware of any data that directly assess mixtures of diuron and linuron. However, these compounds are structurally very similar, have a common mode of action, and produce several common degradates. Therefore, we reasonably assume they can act in combination to increase adverse effects to salmonids and salmonid habitat.

4. Exposure to other stressors of the action including degradates, additional active ingredients, and inert/other ingredients in pesticide products and tank mixes cause adverse effects to salmonids and their habitat.

In addition to exposure to the a.i.s, salmonids and their habitat are likely exposed to other stressors of the action, including degradates and additional active ingredients in formulated products and tank mixes. Salmonid habitats may also be exposed to a number of the approximately 4,000 inert ingredients approved for use in end-use pesticide products by EPA, as well as adjuvants, such as surfactants and other products that are applied as tank mixtures. Once the mixture (formulated pesticide or tank mix) is introduced into the environment, physico-chemical properties of the various compounds will cause them to move through the environment at different rates and partition into different compartments. We expect some percentage of these other stressors will be present in salmonid habitats from spray drift deposition, and from runoff events following application. Salmon and their habitats exposed to these multiple stressors are expected to show a greater response than laboratory animals exposed only to one a.i, thus available toxicity data generally underestimate the response in a field-applied pesticide mixture.

4.a. Exposure to degradates of 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil

The BEs identify many of the degradates of the six a.i.s. However, estimates quantifying potential exposure of listed salmonids and their habitat to these transformation products were not provided. Information on the toxicity of these compounds was reported for few of the degradates and remains a considerable source of uncertainty. We found little information to supplement the BEs. Exposure to some of the environmental degradates may increase adverse effects to salmonids and their habitat. Although toxicity and exposure data is limited for many of these degradates, some data suggest these stressors can add substantial risks. For example, a mesocosm with a 3,4-dichloroaniline (DCA) suggests this degradate of both diuron and linuron may cause persistent toxicity and ecological effects (Maund, Biggs et al. 2009). In general, exposure to degradates of the a.i. is expected to increase adverse effects to salmonids and their habitat.

4.b. Additional Active Ingredients

As discussed in the *Mixture Analysis* section above, pesticide products containing multiple a.i.s are common. While the a.i.s will move through the environment at different rates, it is reasonable to believe that all a.i.s in a given pesticide formulation will co-occur in receiving waters, especially from drift deposition and in the first runoff from the field following application. Examples of some of the formulations mixtures include mixtures of up to 4 herbicides or fungicides. The potential interactive relationships that exist among most of these a.i. combinations have not been defined and are uncertain. Others are likely to cause additive or synergistic effects compared to the single a.i. alone. For example, diuron and linuron likely cause additive effects in plants given their similar structure and mode of action (EPA Reg. No. 352-660). Another product contains captan and two cholinesterase inhibitors, malathion and carbaryl. Some of these a.i.s may be more acutely toxic than the a.i.s specifically considered in this Opinion, although they may have a different mode of action. Others may not be as acutely toxic, but may cause reproductive effects, bioaccumulate, or otherwise adversely affect the salmon or their environment in some way.

2,4-D, triclopyr BEE, diuron, linuron, and captan all provide recommendations for tank mixtures that contain additional a.i.s (Table 105). It is reasonable to assume co-location of a.i.s that are co-applied will occur in the aquatic habitats. Specific interactions between additional a.i.s in products and tank mixes and the a.i.s addressed in this Opinion are unknown, but it is reasonable to assume toxicity of the a.i.s may be enhanced. In general, exposure to other active ingredients in pesticide products and tank mixes is expected to increase adverse effects to salmonids and their habitat.

4. c. Inert/other ingredients

In addition to a.i.s, pesticide products contain other ingredients which are sometimes referred to as the inert ingredients. Some of these ingredients are toxic to aquatic organism by themselves, or can increase the toxicity of the active ingredients. As with

tank mixes, the likelihood of these compounds co-occurring in the water column is difficult to determine with any specificity, but can reasonably be presumed to occur in spray drift deposition and runoff following applications. The other ingredients may make up the majority of the pesticide formulation, but few are required to be specifically identified by pesticide labels. One example of these ingredients are the nonylphenol polyethoxylates, which have been linked to endocrine disruption and were addressed at length in previous Opinions on EPA pesticide registrations (NMFS 2008; NMFS 2009; NMFS 2010). There are however, myriad others, some of which may increase the toxicity of the a.i.s. The majority of a pesticide formulation is often composed of inert ingredients. For example, bensulide formulations considered in this Opinion are >90% inert ingredients. Consequently, salmonid exposure to these ingredients may be greater than exposure to the assessed active ingredient. EPA currently has no specific method of accounting for this potential additional toxicity and risk, but it cannot be ignored. NMFS has opted to address the uncertainty associated with these ingredients in a qualitative sense. Collectively, the available lines of evidence support the overall hypothesis that other stressors of the action cause adverse effects to salmonids and their habitat.

From our review of the available information it is not possible to accurately quantify the contribution of other stressors of the action. These stressors include the additional a.i.s and inert/other ingredients in pesticide formulations as well as tank mixes. These stressors of the action are an important consideration when assessing potential effects on listed salmonids and their habitat. Thus, to provide the benefit of the doubt to the species, we assume these stressors of the action will contribute additional, unquantifiable reductions in fitness to individuals beyond that of 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil.

5. Exposure to other pesticides present in the action area can act in combination with the six a.i.s to increase effects to salmonids and their habitat.

The available toxicity and exposure data support the hypothesis. Environmental mixtures of pesticides are common. We found few data evaluating the response of aquatic species to mixtures containing the the six a.i.s. However, toxicity investigations with pesticide

mixtures reveal that responses of aquatic species are variable, and will depend on the composition of the mixture, concentrations of the a.i.s, and duration of exposure. Additionally, sequential exposures from other pesticides in the action area are reasonably expected to increase effects to salmonids and their habitats if when they impact the same environmental receptors. For examples, in addition to 2,4-D, triclopyr BEE, diuron, and linuron there are many other herbicides used in the action area that may further reduce primary production. Therefore, we assume exposure to other pesticides present in the action area will act in combination with the six a.i.s to increase the effect to salmonids and their habitat.

6. Exposure to elevated temperatures can enhance the toxicity of the stressors of the action.

We reviewed the available information to determine whether empirical data indicated enhanced toxicity at elevated temperatures for the a.i.s assessed in this opinion. However, we located only one study, with chlorothalonil, that assessed the influence of temperature on chlorothalonil response (Davies 1985). That study found reported lower LC_{50s} at higher temperatures, although differences in the LC_{50s} were relatively comparable for the range evaluated (10-16°C). Higher water temperatures can increase the metabolic rate for fish, thus increasing the rate at which they process the toxicant. Depending on the chemical, this may be either beneficial or detrimental. Water temperatures higher than optimum also increase general physiological stress for salmonids making them more susceptible to other stressors. Sufficient data to support or refute this hypothesis is lacking. Therefore we did not assume that elevated temperature would enhance the toxicity of the stressors of the action. A summary of effects on individual fitness for each of the a.i.s is presented in Table 137.

Table 137 Summary of individual-based risk hypotheses.

Risk Hypotheses	Is individual fitness of exposed salmonids compromised?					
	2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chloro- thalonil
<i>1.A. Kill salmonids from direct, acute exposure</i>	yes	yes	yes	no	yes	yes
<i>1. B. Reduce salmonid survival through impacts to growth or development</i>	no	no	yes	yes	no	yes
<i>1. C. Reduce salmonid growth through impacts on the availability and quantity of salmonid prey</i>	yes	yes	yes	yes	no	yes
<i>1. D. Reduce survival, migration, and reproduction through impacts to olfactory-mediated behaviors.</i>	no	no	yes	yes	no	no
<i>2.A. Reduce aquatic primary producers thereby affecting salmonid prey communities and salmonids</i>	yes	yes	yes	yes	no	no
<i>2.B. Reduce riparian vegetation to such an extent that stream temperatures are elevated, erosion increases, and reduction in inputs of woody debris and other organic matter occurs</i>	yes	yes	yes	yes	no	no
<i>3. Exposure to mixtures of diuron and linuron can act in combination to increase adverse effects to assessment endpoints</i>	no	no	yes	yes	no	no
<i>4. Exposure to other stressors of the action including degradates, additional a.i.s and inert/other ingredients in formulations and tank mixes cause adverse effects to salmonids and their habitat.</i>	yes	yes	yes	yes	yes	yes
<i>5. Exposure to other pesticides present in the action area can act in combination with the six a.i.s to increase effects to salmonids and their habitat.</i>	yes	yes	yes	yes	yes	yes
<i>6. Exposure to elevated temperatures can enhance the toxicity of the stressors of the action</i>	no	no	no	no	no	no

Risk Characterization: Evaluation of Critical Habitat Risk Hypotheses

We use the toxicity information presented earlier in the *Effects of the Proposed Action* section to evaluate the scientific lines of evidence that support or refute risk hypotheses developed for critical habitats. We determined that freshwater spawning and rearing sites, migration corridors, estuarine areas, and nearshore marine areas within designated critical habitats are likely to be exposed to the stressors of the action over the 15-year registration duration. We will discuss, and when possible, estimate expected concentrations and durations of exposure for these habitats based on pesticide use information, surface water monitoring data, EPA modeling estimates, and NMFS modeling estimates. For each risk hypothesis below we qualitatively weigh the evidence to determine whether the PCE attributes of water quality and/or prey availability are affected. We ultimately determine whether the degradation of water quality and prey availability within freshwater spawning habitat, freshwater rearing habitat, migration corridors, estuarine areas, and nearshore marine areas will rise to the level expected to reduce the intended conservation role of designated critical habitats - evaluated within the *Integration and Synthesis for Designated Critical Habitat* section. The final conclusion of whether EPA's proposed actions are likely to adversely modify or destroy a species' designated critical habitat is provided in the *Conclusion* section.

Risk hypothesis 1. Exposure to the stressors of the action is sufficient to degrade water quality and substrates in freshwater spawning sites.

Freshwater spawning sites require water quality and substrate conditions that support spawning, incubation, and larval development. The degradation of water quality by exposure to the stressors of the action is indicated by the toxic responses in a variety of aquatic organisms including listed salmonids (see section on *Integration of Exposure and Response* above). Based on allowable application timings of the pesticide products, we expect episodes of water quality degradation to coincide with spawning events within spawning habitats. The levels of contamination expected are highly variable resulting from the diversity of species spawning habitats (small, shallow first and second order

streams to mainstem rivers with variable flow patterns) and year-to-year variation in climate and pesticide applications. All six a.i.s are expected to attain concentrations that degrade water quality within spawning PCEs at some point during the 15-year registration period. The most severe effects to water quality within spawning sites will be those sites that experience multiple applications of the a.i.s, are shallow, low flow systems, and are located in high pesticide use areas such as intensive agricultural or urbanized watersheds.

Contamination of spawning sites by the stressors of the action can degrade water quality in several ways. Exposure to the stressors of the action can impair spawning rituals as well as kill spawning adults which diminishes the purposes for which spawning sites were identified as critical habitat. In vulnerable habitat, expected concentrations of 2,4-D, triclopyr BEE, diuron, captan, and chlorothalonil are sufficient to kill a percentage of spawning adults. Other spawners may experience impaired olfaction from exposure to diuron and linuron which leads to reduced ability to detect spawning events. Other ingredients and degradates of the six a.i.s may also result in degraded water quality. Expected concentrations of the four herbicides are sufficient to kill primary producers and reduce primary production. This can result in a number of undesirable water quality conditions including significant increases in nutrient levels, reduced dissolved oxygen, reduced pH, higher alkalinity, and higher turbidity and conductivity (Brock, Lahr et al. 2000). Degradation of substrate is also expected due to likely impacts of the four herbicides to riparian vegetation – including alterations to the make-up of plant species up to complete plant removal (*e.g.* removal of woody species or emergent vegetation). Alterations or removal of riparian vegetation typically results in decreased bank stability, increased erosion, and increased deposition of fine sediments into the stream channel. Salmon require well oxygenated gravel substrates for successful spawning. We expect the use of these herbicides in and around riparian habitats may result in degraded substrates by increasing gravel imbeddedness in freshwater spawning sites.

Collectively, the overlap of spawning sites with application areas combined with expected concentrations and toxicity effect thresholds to aquatic organisms and riparian vegetation indicates that degradation of the water quality of the spawning PCE is likely for the six a.i.s. and adverse effects to substrate are likely for the four herbicides. We evaluate whether the degradation of this PCE, sites for spawning, in combination with other affected PCEs reduce the conservation value of the 26 designated critical habitats within the *Integration and Synthesis for Designated Critical Habitat* section.

Risk hypothesis 2. Exposure to the stressors of the action is sufficient to degrade water quality, natural cover, and /or reduce prey availability in freshwater rearing sites.

Freshwater rearing sites need to provide good water quality, abundant forage, and cover to support juvenile development. Reductions in any of these attributes can limit the existing and potential carrying capacity of rearing sites and subsequently reduce their conservation value. Recovery of listed salmonid populations is tied closely to the success of juveniles to fully develop, mature, and grow during freshwater residency periods. All species of Pacific salmonids spend some amount of time in freshwater feeding and rearing areas. Chum salmon utilize fresh water for the shortest periods (generally a few days). Chinook, coho, steelhead, and sockeye salmon spend much longer periods rearing in freshwater systems with steelhead trout spending up to several years before ocean migration. Freshwater rearing areas are diverse, extensive, complex sites that can range from small, shallow, intermittent floodplain habitats to channel edges of large river systems. As such, expected concentrations range from some of the highest estimates (via spray drift into floodplain habitats) to some of the lowest estimates (monitoring results from large rivers; see Table 106).

Many freshwater salmonid rearing sites are located in floodplains where shallow, low flow habitats are at high risk of pesticide drift and runoff. These habitats provide some of the most important foraging areas for developing juveniles. Expected floodplain concentrations of the six a.i.s are shown in the *Exposure and Response Integration* above. At these levels, water quality would be effected from the six a.i.s based on concentrations

that exceed toxicity thresholds for lethality to fish (all except linuron), lethality to aquatic invertebrates (all except captan), impacts to olfactory senses in fish (diuron and linuron), cellular damage in fish (captan and chlorothalonil), and damage to plankton and aquatic macrophytes (2,4-D, triclopyr BEE, diuron, and linuron). As discussed above in *Risk Hypothesis 1*, a number of water quality parameters (*e.g.* nutrient concentrations, DO, pH, *etc.*) will be modified by the instream concentrations of the four herbicides in floodplain habitats. Additionally, we expect herbicide modification to riparian will result in decreased shading of these habitats and an increase in the occurrence of temperatures that are stressful to salmonids in freshwater rearing sites.

Triclopyr BEE, diuron, linuron, and chlorothalonil are expected to reach levels that reduce prey abundance; the laboratory toxicity data and exposure estimates indicate that salmonid prey would be severely affected if the lower end of the survival range (shown in the *Exposure and Response Integration* above) is representative of salmonid prey communities. Direct application of 2,4-D butoxyethanol ester at the maximum labeled rate of 4,000 µg/L is also expected to reduce prey through direct toxicity, while other uses of 2,4-D are not. We located few data on the response of real world prey communities to the six a.i.s. We assume that many of salmonid prey items are either as sensitive as or more sensitive than the standard toxicity test organisms for which data exist. This assumption is supported, in part, by comparative data showing that caddisflies, stoneflies, and mayflies are typically more susceptible to pesticide toxicity than laboratory-reared freshwater invertebrates (Peterson, Jepson et al. 2001; Johnson, Jepson et al. 2008). The IBI and other metrics of aquatic community health were reviewed. In areas of intensive agriculture, where we expect use of the stressors of the action, biological integrity is often significantly reduced (Cuffney, Meador et al. 1997). Many of the preferred salmonid prey items are present only in low numbers or absent altogether in these areas. We see similar depauperate communities in urban areas. We recognize many other limiting factors contribute to the poor condition of these aquatic communities. However, these six a.i.s and their formulations may be responsible for a portion of these reductions.

We also expect that in some situations, the herbicidal action of 2,4-D, triclopyr BEE, diuron, and linuron will alter characteristics of the plant communities in freshwater rearing sites decreasing the availability of food or cover for salmonids and their prey. Natural cover that would reasonably be reduced from these four herbicides includes shade and aquatic vegetation (based on exposure and toxicity to aquatic and terrestrial plants). In cases where woody vegetation is eliminated from riparian zones to the extent that bank destabilization occurs, altered stream hydrology could affect the availability of other cover including rocks, side channels, and undercut banks. One extreme example, suggested the application of an herbicide to a riparian zone caused major long-term changes to the hydrology of a stream and degraded the fish habitat (<http://water.epa.gov/scitech/datait/tools/warsss/streamero.cfm>). Herbicide-induced changes to vegetative communities in the riparian zone and aquatic habitat also have implications for the availability of prey as salmonids consume both terrestrial and aquatic insects. As plant communities are modified in riparian zones and aquatic habitats the species that rely on them will also be affected. Habitat responses in these environments are expected to be variable and will depend on the sensitivity of existing plants to 2,4-D, triclopyr BEE, diuron, and linuron. Studies of herbicide responses in test ecosystems indicates the selection pressure from exposure to herbicides can result in reductions to primary production and the composition of primary producers that can adversely impact higher trophic levels. Brock et al. (2000) concluded in their review of herbicides that indirect effects of photosynthetic inhibitors (e.g. diuron and linuron) on consumers and predators occur at concentrations around the EC₅₀ for standard algae taxa; We expect these impacts to occur in freshwater rearing areas given the available exposure and toxicity information. We also expect other effects on the ecosystem such as blooms of insensitive algae that can occur at lower concentrations (e.g. 0.1 of the EC₅₀ of standard algae), and can be a delayed response. The Brock et al. (2000) review of ecosystem impacts indicates that the most dramatic ecological responses may occur with auxin-simulators (e.g. 2,4-D and triclopyr) because aquatic macrophytes are more sensitive than algae. Vascular plants and other macrophytes are important structural elements that

provide important cover to salmonids in their preferred freshwater rearing areas. Aquatic vegetation provides cover to salmonids from avian predation as well as protection from larger fish. In addition, a diverse vascular plant community provides important substrate for an array of insect species upon which young salmon prey. A shift from a macrophyte-dominated community to a plankton-dominated community will reduce the natural cover PCE attribute in freshwater rearing areas and “may lead to drastic shifts in the aquatic community by habitat destruction. The organisms for which aquatic plants play an important role include periphyton, crustaceans, aquatic insects (especially larvae), mollusks, but also certain fish and tadpole species. All these groups may disappear or decrease in numbers as a result of mortality of macrophytes... the disappearance of the vegetation may in some cases lead to a bloom of phytoplankton (Brock, Lahr et al. 2000).” Although such shifts can lead to a temporary increase in primary production that are advantageous to some species, it is expected to result in considerable indirect effects to macrophyte-associated cover and prey resources important to salmonids in freshwater rearing sites. Reductions and removal of cover and prey biomass in floodplain and other habitats that support rearing may substantially reduce this PCE’s role in recovering salmonid populations. Concentrations of the four herbicides in other freshwater habitats that support rearing are also expected to reach levels that reduce both water quality and prey abundance.

Collectively, substantial data indicated that expected concentrations of the six a.i.s are sufficient to adversely affect water quality; 2,4-D, triclopyr BEE, diuron, linuron, and chlorothalonil are sufficient to adversely affect salmonid prey (forage); and concentrations of the four herbicides are expected to degrade the natural cover attribute of freshwater rearing PCEs. Therefore, we evaluate these effects in order to determine whether the conservation value of species’ designated critical habitats will be reduced (see *Integration and Synthesis for Designated Critical Habitat* section below).

Risk hypothesis 3. Exposure to the stressors of the action is sufficient to degrade water quality, natural cover, and/or reduce prey availability in freshwater migration corridors.

Freshwater migration corridors require good water quality, natural cover, and sufficient prey abundance to support juvenile and adult mobility and survival. Contaminating these sites with the stressors of the action degrades water quality, reduces cover, and/or further impedes the mobility and survival of juveniles and adults. Expected contaminant concentrations associated with the use of products containing 2,4-D, triclopyr BEE, diuron, linuron, and chlorothalonil may limit prey availability in migratory sites where juveniles pause to rest and feed during their migration to the ocean. Additionally, exposure of riparian and aquatic habitats to the four herbicides is may reduce the availability of natural cover such as shading and aquatic vegetation. Frequently rest areas such as undercut banks, side channels, submerged and overhanging large wood, log jams, and beaver dams are far and few between in many West Coast salmonid producing streams and rivers. Salmonid recovery plans call for restoration of these sites to improve juvenile survival and overall fitness. Lack of adequate natural cover or prey resources due to the degradation of water quality at these rest areas may cause migrating juveniles to continue downstream thus avoiding needed rest and food, ultimately affecting their health and ability to successfully transition to saltwater environments. Many of these rest areas are located in places where water flow is reduced compared to the main channels. Stressors of the action may persist longer in these areas due to reduced flow.

Additionally, channel-edge habitats that are proximate to applications of the stressors of the action are at risk, increasing the probability of exposure to high concentrations from drift and runoff following application events. Many migratory sites overlap with some of the highest use areas for the stressors of the action such as intensive agricultural valleys. Based on the size, flow rate, and proximity to application sites, exposure durations and concentrations within migratory habitats are expected to be highly variable. That said, we expect uses of the six a.i.s will at times lead to concentrations that will reduce salmonid prey, natural cover, and/or degrade water. The degradation of water quality within migratory sites may affect the mobility of juveniles and adults exposed to diuron

and linuron by impairing their olfaction. The 6 a.i.s are expected to degrade water quality and the four herbicides are expected to degrade natural cover.

Collectively, the available data indicated that expected concentrations of the six a.i.s are sufficient to adversely affect water quality; concentrations of 2,4-D, triclopyr BEE, diuron, linuron, and chlorothalonil are sufficient to reduce the availability of salmonid prey (forage); and concentrations of the four herbicides are sufficient to reduce the availability of natural cover of migratory PCEs. Therefore, we evaluate these effects in order to determine whether the conservation value of species' designated critical habitats will be reduced (See *Integration and Synthesis for Designated Critical Habitat* section below).

Risk hypothesis 4. Exposure to the stressors of the action is sufficient to degrade water quality, natural cover, and /or reduce prey availability in estuarine areas.

Estuarine areas require good water quality to support juvenile and adult physiological transitions between fresh water and salt water as well as to provide juvenile and adult cover and prey resources sufficient to support survival, growth, and maturation. Prey resources for Pacific salmonids within estuaries include a diverse group of organisms - from aquatic invertebrates to small fishes depending on the size of the salmonid. The allowable uses of the stressors of the action overlap with estuaries designated as critical habitat.

Contamination of estuaries occurs via drift, runoff, and atmospheric deposition. Streams and rivers flowing into estuaries act as conveyor belts as they transport the stressors of the action from areas higher in watersheds (Johnson, Grant et al. 1997). We located no estuarine monitoring data specific to the stressors of the action. This is a large data gap as the available exposure data derived for freshwater habitats (EPA modeling estimates, NMFS modeling estimates and monitoring data) are not representative of estuarine habitats. Pacific estuaries are incredibly variable to one another; size, tidal volume, exchange rate, freshwater input, salinity, watershed land uses, trophic structures,

bathymetry, etc., influence and shape estuarine ecosystems (Salo 1991). Estuaries remain dynamic, complex systems that are not completely understood. As such, predictive models are not available to estimate concentrations of pesticides within estuaries. Therefore, we evaluate whether applications of the stressors of the action are allowed within estuarine-containing watersheds and if so, we assume they may contaminate estuarine habitats. Dissipation rates are another source of uncertainty that remains an active area of research. All of the a.i.s are allowed for use in estuarine-containing watersheds.

The available toxicity information for estuarine and marine organisms for the a.i.s is presented in Table 138. The majority of aquatic toxicity data are from survival assays for the sheepshead minnow (fish) and mysid (estuarine invertebrate). The available studies indicate a similar range of sensitivity to the six a.i. between the marine and freshwater species test species. Consequently, we expect similar ecological response when exposure is comparable.

Table 138. Assessment endpoint toxicity values (µg/L) for saltwater aquatic organisms presented in salmonids BEs.

Assessment endpoint	Assessment measure	Range
2,4-D (all forms)		
Fish Survival	estuarine, and marine fish LC ₅₀ (96 h)	3,200 – 465,000
Fish reproduction or larval survival	NOEC/LOEC	56 – 79
Invertebrate survival	estuarine and marine EC ₅₀	1,800 – 401,800
Aquatic primary producer	EC ₅₀	660 - 123,300
	NOEC	538 - 80,000
Triclopyr BEE		
Fish Survival	estuarine, and marine fish LC ₅₀ (96 h)	450 - 760

Assessment endpoint	Assessment measure	Range
Fish reproduction or larval survival	NOEC/LOEC	no studies reported
Invertebrate survival	estuarine and marine EC ₅₀	320 – 2,470
Aquatic primary producer	EC ₅₀	1,170
	NOEC	209
Diuron		
Fish Survival	estuarine, and marine fish LC ₅₀ (96 h)	6,300 – 6,700
Fish reproduction or larval survival	NOEC/LOEC	<440
Invertebrate survival	estuarine and marine EC ₅₀	1,100 – 5,000
Aquatic primary producer	EC ₅₀	no studies reported
	NOEC	
Linuron		
Fish Survival	estuarine, and marine fish LC ₅₀ (96 h)	890
Fish reproduction or larval survival	NOEC/LOEC	no studies reported
Invertebrate survival	estuarine and marine EC ₅₀	890 – 5,400
Aquatic primary producer	EC ₅₀	no studies reported
	NOEC	
Captan		
Fish Survival	estuarine, and marine fish LC ₅₀ (96 h)	1,900
Fish reproduction or larval survival	NOEC/LOEC	no studies reported
Invertebrate survival	estuarine and marine EC ₅₀	no studies reported
Aquatic primary producer	EC ₅₀	no studies reported
	NOEC	
Chlorothalonil		

Assessment endpoint	Assessment measure	Range
Fish Survival	estuarine, and marine fish LC ₅₀ (96 h)	32
Fish reproduction or larval survival	NOEC/LOEC	no studies reported
Invertebrate survival	Estuarine and marine EC ₅₀	3.6 - 320
Aquatic primary producer	EC ₅₀	11

Collectively, the exposure and toxicity information supports that degradation of water quality is expected for 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil. Prey resources for juveniles may also be reduced from pulses of the stressors of the action associated with the use of 2,4-D, triclopyr BEE, diuron, linuron, and chlorothalonil in high risk areas such as distributary channels in tidal flats and channels draining diked agricultural areas where the pesticide products are applied. Applications of the four herbicides adjacent to estuarine habitats are also expected to reduce the available cover, particularly in cases where transport occurs during periods of low flow and neap tides. We discuss the potential for these stressors to reduce the conservation value of estuarine habitats within the *Integration and Synthesis for Designated Critical Habitat* section.

Risk hypothesis 5. Exposure to the stressors of the action is sufficient to degrade water quality, natural cover, and /or reduce prey availability in nearshore marine areas.

Nearshore marine areas require water quality conditions and forage, including aquatic invertebrates and fishes to support growth and maturation. Similar to estuarine sites, nearshore marine sites have very few data on the behavior and dissipation rates of the stressors of the action. More information, yet still insufficient to make a definitive conclusion, is available on the toxicity to a few saltwater organisms. The available acute toxicity information shows that the stressors of the action can kill and reduce growth of marine flora and fauna and have comparable sensitivity to freshwater organisms.

There is no doubt that the stressors of the action contaminate nearshore environments based on allowable uses. However, significant uncertainty arises to the persistence and rate of degradation of the stressors. Fundamental environmental fate data are lacking, not to mention experimental results from environmental realistic exposure scenarios for key salmonid prey taxa including small, forage fish. We found no information on the environmental fate of these materials in nearshore marine habitats. Whether and how frequently the stressors of the action attain toxic levels for sufficient durations within nearshore marine environments remains unknown. We anticipate some level of degradation of water quality of these habitats, particularly for those species with nearshore marine areas of designated critical habitat within Puget Sound. In Puget Sound we expect the greatest deposition and loading from allowable applications as compared to other nearshore marine areas along the California, Oregon, and Washington coasts because of the longer residence time of the water, proximity to right of way uses, stormwater discharges, and inflow from the numerous rivers. For this reason, we discuss effects to the attributes in nearshore marine areas within the *Integration and Synthesis for Designated Critical Habitat* section by evaluating land uses proximate to these habitats.

Summary of the Effects of the Action on PCEs:

We conclude that the available information on exposure and response of aquatic habitats to the stressors of the action supports each of the five risk hypotheses. We expect essential physical and biological features will be reduced in spawning, rearing, migratory, estuarine, and nearshore marine habitats. Next, within the *Integration and Synthesis of Effects to Designated Critical Habitat* section, we evaluate whether these adverse changes to PCEs affect the conservation value of designated critical habitat.

Conclusions Regarding Risk Associated with Specific a.i.s

Below we describe the risk posed by each of the a.i.s to a generic population of salmonids, if those salmonids are present in one or more waterbodies expected to receive input of that a.i. from registered uses. In the ESU/DPS specific summaries, we determine co-occurrence of the a.i. and the fish based primarily on land use overlap with salmon-

bearing waters. In some cases, use sites may not occur directly adjacent to occupied waterbodies but if there are sufficient use sites upstream to reasonably expect exposure concentrations downstream are of concern, we consider these use sites as well. Due to the physico-chemical properties of the a.i.s addressed in this Opinion, we have focused on direct runoff and spray drift from application sites as the primary drivers in exposure. Although the majority of the a.i.s considered have relatively short half-lives in the environment, and are not considered either persistent or bioaccumulative, we do note there is some atmospheric transport from the application sites to more distant environments. Unlike the three OPs considered in NMFS 2008 Opinion (chlorpyrifos, diazinon, and malathion (NMFS 2008c)), all of which were very toxic to both fish and invertebrates, the a.i.s considered in this Opinion vary widely in toxicity, especially to fish.

NMFS does not use the deterministic RQ/LOC approach used by EPA in evaluating risk to salmonids, nor do we consider only the a.i. Although each a.i. has been evaluated separately in the *Effects Analysis*, we consider all stressors of the action, in addition to other stressors such as elevated temperature and toxic chemicals that may already exist in the environment. For the evaluation of the a.i., we consider the full range of toxicity endpoints including sublethal effects, and the full range of EECs

Table 85. Co-occurrence of listed Pacific salmonids with potential application of pesticides to use sites within the salmon's freshwater distribution. Further discussion on the effects of the individual a.i.s is discussed in the *Intetration and Synthesis for Threatened and Endangered Pacific Salmonids* section below.

Cumulative Effects

Cumulative effects include the effects of future state, tribal, local, or private actions that are reasonably certain to occur in the action area considered by this Opinion. Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

During this consultation, NMFS searched for information on future state, tribal, local, or private actions that were reasonably certain to occur in the action area. NMFS conducted electronic searches of business journals, trade journals, and newspapers using Google and other electronic search engines. Those searches produced reports on projected population growth, commercial and industrial growth, and global warming. Trends described below highlight the effects of population growth on existing populations and habitats for all 28 ESUs/DPSs. Changes in the near-term (five-years; 2016) are more likely to occur than longer-term projects (10-years; 2021). Projections are based upon recognized organizations producing best available information and reasonable rough-trend estimates of change stemming from these data. NMFS analysis provides a snapshot of the effects from these future trends on listed ESUs.

The states of the west coast region, which contribute water to major river systems, are projected to have the most rapid growth of any area in the U.S. within the next few decades. California, Idaho, Oregon, and Washington are forecasted to have double digit increases in population for each decade from 2000 to 2030 (USCB 2005). Overall, the west coast region has a projected population of 72.2 million people in 2010. The U.S. Census Bureau predicts this figure will grow to 76.8 million in 2015 and 81.6 million in 2020.

Although general population growth stems from development of metropolitan areas, growth in the western states is projected from the enlargement of smaller cities rather than from major metropolitan areas. Of the 46 western state metropolitan areas that experienced a 10% growth or greater between 2000 and 2008, only seven have populations greater than one million people. Of these major cities, one and two cities are from Oregon and California, respectively. They include Portland-Vancouver-Beaverton, OR (1.81% per year), Riverside-San Bernadino-Ontario, CA (3.31% per year), and Sacramento-Arden-Arcade-Roseville, CA (2.18% per year) (USCB 2009).

As these cities border coastal or riverine systems, diffuse and extensive growth will increase overall volume of contaminant loading from wastewater treatment plants and sediments from sprawling urban and suburban development into riverine, estuarine, and marine habitats. Urban runoff from impervious surfaces and roadways may also contain oil, heavy metals, PAHs, and other chemical pollutants and flow into state surface waters. Inputs of these point and non-point pollution sources into numerous rivers and their tributaries will affect water quality in available spawning and rearing habitat for salmon. Based on the increase in human population growth, we expect an associated increase in the number of NPDES permits issued and the potential listing of more 303(d) waters with high pollutant concentrations in state surface waters. Continued growth into forested and other natural areas will continue the cycle of altering landscapes to the detriment of salmon habitat. Altered landscapes adversely affect the delivery of sediment and gravel and significantly alter stream hydrology and water quality.

Mining has historically been a major component of western state economies. With national output for metals projected to increase by 4.3% annually, output of western mines should increase markedly (Figueroa and Woods 2007). Increases in mining activity will add to existing significant levels of mining contaminants entering river basins. Given this trend, we expect existing water degradation in many western streams that feed into or provide spawning habitat for threatened and endangered salmonid populations will be exacerbated.

As the western states have large tracts of irrigated agriculture, a 2.2% rise in agricultural output is anticipated (Figueroa and Woods 2007). Impacts from heightened agricultural production will likely result in two negative impacts on listed Pacific salmonids. The first impact is the greater use and application of pesticide, fertilizers, and herbicides and their increased concentrations and entry into freshwater systems. Carbaryl, carbofuran, and methomyl, and other pollutants from agricultural runoff may further degrade existing salmonid habitats. Second, increased output and water diversions for agriculture may also place greater demands upon limited water resources. Water diversions will reduce

flow rates and alter habitat throughout freshwater systems. As water is drawn off, contaminants will become more concentrated in these systems, exacerbating contamination issues in habitats for protected species.

The western states are widely known for scenic and natural beauty, and are used recreationally by residents and tourists. Increases in use could place additional strain on the natural state of park and nature areas that are also occupied by protected species. Hiking, camping, and recreational fishing in these natural areas is unlikely to have any extensive effects on water quality.

The above non-federal actions are likely to pose continuous unquantifiable negative effects on listed salmonids addressed in this Opinion. Each activity has negative effects on water quality. They include increases in sedimentation, increased point and non-point pollution discharges, decreased infiltration of rainwater (leading to decreases in shallow groundwater recharge, decreases in hyporrheic flow, and decreases in summer low flows).

Non-federal actions likely to occur in or near surface waters in the action area may also have beneficial effects on the 28 ESUs. They include implementation of riparian improvement measures, fish habitat restoration projects, and best management practices (*e.g.*, associated with timber harvest, grazing, agricultural activities, urban development, road building, recreational activities, and other non-point source pollution controls).

Coupled with EPA's registration of the a.i.s of the past three recent Opinions which include Opinion 1: chlorpyrifos, malathion, and diazinon; Opinion 2: carbaryl, carbofuran, and methomyl; Opinion 3: azinphos-methyl, dimethoate, phorate, methidathion, naled, methyl parathion, disulfoton, fenamiphos, methamidophos, phosmet, ethoprop, and bensulide; the effects from anthropogenic growth on the natural environment will continue to affect and influence the overall distribution, survival, and recovery of Pacific salmonids in California, Idaho, Oregon, and Washington.

Integration and Synthesis for Threatened and Endangered Pacific Salmonids

The *Integration and Synthesis* section describes NMFS' assessment of the potential for EPA's registration of 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil to reduce the reproduction, numbers or distribution of listed Pacific salmonids, taking into account status of the species, the environmental baseline, and cumulative effects.

In the *Risk Characterization* section we described the effects we anticipate for the salmon themselves due to direct toxicity from the active ingredients. We also described anticipated effects due to exposure to other stressors of the action and through more complex ecologic relationships associated with anticipated modifications to the habitat of listed salmonids and interactions of multiple stressors. In this section we analyze the likelihood of effects described in the *Risk Characterization* within the context of species-specific considerations discussed in the *Environmental Baseline* and *Status of Listed Resources* sections. We also consider the ESU/DPS-specific life history characteristics and distribution of pesticide use sites within their watersheds to determine the likelihood of exposure and effects. This is accomplished by considering co-occurrence in both time (detailed in *Appendix 6*) and space (*Appendix 5*) of use sites, as shown (Table 85).

Although we have not done any population modeling specifically for this Opinion, we have in Opinions addressing other a.i.s (NMFS 2008, NMFS 2009, NMFS 2010). In a previous Opinion, we considered lethal effects on a portion of a population (2010, Batch 3 Opinion). This modeling showed that a 15-20% reduction in first year survival associated with direct and indirect effects to salmonids during freshwater rearing was sufficient to cause a significant decrease in the population growth rate (λ).. We used this reduction as one of our benchmarks in determining population-level effects. λ values for most of the salmon populations considered in this Opinion (detailed in *Appendix 2*) currently are near or below one (1). For a population to sustain itself and/or

recover, long-term lambdas need to be ≥ 1.00 . Reductions in population growth rate indicate decreasing abundance for the population and may suggest reduced recovery and jeopardy for the species. Current lambdas integrate a number of environmental factors, including use of pesticides not associated with the federal action addressed in this Opinion.

Based on our analysis, we evaluate whether use of the a.i. as registered will likely reduce the reproduction, numbers, or distribution of a population within each ESU/DPS. This likelihood is expressed qualitatively as low, medium, or high (Table 139- Table 166). Each ESU/DPS is made up of discrete population(s) of salmon or steelhead. Each of these populations support the survival and recovery of the species, but may not all be equally affected by the use of an a.i.(s). Taking into account both the unevenness of use, and the relative importance of various populations to the ESU/DPS, we then determine the potential for appreciable reduction in the reproduction, numbers, or distribution of the species. This is expressed qualitatively as low, medium, or high (Table 139 -Table 166).

In the *Conclusion* section, we present jeopardy and no jeopardy determinations (Table 195 and

Table 196). A high potential for reduction in the reproduction, numbers, or distribution of the species was determined to jeopardize the ESU/DPS. A low potential for reduction in the reproduction, numbers, or distribution of the species was determined to not jeopardize the ESU/DPS. A medium potential for reduction in the reproduction, numbers, or distribution of the species sometimes was determined to jeopardize and sometimes to not jeopardize the ESU/DPS, depending on circumstances associated with population(s) at risk, the relative importance to the ESU/DPS, and the a.i under consideration.

NMFS will continue to evaluate the scientific information regarding the impact of each a.i. to listed salmonids and designated critical habitat. Based on this review, it is possible that NMFS' qualitative determinations and ultimate conclusions may change in the final Biological Opinion.

Summaries of Individual a.i.s

2,4-D acid, amines, and esters

We note that based on available usage data, labels, and other information from applicants we were not able to distinguish separate markets or use sites for the ester and amine forms, thus we have assumed either may be applied on authorized use sites. As described in the *Response* section of the *Effects* chapter, we believe the greater toxicity of the ester forms is related to the rate of uptake. The master label allows for the application of ester products at many sites, including the direct application of the butoxy-ethyl seter (BEE) form to habitats that contain listed salmonids. Due to this fact, we have considered toxicity data from both the ester and amine forms in our evaluation of acute (short-term lethal) effects. Given relatively quick degradation of the ester to acid form in water, we use acid and amine toxicity data in our evaluation of chronic (longer term growth, reproduction, and sub-lethal effects).

Direct water applications are a major concern, as the recommended concentrations in the water exceed all toxicity endpoints for fish, prey, and primary productivity. The butoxy-ethyl ester (BEE) and amine forms are both approved for direct water applications. We are particularly concerned about the BEE, given the rapid uptake by fish. We believe that direct water applications of 2,4-D in accordance with label directions pose a serious risk to listed salmonids. Aquatic weed management may occur under local, state, or federal resource management plans or permitting systems and is often intended to improve the habitat for various beneficial uses. However, these resource management plans and/or permitting systems are not a part of the federal action (*i.e.*, the label).

In Washington State, in-water applications of pesticides require a permit under the National Pollutant Elimination Discharge System (NPDES) (<http://www.ecy.wa.gov/programs/wq/plants/management/aqua028.html>, accessed 02/23/11). Washington has issued general permits for lake applications, and for use in freshwater wetlands, rivers, and estuaries. Both general permits include application timing windows to limit use of 2,4-D during peak periods of salmon use of freshwater habitats for spawning and rearing. While this limits effects, it does not completely eliminate them. Control of weeds in irrigation canals is covered under a different permit, and 2,4-D is not among the pesticides authorized. The website notes that under a court order, 2,4-D BEE may not be used in waters containing listed salmonids. Although not specified on the website, the court order referenced appears to be the injunctions associated with the original *Washington Toxics Coalition* court case, which will be lifted following issuance of the final Opinion for this a.i. We are uncertain if Washington will remove or modify this restriction when the injunction is lifted. There is an individual fisheries management permit issued to the Washington State Department of Fish and Wildlife, but it covers piscicides such as rotenone, not 2,4-D. We believe the current restrictions in place in the state of Washington regarding direct water applications of 2,4-D provide substantial protection to listed salmonids. The restriction against using 2,4-D BEE in waters containing listed salmonids is an important part of that protection.

In California, in-water applications of pesticides require a permit under the National Pollutant Elimination Discharge System (NPDES) (http://www.swrcb.ca.gov/water_issues/programs/npdes/aquatic.shtml, accessed 2/24/11). Use of 2,4-D is addressed in the California permit. The California permit requires the discharger (pesticide applicator) to develop and implement an Aquatic Pesticide Application Plan (APAP). Under terms of the general permit, the discharger must submit a plan describing their analysis of alternatives, and Best Management Practices (BMPs) they may use to minimize effects to non-treated areas. There is an Maximum Concentration Limit (MCL) of 70 µg/L for 2,4-D for receiving waters with a municipal

use (MUN) designation, but no MCL for waters designated as warm or cold water habitat. Nonylphenol-containing surfactants are permitted, but subject to criteria of 6.6 µg/L for freshwater and 1.7 µg/L for saltwater. The only mention of listed species considerations is the following:

“34. This General Permit does not authorize any take of endangered species. The discharge is prohibited from adversely impacting biologically sensitive or critical habitats, including, but not limited to, habitat of species listed under federal or State endangered species laws. To ensure that endangered species issues are raised to the responsible agencies, the State Water Board has notified the U.S. Fish and Wildlife Service, the National Marine Fisheries Service, and the California Department of Fish and Game of this General Permit.” (page 6, Waste Discharge Requirements)

Beyond the language provided above, the California general permit does not contain specific requirements for listed salmonids, but relies instead on the individual applicator to ensure they are not adversely affecting listed species and their habitat. Thus, we do not believe the requirements of the California general permit provide any additional protections for listed salmonids beyond what is currently on the FIFRA label.

There is currently no permitting requirement for direct water applications of pesticides in Oregon, although we did locate a fact describing the proposed permitting system being developed coincident with EPA’s national level Pesticide General Permit (<http://www.deq.state.or.us/wq/wqpermit/docs/general/npdes2300a/ProposedFactSheetPesticideGP.pdf>, accessed 2/24/11). While the proposed permit calls for Integrated Pest Management (IPM) techniques, and requires development of a Pesticides Discharge Management Plan (PDMP), it makes no mention of any specific considerations or limitations for listed species. For these reasons, we do not believe the requirements of the proposed Oregon general permit provide any additional protections for listed salmonids beyond what is currently on the FIFRA label.

Idaho does not have delegated authority under Clean Water Act to develop NPDES permits, thus we assume any direct water applications are governed at the federal level. EPA is currently developing a pesticide general permit. Idaho does have a noxious weed

control plan that includes aquatic weeds such as Eurasian watermilfoil (<http://www.agri.state.id.us/Categories/PlantsInsects/NoxiousWeeds/Documents/general/Overview%20of%20ISDA%20Noxious%20Weed%20Program%202004.pdf>), accessed 2/25/11). There is a state strategic plan for Eurasian watermilfoil, but based on our initial review, it does not contain any specific provisions for listed salmon or other listed species (<http://www.agri.state.id.us/Categories/PlantsInsects/NoxiousWeeds/Documents/Milfoil/EWM%20Strategy%20Final.pdf>). As far as NMFS can ascertain, no programs in Idaho provide any additional protections for listed salmonids beyond what is currently on the FIFRA label.

Based on an analysis of state, regional, and national maps showing roads, railroads, electric transmission lines, and pipelines, we concluded ROW uses are most likely to be concentrated in more developed areas. In more remote areas, roads and railroads are often situated along river valleys, and sometimes in close proximity to the stream or river. These types of rights-of-way are generally associated with larger streams (third-order and above), so we assume EECs from EPA's PRZM-EXAMS modeling are more reflective of concentrations that might be found in these waters than NMFS floodplain EECs. However, unlike triclopyr BEE, evaluation of California's pesticide use reporting and information provided by applicants, indicated that while 2,4-D is used for rights-of-way, it is not a primary market.

2,4-D is registered for a broad range of uses, and is one of the most commonly used herbicides in the U.S. In addition to agricultural uses, ROW uses, and direct water applications, it is also authorized for turf uses, including in homeowner products. Thus, it could be used in any of the landuse classes we evaluate (agriculture, forested, urban/residential). We expect the temporal use pattern to vary among the different use sites, however, applications of 2,4-D are likely to occur throughout the period where plants are actively growing (early spring – late fall for most areas within the salmonids freshwater distribution). Thus, 2,4-D is likely to enter water bodies from multiple sources, and we expect these inputs to be frequent occurrences. The multiple source,

frequent occurrence pattern is a key factor in the risk posed by this a.i. In areas that are either highly developed or heavily agricultural or both we expect frequent inputs of 2,4-D

Overall, for 2,4-D, we anticipate mostly sublethal effects for fish, and little to no direct mortality for terrestrial applications of 2,4-D products. Some of these sublethal effects may manifest essentially as a reduction in available energy for activities such as reproduction, migration, or foraging due to reductions in glycogen storage in the liver. In one field study fish exposed to 2,4-D esters spawned two weeks late (Cope *et al.*, 1970). Systems inhabited by salmonids are often temporally distinct, with runs of listed salmonids and other fish spawning and emerging sequentially. A delay in spawning could mean that a typical food source is not present when the fry emerge, and/or a predator not typically present is in the system, or is larger in size. Such effects would vary widely by system, and are relatively unpredictable. However, if of a sufficient magnitude, such an event could cause reductions in abundance. We are especially concerned about this type of effects in ESUs/DPSs that have only one or two viable populations, and for coho, who have temporally distinct brood years. We do not anticipate large reductions in in-stream primary productivity from 2,4-D. However, any form of the a.i. applied near a waterbody is likely to affect terrestrial plants, especially herbs and forbs. Terrestrial vegetation near the waterbody provides shade, bank stabilization, sediment, chemical and nutrient filtering, and provides a niche for terrestrial invertebrates that are also salmon prey items. We are unable to quantify this effect due to variability in plant susceptibility to the forms of 2,4-D, and differences in species composition and density for various locations. We believe the a.i. will have a detrimental effect on the riparian vegetation. In general, riparian zones with diverse communities of herbaceous plants, shrubs, and trees will be more resilient than narrow grass strips or monocultures.

Triclopyr BEE

Based on use data from California's pesticide use reporting, Doane's use data, and information provided by applicants, a primary market for triclopyr BEE is rights-of-way. Main uses are roads and railroads, with some use on electric transmission lines, substations, and similar industrial locations.

Given the amount of relatively impervious surfaces (pavement, gravel, hard-packed) associated with rights-of-way, concentrations of the a.i. may be higher for these uses than for applications on vegetation and/or more loosely packed soil. We also expect wash-off to be rapid, thus exposure of a group of fish to a high concentration pulse of the ester form of triclopyr BEE is the scenario with which we are most concerned. Based on modeled estimates, this pulse appears to be a high enough concentration to kill a group of fish. If it occurred when a group of adult fish were migrating to spawn, when a group of smolts were emigrating, or in a shallow habitat where fry or larval fish were present, it could cause a major decrease in the population. We stress that we believe this situation is likely to occur rarely. However, rare catastrophic events may have dire consequences for species subject to small population dynamics, so we have considered this scenario in our ESU/DPS specific evaluation.

Based on an analysis of state, regional, and national maps showing roads, railroads, electric transmission lines, and pipelines, we concluded ROW uses are most likely to be concentrated in more developed areas. In more remote areas, roads and railroads are often situated along river valleys, sometimes in close proximity to the stream, rivers, and floodplain habitats. Additionally, rights of way uses are approved for high application rates (8 lbs a.i./A) and a large number of applications (8). However, we expect that due to practical considerations applications to rights-or-way will typically be limited to once or twice per year. High application rates and multiple applications are also approved for noncrop agricultural areas and ephemeral aquatic habitats during the dry period. When these applications occur in close proximity to salmon bearing water they may directly impact salmon. Additionally, direct applications to floodplains, riparian habitat and

ephemeral wetlands will likely modify the habitat in ways that may be detrimental to salmonids and their designated critical habitat.

Given the amount of relatively impervious surfaces (pavement, gravel, hard-packed) associated with rights-of-way, concentrations of the a.i. in salmonid habitats may be higher for these uses than for applications on vegetation and/or more loosely packed soil. We also expect wash-off to be rapid, thus exposure of fish to a high concentration pulse of the ester form of triclopyr BEE is likely. We expect initial concentration of triclopyr BEE to be high enough to kill fish in some instances.

A second important use for triclopyr BEE is in forestry. When scaled up to a lb a.i./A rate, direct (basal) applications are extremely high, but we expect these types of applications will result in lower concentrations than those modeled because they will be applied only in small areas (spot treatments). A number of salmon ESUs/DPSs spawn and/or rear in forested areas, some of which are managed by other federal agencies such as the U.S. Forest Service, the U.S. Bureau of Land Management, and the National Park Service. When these agencies use pesticides on forested lands supporting listed species, it is generally subject to a Section 7 programmatic or site-specific consultation. In the *Environmental Baseline* we describe several of the Biological Opinions based on these consultations. Some permit triclopyr, and some but not all, limit use to the TEA form. All have restrictions on the amount of a.i. applied. Many require “no –apply” buffer zones near water containing listed species, and require low drift, low runoff application methods. Based on these Opinions, we have assumed potential for use of triclopyr BEE near salmon-bearing waters is extremely limited when those waters are located in federally managed forests. However, there are no such restrictions on privately managed forests, and there may or may not be similar restrictions on state or local government managed forests. We assumed triclopyr BEE could be applied to these areas in accordance with label directions.

Diuron

Diuron is classified as moderately toxic to fish and moderately to highly toxic to aquatic invertebrates. It is persistent in aquatic systems, and is frequently detected in ambient water quality monitoring programs (29-47% detections). There may be extended exposure in slow-flowing waters. In faster flowing waters, there may be an extended chemical plume or increased concentrations in certain areas due to multiple input sources. Effects on fish development and reproduction occur at concentrations that have been reported in the ambient monitoring data. Linuron, which is structurally similar to diuron, has been identified as an endocrine disruptor in fish. Although diuron has not been identified as an endocrine disruptor, in absence of data we have assumed the linuron endpoints are applicable given those structural similarities. Effects on primary productivity occur in the range of 10-95 µg/L, and the NOAEC for ecosystem functioning is 0.5 µg/L.

Diuron is used heavily on rights-of-way, and is also authorized for a number of agricultural uses. Rights-of-way uses have much higher application rates (up to 12 lb ai/A) than agricultural uses (1-4 lb ai/A). Although these types of applications often occur in narrow strips, given the linear distances on which it might be used, areas of an acre or larger could conceivably be treated, thus we believe EECs calculated using PRZM-EXAMS are relevant. Additionally, linear rights-of-way such as those associated with roads and railways often occur in close proximity to rivers and shoreline habitats running in parallel for several miles increasing the likelihood of exposure. We simulated a linear application of diuron using AgDrift and found a single swath treatment could result in concentrations detrimental to salmonids and their habitat (*Risk Characterization*).

Based on use data from California's pesticide use reporting, and information provided by applicants, the main uses for diuron are roads and railroads, with some use on electric transmission lines, substations, and similar industrial locations. Using state, regional, and

national maps of roads, railroads, electric transmission lines, and pipelines, we concluded ROW uses are most likely to be concentrated in more developed areas. In more remote areas, roads and railroads are often situated along river valleys, sometimes in close proximity to the stream or river. Rights-of-way are generally associated with larger streams (third-order and above), so we assume EECs from EPA's PRZM-EXAMS modeling are more reflective of concentrations that might be found in these waters than NMFS floodplain EECs.

Agricultural uses are expected to primarily affect primary productivity, but not fish health or prey availability endpoints. We believe ROW uses could occasionally kill fish, and may cause decreased reproduction or endocrine effects. Additionally, it may decrease prey and depress primary productivity. We do not believe it rises to the level where it will cause appreciable reduction in abundance, distribution, or reproduction of listed salmonids in the affected waters.

We are concerned about effects on habitat, from a fish health and primary productivity standpoint, in areas that receive frequent inputs or inputs from multiple sources. In some cases, diuron may affect waters downstream from frequent input areas. An example of such an area is the Lower Columbia River, downstream of Willamette Valley. We expect in some cases diuron will degrade riparian habitat and decrease the productivity of receiving waters.

Linuron

Linuron is structurally very similar to diuron. It has nearly the same fate and toxicity profile, but is registered only for a small number of crops. It has some ROW uses, but those are primarily in an agricultural context. We anticipate most all linuron uses will occur in agricultural areas. Based on information provided by the applicant, and confirmation with pesticide usage data in California and Washington, linuron is used in very limited amounts. We expect minimal exposure in most ESUs/DPSs.

Captan

Captan dissociates rapidly in aquatic systems, so exposure to the parent molecule is expected to be limited. We do not anticipate captan will kill fish, reduce prey abundance or modify primary productivity or riparian vegetation. Sublethal effects are possible, but unlikely as exposure duration is expected to be limited given rapid degradation and may not be of sufficient duration to manifest adverse effects.

Chlorothalonil

Chlorothalonil has two distinct types of uses – on agricultural crops, and on turf. Multiple applications at fairly short intervals (7-14 days) are authorized for both types of uses. Although chlorothalonil is classified as very highly toxic to fish, it degrades quickly in aquatic systems via photolysis ($t_{1/2}=10$ h). In clear, shallow, or sunlit waters we do not expect to be present for extended time following a specific runoff or overspray event. In more turbid, deeper, or shaded waters it is likely to be present longer, as it is stable to hydrolysis, and the aerobic metabolism half life is 7 – 16 days. It is infrequently detected in ambient monitoring programs (0-2% detections), and when detected, concentrations are low compared to acute endpoints, but in the range of several of the chronic endpoints, such as immune response and prey reproduction and growth. However, EECs, which are a better estimator of initial runoff concentrations, exceed assessment endpoints for salmonids, prey, and primary productivity. EECs for turf uses are generally much higher (~ an order of magnitude) than most crop uses. Additionally, a recent targeted monitoring study on managed turf uses measured concentrations of chlorothalonil in a stream with peak concentrations that are greater than salmonid survival endpoints and chronic concentrations that are expected to cause reductions in available prey. We have located one targeted monitoring study on managed turf uses that measured concentrations of chlorothalonil in the stream higher than salmonid survival endpoints. Thus, we anticipate some lethal events for salmon could occur based on both types of use. We believe the likelihood of such events occurring is greater for turf use than for crop use.

“Turf” encompasses a wide range of sites, including golf courses, recreational fields, and managed landscapes around buildings and in parks. In many cases it also includes sod farms, but those are categorized in the agricultural landclass. Based on our initial analysis, we believe there is some likelihood of turf uses near salmon waters in all ESUs/DPSs. We will continue to evaluate turf uses during the period between release of this draft and issuance of the final Opinion.

In highly agricultural areas, we anticipate there will be frequent low-level inputs of chlorothalonil due to the permitted application intervals, and also that it will be from multiple sources. In situations like this, we expect that chronic low concentrations will have reduce primary productivity and/or modify the community structure. We also anticipate these low concentrations may affect fish immune response, making salmon more susceptible to other stressors. Shifts in plant community structure could degrade salmonid habitat by leading to reductions or shifts in invertebrate profiles affecting salmonid prey base or availability.

ESU/DPS Specific Evaluations

Below, we summarize the current status of each species, including baseline stressors. VSP parameters (abundance, growth rate, genetic variability, and spatial structure) are presented as a measure of the ESU/DPS’s relative health. As exposure to a.i.s during the juvenile life stage is of particular concern, we highlight the length of time juveniles are found in shallow, more vulnerable habitats. The number of extant populations that co-occur with agricultural and urban areas is also given.

Puget Sound Chinook Salmon (Threatened Species)

The Puget Sound ESU is comprised of 22 extant populations. Eleven of these populations have declining productivity; the remaining populations are at replacement value. Current spawner abundance is significantly lower than historical estimates. The spatial structure for this species is compromised by extinct and weak populations that are

disproportionately distributed in the mid- to southern Puget Sound and the Strait of Juan de Fuca. The genetic diversity of this ESU has been reduced due to a disproportionate loss of populations exhibiting the early-run life history and past hatchery practices.

The Puget Sound Chinook salmon are faced with many challenges to recovery, including lost and degraded habitat, loss of in-river large wood, poor water quality from land use practices, water diversions, and elevated temperatures. Pesticide use and detections in the ESU's watershed are well documented. NAWQA sampling conducted in 2006 in the Puget Sound basin detected numerous pesticides and other synthetic organic chemicals in streams and rivers.

Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 139. More than 50 percent of the ESU is composed of evergreen, deciduous, or mixed forests. Other pesticide use areas include urban/residential development (15%) and agricultural uses (4%). Cultivated crops (1%) and hay crops and pastures (3%) are primarily distributed on the floodplain and other lowland habitats. The majority of urban/residential land use also occurs within river and stream valleys in lowland areas, and much of the nearshore marine area also consists of urban/residential. Our GIS analysis indicates 22 populations in this ESU are exposed to pesticides applied in agriculture and urban areas. These areas serve as spawning, rearing, and migration habitat for Puget Sound Chinook. Juveniles generally have long freshwater residences of one or more years before migrating to the ocean. Given their long residency period and use of freshwater, estuarine, and nearshore areas, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitats.

We will be providing additional information that details specific considerations for each decision.

Table 139. Puget Sound Chinook Salmon.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Medium	Low
Diuron	Yes	Yes	NA	Yes	Medium	Medium
Linuron	Yes	NA	NA	Limited	Low	Low
Captan	Yes	Yes	NA	NA	Medium	Low
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Lower Columbia River (LCR) Chinook Salmon (Threatened Species)

The LCR Chinook salmon ESU includes 20 fall- and 2 late-fall runs and 9 spring-run populations. The majority of spring-run LCR Chinook salmon populations are nearly extirpated. Total returns for all runs are substantially depressed, and only one population is considered self-sustaining. The spatial structure for this ESU is relatively intact despite a 35% reduction in habitat. The genetic diversity of all populations (except the late fall-runs) has been eroded by large hatchery influences and low effective population size.

Obstacles to the recovery of LCR Chinook salmon include hydropower development, reduced access to habitat, loss of habitat, harvest, elevated water temperature, and sedimentation. NAWQA sampling detected more than 50 pesticides in streams within this ESU's range, ten of which also exceeded EPA's chronic toxicity aquatic life criteria (Wentz, Bonn et al. 1998).

Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 140. The percentage of agriculture lands that overlap with LCR Chinook salmon ESU is about 6 %, with 2% as cultivated crop crops and 4% as hay/pasture. More than 76% of the ESU is composed of evergreen, deciduous forest, and mixed forests.

Urban/residential development (13 %) is a fairly substantial portion of this ESU. Most of the highly developed land and agricultural areas in this ESU's range are adjacent to salmonid habitat. Our GIS analysis indicates that all populations may be exposed to pesticides applied in agriculture and urban areas. Given their long juvenile residency period, use of river mainstem and upstream tributaries for spawning, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitats.

We will be providing additional information that details specific considerations for each decision.

Table 140. Lower Columbia River Chinook Salmon.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low to <i>medium</i>	Low
Diuron	Yes	Yes	NA	Yes	Low to <i>medium</i>	Low
Linuron	Yes	NA	NA	Limited	Low to medium	Low
Captan	Yes	Yes	NA	NA	Medium	Low
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Upper Columbia River (UCR) Spring run Chinook Salmon (Endangered Species)

The UCR Spring-run Chinook salmon ESU is comprised of three extant populations. These populations are affected by low abundances and failing recruitment. The long-term trend for abundance and lambda for all three populations indicate a decline. The ESU's genetic integrity is compromised by periods of low effective population size and a low proportion of natural-origin fish. Spatial structure of this ESU is fairly intact.

Recovery of the UCR Chinook salmon is hindered by altered channel and floodplain morphology, habitat degradation, loss of in-river wood, reduced flow, impaired fish passage and fish mortality from dams, harvest impacts, impaired water quality, and

elevated temperature. Concentrations of azinphos methyl, triallate, chlorpyrifos, diazinon, lindane, and parathion have been detected in surface water samples and all exceeded EPA freshwater chronic criteria for the protection of aquatic life (Williamson, Munn et al. 1998).

Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 141. The percentage of agricultural and developed lands that overlap with UCR Chinook salmon habitat is about 5.4% and 4.7%, respectively. Forested lands make up about 45% of the ESU. Our GIS analysis indicates that all three populations are exposed to pesticides applied in agriculture and urban areas. Fish spawn and rear in the major tributaries leading to the Columbia River between Rock Island and Chief Joseph dams. Given their residency period and use of freshwater tributaries and floodplain areas, juveniles have a high probability of exposure to pesticides that are applied near salmonid aquatic habitats within the range of this ESU.

We will be providing additional information that details specific considerations for each decision.

Table 141. Upper Columbia Spring-Run Chinook Salmon.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	Low
Diuron	Yes	Yes	NA	Yes	Low	Low
Linuron	Yes	NA	NA	Limited	Low	Low
Captan	Yes	Yes	NA	NA	Low	Low
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Snake River (SR) Fall-run Chinook Salmon (Threatened Species)

The SR Fall-run Chinook salmon ESU consists of one population that spawns in the lower mainstem Snake River. Its spatial distribution has been reduced to 10 to 15% of

the historical range. The annual population growth rate for the population is just over replacement, and the ESU remains highly vulnerable due to low abundance. Genetic diversity has been reduced with the loss of additional populations and influx of hatchery raised spawners.

The major threats to this ESU include spawning habitat loss and degradation, impaired stream flows, barriers to fish passage, mortality from hydropower systems, poor water quality, and elevated water temperature.

Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 142. Pesticide use areas for the 6 a.i.s within this ESU's and above the Columbia River migratory corridor include evergreen forests (49%), cultivated crops (15%), pastures (1%), and developed lands (1%).

Historically, SR Fall-run Chinook salmon exhibited a largely ocean-type life history. However, as a consequence of dam construction, this ESU now resides in water that is cooler than the historic spawning areas, and alteration of the Lower Snake River by hydroelectric dams has created a series of low-velocity pools in the Snake River. Thus, Fall-run Chinook salmon in the Snake River Basin now exhibit one of two life histories: ocean-type and reservoir-type ((Connor, Sneva et al. 2005); (Tiffan, Rondorf et al. 2001)). The reservoir-type life history is one where juveniles overwinter in the reservoirs created by the dams, prior to migrating out of the Snake River. SR Fall-run Chinook salmon spend one to four years in the Pacific Ocean before beginning their spawning return migration. Given the freshwater residency period and migration distance traveled along the edges/margins of rivers, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitats.

We will be providing additional information that details specific considerations for each decision.

Table 142. Snake River Fall-run Chinook Salmon.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	Low
Diuron	Yes	Yes	NA	Yes	Low	Low
Linuron	Yes	NA	NA	Limited	Low	Low
Captan	Yes	Yes	NA	NA	Low	Low
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Snake River (SR) Spring/Summer-run Chinook Salmon (Threatened Species)

This ESU includes 31 historical populations. Productivity trends are approaching replacement levels, though most populations are far below their respective interim recovery targets. Many individual populations have highly variable abundance and no positive long-term growth. The genetic diversity and spatial distribution of this ESU are intact.

The major obstacles to the recovery of this ESU include altered channel and floodplain morphology, excessive sediment, reduced stream flow, degraded water quality from land use activities, hydroelectric dams, water diversions, and elevated water temperature.

The percentage of cultivated croplands and developed lands that overlap with SR Spring/Summer-run Chinook salmon habitat are 6.6% and 1.7%, respectively. Our GIS analysis indicates 20 populations in this ESU are exposed to pesticides applied in agriculture and urban areas. Juvenile fish mature in fresh water for one year and may migrate from natal reaches into alternative summer-rearing or overwintering areas.

Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 143. This ESU spawns and rears primarily in the smaller tributaries, many of which are located on U.S. Forest Service lands. Agricultural and urban areas are not common in the watersheds comprising the ESU, and those that are present are clustered mostly around the mainstem Snake and Columbia Rivers. The Snake River is a high-volume, high-flow system, and the salmon use it primarily as a migratory corridor.

We will be providing additional information that details specific considerations for each decision.

Table 143. Snake River Spring/Summer-run Chinook Salmon.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	Low
Diuron	Yes	Yes	NA	Yes	Low	Low
Linuron	Yes	NA	NA	Limited	Low	Low
Captan	Yes	Yes	NA	NA	Low	Low
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Upper Willamette River (UWR) Chinook Salmon (Threatened Species)

The UWR Chinook salmon ESU is composed of seven populations. Of these, only the McKenzie population is producing naturally. Abundance is low for all populations, and growth rates are negative. The spatial distribution of this ESU has been dramatically reduced, with 30 to 40% of the total historic habitat blocked by dams. The genetic diversity of this ESU has been compromised by hatchery stocks and mixing between populations.

The obstacles to recovery for this ESU include loss/degraded floodplain connectivity and stream habitat, reduced stream flow, reduced access to spawning/rearing habitat, degraded water quality, and elevated water temperature. Fifty pesticides were detected in streams that drain both agricultural and urban areas. Ten of these pesticides exceeded EPA criteria for the protection of freshwater aquatic life from chronic toxicity

The percentage of cultivated and developed lands that overlap with UWR Chinook salmon habitat are 10.5% and 9%, respectively. Co-occurrence of agriculture, forestry,

and urban areas with salmonid habitat is shown in Table 144. Our GIS analysis indicates all populations in this ESU may be exposed to pesticides applied in agriculture and urban areas. Juveniles rear in the mainstem Willamette River and floodplain wetlands during the inundation period. Residence periods range from 6 months to over a year, with three distinct emigration runs. Given their residency period and habitat preference, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitat.

We will be providing additional information that details specific considerations for each decision.

Table 144. Upper Willamette River Chinook Salmon.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Medium	Medium
Diuron	Yes	Yes	NA	Yes	Medium	Medium
Linuron	Yes	NA	NA	Limited	Medium	Low to medium
Captan	Yes	Yes	NA	NA	Medium	Medium
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

California Coastal (CC) Chinook Salmon (Threatened Species)

The CC Chinook salmon ESU's spatial structure has been drastically altered through the loss of several historic populations. Genetic diversity has been significantly reduced by the loss of the spring-run and coastal populations. Current population structure is uncertain, though fish are concentrated in 15 geographic locations. Populations in the Eel River and Russian River are larger than some of the others, and are important to the ESU. Overall ESU productivity is low and all populations have low abundance.

The major threats to this ESU's recovery include fisheries, timber harvest, vineyards and other agriculture, introduced fish species, migration barriers, habitat loss or degradation, increased predation, and elevated water temperatures. Pesticides may be used within these watersheds, though very little monitoring has occurred.

Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 145. The percentage of cultivated croplands and developed lands that overlap with CC Chinook salmon habitat are 1% and 5.4%, respectively. Our GIS analysis indicates 15 populations in this ESU are exposed to pesticides applied in agriculture and urban areas. The most abundant populations are in the Eel River and tributaries, and in the Russian River watershed. While there is little overlap of use sites with the habitat of the Eel River populations, there is substantial overlap in the Russian River watershed. Due to the importance of this population to the ESU, likelihood of negative effects ratings were based primarily on the overlap in this watershed. Juveniles rear in freshwater streams for a few months, and may reside in the estuary for an extended period before entering the ocean. Given their residency period and use of estuaries, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitat.

We will be providing additional information that details specific considerations for each decision.

Table 145. California Coastal Chinook Salmon.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	Low
Diuron	Yes	Yes	NA	Yes	Low to medium	Low
Linuron	Yes	NA	NA	Limited	Low	Low
Captan	Yes	Yes	NA	NA	Medium	Medium
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Central Valley (CV) Spring-run Chinook Salmon (Threatened Species)

The CV Spring - run Chinook salmon ESU includes four populations in the upper Sacramento River and three of its tributaries. The spatial distribution has been greatly reduced through extirpation of populations and dams blocking fish passage. Genetic diversity was similarly reduced with the extirpation of all San Joaquin runs. Abundance levels are all severely depressed from historic estimates, though time series data show that all three tributary populations have growth rates just above replacement.

Juvenile emigration in the Sacramento River is highly variable; individuals may migrate as fry or as yearlings. Floodplain habitats are particularly important for CV Spring - run Chinook salmon juveniles during rearing and migration (Sommer, Nobriga et al. 2001; Sommer, Harrell et al. 2005). Given the residency period and use of non - natal tributaries, intermittent streams, and floodplain habitats for rearing and migration, juveniles and adults have a high probability of exposure to pesticides that are applied near their habitat.

The major threats to the recovery of this ESU include impaired or loss of habitat, predation, water management (dams, levees, reservoirs), and impaired water quality.

Pesticides detected in the Sacramento River include thiobencarb, carbofuran, molinate, simazine, metolachlor, and dacthal, chlorpyrifos, carbaryl, and diazinon. Heavy use of agricultural pesticides and the high probability of mixtures increase likelihood of negative effects for this species.

The percentage of cultivated croplands and developed lands that overlap with CV Chinook salmon habitat are 21.3% and 10.8%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 146. Our GIS analysis indicates all four populations in this ESU are exposed to pesticides applied in agriculture and urban areas. Fish must also migrate through the San Francisco-San Pablo-Suisan Bay estuarine complex, which is heavily influenced by input from California's Central Valley.

We will be providing additional information that details specific considerations for each decision.

Table 146. Central Valley Spring-run Chinook Salmon.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Medium	Medium
Diuron	Yes	Yes	NA	Yes	Medium	Medium
Linuron	Yes	NA	NA	Limited	Medium	Medium
Captan	Yes	Yes	NA	NA	Medium	Medium
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Sacramento River Winter-run Chinook Salmon (Endangered Species)

The Sacramento River Winter-run Chinook salmon ESU is now comprised of a single population. This population rears in the mainstem of the Sacramento River below Keswick Dam. Abundance and productivity have fluctuated greatly over the past two decades. The genetic diversity of this population has been reduced through small

population sizes and the influence of hatchery fish. The large fluctuations in productivity and abundance indicate that the species is highly vulnerable to extinction.

The obstacles to the recovery of this ESU are impaired or loss of habitat, predation, water management (dams, levees, and reservoirs), and increased water temperatures. Today, the ESU depends on reservoir storage and releases for access to cold water. Pesticides frequently detected in the Sacramento River include thiobencarb, carbofuran, molinate, simazine, metolachlor, dacthal, chlorpyrifos, carbaryl, and diazinon. Heavy use of agricultural pesticides and the high probability of mixtures increase likelihood of negative effects for this species. Juveniles rearing in the river system and floodplains may encounter high concentrations of pollutants at the onset of the rainy season.

Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 147. The percentage of cultivated croplands and developed lands that overlap with Sacramento River Winter-run Chinook salmon are 25% and 10%, respectively. Our GIS analysis indicates the sole winter - run population in this ESU is exposed to pesticides applied in agriculture and urban areas. Juvenile winter-run fish are found in the Delta primarily from November through early May, though some spend up to 10 months in the river system. Given their residency period and use of the Sacramento River and Delta for rearing, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitat.

We will be providing additional information that details specific considerations for each decision.

Table 147. Sacramento Winter-run Chinook Salmon.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Medium	Medium
Diuron	Yes	Yes	NA	Yes	Medium	Medium
Linuron	Yes	NA	NA	Limited	Medium	Medium
Captan	Yes	Yes	NA	NA	Medium	Medium
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Hood Canal Summer-run Chum Salmon (Threatened Species)

This ESU has two remaining independent populations. Much of the historical spatial structure has been lost; most all populations on the eastern side of the canal are extirpated. The genetic diversity of the ESU has also declined. The two populations have long-term trends above replacement, though abundance is very low. The life history of this ESU strongly influences the potential for exposure. Following emergence, fish typically migrate quickly to nearshore marine areas in Puget Sound to rear and grow. Average rearing time for juveniles in Hood Canal is around 23 days before emigration.

The major threats to the survival and recovery of this ESU include habitat (floodplain, estuarine, and riparian) degradation, reduced stream flow, sedimentation, and hatcheries. The widespread loss of estuary and lower floodplain habitat has impacted the ESU's spatial structure and connectivity. NAWQA detections in surface waters in the Puget Sound Basin reported 26 of 47 analyzed pesticides.

Land use within the ESU is predominantly forested (73%), open water (17%), urban/residential (9%), and agriculture (2%). The percentage of cultivated croplands and developed lands that overlap with HC Summer-run chum salmon habitat is about 0.04% and 8.9%, respectively. Most of the agriculture and urban/residential occurs within river

and stream valleys in lowland areas. Nearshore marine areas are frequently adjacent to urban/residential areas. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 148. Our GIS analysis indicates that both populations of HC Summer-run chum salmon may be exposed to pesticides applied in agriculture and urban areas

We will be providing additional information that details specific considerations for each decision.

Table 148. Hood Canal Summer-run Chum Salmon.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	Low
Diuron	Yes	Yes	NA	Yes	Low	Low
Linuron	Yes	NA	NA	Limited	Low	Low
Captan	Yes	Yes	NA	NA	Low	Low
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Columbia River (CR) Chum Salmon (Threatened Species)

This ESU has been reduced to two populations: the Lower Gorge tributaries and Grays River. The population abundances for the Grays River and Lower Gorge are significantly depressed. Short- and long-term productivity trends for these populations are at or below replacement. Much of the genetic diversity of this population has been lost due to the extirpation of 15 populations.

The major threats to this ESU include overharvests, hatcheries, hydromodification, habitat loss, elevated temperatures, and poor water quality. Of the salmonids, chum salmon are most averse to negotiating obstacles in their migratory pathway. Thus, they are more highly impacted by the Columbia River hydropower system – specifically the Bonneville Dam (Johnson, Grant et al. 1997).

The percentage of cultivated croplands, hay/pasture, and developed lands that overlap with CR chum salmon habitat is about 2%, 5%, and 15%, respectively. More than 50% of the ESU is covered by deciduous, evergreen, or mixed forests. Within the ESU, agriculture and development are predominantly distributed in the low-lying areas near the Columbia River and its tributaries. Our GIS analysis indicates 14 populations of CR chum salmon are exposed to pesticides applied in agriculture, developed, and forested areas. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 149.

Chum salmon spawning migration in the Columbia River occurs in the late fall, from mid-October to December. They primarily spawn along the edges of the mainstem or in tributaries or side channels. The fry emerge between March and May and emigrate shortly thereafter to nearshore estuarine environments (Salo 1991). Juveniles spend around 24 days feeding in the estuary. The Columbia River estuary is extremely large with tidal influence extending from its mouth at the Pacific Ocean to the Bonneville Dam, 235 km upstream.

We will be providing additional information that details specific considerations for each decision.

Table 149. Columbia River chum Salmon

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	Low
Diuron	Yes	Yes	NA	Yes	Low	Low
Linuron	Yes	NA	NA	Limited	Low	Low
Captan	Yes	Yes	NA	NA	Low	Low
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Lower Columbia River Coho Salmon (Threatened Species)

The LCR coho salmon ESU now consists of two populations found in the Sandy and Clackamas Rivers. Both populations have low levels of abundance. The diversity of populations has been eroded by large hatchery influences and low effective population sizes. The spatial structure for this ESU has also been drastically reduced compared to historical levels.

The major obstacles to LCR coho salmon's survival and recovery are reduced water flow from irrigation diversions and hydroelectric dams, degraded water quality, and elevated temperature. NAWQA sampling in surface waters within the ESU range detected more than 50 pesticides in streams. Ten pesticides exceeded EPA's criteria for the protection of aquatic life from chronic toxicity, including azinphos methyl.

The percentage of cultivated crop lands overlap with LCR coho ESU is about 6 %, 4% as hay/pasture land and 2% as cultivated crop land. More than 76% of the ESU is composed of evergreen, deciduous forest, and mixed forests. Urban/residential development lands (12%) were a fairly substantial portion of this ESU. The percentage of cultivated croplands and developed lands that overlap with LCR chum salmon habitat are 2% and 11.7%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 150. Our GIS analysis indicates both populations of LCR coho salmon may be exposed to pesticides applied in agriculture and urban areas. Juveniles rear in fresh water for more than a year. During the day, they show a preference for near-shore habitats and use floodplain habitats (Johnson 1991).

We will be providing additional information that details specific considerations for each decision.

Table 150. Lower Columbia River coho Salmon.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Medium	Medium
Diuron	Yes	Yes	NA	Yes	Medium	Low
Linuron	Yes	NA	NA	Limited	Medium	Medium
Captan	Yes	Yes	NA	NA	Medium	Medium
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Oregon Coast (OC) Coho Salmon (Threatened Species)

The OC coho salmon ESU includes 13 functionally independent populations. Current abundance levels are less than 10% of historic populations. Long-term trends in ESU productivity remain strongly (Good, Waples et al. 2005). Spatial distribution is relatively intact. Populations within the ESU experience recruitment failure and long-term negative growth. As with other coho, there is a 3 year brood cycle, and depletion of a specific brood year may reduce the resiliency of the ESU.

The major threats to this ESU include reduced habitat complexity, loss of overwintering habitat, excessive sediment, habitat degradation, elevated temperature, water diversions, and poor water quality.

The percentage of cultivated croplands and developed lands that overlap with OC coho salmon habitat are 0.23% and 6.6%, respectively. Most of the cropland is hay/pasture, and is primarily located in the Umpqua watersheds. While this is an important population for this ESU, there are a number of other functionally independent populations in other watersheds with less overlap. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 151. Our GIS analysis indicates

all 13 populations of OC coho salmon may be exposed to pesticides applied in agriculture and urban areas. Juvenile coho salmon are often found in small streams less than five ft wide and rear in fresh water for 18 months.

We will be providing additional information that details specific considerations for each decision.

Table 151. Oregon Coast Coho Salmon.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	Low
Diuron	Yes	Yes	NA	Yes	Low	Low
Linuron	Yes	NA	NA	Limited	Low	Low
Captan	Yes	Yes	NA	NA	Low	Low
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

*Southern Oregon/Northern California Coast (SONCC) Coho Salmon
(Threatened Species)*

The SONCC coho salmon ESU includes coho salmon in streams between Cape Blanco, Oregon, and Punta Gorda, California. The disproportionate loss of southern populations has decreased the genetic diversity of this ESU. Coho distribution within individual watersheds has been reduced as well. There is very limited information on population growth rates for this ESU. Available data indicates that the Eel River and southern populations have critically low abundances. Coho have a 3 year brood cycle, and depletion of a specific brood year may reduce the resiliency of the ESU.

The major obstacles to the survival and recovery of this ESU include habitat loss and degradation, reduced stream flow, migratory barriers, timber harvest, agricultural activities, water management, and elevated temperatures.

The percentage of cultivated croplands and developed lands that overlap with SONCC coho salmon habitat are 2.5% and 4.3%, respectively. Our GIS analysis indicates that fish may be exposed to pesticides applied in agriculture and urban areas in all watersheds. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 152. As little population data were available for this ESU, we were not able to determine if agricultural and developed areas, which cluster in certain watersheds, co-occur with important populations. Areas with more cropland include the Scott and Shasta watersheds in the Klamath basin, and the Upper and Middle Rough River⁹ watersheds. Of the development in this ESU, much is in the Rough River basin, with most of the rest distributed along the coastline and estuaries. The fry rear in backwater, side channels, and shallow channel edges for up to 18 months.

We will be providing additional information that details specific considerations for each decision.

Table 152. Southern Oregon/Northern California Coast Coho Salmon.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	Low
Diuron	Yes	Yes	NA	Yes	Low	Low
Linuron	Yes	NA	NA	Limited	Low	Low
Captan	Yes	Yes	NA	NA	Low	Low
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

⁹ The Rough River is also be referred to as the Rouge or Rouge River in other publications, maps, or websites

Central California Coast (CCC) Coho Salmon (Endangered Species)

The CCC coho salmon ESU includes 11 independent populations. The spatial structure for CCC coho salmon has been substantially modified due to lack of viable source populations and loss of dependent populations. Long-term population trends are unknown, though all populations have very low abundances. This year's low return suggests that all three year classes are faring poorly across the species' range. Loss of a specific year class may decrease the overall resiliency of the population. Juveniles rear for 18 months, spending two winters in fresh water.

The major threats to the survival and recovery the ESU include loss of riparian cover, elevated water temperature, alteration of channel morphology, loss of winter habitat, and siltation. Highly contaminated runoff into the Russian River, San Francisco Bay, and into rivers south of the Golden Gate Bridge is expected during the first fall storms.

The percentage of cultivated croplands and developed lands that overlap with CCC coho salmon habitat are 2.3% and 9.4%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 153. Much of the development is centered around San Francisco Bay, and there are also developed areas and agriculture in the Russian River watershed. Coho in the San Francisco Bay are considered effectively extirpated, and the Russian River, which was once a source population for this ESU, is in serious decline (Spence, Bjorkstedt et al. 2008). Our GIS analysis indicates that all 11 populations may be exposed to pesticides applied in agriculture and urban areas. The majority of the salmon remaining is in the northern, undeveloped watersheds around the Navarro and Big Rivers.

We will be providing additional information that details specific considerations for each decision.

Table 153. Central California Coast Coho Salmon.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low to medium	Low
Diuron	Yes	Yes	NA	Yes	Medium	Medium
Linuron	Yes	NA	NA	Limited	Low	Low
Captan	Yes	Yes	NA	NA	Medium	Medium
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Ozette Lake Sockeye Salmon (Threatened Species)

The Ozette Lake sockeye salmon ESU consists of a single population made up of five spawning aggregations. Uncertainty remains on the growth rate and productivity of the natural component of the ESU. Genetic differences occur between age cohorts and different age groups do not spawn with each other. Genetic diversity within the ESU, however, is low. Spatial structure of the population has been altered, as only two beaches are known to be used for spawning (Haggerty, Ritchie et al. 2007). Overall abundance is also significantly depressed.

Major threats to this population include degraded habitat, loss of in-river large wood, and siltation of spawning habitat (Haggerty, Ritchie et al. 2007). Roughly 77% of the land in Ozette Basin is managed for timber production (Jacobs, Larson et al. 1996).

Ozette Lake is in a sparsely populated area, with less than 1% of land developed within the range of this ESU. Similarly, there is no cultivated cropland. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 154. Our GIS analysis indicates that Ozette Lake sockeye salmon have minimal risk of exposure to pesticides applied for agricultural or urban uses. They may be at risk of exposure from forestry related uses. Fry rear in the limnetic zone of Ozette Lake for a full year.

The life histories of this ESU strongly influence the potential for exposure to the 6 a.i.s. Adult spawners enter Ozette River from April to early August and may remain in Ozette Lake for extended periods before spawning (October- February). Spawning occurs along the lakeshore and historically in some of the lakes' tributaries. Fry migrate immediately to the lake where they rear for a year or so before entering the ocean. Land use of this ESU is primarily forest with private, state, and federal ownership (86% forested, 13% open water, 1% developed land, 0% agriculture). No crops were indentified within the NLCD data for this ESU. The entire circumference of the lake is within Olympic National Park. The predominant pesticide use sites (*i.e.*, urban/residential and forestry uses) overlap with the Lake's freshwater tributaries. As such, the greatest risk of exposure is to those sockeye that utilize freshwater tributary habitats. Based on juvenile sockeye's lake rearing, we do not anticipate reductions in prey within the tributaries to affect juvenile growth. Direct effects to fish remain a concern within tributaries. No effect determinations were made for those a.i.s that showed little to no overlap of sockeye with labeled uses. Although no cropland occurred within the 2.5 km area analyzed, private residences along tributaries may have small, non-commercial crops where applications of the a.i.s could occur. We assumed it is unlikely that restricted use pesticides would be applied in these situations. Not likely to adversely affect determinations were made for those a.i.s with very low probability of exposure and expect effects to be discountable or insignificant.

We will be providing additional information that details specific considerations for each decision.

Table 154. Ozette Lake sockeye Salmon.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/Residential	Forest	Rights-of-way	Populations	Species
2,4-D	No	Yes	Yes	Yes	High	High
Triclopyr BEE	No	Yes	Yes	Yes	Low to <i>medium</i>	Low
Diuron	No	Yes	NA	Yes	Low	Low
Linuron	No	NA	NA	Limited	Low	Low
Captan	No	Yes	NA	NA	Low	Low
Chlorothalonil	No	Yes	Yes	NA	Low	Low

NA: Not applicable

Snake River Sockeye Salmon (Endangered Species)

The SR sockeye salmon ESU is comprised of one remaining population in Redfish Lake, Idaho. Abundance and productivity are highly variable; around 30 fish of hatchery origin return to spawn each year (NMFS 2008). However, this figure has increased to adults numbering in the hundreds over the last two years. The ESU's genetic diversity has been reduced based on low population abundance and a high proportion of hatchery-origin fish.

The major threats to the survival and recovery of this ESU include altered channel morphology, impaired tributary and stream flow and passage, migration barriers, degraded water quality, hydromodification of the Columbia and Snake Rivers, and fish mortality from hydropower systems.

About 1% of the land surrounding Redfish Lake has been developed, and another 1% is used for agriculture, primarily hay and pasture. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 155. Juvenile sockeye remain in the lake for one to three years before migrating through the Snake and Columbia Rivers for several hundred miles to the ocean. Given the distance traveled between Redfish Lake and the ocean, juveniles and returning adults have a high probability of exposure to

pesticides that are applied near salmonid habitats during migration. More than 50% of the ESUs is in evergreen forests. Consequently, forestry uses are the major source of pesticide exposure during spawning and rearing activities.

We will be providing additional information that details specific considerations for each decision.

Table 155. Snake River Sockeye Salmon.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	Low
Diuron	Yes	Yes	NA	Yes	Low	Low
Linuron	Yes	NA	NA	Limited	Low	Low
Captan	Yes	Yes	NA	NA	Low	Low
Chlorothalonil	Yes	Yes	Yes	NA	Low	Low

NA: Not applicable

Puget Sound Steelhead (Threatened Species)

The Puget Sound steelhead is comprised of 53 populations (37 winter-run and 16 summer-run). Summer-run populations are concentrated in northern Puget Sound and Hood Canal. The WDFW 2002 stock assessment categorized 5 populations as healthy, 19 as depressed, 1 as critical, and 27 of unknown status (Washington Department of Fish and Wildlife (WDFW) 2002). Median population growth rates indicate declining population growth for nearly all populations in the DPS (NMFS 2005). Overall, the DPS experiences declining abundance, reduced genetic diversity, and abbreviated spatial complexity.

The major threats to the survival and recovery of this DPS include habitat degradation, water diversions, poor water quality, hatchery domestication, and elevated temperature. Over two million people inhabit the area, with most development occurring along rivers

and coastline. NAWQA sampling conducted in 2006 within the Puget Sound basin detected 26 pesticides and 74 other synthetic organic chemicals in streams and rivers.

More than 50 percent of the ESU is composed of evergreen, deciduous, or mixed forests. Other pesticide use areas include urban/residential development (15%) and agricultural uses (4%). Cultivated crops (1%) and hay crops and pastures (3%) are primarily distributed on the floodplain and other lowland habitats. The majority of urban/residential also occurs within river and stream valleys in lowland areas, and much of the nearshore marine area also consists of urban/residential development. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 156. Our GIS analysis indicates all populations in this DPS are exposed to pesticides applied in agriculture and urban areas. Fry usually inhabit shallow water along banks of stream or aquatic habitats on stream margins. Juveniles rear in a wide variety of freshwater habitats, generally for two years with a minority migrating to the ocean as one or three-year olds.

We will be providing additional information that details specific considerations for each decision.

Table 156. Puget Sound steelhead.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Medium	Low
Diuron	Yes	Yes	NA	Yes	Medium	Medium
Linuron	Yes	NA	NA	Limited	Medium	Low
Captan	Yes	Yes	NA	NA	Medium	Low
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Lower Columbia River Steelhead (Threatened Species)

The LCR steelhead DPS includes 23 extant populations. Spatial structure within the DPS, especially in Washington, has been substantially reduced by the loss of access to the upper portions of some basins from tributary hydropower development. Many of the populations in this DPS are small, and the long- and short-term trends in abundance of all individual populations are negative. The genetic diversity of this DPS has also been substantially reduced.

The major threats to this DPS include dams, water diversions, destruction/ degradation of habitat, altered channel morphology, reduced floodplain connectivity, sedimentation, reduced stream flow, land use practices, poor water quality, and elevated water temperature. NAWQA sampling detected more than 50 pesticides. Ten pesticides exceeded EPA's criteria for the protection of aquatic life from chronic toxicity.

The percentage of cultivated crop lands overlap with LCR Steelhead DPS is about 7%, 4.5 % as hay/pasture land and 2.5% as cultivated crop land. More than 61% of the DPS is composed of evergreen, deciduous forest, and mixed forests. Urban/residential development lands (12%) were a fairly substantial portion of this DPS. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 157. Our GIS analysis indicates all populations are exposed to pesticides applied in agriculture and urban areas. Juveniles typically rear in floodplain habitats associated with their natal rivers and streams for more than a year, and remain in fresh water systems for at least two years.

We will be providing additional information that details specific considerations for each decision.

Table 157. Lower Columbia River steelhead.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low to <i>medium</i>	Low
Diuron	Yes	Yes	NA	Yes	Low to <i>medium</i>	Low
Linuron	Yes	NA	NA	Limited	Low	Low
Captan	Yes	Yes	NA	NA	Medium	Low
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Upper Willamette River Steelhead (Threatened Species)

The UWR steelhead DPS is comprised of four extant populations that occupy tributaries draining the east side of the UWR basin. Populations within this DPS have been declining and have exhibited large fluctuations in abundance. Abundance is moderately depressed for the entire DPS. The DPS's spatial distribution and genetic diversity are moderately intact.

The major threats to the survival and recovery of this DPS include habitat loss due to blockages, lost or degraded floodplain connectivity, and degraded water quality within the Willamette mainstem and the lower reaches of its tributaries. Fifty pesticides were detected in streams that drain both agricultural and urban areas. Forty-nine pesticides were detected in streams draining agricultural land, while 25 pesticides were detected in streams draining urban areas. Ten of these pesticides, including azinphos methyl, exceeded EPA criteria for the protection of freshwater aquatic life.

The percentage of cultivated crop lands and developed lands overlapping with this DPS are 14.5% and 10%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 158. Our GIS analysis indicates all four

populations in this DPS are exposed to pesticides applied in agriculture and urban areas. After emergence, steelhead fry typically rear in floodplain habitats associated with their natal rivers and streams for two years. Given their residency period and habitat preference, juveniles and migrating adults have a high probability of exposure to pesticides that are applied near their habitat.

We will be providing additional information that details specific considerations for each decision.

Table 158. Upper Willamette Steelhead.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Medium	Medium
Diuron	Yes	Yes	NA	Yes	Medium	Medium
Linuron	Yes	NA	NA	Limited	Medium	Medium
Captan	Yes	Yes	NA	NA	Medium	Medium
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Middle Columbia River Steelhead (Threatened Status)

The MCR steelhead DPS includes 16 extant populations in Oregon and Washington. The spatial structure of this population is relatively intact. The genetic diversity has been compromised by interbreeding with resident and hatchery fish. Population growth rates are near replacement, though abundances are depressed in relation to historic levels.

The major threats to this DPS include altered floodplain and channel morphology, sedimentation, reduced stream flow altered hydrograph, migratory barriers, hydroelectric system mortalities, agricultural practices, poor water quality, and elevated water temperature. Seventy-six pesticide compounds were detected within the Yakima River Basin.

The percentage of cultivated crop lands and developed lands within the range of this DPS are 17% and 3%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 159. Our GIS analysis indicates 16 populations are exposed to pesticides applied in agricultural and urban areas. Swim-up fry usually inhabit shallow water along banks of streams or aquatic habitats on stream margins. Juveniles rear in a variety of freshwater habitat for two years . Essentially there is a continuous run of steelhead throughout the year thus adults and rearing juveniles are in freshwater habitats throughout the year.

We will be providing additional information that details specific considerations for each decision.

Table 159. Middle Columbia River steelhead.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low to medium	Low
Diuron	Yes	Yes	NA	Yes	Low	Low
Linuron	Yes	NA	NA	Limited	Medium	Low
Captan	Yes	Yes	NA	NA	Medium	Medium
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Upper Columbia River Steelhead (Endangered Species)

The UCR steelhead DPS consists of four extant populations in Washington State. Abundance data indicate that these populations are below the minimum threshold for recovery and have negative growth rates. Adult returns are dominated by hatchery fish and experience reduced genetic diversity from homogenization of populations. The spatial structure of this DPS has been severely altered, with 50% of its habitat cutoff by the Grand Coulee Dam.

The major obstacles to the survival and recovery of the UCR steelhead include hatcheries, dams that impede and block fish migration, altered floodplain and channel morphology, water diversions which reduce water quantity, loss of LWD, destruction of riparian habitat, harvest, hydroelectric system mortality, land use practices, poor water quality, and elevated water temperature. Pesticides have been detected in UCR steelhead freshwater habitats. Concentrations of six pesticides exceeded the guidelines for aquatic life.

The percentage of cultivated crop lands and developed lands within the range of the ESU are 13% and 4%, respectively. Our GIS analysis indicates all 4 populations in this DPS are exposed to pesticides applied in agriculture and urban areas. Newly emerged fry move about considerably and seek suitable rearing habitat, such as stream margins or cascades. The majority of juveniles smolt as two-year olds, though some individuals may rear for as long as seven years in these fresh water systems.

Co-occurrence of agriculture, forestry and urban areas with salmonid habitat is shown in Table 160. There is some agriculture in the spawning and rearing areas in the Wenatchee, Methow, and Okenogan watersheds. In the Entiat, there is intense agriculture outside the buffer, in the Upper Columbia Irrigation District. The water is heavily used and re-used in irrigation. We expect that the fish will also be exposed to a number of the a.i.s on their migratory pathway along the Columbia River, where the valley is heavily agricultural. A portion of the waters the salmonids use are 303 (d) listed for high temperature, and we expect this will exacerbate effects of the a.i.s.

We will be providing additional information that details specific considerations for each decision.

Table 160. Upper Columbia River steelhead.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	Low
Diuron	Yes	Yes	NA	Yes	Low	Low
Linuron	Yes	NA	NA	Limited	Low	Low
Captan	Yes	Yes	NA	NA	Low	Low
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

SNAKE RIVER BASIN STEELHEAD (THREATENED SPECIES)

The SR basin steelhead DPS includes 23 populations that are spatially distributed in each of the six major geographic areas in the Snake River basin (Good, Waples et al. 2005).

The historic spatial structure is relatively unaltered. While population growth rates show mixed long- and short-term trends in productivity, overall abundances remain well below their interim recovery criteria. Genetic diversity has been reduced, particularly for the B-run steelhead, those whose life history pattern includes spending two or more years in freshwater, and two or more years in the ocean before their upriver migration. A-run steelhead are smaller, have a shorter freshwater and ocean residence.

The major threats to the survival and recovery of this DPS include hatcheries, harvest impacts, altered floodplain and channel morphology, hydrosystem mortality, water diversions, sedimentation, degraded water quality, and elevated temperature. Pesticides have been detected in SR basin steelhead freshwater habitats, including eptam, atrazine, desethylatrazine, metolachlor, and alachlor.

SR basin steelhead are generally classified as summer-run fish. They enter the Columbia River from late June to October. After remaining in the river through the winter, SR basin steelhead spawn the following spring (March to May). Juveniles typically rear in

floodplain habitats associated with their natal rivers and streams for more than a year. SR basin steelhead typically smolt after two or three years. During their freshwater residence they may be exposed to pesticides used for a variety of uses. Potential exposure from use within the DPS includes use on evergreen forests (52%), agricultural lands including use on cultivated crops (8%) and hay/pasture (1%), and use in urban/residential or other developed areas (2%). Our GIS analysis indicates 15 populations of this DPS could be exposed to pesticides applied for these uses during their freshwater rearing period. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 161.

We will be providing additional information that details specific considerations for each decision.

Table 161. Snake River steelhead.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	Low
Diuron	Yes	Yes	NA	Yes	Low	Low
Linuron	Yes	NA	NA	Limited	Low	Low
Captan	Yes	Yes	NA	NA	Low	Low
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Northern California Steelhead (Threatened Species)

The NC steelhead DPS includes 15 historically independent populations of winter steelhead and 4 extant populations of summer steelhead. The loss of summer-run steelhead populations has significantly reduced the genetic diversity. Most populations are in decline and have low abundances and production. Although the DPS spatial structure is relatively intact, the distribution within most watersheds has been restricted

by physical and temperature barriers. Juvenile steelhead remain in fresh water for two or more years, rearing in streams and lagoons.

The major threats to the survival and recovery of this DPS include land use practices, migratory barriers, timber harvest, loss of large woody debris, reduced riparian vegetation, elevated water temperature, increased predation, and barriers that limit access to tributaries.

The percentage of cultivated crop lands and developed lands overlapping with NC steelhead habitat are also less than 1% and 19%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 162. Our GIS analysis indicates 19 populations of NC steelhead are exposed to pesticides applied in urban areas. Of these, 15 are also exposed to pesticides applied in agriculture areas. However, there are few areas of concentrated agriculture. Most appears to hay/pasture, concentrated in the Lower Eel watershed and some of the other coastal valleys. Development is concentrated primarily near Eureka, on the coast in the Mad River and Redwood Creek watersheds. Much of the land area in this DPS is heavily forested, and there are a number of state and national parks.

We will be providing additional information that details specific considerations for each decision.

Table 162. Northern California steelhead.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	Low
Diuron	Yes	Yes	NA	Yes	Low	Low
Linuron	Yes	NA	NA	Limited	Low	Low
Captan	Yes	Yes	NA	NA	Low	Low
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Central California Coast (CCC) Steelhead (Threatened Species)

The CCC steelhead DPS includes nine historic independent populations, all of which are nearly extirpated. Data on abundance and population growth rates are scarce, but available information strongly suggests that no population is viable. The loss of spatial structure and hatchery influences have likely reduced the genetic diversity for this DPS. Juvenile steelhead remain in fresh water for one or more years rearing in small tributaries and floodplain habitats. Age to smoltification for this DPS is typically 1 to 4 years. Steelhead have a more adaptive life history than some of the other salmon species, including overlapping generations and iteropary.

The major threats to this DPS include dams and other migration barriers, urbanization and channel modification, agricultural activities, predators, hatcheries, and water diversions which affect water quantity and stream temperature. Throughout the species' range, habitat conditions and quality have been degraded by a lack of channel complexity, eroded banks, turbid and contaminated water, low summer flow and high water temperatures, and restricted access to cooler head waters from migration barriers.

High densities of crop farming occur throughout the San Joaquin Basin, the Delta, and along the lower Sacramento River. There is also agriculture in the Russian River valley. The Russian River population is one of the largest runs. Southern portions of DPS include the heavily developed areas around San Francisco Bay. The percentage of cultivated croplands and developed lands that overlap with CCV steelhead habitat are 27% and 10%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 163. Most of the watersheds in this DPS are heavily developed, and/or have intensive agriculture in the river valley. A number of the populations must migrate through the San Francisco-San Pablo-Suisan Bay estuarine complex, which is heavily influenced by input from California's Central Valley.

We will be providing additional information that details specific considerations for each decision.

Table 163. Central California Coast steelhead.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Medium	Medium
Diuron	Yes	Yes	NA	Yes	Medium	Medium
Linuron	Yes	NA	NA	Limited	Low to medium	Low
Captan	Yes	Yes	NA	NA	Medium	Medium
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

California Central Valley (CCV) Steelhead (Threatened Species)

The CCV steelhead DPS consisted of 81 historical and independent populations. The spatial structure of the CCV steelhead has been greatly reduced by loss of habitat diversity and tributary access from dams. Available information shows a significant long-term downward trend in abundance for this DPS (NMFS 2009). Population losses

and reduction in abundance have reduced the genetic diversity that existed within the DPS.

The major threats to the survival and recovery of this DPS include dams and other migration barriers, urbanization and channel modification, agricultural activities, non-native predators, hatcheries, large scale water management and diversions, habitat degradation, increased water temperature, and decreased water quality from contaminants including pesticides. Numerous NAWQA, CDPR, and other assessments found high concentration of contaminants in both the San Joaquin and Sacramento Rivers and their tributaries. Monitoring in the San Joaquin basin found seven pesticides exceeded criteria for the protection of aquatic life which included OPs.

The percentage of agriculture, developed, and forested lands that overlap with CCV steelhead habitat are 32%, 10%, and 58%, respectively. Heavy use of agricultural pesticides and the high probability of mixtures increase likelihood of negative effects for this species. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 164. Our GIS analysis indicates that CCV steelhead are exposed to pesticides applied in urban areas and agriculture areas. Juveniles feed and rear in a variety of habitats, including the Sacramento River, the Delta, non-natal intermittent tributaries, tidal marshes, non-tidal freshwater marshes, and other shallow areas in the Delta as rearing areas for short periods during out-migration to the sea. Essentially there is a continuous run of steelhead throughout the year thus adults and rearing juveniles are in freshwater habitats throughout the year. Juveniles typically rear for multiple years in freshwaters where they rely upon a variety of prey dependent on their age and size.

We will be providing additional information that details specific considerations for each decision.

Table 164. California Central Valley steelhead.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Medium	Medium
Diuron	Yes	Yes	NA	Yes	Medium	Medium
Linuron	Yes	NA	NA	Limited	Medium	Medium
Captan	Yes	Yes	NA	NA	Medium	Medium
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

South-Central California Coast (S-CCC) Steelhead (Threatened Species)

The S-CCC steelhead DPS includes all naturally spawned steelhead in streams from the Pajaro River to the Santa Maria River. Population growth rates are unknown, though abundances are very depressed. Generally, juvenile steelhead remain in fresh water for one or more years before migrating downstream to smolt. Steelhead have a more adaptive life history than some of the other species, including overlapping generations, and iteropary. Following emergence, fry rear in smaller tributaries and floodplain habitats

Little information is available on the spatial structure or genetic diversity of this DPS. Because of the lack of information as to which populations are more important to the DPS, we have given the benefit of doubt to the species, and assumed that the populations in the mainstem of the Salinas and Pajaro Rivers, both of which have areas of intensive agriculture and development, are important.

The major obstacles to the survival and recovery of this DPS include dams and other migration barriers, urbanization and channel modification, agricultural activities, wildfires, eroded banks, increased water temperature, and decreased water quality from contaminants.

The percentage of cultivated crop lands and developed lands that overlap with this DPS' range are 7% and 10%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 165. Agriculture is the dominant land use in the Salinas River valley, and there are areas of intense agriculture in the Pajaro watershed as well. Areas higher in the Salinas and Pajaro watersheds and along some of the coastal areas are less affected. Crops for which phorate may be used appeared unlikely to be planted in this area.

We will be providing additional information that details specific considerations for each decision.

Table 165. South-Central California Coast steelhead.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Medium	Medium
Diuron	Yes	Yes	NA	Yes	Medium	Low to medium
Linuron	Yes	NA	NA	Limited	Medium	Medium
Captan	Yes	Yes	NA	NA	Medium	Medium
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Southern California (SC) Steelhead (Endangered Species)

The SC steelhead DPS includes populations in five major and several small coastal river basins in California from the Santa Maria River to the U.S.–Mexican border. Long-term estimates and population trends are lacking for the streams within the DPS. The DPS experiences reduced and fragmented distribution, and large variations in annual spawner runs. Abundance is extremely low. SC steelhead juveniles may rear in fresh water or at

the upper end of coastal lagoons for the first or second summer before migrating downstream to smolt.

The major threats to this DPS include dams and other migration barriers, urbanization and channel modification, agricultural activities, wildfires, and compromised water quality. The NAWQA analysis detected more than 58 pesticides in ground and surface waters within the heavily populated Santa Ana basin.

The percentage of cultivated crop lands and developed lands within SC steelhead habitat are about 5% and 34%, respectively. Co-occurrence of agriculture, forestry, and urban areas with salmonid habitat is shown in Table 166. All of the rivers are affected by anthropogenic inputs.

We will be providing additional information that details specific considerations for each decision.

Table 166. Southern California steelhead.

a.i.	Co-occurrence				Potential for reduction in reproduction, numbers, or distribution	
	Agriculture	Urban/Residential	Forest	Rights-of-way	Populations	Species
2,4-D	Yes	Yes	Yes	Yes	High	high
Triclopyr BEE	Yes	Yes	Yes	Yes	Medium	Low
Diuron	Yes	Yes	NA	Yes	Medium	Medium
Linuron	Yes	NA	NA	Limited	Medium	Medium
Captan	Yes	Yes	NA	NA	Medium	Medium
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Medium

NA: Not applicable

Table 167. Potential for reduction in reproduction, abundance, or distribution.

Species	ESU/DPS	Herbicides				Fungicides	
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chlorothalonil
Chinook	Puget Sound	High	Low	Medium	Low	Low	Medium
	Lower Columbia River	High	Low	Low	Low	Low	Medium
	Upper Columbia River Spring - Run	High	Low	Low	Low	Low	Medium
	Snake River Fall - Run	High	Low	Low	Low	Low	Medium
	Snake River Spring/Summer - Run	High	Low	Low	Low	Low	Medium
	Upper Willamette River	High	Medium	Medium	Medium	Medium	Medium
	California Coastal	High	Low	Low	Low	Medium	Medium
	Central Valley Spring - Run	High	Medium	Medium	Medium	Medium	Medium
	Sacramento River Winter - Run	High	Medium	Medium	Medium	Medium	Medium
Chum	Hood Canal Summer - Run	High	Low	Low	Low	Low	Medium
	Columbia River	High	Low	Low	Low	Low	Medium
Coho	Lower Columbia River	High	Medium	Low	Medium	Medium	Medium
	Oregon Coast	High	Low	Low	Low	Low	Medium
	Southern Oregon and Northern California Coast	High	Low	Low	Low	Low	Medium
	Central California Coast	High	Low	Medium	Low	Medium	Medium
Sockeye	Ozette Lake	High	Low	Low	Low	Low	Medium
	Snake River	High	Low	Low	Low	Low	Medium
Steelhead	Puget Sound	High	Low	Medium	Low	Low	Medium
	Lower Columbia River	High	Low	Low	Low	Low	Medium
	Upper Willamette River	High	Medium	Medium	Medium	Medium	Medium
	Middle Columbia River	High	Low	Low	Low	Medium	Medium
	Upper Columbia River	High	Low	Low	Low	Low	Medium
	Snake River	High	Low	Low	Low	Low	Medium
	Northern California	High	Low	Low	Low	Low	Medium
	Central California Coast	High	Medium	Medium	Low	Medium	Medium
	California Central Valley	High	Medium	Medium	Medium	Medium	Medium
	South-Central California Coast	High	Medium	Medium	Medium	Medium	Medium
	Southern California	High	Low	Medium	Medium	Medium	Medium

Integration and Synthesis for Designated Critical Habitat

The *Integration and Synthesis of Effects to Designated Critical Habitat* section describes NMFS' assessment of the likelihood that EPA's registration of 2,4-D, triclopyr BEE, diuron, linuron, captan and chlorothalonil will destroy or adversely modify designated critical habitat for 26 of 28

ESUs/DPSs covered in this Opinion. Critical habitat has not been designated for the LCR coho salmon and Puget Sound steelhead.

All species addressed in this Opinion have similar PCEs, as described in Table 9 in the *Approach to the Assessment* and as describe in the *Status of Listed Resources*. These PCEs are sites that support one or more life stages and include

1. freshwater rearing sites;
2. freshwater migration corridors;
3. estuarine areas;
4. nearshore marine areas; and
5. offshore marine areas,

which contain physical or biological features essential to the conservation of the ESU/DPS.

Essential physical and biological features include water quality, substrate, prey availability, and natural cover. In the *Risk Characterization* section we concluded that these attributes will likely be reduced in spawning, rearing, migratory, esturine, and nearshore marine habitats. Within this section we evaluate whether these adverse changes to PCEs affect the conservation value of designated critical habitat. Destruction or adverse modification of designated critical habitat is evaluated in this Opinion based on whether the stressors of the action are expected to cause reductions or community modifications in the in- and near-stream plant communities or reductions in water quality that may cause fish to have impaired health or greater susceptibility to other stressors.

As noted in the recovery plans, during all freshwater life stages, salmonids require cool water, free of contaminants. Water free of contaminants promotes normal fish behavior for successful migration, spawning, and juvenile rearing. In the juvenile life stage, salmonids also require stream habitat providing adequate forage. Sufficient forage is necessary for juveniles to maintain growth which subsequently reduces freshwater predation mortality, increases overwintering success, initiates smoltification, and improves their survival at sea.

The stressors of the action include the a.i.s, degradates, inert/other ingredients in formulations, surfactants, and tank mixtures; and their individual and collective interactions when applied in agricultural, urban, and residential landscapes throughout the action area. Most of our quantitative analysis is based on exposure and response data for only the a.i.s, although there may be substantial toxicity from some of the other stressors of the action. Data are not available for some of these other stressors, thus they are evaluated in a qualitative fashion.

We start with the analyses presented in the *Effects of the Proposed Action to Threatened and Endangered Pacific Salmonids* chapter. Modeling EECs and monitoring data are not ESU/DPS specific. Inherent in the modeling used to determine some of the EECs is the assumption that the pesticide is applied in a location next to or draining into designated critical habitat. Monitoring data may reflect pesticide applications proximate to the waterbody, or resulting from more distant uses in the watershed or airshed. In the *Exposure* NMFS used a GIS overlay containing landuse classifications and salmon distributions to determine overlap of application sites and designated critical habitat. Because cropping patterns and registered use sites may change over time, landuse classifications (agricultural, forestry, urban/developed) are used rather than specific crops. Details of the GIS analysis and the maps are provided in *Appendix 5*. Occurrence of land uses where specific a.i.s could be applied near salmon-bearing waters for each ESU/DPS is shown in Table 168 through Table 193.

Based on the risk associated with the specific a.i., co-occurrence of land uses where that a.i. may be applied, other factors described in the *Baseline*, *Status of the Species*, and *Cumulative Effects*, and the conservation values of the watersheds, we determine the potential for use of that a.i. to adversely affect the PCEs. This is expressed qualitatively as low, medium, or high (Table 168 through Table 193).

Taking into account both the unevenness of use of a.i.s, and the conservation value of the various watersheds we then determine the potential for appreciable reduction in that conservation value. We considered the conservation value appreciably reduced if effects were sufficient to cause long-term or permanent shifts in the plant communities, or were anticipated to be temporally

persistent due to chemical properties of the a.i. or frequent inputs and occurred in a significant number of watersheds in the ESU/DPS. Appreciable reductions are described as yes or no [note: We will be providing additional information that details specific considerations for each decision in the final draft], and listed in Table 168 through Table 193. In the *Conclusion* section, we present adverse modification and no adverse modification determinations which are summarized in Table 196.

.

Designated Critical Habitat Specific Evaluations for Each a.i.

Below, we summarize the current status of high and medium conservation value watersheds for each species, including baseline stressors. As exposure to the stressors of the action in salmonid spawning, rearing, and migration habitat is of concern, we highlight exposure from the stressors in shallow, more vulnerable habitats. The number of exposed watersheds that co-occur with agricultural and urban areas is also given. Using both chemical and species habitat information, we determine whether the stressors associated with each a.i. will co-occur and have negative effects on PCEs and if those effects will cause an appreciable decline in the conservation value of that habitat. We present co-occurrence of salmonid PCEs with land use and the six a.i.s in a table format for each species.

Puget Sound Chinook Salmon

Of 61 assessed watersheds (HUC 5), 40 and 9 are of high and medium conservation value, respectively. Nineteen nearshore marine areas are also of high conservation value. Of the high value conservation watersheds, 32 and 40 are exposed to pesticides from agriculture and urban land uses, respectively. Among the medium value watersheds, six and nine are exposed to pesticides from agriculture and urban land uses, respectively. These areas serve as spawning, rearing, and migration habitat for Puget Sound Chinook salmon.

Migration, spawning, and rearing PCEs in upper watersheds of most river systems, and in the lower alluvial valleys of mid- to southern Puget Sound and the Strait of Juan de Fuca have been heavily altered by forestry, agriculture, and urban land uses. These activities have resulted in the loss of floodplain habitat, reduced substrate conditions for spawning and incubation, and degraded water quality. Estuary PCEs in the northwest Puget Sound are also degraded from impaired water quality (*e.g.*, contaminants), altered salinity conditions, lack of natural cover, and modification of and lack of access to tidal marshes and their channels.

Spawning, rearing, and migration PCEs in these exposed watersheds likely experience a reduction in water quality and prey abundance in freshwater, estuarine, and nearshore areas, especially during allowable pesticide applications adjacent to or within Puget Sound Chinook salmon habitat. Based on concentrations of agriculture and developed areas in the Lower Skagit Valley, we expect reductions in spawning, rearing, and migration PCEs in watersheds for this valley. As elevated water temperature prevents this ESU from inhabiting about 374 km of streams within its range, suitable PCE conditions in remaining available species habitat become important for ensuring long-term species conservation.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 168. More than 50% of the ESU is composed of evergreen, deciduous, or mixed forests. Other pesticide use areas include urban/residential development (15%) and agricultural uses (4%). Cultivated crops (1%) and hay crops and pastures (3%) are primarily distributed on the floodplain and other lowland habitats. The majority of urban/residential land use also occurs within river and stream valleys in lowland areas, much of the nearshore marine area also consists of urban/residential.

We will be providing additional information that details specific considerations for each decision.

Table 168. Co-occurrence of land use types and likelihood of a reduction in PCEs for Puget Sound Chinook salmon.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	High
Triclopyr BEE	Yes	Yes	Yes	Yes	Low to Medium	No
Diuron	Yes	Yes	NA	Yes	Medium	Yes
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Lower Columbia River (LCR) Chinook Salmon

Thirty-one and 13 watersheds are of high and medium conservation value, respectively. Four additional unoccupied watersheds received a “possibly high” rating for species conservation as well. Our GIS analysis indicates 26 of 31 high conservation value watersheds are exposed to pesticide applications from agriculture and urban land uses, respectively. All 13 medium conservation watersheds are also exposed to pesticide applications from both land uses.

Spawning and rearing PCEs for LCR Chinook salmon have been degraded by timber harvests, agriculture, and urbanization. These land uses have reduced floodplain connectivity and water quality, and removed natural cover in several rivers. Hydropower development projects have also reduced the timing and magnitude of water flows, thereby altering required water quantity to form and maintain physical habitat conditions for juvenile fish growth and mobility. Migration PCEs are also affected by several dams along the migration route used by adult and juvenile fish. The survival of yearlings in the ocean is also affected by habitat conditions in the estuary, such as changes in food availability and the presence of contaminants.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated on Table 169. The percentage of agriculture lands that overlap with LCR Chinook salmon is about 6% and 2% as

cultivated crops and 4% as hay/pasture. More than 76% of the ESU is composed of evergreen, deciduous forest, and mixed forests. Urban/residential development (13%) is a fairly substantial portion of this ESU. Most of the highly developed land and agricultural areas in this ESU's range are adjacent to salmonid habitat.

Spawning and migration PCEs in these exposed watersheds, as well as the river mainstem, and upstream tributaries likely experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems. As elevated water temperature prevents LCR Chinook salmon from inhabiting about 275 km of streams within its range, suitable PCE conditions in available species habitat are important for ensuring long-term species conservation.

We will be providing additional information that details specific considerations for each decision.

Table 169 Co-occurrence of land use types and likelihood of a reduction in PCEs for Lower Columbia River Chinook salmon.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Medium	Yes
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Upper Columbia River (UCR) Spring-run Chinook Salmon

Twenty-six and five watersheds are of high and medium conservation value, respectively. Our GIS analysis indicates 23 and 26 high conservation watersheds are exposed to pesticide

applications from agriculture and urban land uses, respectively. All medium conservation value watersheds are also exposed to pesticides from both land uses.

Urbanization in lower reaches, irrigation and diversion in the major upper drainages, and grazing in the middle reaches have degraded spawning and rearing PCEs in tributary systems. Migration PCEs for adult and juvenile fish are heavily degraded by Columbia River federal dam projects and a number of mid-Columbia River Public Utility District dam projects.

Co-occurrence of agricultural, urban/residential, and forestry land uses is indicated on Table 170. The percentage of agricultural and developed lands that overlap with UCR Chinook salmon habitat is about 5.4% and 4.7%, respectively. Forested lands compose about 45% of the ESU. Fish spawn and rear in the major tributaries leading to the Columbia River between Rock Island and Chief Joseph dams. Spawning, rearing, and migration PCEs likely experience a reduction in water quality and prey abundance especially in freshwater tributaries and shallow low flow floodplain habitats during allowable applications adjacent to UCR Spring-run Chinook salmon habitat.

We will be providing additional information that details specific considerations for each decision.

Table 170. Co-occurrence of land use types and likelihood of a reduction in PCEs for Upper Columbia River Chinook Salmon.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Low	No
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Snake River (SR) Fall-run Chinook Salmon

Individual watersheds within the range of SR Fall-run Chinook salmon have not been evaluated by the CHART team for their conservation value. However, the Lower Columbia River corridor is of high conservation value as it connects several populations with the ocean and is used by rearing/migrating juveniles and migrating adults. The Columbia River estuary is also a unique and essential area for juveniles and adults making the physiological transition between life in freshwater and marine habitats.

Baseline conditions for this ESU include reduced spawning habitat and impaired stream flows and barriers to fish passage in tributaries from hydroelectric dams. Stream water quality and biological communities in the downstream portion of the upper Snake River basin are also degraded. We note that elevated water temperature currently prevents SR Fall-run Chinook salmon from inhabiting 2,401 km of streams within its range.

In lieu of CHART data on the conservation value ratings of salmonid watersheds, we recognize that all watersheds within the range of SR Fall-run Chinook salmon are of high conservation value. We used GIS data to assess the overlap between spawning and migration PCEs and use sites and their exposure in the Columbia River estuary and migratory corridor.

Co-occurrence of agriculture, forestry, and urban areas with salmonid habitats is shown in Table 171. Pesticide use areas for the 11 a.i.s within the range for this ESU and above the Columbia River migratory corridor include evergreen forests (49%), cultivated crops (15%), pastures (1%), and developed lands (1%).

We will be providing additional information that details specific considerations for each decision.

Table 171. Co-occurrence of land use types and likelihood of a reduction in PCEs for Snake River Fall-run Chinook Salmon.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Low	No
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Sneke River (SR) Spring/Summer-run Chinook Salmon

Watersheds within the range of SR Spring/Summer-run Chinook salmon were not evaluated by the CHART team for their conservation value. However, the Lower Columbia River is of high conservation value as it connects every population with the ocean and is used by rearing/migrating juveniles and migrating adults. Juveniles of this ESU rely on adequate fresh water quality and prey abundance for migrating and rearing in freshwater habitats including migratory routes from natal reaches leading to alternative summer-rearing or overwintering areas.

Spawning and juvenile PCEs are regionally degraded by changes in flow quantity, water quality, and loss of cover. Juvenile and adult migrations are obstructed by reduced access stemming from altered flow regimes from hydroelectric dams. As elevated water temperature prevents SR Spring/Summer-run Chinook salmon from inhabiting 1,596.3 km of streams within its range, suitable PCE conditions in remaining species habitat become important for ensuring the long-term conservation for this species.

Co-occurrence of agriculture, urban/residential, and forestry land use is shown in Table 172. This ESU spawns and rears primarily in the smaller tributaries, many of which are located on U.S. Forest Service lands. Agricultural and urban areas are not common in the watersheds

comprising the ESU, and those that are present are clustered mostly around the mainstem Snake and Columbia Rivers. The Snake River is a high-volume, high-flow system, and salmon use it primarily as a migratory corridor.

We will be providing additional information that details specific considerations for each decision.

Table 172. Co-occurrence of land use types and likelihood of a reduction in PCEs for Snake River Spring/Summer-run Chinook Salmon.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Low	No
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Upper Willamette River (UWR) Chinook Salmon

Of 59 assessed watersheds, 22 and 18 are of high and medium conservation value, respectively. The lower Willamette/Columbia River rearing/migration corridor downstream of the spawning range is also of high conservation value. Our GIS analysis indicates 15 and 19 high conservation watersheds are exposed to pesticide applications from agriculture and urban land uses, respectively. Of the medium conservation watersheds, 13 and 12 are also exposed to pesticide applications from the above respective land uses.

Migration and rearing PCEs have been degraded by dams altering migration timing and water management. Migration, rearing, and estuary PCEs are also degraded by the loss of riparian vegetation and instream cover. Water quality is also degraded in floodplain rearing habitat along the lower Willamette River. As elevated water temperature prevents UWR Chinook salmon

from inhabiting 2,468 km of waters within its range, PCE conditions in remaining species habitat are important for ensuring long-term conservation for this species.

Co-occurrence of agriculture, urban/residential, and forestry land uses is indicated on Table 173. The percentage of cultivated and develop lands that overlap with UWR Chinook salmon habitat are 10.5% and 9%, respectively. Spawning, rearing, and migration freshwater (PCEs in these exposed watersheds (including mainstem and floodplain wetlands) likely experience reductions in water quality and prey abundance.

We will be providing additional information that details specific considerations for each decision.

Table 173. Co-occurrence of land use types and likelihood of a reduction in PCEs for Upper Willamette River Chinook Salmon.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	Medium	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Medium	No
Diuron	Yes	Yes	NA	Yes	Medium	Yes
Linuron	Yes	NA	NA	Limited	Medium	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

California Coastal (CC) Chinook Salmon

Of 45 occupied watersheds, 27 and 10 are of high and medium conservation value, respectively. Our GIS analysis indicates 8 and 27 high conservation watersheds are exposed to pesticides from agriculture and urban land uses, respectively. Of the medium conservation watersheds, 4 and 10 are exposed to pesticide applications from the above respective land uses.

The spawning PCE in coastal streams have been degraded from timber harvests. Rearing and migration PCEs in the Russian River have also been impacted by agriculture and urban areas. Water management for dams within the Russian and Eel River watersheds maintain high flows and warm water during summer which indirectly benefits the introduced Sacramento pikeminnow, a predatory fish on CC Chinook salmon along migration corridors. The estuary PCE has also been degraded from breaches of the sandbar at the mouth of the Russian River causing periodic mixing of salt water. This condition alters the water quality and salinity conditions for the juvenile physiological transitions between fresh and salt water. Current PCE conditions likely maintain a low population abundance across the ESU.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated on Table 174. The percentage of cultivated croplands and developed lands that overlap with CC Chinook salmon habitat are 1% and 5.4%, respectively. There is substantial overlap with use sites in the Russian River watershed. Rearing and migration PCEs in freshwater and estuaries in this watershed likely experience reductions in water quality and prey abundance.

We will be providing additional information that details specific considerations for each decision.

Table 174. Co-occurrence of land use types and likelihood of a reduction in PCEs for Central California Chinook Salmon.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Low	No
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Central Valley (CV) Spring-run Chinook Salmon

Of 38 occupied watersheds, 28 and 3 are of high and medium conservation value, respectively. Four of these watersheds comprise portions of the San Francisco-San Pablo-Suisun Bay estuarine complex which provides rearing and migratory habitat for CV Spring-run Chinook salmon. Our GIS analysis indicates 17 and 28 high conservation value watersheds are exposed to pesticides from agriculture and urban land uses, respectively. Of the medium conservation watersheds, two and three watersheds are exposed to pesticide applications from the above land uses as well.

Spawning and rearing PCEs are currently degraded by elevated water temperature and lost access to historic spawning areas in upper watersheds with cool and clean water throughout the summer. The rearing PCE is degraded and is affected by loss of floodplain habitat connectivity from the mainstem of larger rivers through the Sacramento River watershed, thereby reducing effective foraging. The migration PCE is degraded by lack of natural cover along the migration corridors. Juvenile migration is further obstructed by water diversions along the Sacramento River and by two large state and federal water-export facilities in the Sacramento-San Joaquin Delta. Agriculture and urban runoff containing a suite of pollutants further impair water quality of receiving systems used by this species.

Intensive agricultural development occurs in the California Central Valley and may impact waters draining into the Sacramento River. We further expect rearing and migration PCEs in non-natal tributaries, intermittent streams, and floodplain habitats may also experience likely reductions in water quality and prey abundance.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated on Table 175. The percentage of cultivated croplands and developed lands that overlap with CV Chinook salmon habitat are 21.3% and 10.8%, respectively. Migration PCEs in the San Francisco-San Pablo-Suisun Bay estuaries complex, which are heavily influenced by input from California's Central Valley likely experience reductions in water quality and prey abundance.

We will be providing additional information that details specific considerations for each decision.

Table 175. Co-occurrence of land use types and likelihood of a reduction in PCEs for Central Valley Spring-run Chinook Salmon.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	Medium	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Medium	No
Diuron	Yes	Yes	NA	Yes	Medium	Yes
Linuron	Yes	NA	NA	Limited	Medium	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Sacramento River Winter-run Chinook Salmon

Individual subbasins or river sections were not evaluated for their conservation value. However, the entire Sacramento River and the Delta are considered of high conservation value for spawning, rearing, and migration.

Spawning and rearing PCEs are currently degraded by elevated water temperature and lost access to historic spawning areas in upper watersheds with cool and clean water throughout the summer. The rearing PCE is degraded and is affected by loss of floodplain habitat connection from the mainstem of larger rivers through the Sacramento River watershed, thereby reducing effective foraging. The migration PCE is degraded by lack of natural cover along the migration corridors. Juvenile migration is further obstructed by water diversions along the Sacramento River and by two large state and federal water-export facilities in the Sacramento-San Joaquin Delta.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated on Table 176. The percentage of cultivated croplands and developed lands that overlap with SR Winter-run Chinook salmon are 25% and 10%, respectively. As agriculture and urban land uses occur in the Sacramento River watershed and in the Sacramento-San Joaquin Delta, we expect rearing and spawning PCEs in floodplain habitat and the Sacramento River may experience reductions in water quality and prey abundance.

We will be providing additional information that details specific considerations for each decision.

Table 176. Co-occurrence of land use types and likelihood of a reduction in PCEs for Sacramento River Winter-run Chinook Salmon.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Medium	No
Diuron	Yes	Yes	NA	Yes	Medium	Yes
Linuron	Yes	NA	NA	Limited	Medium	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Hood Canal Summer-run Chum Salmon

Of 12 assessed watersheds, nine and three are of high and medium conservation value, respectively. Five nearshore marine areas were also rated as high conservation value. Many of the watersheds have less than four miles of spawning habitat and none are greater than 8.5 miles in length. Our GIS analysis indicates seven and nine high conservation value watersheds are exposed to pesticides from agriculture and urban land uses, respectively. All three medium conservation watersheds are exposed to both land uses as well.

The spawning PCE is degraded by excessive fine sediment in gravel. The rearing PCE is degraded by loss of access to sloughs in the estuary and nearshore areas and excessive predation.

Migration and rearing PCEs in estuaries are impaired by the loss of functional floodplain areas. These degraded conditions likely maintain low population abundance across the ESU.

Co-occurrence with agriculture, urban/residential, and forestry land uses in Table 177. Land use within the range of this ESU is predominantly forested (73%), open water (17%), urban/residential (9%), and agriculture (2%). The percentage of cultivated croplands and developed lands that overall with HC Summer-run chum salmon habitat is about 9.94% and 8.9%, respectively. Most of the agriculture and urban/residential uses occur within rivers and stream valleys in lowland areas. Nearshore marine areas are frequently adjacent to urban/residential areas. Given these uses, spawning and migration PCEs in streams, estuaries, and nearshore marine areas may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

We will be providing additional information that details specific considerations for each decision.

Table 177. Co-occurrence of land use types and likelihood of a reduction in PCEs for Hood Canal Summer-run Chum Salmon.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Low	No
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Columbia River (CR) Chum Salmon

Of 19 assessed watersheds, 16 and 3 are of high and medium conservation value, respectively. Our GIS analysis indicates all high and medium conservation value watersheds are exposed to

pesticide applications from agriculture, developed areas, and forestry adjacent to CR chum salmon habitat.

The migration PCE for this species has been significantly impacted by dams obstructing adult migration and access to historic spawning sites. Water quality and cover for estuary and rearing PCEs have decreased and are not likely to maintain their intended function to conserve the species. Elevated water temperature further prevents CR chum salmon from inhabiting 272.8 km of waters within its range.

Co-occurrence of agriculture, urban/residential, and forestry land uses is indicated on Table 178. The percentage of cultivated croplands, hay/pasture, and developed lands that overlap with CR chum salmon habitat is 2%, 5%, and 15%, respectively. More than 50% of the range of the ESU is covered by deciduous, evergreen, or mixed forests. Within the ESU, agricultural and development are predominantly distributed in the low-lying areas near the Columbia River and its tributaries. Given these uses the rearing and migration PCEs along the edges of the mainstem or in tributaries and side channels of freshwater and estuarine systems may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

We will be providing additional information that details specific considerations for each decision.

Table 178. Co-occurrence of land use types and likelihood of a reduction in PCEs for Columbia River Chum Salmon.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Low	No
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Oregon Coast (OC) Coho Salmon

Of 80 watersheds, 45 and 27 are of high and medium conservation value, respectively. Our GIS analysis indicates 39 and 44 high conservation watersheds are exposed to pesticides from agriculture and urban areas, respectively. Of the medium conservation watersheds, 18 and 23 are exposed to pesticide applications from the above respective land uses.

The rearing PCE has been degraded by elevated water temperature in 29 of the 80 HUC 5 watersheds. Elevated temperature further prevents OC coho salmon from inhabiting 3,716 km of waters within its range. Twelve watersheds have reduced water quality from contaminants and excessive nutrition.

Co-occurrence of agriculture, urban/residential, and forestry land uses is indicated in Table 179. The percentage of cultivated croplands and developed lands that overlap with OC coho salmon habitat are 0.23% and 6.6%, respectively. Most of the cropland is hay/pasture and is primarily located in the Umpqua watersheds. Given these uses, we expect a low likelihood of freshwater rearing PCE in small streams to experience reductions in water quality and prey abundance.

We will be providing additional information that details specific considerations for each decision.

Table 179. Co-occurrence of land use types and likelihood of a reduction in PCEs for Oregon Coast Coho Salmon.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Low	No
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Southern Oregon/Northern California Coast (SONCC) Coho Salmon

Although watersheds within this ESU were not evaluated for their conservation value, the northern coastal streams that are designated as critical habitat are of good quality. Throughout this ESU's range, the spawning PCE has been degraded by fines in spawning gravel from logging. The rearing PCE has been considerably degraded in many inland watersheds by the loss of riparian vegetation, resulting in unsuitable high temperatures. Rearing and migration PCEs have been reduced by the disconnection of floodplain and off-channel habitats in low gradient reaches of streams. Elevated water temperature further prevents SONCC coho salmon from inhabiting 3,249.2 km of waters within its range.

Co-occurrence of agriculture, urban/residential, and forestry land uses is indicated on Table 180. The percentage of cultivated croplands and developed lands that overlap with SONCC coho salmon habitat are 2.5% and 4.3%, respectively. Areas with more cropland include the Scott and Shasta watersheds in the Klamath basin and the Upper and Middle rough River watersheds. Of the development in this ESU, much is in the rough River basin, with remaining development distributed along the coastline and estuaries. Given these uses, we expect rearing and migration PCEs in backwater, side channels, and shallow channel edge fish habitat may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

We will be providing additional information that details specific considerations for each decision.

Table 180. Co-occurrence of land use types and likelihood of a reduction in PCEs for Southern Oregon Northern California Coast Coho Salmon.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Low	No
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Central California Coast (CCC) Coho Salmon

Individual watersheds have not been evaluated for their conservation value. Nevertheless, there is a distinct trend of increasing degradation in quality and quantity of all PCEs as the habitat progresses south through the species range along the Lost Coast to Navarro Point and the Santa Cruz Mountains. Spawning and incubation substrate and juvenile rearing habitat are generally degraded.

Co-occurrence of agriculture, urban/residential, and forestry land uses is indicated on Table 181. The percentage of cultivated croplands and developed lands that overlap with CCC coho salmon habitat are 2.3% and 9.4%, respectively. Much of the development is centered around San Francisco Bay, and developed and agricultural areas also occur in the Russian River watershed. The northern, undeveloped watersheds around the Navarro and Big Rivers are used by the majority of this species. Given these land uses, we expect the freshwater rearing PCE may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to freshwater systems.

We will be providing additional information that details specific considerations for each decision.

Table 181. Co-occurrence of land use types and likelihood of a reduction in PCEs for Central California Coast Coho Salmon.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Medium	No
Linuron	Yes	NA	NA	Limited	Medium	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Ozette Lake Sockeye Salmon

The Ozette Lake watershed is of high conservation value. The entire circumference of the lake is within Olympic National Park. Ozette Lake and portions of three tributaries support spawning and rearing PCEs. Ozette River supports rearing and migration PCEs; its river mouth also provides estuarine habitat. Migration habitat is also affected by low water flow in summer and elevated water temperature which pose as a thermal barrier for migration.

Spawning habitat has been affected by the loss of tributary spawning areas, low water levels in summer, and vegetation and sediment that have reduced the quantity and suitability of beaches for spawning. The rearing PCE is degraded by excessive predation, competition with non-native species, and loss of rearing habitat. Migration habitat is affected by high water temperatures and low water flows in summer.

Co-occurrence of urban/residential and forestry uses is indicated on Table 182. Ozette Lake is in a sparsely populated area, with less than 1% of land developed within the range of this ESU. Similarly, there is no cultivated cropland. However, salmonid habitat may be at risk of exposure from forestry-related uses. Land use is primarily forest with private, state, and federal ownership

(86% forested, 13% open water, 1% developed land, 0% agriculture). The predominant pesticide use sites (*i.e.*, urban/residential and forestry) overlap with the Lake's freshwater tributaries.

Thus, the greatest risk of exposure to freshwater PCEs are in tributary habitats. However, we do not expect a reduction in prey abundance within these tributaries. Although private residences along tributaries may have small, non-commercial crops for pesticide applications, it is unlikely that restricted use pesticides would be applied.

We will be providing additional information that details specific considerations for each decision.

Table 182. Co-occurrence of land use types and likelihood of a reduction in PCEs for Ozette Lake Sockeye Salmon.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	No	Yes	Yes	Yes	High	Yes
Triclopyr BEE	No	Yea	Yes	Yes	Low	No
Diuron	No	Yes	NA	Yes	Low	No
Linuron	No	NA	NA	Limited	NA	No
Captan	No	Yes	NA	NA	Low	No
Chlorothalonil	No	Yes	Yes	NA	Low	No

Snake River Sockeye Salmon

Conservation values of individual watersheds have not been reported. Nevertheless, all areas occupied and used by migrating SR sockeye are considered of high conservation value as this species is limited to a single lake within the SR basin.

The quality and quantity of rearing and migration PCEs have been reduced by land uses that disrupt access to foraging areas, increase the amount of fines in the stream substrate, and reduce instream cover. Water quality is impaired by a suite of anthropogenic pollutants which enter surface waters and riverine sediments from the headwaters of the Salmon River to the Columbia River estuary. The migration PCE is also affected by four dams in the SR basins that obstructs

migration and increases mortality of downstream migrating juveniles. Given the migration distance traveled by this species, adequate passage conditions (water quality and quantity available at specific times) is critical.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 183. About 1% of the land surrounding Red Fish Lake has been developed, and another 1% is used for agriculture, primarily hay and pasture. More than 50% of range of this ESU is in evergreen forests. Consequently, forestry uses are the major source of exposure in spawning and rearing habitats. Given the limited uses around Red Fish Lake, we expect the migration PCE may experience some reductions in water quality and abundance during allowing pesticide applications adjacent to the Snake and Columbia Rivers leading to the ocean.

We will be providing additional information that details specific considerations for each decision.

Table 183. Co-occurrence of land use types and likelihood of a reduction in PCEs for Snake River Sockeye Salmon.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Low	No
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Low	No

Lower Columbia River Steelhead

Of 41 watersheds listed as critical habitat for LCR steelhead, 28 and 11 are of high and medium conservation value, respectively. Our GIS analysis indicates 21 and 26 high conservation watersheds are exposed to pesticides from agriculture and urban/residential land uses,

respectively. Of the medium conservation watersheds, 11 and 10 are also exposed to pesticide applications from the above respective land uses.

The water quality of the rearing PCE within the lower portion and alluvial valleys of many watersheds has been degraded by agricultural runoff into tributaries reaches and the mainstem Columbia River. Consequently, invertebrate production in these aquatic systems is also affected. Elevated water temperature further prevents LCR steelhead from inhabiting 341.5 km of waters within its range.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 184. The percentage of cultivated crop lands that overlap with LCR steelhead is about 7%, with 4.5% as hay/pasture land and 2.5% as cultivated cropland. More than 61% of the range of this DPS is composed of evergreen, deciduous forest, and mixed forests. Urban/residential development lands (12%) were a fairly substantial portion of this DPS. Given these uses, we expect the freshwater rearing PCE in floodplain habitats, and natal rivers and streams, may experience reductions in water quality and prey abundance during allowable pesticide applications to these systems.

We will be providing additional information that details specific considerations for each decision.

Table 184. Co-occurrence of land use types and likelihood of a reduction in PCEs for Lower Columbia River Steelhead.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Medium	Yes
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Upper Willamette River Steelhead

Of the watersheds assessed, 14 and 6 are of high and medium conservation value, respectively. Our GIS analysis indicates all high and medium conservation value watersheds are exposed to pesticide applications from agriculture and urban areas adjacent to UWR steelhead critical habitat.

Existing water quality necessary for juvenile rearing within many watersheds have been impaired by pollutants in agricultural runoff. Consequently, invertebrate production for salmonids in several watersheds and in the mainstem Columbia River is affected. As several dams obstruct migrating fish along the migratory corridor, the migration PCE is also reduced by these features. Elevated water temperature further prevents UWR steelhead from inhabiting 1,668 km of waters within its range.

Co-occurrence of agriculture, urban/residential, and forestry land uses is indicated in Table 185. The percentage of cultivated crop land and developed lands are 14.5% and 10%, respectively. Given these uses, we expect the freshwater rearing PCE in floodplain habitats, rivers, and streams may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

We will be providing additional information that details specific considerations for each decision.

Table 185. Co-occurrence of land use types and likelihood of a reduction in PCEs for Upper Willamette River Steelhead.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	Medium	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Medium	Yes
Linuron	Yes	NA	NA	Limited	Medium	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Middle Columbia River Steelhead

Of the 106 assessed watersheds, 73 and 24 are of high and medium conservation value, respectively. The lower Columbia River rearing/migration corridor downstream of the spawning range is also of high conservation value. Our GIS analysis indicates 67 and 68 high conservation watersheds are exposed to pesticides from agriculture and urban areas, respectively. Of the medium conservation watersheds, 23 and 24 watersheds are also exposed to pesticide applications from the above respective land uses.

The current condition of critical habitat for MCR steelhead is moderately degraded. The water quality attribute for the rearing PCE within many watersheds is reduced. Consequently, invertebrate production in these watersheds and in the mainstem Columbia River is also reduced. Loss of riparian vegetation to grazing has resulted in elevated water temperature in the John Day Basin. Elevated water temperature prevents MCR steelhead from inhabiting 3,727.9 km of waters within its range. In the Yakima River, 72 streams and river segments are also listed as impaired waters and 83% exceed temperature standards. As several dams obstruct fish along their migratory corridor, these features further degrade the migration PCE.

Co-occurrence of agriculture, urban/residential, and forestry land uses is indicated on Table 186. The percentage of cultivated crop lands and developed lands within the range of this DPS are 17% and 3%, respectively. Given the continuous run of steelhead throughout the year, the conditions of the rearing PCE in freshwater habitat is important for adult and rearing juveniles. Given the above land uses, the freshwater rearing PCE in shallow water along the banks of streams or aquatic habitats on stream margins are vulnerable to reductions in water quality and prey abundance during allowable pesticide applications to these systems.

We will be providing additional information that details specific considerations for each decision.

Table 186. Co-occurrence of land use types and likelihood of a reduction in PCEs for Middle Columbia River Steelhead.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Low	No
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Upper Columbia River Steelhead

Of the 41 watersheds occupied by UCR steelhead, 31 and 7 are of high and medium conservation value, respectively. The lower Columbia River rearing/migration corridor downstream of the species' spawning range is also of high conservation value. Our GIS analysis indicates 28 and 31 high conservation watersheds are exposed to pesticides from agriculture and urban areas, respectively. All seven medium conservation watersheds are exposed to pesticide applications from the above land uses.

The current condition of UCR steelhead critical habitat is moderately degraded. Habitat quality in tributary streams range from excellent to poor. Water quality for the rearing PCEs within many watersheds has been reduced from agriculture runoff. Consequently, invertebrate production in several watersheds and in the mainstem Columbia River is also reduced. Several dams obstruct fish migrating through the migratory corridor and further impact the migration PCEs.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 187. The percentage of cultivated crop lands and developed lands within the range of this DPS are 13% and 4%, respectively. There is some agriculture in the spawning and rearing areas in the Wenatchee, Methow, and Okenogan watersheds. Intense agriculture occurs outside the 2.5 km

buffer and in the Upper Columbia Irrigation District within the Entiat watershed. The water is heavily used and re-used for irrigation. We expect reductions in water quality and prey abundance for the migration PCE along the Columbia River, where the valley is heavily agricultural. Reductions in water quality and prey abundance are also expected for the freshwater rearing PCE during allowable pesticide applications adjacent to stream margins or cascades.

We will be providing additional information that details specific considerations for each decision.

Table 187. Co-occurrence of land use types and likelihood of a reduction in PCEs for Upper Columbia River Steelhead.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Low	No
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Snake River Basin Steelhead

Of the watersheds assessed, 229 and 41 are of high and medium conservation value, respectively. The Columbia River migration corridor is also of high conservation value. Our GIS analysis indicates 163 and 99 high conservation watersheds are exposed to pesticides from agriculture and urban areas, respectively. Of the medium conservation watersheds, 34 and 28 are also exposed to pesticide applications from the above land uses.

The current condition of SR basin steelhead critical habitat is moderately degraded. Water quality conditions for rearing PCEs within many watersheds have been degraded from contaminants in agricultural runoff. Consequently, invertebrate communities in several

watersheds and in the mainstem Columbia River are negatively impacted. Loss of riparian vegetation to grazing has resulted in elevated water temperature in the John Day basin. These conditions have reduced the rearing PCE. As several dams obstruct adult fish migrating along the migratory corridor, the migration PCE is also negatively impacted. Elevated water temperature further prevents SR basin steelhead from inhabiting 3,282 km of waters within its range.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 188. Land uses within the range of this DPS include evergreen forests (52%), agricultural lands including cultivated crops (8%), and hay/pasture (1%), and urban/residential or developed areas (2%). Given these land uses, the rearing PCE in freshwater habitats may be exposed during allowable pesticide applications adjacent to exposed systems. Consequently, we expect reductions in the water quality and prey abundance in these systems.

We will be providing additional information that details specific considerations for each decision.

Table 188. Co-occurrence of land use types and likelihood of a reduction in PCEs for Snake River Basin Steelhead.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Low	No
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Northern California Steelhead

Of the 50 assessed watersheds, 27 and 14 are of high and medium conservation value, respectively. Two estuarine habitat areas used for rearing and migration (Humboldt Bay and the

Eel River Estuary) are also of high conservation value. Our GIS analysis indicates 10 and 27 high conservation watersheds are exposed to agriculture and urban areas, respectively. Of the medium conservation watersheds, 2 and 14 are also exposed to pesticide applications from the same above land uses, respectively.

The current condition of critical habitat for NC steelhead is moderately degraded. Removal of riparian vegetation within portions of its range promotes elevated water temperature and consequently affects the rearing PCE in freshwater and estuaries. Spawning PCE attributes such as the quality of substrate supporting spawning, incubation, and larval development are degraded by silt and sediment fines in the spawning gravel. Access to tributaries in many watersheds is affected by bridges, culverts, and forest road construction. Consequently, these uses reduce the function of the migration PCE for adults.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated on Table 189. The percentage of cultivated crop lands and developed lands overlapping with NC steelhead habitat are less than 1% and 19%, respectively. There are few areas of concentrated agriculture and most appear to be hay/pasture and are concentrated in the Lower Eel watershed and some of the other coastal valleys. Development is concentrated primarily near Eureka, on the coast in the Mad River and Redwood Creek watersheds. Much of the land area in this DPS is heavily forested, and there is a number of state and national parks. Given these land uses we expect the rearing PCE in freshwater streams and lagoons may experience some reductions in water quality and prey abundance during allowing pesticide applications adjacent to these systems.

We will be providing additional information that details specific considerations for each decision.

Table 189. Co-occurrence of land use types and likelihood of a reduction in PCEs for Northern California Steelhead.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Low	No
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Central California Coast (CCC) Steelhead

Of 47 occupied watersheds, 19 and 15 are of high and medium conservation value, respectively. Our GIS analysis indicates 12 and 15 high conservation watersheds are exposed to pesticide applications from agriculture and urban areas, respectively. Of the medium conservation watersheds, 8 and 13 are also exposed to the above land uses areas, respectively. Throughout the species' range, habitat conditions and quality have been degraded by a lack of channel complexity, eroded banks, turbid and contaminated water, low summer flow and high water temperatures, multiple contaminants found at toxic levels, and restricted access to cooler head waters from migration barriers.

The current condition of designated critical habitat for CCC steelhead is poor. The spawning PCE is impacted by sediment fines in the spawning gravel, which limits the production of aquatic stream insects adapted to running water. Elevated water temperature and impaired water quality have further reduced the quality, quantity, and function of the rearing PCE within most streams.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 190. The percentage of cultivated croplands and developed lands that overlap with CCV steelhead habitat

are 27% and 10%, respectively. High densities of crop farming occur throughout the San Joaquin Basin, the Delta, and along the lower Sacramento River. Agriculture also occurs in the Russian River valley. Most of the watersheds in this DPS are heavily developed, and/or have intensive agriculture in the river valley. Given these land uses, rearing and migration PCEs in small freshwater tributaries and floodplains and the San Francisco-San Pablo-Suisan Bay estuarine complex may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

We will be providing additional information that details specific considerations for each decision.

Table 190. Co-occurrence of land use types and likelihood of a reduction in PCEs for Central California Coast Steelhead.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Medium	Yes
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

California Central Valley (CCV) Steelhead

Of 67 occupied watersheds, 37 and 18 are of high and medium conservation value, respectively. Our GIS analysis indicates 24 and 37 high conservation watersheds are exposed to pesticide applications from agriculture and urban areas, respectively. Of the medium conservation watersheds, 14 and 17 watersheds are exposed to pesticide applications from the above land uses, respectively.

The current condition of CCV steelhead critical habitat is degraded and does not function well for ensuring species recovery. The Sacramento-San Joaquin River Delta serves little function for

juvenile CCV steelhead rearing and their physiological transition to salt water. Water flow and temperature, especially during the summer months affect the condition of the spawning PCE in floodplains and flood bypasses. The rearing PCE is degraded by channelized, leveed, and riprapped river reaches and sloughs in the Sacramento-San Joaquin system. Stream channels commonly have elevated water temperature. The current condition of migration corridors is poor. Both migration and rearing PCEs are affected by dense urbanization and agriculture along the mainstems and in the Delta which contribute to reduced water quality from contaminants in runoff. The RBDD gates obstruct migrating juveniles and adults. State and federal government pumps and associated fish facilities alter flow in the Delta and consequently obstruct migrations along the migratory corridor.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 191. The percentage of agriculture, developed, and forested lands that overlap with CCV steelhead habitat are 32%, 10%, and 58%, respectively. Heavy uses of agricultural pesticides and the high probability of mixtures increase the likelihood of negative effects on PCEs and critical habitat. AS there is a continuous run of steelhead throughout the year, the conditions of the rearing PCE in a variety of habitat are important for this DPS. Given these land uses, freshwater rearing and migration PCEs in the Sacramento River, the Delta, tributaries, tidal and non-tidal marshes, and other shallow areas in the Delta may experience reductions in water quality and prey abundance during allowable pesticide applications adjacent to these systems.

We will be providing additional information that details specific considerations for each decision.

Table 191. Co-occurrence of land use types and likelihood of a reduction in PCEs for California Central Valley Steelhead.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Medium	No
Diuron	Yes	Yes	NA	Yes	Medium	Yes
Linuron	Yes	NA	NA	Limited	Medium	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

South-Central California Coast (S-CCC) Steelhead

Of 29 occupied watersheds, 12 and 11 are of high and medium conservation value, respectively. Our GIS analysis indicates all high conservation watersheds are exposed to pesticide applications from agriculture and urban areas. Of the medium conservation watersheds, 9 and 11 watersheds are exposed to pesticide applications from agriculture and urban areas, respectively.

Migration and rearing PCEs are degraded throughout critical habitat by elevated water temperature and contaminants from urban and agricultural runoff. The estuarine PCE is further affected when estuaries are breached and receive contaminant inputs from runoff.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 192. The percentage of cultivated crop land and developed lands that overlap with this DPS' range are 7% and 10%, respectively. Agriculture is the dominant land use in the Salinas River valley, and there are areas of intense agriculture in the Pajaro watershed as well. Areas higher in the Salinas and Pajaro watersheds and along some of the coastal areas are less affected. Given these uses, we expect the rearing PCE in smaller freshwater tributaries and floodplain habitats may be

exposed during pesticide application adjacent to these systems. Consequently, reductions in water quality and prey abundance are expected.

We will be providing additional information that details specific considerations for each decision.

Table 192. Co-occurrence of land use types and likelihood of a reduction in PCEs for South-Central California Coast Steelhead.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Medium	No
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Southern California (SC) Steelhead

Of 29 freshwater and estuarine watersheds, 21 and 5 are of high and medium conservation value, respectively. Our GIS analysis indicates 15 and 21 high conservation watersheds are exposed to pesticide applications from agriculture and urban areas, respectively. Of the medium conservation watersheds, all five watersheds are exposed to pesticide applications from the same above land uses.

All PCEs are affected by degraded water quality from pollutants in urban and agricultural runoff. Elevated water temperature and low water flow impact rearing and migration PCEs. The spawning PCE is affected by erosive geology and land use activities that result in an excessive amount of fines in the spawning gravel of most rivers.

Co-occurrence of agriculture, urban/residential, and forestry uses is indicated in Table 193. The percentage of cultivated crop land and developed lands within SC steelhead habitat are about 5% and 34%, respectively. All of the rivers are affected by anthropogenic inputs. Given these land uses we expect some level of exposure for rearing and migration PCEs in fresh water, coastal lagoons, and downstream migration habitats.

We will be providing additional information that details specific considerations for each decision.

Table 193. Co-occurrence of land use types and likelihood of adverse effects for Southern California Steelhead.

a.i.	Co-occurrence				Reduction in PCEs	
	Agriculture	Urban/ Residential	Forest	Rights-of-way	PCEs Affected	Appreciable
2,4-D	Yes	Yes	Yes	Yes	High	Yes
Triclopyr BEE	Yes	Yes	Yes	Yes	Low	No
Diuron	Yes	Yes	NA	Yes	Medium	No
Linuron	Yes	NA	NA	Limited	Low	No
Captan	Yes	Yes	NA	NA	Low	No
Chlorothalonil	Yes	Yes	Yes	NA	Medium	Yes

Table 194. Appreciable reduction in conservation value of critical habitat.

Species	ESU/DPS	Herbicides				Fungicides	
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chlorothalonil
Chinook	Puget Sound	Yes	No	Yes	No	No	Yes
	Lower Columbia River	Yes	No	Yes	No	No	Yes
	Upper Columbia River Spring - Run	Yes	No	No	No	No	Yes
	Snake River Fall - Run	Yes	No	No	No	No	Yes
	Snake River Spring/Summer - Run	Yes	No	No	No	No	Yes
	Upper Willamette River	Yes	No	Yes	No	No	Yes
	California Coastal	Yes	No	No	No	No	Yes
	Central Valley Spring - Run	Yes	No	Yes	No	No	Yes
	Sacramento River Winter - Run	Yes	No	Yes	No	No	Yes
Chum	Hood Canal Summer - Run	Yes	No	No	No	No	Yes
	Columbia River	Yes	No	No	No	No	Yes
Coho	Lower Columbia River	NA	NA	NA	NA	NA	NA
	Oregon Coast	Yes	No	No	No	No	Yes
	Southern Oregon and Northern California Coast	Yes	No	No	No	No	Yes
	Central California Coast	Yes	No	No	No	No	Yes
Sockeye	Ozette Lake	Yes	No	No	No	No	No
	Snake River	Yes	No	No	No	No	No
Steelhead	Puget Sound	NA	NA	NA	NA	NA	NA
	Lower Columbia River	Yes	No	Yes	No	No	Yes
	Upper Willamette River	Yes	No	Yes	No	No	Yes
	Middle Columbia River	Yes	No	No	No	No	Yes
	Upper Columbia River	Yes	No	No	No	No	Yes
	Snake River	Yes	No	No	No	No	Yes
	Northern California	Yes	No	No	No	No	Yes
	Central California Coast	Yes	No	Yes	No	No	Yes
	California Central Valley	Yes	No	Yes	No	No	Yes
	South-Central California Coast	Yes	No	No	No	No	Yes
	Southern California	Yes	No	No	No	No	Yes

DRAFT 03/01/2011

This page left blank intentionally.

Conclusion

In the *Integration and Synthesis of Effects to Listed Species* section, we described NMFS' assessment of the likelihood of negative effects posed to the survival and recovery of listed Pacific salmonids as a result of EPA's registration of 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil. The likelihood of effects assigned to each ESU/DPS for each a.i. reflects NMFS' evaluation of the likelihood that a compound will cause reductions in species viability. Separate determinations were made for each a.i./species pairing. We expect that the registration of 2,4-D will jeopardize the existence of all listed salmonids ESUs/DPSs, and that the registration of chlorothalonil will jeopardize the existence of some listed salmonids. We expect that the remaining a.i.s will have a negative impact on most listed salmonids. However, we do not expect that these effects will jeopardize the existence or recovery of any listed ESU/DPS (Table 195).

The *Integration and Synthesis of Effects to Designated Critical Habitat* section described NMFS' determination as to whether the proposed action will likely destroy or adversely modify critical habitat for ESA-listed species. Separate determinations were made for all critical habitat/a.i. pairs. The PCE attributes were evaluated to determine the likelihood of reducing the quality of spawning, rearing, migration, and estuarine PCEs. We expect stressors of the action primarily affect water quality, prey and plant-based components of the critical habitat. Any substantial likelihood of negative effects reduces the ability of critical habitat to support survival and recovery of the species. We do not expect the registration of triclopyr BEE, linuron, or captan to result in the adverse modification critical habitat of any listed salmonids. We do expect that registered uses of diuron and chlorothalonil may adversely modify the critical habitat of some listed Pacific salmonids, and that the registered uses of 2,4-D will adversely modify the designated critical habitat of all ESUs/DPSs. Final determinations for the adverse modification of critical habitat are given below in Table 196.

Table 195. Jeopardy determinations for a.i.s.

Species	ESU	Herbicides				Fungicides	
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chlorothalonil
Chinook	Puget Sound	Jeopardy	No	No	No	No	Jeopardy
	Lower Columbia River	Jeopardy	No	No	No	No	Jeopardy
	Upper Columbia River Spring - Run	Jeopardy	No	No	No	No	Jeopardy
	Snake River Fall - Run	Jeopardy	No	No	No	No	Jeopardy
	Snake River Spring/Summer - Run	Jeopardy	No	No	No	No	Jeopardy
	Upper Willamette River	Jeopardy	No	No	No	No	Jeopardy
	California Coastal	Jeopardy	No	No	No	No	Jeopardy
	Central Valley Spring - Run	Jeopardy	No	No	No	No	Jeopardy
	Sacramento River Winter - Run	Jeopardy	No	No	No	No	Jeopardy
Chum	Hood Canal Summer - Run	Jeopardy	No	No	No	No	Jeopardy
	Columbia River	Jeopardy	No	No	No	No	Jeopardy
Coho	Lower Columbia River	Jeopardy	No	No	No	No	Jeopardy
	Oregon Coast	Jeopardy	No	No	No	No	Jeopardy
	Southern Oregon and Northern California Coast	Jeopardy	No	No	No	No	Jeopardy
	Central California Coast	Jeopardy	No	No	No	No	Jeopardy
Sockeye	Ozette Lake	Jeopardy	No	No	No	No	No
	Snake River	Jeopardy	No	No	No	No	No
Steelhead	Puget Sound	Jeopardy	No	No	No	No	Jeopardy
	Lower Columbia River	Jeopardy	No	No	No	No	Jeopardy
	Upper Willamette River	Jeopardy	No	No	No	No	Jeopardy
	Middle Columbia River	Jeopardy	No	No	No	No	Jeopardy
	Upper Columbia River	Jeopardy	No	No	No	No	Jeopardy
	Snake River	Jeopardy	No	No	No	No	Jeopardy
	Northern California	Jeopardy	No	No	No	No	Jeopardy
	Central California Coast	Jeopardy	No	No	No	No	Jeopardy
	California Central Valley	Jeopardy	No	No	No	No	Jeopardy
	South-Central California Coast	Jeopardy	No	No	No	No	Jeopardy

	Southern California	Jeopardy	No	No	No	No	Jeopardy
--	---------------------	-----------------	----	----	----	----	-----------------

Table 196. Adverse modification determinations.

Species	ESU	Herbicides				Fungicides	
		2,4-D	Triclopyr BEE	Diuron	Linuron	Captan	Chlorothalonil
Chinook	Puget Sound	Ad Mod	No	Ad Mod	No	No	Ad Mod
	Lower Columbia River	Ad Mod	No	Ad Mod	No	No	Ad Mod
	Upper Columbia River Spring - Run	Ad Mod	No	No	No	No	Ad Mod
	Snake River Fall - Run	Ad Mod	No	No	No	No	Ad Mod
	Snake River Spring/Summer - Run	Ad Mod	No	No	No	No	Ad Mod
	Upper Willamette River	Ad Mod	No	Ad Mod	No	No	Ad Mod
	California Coastal	Ad Mod	No	No	No	No	Ad Mod
	Central Valley Spring - Run	Ad Mod	No	Ad Mod	No	No	Ad Mod
	Sacramento River Winter - Run	Ad Mod	No	Ad Mod	No	No	Ad Mod
Chum	Hood Canal Summer - Run	Ad Mod	No	No	No	No	Ad Mod
	Columbia River	Ad Mod	No	No	No	No	Ad Mod
Coho	Lower Columbia River	<i>NA</i>	<i>NA</i>	<i>NA</i>	<i>NA</i>	<i>NA</i>	<i>NA</i>
	Oregon Coast	Ad Mod	No	No	No	No	Ad Mod
	Southern Oregon and Northern California Coast	Ad Mod	No	No	No	No	Ad Mod
	Central California Coast	Ad Mod	No	No	No	No	Ad Mod
Sockeye	Ozette Lake	Ad Mod	No	No	No	No	No
	Snake River	Ad Mod	No	No	No	No	No
Steelhead	Puget Sound	<i>NA</i>	<i>NA</i>	<i>NA</i>	<i>NA</i>	<i>NA</i>	<i>NA</i>
	Lower Columbia River	Ad Mod	No	Ad Mod	No	No	Ad Mod
	Upper Willamette River	Ad Mod	No	Ad Mod	No	No	Ad Mod
	Middle Columbia River	Ad Mod	No	No	No	No	Ad Mod
	Upper Columbia River	Ad Mod	No	No	No	No	Ad Mod
	Snake River	Ad Mod	No	No	No	No	Ad Mod
	Northern California	Ad Mod	No	No	No	No	Ad Mod

	Central California Coast	Ad Mod	No	Ad Mod	No	No	Ad Mod
	California Central Valley	Ad Mod	No	Ad Mod	No	No	Ad Mod
	South-Central California Coast	Ad Mod	No	No	No	No	Ad Mod
	Southern California	Ad Mod	No	No	No	No	Ad Mod

Reasonable and Prudent Alternatives

Regulations (50 CFR §402.02) implementing section 7 of the ESA define reasonable and prudent alternatives as alternative actions, identified during formal consultation, that: (1) can be implemented in a manner consistent with the intended purpose of the action; (2) can be implemented consistent with the scope of the action agency's legal authority and jurisdiction; (3) are economically and technologically feasible; and (4) NMFS believes would avoid the likelihood of jeopardizing the continued existence of listed species or resulting in the destruction or adverse modification of critical habitat.

This Opinion has concluded that EPA's proposed registration of pesticides containing 2, 4-D and chlorothalonil are likely to jeopardize the continued existence of one or more of the 28 endangered and threatened Pacific salmonids and 2,4-D, diuron, and chlorothalonil are likely to adversely modify or destroy designated critical habitat for one or more of the 28 threatened and endangered salmonids. NMFS reached these conclusions because predicted concentrations of these three a.i.s in salmonid habitats, particularly in floodplain habitats¹⁰, are likely to cause adverse effects to Pacific salmonids, water quality, salmonid prey, natural cover, and/or substrate in freshwater rearing, spawning, and foraging areas.

The Reasonable and Prudent Alternative (RPA) accounts for the following issues: (1) the action will result in exposure to other chemical stressors in addition to the a.i. that may increase the risk of the action to listed species, including unspecified inert ingredients, adjuvants, and tank mixes; (2) exposure to chemical mixtures containing the a.i.s; and (3) exposure to other chemicals and physical stressors in the baseline habitat will likely intensify response to the a.i.s.

¹⁰ Floodplain habitat – water bodies and/or inundated areas that are connected (accessible to salmonid juveniles) seasonally or annually to the main channel of a stream including but not limited to features such as side channels, alcoves, ox bows, ditches, and tributaries.

Main channel –the stream channel that includes the thalweg (longitudinal continuous deepest portion of the channel).

The action as implemented under the RPA will remove the likelihood of jeopardy and adverse modification by reducing the concentrations of each of these a.i.s and their associated stressors of the action within the designated critical habitat. In the proposed RPA, NMFS does not attempt to ensure there is no take of listed species. NMFS believes take will occur, and has provided an incidental take statement exempting that take from the take prohibitions, so long as the action is conducted according to the RPA and reasonable and prudent measures (RPM). Avoiding take altogether would most likely entail canceling registration, or prohibiting use in watersheds inhabited by salmonids. The goal of the RPA is to reduce exposure to ensure that the action is not likely to jeopardize listed species, destroy or adversely modify critical habitat.

The RPA is comprised of seven required elements that must be implemented in its entirety within one year of the EPA's receipt of this Opinion to ensure the registration of these pesticides is not likely to jeopardize endangered or threatened Pacific salmonids under the jurisdiction of NMFS or destroy or adversely modify critical habitat designated for these species. For each active ingredient, the elements of the RPA apply only to those ESUs/DPSs where NMFS has determined that registration of that a.i. is likely to jeopardize listed species and/or destroy or adversely modify designated critical habitat (Table 195 and Table 196). These elements rely upon recognized practices for reducing the loading of pesticide products into aquatic habitats.

Specific Elements of the Reasonable and Prudent Alternative

Elements 1-6 shall be specified on FIFRA labels of all pesticide products containing 2,4-D, diuron, and chlorothalonil. Alternatively, the label could direct pesticide users to the EPA's Endangered Species Protection Program (ESPP) bulletins that specify elements 1-6. For purposes of this RPA salmonid habitats are defined as freshwaters, estuarine habitats, and nearshore marine habitats including bays within the ESU/DPS ranges including migratory corridors. The freshwater habitats include intermittent streams and other habitats temporally connected to salmonid-bearing waters when those habitats contain water. Freshwater habitats

also include all known types of floodplain habitats as well as drainages, ditches, and other man-made conveyances to salmonid habitats that lack salmonid exclusion devices (*e.g.*, screens).

Element 1. Do not apply pesticide products containing 2,4-D, diuron, or chlorothalonil when wind speeds are greater than or equal to 10 mph.

Element 2. Do not apply pesticide products containing 2,4-D, diuron, or chlorothalonil when soil moisture is at field capacity, or when a storm event likely to produce runoff from the treated area is forecasted by to occur within 48 h following application by NOAA/NWS (National Weather Service) or other similar forecasting service.

Element 3. 2,4-D specific requirements

- a) Do not apply pesticide products containing 2,4-D butoxyester ethyl directly to salmonid habitats.
- b) Do not apply products containing 2,4-D directly to salmonid habitats
 - a. except to control non-native (exotic) invasive species;
 - b. outside of timing windows to be developed by NMFS for each ESU/DPS; and
 - c. when the control of non-native (exotic) invasive species cannot be accomplished without adverse effects to native aquatic plants.

NMFS reserves the right to make alterations to spray timing windows as new information becomes available on listed species spatial and temporal distributions.
- c) Within the ESU/DPSs boundary of Upper Willamette River Chinook, Central Valley Spring-Run Chinook, Sacramento River Winter-Run Chinook, Upper Willamette River Steelhead, Central California Coast Steelhead, and California Central Valley Steelhead:
 - a. Do not apply 2,4-D directly to riparian habitat where applications will decrease shading to aquatic habitat, decrease bank stability, or increase

erosion.¹¹ EPA will implement NMFS approved risk reduction measures to ensure maximum concentrations of 2,4-D do not exceed 10 µg/L¹² in these salmonid habitats.

Element 4. Diuron-specific requirements

- a) EPA will implement NMFS approved risk reduction measures to ensure diuron drift to riparian habitat does not exceed 0.05 lbs/A¹². Diuron impacts terrestrial plant growth at 0.002 – 0.021 lbs/A based on the EC₂₅ for vegetative vigor in dicotyledonous and monocotyledonous plants. It also reduces seedling emergence at 0.075 – 0.099 based (EC₂₅ for crop species). Although the data suggests 0.05 lbs/A will harm riparian vegetation we believe that under most circumstances it will not alter the function of the riparian zone to the extent that attributes important to salmonids are negatively impacted.
- b) When a riparian zone¹¹ is not present, EPA will implement NMFS approved risk reduction measures to ensure maximum concentrations of diuron do not exceed 2.9 µg/L¹² in salmonid habitats

Element 5. Chlorothalonil-specific requirements. EPA will implement NMFS approved risk reduction measures to address effects from turf and agricultural uses of chlorothalonil products.

¹¹ Riparian habitat is the transitional zone between terrestrial and aquatic ecosystems. It is an area of vegetation adjacent to rivers, streams, lakes, and wetland areas, typically with a high water table and with certain plants amongst the vegetation community that are dependent on moist soil conditions.

¹² EPA will implement NMFS approved risk reduction measures to ensure maximum concentrations of the a.i.s predicted in salmonid habitats or riparian zones will not exceed the specified value. These values represent the highest concentrations that may be achieved in salmonid habitats, rather than time-weighted average concentrations, considering the range in potential droplet size spectrum, release heights, wind speeds, and wind directions that may be associated with all labeled application methods (*e.g.*, agricultural applications, vector control in public health programs, *etc.*). The maximum predicted concentrations shall account for potential contributions from both runoff and drift to salmonid habitats, as appropriate. Risk reduction measures shall account for the predicted maximum concentrations in all salmonid habitats, including a modeled floodplain habitat of 1-2 m wide and 0.1 m deep. They shall also account for potential increases in aquatic concentrations associated with the maximum application rate and the maximum number of times an a.i. may be applied per season according to label restrictions.

Element 6. Report all incidents of fish mortality that occur within the vicinity of the treatment area, including areas downstream and downwind, in the four days following application of and of these a.i.s to EPA OPP (703-305-7695). Alternatively, these incidents may be reported to the pesticide manufacturer through the phone number on the product label once EPA modifies FIFRA 6(a)2 to require registrants to report all fish kills immediately, regardless of incident classification (*i.e.* both minor and major incidents). EPA shall submit an annual report to NMFS OPR that identifies the total number of fish affected and incident locations.

Element 7. In addition to the labeling requirements above, EPA shall develop and implement a NMFS-approved effectiveness monitoring plan for floodplain habitats, and produce annual reports of the results. The plan shall identify representative floodplain habitats prone to drift and runoff of pesticides within agricultural areas. The representative floodplain habitat sampling sites shall include floodplain habitats currently used by threatened and endangered Pacific salmonids, as identified by NMFS biologists. Sampling sites include at least two sites for each general species (*i.e.*, coho salmon, chum salmon, steelhead, sockeye salmon, and ocean-type Chinook and stream-type Chinook salmon). Sampling shall consist of daily collection of surface water samples for seven consecutive days during three periods of high application for 2,4-D, diuron, and chlorothalonil. The report shall be submitted to NMFS OPR and will summarize annual monitoring data and provide all raw data.

Because this Opinion has found jeopardy and destruction or adverse modification to designated critical habitat, the EPA is required to notify NMFS of its final decision on the implementation of the reasonable and prudent alternatives (50 CFR §402.15(b)).

Incidental Take Statement

Section 9(a)(1) of the ESA prohibits the taking of endangered species without a specific permit or exemption. Protective regulations adopted pursuant to section 4(d) of the ESA extend the

prohibition to threatened species. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct (50 CFR 222.102).

Harm is further defined by NMFS to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity conducted by the Federal agency or applicant (50 CFR 402.02). Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action, whether implemented as proposed or as modified by reasonable and prudent alternatives, is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

Amount or Extent of Take

As described earlier in this Opinion, this is a consultation on the EPA's registration of pesticide products containing 2,4-D, triclopyr BEE, diuron, linuron, captan, chlorothalonil, and their formulations as they are used in the Pacific Northwest and California and the affects of these applications on listed ESUs/DPSs of Pacific salmonids. The EPA authorizes use of these pesticide products for pest control purposes across multiple landscapes as described in the *Description of the Proposed Action* and elsewhere in the document. The goal of this Opinion is to evaluate the impacts to NMFS' listed resources from the EPA's broad authorization of applied pesticide products. This Opinion is a partial consultation because pursuant to the court's order, EPA sought consultation on only 26 listed Pacific salmonids under NMFS' jurisdiction. However, even though the court's order did not address the two more recently listed ESUs and DPSs, NMFS analyzed the impacts of EPA's actions to them because they belong to the same taxon and the analysis requires consideration of the same information. Consultation with NMFS will be completed when EPA makes effect determinations on all remaining species under NMFS' jurisdiction and consults with NMFS as necessary.

For this Opinion, NMFS anticipates the general direct and indirect effects that would occur from EPA's registration of pesticide products across the states of California, Idaho, Oregon, and Washington to 28 listed Pacific salmonids under NMFS' jurisdiction during the 15-year duration of the proposed action. Recent and historical surveys indicate that listed salmonids occur in the action area, in places where they will be exposed to the stressors of the action. The RPAs provided in this Opinion are designed to reduce this exposure but not eliminate it. Pesticide runoff and drift of 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil are most likely to reach streams and other aquatic sites when they are applied to crops and other land use settings located adjacent to wetlands, riparian areas, ditches, off-channel habitats, and intermittent streams. These inputs into aquatic habitats are especially high when rainfall immediately follows applications. The effects of pesticides and other contaminants found in right of ways and urban runoff, especially from areas with a high degree of impervious surfaces, may also exacerbate degraded water quality conditions of receiving waters used by salmon. Urban runoff is also generally warmer in temperature, and elevated water temperature poses negative effects on certain life history phases for salmon. The range of effects of the 6 a.i.s on salmonids includes direct toxicological effects such as impaired growth, reproduction, olfaction, and survival; reductions in energy for activities such as reproduction, migration, foraging; reductions in prey base, increased susceptibility to predation and disease, potential delays in spawning. Adverse impacts to riparian vegetation could lead to increased water temperature, increased sedimentation from bank instability, reductions in cover, alterations to or decreases in prey production, reduction in chemical and nutrient filtering from upland sources. Impacts to aquatic vegetation would reduce natural cover, alter or reduce the prey base, affect growth and lead to an increased susceptibility to predation. Thus, we expect some exposed fish will respond to these effects by changing normal behaviors. In some cases, fish may die, be injured, or suffer sublethal effects. These results are not the purpose of the proposed action. Therefore, incidental take of listed salmonids is reasonably certain to occur over the 15-year duration of the proposed action.

Given the variability of real-life conditions, the broad nature and scope of the proposed action, and the migratory nature of salmon, the best scientific and commercial data available are not

sufficient to enable NMFS to estimate a specific amount of incidental take associated with the proposed action. As explained in the *Description of the Proposed Action* and the *Effects of the Proposed Action* sections, NMFS identified multiple uncertainties associated with the proposed action. Areas of uncertainty include:

1. Unable to quantify effect of herbicides on salmon habitat due to variability in plant susceptibility to the herbicides and variability in species composition and density in the various locations.
2. Incomplete information on the proposed action (*i.e.*, no master labels summarizing all stressors of the action and all authorized uses of pesticide products). ;
3. Limited use and exposure data on stressors of the action for non-agricultural uses of these pesticides;
4. Minimal information on exposure and toxicity for pesticide formulations, adjuvants, and other/inert ingredients within registered formulations;
5. Little information on permitted tank mixtures and associated exposure estimates;
6. Limited data on toxicity of environmental mixtures;
7. Responses from exposure to combinations of the 6 a.i.s and other stressors in the baseline;
8. Annual variable conditions regarding land use, crop cover, and pest pressure;
9. Variable temporal and spatial conditions within each ESU, especially at the population-level; and
10. Variable conditions of water bodies in which salmonids live.

NMFS therefore identifies, as a surrogate for the allowable extent of take, the ability of this action to proceed without any fish kills attributed to the legal use of 2,4-D, triclopyr BEE, diuron, linuron, captan, or chlorothalonil, or any compounds, degradates, or mixtures in aquatic habitats containing individuals from any ESU/DPS. Because of the difficulty of detecting salmonid deaths, the fishes killed do not have to be listed salmonids. In general, salmonids appear to be more sensitive to these a.i.s than many other species of fish, so that if there are kills of other freshwater fishes attributed to use of these pesticides, it is likely that salmonids have also died, even if no dead salmonids can be located. In addition, if stream conditions due to pesticide use kill less sensitive fishes in certain areas, the potential for lethal and non-lethal takes in downstream areas increases. A fish kill is considered attributable to one of these 6 ingredients, its metabolites, or degradates, if the a.i is known to have been applied in the vicinity

and may reasonably be supposed to have run off or drifted into the affected area, and if surface water samples, or pathology indicate lethal levels of the a.i.(s).

NMFS notes that with increased monitoring and study of the impact of these pesticides on water quality, particularly water quality in off-channel habitats, NMFS will be able to refine this incidental take statement, and future incidental take statements, to allow other measures of the extent of take.

Reasonable and Prudent Measures

The measures described below are non-discretionary measures to avoid or minimize take that must be undertaken by the EPA so that they become binding conditions of any grant or permit issued to the applicant(s), as appropriate, for the exemption in section 7(o)(2) to apply. The EPA has a continuing duty to regulate the activity covered by this incidental take statement. If the EPA (1) fails to assume and implement the terms and conditions or (2) fails to require the applicant(s) to adhere to the terms and conditions of the incidental take statement through enforceable terms that are added to the permit or grant document, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, the EPA must report the progress of the action and its impact on the species to NMFS OPR as specified in the incidental take statement [50 CFR§402.14(i)(3)].

To satisfy its obligations pursuant to section 7(a)(2) of the ESA, the EPA must monitor (a) the direct, indirect, and cumulative impacts of its long-term registration of pesticide products containing 2,4-D, triclopyr BEE, diuron, linuron, captan, or chlorothalonil; (b) evaluate the direct, indirect, or cumulative impacts of pesticide misapplications in the aquatic habitats in which they occur; and (c) the consequences of those effects on listed Pacific salmonids under NMFS' jurisdiction. The purpose of the monitoring program is for the EPA to use the results of the monitoring data and modify the registration process in order to reduce exposure and minimize the effect of exposure where pesticides will occur in salmonid habitat. NMFS believes

all measures described as part of the proposed action, together with use of the Reasonable and Prudent Measures and Terms and Conditions described below, are necessary and appropriate to minimize the likelihood of incidental take of listed species due to implementation of the proposed action.

The EPA shall:

1. Minimize the amount and extent of incidental take from use of pesticide products containing 2,4-D, triclopyr BEE, diuron, linuron, captan, or chlorothalonil by reducing the potential of chemicals to reach salmon-bearing waters;
2. Monitor any incidental take or surrogate measure of take that occurs from the action; and
3. Report annually to NMFS OPR on the monitoring results from the previous year.

Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, within one year following the date of issuance of this Opinion, the EPA must comply with the following terms and conditions.

These terms and conditions implement the reasonable and prudent measures described above.

These terms and conditions are non-discretionary.

1. a. Do not apply pesticide products when wind speeds are greater than or equal to 10 mph.

b. Do not apply pesticide products when soil moisture is at field capacity, or when a storm event likely to produce runoff from the treated area is forecasted by to occur within 48 h following application by NOAA/NWS (National Weather Service) or other similar forecasting service.
2. a. EPA shall include the following instructions requiring reporting of fish kills either on the labels for all products containing 2,4-D, triclopyr BEE, diuron, linuron, captan, and chlorothalonil or in ESPP Bulletins:

NOTICE: Incidents where salmon appear injured or killed as a result of pesticide applications shall be reported to NMFS OPR at 301-713-1401 and EPA at 703-305-7695.

The finder should leave the fish alone, make note of any circumstances likely causing the death or injury, location and number of fish involved, and take photographs, if possible. Adult fish should generally not be disturbed unless circumstances arise where an adult fish is obviously injured or killed by pesticide exposure, or some unnatural cause. The finder may be asked to carry out instructions provided by NMFS OPR to collect specimens or take other measures to ensure that evidence intrinsic to the specimen is preserved.

b. EPA shall report to NMFS OPR any incidences regarding 2,4-D, triclopyr BEE, diuron, linuron, captan, or chlorothalonil effects on aquatic ecosystems added to its incident database that it has classified as probable or highly probable.

3. EPA shall provide OPR a commencement date for annual reporting of monitoring results.

Conservation Recommendations

Section 7(a) (1) of the ESA directs federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

The following conservation recommendations would provide information for future consultations involving future authorizations of pesticide a.i.s that may affect listed species:

1. Conduct mixture toxicity analysis in screening-level and endangered species biological evaluations;
2. Develop models to estimate pesticide concentrations in off-channel habitats; and
3. Develop models to estimate pesticide concentrations in aquatic habitats associated with non-agricultural applications, particularly in residential and industrial environments.

In order for NMFS to be kept informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, the EPA should notify NMFS OPR of any conservation recommendations it implements in the final action.

Conservation Recommendations

Section 7(a) (1) of the ESA directs federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

The following conservation recommendations would provide information for future consultations involving future authorizations of pesticide a.i.s that may affect listed species:

4. Conduct mixture toxicity analysis in screening-level and endangered species biological evaluations;
5. Develop models to estimate pesticide concentrations in off-channel habitats; and
6. Develop models to estimate pesticide concentrations in aquatic habitats associated with non-agricultural applications, particularly in residential and industrial environments.

In order for NMFS to be kept informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, the EPA should notify NMFS OPR of any conservation recommendations it implements in the final action.

Reinitiation Notice

This concludes formal consultation on the EPA's proposed registration of pesticide products containing 2,4-D, triclopyr BEE, diuron, linuron, captan, chlorothalonil, and their formulations to ESA-listed Pacific salmonids under the jurisdiction of the NMFS. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the extent of take specified in the *Incidental Take Statement* is exceeded; (2) new information reveals effects of this action that may affect listed species or designated critical habitat in a manner or to an extent not previously considered in this biological opinion; (3) the identified action is subsequently modified in a manner that causes an effect to the listed species or critical

habitat that was not considered in this Opinion; or (4) a new species is listed or critical habitat designated that may be affected by the identified action. If reinitiation of consultation appears warranted due to one or more of the above circumstances, EPA must contact NMFS OPR. In the event reinitiation conditions (1), (2), or (3) is met, reinitiation will be only for the a.i.(s) which meet that condition, not for all 6 a.i.s considered in the Opinion. If none of these reinitiation triggers are met within the next 15 years, then reinitiation will be required because the Opinion only covers the action for 15 years.

References

- Ackers, J. T., M. F. Johnston, et al. (2000). "Immunodetection of hepatic peroxisomal PMP70 as an indicator of peroxisomal proliferation in the mummichog, *Fundulus heteroclitus*." Marine Environmental Research 50: 361-365.
- Adams, P. (2000). Status review update for the steelhead Northern California Evolutionary Significant Unit. Status Review. Tiburon, California, Southwest Fisheries Science Center, Santa Cruz/Tiburon Laboratory: 12.
- Ajawani, S. (1956). "A review of Lake Washington watershed, historical, biological, and limnological." M.s. thesis, University of Washington, Seattle, Washington.
- Allan, J. D. and M. M. Castillo (2008). Stream Ecology: Structure and Function of Running Waters. The Netherlands, Springer.
- Allan, J. D., M. S. Wipfli, et al. (2003). "Influence of streamside vegetation on inputs of terrestrial invertebrates to salmonid food webs." Canadian Journal of Fisheries and Aquatic Sciences 60(3): 309-320.
- Anderson, B. S., B. M. Phillips, et al. (2006). "Identifying primary stressors impacting macroinvertebrates in the Salinas River (California, USA): Relative effects of pesticides and suspended particles." Environmental Pollution 141(3): 402-408.
- Anderson, P. D., D. Dugger, et al. (2007). Surface water monitoring program for pesticides in salmonid-bearing streams, 2006 monitoring data summary. Report. Olympia, Washington, Washington State Department of Ecology: 43.
- Anderson, S. E. (1999). Use of off-channel freshwater wetlands by juvenile chinook and other salmonids: Potential for habitat restoration in Puget Sound, The Evergreen State College. Masters: 73.
- Andrieux, P., B. Lennartz, et al. (1997). "Diuron and simazine losses to runoff water in mediterranean vineyards. ." Journal of environmental quality 26: 1493 - 1502.
- Anonymous (2007). Biological Assessment of the effects of the Federal Columbia River Power System and mainstem effects of other tributary actions on anadromous salmonid species listed under the Endangered Species Act. Biological Assessment, Corps, Bonneville Power Administration, and Reclamation.

- Antolos, M., D. D. Roby, et al. (2005). "Caspian tern predation on juvenile salmonids in the Mid-Columbia River." Transaction of American Fisheries Society(134): 466-480.
- Aspelin, A. L. and A. H. Grube (1999). Pesticides industry sales and usage-1996 and 1997 market estimates. Pesticide Industry Sales and Usage Report, EPA, Biological and Economic Analysis Division: 39.
- Austin, A. P., G. E. Harris, et al. (1991). "Impacts of an organophosphate herbicide (glyphosate) on periphyton communities developed in experimental streams." Bulletin of Environmental Contamination and Toxicology 47(1): 29-35.
- Baier-Anderson, C. and R. S. Anderson (1998). "Evaluation of the immunotoxicity of chlorothalonil to striped bass phagocytes following in vitro exposure." Environmental Toxicology and Chemistry 17(8): 1546-1551.
- Bakkala, R. G. (1970). "Synopsis of biological data on the chum salmon, *Oncorhynchus keta* (Walbaum) 1792." FAO Fish. Synop. 41; U.S. Fish. Wildl. Serv. Circ. 315: 89 p.
- Barnhart, R. A. (1986). "Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Pacific Southwest): steelhead. ." Technical Report, TR-EL-82-4/872-11-60. Humboldt State University, Arcata, California.
- Barron, M. G., M. A. Mayers, et al. (1990). "Pharmacokinetics and metabolism of triclopyr butoxyethyl ester in coho salmon." Aquatic Toxicology 16(19-32).
- Bax, N. J. (1983). *The early marine migration of juvenile chum salmon (Oncorhynchus keta) through Hood Canal - its variability and consequences.* Seattle, Washington, University of Washington. Ph.D.: 196.
- Baxter, C. V., K. D. Fresh, et al. (2007). "Invading rainbow trout usurp a terrestrial prey subsidy from native charr and reduce their growth and abundance." Oecologia 153: 461-470.
- Beamer, E. M., B. Hayman, et al. (2005). Appendix C: Linking freshwater habitat to Skagit Chinook salmon recovery. Skagit Chinook Recovery Plan, Skagit River System Cooperative and Washington Department of Fish and Wildlife: 24.
- Beamish, R. J. and D. R. Bouillon (1993). "Pacific salmon production trends in relation to climate." Canadian Journal of Fisheries and Aquatic Sciences 50: 1002-1016.

- Beamish, R. J. and C.-E. M. Neville (1995). "Pacific salmon and Pacific herring mortalities in the Fraser River plume caused by river lamprey (*Lampetra ayresi*). " Canadian Journal of Fisheries and Aquatic Sciences(52): 644-650.
- Beamish, R. J., D. Noakes, et al. (2000). "Trends in coho marine survival in relation to the regime concept." Fisheries Oceanography 9(1): 114-119.
- Beamish, R. J., R. M. Sweeting, et al. (2009). "Planning the management of Pacific salmon in a changing climate." American Fisheries Society Symposium 69: 155-173.
- Beamish, R. J., B. L. Thomson, et al. (1992). "Spiny dogfish predation on chinook and coho salmon and the potential effects on hatchery-produced salmon." Transaction of American Fisheries Society 121(4): 444-445.
- Beecher, H. A., B. A. Caldwell, et al. (2002). "Evaluation of depth and velocity preferences of juvenile coho salmon in Washington streams." North American Journal of Fisheries Management 22(3): 785-795.
- Beechie, T. and S. Bolton (1999). "An approach to restoring salmonid habitat-forming processes in Pacific Northwest watersheds." Fisheries 24: 6-15.
- Beechie, T. J., E. Beamer, et al. (1994). "Estimating coho salmon rearing habitat and smolt production losses in a large river basin, and implications for habitat restoration." North American Journal for Fisheries Management 14: 797-811.
- Beechie, T. J., M. Liermann, et al. (2005). "A classification of habitat types in a large river and their use by juvenile salmonids." Transactions of the American Fisheries Society 134(3): 717-729.
- Belitz, K., S. N. Hamlin, et al. (2004). *Water Quality in the Santa Ana Basin, California, 1999-2001. U.S. Geological Survey Circular*. Reston, Virginia, U.S. Department of the Interior. 1238.
- Bell, M. C. (1973). Fisheries handbook of engineering requirements and biological criteria. Portland, OR, United States Army Corps of Engineers, Fisheries Engineering Research Program.
- Benda, L. E., D. J. Miller, et al. (2001). *Dynamic landscape systems. River Ecology and Management, Lessons from the Pacific Coastal Ecoregion*. R. J. Naiman and R. E. Bilby. New York, Springer-Verlag New York, Inc.: 261-288.

- Bennet, W. A. (2005, September 2005). "Critical assessment of the delta smelt population in the San Francisco Estuary, California." San Francisco Estuary and Watershed Science Retrieved 2, 3.
- Bernard, B. K. and E. B. Gordon (2000). "An evaluation of the common mechanism approach to the Food Quality Protection Act: captan and four related fungicides, a practical example." International Journal of Toxicology 19: 43-61.
- Bigler, B. (1985). "Kotzebue Sound chum salmon (*Oncorhynchus keta*) escapement and return data, 1962-1984." Alaska Dep. Fish Game ADF&G Tech Data Rep. 149: 112 p.
- Bilby, R. E. and P. A. Bisson (1987). "Emigration and production of hatchery coho salmon (*Oncorhynchus kisutch*) stocked in streams draining an old-growth and clear-cut watershed." Canadian Journal of Fisheries and Aquatic Sciences 44: 1397-1407.
- Bilby, R. E. and P. A. Bisson (2001). Function and distribution of large woody debris. River Ecology and Management, Lessons from the Pacific Coastal Ecoregion. R. J. Naiman and R. E. Bilby. New York, Springer-Verlag, Inc.: 324-346.
- Bilby, R. E., B. R. Fransen, et al. (2001). "Preliminary evaluation of the use of nitrogen stable isotope ratios to establish escapement levels for pacific salmon." Fisheries 26(1): 6-14.
- Bilton, H. T. and G. L. Robins (1973). "Effects of starvation and subsequent feeding on survival and growth of Fulton Channel salmon fry (*Oncorhynchus nerka*)." Journal Fisheries Research Board of Canada 30(1): 1-5.
- Binelli, A., F. Ricciardi, et al. (2005). "Screening of POP pollution by AChE and EROD activities in Zebra mussels from the Italian Great Lakes." Chemosphere 61(8): 1074-1082.
- Bird, S. L., S. G. Perry, et al. (2002). "Evaluation of the AgDisp aerial spray algorithms in the AgDrift model." Environmental Toxicology and Chemistry 21(3): 672-681.
- Bisson, P. A. and R. E. Bilby (2001). Organic matter and trophic dynamics. River Ecology and Management, Lessons from the Pacific Coastal Ecoregion. R. J. Naiman and R. E. Bilby. New York, Springer-Verlag, Inc.: 373-398.
- Bisson, P. A., J. L. Nielsen, et al. (1982). A system of naming habitat types in small streams, with examples of habitat utilization by salmonids during low streamflow. Acquisition and utilization of aquatic habitat inventory information. N. B. Armantrout. Bethesda, Maryland, American Fisheries Society, Western Division.: 62-73.

- Bisson, P. A., B. E. Rieman, et al. (2003). "Fire and aquatic ecosystems of the western USA: current knowledge and key questions." Forest Ecology and Management(178): 213-229.
- Bisson, P. A., K. Sullivan, et al. (1988). "Channel hydraulics, habitat use, and body form of juvenile coho salmon, steelhead, and cutthroat trout in streams." Transaction of American Fisheries Society 117: 262-273.
- Bjorkstedt, E. P., B. C. Spence, et al. (2005). An analysis of historical population structure for evolutionarily significant units of Chinook salmon, coho salmon, and steelhead in the North-Central California Coast Recovery Domain. NOAA Technical Memorandum, U.S. Department of Commerce: 210.
- Bjornn, T., D. Craddock, et al. (1968). "Migration and survival of Redfish Lake, Idaho, Sockeye salmon, *Oncorhynchus nerka*." Transactions of the American Fisheries Society 97: 360-373.
- Bjornn, T. C. and N. Horner (1980). "Biological criteria for classification of Pacific salmon and steelhead as threatened or endangered under the Endangered Species Act." Idaho Cooperative Fisheries Research Unit for NMFS.
- Bjornn, T. C. and D. W. Reiser (1991). Habitat requirements of salmonids in streams. Influences of forest and rangeland management on salmonid fishes and their habitats. M. R. Meehan, American Fisheries Society Special Publications. 19: 83-138.
- Blanchet, S., G. Loot, et al. (2008). "Competition, predation and flow rate as mediators of direct and indirect effects in a stream food chain." Oecologia 157(1): 93-104.
- Blum, J. P. (1988). "Assessment of factors affecting sockeye salmon (*Oncorhynchus nerka*) production in Ozette Lake, WA." Masters Thesis, University of Washington, Seattle, Washington.
- Boles, G. L. (1988). Water temperature effects on Chinook salmon (*Oncorhynchus tshawytscha*) with emphasis on the Sacramento River: a literature review. Report. Sacramento, CA, California Department of Fish and Game, Northern District.
- Bortleson, G. C., M. J. Chrzastowski, et al. (1980). "Historical changes of shoreline and wetland at eleven major deltas in the Puget Sound region, Washington." U.S. Geological Survey, Hydrologic Investigations Atlas HA-617, U.S. Department of Justice and Bureau of Indian Affairs, Reston, Virginia.

- Bortleson, G. C. and J. C. Ebbert (2000). "Occurrence of pesticides in streams and ground water in the Puget Sound basin, Washington, and British Columbia, 1996-98." United States Geological Survey, Water-Resources Investigations Report 00-4118, Tacoma, Washington.
- Boughton, D. A., P. B. Adams, et al. (2007). Viability criteria for steelhead of the south-central and southern California coast. NOAA Technical Memorandum, National Marine Fisheries Service. Draft: 33.
- Boughton, D. A., H. Fish, et al. (2005). Contraction of the southern range limit for anadromous *Oncorhynchus mykiss*. NOAA Technical Memorandum, U.S. Department of Commerce. TM-NMFS-SWFSC-380: 21.
- Bouldin, J. L., J. L. Farris, et al. (2007). "Assessment of diazinon toxicity in sediment and water of constructed wetlands using deployed *Corbicula fluminea* and laboratory testing." Archives of Environmental Contamination and Toxicology 53(2): 174-182.
- Bowman, K. E. and G. W. Minshall (2000). "Assessment of short- and mid-term effects of wildlife on habitat structure of the Payette National Forest." Prepared by the Stream Ecology Center, Idaho State University for the Payette National Forest: 45 p.
- Bradford, M. J., G. C. Taylor, et al. (1997). "Empirical review of coho salmon smolt abundance and prediction of smolt production at the regional level." Transactions of the American Fisheries Society 126: 49-64.
- Brain, R. A., D. B. Perkins, et al. (2010). Review and assessment of the potential impact of chlorothalonil on endangered and threatened salmonid species in the Pacific Northwest based on reproductive, physiological, and behavioral considerations and recent environmental monitoring data. Greensboro, NC, Syngenta Crop Protection, Inc.: 53.
- Braun, A. L. and L. S. Hawkins (1991). Presence of bromacil, diuron, and simazine in surface water runoff from agricultural fields and non-crop sites in Tulare County, California. Department of Pesticide Regulation, Pest Management and Analysis Program, pub. PM91-1.
- Brazner, J. C. and E. R. Kline (1990). "Effects of chlorpyrifos on the diet and growth of larval fathead minnows, *Pimephales promelas*, in littoral enclosures." Canadian Journal of Fisheries and Aquatic Sciences 47: 1157-1165.

- Brennan, J. S., K. F. Higgins, et al. (2004). "Juvenile salmon composition, timing, distribution, and diet in the marine nearshore waters of central Puget Sound in 2001-2002 " King County Department of Natural Resources and Parks, Seattle, Washington: 164 p.
- Brett, J. R. (1952). "Temperature tolerance in young Pacific salmon, genus *Oncorhynchus*." Journal Fisheries Research Board of Canada 9(6): 265-323.
- Brett, J. R. (1995). "Energetics." Physiological ecology of Pacific salmon. Edited by C.Groot, L. Margolis, and W.C. Clarke. University of British Columbia Press, Vancouver, B.C.: pp. 3-68.
- Brock, T. C. M., S. J. H. Crum, et al. (2004). "Comparing aquatic risk assessment methods for the photosynthesis-inhibiting herbicides metribuzin and metamitron." Environmental Pollution 130(3): 403-426.
- Brock, T. C. M., J. Lahr, et al. (2000). Ecological risk of pesticides in freshwater ecosystems. Part 1: herbicides. Alterra Report. Wageningen, The Netherlands.
- Brodeur, R. D., J. P. Fisher, et al. (2004). "Juvenile salmonid distribution, growth, condition, origin, and environmental and species associations in the Northern California Current." Fish. Bull 102: 25-46.
- Brown, G. T. (1985). The role of abandoned stream channels as over-wintering habitat for juvenile salmonids. The Faculty of Graduate Studies, Department of Forestry. British Columbia, The University of British Columbia: 134.
- Brown, L. R., P. B. Moyle, et al. (1994). "Historical decline and current status of coho salmon in California." North American Journal of Fisheries Management 14: 237 -261.
- Brust, K., O. Licht, et al. (2001). "Effects of terbutryn on aufwuchs and *Lumbriculus variegatus* in artificial indoor streams." Environmental Toxicology and Chemistry 20(9): 2000-2007.
- Buchwalter, D. B., J. J. Jenkins, et al. (2003). "Temperature influences on water permeability and chlorpyrifos uptake in aquatic insects with differing respiratory strategies." Environmental Toxicology and Chemistry 22(11): 2806-2812.
- Buchwalter, D. B., J. F. Sandahl, et al. (2004). "Roles of uptake, biotransformation, and target site sensitivity in determining the differential toxicity of chlorpyrifos to second to fourth instar *Chironomus riparius* (Meigen)." Aquatic Toxicology 66(2): 149-157.

- Bugert, R. M., T. C. Bjornn, et al. (1991). "Summer habitat use by young salmonids and their responses to cover and predators in a small southeast Alaskan stream." Transaction of American Fisheries Society 120: 474-485.
- Burgner, R. L. (1991). "The life history of sockeye salmon (*Oncorhynchus nerka*). " in Groot, C. and L. Margolis (editors). Life history of Pacific salmon. University of British Columbia Press, Vancouver, British Columbia, Canada.: 3-117.
- Burke, C., P. Anderson, et al. (2006). Surface water monitoring program for pesticides in salmonid-bearing streams, 2003-2005. Report, Washington State Department of Ecology: 114.
- Burton, C. A., J. A. Izbicki, et al. (1998). "Water-quality trends in the Santa Ana River at MWD Crossing and below Prado Dam, Riverside County, California." U.S. Geological Survey Water-Resources Investigations Report 97-4173. Sacramento, California.
- Busack, C. (1990). "Yakima/Klickitat production project genetic risk assessment." Genetics Unit, Washington Department of Fisheries, Olympia, Washington.
- Busby, P. J., T. C. Wainwright, et al. (1996). Status review of steelhead from Washington, Oregon, and California. NOAA Technical Memorandum. Seattle, Washington, U.S. Department of Commerce, Northwest Fisheries Science Center.
- Bustard, D. R. and D. W. Narver (1975). "Aspects of the winter ecology of juvenile coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Salmo gairdneri*). " Journal of the Fisheries Board of Canada 32: 667-680.
- Bustard, D. R. and D. W. Narver (1975). "Preferences of juvenile coho salmon (*Oncorhynchus kisutch*) and cutthroat trout (*Salmo clarki*) relative to simulated alteration of winter habitat." Journal of the Fisheries Board of Canada 32: 681-687.
- Caffrey, J. M. (1996). "Glyphosate in fisheries management." Hydrobiologia 340: 259-263.
- Call, D., L. Brooke, et al. (1983). Toxicity, bioconcentration, and metabolism of five herbicides in freshwater fish, Prepared by University of Wisconsin, Center for Lake Superior Environmental Studies for the EPA: 113.
- Call, D. J., L. T. Brooke, et al. (1987). "Bromachil and diuron herbicides: Toxicity, uptake, and elimination in freshwater fish." Archives of Environmental Contamination and Toxicology 16: 607-613.

Cannata, S. P. (1998). *Observations of steelhead trout (Oncorhynchus mykiss), coho salmon (O. kisutch) and water quality of the Navarro River estuary/lagoon May 1996 to December 1997. Draft Report.* Arcada, California, Humboldt State University Foundation: 48, plus tables and figures.

Carpenter, K. D., S. Sobieszczyk, et al. (2008). *Pesticide Occurrence and distribution in the lower Clackamas basin, Oregon, 2000-2005. U.S. Geological Survey Scientific Investigations, U.S. Department of the Interior: 98.*

Carter, J. L. and V. H. Resh (2005). *Pacific Coast rivers of the coterminous United States. Rivers of North America.* A. C. Benke and C. E. Cushing. Burlington, Massachusetts, Elsevier Academic Press: 541-590.

Casillas, E. (1999). *"Role of the Columbia River estuary and plume in salmon productivity." in Ocean conditions and the management of the Columbia River salmon, proceeding of a symposium, July 1, 1999.*

CBFWA (1990). *"Review of the history, development, and management of anadromous fish production facilities in the Columbia River basin." Columbia Basin Fish and Wildlife Authority, Portland, Oregon.*

CDFG (1993). *Restoring Central Valley streams: A plan for action. Report.* Yountville, California, CDFG: 217.

CDFG (1998). *Report to the Fish and Game Commission: a status review of the spring-run chinook salmon (Oncorhynchus tshawytscha) in the Sacramento River drainage. Candidate Species Status Report 98-01,* California Department of Fish and Game.

CDFG (2003). *"September 2002 Klamath River fish kill: preliminary analysis of contributing factors." ?????? ??????: ??-??*

CDFG (2007). *"Final 2006 California Commercial Landings." Available: <http://www.dfg.ca.gov/marine/landings06.asp> (February 2008).*

CDFG (2008). *Anadromous fish restoration program, GRANDTAB salmon data <http://www.delta.dfg.ca.gov/afpr/>, USFWS.*

CDPR (2008). *California department of pesticide regulation's surface water database. <http://www.cdpr.ca.gov/docs/emon/surfwtr/surfdes.htm>.*

- CDPR (2009). *Summary of pesticide use data 2008, indexed by commodity. California Department of Pesticide Regulation. Sacramento, CA.*
- CDPR (2010). *Summary of pesticide use data 2009, indexed by commodity. California Department of Pesticide Regulation. Sacramento, CA.*
- CEPA. (2007). "California EPA TMDL Program - Summary statistics for 2006 303(d) List. Download September 26, 2008. Available at http://www.swrcb.ca.gov/water_issues/programs/tmdl/303d_lists2006_approved.shtml."
- Chapman, D., G. T. Hillman, et al. (1994). "Status of summer/fall Chinook salmon in the mid-Columbia region." Report to Chelan, Douglas, and Grant County PUDs, Boise, Idaho.
- Chapman, D. and K. Witty (1993). "Habitat of weak salmon stocks in the Snake River basin and feasible recovery measures." Report to the Bonneville Power Administration, DOE/BP-99654-1, Portland, Oregon.
- Chapman, D. J. and B. E. Julius (2005). "The use of preventative projects as compensatory restoration." Journal of Coastal Research 40(Special Issue): 120-131.
- Chapman, D. W. and T. C. Bjornn (1969). Distribution of salmonids in streams, with special reference to food and feeding. Symposium on salmon and trout in streams. T. G. Northcote, H.R. MacMillan Lectures in Fisheries, University of British Columbia, Institute of Fisheries, Vancouver: 153 - 176.
- Clark, G. M., T. R. Maret, et al. (1998). Water quality in the upper Snake River Basin, Idaho and Wyoming, 1992-1995. U.S. Geological Survey Circular 1160.
- Climate Impacts Group (CIG) (2004). Overview of climate change impacts in the U.S. Pacific Northwest. Report. Seattle, Washington, University of Washington.
- Colgrove, D. J. and J. W. Wood (1966). "Occurrence and control of *Chondrococcus columnaris* as related to Fraser River sockeye salmon." Int. Pac. Salmon Fish. Comm. Prog. Rep. No. 15.
- Collier, T. K., J. E. O'Neill, et al. (2006). Toxic chemical contaminants and Puget Sound. Report. Seattle and Olympia, Washington, NMFS, Northwest Fisheries Science Center, Environmental Conservation Division and Washington Department of Fish and Wildlife, Fish Program, Marine Resources.

- Collins, B. D. and A. J. Sheikh (2005). "Historical reconstruction, classification, and change analysis of Puget Sound tidal marshes." Final report to Washington Department of Natural Resources Aquatic Resources Division, Olympia, Washington. Available: http://riverhistory.ess.washington.edu/project_reports/finalrpt_rev_aug12_2005.pdf (February 2008).
- Collis, K. (2007). "Research, monitoring, and evaluation of avian predation on salmonid smolts in the lower and mid-Columbia River - 2006 season summary." U.S. Geological survey report to Bonneville Power Administration and the U.S. Army Corps of Engineers.
- Conaway, C. H., S. Squire, et al. (2003). "Mercury speciation in the San Francisco Bay estuary." Marine Chemistry 80(2-3): 199-225.
- Connor, W. P., H. L. Burge, et al. (2002). "Juvenile life history of wild fall Chinook salmon in the Snake and Clearwater Rivers." North American Journal of Fisheries Management 22: 703-712.
- Connor, W. P., J. G. Sneva, et al. (2005). "Two alternative juvenile life history types for fall Chinook salmon in the Snake River basin." Transactions of the American Fisheries Society 134: 291-304.
- Cook, D. (2008). Chinook salmon spawning study Russian River fall 2002-2007. Report. Santa Rosa, California, Sonoma County Water Agency: 24.
- Cope, O. B., E. M. Wood, et al. (1970). "Some chronic effects of 2,4-D on the bluegill (*Lepomis macrochirus*)." Transactions of the American Fisheries Society 90(1): 1-12.
- Corps, BPA, et al. (2007). *Biological Assessment of the Effects of the Federal Columbia River Power System and Mainstem Effects of Other Tributary Actions on Anadromous Salmonid Species Listed under the Endangered Species Act.*
- Crain, D. A., L. J. Guilette Jr., et al. (1997). "Alterations in steroidogenesis in alligators (*Alligator mississippiensis*) exposed naturally and experimentally to environmental contaminants." Environmental Health Perspectives 105(5): 528-533.
- Cremlyn, R. J. (1991). Agrochemicals: Preparation and mode of action. New York, NY, John Wiley and Sons.

- Crewson, M. J., M. J. Haggerty, et al. (2001). *Genetic characterization of Lake Ozette sockeye salmon for enhancement recovery strategies. Report. Neah and Olympia, Washington, Makah Fisheries Management, Washington Department of Fish and Wildlife, and Northwest Indian Fisheries Commission.*
- Crommentuijn, T., D. Sijm, et al. (2000). "Maximum permissible and negligible concentrations for some organic substances and pesticides." Journal of Environmental Management 58: 297-312.
- Cuffney, T. F., M. R. Meador, et al. (1997). *Distribution of fish, benthic invertebrate, and algal communities in relation to physical and chemical conditions, Yakima River Basin, Washington, 1990. Report, U.S. Geological Survey: 1-94.*
- Cuffney, T. R., M. R. Meador, et al. (1997). "Distribution of fish, benthic invertebrate, and algal communities in relation to physical and chemical conditions, Yakima River Basin, Washington 1990." U.S. Geological Survey, Water Resources- Investigation Report 96-4280, Raleigh, North Carolina.
- Cuppen, J. G. M., P. J. Van den Brink, et al. (1997). "Sensitivity of macrophyte-dominated freshwater microcosms to chronic levels of the herbicide linuron. II: Community metabolism and invertebrates." Ecotoxicology and Environmental Safety 38: 25-35.
- Daam, M. A., A. M. F. Rodrigues, et al. (2009). "Ecological effects of the herbicide linuron in tropical freshwater microcosms." Ecotoxicology and Environmental Safety 72: 410-423.
- Daam, M. A. and P. J. Van den Brink (2007). "Effects of chlorpyrifos, carbendazim, and linuron on the ecology of small indoor aquatic microcosm." Archives of Environmental Contamination and Toxicology 53: 22-35.
- Daam, M. A., P. J. Van den Brink, et al. (2009). "Comparison of fate and ecological effects of the herbicide linuron in freshwater ecosystems between tropical and temperate regions." Ecotoxicology and Environmental Safety 72: 424-433.
- Davies, P. E. (1985). "The toxicology and metabolism of chlorothalonil in fish. II. Glutathione conjugates and protein binding." Aquatic Toxicology 7: 265-275.
- Davies, P. E. (1985). "The toxicology and metabolism of chlorothalonil in fish. III. Metabolism, enzymatics and detoxification in *Salmo* spp and *Galaxias* spp." Aquatic Toxicology 7: 277-299.

- Davies, P. E. (1985). "The toxicology and metabolism of chlorothalonil in fish. IV. Zinc coexposure and the significance of metallothionein in detoxification in *Salmo gairdneri*." Aquatic Toxicology 7: 301-306.
- Davies, P. E., L. S. J. Cook, et al. (1994). "Sublethal Responses to Pesticides of Several Species of Australian Freshwater Fish and Crustaceans and Rainbow Trout." Environmental Toxicology and Chemistry 13(8): 1341-1354.
- Davies, P. E. and R. W. G. White (1985). "The toxicology and metabolism of chlorothalonil in fish. I. Lethal levels for *Salmo gairdneri*, *Galaxias maculatus*, *G. truttaceus*, *G. auratus* and the fate of ^{14}C -TCIN." Aquatic Toxicology 7: 93-105.
- DeLorenzo, M. E., J. Lauth, et al. (1999). "Atrazine effects on the microbial food web in tidal creek mesocosms." Aquatic Toxicology 46(3-4): 241-251.
- Demko, D. B. and S. P. Cramer (2000). "Effects of pulse flows on juvenile Chinook migration in the Stanislaus River." Report of S.P. Cramer & Associates, Inc. to South San Joaquin Irrigation District, Manteca, California and Oakdale Irrigation District, Oakdale, California.
- DeNoyelles, F., W. D. Kettle, et al. (1982). "The responses of plankton communities in experimental ponds to atrazine, the most heavily used pesticide in the United States." Ecology 63(5): 1285-1293.
- Dewey, S. L. (1986). "Effects of the Herbicide Atrazine on Aquatic Insect Community Structure and Emergence." Ecology 67(1): 148-162.
- Dill, L. M. and A. H. G. Fraser (1984). "Risk of predation and the feeding behavior of juvenile coho salmon (*Oncorhynchus kisutch*)." Behavioral and Ecological Sociobiology 16: 65-71.
- Dill, L. M., R. C. Ydenberg, et al. (1981). "Food abundance and territory size in juvenile coho salmon (*Oncorhynchus kisutch*)." Canadian Journal Of Zoology 59: 1801-1809.
- Dineen, G. S., S. C. Harrison, et al. (2007). "Growth, production and bioenergetics of brown trout in upland streams with contrasting riparian vegetation." Freshwater Biology 52: 771-783.
- Dolloff, C. A. (1987). "Seasonal population characteristics and habitat use by juvenile coho salmon in a small southeast Alaska stream." Transaction of American Fisheries Society 116: 829 - 838.

- Domagalski, J. (2000). "Pesticides in surface water measured at select sites in the Sacramento River Basin, California, 1996–1998." U.S. GEOLOGICAL SURVEY Water-Resources Investigations Report 00-4203 National Water-Quality Assessment Program.
- Dow, A. (2010). Letter from Dow AgroScience (John Jachetta) to EPA (Arthur-Jean Williams) regarding triclopyr BEE ESA consultation with National Marine Fisheries Service dated October 7, 2010.
- Dröge, W. and R. Breitzkreutz (2000). "Glutathione and immune function." Proceedings of the Nutrition Society 59(4): 595-600.
- Dubrovsky, N. M., C. R. Kratzer, et al. (1998). "Water quality in the San Joaquin-Tulare Basins, California, 1992-1995." U.S. Department of the Interior, U.S. Geological Survey circular 1159, Reston, Virginia. Available:
<http://pubs.usgs.gov/circ/circ1159/circ1159.pdf> (February 2008).
- Ebbert, J. and S. Embry (2001). "Pesticides in surface water of the Yakima River Basin, Washington, 1999-2000 - their occurrence and an assessment of factors affecting concentrations and loads." U.S. Geological Survey, Water Investigations Report 01-4211, Portland Oregon.
- Ebbert, J. C., S. S. Embrey, et al. (2000). "Water quality in the Puget Sound basin, Washington and British Columbia, 1996-98." U.S. Department of the Interior, U.S. Geological Survey Circular 1216, Reston, Virginia. Available:
<http://pubs.usgs.gov/circ/circ1216/pdf/circ1216.pdf> (February 2008).
- Ehlke, R. D. and K. Keller (2003). "2002 Chum salmon spawnign ground surveys on the mainstem Columbia River and its Washington tributaries." Prepared for the Pacific States Marine Fisheries Commission, Vancouver, Washington by the Bonneville Power Administration, Portland, Oregon
- EPA (1995). Registration Eligibility Decision (RED). Linuron. Office of Pesticide Programs.
- EPA (1998). "Guidelines for ecological risk assessment." Federal Register 63(93): 26846-26924.
- EPA (1998). Registration Eligibility Decision (RED). Triclopyr. Office of Pesticide Programs. RED.
- EPA (1999). Folpet RED facts. Prevention, pesticides, and toxic substances. Washington, D.C., EPA.

EPA (1999). Registration Eligibility Decision (RED). Chlorothalonil. Office of Pesticide Programs.

EPA (2003). Captan analysis of risks to endangered and threatened Salmon and Steelhead. Office of Pesticide Programs.

EPA (2003). Chlorothalonil analysis of risks to endangered and threatened Salmon and Steelhead. Office of Pesticide Programs.

EPA (2003). Diuron analysis of risks to endangered and threatened Salmon and Steelhead. Office of Pesticide Programs.

EPA (2003). Registration Eligibility Decision (RED). Diuron. Office of Pesticide Programs. RED.

EPA (2004). 2,4-Dichlorophenoxyacetic acid analysis of risks to endangered and threatened Salmon and Steelhead. Office of Pesticide Programs. Washington, D.C., Environmental Protection Agency, Environmental Field Branch, Office of Pesticide Programs.

EPA (2004). Linuron analysis of risks to endangered and threatened Salmon and Steelhead. Office of Pesticide Programs.

EPA (2004). November 24, 2004 Notices. OPP-2004-0296, FRL-7685-3. EPA.

EPA (2004). Overview of the ecological risk assessment process in the Office of Pesticide Programs, U.S. Environmental Protection Agency - Endangered and threatened species effects determinations, Environmental Protection Agency - Office of Pesticide Programs.

EPA (2004). Registration Eligibility Decision (RED). Captan. Office of Pesticide Programs.

EPA (2004). Triclopyr butoxyethyl ester analysis of risks to endangered and threatened Salmon and Steelhead. Office of Pesticide Programs. Washington, D.C., Environmental Protection Agency, Environmental Field Branch, Office of Pesticide Programs.

EPA (2005). Registration Eligibility Decision (RED). 2,4-D. Office of Pesticide Programs.

EPA (2006). Registration eligibility decision (RED) for imazapyr. EPA. Office of Pesticide Programs.

EPA (2007). Environmental Protection Agency, Fish and Wildlife Service, and National Marine Fisheries Service Pesticide Coordination Meeting. December 10 - 12, 2007. National Conservation Training Center, Shepherdstown, WV

EPA (2007). Potential risks of labeled chlorothalonil uses to the federally listed California red-legged frog (Rana aurora draytonii). Pesticide effects determination. Office of Pesticide Programs.

EPA (2007). Risks of captan use to federally threatened California red-legged frog (Rana aurora draytonii). Pesticide effect determination. Office of Protected Resources.

EPA (2008). Risks of linuron use to federally threatened California red-legged frog (Rana aurora draytonii). Pesticide effects determination. Office of Pesticide Programs.

EPA (2009). Risks of 2,4-D Use to the Federally Threatened California Red-legged Frog (Rana aurora draytonii) and Alameda Whipsnake (Masticophis lateralis euryxanthus). Office of Pesticide Programs. Washington, D.C., Environmental Protection Agency, Environmental Fate and Effects Division, Office of Pesticide Programs.

EPA (2009). Risks of diuron use to federally threatened California red-legged frog (Rana aurora draytonii). Effect determination. Office of Pesticide Programs.

EPA (2009). Risks of triclopyr use to federally threatened California red-legged frog (Rana aurora draytonii). Pesticide Effects Determination. Office of Pesticide Programs. Washington, D.C., Environmental Protection Agency, Environmental Fate and Effects Division, Office of Pesticide Programs.

Everest, F. H. and D. W. Chapman (1972). "Habitat selection and spatial interaction by juvenile Chinook salmon and steelhead trout." Journal of the Fisheries Research Board of Canada 29: 91 - 100.

Fausch, K. D. (1984). "Profitable stream positions for salmonids: relating specific growth rate to net energy gain." Canadian Journal Of Zoology 62: 441-451.

Fausch, K. D. (1993). "Experimental analysis of microhabitat selection by juvenile steelhead (Oncorhynchus mykiss) and coho salmon (O. kisutch) in a British Columbia stream." Canadian Journal of Fisheries and Aquatic Sciences 50: 1198-1207.

Fausch, K. D. and T. G. Northcote (1992). "Large woody debris and salmonid habitat in a small coastal British Columbia stream." Canadian Journal of Fisheries and Aquatic Sciences 49: 682-693.

Ferguson, J. (2006). *Estimated survival of juvenile salmonids through the lower Columbia River and estuary, and mortality due to avian predation* Seattle, WA, NMFS, Northwest Fisheries Science Center: 2.

Figueroa, E. B. and R. A. Woods (2007). "Industry output and employment projections to 2016." Monthly Labor Review 130: 53-85.

Finney, B. P., I. Gregory-Eaves, et al. (2002). "Fisheries productivity in the northeastern Pacific Ocean over the past 2200 years." Nature 416: 729-733.

Fischer, J. R., M. C. Quist, et al. (2010). "Assemblage and Population-Level Responses of Stream Fish to Riparian Buffers at Multiple Spatial Scales." Transactions of the American Fisheries Society 139(1): 185-200.

Fisher, F. W. (1994). "Past and present status of Central Valley Chinook salmon." Conservation Biology 8: 870-873.

Fisher, T. and R. Hinrichsen (2006). "Abundance-based trend results for Columbia Basin salmon and steelhead ESUs. ." Bonneville Power Administration, Portland, Oregon.

Flagg, T. A., F. W. Waknitz, et al. (1995). *The effect of hatcheries on native coho salmon populations in the lower Columbia River. Uses and effects of cultured fishes in aquatic systems.* H. Schramm and R. Piper. Bethesda, MD, American Fisheries Society: 366-375.

Foott, J. S., R. Harmon, et al. (2003). "Ceratomyxosis resistance in juvenile Chinook salmon and steelhead trout from the Klamath River, 2002 Investigational Report." U.S. Fish and Wildlife Service, California-Nevada Fish Health Center, Anderson, California. [768 Kb] at KRIS Web site.

Ford, J. K. B. and G. M. Ellis (2006). "Selective foraging by fish-eating killer whales *Orcinus orca* in British Columbia." Marine Ecology Progress Series(316): 185-199.

Foss, S. F., P. R. Ode, et al. (2007). "Non-indigenous aquatic organisms in the coastal waters of California. ." California Fish and Game 93(3): 111-129.

- Francis, R. C. and S. R. Hare (1994). "Decadal-scale regime shifts in the large marine ecosystems of the North-east Pacific: A case for historical science." Fisheries Oceanography 3(4): 279-291.
- Fresh, K. D. (1997). *The role of competition and predation in the decline of Pacific salmon and steelhead.* Pacific salmon and their ecosystems: status and future options D. J. Stouder, P. A. Bisson and R. J. Naiman. New York, Chapman and Hall: 245-275.
- Fresh, K. L., E. Casillas, et al. (2005). *Role of the estuary in the recovery of Columbia River Basin salmon and steelhead: An evaluation of the effects of selected factors on population viability.* NOAA Technical Memorandum. Seattle, Washington, U.S. Department of Commerce: 155.
- Friesen, T. A. and D. L. Ward (1999). "Management of the northern pikeminnow and implications for juvenile salmonid survival in the lower Columbia and Snake rivers." North American Journal of Fisheries Management(19): 406-420.
- Fry, D. H., Jr., (1961). "King salmon spawning stocks of the California Central Valley, 1940-1959." California Department of Fish and Game 47(1): 55-71.
- Fuhrer, G. J., J. L. Morace, et al. (2004). "Water quality in the Yakima Basin, Washington, 1999-2000." U.S. Department of the Interior, U.S. Geological Survey Circular 1237, water research investigations report 03-4026, Portland, Oregon. Available: <http://pubs.usgs.gov/wri/wri034026/pdf/wri034026.pdf> (February 2008).
- Fulton, L. A. (1968). "Spawning areas and abundance of Chinook salmon, *Oncorhynchus tshawytscha*, in the Columbia River basin - past and present." U.S. Fish and Wildlife Service Spec. Sci. Rep. Fish. 571.
- Fulton, L. A. (1970). "Spawning areas and abundance of steelhead trout and coho, sockeye, and chum salmon in the Columbia River basin - past and present." NOAA Technical Report, NMFS-SSRF 618.
- Gagnon, M. M. and R. A. Rawson (2009). "Diuron increases spinal deformity in early life-stage pink snapper *Pagrus auratus*." Marine Pollution Bulletin 58: 1078-1095.
- Geisy, J. P., K. R. Solomon, et al. (1999). Chlorpyrifos: Ecological risk assessment in North American aquatic environments. New York City, Springer-Verlag.

- Giannico, G. R. (2000). "Habitat selection by juvenile coho salmon in response to food and woody debris manipulations in suburban and rural stream sections." Canadian Journal of Fisheries and Aquatic Sciences 57: 1804-1813.
- Giannico, G. R. and M. C. Healey (1999). "Ideal free distribution theory as a tool to examine juvenile coho salmon (*Oncorhynchus kisutch*) habitat choice under different conditions of food abundance and cover." Canadian Journal of Fisheries and Aquatic Sciences 56: 2362 - 2373.
- Giddings, J. M. (2009). Carbaryl exposure and potential effects on threatened and endangered salmonids. CSI-USA, Compliance Services International for Bayer CropSciences LP: 73.
- Gilliom, R. J. (2007). "Pesticides in U.S. streams and groundwater." Environmental Science & Technology 41(10): 3407-3413.
- Gilliom, R. J., J. E. Barbash, et al. (2006). "The quality of our nation's waters; pesticides in the nation's streams and groundwater, 1992-2001." U.S. Geological Survey circular 1291: 52 p.
- Gomez, L., J. Masot, et al. (1998). "Acute 2,4-D poisoning in tench (*Tinca tinca* L.): Lesions in the hematopoietic section of the kidney." Archives of Environmental Contamination and Toxicology 35: 479-483.
- Good, J. W. (2000). 3.3 Summary and current status of Oregon's estuarine ecosystems. Oregon State of the Environment: Full Report. Salem, OR.
- Good, T. P., R. S. Waples, et al. (2005). Updated status of federally listed ESUs of West Coast salmon and steelhead. NOAA Technical Memorandum. Seattle, Washington, U.S. Department of Commerce: 1-598.
- Gosselin, R. E., R. P. Smith, et al. (1984). Clinical toxicology of commercial products. Baltimore, MD, Williams and Wilkins.
- Greco, L., E. Capri, et al. (2007). "Biochemical responses in *Salmo salar* muscle following exposure to ethynylestradiol and tributyltin." Chemosphere 68(3): 564-571.
- Gregory, S. (2000). 3.5 Summary of current status and health of Oregon's riparian areas. Oregon State of the Environment: Full Report. Salem, OR, Oregon Progress Board.

- Gregory, S. V. and P. A. Bisson (1997). "Degradation and loss of anadromous salmonid habitat in the Pacific Northwest." pages 277-314 in Stouder D.J, Bisson, P.A, Naiman, R.J. (editors) Pacific salmon and their ecosystems. New York: Chapman and Hall.
- Greswell, R. E. (1999). "Fire and aquatic ecosystems in forested biomes of North America." Transactions of the American Fisheries Society 128(2): 193-221.
- Groot, C. and L. Margolis (1991). Pacific salmon life histories. Vancouver, BC, Canada, Vancouver University of British Columbia Press.
- Gruessner, B. and M. C. Watzin (1996). "Response of aquatic communities from a Vermont stream to environmentally realistic atrazine exposure in laboratory microcosms." Environmental Toxicology and Chemistry 15(4): 410-419.
- Guillen, G. (2003). "Klamath River fish die-off, September 2002: causative factors of mortality." Report number AFWO-F-02-03. U.S. Fish and Wildlife Service, Arcata Fish and Wildlife Office. Arcata, California 128 p.
- Gustafson, R. G., T. C. Wainwright, et al. (1997). Status review of sockeye salmon from Washington and Oregon. NOAA Technical Memorandum U.S. Department of Commerce: 282.
- Haggerty, M. J., A. C. Ritchie, et al. (2007). "Lake Ozette sockeye limiting factors analysis: draft 8_1." Prepared for the Makah Indian Tribe and NOAA Fisheries in cooperation with the Lake Ozette Sockeye steering Committee, Port, Angeles, Washington.
- Hall, L. W., W. D. Killen, et al. (2006). "Characterization of benthic communities and physical habitat in the Stanislaus, Tuolumne, and Merced Rivers, California." Environmental Monitoring and Assessment 115(1-3): 223-264.
- Hallock, R. J., W. F. Van Woert, et al. (1961). "An evaluation of stocking hatchery-reared steelhead rainbow trout (*Salmo gairdneri gairdneri*) in the Sacramento River System." Fisheries Bulletin 114: 1-74.
- Hanson, B., R. W. Baird, et al. (2005). "Focal behavioral observations and fish-eating killer whales: improving our understanding of foraging behaviour and prey selection." Abstract at the 16th Biennial Conference on the Biology of Marine Mammals, San Diego, California.
- Hanson, C. H. (1997). Acute temperature tolerance of juvenile Chinook salmon from the Mokelumne River. Walnut Creek, California, Hanson Environmental, Inc.: 11.

- Hard, J., R. P. J. Jones, et al. (1992). *Pacific salmon and artificial propagation under the endangered species act. NOAA Technical Memorandum*. Seattle, Washington, U.S. Department of Commerce.
- Hare, S. R. and N. J. Mantua. (2000). "Empirical evidence for North Pacific regime shifts in 1977 and 1989." *Progress in Oceanography* 16-Feb(4-Feb): 103-145.
- Hartgers, E. M., G. H. Aalderink, et al. (1998). "Ecotoxicological threshold levels of a mixture of herbicides (atrazine, diuron and metolachlor) in freshwater microcosms." *Aquatic Ecology* 32: 135-152.
- Hartt, A. C. and M. B. Dell (1986). "Early oceanic migrations and growth of juvenile Pacific salmon and steelhead trout." *International North Pacific Fisheries Commission Bulletin* 47.
- Harvey, C. J., B. J. Peterson, et al. (1998). "Biological Responses to Fertilization of Oksrukuyik Creek, a Tundra Stream." *Journal of the North American Benthological Society* 17(2): 190-209.
- Hayes, S. A., M. H. Bond, et al. (2004). "Interactions between endangered wild and hatchery salmonids: can the pitfalls of artificial propagation be avoided in small coastal streams?" *Journal of Fish Biology* 65: 101 - 121.
- Hayman, R. A., E. M. Beamer, et al. (1996). "FY 1995 Skaig River chinook restoration research." *Report by Skagit System Cooperative, La Conner, Washington: 54p + Appendices.*
- Healey, M. C. (1982). *Juvenile Pacific salmon in estuaries: the life support system. Esuarine Comparisons*. V. S. Kennedy. New York, NY, Academic Press: 315-341.
- Healey, M. C. (1991). *Life history of Chinook salmon (Oncorhynchus tshawytscha). Pacific salmon life histories*. C. Groot and L. Margolis. Vancouver, Canada, University of British Columbia Press: 311-394.
- Hebdon, J. L., P. Kline, et al. (2004). "Evaluating reintroduction strategies for Redfish Lake sockeye salmon captive broodstock strategy." pages 401-413 in Nickum, M.J., P.M. Mazik, J.G. Nickum, and D.D. MacKinlay (editors). Propogated fish in resource management. American Fisheries Society, Symposium 44, American Fisheries Society, Bethesda, Maryland.

- Henning, J. A., R. E. Gresswell, et al. (2006). "Juvenile salmonid use of freshwater emergent wetlands in the floodplain and its implications for conservation management." North American Journal of Fisheries Management 26: 367-376.
- Hernando, M. D., M. Ejerhoon, et al. (2003). "Combined toxicity effects of MTBE and pesticides measured with *Vibrio fishcheri* and *Daphnia magna* bioassays." Water Research 37: 4091-4098.
- Hinck, J. E., C. J. Schmitt, et al. (2004). "Biomonitoring of environmental status and trends (BEST) program: environmental contaminants and their effects on fish in the Columbia River basin." U.S. Department of the Interior, U.S. Geological Survey, Columbia Environmental Research Center, scientific investigation report 2004-5154, Columbia, Missouri.
- Hirsch, R. M., W.M. Alley, and W.G. Wilbur (1988). Concepts for a national water-quality assessment program.
- Howell, P., K. Jones, et al. (1985). Stock assessment of Columbia River anadromous salmonids. Volume II: Steelhead stock summaries stock transfer guidelines-information needs. Final report to Bonneville Power Administration. Contract. Portland, Oregon, Bonneville Power Administration. II: 1032.
- Hughes, J. S. (1973). Acute toxicity of thirty chemicals to striped bass (*Morone saxatilis*). Western Association of State Game and Fish Commissioners.
- Huryn, A. D. (1996). "An appraisal of the Allen paradox in a New Zealand trout stream." Limnology and Oceanography 41(2): 243-252.
- Huryn, A. D. (1998). "Ecosystem-level evidence for top-down and bottom-up control of production in a grassland stream system." Oecologia 115(1-2): 173-183.
- ICTRT (2003). Independent populations of chinook, steelhead, and sockeye for listed evolutionarily significant units within the Interior Columbia River Domain. Report. Seattle, Washington, NMFS, Northwest Fisheries Science Center: 173.
- ICTRT (2007). Viability criteria for application to Interior Columbia Basin salmonid ESUs. Technical Review Draft Report. Seattle, Washington, NMFS, Northwest Fisheries Science Center: 90.
- ICTRT (2008). Entiat spring Chinook population. Technical Working Draft Report. Seattle, Washington, NMFS, Northwest Fisheries Science Center: 13.

ICTRT (2008). *Methow spring Chinook salmon. Working Draft*. Seattle, Washington, NMFS, Northwest Fisheries Science Center: 12.

ICTRT (2008). *Wenatchee River spring Chinook population. Technical Working Draft Report*. Seattle, Washington, NMFS, Northwest Fisheries Science Center: 13.

Incardona, J. P., T. K. Collier, et al. (2004). "Defects in cardiac function precede morphological abnormalities in fish embryos exposed to polycyclic aromatic hydrocarbons." Toxicology and Applied Pharmacology 196: 191205.

IPCC (2001). *Climate change 2001: impacts adaptation and vulnerability. Summary for policy makers and technical summary*. Geneva, Switzerland, IPCC.

IPCC (2007). *Contribution of working groups I, II, and III to the fourth assessment report of the intergovernmental panel on climate change*. R. K. Pachauri and A. Reisinger. Geneva, Switzerland, Intergovernmental Panel on Climate Change: 104.

ISG (1996). "Return to the river: restoration of salmonid fishes in the Columbia River ecosystem." Northwest Power Planning Council, Independent Science Group report 96-6, Portland, Oregon. Available: <http://www.ecy.wa.gov/programs/WR/wstf/images/pdf/normrivr.pdf> (February 2008).

Jacobs, R., G. Larson, et al. (1996). *The sockeye salmon *Oncorhynchus nerka* population in Lake Ozette, Washington, USA. Technical Report Available from Denver Service Center, Technical Information Center, P. O. Box 25287, Denver, Colorado 80225-0287: 104.*

Jarrard, H. E., K. R. Delaney, et al. (2004). "Impacts of carbamate pesticides on olfactory neurophysiology and cholinesterase activity in coho salmon (*Oncorhynchus kisutch*)."
Aquatic Toxicology 69: 133-148.

Jeffres, C. A., J. J. Opperman, et al. (2008). "Ephemeral floodplain habitats provide best growth conditions for juvenile Chinook salmon in a California river." Environmental Biology of Fishes 83: 449-458.

Jenkins, D. G. and A. L. Buikema (1998). "Do similar communities develop in similar sites? A test with zooplankton structure and function." Ecological Monographs 68(3): 421-443.

Jenkins, J., P. Jepson, et al. (2004). *Watershed-based ecological risk assessment of pesticide use in Western Oregon: A conceptual framework. Report to Oregon Department of Agriculture*.

Jepsen, D. B. and K. A. Leader (2008). Abundance monitoring of juvenile salmonids in Oregon coastal and Lower Columbia streams, 2007. Monitoring Program Report. Salem, OR, Oregon Department of Fish and Wildlife. OPSW-ODFW-2008-1: 19.

JISAO (2007). "Pacific Northwest impacts of climate change." accessed at <http://www.jisao.washington.edu/main.html> accessed on 6/22/08: 9.

Jobling, S., T. Reynolds, et al. (1995). "A variety of environmentally persistent chemical, including some phthalate plasticizers, are weakly estrogenic." Environmental Health Perspectives 103(6): 582-587.

Johnson, A. and J. Cowles (2003). Washington state surface water monitoring program for pesticides in salmon habitat for two index watersheds: A study for the Washington State Department of Agriculture conducted by the Washington State Department of Ecology. Publication No. 03-03-104. : 38 p.

Johnson, A. and A. Newman (1983). "Water quality in the gap-to-gap reach of the Yakima River, June - October 1982." Washington Department of Ecology, Olympia, Washington. Available: <http://www.ecy.wa.gov/biblio/83e17.html> (February 2008).

Johnson, K. R., P. C. Jepson, et al. (2008). "Esfenvalerate-induced case-abandonment in the larvae of the caddisfly (*Brachycentrus americanus*)." Environmental Toxicology and Chemistry 27(2): 397-403.

Johnson, L., T. K. Collier, et al. (2002). "An analysis in support of sediment quality thresholds for polycyclic aromatic hydrocarbons (PAHs) to protect estuarine fish." Aquatic Conservation: Marine Freshwater Ecosystems 12: 517-538.

Johnson, O. W., T. A. Flagg, et al. (1991). Status review for Lower Columbia River coho salmon. NOAA Technical Memorandum. Seattle, WA, Department of Commerce. NMFS F/NWC-202.

Johnson, O. W., W. S. Grant, et al. (1997). Status review of chum salmon from Washington, Oregon, and California. NOAA Technical Memorandum. Seattle, WA, U.S. Department of Commerce: 280.

Johnson, S. W., J. F. Thedinga, et al. (1992). "Life history of juvenile ocean-type Chinook salmon (*Oncorhynchus tshawytscha*) in the Situk River, Alaska." Canadian Journal of Fisheries and Aquatic Sciences 49(12): 2621-2629.

Johnson, W. W. and M. T. Finley (1980). *Handbook of acute toxicity of chemicals to fish and aquatic invertebrates*, USFWS (EPA MRID 40094602).

Jolly, C., I. Katsiadaki, et al. (2009). "detection of the anti-androgenic effect of endocrine disrupting environmental contaminants using in vivo and in vitro assays in the three-spined stickleback." *Aquatic Toxicology* 92: 228-239.

Joy, J. (2002). "Upper Yakima River basin suspended sediment and organochlorine pesticide total maximum daily load evaluation." *Washington Department of Ecology, publication number 02-30-012, Olympia, Washington. Available: <http://www.ecy.wa.gov/biblio/0203012.html> (February 2008).*

Joy, J. and A. Madrone (2002). "Data summary: upper Yakima River basin suspended sediment and organochlorine TMDL evaluation." *Washington Department of Ecology, publication number 02-30-032, Olympia, Washington. Available: <http://www.ecy.wa.gov/biblio/0203032.html> (February 2008).*

Juttner, I., A. Peither, et al. (1995). "An outdood mesocosm study to assess ecotoxicological effects of atrazine on a natural plankton community." *Archives of Environmental Contamination and Toxicology* 29(4): 435-441.

Kagan, J. S., J. C. Hak, et al. (1999). "Oregon gap analysis project final report:a geographic approach to planning for biological diversity." *Oregon Natural Heritage Program, Portland, Oregon.*

Kalleberg, H. (1958). "Observations in a stream tank of territoriality and competition in juvenile salmon and trout (*Salmo salar* L. and *Salmo trutta* L.)." *Report of the Institute of Freshwater Research, Drottningholm* 39: 55-98.

Kammerer, J. C. (1990). "Largest rivers in the United States." *Water Fact Sheet. U.S. Department of the Interior, U.S. Geological Survey, open file report 87-242, Reston, Virginia. Available: <http://pubs.usgs.gov/of/1987/ofr87-242/pdf/ofr87242.pdf> (February 2008).*

Kamrin, M. (1997). *Pesticide profiles: toxicity, environmental impact, and fate*, CRC Press.

Kasai, F. and T. Hanazato (1995). "Effects of the triazine herbicide, Simetryn, on freshwater plankton communities in experimental ponds." *Environmental Pollution* 89(2): 197-202.

- Kashian, D. R. and S. I. Dodson (2001). "Effects of common-use pesticides on developmental and reproductive processes in *Daphnia*." Toxicology and Industrial Health 18: 225-235.
- Kemmerich, J. (1945). "A review of the artificial propagation and transplantation of the sockeye salmon of the Puget Sound area in the State of Washington conducted by the federal government from 1896 to 1945." Report to the Regional Director, U.S. Fish and Wildlife Service.
- Kiely, T., D. Donaldson, et al. (2004). Pesticides industry sales and usage 2000 and 2001 market estimates, U.S. Environmental Protection Agency, Biological and Economic Analysis Division: 33.
- Kier Associates (1991). "Long range plan for the Klamath River Basin Conservatin Area Fishery Restoration Program." U.S. Fish and Wildlife Service Klamath River Fishery Resource Office. Yreka, California. [8.5 MB] at KRIS Web site: 403 p.
- King County (2002). "Literature review of endocrine disruptors in secondary treated effluent: toxicological effects in aquatic organisms. King County Department of Natural Resources, Seattle, Washington 98104."
- King, K. W. and J. C. Balogh (2010). "Chlorothalonil and 2,4-D losses in surface water discharge from a manged turf watershed." Journal of environmental monitoring 12: 1601 - 1612.
- Knauer, K., A. Sobek, et al. (2007). "Reduced toxicity of diuron to freshwater green alga *Psuedokirchneriella subcapitata* in the presence of black carbon." Aquatic Toxicology 83: 143-148.
- Koehler, M. E., K. L. Fresh, et al. (2006). "Diet and bioenergetics of lake-rearing juvenile Chinook salmon in Lake Washington." Transaction of American Fisheries Sociery 135: 1580-1591.
- Kolpin, D. W., E. T. Furlong, et al. (2002). "Pharmaceuticals, hormones, and other organic wastewater contaminants in US streams, 1999-2000: A national reconnaissance." Environmental Science & Technology 36(6): 1202-1211.
- Koplin, D. W., E. T. Furlong, et al. (2002). "Pharmaceuticals, hormones, and other organic wastewater contaminants in U.S. Streams,1999-2000: A national reconnaissance." Environmental Science and Technology 36(6): 1202-1211.

- kostow, K. K., A. R. Marshall, et al. (3003). "Naturally spawning hatchery steelhead contribute to smolt production but experience low reproductive success." Transactions of the American Fisheries Society 132: 780-790.*
- Kreutzweiser, D. P. and S. S. Capell (1992). "A simple stream-side test system for determining acute lethal and behavioral effects of pesticides on aquatic insects." Environmental Toxicology and Chemistry 11: 993-999.*
- Krieger, R. I. and W. C. Krieger (2001). Handbook of pesticides toxicology: Principles. New York, NY, Academic Press: 1711-1742.*
- Kruckeberg, A. R. (1991). "The natural history of Puget Sound Country." University of Washington Press, Seattle, Washington.*
- La Riviere, M. G. (1991). "The Ozette Lake sockeye salmon enhancement program." Makah Fisheries Management Department, unpubl. Rep.: 9 p.*
- Lambright, C., J. Ostby, et al. (2000). "Cellular and molecular mechanisms of action of linuron: an antiandrogenic herbicide that produces reproductive malformations in male rats." Toxicological Sciences 56: 389-399.*
- Lande, R. (1993). "Risks of population extinction from demographic and environmental stochasticity and random catastrophs." The American Naturalist 142: 911-927.*
- Laufle, J. C., G. B. Pauley, et al. (1986). "Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Pacific northwest): coho salmon." National Coastal Ecosystem Team, Division of Biological Services, Research and Development, Fish and Wildlife Service, U.S. Department of Interior, Vicksburg, Mississippi.*
- Lawson, P. W., E. P. Bjorkstedt, et al. (2007). "Identification of historical populations of coho salmon (*Oncorhynchus kisutch*) in the Oregon coast evolutionary significant unit." NOAA Technical Memorandum NMFS-NWFSC-79: 120.*
- LCFRB (2004). "Lower Columbia salmon recovery and fish and wildlife subbasin plan."*
- Lehninger, A. L. (1975). Biochemistry. New York, NY, Worth Publishers.*
- Leidy, R. A. (1984). "Distribution and ecology of stream fishes in the San Francisco Bay drainage." Hilgardia 52: 1-175.*

- Leidy, R. A., G. S. Becker, et al. (2005). *Historical distribution and current status of steelhead/rainbow trout (Oncorhynchus mykiss) in streams of the San Francisco Estuary, California*. Oakland, CA, Center for Ecosystem Management and Restoration.
- Levin, P. S. (2003). "Regional differences in responses of Chinook salmon populations to large-scale climatic patterns." Journal of Biogeography 30(5): 711-717.
- Lichatowich, J. A. (1999). *"Salmon without rivers. A history of the Pacific salmon crisis."* Island Press, Washington D.C.
- Liebig, M., G. Schmidt, et al. (2008). "Direct and indirect effects of pollutants on algae and algivorous ciliates in an aquatic indoor microcosm." Aquatic Toxicology 88(2): 102-110.
- Liess, M. and R. Schulz (1999). "Linking insecticide contamination and population response in an agricultural stream." Environmental Toxicology and Chemistry 18: 1948-1955.
- Likens, G. E., F. H. Bormann, et al. (1970). "Effects of forest cutting and herbicide treatment on nutrient budgets in Hubbard Brook watershed ecosystem." Ecological Monographs 40(1): 23-&.
- Limm, M. P. and M. P. Marchetti (2009). "Juvenile Chinook salmon (*Oncorhynchus tshawytscha*) growth in off-channel and main-channel habitats on the Sacramento River, CA using otolith increment widths." Environmental Biology of Fishes 85: 141-151.
- Lin, J., J. Hetrick, et al. (1998). PRZM-EXAMS Overview for the July 29, 2008 Science Advisory Panel meeting, Office of Pesticide Programs.
- Lindley, S. T., R. S. Schick, et al. (2006). "Historical population structure of Central Valley steelhead and its alterations by dams." San Francisco Estuary & Watershed Science 4(1): 1-19.
- Linley, T. J. (2001). "A comparison of first feeding characteristics in two populations of Chinook salmon." Transaction of American Fisheries Society 130: 519-525.
- Long, J. W. and M. R. Siegel (1975). "Mechanism of action and fate of the fungicide chlorothalonil (2,4,5,6-tetrachloroisophthalonitrile) in biological systems: 2. In vitro reactions." Chemico-Biological Interactions 10(6): 383-394.

- Lukens, R. J. (1966). "The fungitoxicity of compounds containing trichloromethylthio-group." Journal of Agricultural and Food Chemistry 14: 365-367.
- Magnusson, A. and R. Hilborn (2003). "Estuarine Influence on Survival Rates of Coho (*Oncorhynchus kisutch*) and Chinook Salmon (*Oncorhynchus tshawytscha*) Released from Hatcheries on the U. S. Pacific Coast." Estuaries 26(4): 1094-1103.
- Makah Fisheries Management (2000). "Lake Ozette sockeye hatchery and genetic management plan. Biological assessment Section 7 consultation." available from Makah Fisheries, P. O. Box 115, Neah Bay, Washington 98357.
- Mangel, M. (1994). "Climate change and salmonid life history variation." Deep-Sea Research 41(1): 75-106.
- Mantua, N. J., S. R. Hare, et al. (1997). "A pacific interdecadal climate oscillation with impacts on salmon production." Bulletin of the American Meteorological Society(78): 1069-1079.
- Markle, J., O. Kalman, et al. (2005). Mitigating Diazinon runoff into waterways from fruit and nut orchards in the Sacramento Valley. Coalition for Urban/Rural Environmental Stewardship. Report funded by CALFED.: 23.
- Marr, J. C. (1943). "Age, length, and weight studies of three species of Columbia River salmon (*Oncorhynchus keta*, *O. gorbusha*, and *O. kisutch*)." Stanford Ichthyology Bulletin 2: 157-197.
- Marshall, A. R., C. Smith, et al. (1995). "Genetic diversity units and major ancestral lineages for chinook salmon in Washington." in C. Busack and J.B. Shaklee (editors). Genetic diversity units and major ancestral lineages of salmonid fishes in Washington. Wash. Dep. Fish Wildl. Tech. Rep. RAD 9502.
- Marshall, D. E. and E. W. Britton (1990). "Carrying capacity of coho salmon streams." Canadian Reportns on Fisheries and Aquatic Science(2058): 1-32.
- Martins, J., M. Saker, et al. (2007). "Oxygen consumption by *Daphnia magna* Straus as a marker of chemical stress in the aquatic environment." Environmental Toxicology and Chemistry 26(9): 1987-1991.
- Mason, J. C. (1976). "Responses of uneryearling coho salmon to supplemental feeding I a natural stream." Journal of Wildlife Management 40: 775-778.

- Matthews, G. M. and R. S. Waples (1991). Status review for Snake River spring and summer Chinook salmon. NOAA Technical Memorandum. U.S. Department of Commerce, National Marine Fisheries Service. NMFS F/NWC-200.
- Maund, S., J. Biggs, et al. (2009). "The influence of simulated immigration and chemical persistence on recovery of macroinvertebrates from cypermethrin and 3,4-dichloroaniline exposure in aquatic microcosms." Pest Management Science 65(6): 678-687.
- Mayer, F. L. and M. R. Ellersieck (1986). Manual of acute toxicity: Interpretation and data base for 410 chemicals and 66 species of freshwater animals, USFWS. Resource Publication 160: 579.
- Mayes, M. A., D. C. Dill, et al. (1984). "Triclopyr triethylamine salt toxicity to life stages of the fathead minnow (*Pimephales promelas* Rafinesque)." Bulletin of Environmental Contamination and Toxicology 33: 339-347.
- McBride, J. R., H. M. Dye, et al. (1981). "Stress response of juvenile sockeye salmon (*Oncorhynchus nerka*) to the butoxyethanol ester of 2,4-Dichlorophenoxyacetic acid." Bulletin of Environmental Contamination and Toxicology 27: 877-884.
- McClure, M. M., E. E. Holmes, et al. (2003). "A large-scale, multispecies status assessment: Anadromous salmonids in the Columbia River basin." Ecological Applications 13(4): 964-989.
- McCormick, S. D., D. T. Lerner, et al. (2009). "Taking it with you when you go: How perturbations to the freshwater environment, including temperature, dams, and contaminants, affect marine survival of salmon." American Fisheries Society Symposium(69): 195-214.
- McCullough, D. A. (1999). A review and synthesis of effects of alterations to the water temperature regime on freshwater life stages of salmonids, with special reference to Chinook salmon. Report prepared for U.S. Environmental Protection Agency. Portland, Oregon, Columbia River Inter-Tribal Fish Commission: 279.
- McElhany, P., T. Backman, et al. (2003). Interim report on viability criteria for Willamette and Lower Columbia basin Pacific salmonids. Willamette/Lower Columbia Technical Recovery Team Report. Seattle, WA, National Marine Fisheries Service: 81.

- McElhany, P., T. Backman, et al. (2004). Status evaluation of salmon and steelhead populations in the Willamette and Lower Columbia River basins. Status Review. Willamette/Lower Columbia Technical Recovery Team. Seattle, WA, National Marine Fisheries Service, Northwest Fisheries Science Center: 114.
- McElhany, P., C. Busack, et al. (2006). Revised viability criteria for salmon and steelhead in the Willamette and Lower Columbia basins. Willamette/Lower Columbia Technical Recovery Team Report, NMFS and Oregon Department of Fish and Game. Review Draft: 178 p.
- McElhany, P., M. Chilcote, et al. (2007). Viability status of Oregon salmon and steelhead populations in the Willamette and lower Columbia basins. Willamette/Lower Columbia Technical Recovery Team Report. Seattle, Washington, NMFS and Oregon Department of Fish and Wildlife. Review Draft.
- McElhany, P., M. H. Ruckelshaus, et al. (2000). Viable salmonid populations and the recovery of evolutionarily significant units. NOAA Technical Memorandum. Seattle, Washington, U.S. Department of Commerce. NMFS-NWFSC-42: 156.
- McEwan, D. and T. A. Jackson (1996). "Steelhead restoration and management plan for California." California Department of Fish and Game, Sacramento, California.
- McEwan, D. R. (2001). Central Valley steelhead. Contributions to the biology of the Central Valley salmonids. R. L. Brown. Sacramento, California, California Department of Fish and Game: 1-43.
- McFarlane, G. A., S. Hayes, et al. (2008). Coho and Chinook salmon decline in California during the spawning seasons of 2007/08. Unpublished Report.
- McMahon, T. E. (1983). Habitat suitability index models: coho salmon, U.S. Department of Interior, Fish and Wildlife Service: 29.
- Meador, J., J. Stein, et al. (1995). "A review of bioaccumulation of polycyclic aromatic hydrocarbons by marine organisms." Environmental Contamination and Toxicology 143: 79-165.
- Meehan, W. R. and T. R. Bjornn (1991). "Salmonid distributions and life histories: cutthroat trout. Influence of forest and rangeland management on salmonid fishes and their habitats. ." Bethesda, American Fisheries Society Special Publication 19: 66-67.

- Menzie, C. M. (1969). *Metabolism of pesticides*. Washington, D.C., U.S. Department of Interior, Bureau of Sport Fisheries Wildlife.
- Merz, J. (2002). "Seasonal feeding habits, growth, and movement of steelhead trout in the lower Mokelumne River, California." California Fish and Game 88(3): 95-111.
- Metcalf, N. B., N. H. C. Fraser, et al. (1999). "Food availability and the nocturnal vs. diurnal foraging trade-off in juvenile salmon." Journal of Animal Ecology 68(2): 371-381.
- Michael, J. L. (2003). "Environmental fate and impacts of sulfometuron on watersheds in the southern United States." Journal of Environmental Quality 32(2): 456-465.
- Michael, J. L., D. P. Batzer, et al. (2006). "Fate of the herbicide sulfometuron methyl (Oust((R))) and effects on invertebrates in drainages of an intensively managed plantation." Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 36(10): 2497-2504.
- Michael, J. L., E. C. Webber, et al. (1999). "Hexazinone dissipation in forest ecosystems and impacts on aquatic communities." Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 29(7): 1170-1181.
- Miller, B. A. and S. Sadro (2003). "Residence time and seasonal movements of juvenile coho salmon in the ecotone and lower estuary of Winchester Creek, South Slough, Oregon." Transactions of the American Fisheries Society 132(3): 546-559.
- Miller, J. A. and C. A. Simenstad (1997). "A comparative assessment of a natural and created estuarine slough as rearing habitat for juvenile Chinook and coho salmon." Estuaries 20(4): 792-806.
- Miller, R. J. and E. L. Brannon (1982). "The origin and development of life history patterns in Pacific salmonids." pages 296-309 in Brannon, E.L. and E.O. Salo (editors). Proceedings of the Salmon and Trout Migratory Behavior Symposium. School of Fisheries, University of Washington, Seattle, Washington.
- Miller, R. R., J. D. Williams, et al. (1989). "Extinctions of North American fishes during the last century." Fisheries 14(6): 22-38.
- Mills, S. K. and J. H. Beatty (1979). "The propensity interpretation of fitness." Philosophy of Science 46: 263-286.

- Minshall, G. W., T. V. Royer, et al. (2001). "Response of the Cache Creek macroinvertebrates during the first 10 years following disturbance by the 1988 Yellowstone wildfires." Canadian Journal of Fisheries and Aquatic Sciences 58(6): 1077-1088.
- Montgomery, D. R., E. M. Beamer, G. R. Pess, and T. P. Quinn (1999). "Channel type and salmonid spawning distribution and abundance." Canadian Journal of Fisheries and Aquatic Sciences 56: 377-387.
- Moore, M. M. (1980). Factors influencing survival of juvenile steelhead rainbow trout (*Salmo gairdnerii gairdnerii*) in the Ventura River. Arcata, CA, Humboldt State University. Master of Science: 82.
- Morley, S. A., P. S. Garcia, et al. (2005). "Juvenile salmonid (*Oncorhynchus* spp.) use of constructed and natural side channels in Pacific Northwest rivers." Canadian Journal of Fisheries and Aquatic Sciences 62: 2811-2821.
- Morris, W. F. and D. F. Doak (2002). Quantitative conservation biology. Theory and practice of population viability analysis. Sunderland, MA, Sinauer Associates Inc.
- Mouchet, F., L. Gauchet, et al. (2006). "Comparative evaluation of genotoxicity of captan in amphibian larvae (*Xenopus laevis* and *Pleurodeles waltl*) using comet assay and the micronucleus test." Environmental Toxicology and Chemistry 21(3): 264-277.
- Mount, J. F. (1995). "California rivers: the conflict between fluvial process and land use." University of California Press, Berkeley and Los Angeles, California.
- Mukherjee, P., S. Alam, et al. (2006). "Persistence and dissipation of linuron (Alfalon-50wp) in pea cropped soil and its effects on soil microorganisms." Bulletin of Environmental Contamination and Toxicology 76: 407-414.
- Mundie, J. H., K. S. Simpson, et al. (1991). "Responses of stream periphyton and benthic insects to increases in dissolved inorganic phosphorus in a mesocosm." Canadian Journal of Fisheries and Aquatic Sciences 48(11): 2061-2072.
- Myers, J., C. Busack, et al. (2006). Historical population structure of Pacific salmonids in the Willamette River and Columbia River basins. NOAA Technical Memorandum. Seattle, Washington, U.S. Department of Commerce. NMFS-NWFSC-73: 311.
- Myers, J. M., R. G. Kope, et al. (1998). Status review of chinook salmon from Washington, Idaho, Oregon, and California. NOAA Technical Memorandum. Seattle, Washington, U.S. Department of Commerce. NMFS-NWFSC-35: 443.

- Neary, D. G., P. B. Bush, et al. (1993). "Fate, Dissipation and Environmental-Effects of Pesticides in Southern Forests - a Review of a Decade of Research Progress." Environmental Toxicology and Chemistry 12(3): 411-428.
- Nebeker, A. V. and G. S. Schuytema (1998). "Chronic effects of the herbicide diuron in freshwater cladocerans, amphipods, midges, monnows, worms, and snails." Archives of Environmental Contamination and Toxicology 35: 441-446.
- Neuwoehner, J., T. Zilberman, et al. (2010). "QSAR-analysis and mixture toxicity as diagnostic tools: influence of degradation on the toxicity mode of action of diuron in algae and daphnids." Aquatic Toxicology 97: 58-67.
- Nicholas, J., B. McIntosh, et al. (2005). Oregon coastal coho assessment. Salem, OR, Oregon Watershed Enhancement Board and Oregon Department of Fish and Wildlife: 69.
- Nickelson, T. E., J. W. Nicholas, et al. (1992). "Status of anadromous salmonids in Oregon coastal basins." Oregon Department of Fish and Wildlife, Resource Development Section and Ocean Salmon Management, Oregon Department of Fish and Wildlife, Portland, Oregon.
- Nickelson, T. E., J. D. Rodgers, et al. (1992). "Seasonal changes in habitat use by juvenile coho salmon (*Oncorhynchus kisutch*) in Oregon coastal streams." Canadian Journal of Fisheries and Aquatic Sciences 49: 783-789.
- Nickelson, T. E., M. F. Solazzi, et al. (1992). "Effectiveness of selected stream improvement techniques to create suitable summer and winter rearing habitat for juvenile coho salmon (*Oncorhynchus kisutch*) in Oregon coastal streams." Canadian Journal of Fisheries and Aquatic Sciences 49: 790-794.
- Nielsen, L. W. and I. Dahllof (2007). "Direct and indirect effects of the herbicides Glyphosate, Bentazone and MCPA on eelgrass (*Zostera marina*)." Aquatic Toxicology 82(1): 47-54.
- NLCD. (2001). "National land cover data. Multi-resolution land characteristics consortium. Downloaded May 2008. Available at: <http://www.mrlc.gov/index.php>".
- NMA (2007). "2004 State Mining Statistics." Available: http://www.nma.org/statistics/state_statistics_2004.asp# (February 2008).

NMFS (1999). Updated review of the status of the Upper Willamette River and Middle Columbia River ESUs of steelhead. Status Review. U.S. Department of Commerce, Seattle, WA, National Marine Fisheries Service: 44.

*NMFS (2001). Status review update for coho salmon (*Oncorhynchus kisutch*) from the Central California Coast and the California portion of the Southern Oregon/Northern California Coasts Evolutionary Significant Units. Status Review. Santa Cruz, California, National Marine Fisheries Service, Southwest Fisheries Science Center: 34.*

NMFS (2005). Final assessment NOAA Fisheries' Critical Habitat Review Teams for 12 evolutionary significant units of West Coast salmon and steelhead. CHART Report. Portland, Oregon, National Marine Fisheries Service.

NMFS (2005). "Final assessment of NOAA Fisheries' critical habitat analytical review teams for 12 evolutionarily significant units of West Coast salmon and steelhead." available online at [http://www.nwr.noaa.gov/Salmon-Habitat/Critical-Habitat/2005 Biological - Teams-Report](http://www.nwr.noaa.gov/Salmon-Habitat/Critical-Habitat/2005%20Biological%20Teams-Report). accessed July 2008.

NMFS (2005). Final assessment of the National Marine Fisheries Service's critical habitat analytical review teams (CHARTs) for seven salmon and steelhead evolutionarily significant units (ESUs) in California. CHART report, Long Beach, California, Protected Resources, National Marine Fisheries Service: 23 + appendices.

NMFS (2005). Status review update for Puget Sound steelhead. Status Review. Seattle, Washington, National Marine Fisheries Service, Northwest Fisheries Science Center: 112.

NMFS (2006). Draft Columbia River estuary recovery plan module, Lower Columbia River Estuary Partnership under contract to NOAA Fisheries.

NMFS (2007). Biological Opinion for U.S. Department of Agriculture, Forest Service aerial application of eight long-term retardants on all forest service lands. Biological Opinion. Silver Spring, Maryland, U.S. Department of Commerce.

NMFS (2007). Request for endangered species act section 7 informal consultation on the EPA's reregistration and use of atrazine in the Chesapeake Bay watershed, September 1, 2006.

NMFS (2008). *Biological opinion for water supply, flood control operations, and channel maintenance conducted by the U.S. Army Corps of Engineers, the Sonoma County Water Agency, and the Mendocino County Russian River Flood Control and Water Conservation Improvement District in the Russian River watershed. Biological Opinion*. Santa Rosa, California, U.S. Department of Commerce: 367.

NMFS (2008). *Endangered Species Act - Section 7 Consultation Final Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation Implementation of the National Flood Insurance Program in the State of Washington Phase One Document - Puget Sound Region. Biological Opinion*, U.S. Department of Commerce: 238.

NMFS (2008). *Endangered Species Act section 7 consultation: biological opinion on Environmental Protection Agency registration of pesticides containing chlorpyrifos, diazinon, and malathion. Biological Opinion*. Silver Spring, Maryland, U.S. Department of Commerce: 482.

NMFS (2008). *Endangered Species Act Section 7(a)(2) Consultation Biological Opinion and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation on remand for operation of the Federal Columbia River Power System, 11 Bureau of Reclamation Projects in the Columbia Basin and ESA Section 10(a)(1)(A) permit for juvenile fish transportation program. Biological Opinion*, U.S. Department of Commerce.

NMFS (2008). *National Marine Fisheries Service Endangered Species Act Section 7 Consultation, Biological Opinion: Environmental Protection Agency Registration of Pesticides Containing Chlorpyrifos, Diazinon, and Malathion. Biological Opinion*. Washington, D.C., U.S. Department of Commerce.

NMFS (2008). *National Marine Fisheries Service endangered species act section 7 consultation: Environmental Protection Agency registration of pesticides containing chlorpyrifos, diazinon, and malathion. Biological Opinion*. Silver Spring, Maryland, U.S. Department of Commerce: 482.

NMFS (2008). *Proposed recovery plan for Lake Ozette sockeye salmon (*Oncorhynchus nerka*). Draft Recovery Plan*. Seattle, Washington, NMFS, Northwest Regional Office, Salmon Recovery Division.

NMFS (2009). *Biological Opinion and Conference Opinion on the long-term operations of the Central Valley Project and State Water Project. Biological Opinion*. Sacramento, California, U.S. Department of Commerce: 844.

NMFS (2009). Endangered Species Act – Section 7 Consultation Biological Opinion & Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation Nez Perce National Forest Noxious Weeds Programmatic, Middle Salmon River 17090207, Lower Salmon River 17060209, Main Salmon River 17060209, Little Salmon River 17060210, Selway River 17060302 and 17060302, Middle Fork Clearwater River 17060304, and South Fork Clearwater River 17060305, Idaho County, Idaho. Biological Opinion.

NMFS (2009). Endangered Species Act – Section 7 Consultation Biological Opinion & Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation:

Umatilla and Wallowa-Whitman National Forests Invasive Plant Treatments Project

Baker, Grant, Morrow, Umatilla, Union, Wallowa, and Wheeler Counties, Oregon

Asotin, Columbia, Garfield, and Walla Walla Counties, Washington; and Adams and Nez Perce

Counties, Idaho. Biological Opinion. Seattle, WA, U.S. Department of Commerce.

NMFS (2009). Endangered Species Act section 7 consultation: biological opinion on Environmental Protection Agency registration of pesticides containing carbaryl, carbofuran, and methomyl. Biological Opinion. Silver Spring, Maryland, U.S. Department of Commerce: 607.

NMFS (2009). NMFS Endangered Species Act Section 7 Consultation, Biological Opinion: Environmental Protection Agency Registration of Pesticides Containing Carbaryl, Carbofuran, and Methomyl. Biological Opinion. Washington, D.C., U.S. Department of Commerce.

NMFS (2010). NMFS Endangered Species Act Section 7 Consultation, Biological Opinion: Environmental Protection Agency registration of pesticides containing azinphos methyl, bensulide, dimethoate, disulfoton, ethoprop, fenamiphos, naled, methamidophos, methidathion, methyl parathion, phorate and phosmet. Biological Opinion. Washington, D.C., U.S. Department of Commerce.

*NMFS Salmon Recovery Division (2008). Proposed recovery plan for Lake Ozette sockeye salmon (*Oncorhynchus nerka*). Report. Seattle, Washington, U.S. Department of Commerce 378.*

NRC (1996). "Upstream: Salmon and society in the Pacific northwest." National Academy Press, Washington, D.C. Available:
http://books.nap.edu/openbook.php?record_id=4976&page=381 (February 2008).

NRC (2003). *Endangered and threatened fishes in the Klamath River Basin: Causes of decline and strategies for recovery.* Washington DC, National Research Council: 358.

NRC (2004). "Managing the Columbia River: Instream flows, water withdrawals, and salmon survival." National Academy Press, Washington D.C. Available:
http://www.nap.edu/catalog.php?record_id=10962#toc (February 2008).

NWPPC (1986). "Compilation of information on salmon and steelhead losses in the Columbia River basin." Report to the Northwest Power Planning Council, Portland, Oregon.
Available: <http://www.nwcouncil.org/library/1986/Compilation.htm> (February 2008)

O'Neill, S. M., G. M. Ylitalo, et al. (2006). "Regional patterns of persistent organic pollutants in five Pacific salmon species (*Oncorhynchus* spp) and their contributions to contaminant levels in northern and southern resident killer whales (*Orcinus orca*)."?????? ??????: ???

ODFW (2005). *Oregon plan for salmon and watersheds: Oregon coast coho assessment habitat*, Oregon Department of Fish and Wildlife: 61.

ODFW. (2007). "Oregon Department of Fish and Wildlife. Fish Hatchery Management Plan. Available online at: http://www.dfw.state.or.us/fish/nfcp/hatchery_mgmt.pdf ".

ODFW (2009). *Coho salmon inventory project: Coho abundance.*
<http://oregonstate.edu/dept/ODFW/spawn/cohoabund.htm>, ODFW, Corvallis, Oregon.
Updated August 19, 2009.

Olegario, A. O. (2006). *Over-wintering diet, growth, and prey availability to juvenile coho salmon (*Onchorhynchus kisutch*) in the West Fork Smith Riever, Oregon.* Department of Fisheries and Wildlife, Corvallis, OR, Oregon State University. Master Thesis: 57.

Opperman, J. J. and A. M. Merenlender (2004). "The effectiveness of riparian restoration for improving instream fish habitat in four hardwood-dominated California streams." North American Journal of Fisheries Management 24: 822-834.

Orr, J. C., V. J. Fabry, et al. (2005). "Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms." Nature 43(681-686).

Palmisano, J. F., R. H. Ellis, et al. (1993). "The impact of environmental and management factors on Washington's wild anadromous salmon and trout." Washington Forest Protection Association and Washington Department of Natural Resources, Olympia, Washington.

Parkinson, A. (1996). Biotransformation of xenobiotics. Toxicology: the basic science of poisons. Casarett and Doull. New York, NY, McGraw-Hill.

Paul, M. J. and J. L. Meyer (2001). "Streams in the urban landscape." Annual Review Ecological System 32: 333-365.

Pearcy, W. G. (1992). "Ocean ecology of North Pacific salmonids." University of Washington. University of Washington Press, Seattle, Washington: 179 p.

Pearcy, W. G. (1997). "What have we learned in the last decade? What are research priorities?" in Emmett, R.L. and M.H. Schiewe (editors). NOAA-NMFS-NWFSC TM-29: Estuarine and Ocean Survival of Northeastern Pacific Salmon: Proceedings of the workshop. U.S. Dep. Commer.: 5 p.

Perez, G. L., A. Torremorell, et al. (2007). "Effects of the herbicide roundup on freshwater microbial communities: A mesocosm study." Ecological Applications 17(8): 2310-2322.

Perry, R. W., M. J. Bradford, et al. (2003). "Effects of disturbance on contribution of energy sources to growth of juvenile chinook salmon (*Oncorhynchus tshawytscha*) in boreal streams." Canadian Journal of Fisheries and Aquatic Sciences 60(4): 390-400.

Pert, H. A. (1987). Winter food habits of coastal juvenile steelhead and coho salmon in Pudding Creek, Northern California. Wildland Resources Science. Humboldt, California, Humboldt State University. M.S.: 75.

Pesce, S., C. Fajon, et al. (2006). "Effects of the phenylurea herbicide diuron on natural riverine microbial communities in an experimental study." Aquatic Toxicology 78(4): 303-314.

Pess, G. R., D. R. Montgomery, et al. (2002). "Landscape characteristics, land use, and coho salmon (*Oncorhynchus kisutch*) abundance, Snohomish River, Wash., U.S.A." Canadian Journal of Fisheries and Aquatic Sciences 59: 613-623.

Peterson, J. L., P. C. Jepson, et al. (2001). "A test system to evaluate the susceptibility of Oregon, USA, native stream invertebrates to triclopyr and carbaryl." Environmental Toxicology and Chemistry 20(10): 2205-2214.

Peterson, N. P. (1982). "Immigration of juvenile coho salmon (*Oncorhynchus kisutch*) into riverine ponds." Canadian Journal of Fisheries and Aquatic Sciences 39: 1308-1310.

Phelps, S. R., S. A. Leider, et al. (1997). Genetic analysis of Washington steelhead: preliminary results incorporating 36 new collections from 1995 and 1996. Progress Report. W. D. o. F. a. Wildlife. Olympia, WA.

Pitcher, T. J. (1986). "Functions of shoaling in teleosts." pages 294-337 in Fisher, T.J. (editor). The behavior of teleost fishes. John Hopkins University Press, Baltimore, Maryland.

PNERC (2002). Willamette River Basin Planning Atlas: Trajectories of Environmental and Ecological Change. . D. Hulse, Gregory, S., & Baker, J. Eugene, OR, Institute for a Sustainable Environment, University of Oregon.

Pollock, M. M., G. R. Pess, et al. (2004). "The importance of beaver ponds to coho salmon production in the Stillaguamish River basin, Washington, USA." North American Journal of Fisheries Management 24(3): 749-760.

Powell, S., R. Neal, et al. (1996). Runoff and leaching of simazine and diuron used on highway rights-of-way. Environmental Hazards Assessment Program. EH 96-03. S. O. CALIFORNIA, E. P. Agency and D. o. P. Regulation.

Pratt, J. R., A. E. Melendez, et al. (1997). "Predicting the ecological effects of herbicides." Ecological Applications 7(4): 1117-1124.

PSAT (2004). "State of the Sound 2004." State of Washington, Office of the Governor, Olympia, Washington. Available:
http://www.psat.wa.gov/Publications/StateSound2004/141963_811.pdf (February 2008).

PSAT (2005). "State of the Sound 2004. Puget Sound Action Team, Office of the Governor, Olympia, WA."

PSAT (2007). "State of the Sound 2007." State of Washington, Office of the Governor, Publication Number PSAT 07-01, Olympia, Washington. Available:
http://www.psat.wa.gov/Publications/state_sound07/2007_stateofthesound.pdf
(February 2008).

Quinn, T. P. and N. P. Peterson (1996). "The influence of habitat complexity and fish size on over-winter survival and growth of individually marked juvenile coho salmon (*Oncorhynchus kisutch*) in Big Beef creek, Washington." Canadian Journal of Fisheries and Aquatic Sciences 53(7): 1555-1564.

Rakitsky, V. N., V. A. Koblyakov, et al. (2000). "Nongenotoxic (epigenetic) carcinogens: pesticides as an example. A critical review." Teratogenesis, Carcinogenesis, and Mutagenesis 20: 229-240.

Rawson, K., N. J. Sands, et al. (2009). Viability criteria for the Lake Ozette sockeye salmon evolutionarily significant unit. NOAA Technical Memorandum. Seattle, Washington, Department of Commerce: 38.

Reimers, P. E. (1973). "The length of residence of fall Chinook in Sixes River, Oregon." Research Report of the Fisheries Commission of Oregon 4(2): 43.

Reinhardt, U. G. (1999). "Predation risk breaks size-dependent dominance in juvenile coho salmon (*Oncorhynchus kisutch*) and provides growth opportunities for risk-prone individuals." Canadian Journal of Fisheries and Aquatic Sciences 56: 1206 - 1212.

Reisenbichler, R. R. (1997). Genetic factors contributing to declines of anadromous salmonids in the Pacific Northwest. Pacific salmon and their ecosystems: status and future options. D. J. Stouder, P. A. Bisson and R. J. Naiman. New York, Chapman and Hall: 223-244.

Relyea, R. A. (2009). "A cocktail of contaminants: how mixtures of pesticides at low concentrations affect aquatic communities." Oecologia 159(2): 363-376.

Rich, W. H. (1942). "The salmon runs of the Columbia River in 1938." Fishery Bulletin 50(37): 103-147.

Richardson, J. S., E. Taylor, et al. (2010). "Do riparian zones qualify as critical habitat for endangered freshwater fishes?" Canadian Journal of Fisheries and Aquatic Sciences 67: 1197-1204.

Richter, J. E. (2002). "Urban runoff water quality: a salmonid's perspective." Available online at http://www.4sos.org/wssupport/ws_rest/assess.asp accessed on 7/7/08: 5 p.

Rieman, B. E., D. L. Myers, et al. (1994). "Use of otolith microchemistry to discriminate *Oncorhynchus nerka* of resident and anadromous origin." Canadian Journal of Fisheries and Aquatic Sciences 51(1): 68-77.

Riggs, L. A. (1990). "Principles for genetic conservation and production quality: results of a scientific and technical clarification and revision." Unpublished report prepared for the Northwest Power Planning Council, prepared by Genetic Resource Consultants.

Rinne, J. N. (2004). "Forests, fish and fire: relationships and management implications for fishes in the southwestern USA." pages 151-156 in G.J. Scrimgeour, G.Eisler, B. McCulloch, U. Silins, and M. Monita (editors). Forest Land - Fish Conference II - Ecosystem Stewardship through Collaboration. Proceedings of the Forest-Land-Fish Conference II, April 26-28, 2004. Edmonton, Alberta, Canada.

Ritchie, A. C. (2005). Lake Ozette shoreline Morphology: 1953-2003. Progress Report to Olympic National Park. Port Angeles, Washington: 9.

Roby, D. D., K. Colis, et al. (2008). Research, monitoring, and evaluation of avian predation on salmonid smolts in the lower and Mid-Columbia River. Draft 2007 Season Summary. Corvallis, OR, USGS - Oregon Cooperative Fish and Wildlife Research Unit Department of Fisheries and Wildlife: 85.

Roby, D. D., K. Collis, et al. (2006). Research, monitoring, and evaluation of avian predation on salmonid smolts in the lower and mid-Columbia River. Draft Season Summary submitted to the Bonneville Power Administration and the U.S. Army Corps of Engineers. Portland, Oregon: 130 p.

Rogers, C. A. and D. L. Stalling (1972). "Dynamics of an ester of 2,4-D in organs of three species of fish." Weed Science 20: 101-105.

Rohr, J. R. and P. W. Crumrine (2005). "Effects of an herbicide and an insecticide on pond community structure and processes." Ecological Applications 15(4): 1135-1147.

Rohr, J. R., A. A. Elskus, et al. (2004). "Multiple stressors and salamanders: Effects of an herbicide, food limitation, and hydroperiod." Ecological Applications 14(4): 1028-1040.

- Rondorf, D. W., G. A. Gray, et al. (1990). "Feeding ecology of subyearling Chinook salmon in riverine and reservoir habitats of the Columbia River." Transaction of American Fisheries Society 119: 16-24.
- Roni, P. (2002). "Habitat use by fishes and pacific giant salamanders in small western Oregon and Washington streams." Transactions of the American Fisheries Society 131(4): 743-761.
- Rosemond, A. D., P. J. Mulholland, et al. (2000). "Seasonally shifting limitation of stream periphyton: response of algal populations and assemblage biomass and productivity to variation in light, nutrients, and herbivores." Canadian Journal of Fisheries and Aquatic Sciences 57(1): 66-75.
- Rosenfeld, J., M. Porter, et al. (2000). "Habitat factors affecting the abundance and distribution of juvenile cutthroat trout (*Oncorhynchus clarki*) and coho salmon (*Oncorhynchus kisutch*)." Canadian Journal of Fisheries and Aquatic Sciences 57: 766 - 774.
- Rosenfeld, J. S., T. Leiter, et al. (2005). "Food abundance and fish density alters habitat selection, growth, and habitat suitability curves for juvenile coho salmon (*Oncorhynchus kisutch*)." Canadian Journal of Fisheries and Aquatic Sciences 62: 1691-1701.
- Rosetta, T. and D. Borys (1996). "Identification of sources of pollutants to the lower Columbia River basin. Draft report. Prepared for the Lower Columbia River Bi-State Program." Oregon Department of Environmental Quality. Available:
<http://www.lcrep.org/pdfs/58.%2002750.pdf> (February 2008).
- Ruckelshaus, M. H., K. P. Currens, et al. (2006). Independent populations of Chinook salmon in Puget Sound. NOAA Technical Memorandum, U.S. Department of Commerce. NMFS-NWFSC-78: 125.
- Ruckelshaus, M. H. and M. M. McClure (2007). "Sound Science: synthesizing ecological and socioeconomic information about the Puget Sound ecosystem." Prepared in Cooperation with the Sound Science collaborative team. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle, Washington. Available:
http://www.nwfsc.noaa.gov/research/shared/sound_science/index.cfm (February 2008).

- Ruggerone, G. T. and F. A. Goetz (2004). "Survival of Puget Sound Chinook salmon (*Oncorhynchus tshawytscha*) in response to climate-induced competition with pink salmon (*Oncorhynchus gorbuscha*)."
Canadian Journal of Fisheries and Aquatic Sciences 61: 1756-1770.
- Ryan, B. A., M. Carper, et al. (2006). "Alternative barging strategies to improve survival of transported juvenile salmonids." Draft report sent to U.S. Army Corps of Engineers, Walla Walla, Washington.
- Salo, E. O. (1991). "Life history of chum salmon " in Pacific Salmon Life Histories. C.Groot and L.Margolis (editors). University of British Columbia Press, Vancouver, British Columbia, Canada
- Sandercock, F. K. (1991). Life history of coho salmon. Pacific Salmon Life Histories. C. Groot and L. Margolis. Vancouver, BC, University of British Columbia: 395 - 446.
- Sanderson, B. L., K. A. Barnas, et al. (2009). "Nonindigenous species of the Pacific Northwest: An overlooked risk to endangered salmon?" Bioscience 59(3): 245-256.
- Sands, N. J., K. Rawson, et al. (2007). Dawgz'n the hood: The Hood Canal summer chum salmon ESU. Draft TRT Report. Seattle, Washington, Puget Sound Technical Recovery Team: 64.
- Sazaki, S. (1966). Distribution and food habits of king salmon, *Oncorhynchus tshawytscha*, and steelhead rainbow trout, *Salmo gairdnerii*, in the Sacramento-San Joaquin delta. Ecological studies of the Sacramento-San Joaquin Delta. J. L. Turner and D. W. Kelly. 136.
- Schafer, R. B., T. Caquet, et al. (2007). "Effects of pesticides on community structure and ecosystem functions in agricultural streams of three biogeographical regions in Europe." Science of the Total Environment 382(2-3): 272-285.
- Schroder, S. L. (1977). "Assessment of production of chum salmon fry from the Big Beef Creek spawning channel." Completion report. Univ. Wash.Fish. Res. Inst. FRI-UW 7718: 77 p.
- Schroder, S. L., K. V. Koski, et al. (1974). "Big Beef Creek studies." in: Research in fisheries 1973. Univ. Wash. Coll. Fish. Contrib. 390: p. 26-27.
- Seaber, P. R., F. P. Kapinos, et al. (1987). Hydrologic Unit Maps, U. S. Geologic Survey.

- Sergeant, C. J. and D. A. Beauchamp (2006). "Effects of physical habitat ontogeny on lentic habitat preference of juvenile Chinook salmon." Transaction of American Fisheries Society 135: 1191-1204.
- Shapovalov, L. and A. C. Taft (1954). "The life histories of the steelhead rainbow trout (*Salmo gairdneri gairdneri*) and silver salmon (*Oncorhynchus kisutch*) with special reference to Waddell Creek, California, and recommendations regarding their management." California Department of Fish and Game Bulletin 98: 375.
- Sharma, R. and R. Hilborn (2001). "Empirical relationships between watershed characteristics and coho salmon (*Oncorhynchus kisutch*) smolt abundance in 14 western Washington streams." Canadian Journal of Fisheries and Aquatic Sciences 58: 1453-1463.
- Shelley, L. K., S. K. Balfry, et al. (2009). "Immunotoxicological effects of a sub-chronic exposure to selected current-use pesticides in rainbow trout (*Oncorhynchus mykiss*)."
Aquatic Toxicology 92: 95-103.
- Shirvell, C. S. (1990). "Role of instream rootwads as juvenile coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*O. mukiss*) cover habitat under varying streamflows." Canadian Journal of Fisheries and Aquatic Sciences 47: 852-861.
- Slater, D. W. (1963). Winter-run Chinook salmon in the Sacramento River, California, with notes on water temperature requirements at spawning. Special Science Report Fisheries, U.S. Fish and Wildlife Service. 461: 9.
- Smith, C. J. (2005). *Salmon habitat limiting factors in Washington State*. Olympia, Washington, Washington State Conservation Commission: 222.
- Smith, S. G., W. D. Muir, et al. (2003). "Influence of river conditions on survival and travel time of Snake River subyearling fall Chinook salmon." North American Journal of Fisheries Management 23: 939-961.
- Sogard, S., T. H. Williams, et al. (2009). "Seasonal patterns of abundance, growth, and site fidelity of juvenile steelhead in a small coastal California stream." Transaction of American Fisheries Society 138(3): 549-563.
- Sommer, T. R., W. C. Harrell, et al. (2005). "Habitat use and stranding risk of juvenile Chinook salmon on a seasonal floodplain." North American Journal of Fisheries Management 25: 1493-1504.

Sommer, T. R., M. L. Nobriga, et al. (2001). "Floodplain rearing of juvenile chinook salmon: evidence of enhanced growth and survival." Canadian Journal of Fisheries and Aquatic Sciences 58: 325-333.

Sonoma County Water Agency (SCWA) (2002). Documenting biodiversity of coastal salmon (*Oncorhynchus* spp.) in Northern California. Final Report. Bodega Bay, California, Bodega Marine Laboratory, University of California at Davis: 81.

Sonoma County Water Agency (SCWA) (2008). Chinook salmon in the Russian River, http://www.scwa.ca.gov/environment/natural_resources/chinook_salmon.php.

Spence, B. C., E. P. Bjorkstedt, et al. (2008). "A framework for assessing the viability of threatened and endangered salmon and steelhead in the north-central California Coast Recovery Domain." NOAA Technical Memorandum NMFS NOAA-TM-NMFS-SWFSC-423: 173.

Spence, B. C., G. A. Lomnický, et al. (1996). "An ecosystem approach to salmonid conservation." Man Tech Environmental Research Services Corps, TR-4501-96-6057, Corvallis, Oregon.

Spurlock, F., C. Garretson, et al. (1997). Runoff from citrus orchard middles: comparison of three herbicides and effect of organosilicon surfactant. California Department of Pesticide Regulation. Environmental Hazards Assessment Program. EH 97-02.

Stanford, J. A., F. R. Hauer, et al. (2005). "Columbia River basin." Pages 591-653 in A. C. Benke and C. E. Cushing, editors. Rivers of North America. Elsevier Academic Press, Burlington, Massachusetts. Available:
http://books.google.com/books?id=faOUIwkiYFIC&pg=RA3-PA541&lpg=RA3-PA541&dq=pacific+coast+rivers+of+the+coterminous+united+states&source=web&ots=pMpyECFaA&sig=FkGrIiwgkfDyHxXCWXRaK_XSvU#PPRI,M1 (February 2008).

Stearns, S. C. (1982). "The evolution of life histories." New York, New York, Oxford University Press.

Stein, R. A., P. E. Reimers, et al. (1972). "Social interaction between juvenile coho (*Oncorhynchus kisutch*) and fall Chinook salmon (*O. tshawytscha*) in Six Rivers, Oregon." Journal Fisheries Research Board of Canada 29: 1748-1737.

Steward, C. R. and T. C. Bjornn (1990). "Supplementation of salmon and steelhead stocks with hatchery fish: a synthesis of published literature." Bonneville Power Administration, Technical Report 90-1, Portland, Oregon.

- Swales, S., F. Caron, et al. (1988). "Overwintering habitats of coho salmon (*Oncorhynchus kisutch*) and other juvenile salmonids in the Keogh River system, British Columbia." Canadian Journal of Zoology 66: 254-261.
- Swift III, C. H. (1979). Preferred stream discharges for salmon spawning and rearing in Washington. USGS Open-File Report 77-422 USGS: 51p.
- Sytsma, M. D., J. R. Cordell, et al. (2004). Lower Columbia River aquatic nonindigenous species survey 2001-2004. Final Technical Report. October. Prepared for the US Coast Guard and the US Fish and Wildlife Service.
- Tabor, R. A., H. A. Gearns, et al. (2006). Nearshore habitat use by Chinook salmon in lentic systems of the Lake Washington basin. Annual Report. Lacey, Washington, U.S. Fish and Wildlife Service: 94.
- Teel, D. J., C. Baker, et al. (2009). "Genetic stock composition of subyearling Chinook salmon in seasonal floodplains wetlands of the Lower Willamette River, Oregon." Transaction of American Fisheries Society 138: 211-217.
- Teske, M. E. (2001). A user's guide for AgDrift 2.04: A tiered approach for the assessment of spray drift of pesticides, Spray Drift Task Force: 128.
- Thompson, D. G., B. Stanik, et al. (1991). "Fate of triclopyr ester (RELEASE) in a boreal forest stream." Environmental Toxicology and Chemistry 10: 619 - 632.
- Thorpe, J. E. (1994). "Salmonid fishes and the estuarine environment." Estuaries 17(1A): 76 - 93.
- Tierney, K. B., D. H. Baldwin, et al. (2010). "Olfactory toxicity in fishes." Aquatic Toxicology 96(2010): 2-26.
- Tierney, K. B., P. S. Ross, et al. (2006). "Changes in juvenile coho salmon electro-olfactogram during and after short-term exposure to current-use pesticides." Environmental Toxicology and Chemistry 25(10): 2809-2817.
- Tierney, K. B., P. S. Ross, et al. (2007). "Linuron and carbaryl differentially impair baseline amino acid and bile salt olfactory responses in three salmonids." Toxicology 231(2-3): 175-187.

- Tierney, K. B., J. L. Sampson, et al. (2008). "Salmon olfaction is impaired by an environmentally realistic pesticide mixture." Environmental Science and Technology 42(13): 4996-5001.
- Tiffan, K. F., L. O. Clark, et al. (2006). "Variables influencing the presence of subyearlings fall Chinook salmon in shoreline habitats of the Hanford Reach, Columbia River." North American Journal of Fisheries Management 26: 351-360.
- Tiffan, K. F., D. W. Rondorf, et al. (2001). Post-release attributes and survival of hatchery and natural fall Chinook salmon in the Snake River annual report 1999. Report to Bonneville Power Administration, USGS, Cook, Washington; U.S. FWS, Ahshaka, Idaho: 140.
- Tillman, R. W., M. R. Siegel, et al. (1973). "Mechanism of action and fate of the fungicide chlorothalonil (2,4,5,6-tetrachloroisophthalonitrile) in biological systems : I. Reactions with cells and subcellular components of *Saccharomyces pastorianus*." Pesticide Biochemistry and Physiology 3(2): 160-167.
- Tixier, C., M. Sancelme, et al. (2001). "Degradation products of a phenylurea herbicide, diuron: synthesis, ecotoxicity and biotransformation." Environmental Toxicology and Chemistry 20(7): 1381-1389.
- Torres-Ruiz, M., J. D. Wehr, et al. (2007). "Trophic relations in a stream food web: importance of fatty acids for macroinvertebrate consumers." Journal of the North American Benthological Society 26(3): 509-522.
- Tschaplinski, P. J. and G. F. Hartman (1983). "Winter distribution of juvenile coho salmon (*Oncorhynchus kisutch*) before and after logging in Carnation Creek, British Columbia, and some implications for overwinter survival." Canadian Journal of Fisheries and Aquatic Sciences 40: 452-461.
- USCB (2005). *Interim State Population Projections, 2005*. P. Division, U.S. Census Bureau.
- USCB (2009). *Cumulative estimates of population change for metropolitan statistical areas and rankings: April 1, 2000 to July 1, 2008 (Table 7)*.
- USFS (2000). *Fire recharges native fisheries*, U.S. Department of Agriculture, Forest Service - Northern Region: 3.
- USFWS (2008). *Informal consultation on the effects of atrazine re-registration on the endangered Alabama sturgeon and the endangered dwarf wedge-mussel*.

- USFWS and Reclamation (2007). *A compilation and analysis of anadromous fish monitoring data from the Central Valley of California 1992-2006. Comprehensive Assessment and Monitoring Program (CAMP). Sacramento, California, U.S. Department of the Interior: 99.*
- Van den Brink, P. J., N. Blake, et al. (2006). "Predictive value of species sensitivity distributions for effects of herbicides in freshwater ecosystems." Human and Ecological Risk Assessment 12(4): 645-674.
- Van den Brink, P. J., S. J. H. Crum, et al. (2009). "Effects of a herbicide-insecticide mixture in freshwater microcosms: Risk assessment and ecological effect chain." Environmental Pollution 157(1): 237-249.
- Varanasi, U., J. Stein, et al. (1989). "Biotransformation and disposition of PAH in fish." Metabolism of polycyclic aromatic hydrocarbons in the aquatic environment, Varansi U. (ed). CRC Press: Boca Raton, Florida: 93-149.
- Varanasi, U., J. Stein, et al. (1992). "Chlorinated and aromatic hydrocarbons in bottom sediments, fish and marine mammals in U.S. coastal waters; laboratory and field studies of metabolism and accumulation." Persistent Pollutants in Marine Ecosystems. C.H. Walker and D.R. Livingstone (eds). Permagon Press: New York: 83-115.
- Walker, R. L. and J. S. Foott (1993). "Disease survey of Klamath River salmonid smolt populations, 1992." U.S. Fish and Wildlife Service, California-Nevada Fish Health Center, Anderson, California. [150 Kb] at KRIS Website: 46 p.
- Wallace, J. B., S. L. Eggert, et al. (1999). "Effects of resource limitation on a detrital-based ecosystem." Ecological Monographs 69(4): 409-442.
- Washington Department of Fish and Wildlife (WDFW) (1993). *1992 Washington state salmon and steelhead stock inventory (SASSI) Olympia, Washington, WDFW and Western Washington Treaty Indian Tribes.*
- Washington Department of Fish and Wildlife (WDFW) (2002). *Salmon stock inventory (SaSI) 2002.* <http://wdfw.wa.gov/fish/sasi/>.
- Washington Department of Fish and Wildlife (WDFW). (2005). "Washington Department of Fish and Wildlife.Methow River Steelhead Hatchery and Genetic Management Plan. Available online at: <http://wdfw.wa.gov/hat/hgmp/>."

- Washington Department of Fish and Wildlife (WDFW) (2008). *Oncorhynchus mykiss*: Assessment of Washington State's anadromous populations and programs Preliminary Draft. J. B. Scott, Jr. and W. T. Gill. Olympia, Washington, WDFW: 424.
- Washington Department of Fish and Wildlife (WDFW) (2009). *SalmonScape online database* <http://fortress.wa.gov/dfw/gispublic/apps/salmonscape/default.htm>, Washington Department of Fish and Wildlife.
- Weitkamp, L. A., T. C. Wainwright, et al. (1995). "Status review of coho salmon from Washington, Oregon, and California." U.S. Dep. of Commer., NOAA Tech. Memo. NMFS-NWFSC 24: 258 p.
- Welsh, H. H., G. R. Hodgson, et al. (2001). "Distribution of juvenile coho salmon (*Oncorhynchus kistuch*) in relation to water temperature in tributaries of a Northern California watershed determining management thresholds for an impaired cold-water adapted fauna." in review for publication in the North American Journal of Fisheries Management(21): 464-470.
- Wendt-Rasch, L., P. Pirzadeh, et al. (2003). "Effects of metsulfuron methyl and cypermethrin exposure on freshwater model ecosystems." Aquatic Toxicology 63(3): 243-256.
- Wentz, D. A., B. A. Bonn, et al. (1998). Water quality in the Willamette Basin, Oregon 1991-95. USGS circular 1161.
- Wheeler, A. P., P. L. Angermeier, et al. (2005). "Impacts of new highways and subsequent landscape urbanization on stream habitat and biota." Fisheries Science 13: 141-164.
- Widenfalk, A., J. M. Svensson, et al. (2004). "Effects of the pesticides captan, deltamethrin, isoproturon, and pirimicarb on the microbial community of a freshwater sediment." Environmental Toxicology and Chemistry 23(8): 1920-1927.
- Wilcove, D. S., D. Rothstein, et al. (1998). "Quantifying threats to imperiled species in the United States: Assessing the relative importance of habitat destruction, alien species, pollution, overexploitation, and disease." Bioscience 48(8): 607-615.
- Williams, T. H., E. P. Bjorkstedt, et al. (2006). Historical population structure of coho salmon in the southern Oregon/northern California coasts evolutionarily significant unit. NOAA Technical Memorandum. Santa Cruz, CA, U.S. Department of Commerce. NMFS-SWFSC- 390: 71.

- Williamson, A. K., M. D. Munn, et al. (1998). *Water Quality in the Central Columbia Plateau, Washington and Idaho, 1992-95*: U.S. Geological Survey Circular 1144, on line at: <http://water.usgs.gov/pubs/circ114>.
- WSDA (2010). *2,4-D use summary*. Washington State Department of Agriculture.
- WSDA (2010). *Captan use summary*. Washington State Department of Agriculture.
- WSDA (2010). *Chlorothalonil use summary*. Washington State Department of Agriculture.
- WSDA (2010). *Diuron use summary*. Washington State Department of Agriculture.
- WSDA (2010). *Linuron use summary*. Washington State Department of Agriculture.
- WSDE. (2006). "Washington State Department of Ecology - Water quality standards for surfaces waters of the state of Washington chapter 173-201A WAC. Publication Number 06-10-091. Available at: <http://www.ecy.wa.gov/pubs/0610091.pdf>."
- Wu, L. (2000). "TMDLs: a significant change in water quality regulation enforcement." accessed at <http://esce.ucr.edu/soilwater/winter2000.htm> on 7/7/08: 5 p.
- Wunderlich, R. C., B. D. Winter, et al. (1994). "Restoration of the Elwha River ecosystem." Fisheries 19((8)): 11-19.
- Wydoski, R. S. and R. R. Whitney (1979). "Inland fishes of Washington." University of Washington. University of Washington Press, Seattle, Washington.
- Yates, D., H. Galbraith, et al. (2008). "Climate warming, water storage, and Chinook salmon in California's Sacramento Valley." Climate Change 91(3-4): 335-350.
- Yoshiyama, R. M., E. R. Gerstung, et al. (1998). "Historical abundance and decline of Chinook salmon in the Central Valley region of California." North American Journal of Fisheries Management 18: 487-521.
- Zedonis, P. A. (1992). *The biology of steelhead (Onchorynchus mykiss) in the Mattole River estuary/lagoon, California*. Arcata, CA, Humboldt State University. Masters Thesis: 77.

Appendix 1 – Population Modeling

Introduction

To assess the potential for adverse impacts of the anticholinesterase insecticides on Pacific salmon populations, a model was developed that explicitly links impairments in the biochemistry, behavior, prey availability and somatic growth of individual salmon to the productivity of salmon populations. More specifically, the model connects known effects of the pesticides on salmon physiology and behavior with community-level effects on salmon prey to estimate population-level effects on salmon. The model used here is an extension of one developed for investigating the direct effects of pesticides on the biochemistry, behavior and growth of ocean-type Chinook salmon (Baldwin et al., 2009).

In the freshwater portion of their life, Pacific salmon may be exposed to insecticides that act by inhibiting acetylcholinesterase (AChE). Acetylcholinesterase is a crucial enzyme in the proper functioning of cholinergic synapses in the central and peripheral nervous systems of vertebrates and invertebrates. Of consequence to salmon, anticholinesterase insecticides have been shown to interfere with salmon swimming behavior (Beauvais et al. 2000, Brewer et al. 2001, Sandahl et al. 2005), feeding behavior (Sandahl et al. 2005), foraging behavior (Morgan and Kiceniuk 1990), homing behavior (Scholz et al. 2000), antipredator behaviors (Scholz et al. 2000) and reproductive physiology (Moore and Waring 1996, Waring and Moore 1997, Scholz et al. 2000).

Anticholinesterase insecticides have also been found to reduce benthic densities of aquatic invertebrates and alter the composition of aquatic communities (Liess and Schulz 1999, Schulz and Liess 1999, Schulz et al. 2002, Fleeger et al. 2003, Schulz 2004, Chang et al. 2005, Relyea 2005). Spray drift and runoff from agricultural and urban areas can expose aquatic invertebrates to relatively low concentrations of insecticides for as little as minutes or hours, but populations of many taxa can take months or even years to recover to pre-exposure or reference densities (Wallace et al. 1991, Liess and Schulz 1999, Anderson et al. 2003, Stark et al. 2004). For

example, when an aquatic macroinvertebrate community in a German stream was exposed to runoff containing parathion (an acetylcholinesterase inhibitor) and fenvalerate (another commonly used insecticide), eight of eleven abundant species disappeared and the remaining three were reduced in abundance (Liess and Schulz 1999). Long-term changes in invertebrate densities and community composition likely result in reductions in salmon prey availability. Therefore, in addition to the direct impacts that acetylcholinesterase inhibitors have on salmon, there may also be, independently, significant indirect effects to salmon via their prey (Peterson et al. 2001a). Wild juvenile salmon feed primarily on invertebrates in the water column and those trapped on the water's surface, actively selecting the largest items available (Healey 1991, Quinn 2005). Salmon are often found to be food limited (Quinn 2005), suggesting that a reduction in prey number or size due to insecticide exposure may further stress salmon. For example, Davies and Cook (1993) found that several months following a spray drift event, benthic and drift densities were still reduced in exposed stream reaches. Consequently, brown trout in the exposed reaches fed less and grew at a slower rate compared to those in unexposed stream reaches (Davies and Cook 1993). Although the insecticide in their study was cypermethrin (a pyrethroid), similar reductions in macroinvertebrate density and recovery times have been found in studies with acetylcholinesterase inhibitors (Liess and Schulz 1999, Schulz et al. 2002), suggesting indirect effects to salmon via prey availability may be similar.

One likely biological consequence of reduced swimming, feeding, foraging, and prey availability is a reduction in food uptake and, subsequently, a reduction in somatic growth of exposed fish. Juvenile growth is a critical determinant of freshwater and marine survival for Chinook salmon (Higgs et al. 1995). Reductions in the somatic growth rate of salmon fry and smolts are believed to result in increased size-dependent mortality (Healey 1982, West and Larkin 1987, Zabel and Achord 2004). Zabel and Achord (2004) observed size-dependent survival for juvenile salmon during the freshwater phase of their outmigration. Mortality is also higher among smaller and slower growing salmon because they are more susceptible to predation during their first winter (Healey 1982, Holtby et al. 1990, Beamish and Mahnken 2001). These studies suggest that factors affecting the organism and reducing somatic growth, such as anticholinesterase

insecticide exposure, could result in decreased first-year survival and, thus, reduce population productivity.

Changes to the size of juvenile salmon from exposure to anticholinesterase pesticides were linked to salmon population demographics. We used size-dependent survival of juveniles during a period of their first year of life. We did this by constructing and analyzing general life-history matrix models for coho salmon (*Oncorhynchus kisutch*), sockeye salmon (*O. nerka*) and ocean-type and stream-type Chinook salmon (*O. tshawytscha*). A steelhead (*O. mykiss*) life-history model was not constructed due to the lack of demographic information relating to the proportions of resident and anadromous individuals, the freshwater residence time of steelhead, and rates of repeated spawning. The basic salmonid life history modeled consisted of hatching and rearing in freshwater, smoltification in estuaries, migration to the ocean, maturation at sea, and returning to the natal freshwater stream for spawning followed shortly by death. Differences between the modeled strategies are lifespan of the female, time to reproductive maturity, and the number and relative contribution of the reproductive age classes (Figure 1). The coho females we modeled reach reproductive maturity at age 3 and provide all of the reproductive contribution. Sockeye females reach maturity at age 4 or 5, but the majority of reproductive contributions are provided by age 4 females. Chinook females can mature at age 3, 4 or 5, with the majority of the reproductive contribution from ages 4 and 5. The primary difference between the ocean-type and stream-type Chinook is the juvenile freshwater residence with ocean-type juveniles migrating to the ocean as subyearlings and stream-type overwintering in freshwater and migrating to the ocean as yearlings. The models depicted general populations representing each life-history strategy and were constructed based upon literature data described below. Specific populations were not modeled due to the difficulty in finding sufficient demographic and reproductive data for a single population. Our modeled populations were not designed to represent particular salmon population segments, and they did not incorporate potentially influential life history information that may vary among populations. This includes, for example, density-dependent effects on juvenile growth and survival as well as the effects of adult migration (*i.e.*, straying) on adult spawner abundance. Our results using a more simplified and generic model for show how improving water quality conditions by reducing the pesticide load could potentially impact

population viability and rate of recovery. This should allow resource managers to consider pesticides at the same biological scale as physical and biological stressors when prioritizing habitat restoration activities.

A separate acute toxicity model was constructed that estimated the population-level impacts of juvenile mortality resulting from exposure to lethal concentrations of the 12 pesticides. These models excluded sublethal and indirect effects of the pesticide exposures and focused on the population-level outcomes resulting from an annual exposure of juveniles to a pesticide. The lethal impact was implemented as a change in first year survival for each of the salmon life-history strategies.

The overall model endpoint used to assess population-level impacts for both the growth and acute lethality models was the percent change in the intrinsic population growth rate (λ) resulting from the pesticide exposure. Change in λ is an accepted population parameter often used in evaluating population productivity, status, and viability. The National Marine Fisheries Service uses changes in λ when estimating the status of species, conducting risk and viability assessments, developing Endangered Species Recovery Plans, composing Biological Opinions, and communicating with other federal, state and local agencies (McClure et al. 2003). While values of $\lambda < 1.0$ indicate a declining population, negative changes in λ greater than the natural variability for the population indicate a loss of productivity. This can be a cause for concern since the decline could make a population more susceptible to dropping below 1.0 due to impacts from multiple stressors.

The following models were developed to serve as a means to assess the potential effects on ESA-listed salmon populations from exposure to AChE inhibiting pesticides, including n-methyl carbamates and organophosphorus insecticides. The growth model focuses on the impacts to prey abundance and a salmon's ability to feed which are integrated into reductions in juvenile growth. Assessing the results from different pesticide exposure scenarios relative to a control (i.e. unexposed) scenario can indicate the potential for sublethal pesticide exposures to lead to changes in the somatic growth and survival of individual subyearling salmon. Consequently,

subsequent changes in salmon population dynamics as indicated by percent change in a population's intrinsic rate of increase assists us in forecasting the potential population-level impacts to listed populations. Also, the model helps us understand the potential influence of life-history strategies that might explain differential results within the species modeled.

Methods

The model consists of two parts, an organismal portion and a population portion. The organismal portion of the model links AChE inhibition and reduced prey abundance due to insecticide exposure to potential reductions in the growth of individual fish. The population portion of the model links the sizes of individual subyearling salmon to their survival and the subsequent growth of the population. Models were constructed using MATLAB 7.7.0 (R2008b) (The MathWorks, Inc. Natick, MA).

Organismal Model

For the organismal model a relationship between AChE activity and somatic growth of salmonid fingerlings was developed using a series of relationships between pesticide exposure, AChE activity, feeding behavior, food uptake, and somatic growth rate (Figures 2-4). The model incorporates empirical data when available. Since growth and toxicity data are limited, extrapolation from one salmon species to the others was done with the assumption that the salmon stocks would exhibit similar physiological and toxicological responses. Sigmoidal dose-response relationships based upon the AChE inhibition EC50 values and their slopes are used to determine the level of AChE activity (Figure 2A, 2B, 2C) from the exposure concentration of each pesticide exposure or pulse.

A linear relationship based on empirical data related AChE activity to feeding behavior (Sandahl et al. 2005, Figure 2D). Feeding behavior was then assumed to be directly proportional to food uptake, defined as potential ration (Figure 2E). The potential ration expresses the amount of food the organism can consume when prey abundance is not limiting. Potential ration over time (Figure 2F) depicts how the food intake of individual fish changes in response to the behavioral effects of the pesticide exposure over the modeled growth period. Potential ration is equal to final ration if no effects on prey abundance are incorporated (Figure 4). If effects of pesticide exposure on prey abundance are incorporated, final ration is the product of potential ration (relating to the fish's ability to capture prey, Figure 2) and the relative abundance of prey available following exposure (Figure 3). Next, additional empirical data (e.g. Weatherley and

Gill 1995) defined the relationship between final ration and somatic growth rate (Figure 4C). While the empirical relationship is more complex (e.g. somatic growth rate plateaus at rations above maximum feeding), a linear model was considered sufficient for the overall purpose of this model. Finally, the model combines these linear models relating AChE activity to feeding behavior, feeding behavior to potential ration, and final ration to somatic growth rate to produce a linear relationship between AChE activity and somatic growth rate (Figure 4D). One important assumption of the model is that the relationships are stable, i.e. do not change with time. The relationships would need to be modified to incorporate time as a variable if, for example, fish are shown to compensate over time for reduced AChE activity to improve their feeding behavior and increase food uptake.

Selection of aquatic invertebrate toxicity values to represent salmonid prey items:

The model requires an EC_{50} for each pesticide (defined as a 50% reduction in the biomass of salmonid prey items) and a corresponding slope. The term “ EC_{50} ” will be used in this section to describe short-term survival data for aquatic invertebrates (death and immobility). To determine what levels of the OPs reduce aquatic invertebrate numbers, we reviewed the available field and laboratory studies. We found a wide spectrum of available data for the 10 a.i.s (2 a.i.s were not modeled: fenamiphos and bensulide). We did not locate a field study that measured aquatic community response to a range of concentrations of these pesticides. Therefore, we did not select concentration data from field experiments as we did in NMFS’ 2008 Opinion on the registration of chlorpyrifos, diazinon, and malathion {Kuivila, 1995 #2083}. Due to the scarcity of data for many of the a.i.s, we did not develop probability plots. Instead, we selected the lowest available survival EC_{50} for *D. magna* for each a.i. to represent the salmonid prey community EC_{50} because *D. magna* data were available for all a.i.s (Table 5).

The models allow exposures that can include multiple AChE-inhibiting pesticides over various time pulses. Sigmoidal dose-response relationships, at steady-state, between each single pesticide exposure and 1) AChE activity and 2) relative prey abundance are modeled using specific EC_{50} s and EC_{50} s and slopes (Figure 2B and 3B). The timecourse for each exposure was built into the model as a pulse with a defined start and end during which the exposure remained constant

(Figure 2A and 3A). The timecourse for AChE activity, on the other hand, was modeled using two single-order exponential functions, one for the time required for the exposure to reach full effect and the other for time required for complete recovery following the end of the exposure (time-to-effect_{AChE activity} and time-to-recovery_{AChE activity}, respectively; Figure 2C). The apparent activity level was back-calculated to result in a relative concentration (concentration/ AChE inhibition EC₅₀) for each day of the growth period for each pulse. The relative concentration for each day was summed across all the pulses to result in a total apparent concentration for each day. The sigmoid slope used in the calculation of AChE activity using the apparent concentration was the arithmetic mean of the sigmoid slopes for each pesticide present on each day. The timecourse for relative prey abundance was modeled incorporating a one day spike in prey drift relative to the toxicity and available prey base followed by a drop in abundance due to the toxic impacts (Figure 3C). Recovery is assumed to be due to a constant influx of invertebrates from connected habitats (aquatic and terrestrial) that are not exposed to the pesticide. Incoming organisms are subject to toxicity if pesticides are still present and this alters the rate of recovery during exposures. Incorporating dynamic effects and recovery variables allows the model to simulate differences in the pharmacokinetics (e.g. the rates of uptake from the environment and of detoxification) of various pesticides and simulate differences in invertebrate community response and recovery rates (see below).

The relationship between final ration and somatic growth rate (Figure 4C) produces a relationship representing somatic growth rate over time (Figure 4D), which is then used to model individual growth rate and size over time. The growth models were run for 1000 individual fish, with initial weight selected from a normal distribution with a mean of 1.0 g and standard deviation of 0.1 g. The size of 1.0 g was chosen to represent subyearling size in the spring prior to the onset of pesticide application. For each iteration of the model (one day for the organismal model), the somatic growth rate is calculated for each fish by selecting the parameter values from normal distributions with specified means and standard deviations (Table 1). The weight for each fish is then adjusted based on the calculated growth rate to generate a new weight for the next iteration. The length (days) to run the growth portion of the model was selected to represent the time from when the fish enter the linear portion of their growth trajectory in the mid to late

spring until they change their growth pattern in the fall due to reductions in temperature and resources or until they migrate out of the system. The outputs of the organismal model that are handed to the population models consist of mean weights (with standard deviations) after the species-appropriate growth period (Table 2). A sensitivity analysis was run to determine the influence of the parameter values on the output of the growth model.

The option of exposing only a specified percent of the population to the pesticide(s) during the somatic growth period is provided. The exposed percent of the population is applied to the number of individuals run in the individual growth model. After running all 1000 individual growth trajectories (with X% exposed and 100-X% control) the mean weight and standard deviation of the whole is determined and handed to the population model to run as the size distribution of the impacted population.

The parameter values defining control conditions that are constant for all the modeled species are listed in Table 1. Model parameters such as the length of the growth period and control daily growth rate that are species specific are listed in Table 2. Each exposure scenario was defined by a concentration and exposure time for each pesticide. The duration of time until full effect for the pesticides was assumed to be within a few days (Ferrari et al. 2004), with a half-life of 0.5 days.

For prey, it is assumed there is a constant, independent influx of prey from upstream habitats that will eventually (depending on the rate selected) return prey abundance to 1. As mentioned above, however, these invertebrates are subject to exposure once added to the system, and therefore prey recovery rate is a product of the influx rate as well as the exposure scenario. While recovery rates reported in the literature vary, it is assumed a 1% recovery rate is ecologically realistic (Ward et al. 1995, Van den Brink et al. 1996, Colville et al. 2008). It was also assumed that regardless of the exposure scenario, relative prey abundance would not drop below a specific floor (Figure 3B). This assumption depends on a minimal yet constant terrestrial subsidy of prey and/or an aquatic community with tolerant individuals that would be available as prey, regardless of pesticide exposure and in addition to the constant recovery rate. No studies specify floors per se, but studies quantifying invertebrate densities following highly toxic exposures indicate a floor of

0.2 is ecologically realistic (i.e. regardless of the exposure, 20% of a fish's ration will be available daily; e.g., Cuffney et al. 1984). Finally, because prey availability has been found to increase dramatically albeit briefly following pesticide exposures (due to immediate mortality and/or emigration of benthic prey into the water column; Davies and Cook 1993, Schulz 2004), a one-day prey spike is included for the day following an exposure. The relative magnitude of the spike is calculated as the product of the standing prey availability the day prior to exposure (minus the floor), the toxicity of the exposure, and a constant of 20. This calculation therefore accounts for the potential prey that are available and the severity of the exposure. The spike will be greater when more prey are available and/or the toxicity of the exposure is greater; alternatively, the spike will be small when few prey are available and/or the exposure toxicity is low.

Below are the mathematical equations used to derive Figures 2, 3, and 4.

Figures 2A and 3A use a step function:

time < start; exposure = 0

start ≤ time ≤ end; exposure = exposure concentration(s)

time > end; exposure = 0.

Figures 2B and 3B use a sigmoid function:

$y = \text{bottom} + (\text{top} - \text{bottom}) / (1 + (\text{exposure concentration} / \text{EC}_{50})^{\text{slope}})$.

For 2B, y = AChE activity, top = Ac, bottom = 0.

For Figure 3B, y = prey abundance, top = Pc (in this case 1), bottom = Pf.

Figures 2D, 2E, and 4C use a linear function (the point-slope form of a line):

$y = m * (x - x1) + y1$.

For 2D, m = Mfa, x1 = Ac, and y1 = Fc.

For 2E, m = Mrf (computed as Rc/Fc), x1 = Fc, and y1 = Rc.

For 4C, m = Mgr, x1 = Rc, and y1 = Gc.

Figure 2C uses a series of exponential functions:

$$\begin{aligned} \text{time} < \text{start}; y &= c \\ \text{start} \leq \text{time} \leq \text{end}; y &= c - (c - i) * (1 - \exp(-ke * (\text{time} - \text{start}))) \\ \text{time} > \text{end}; y &= c - (c - i) * (1 - \exp(-ke * (\text{end} - \text{start}))) \\ y &= ye + (c - ye) * (1 - \exp(-kr * (\text{time} - \text{end}))). \end{aligned}$$

For Figure 2C, $c = A_c$, $i = A_i$, $ke = \ln(2)/AChE \text{ effect half-life}$, $kr = \ln(2)/AChE \text{ recovery half-life}$. For Figure 2C the value of ye is calculated to determine the amount of inhibition that is reached during the exposure time, which may not be long enough to reach the maximum level of inhibition.

For Figure 3C, an exposure pulse would result in a 1-day spike followed by a decline to the impacted level based upon the prey toxicity. During exposures resulting in low prey toxicity, toxicity-limited recovery can occur. After exposure ends a constant rate of recovery proceeds until control drift is reached or another exposure occurs

$$\begin{aligned} \text{preyavail} &= \text{preydrift}(\text{day}-1) - \text{floor}; \\ \text{preytox} &= 1 / (1 + (\text{concentration})^{\text{preyslope}}); \\ \text{preyrecreate} &= 0.01; \\ \text{preydriftrec} &= \text{preyrecreate} * \text{preytox}. \\ \text{time} = \text{start}; \text{spike} &= (-1 + 10^{(1.654 * \text{preyavail})}) * (1 - \text{preytox}) \\ \text{preydrift} &= \text{preydrift} + \text{spike} \\ \text{start} \leq \text{time} \leq \text{end}; \text{preydrift} &= (\text{preyavail} * \text{preytox}) + \text{preydriftrec} + \text{floor}; \\ \text{time} > \text{end}; \text{preydrift} &= \text{preydrift}(\text{day}-1) + \text{preydriftrec} \end{aligned}$$

Figure 2F is generated by using the output of Figure 2C for a given time as the input for 2D and using the resulting output of 2D as the input for 2E. The resulting output of 2E produces a single time point in the relationship in 2F. Performing this series of computations across multiple days produces the entire relationship in 2F. 4D is generated by taking the outputs of 4A and 4B for the same day. Note the relationship of 4A is equivalent to 2F. The resulting outputs of 4A and 4B are multiplied to produce a final ration for a given day. The prey abundance (4B) available for consumption during a prey

spike is capped at a maximum of $1.5 \times \text{control drift}$ to provide a limited benefit to the individual fish. The final ration is used as input for 4C to generate 4D.

Population Model

The weight distributions from the organismal growth portion of the model are used to calculate size-dependent first-year survival for a life-history matrix population model for each species and life-history type. This incorporates the impact that reductions in size could have on population growth rate and abundance. The first-year survival element of the transition matrix incorporates a size-dependent survival rate for a three- or four-month interval (depending upon the species) which takes the juveniles up to 12 months of age. This time represents the 4-month early winter survival in freshwater for stream-type Chinook, coho, and sockeye models. For ocean-type Chinook, it is the 3-month period the subyearling smolt spend in the estuary and nearshore habitats (i.e. estuary survival). The weight distributions from the organismal model are converted to length distributions by applying condition factors from data for each modeled species (cf; 0.0095 for sockeye and 0.0115 for all others) as shown in Equation L.

$$\text{Equation L: length(mm)} = ((\text{fish weight(g)}/\text{cf})^{(1/3)}) * 10$$

The relationship between length and early winter or estuary survival rate was adapted from Zabel and Achord (2004) to match the survival rate for each control model population (Howell et al. 1985, Kostow 1995, Myers et al. 2006). The relationship is based on the length of a subyearling salmon relative to the mean length of other competing subyearling salmon of the same species in the system, Equation D, and relates that relative difference to size-dependent survival based upon Equation S. The values for α and resulting size-dependent survival (survival ϕ) for control runs for each species are listed in Table 2. The constant α is a species-specific parameter defined such that it produces the correct control survival ϕ value when Δlength equals zero.

$$\text{Equation D: } \Delta\text{length} = \text{fish length(mm)} - \text{mean length(mm)}$$

$$\text{Equation S: Survival } \phi = (e^{(\alpha + (0.0329 * \Delta\text{length}))}) / (1 + e^{(\alpha + (0.0329 * \Delta\text{length}))})$$

Randomly selecting length values from the normal distribution calculated from the organismal model output size and applying equations 1 and 2 generates a size-dependent survival probability for each fish. This process was replicated 1000 times for each exposure scenario and simultaneously 1000 times for the paired control scenario and results in a mean size-dependent survival rate for each population. The resulting size-dependent survival rates are inserted in the

calculation of first-year survival in the respective control and pesticide-exposed transition matrices.

The investigation of population-level responses to pesticide exposures uses life-history projection matrix models. Individuals within a population exhibit various growth, reproduction, and survivorship rates depending on their developmental or life-history stage or age. These age specific characteristics are depicted in the life-history graph (Figure 1A-D) in which transitions are depicted as arrows. The nonzero matrix elements represent transitions corresponding to reproductive contribution or survival, located in the top row and the subdiagonal of the matrix, respectively (Figure 1E). The survival transitions in the life-history graph are incorporated into the $n \times n$ square matrix (A) by assigning each age a number (1 through n) and each transition from age i to age j becomes the element a_{ij} of matrix A (i = row, j = column) and represent the proportion of the individuals in each age passing to the next age as a result of survival. The reproductive element (a_{1j}) gives the number of offspring that hatch per individual in the contributing age, j . The reproductive element value incorporates the proportion of females in each age, the proportion of females in the age that are sexually mature, fecundity, fertilization success, and hatch success.

In order to understand the relative impacts of a short-term pesticide exposure on exposed vs. unexposed fish, we used parameters for an idealized control population that exhibits an increasing population growth rate. All characteristics exhibit density independent dynamics. The models assume closed systems, allowing no migration impact on population size. No stochastic impacts are included beyond natural variability as represented by selecting parameter values from a normal distribution about a mean each model iteration (year). Ocean conditions, freshwater habitat, fishing pressure, and marine resource availability were assumed constant and density independent.

In the model an individual fish experiences an exposure scenario once as a subyearling (during its first spring) and never again. The pesticide exposure is assumed to occur annually. All subyearlings within a given population are assumed to be exposed to the pesticide. No other age

classes experience the exposure. The model integrates this as every brood class being exposed as subyearlings and thus the vital demographic rates of the transition matrix are continually impacted in the same manner. Regardless of the level of AChE inhibition due to the direct exposure, only the sublethal effects are incorporated in the models.

The model recalculates first-year survival for each run using a size-dependent survival value selected from a normal distribution with the mean and standard deviation produced by Equation S. Population model output consists of the percent change in lambda from the unexposed control populations derived from the mean of two thousand calculations of both the unexposed control population and the pesticide exposed population. Change in lambda, representing alterations to the population productivity, was selected as the primary model output for reasons outlined previously.

A prospective analysis of the transition matrix, A , (Caswell 2001) explored the intrinsic population growth rate as a function of the vital rates. The intrinsic population growth rate, λ , equals the dominant eigenvalue of A and was calculated using matrix analysis software (MATLAB version 7.7.0 by The Math Works Inc., Natick, MA). Therefore λ is calculated directly from the matrix and running projections of abundances over time is redundant and unnecessary. The stable age distribution, the proportional distribution of individuals among the ages when the population is at equilibrium, is calculated as the right normalized eigenvector corresponding to the dominant eigenvalue λ . Variability was integrated by repeating the calculation of λ 2000 times selecting the values in the transition matrix from their normal distribution defined by the mean standard deviation. The influence of each matrix element, a_{ij} , on λ was assessed by calculating the sensitivity values for A . The sensitivity of matrix element a_{ij} equals the rate of change in λ with respect to a_{ij} , defined by $\delta\lambda / \delta a_{ij}$. Higher sensitivity values indicate greater influence on λ . The elasticity of matrix element a_{ij} is defined as the proportional change in λ relative to the proportional change in a_{ij} , and equals (a_{ij}/λ) times the sensitivity of a_{ij} . One characteristic of elasticity analysis is that the elasticity values for a transition matrix sum to

unity (one). The unity characteristic also allows comparison of the influence of transition elements and comparison across matrices.

Due to differences in the life-history strategies, specifically lifespan, age at reproduction and first year residence and migration habits, four life-history models were constructed. This was done to encompass the different responses to freshwater pesticide exposures and assess potentially different population-level responses. Separate models were constructed for coho, sockeye, ocean-type and stream-type Chinook. In all cases transition values were determined from literature data on survival and reproductive characteristics of each species.

A life-history model was constructed for coho salmon (*O. kisutch*) with a maximum age of 3. Spawning occurs in late fall and early winter with emergence from March to May. Fry spend 14-18 months in freshwater, smolt and spend 16-20 months in the saltwater before returning to spawn (Pess et al. 2002). Survival numbers were summarized in Knudsen et al. (2002) as follows. The average fecundity of each female is 4500 with a standard deviation of 500. The observed number of males:females was 1:1. Survival from spawning to emergence is 0.3 (0.07). Survival from emergence to smolt is 0.0296 (0.00029) and marine survival is 0.05 (0.01). All parameters followed a normal distribution (Knudson et al. 2002). The calculated values used in the matrix are listed in Table 3. The growth period for first year coho was set at 180 days to represent the time from mid-spring to mid-fall when the temperatures and resources drop and somatic growth slows (Knudson et al. 2002).

Life-history models for sockeye salmon (*O. nerka*) were based upon the lake wintering populations of Lake Washington, Washington, USA. These female sockeye salmon spend one winter in freshwater, then migrate to the ocean to spend three to four winters before returning to spawn at ages 4 or 5. Jacks return at age 2 after only one winter in the ocean. The age proportion of returning adults is 0.03, 0.82, and 0.15 for ages 3, 4 and 5, respectively (Gustafson et al. 1997). All age 3 returning adults are males. Hatch rate and first year survival were calculated from brood year data on escapement, resulting presmolts and returning adults (Pauley et al. 1989) and fecundity (McGurk 2000). Fecundity values for age 4 females were 3374 (473) and for age 5

females were 4058 (557) (McGurk 2000). First year survival rates were 0.737/month (Gustafson et al. 1997). Ocean survival rates were calculated based upon brood data and the findings that 90% of ocean mortality occurs during the first 4 months of ocean residence (Pauley et al. 1989). Matrix values used in the sockeye baseline model are listed in Table 3. The 168 day growth period represents the time from lake entry to early fall when the temperature drops and somatic growth slows (Gustafson et al. 1997).

A life-history model was constructed for ocean-type Chinook salmon (*O. tshawytscha*) with a maximum female age of 5 and reproductive maturity at ages 3, 4 or 5. Ocean-type Chinook migrate from their natal stream within a couple months of hatching and spend several months rearing in estuary and nearshore habitats before continuing on to the open ocean. Transition values were determined from literature data on survival and reproductive characteristics from several ocean-type Chinook populations in the Columbia River system (Healey and Heard 1984, Howell et al. 1985, Roni and Quinn 1995, Ratner et al. 1997, PSCCTC 2002, Green and Beechie 2004). The sex ratio of spawners was approximately 1:1. Estimated size-based fecundity of 4511(65), 5184(89), and 5812(102) was calculated based on data from Howell et al., 1985, using length-fecundity relationships from Healy and Heard (1984). Control matrix values for the Chinook model are listed in Table 3. The growth period of 140 days encompasses the time the fish rear in freshwater prior to entering the estuary and open ocean. The first three months of estuary/ocean survival are the size-dependent stage. Size data for determining subyearling Chinook condition indices came from data collected in the lower Columbia River and estuary (Johnson et al. 2007).

An age-structured life-history matrix model for stream-type Chinook salmon with a maximum age of 5 was defined based upon literature data on Yakima River spring Chinook from Knudsen et al. (2006) and Fast et al. (1988), with sex ratios of 0.035, 0.62 and 0.62 for females spawning at ages 3, 4, and 5, respectively. Length data from Fast et al. (1988) was used to calculate fecundity from the length-fecundity relationships in Healy and Heard (1984). The 184-day growth period produces control fish with a mean size of 96mm, within the observed range

documented in the fall prior to the first winter (Beckman et al. 2000). The size-dependent survival encompasses the 4 early winter months, up until the fish are 12 months old.

Acute Toxicity Model

In order to estimate the population-level responses of exposure to lethal pesticide concentrations, acute mortality models were constructed based upon the control life-history matrices described above. The acute responses are modeled as direct reduction in the first year survival rate (S1). Two options are available to run, direct mortality estimates and exposure scenarios. Direct mortality can be input as percent mortality and is multiplied by the first-year survival rate in the transition matrix. For the exposure scenarios all subyearling salmon are assumed to be exposed in each scenario. Exposures are assumed to result in a cumulative reduction in survival as defined by the concentration and the dose-response curve as defined by the LC_{50} and slope for each pesticide. A sigmoid dose-response relationship is used to accurately handle responses well away from LC_{50} and to be consistent with other dose-response relationships. The model inputs for each scenario are the exposure concentration and acute fish LC_{50} , as well as the sigmoid slope for the LC_{50} . For a given concentration a pesticide survival rate (1-mortality) is calculated and is multiplied by the control first-year survival rate, producing an exposed scenario first-year survival for the life-history matrix. Variability is incorporated as described above using mean and standard deviation of normally distributed survival and reproductive rates and model output consists of the percent change in lambda from unexposed control populations derived from the mean of 10000 calculations of both the unexposed control population and the pesticide exposed population. The percent change in lambda is considered different from control when the difference is greater than the percent of one standard deviation from the control lambda.

Results

Sensitivity Analysis

A sensitivity analysis conducted on the organismal model revealed that changes in the control somatic growth rate had the greatest influence on the final weights (Table 1). While this parameter value was experimentally derived for another species (sockeye salmon; Brett et al. 1969), this value was adapted for each model species and is within the variability reported in the literature for other salmonids (reviewed in Weatherley and Gill 1995). Other parameters related to the daily growth rate calculation, including the growth to ration slope (Mgr) and the control ration produced strong sensitivity values. Initial weight, the prey recovery rate and the prey floor

also strongly influenced the final weight values (Table 1). Large changes (0.5 to 2X) in the other key parameters produced proportionate changes in final weight.

The sensitivity analysis of all four of the control population matrices predicted the greatest changes in population growth rate (λ) result from changes in first-year survival. Parameter values and their corresponding sensitivity values are listed in Table 4. The elasticity values for the transition matrices also corresponded to the driving influence of first-year survival, with contributions to lambda of 0.33 for coho, 0.29 for ocean-type Chinook, 0.25 for stream-type Chinook, and 0.24 for sockeye.

Model Output

Organismal and population model outputs for all scenarios are summarized as graphs in the main text of the Opinion. As expected, greater changes in population growth resulted from longer exposures to the pesticides. The factors driving the level of change in lambda were the Prey Drift and relative AChE Activity parameters determined by the toxicity values for each pesticide (Table 3).

Output from the acute toxicity models was presented in the Risk Characterization section of the main text. Increases in direct mortality during the first year of life produced large impacts on the population growth rates for all the life-history strategies.

While strong trends in effects were seen for each pesticide across all four life-history strategies modeled, some slight differences were apparent. The similarity in patterns likely stems from using the same toxicity values for all four models, while the differences are consequences of distinctions between the life-history matrices. The stream-type Chinook and sockeye models produced very similar results as measured as the percent change in population growth rate. The ocean-type Chinook and coho models output produced the greatest changes in lambda resulting from the pesticide exposures. When looking for similarities in parameters to explain the ranking, no single life history parameter or characteristic, such as lifespan, reproductive ages, age distribution, lambda and standard deviation, or first-year survival show a pattern that matches

this consistent output. Combining these factors into the transition matrix for each life-history and conducting the sensitivity and elasticity analyses revealed that changes in first-year survival produced the greatest changes in λ . In addition, the elasticity analysis can be used to predict relative contribution to λ from changes in first-year survival on a per unit basis. As detailed by the elasticity values reported above, the same change in first-year survival will produce a slightly greater change in the population growth rate for coho and ocean-type Chinook than for stream-type Chinook and sockeye. While some life-history characteristics may lead a population to be more vulnerable to an impact, the culmination of age structure, survival and reproductive rates as a whole strongly influences the population-level response.

Captions for figures

Figure 1: Life-History Graphs and Transition Matrix for coho (A), sockeye (B) and Chinook (C) salmon. The life-history graph for a population labeled by age, with each transition element labeled according to the matrix position, a_{ij} , i row and j column. Dashed lines represent reproductive contribution and solid lines represent survival transitions. D) The transition matrix for the life-history graph depicted in C.

Figure 2: Relationships used to link anticholinesterase exposure to the organism's ability to acquire food (potential ration). See text for details. Relationships in B, C, and D utilize empirical data. Closed circles represent control conditions. Open circles represent the exposed (inhibited) condition. A) Representation of a constant level of anticholinesterase pesticide exposure (either a single compound or mixtures). B) Sigmoidal relationship between exposure concentration and steady-state acetylcholinesterase (AChE) activity showing a dose-dependent reduction defined by control activity (horizontal line, A_c), sigmoidal (i.e. hille) slope (AChE slope), and the concentration producing 50% inhibition (vertical line, EC_{50}). C) Timecourse of acetylcholinesterase inhibition based on modeling the time-to-effect and time-to-recovery as single exponential curves with different time-constants. At the start of the exposure AChE activity will be at control and then decline toward the inhibited activity (A_i) based on Panel B. D) Linear model relating acetylcholinesterase activity to feeding behavior using a line that passes through the feeding (F_c) and activity (A_c) control conditions with a slope of M_{fa} . E) The relationship between feeding behavior and the potential ratio an organism could acquire (if not food limited) used a line passing through the control conditions (F_c as in Panel D and the control ration, R_c) and through the origin producing a slope (M_{rf}) equal to R_c/F_c . F) Timecourse for effect of exposure to anticholinesterase on potential ration produced by combining C & E.

Figure 3: Relationships used to link anticholinesterase exposure to the availability of prey. See text for details. Relationships in B and C utilize empirical data. Closed circles represent control conditions. Open circles represent the exposed (inhibited) condition. A) Representation of a constant level of anticholinesterase pesticide exposure (either single compound or mixtures). B) Sigmoidal relationship between exposure concentration and relative prey abundance showing a dose-dependent reduction defined by control abundance (horizontal line at 1, P_c), sigmoid (i.e. hille) slope (prey slope), the concentration producing a 50% reduction in prey (vertical line, EC_{50}), and a minimum abundance always present (horizontal line denoted as floor, P_f). C) Timecourse of prey abundance including a 1-day spike in prey drift relative to the available prey and the level of toxicity followed by a drop to the level of impact or the floor whichever is greater. During extended exposures at low toxicity recovery can begin at the constant prey influx rate multiplied by the current level of toxicity. After exposure recovery to control prey drift is at the constant rate of influx from upstream habitats.

Figure 4: Relationships used to link anticholinesterase exposure to growth rate relating to long-term weight gain of each fish. See text for details. Relationships in A, B, and C utilize empirical

data. Closed circles represent control conditions. Open circles (e.g. Ai) represent the exposed (inhibited) condition. A&B) Relationships describing the Timecourse of the effects of anticholinesterase exposure on the organisms ability to capture food (Panel A, potential ration) and the availability of food to capture (Panel B, relative prey abundance). The figures are the same as those in Figures 2F and 3C, respectively. For a given exposure concentration and time, multiplying potential ration by relative prey abundance yields the final ration acquired by the organism. C) A linear model was used to relate final ration to growth rate using a line passing through the control conditions and through the maintenance condition with a slope denoted by Mgr. D) Timecourse for effect of exposure to anticholinesterase on growth rate produced by combining A, B, & C.

Table 1. List of values used for control parameters to model organismal growth and the model sensitivity to changes in the parameter.

Parameter	Value ¹	Error ²	Sensitivity ³
acetylcholinesterase activity (Ac)	1.0 ^{4,5}	0.06 ⁵	-0.167
feeding (Fc)	1.0 ^{4,5}	0.05 ⁵	0.088
ration (Rc)	5% weight/day ⁶	0.05 ⁷	-0.547
feeding vs. activity slope (Mfa)	1.0 ⁵	0.1 ⁵	-0.047
ration vs. feeding slope (Mrf)	5 (Rc/Fc)	-	-
growth vs. ration slope (Mgr)	0.35 ⁶	0.02 ⁶	-0.547
growth vs. activity slope (Mga)	1.75 (Mfa*Mrf*Mgr)	-	-
initial weight	1 gram ⁸	0.1 ⁸	1.00
control prey drift	1.0 ⁴	0.05 ¹¹	0.116
AChE impact time-to-effect (t _{1/2})	0.5 day ⁹	n/a	0.005
AChE time-to-recovery (t _{1/2})	30 days ¹⁰	n/a	-0.0001
prey floor	0.20 ¹¹	n/a	0.178
prey recovery rate	0.01 ¹²	n/a	0.323
somatic growth rate (Gc)	1.3 ¹³	0.06 ⁶	2.531

¹ mean value of a normal distribution used in the model or constant value when no corresponding error is listed

² standard deviation of the normal distribution used in the model

³ mean sensitivity when baseline parameter is changed over range of 0.5 to 2-fold

⁴ other values relative to control

⁵ derived from Sandahl et al. 2005

⁶ derived from Brett et al. 1969

⁷ data from Brett et al. 1969 has no variability (ration was the independent variable) so a variability of 1% was selected to introduce some variability

⁸ consistent with field-collected data for juvenile Chinook (Nelson et al. 2004)

⁹ estimated from Ferrari et al. 2004

¹⁰ consistent with Eder et al., 2007; Ferrari et al., 2004; Chambers et al., 2002

¹¹ estimated from Van den Brink et al. 1996

¹² derived from Ward et al. 1995, Van den Brink et al. 1996, Colville et al. 2008

¹³ derived from Brett et al. 1969 and adapted for ocean-type Chinook, used for sensitivity analysis

Table 2. Species specific control parameters to model organismal growth and survival rates. Growth period and survival rate are determined from the literature data listed for each species. Gc and α were calculated to make the basic model produce the appropriate size and survival values from the literature.

	Chinook Stream-type ¹	Chinook Ocean-type ²	Coho ³	Sockeye ⁴
days to run organismal growth model	184	140	184	168
growth rate % body wt/day (Gc)	1.28	1.30	0.90	1.183
α from equation S	-0.33	-1.99	-0.802	-0.871
Control Survival ϕ	0.418	0.169	0.310	0.295

¹ Values from data in Healy and Heard 1984, Fast et al. 1988, Beckman et al. 2000, Knudsen et al. 2006

² Values from data in Healey and Heard 1984, Howell et al. 1985, Roni and Quinn 1995, Ratner et al. 1997, PSCCTC 2002, Green and Beechie 2004, Johnson et al. 2007

³ Values from data in Pess et al. 2002, Knudsen et al. 2002

⁴ Values from data in Pauley et al. 1989, Gustafson et al. 1997, McGurk 2000

Table 3. Effects values (ug/L) and slopes for AChE activity, acute fish lethality, and prey abundance dose-response curves.

compound	AChE Activity EC ₅₀ ¹ ug/L	AChE Activity slope	Fish lethality LC ₅₀ ² ug/L	Fish lethality slope ³	Prey Abundance EC ₅₀ ⁴ ug/L	Prey Abundance Slope ⁵
Azinphos-methyl	0.1639	0.99	1.2	3.63	1.13	3.63
Bensulide	180	0.99	720	3.63	580	3.63
Dimethoate	273.4	0.99	6200	3.63	3320	3.63
Disulfoton	485.5	0.99	1850	3.63	13	3.63
Ethoprop	90.62	0.99	1020	3.63	44	3.63
Fenamiphos			68	3.63	1.3	3.63
Methamidophos	10000	0.99	25000	3.63	26	3.63
Methidathion	1.123	0.99	6.6	3.63	3	3.63
Methyl parathion	28.75	0.99	1850	3.63	0.14	3.63
Naled	7.848	0.99	87	3.63	0.3	3.63
Phorate	0.5697	0.99	13	3.63	37	3.63
Phosmet	3.25	0.99	150	3.63	5.6	3.63

¹ Values from Laetz et al.

² Values from EPA BEs

³ sigmoidal slope that produces responses with a probit slope of Peterson et al. 2001a, see text.

⁴ Values from analysis of global search of reported LC₅₀ and EC_{50s} reported in EPA's Ecotox database. See text.

⁵ Values from Peterson et al. 2001a

Table 4. Matrix transition element and sensitivity (S) and elasticity (E) values for each model species. These control values are listed by the transition element taken from the life-history graphs as depicted in Figure 1 and the literature data described in the method text. Blank cells indicate elements that are not in the transition matrix for a particular species. The influence of each matrix element on λ was assessed by calculating the sensitivity (S) and elasticity (E) values for A. The sensitivity of matrix element a_{ij} equals the rate of change in λ with respect to the transition element, defined by $\delta\lambda/\delta a$. The elasticity of transition element a_{ij} is defined as the proportional change in λ relative to the proportional change in a_{ij} , and equals (a_{ij}/λ) times the sensitivity of a_{ij} . Elasticity values allow comparison of the influence of individual transition elements and comparison across matrices.

Transition Element	Chinook Stream-type			Chinook Ocean-type			Coho			Sockeye		
	Value ¹	S	E	Value ²	S	E	Value ³	S	E	Value ⁴	S	E
S1	0.0643	3.844	0.247	0.0056	57.13	0.292	0.0296	11.59	0.333	0.0257	9.441	0.239
S2	0.1160	2.132	0.247	0.48	0.670	0.292	0.0505	6.809	0.333	0.183	1.326	0.239
S3	0.17005	1.448	0.246	0.246	0.476	0.106				0.499	0.486	0.239
S4	0.04	0.319	0.0127	0.136	0.136	0.0168				0.1377	0.322	0.0437
R3	0.5807	0.00184	0.0011	313.8	0.0006	0.186	732.8	0.000469	0.333			
R4	746.73	0.000313	0.233	677.1	0.000146	0.0896				379.57	0.000537	0.195
R5	1020.36	1.25E-05	0.0127	1028	1.80E-05	0.0168				608.7	7.28E-05	0.0437

¹ Value calculated from data in Healy and Heard 1984, Fast et al. 1988, Beckman et al. 2000, Knudsen et al. 2006

² Value calculated from data in Healey and Heard 1984, Howell et al. 1985, Roni and Quinn 1995, Ratner et al. 1997, PSCCTC 2002, Green and Beechie 2004, Johnson et al. 2007

³ Value calculated from data in Pess et al. 2002, Knudsen et al. 2002

⁴ Value calculated from data in Pauley et al. 1989, Gustafson et al. 1997, McGurk 2000

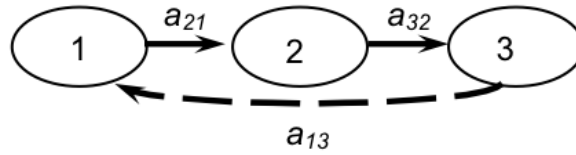
Table 5. 48 h survival EC₅₀s of *Daphnia magna*.

Organophosphate	<i>Daphnia magna</i> 48 hr EC ₅₀ (µg/L) (95% CI)	Data Source
Azinphos methyl	1.13	MRID 00068678
Dimethoate	3320 (1730-4120)	Song, M.Y., J.D. Stark, J.J. Brown. 1997. Comparative Toxicity of Four Insecticides, Including Imidacloprid and Tebufenozide, to Four Aquatic. Environ.Toxicol.Chem. 16(12):2494-2500
Disulfoton	13	MRID 00143401
Ethoprop	44	MRID 00068325
Methamidophos	26 (20-34)	MRID 00041311)
Methidathion	3	MRID 42081704 (1991)
Methyl parathion	0.14 (0.09-0.2)	MRID 40094602
Naled	0.3	MRID BA0NAL02
Phorate	37 (30-44)	MRID 0161825
Phosmet	5.6	MRID 00063194,

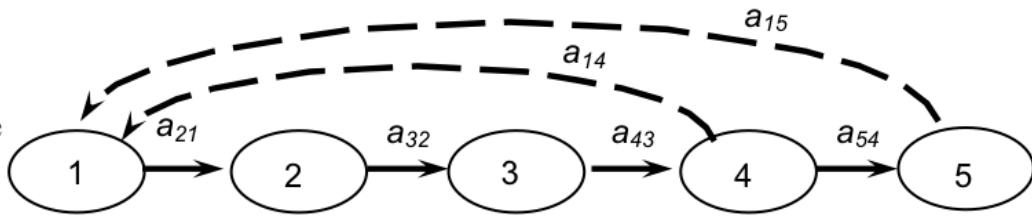
For this table, unless noted otherwise EC₅₀ from EPA documents, and is referred to by MRID number.

Figure 1.

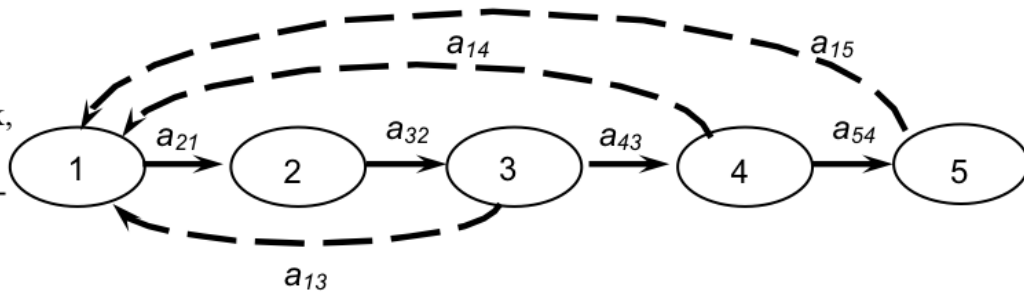
A. Coho



B. Sockeye



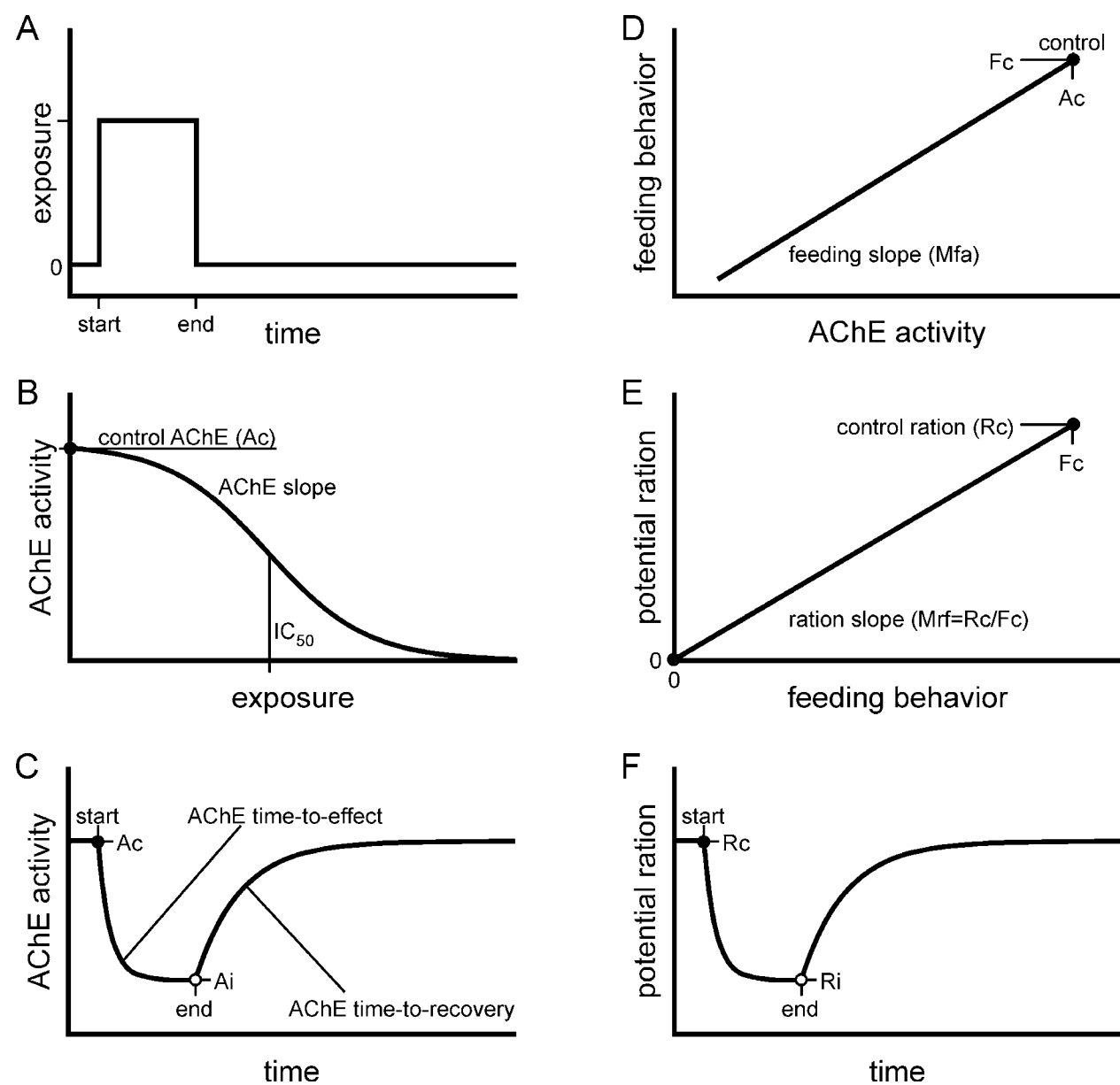
C. Chinook,
ocean-type
and stream-
type



D. Transition
Matrix

$$A = \begin{bmatrix} 0 & 0 & R3=a_{13} & R4=a_{14} & R5=a_{15} \\ S1=a_{21} & 0 & 0 & 0 & 0 \\ 0 & S2=a_{32} & 0 & 0 & 0 \\ 0 & 0 & S3=a_{43} & 0 & 0 \\ 0 & 0 & 0 & S4=a_{54} & 0 \end{bmatrix}$$

Figure 2.



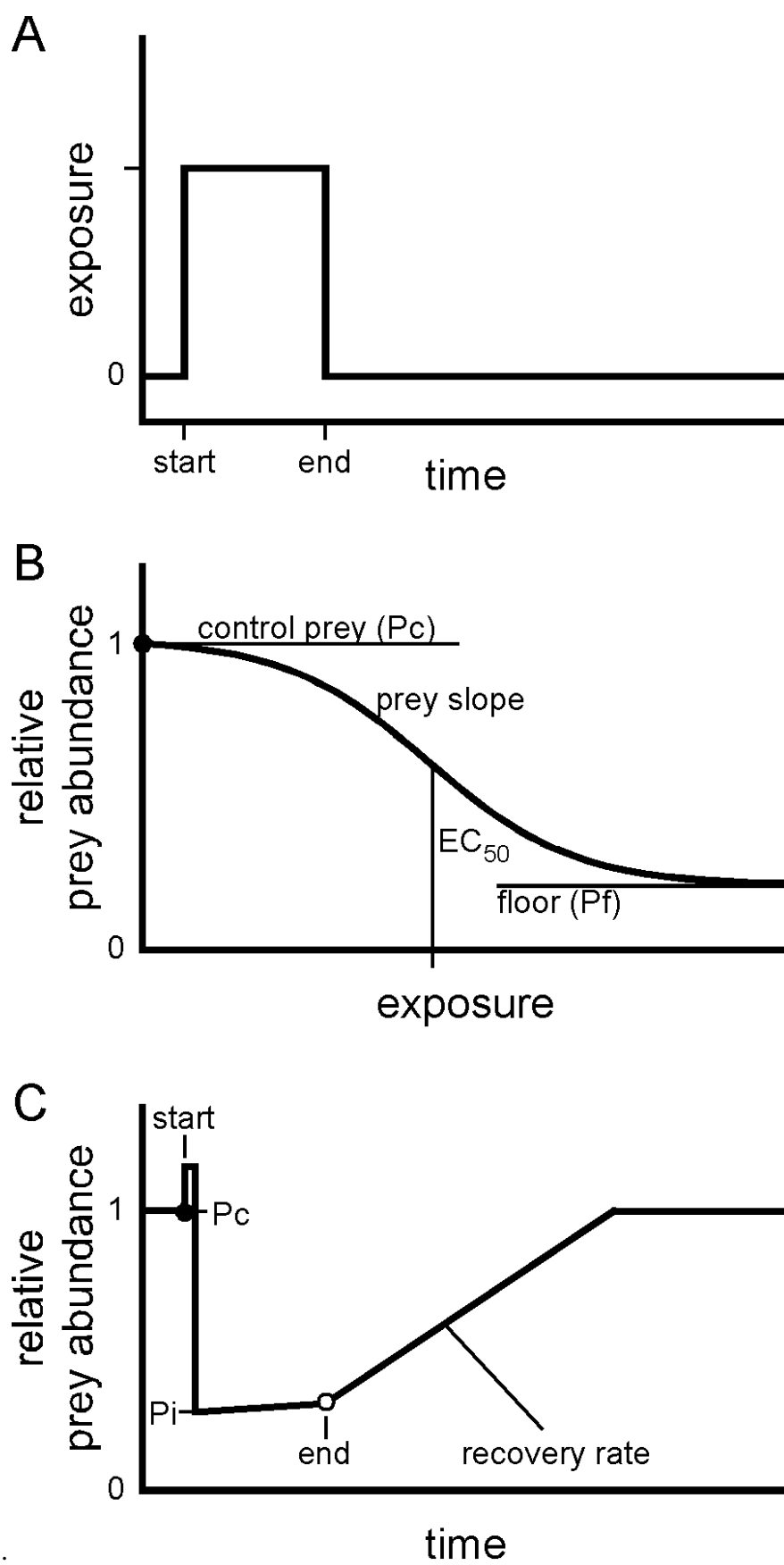


Figure 3.

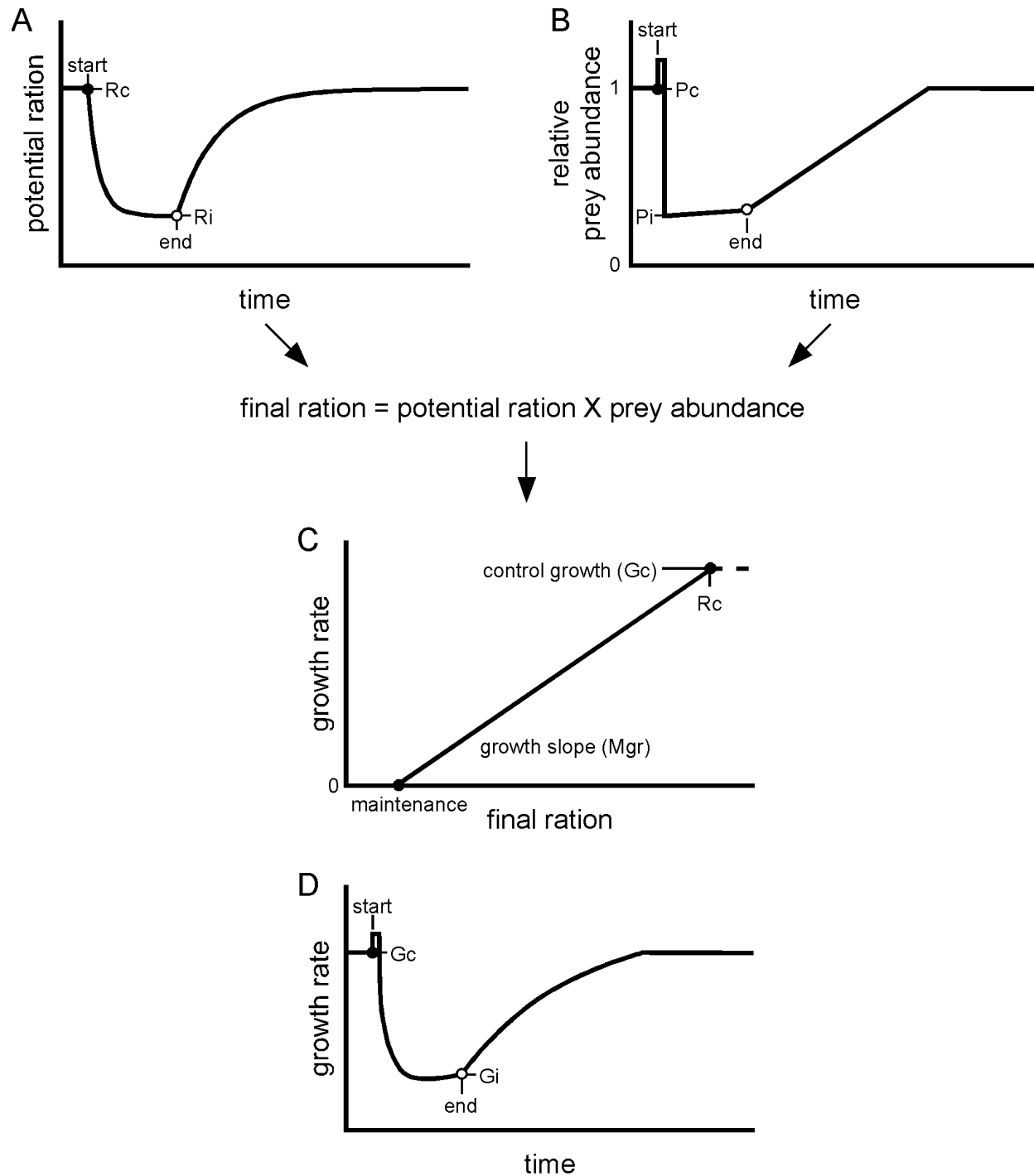


Figure 4.

References

- Anderson, B.S., Hunt, J.W., Phillips, B.M., Nicely, P.A., Gilbert, K.D., DeVlaming, V., Connor, V., Richard, N., and Tjeerdemat, R.S. 2003. Ecotoxicologic impacts of agricultural drain water in the Salinas River, California, USA. *Environmental Toxicology and Chemistry* 22:2375-2384
- Baldwin, D.B., Spromberg, J.A., Collier, T.K., and Scholz, N.L. 2009. A fish of many scales: extrapolating sublethal pesticide exposures to the productivity of wild salmon populations. *Ecological Applications*. In press.
- Beamish, R.J., and Mahnken, C. 2001. A critical size and period hypothesis to explain natural regulation of salmon abundance and the linkage to climate and climate change. *Progress in Oceanography* 49: 423–437.
- Beauvais, S.L., Jones, S.B., Brewer, S.K., and Little, E.E. 2000. Physiological measures of neurotoxicity of diazinon and malathion to larval rainbow trout (*Oncorhynchus mykiss*) and their correlation with behavioral measures. *Environmental Toxicology and Chemistry* 19:1875-1880.
- Beckman, B.R., Larsen, D.A., Sharpe, C., Lee-Pawlak, B., Schreck, C.B., and Dickhoff, W.W. 2000. Physiological status of naturally reared juvenile spring Chinook salmon in the Yakima River: seasonal dynamics and changes associated with smolting. *Transactions of the American Fisheries Society* 129:727–753.
- Brett, J.R., Shelbourn, J.E., and Shoop, C.T. 1969. Growth rate and body composition of fingerling sockeye salmon, *Oncorhynchus nerka*, in relation to temperature and ration size. *Journal of the Fisheries Board of Canada* 26:2363-2394.
- Brewer, S.K., Little, E.E., DeLonay, A.J., Beauvais, S.L., Jones, S.B. and Ellersieck, M.R. 2001. Behavioral dysfunctions correlate to altered physiology in rainbow trout (*Oncorhynchus mykiss*) exposed to cholinesterase-inhibiting chemicals. *Archives of Environmental Contamination and Toxicology* 40:70-76.
- Caswell, H. 2001. Matrix population models: Construction, analysis, and interpretation. Sunderland, MA, USA: Sinauer Assoc.

- Chang, K.H., Sakamoto, M., and Hanazato, T. 2005. Impact of pesticide application on zooplankton communities with different densities of invertebrate predators: An experimental analysis using small-scale mesocosms. *Aquatic Toxicology* 72:373-382.
- Cold, A., and Forbes, V.E. 2004. Consequences of a short pulse of pesticide exposure for survival and reproduction of *Gammarus pulex*. *Aquatic Toxicology* 67: 287-299.
- Colville, A., Jones, P., Pablo, F., Krassoi, F., Hose, G., and Lim, R. 2008. Effects of chlorpyrifos on macroinvertebrate communities in coastal stream mesocosms. *Ecotoxicology* 17: 173-180.
- Cuffney, T.F., Wallace, J.B., and Webster, J.R. 1984. Pesticide manipulation of a headwater stream: invertebrate responses and their significance for ecosystem processes. *Freshwater Invertebrate Biology* 3: 153-171.
- Davies, P.E., and Cook, L.S.J. 1993. Catastrophic macroinvertebrate drift and sublethal effects on brown trout, *Salmo trutta*, caused by cypermethrin spraying on a Tasmanian stream. *Aquatic Toxicology* 27(3-4):201-224.
- Fast, D.E., Hubble, J.D., and Kohn, M.S. 1988. Yakima River Spring Chinook Enhancement Study, Annual Report FY 1988. U.S. Department of Energy, Bonneville Power Administration Division of Fish and Wildlife. Project No. 82-16. 101pp.
- Ferrari, A., Venturino, A., and Pechen de D'Angelo, A.M. 2004. Time course of brain cholinesterase inhibition and recovery following acute and subacute azinphosmethyl, parathion, and carbaryl exposure in the goldfish (*Carassius auratus*). *Ecotoxicology and Environmental Safety* 57:420-425.
- Fleeger, J.W., Carman, K.R., and Nisbet, R.M. 2003. Indirect effects of contaminants in aquatic ecosystems. *Science of the Total Environment* 317:207-233.
- Forbes, V.E., and Cold, A. 2005. Effects of the pyrethroid esfenvalerate on life-cycle traits and population dynamics of *Chironomus riparius* - Importance of exposure scenario. *Environmental Toxicology and Chemistry* 24: 78-86.
- Greene, C.M., and Beechie, T.J. 2004. Consequences of potential density-dependent mechanisms on recovery of ocean-type Chinook salmon (*Oncorhynchus tshawytscha*). *Canadian Journal of Fisheries and Aquatic Sciences* 61:590-602.

- Gustafson, R.G., Wainwright, T.C., Winans, G.A., Waknitz, F.W., Parker, L.T., and Waples, R.S. 1997. Status review of sockeye salmon from Washington and Oregon. U.S. Dept. Commerce, NOAA Tech. Memo. NMFS-NWFSC-33, 282 pp.
- Healey, M.C. 1982. Timing and relative intensity of size-selective mortality of juvenile chum salmon (*Oncorhynchus keta*) during early sea life. *Canadian Journal of Fisheries and Aquatic Sciences* 39:952-957.
- Healey, M.C. 1991. Life history of Chinook salmon (*Oncorhynchus tshawytscha*). Pages 311–394 in C. Groot and L. Margolis, editors. Pacific salmon life histories. University of British Columbia Press, Vancouver, Canada.
- Healey, M.C., and Heard, W.R. 1984. Inter- and intra-population variation in the fecundity of Chinook salmon (*Oncorhynchus tshawytscha*) and its relevance to life history theory. *Canadian Journal of Fisheries and Aquatic Sciences* 41:476-483.
- Higgs, D.A., MacDonald, J.S., Levings, C.D., and Dosanjh, B.S. 1995. Nutrition and feeding habits in relation to life history stage. Pages 159-315. in C. Groot, L. Margolis, and W.C. Clarke, editors. Physiological Ecology of Pacific Salmon. University of British Columbia Press, Vancouver, Canada.
- Holtby, L.B., Andersen, B.C., and Kadowak, R.K. 1990. Importance of smolt size and early ocean growth to interannual variability in marine survival of coho salmon (*Oncorhynchus kisutch*). *Canadian Journal of Fisheries and Aquatic Sciences* 47:2181-2194.
- Howell, P., Jones, K., Scarnecchia, D., LaVoy, L., Kendra, W., Ortmann, D., Neff, C., Petrosky, C., and Thurow, R. 1985. Stock assessment of Columbia River anadromous salmonids Volume I: Chinook, coho, chum, and sockeye salmon stock summaries. Final Report to Bonneville Power Administration. Bonneville Power Administration, P.O Box 3621, Portland OR 97208, DE-AI79-84BP12737, Project No. 83-335. 579 p.
- Johnson, K.R., Jepson, P.C. and Jenkins, J.J. 2008. Esfenvalerate-induced case-abandonment in the larvae of the caddisfly (*Brachycentrus americanus*). *Environmental Toxicology and Chemistry* 27: 397-403.
- Johnson, L.L., Ylitalo, G.M., Arkoosh, M.R., Kagley, A.N., Stafford, C.L., Bolton, J.L., Buzitis, J., Anulacion, B.F., and Collier, T.K. 2007. Contaminant exposure in outmigrant juvenile

- salmon from Pacific Northwest estuaries. *Environmental Monitoring and Assessment* 124:167-194.
- Knudsen, C.M., Schroder, S.L., Busack, C.A., Johnston, M.V., Pearsons, T.N., Bosch, W.J., and Fast, D.E. 2006. Comparison of life history traits between first-generation hatchery and wild upper Yakima River spring Chinook salmon. *Transactions of the American Fisheries Society* 135:1130-1144.
- Knudsen, E.E., Symmes, E.W., and Margraf, F.J. 2002. Searching for an ecological life history approach to salmon escapement management. *American Fisheries Society Symposium* 34:261-276.
- Kostow, K. 1995. Biennial Report on the Status of Wild Fish in Oregon. Oreg. Dep. Fish Wildl. Rep., 217 p. + app. (Available from Oregon Department of Fish and Wildlife, P.O. Box 59, Portland, OR 97207.)
- Labenia, J.S., Baldwin, D.H., French, B.L., Davis, J.W., and Scholz, N.L. 2007. Behavioral impairment and increased predation mortality in cutthroat trout exposed to carbaryl. *Marine Ecology Progress Series* 329: 1–11.
- Laetz, C.A., Baldwin, D.H., Collier, T.K., Hebert, V., Stark, J.D., and Scholz, N.L. 2009. The synergistic toxicity of pesticide mixtures: implications for risk assessment and the conservation of endangered Pacific salmon. *Environmental Health Perspectives* 117:348-353.
- Liess, M., and Schulz, R. 1999. Linking insecticide contamination and population response in an agricultural stream. *Environmental Toxicology and Chemistry* 18(9):1948-1955.
- McClure, M.M., Holmes, E.E., Sanderson, B.L., and Jordan, C.E. 2003. A large-scale, multispecies status assessment: anadromous salmonids in the Columbia River Basin. *Ecological Applications* 13(4):964-989.
- McGurk, M.D. 2000. Comparison of fecundity–length–latitude relationships between nonanadromous (kokanee) and anadromous sockeye salmon (*Oncorhynchus nerka*). *Canadian Journal of Zoology* 78: 1791–1805.
- Moore, A., and Waring, C.P. 1996. Sublethal effects of the pesticide Diazinon on olfactory function in mature male Atlantic salmon parr. *Journal of Fish Biology* 48:758-775.

- Morgan, M.J., and Kiceniuk, J.W. 1990. Effect of fenitrothion on the foraging behavior of juvenile Atlantic salmon. *Environmental Toxicology and Chemistry* 9:489-495.
- Myers, J., Busack, C., Rawding, D., Marshall, A., Teel, D., Van Doornik, D.M., and Maher, M.T. 2006. Historical population structure of Pacific salmonids in the Willamette River and lower Columbia River basins. U.S. Dept. Commerce, NOAA Tech. Memo. NMFS-NWFSC-73, 311 p.
- Nelson, T. S., G. Ruggerone, H. Kim, R. Schaefer, and M. Boles. 2004. Juvenile Chinook migration, growth and habitat use in the Lower Green River, Duwamish River and Nearshore of Elliott Bay 2001-2003, Draft Report. King County Department of Natural Resources and Parks. Seattle, Washington.
- PSCCTC (Pacific Salmon Commission Chinook Technical Committee). 2002. Pacific Salmon Commission Joint Chinook Technical Committee Report: Annual Exploitation Rate Analysis and Model Calibration. Report TCCHINOOK (02)-3. Vancouver, British Columbia, Canada.
- Pauley, G.B., Risher, R., and Thomas, G.L. 1989. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Pacific Northwest) - sockeye salmon. U.S. Fish Wildl. Serv., Biol. Rep. 82(11.116). U.S. Army Corps of Engineers, TR EL-82-4, 22 p.
- Pess, G.R., Montgomery, D.R., Steel, E.A., Bilby, R.E., Feist, B.E., and Greenberg, H.M. 2002. Landscape characteristics, land use, and coho salmon (*Oncorhynchus kisutch*) abundance, Snohomish River, Wash., U.S.A. *Canadian Journal of Fisheries and Aquatic Sciences* 59: 613–623.
- Peterson, J.L., Jepson, P.C., and Jenkins, J.J. 2001a. A test system to evaluate the susceptibility of Oregon, USA, native stream invertebrates to triclopyr and carbaryl. *Environmental Toxicology and Chemistry* 20:2205-2214.
- Peterson, J.L., Jepson, P.C., and Jenkins, J.J. 2001b. Effect of varying pesticide exposure duration and concentration on the toxicity of carbaryl to two field-collected stream invertebrates, *Calineuria californica* (Plecoptera : Perlidae) and *Cinygma* sp (Ephemeroptera : Heptageniidae). *Environmental Toxicology and Chemistry* 20: 2215-2223.
- Quinn, T.P. 2005. The behavior and ecology of Pacific salmon and trout. University of Washington Press. Seattle, WA. 378pp.

- Ratner, S., Lande, R., and Roper, B.B. 1997. Population viability analysis of spring Chinook salmon in the South Umpqua River, Oregon. *Conservation Biology* 11:879-889.
- Relyea, R.A. 2005. The impact of insecticides and herbicides on the biodiversity and productivity of aquatic communities. *Ecological Applications* 15(2): 618–627.
- Roni, P., and Quinn, T.P. 1995. Geographic variation in size and age of North American Chinook salmon. *North American Journal of Fisheries Management* 15:325-345.
- Sakamoto, M., Chang, K.H., and Hanazato, T. 2006. Inhibition of development of anti-predator morphology in the small cladoceran *Bosmina* by an insecticide: impact of an anthropogenic chemical on prey-predator interactions. *Freshwater Biology* 51: 1974-1983.
- Sandahl, J.F., Baldwin, D.H., Jenkins, J.J., and Scholz, N.L. 2005. Comparative thresholds for acetylcholinesterase inhibition and behavioral impairment in coho salmon exposed to chlorpyrifos. *Environmental Toxicology and Chemistry* 24:136-145.
- Scholz, N.L., Truelove, N.K., French, B.L., Berejikian, B.A., Quinn, T.P., Casillas, E., and Collier, T.K. 2000. Diazinon disrupts antipredator and homing behaviors in Chinook salmon (*Oncorhynchus tshawytscha*). *Canadian Journal of Fisheries and Aquatic Sciences* 57(9): 1911–1918.
- Schulz, R. 2004. Field studies on exposure, effects, and risk mitigation of aquatic nonpoint-source insecticide pollution: A review. *Journal of Environmental Quality* 33:419-448.
- Schulz, R., and Liess, M. 1999. A field study of the effects of agriculturally derived insecticide input on stream macroinvertebrate dynamics. *Aquatic Toxicology* 46:155-176.
- Schulz, R., and Liess, M. 2001a. Acute and chronic effects of particle-associated fenvalerate on stream macroinvertebrates: A runoff simulation study using outdoor microcosms. *Archives of Environmental Contamination and Toxicology* 40: 481-488.
- Schulz, R., and Liess, M. 2001b. Toxicity of aqueous-phase and suspended particle-associated fenvalerate: Chronic effects after pulse-dosed exposure of *Limnephilus lunatus* (Trichoptera). *Environmental Toxicology and Chemistry* 20: 185-190.
- Schulz, R., Thiere, G., and Dabrowski, J.M. 2002. A combined microcosm and field approach to evaluate the aquatic toxicity of azinphosmethyl to stream communities. *Environmental Toxicology and Chemistry* 21:2172-2178.

- Stark, J.D., Banks, J.E., and Vargas, R. 2004. How risky is risk assessment: The role that life history strategies play in susceptibility of species to stress. *Proceedings of the National Academy of Sciences, USA* 101:732-736.
- Van den Brink, P.J., van Wijngaarden, R.P.A., Lucassen, W.G.H., Brock, T.C.M. and Leeuwangh, P. 1996. Effects of the insecticide Dursban 4E (active ingredient chlorpyrifos) in outdoor experimental ditches: II. Invertebrate community responses and recovery. *Environmental Toxicology and Chemistry* 15(7):1143-1153.
- Wallace, J.B., Huryn, A.D., and Lugthart, G.J. 1991. Colonization of a headwater stream during 3 years of seasonal insecticidal applications. *Hydrobiologia* 211: 65-76.
- Ward, S., Arthington, A.H., and Pusey, B.J. 1995. The effects of a chronic application of chlorpyrifos on the macroinvertebrate fauna in an outdoor artificial stream system – species responses. *Ecotoxicology and Environmental Safety* 30: 2-23.
- Waring, C.P., and Moore, A.P. 1997. Sublethal effects of a carbamate pesticide on pheromonal mediated endocrine function in mature male Atlantic salmon (*Salmo salar*) parr. *Fish Physiology and Biochemistry* 17:203-211.
- Weatherley, A.H., and Gill, H.S. 1995. Growth. Pages 103-158. in C. Groot, L. Margolis, and W.C. Clarke, editors. *Physiological Ecology of Pacific Salmon*. University of British Columbia Press, Vancouver, Canada.
- West, C.J., and Larkin, P.A. 1987. Evidence of size-selective mortality of juvenile sockeye salmon (*Oncorhynchus nerka*) in Babine Lake, British Columbia. *Canadian Journal of Fisheries and Aquatic Sciences* 44: 712-721.
- Zabel, R.W., and Achord, S. 2004. Relating size of juveniles to survival within and among populations of Chinook salmon. *Ecology* 85:795-806.

Appendix 1. Species and Population Annual Rates of Growth

Chinook Salmon

ESU	Population	$\lambda - H=0$	95% CI -lower	95% CI - upper
California Coastal	Eel River	N/A	N/A	N/A
	Redwood Creek	N/A	N/A	N/A
	Mad River	N/A	N/A	N/A
	Humboldt Bay tributaries	N/A	N/A	N/A
	Bear River	N/A	N/A	N/A
	Mattole River	N/A	N/A	N/A
	Tenmile to Gualala	N/A	N/A	N/A
	Russain River	N/A	N/A	N/A
Central Valley Spring - Run (Good et al., 2005 - 90% CI)	Butte Creek - spring run	1.300	1.060	1.600
	Deer Creek - spring run	1.170	1.040	1.350
	Mill Creek - spring run	1.190	1.000	1.470
Lower Columbia River (Good et al., 2005) (# = McElhany et al., 2007)	Youngs Bay	N/A	N/A	N/A
	Grays River - fall run	0.944	0.739	1.204
	Big Creek	N/A	N/A	N/A
	Elochoman River - fall run	1.037	0.813	1.323
	Clatskanie River #	0.990	0.824	1.189
	Mill, Abernathy, Germany Creeks - fall run	0.981	0.769	1.252
	Scappoose Creek	N/A	N/A	N/A
	Coweeman River - fall run	1.092	0.855	1.393
	Lower Cowlitz River - fall run	0.998	0.776	1.282
	Upper Cowlitz River - fall run	N/A	N/A	N/A
	Toutle River - fall run	N/A	N/A	N/A
	Kalamaha River - fall run	0.937	0.763	1.242
	Salmon Creek / Lewis River - fall run	0.984	0.771	1.256
	Clackamas River - fall run	N/A	N/A	N/A
	Washougal River - fall run	1.025	0.803	1.308
	Sandy River - fall run	N/A	N/A	N/A
	Lower Gorge tributaries	N/A	N/A	N/A
	Upper Gorge tributaries - fall run	0.959	0.751	1.224
	Hood River - fall run	N/A	N/A	N/A
	Big White Salmon River - fall run	0.963	0.755	1.229
	Sandy River - late fall run	0.943	0.715	1.243
	North Fork Lewis River - late fall run	0.968	0.756	1.204
	Upper Cowlitz River - spring run	N/A	N/A	N/A
	Cispus River	N/A	N/A	N/A
	Tilton River	N/A	N/A	N/A
	Toutle River - spring run	N/A	N/A	N/A
	Kalamaha River - spring run	N/A	N/A	N/A
	Lewis River - spring run	N/A	N/A	N/A
	Sandy River - spring run #	0.961	0.853	1.083
	Big White Salmon River - spring run	N/A	N/A	N/A
	Hood River - spring run	N/A	N/A	N/A

Chinook Salmon (continued)

ESU	Population	$\lambda - H=0$	95% CI -lower	95% CI - upper
Upper Columbia River Spring - Run (FCRPS)	Methow River	1.100	N/A	N/A
	Twisp River	N/A	N/A	N/A
	Chewuch River	N/A	N/A	N/A
	Lost / Early River	N/A	N/A	N/A
	Entiat River	0.990	N/A	N/A
	Wenatchee River	1.010	N/A	N/A
	Chiawawa River	N/A	N/A	N/A
	Nason River	N/A	N/A	N/A
	Upper Wenatchee River	N/A	N/A	N/A
	White River	N/A	N/A	N/A
	Little Wenatchee River	N/A	N/A	N/A
Puget Sound (only have λ where hatchery fish = native fish), (Good et al., 2005)	Nooksack - North Fork	0.750	0.680	0.820
	Nooksack - South Fork	0.940	0.880	0.990
	Lower Skagit	1.050	0.960	1.140
	Upper Skagit	1.050	0.990	1.110
	Upper Cascade	1.060	1.010	1.110
	Lower Sauk	1.010	0.890	1.130
	Upper Sauk	0.960	0.900	1.020
	Suiattle	0.990	0.930	1.050
	Stillaguamish - North Fork	0.920	0.880	0.960
	Stillaguamish - South Fork	0.990	0.970	1.010
	Skykomish	0.870	0.840	0.900
	Snoqualmie	1.000	0.960	1.040
	North Lake Washington	1.070	1.000	1.140
	Cedar	0.990	0.920	1.060
	Green	0.670	0.610	0.730
	White	1.160	1.100	1.220
	Puyallup	0.950	0.890	1.010
	Nisqually	1.040	0.970	1.110
	Skokomish	1.040	1.000	1.080
	Dosewallips	1.170	1.070	1.270
	Duckabush	N/A	N/A	N/A
	Hamma Hamma	N/A	N/A	N/A
	Mid Hood Canal	N/A	N/A	N/A
	Dungeness	1.090	0.980	1.200
	Elwha	0.950	0.840	1.060
Sacramento River Winter - Run (Good, 2005 - 90% CI)	Sacramento River - winter run	0.970	0.870	1.090

Chinook Salmon (continued)

ESU	Population	$\lambda - H=0$	95% CI -lower	95% CI - upper
Snake River Fall - Run (Good, 2005)	Lower Snake River	1.024	N/A	N/A
Snake River Spring/Summer - Run (FCRPS)	Tucannon River	1.000	N/A	N/A
	Wenaha River	1.100	N/A	N/A
	Wallowa River	N/A	N/A	N/A
	Lostine River	1.050	N/A	N/A
	Minam River	1.050	N/A	N/A
	Catherine Creek	0.970	N/A	N/A
	Upper Grande Ronde River	N/A	N/A	N/A
	South Fork Salmon River	1.110	N/A	N/A
	Secesh River	1.070	N/A	N/A
	Johnson Creek	N/A	N/A	N/A
	Big Creek Spring Run	1.090	N/A	N/A
	Big Creek Summer Run	1.090	N/A	N/A
	Loon Creek	N/A	N/A	N/A
	Marsh Creek	1.080	N/A	N/A
	Bear Valley / Elk Creek	1.100	N/A	N/A
	North Fork Salmon River	N/A	N/A	N/A
	Lemhi River	1.020	N/A	N/A
	Pahsimeroi River	1.080	N/A	N/A
	East Fork Salmon Spring Run	1.040	N/A	N/A
	East Fork Salmon Summer Run	1.040	N/A	N/A
	Yankee Fork Spring Run	N/A	N/A	N/A
	Yankee Fork Summer Run	N/A	N/A	N/A
	Valley Creek Spring Run	N/A	N/A	N/A
	Valley Creek Summer Run	N/A	N/A	N/A
	Upper Salmon Spring Run	1.060	N/A	N/A
	Upper Salmon Summer Run	1.060	N/A	N/A
	Alturas Lake Creek	N/A	N/A	N/A
	Imnaha River	1.050	N/A	N/A
	Big Sheep Creek	N/A	N/A	N/A
	Lick Creek	N/A	N/A	N/A
Upper Willamette River (McElhany et al., 2007)	Clackamas River	0.967	0.849	1.102
	Molalla River	N/A	N/A	N/A
	North Santiam River	N/A	N/A	N/A
	South Santiam River	N/A	N/A	N/A
	Calapooia River	N/A	N/A	N/A
	McKenzie River	0.927	0.761	1.129
	Middle Fork Willamette River	N/A	N/A	N/A
	Upper Fork Willamette River	N/A	N/A	N/A

Chum Salmon

ESU	Population	$\lambda - H=0$	95% CI -lower	95% CI - upper
Columbia River	Youngs Bay	N/A	N/A	N/A
	Grays River	0.954	0.855	1.064
	Big Creek	N/A	N/A	N/A
	Elochoman River	N/A	N/A	N/A
	Clatskanie River	N/A	N/A	N/A
	Mill, Abernathy and German Creeks	N/A	N/A	N/A
	Scappoose Creek	N/A	N/A	N/A
	Cowlitz River	N/A	N/A	N/A
	Kalama River	N/A	N/A	N/A
	Lewis River	N/A	N/A	N/A
	Salmon Creek	N/A	N/A	N/A
	Clackamas River	N/A	N/A	N/A
	Sandy River	N/A	N/A	N/A
	Washougal River	N/A	N/A	N/A
	Lower Gorge tributaries	0.984	0.883	1.096
	Upper Gorge tributaries	N/A	N/A	N/A
Hood Canal Summer - Run (only have λ where hatchery fish reproductive potential = native fish; Good et. al., 2005)	Jimmycomelately Creek	0.850	0.690	1.010
	Salmon / Snow Creeks	1.230	1.130	1.330
	Big / Little Quilcene rivers	1.390	1.170	1.610
	Lilliwaup Creek	1.190	0.750	1.630
	Hamma Hamma River	1.300	1.110	1.490
	Duckabush River	1.100	0.930	1.270
	Dosewallips River	1.170	0.930	1.410
	Union River	1.150	1.050	1.250
	Chimacum Creek	N/A	N/A	N/A
	Big Beef Creek	N/A	N/A	N/A
	Dewetto Creek	N/A	N/A	N/A

Coho Salmon

ESU	Population	$\lambda - H=0$	95% CI -lower	95% CI - upper
Central California Coast	Ten Mile River	N/A	N/A	N/A
	Noyo River	N/A	N/A	N/A
	Big River	N/A	N/A	N/A
	Navarro River	N/A	N/A	N/A
	Garcia River	N/A	N/A	N/A
	Other Mendacino County Rivers	N/A	N/A	N/A
	Gualala River	N/A	N/A	N/A
	Russain River	N/A	N/A	N/A
	Other Sonoma County Rivers	N/A	N/A	N/A
	Martin County	N/A	N/A	N/A
	San Mateo County	N/A	N/A	N/A
	Santa Cruz County	N/A	N/A	N/A
	San Lorenzo River	N/A	N/A	N/A
Lower Columbia River (Good et al., 2005)	Youngs Bay	N/A	N/A	N/A
	Grays River	N/A	N/A	N/A
	Elochoman River	N/A	N/A	N/A
	Clatskanie River	N/A	N/A	N/A
	Mill, Abernathy, Germany Creeks	N/A	N/A	N/A
	Scappose Creek	N/A	N/A	N/A
	Cispus River	N/A	N/A	N/A
	Tilton River	N/A	N/A	N/A
	Upper Cowlitz River	N/A	N/A	N/A
	Lower Cowlitz River	N/A	N/A	N/A
	North Fork Toutle River	N/A	N/A	N/A
	South Fork Toutle River	N/A	N/A	N/A
	Coweeman River	N/A	N/A	N/A
	Kalama River	N/A	N/A	N/A
	North Fork Lewis River	N/A	N/A	N/A
	East Fork Lewis River	N/A	N/A	N/A
	Upper Clackamas River	1.028	0.898	1.177
	Lower Clackamas River	N/A	N/A	N/A
	Salmon Creek	N/A	N/A	N/A
	Upper Sandy River	1.102	0.874	1.172
	Lower Sandy River	N/A	N/A	N/A
	Washougal River	N/A	N/A	N/A
	Lower Columbia River gorge tributaries	N/A	N/A	N/A
	White Salmon	N/A	N/A	N/A
	Upper Columbia River gorge tributaries	N/A	N/A	N/A
	Hood River	N/A	N/A	N/A

Coho Salmon (continued)

ESU	Population	$\lambda - H=0$	95% CI -lower	95% CI - upper
Southern Oregon and Northern California Coast	Southern Oregon and Northern California Coast	N/A	N/A	N/A
Oregon Coast	Necanicum	N/A	N/A	N/A
	Nehalem	N/A	N/A	N/A
	Tillamook	N/A	N/A	N/A
	Nestucca	N/A	N/A	N/A
	Siletz	N/A	N/A	N/A
	Yaquina	N/A	N/A	N/A
	Alsea	N/A	N/A	N/A
	Siuslaw	N/A	N/A	N/A
	Umpqua	N/A	N/A	N/A
	Coos	N/A	N/A	N/A
	Coquille	N/A	N/A	N/A

Sockeye Salmon

ESU	Population	$\lambda - H=0$	95% CI -lower	95% CI - upper
Ozette Lake	Ozette Lake	N/A	N/A	N/A
Snake River	Snake River	N/A	N/A	N/A

Steelhead

DPS	Population	$\lambda - H=0$	95% CI -lower	95% CI - upper
Central California Coast (Good et al., 2005)	Russain River	N/A	N/A	N/A
	Lagunitas	N/A	N/A	N/A
	San Gregorio	N/A	N/A	N/A
	Waddell Creek	N/A	N/A	N/A
	Scott Creek	N/A	N/A	N/A
	San Vicente Creek	N/A	N/A	N/A
	San Lorenzo River	N/A	N/A	N/A
	Soquel Creek	N/A	N/A	N/A
	Aptos Creek	N/A	N/A	N/A
California Central Valley (Good et al., 2005)	Sacramento River	0.950	0.900	1.020
Lower Columbia River (Good et al., 2005)	Cispus River	N/A	N/A	N/A
	Tilton River	N/A	N/A	N/A
	Upper Cowlitz River	N/A	N/A	N/A
	Lower Cowlitz River	N/A	N/A	N/A
	Coweeman River	0.908	0.792	1.041
	South Fork Toutle River	0.938	0.830	1.059
	North Fork Toutle River	1.062	0.915	1.233
	Kalama River - winter run	1.010	0.9130	1.117
	Kalama River - summer run	0.981	0.889	1.083
	North Fork Lewis River - winter run	N/A	N/A	N/A
	North Fork Lewis River - summer run	N/A	N/A	N/A
	East Fork Lewis River - winter run	N/A	N/A	N/A
	East Fork Lewis River - summer run	N/A	N/A	N/A
	Salmon Creek	N/A	N/A	N/A
	Washougal River - winter run	N/A	N/A	N/A
	Washougal River - summer run	1.003	0.884	1.138
	Clackamas River	0.971	0.901	1.047
	Sandy River	0.945	0.850	1.051
	Lower Columbia gorge tributaries	N/A	N/A	N/A
	Upper Columbia gorge tributaries	N/A	N/A	N/A

Steelhead (continued)

DPS	Population	$\lambda - H=0$	95% CI -lower	95% CI - upper
Middle Columbia River (Good et al., 2005)	Klickitat River	N/A	N/A	N/A
	Yakima River	1.009	N/A	N/A
	Fifteenmile Creek	0.981	N/A	N/A
	Deschutes River	1.022	N/A	N/A
	John Day - upper main stream	0.975	N/A	N/A
	John Day - lower main stream	0.981	N/A	N/A
	John Day - upper north fork	1.011	N/A	N/A
	John Day - lower north fork	1.013	N/A	N/A
	John Day - middle fork	0.966	N/A	N/A
	John Day - south fork	0.967	N/A	N/A
	Umatilla River	1.007	N/A	N/A
	Touchet River	0.961	N/A	N/A
Northern California (Good et al., 2005)	Redwood Creek	N/A	N/A	N/A
	Mad River - winter run	1.000	0.930	1.050
	Eel River - summer run	0.980	0.930	1.040
	Mattole River	N/A	N/A	N/A
	Ten Mile river	N/A	N/A	N/A
	Noyo River	N/A	N/A	N/A
	Big River	N/A	N/A	N/A
	Navarro River	N/A	N/A	N/A
	Garcia River	N/A	N/A	N/A
	Gualala River	N/A	N/A	N/A
	Other Humboldt County streams	N/A	N/A	N/A
	Other Mendocino County streams	N/A	N/A	N/A
Puget Sound*	Puget Sound	N/A	N/A	N/A
Snake River (Good et al., 2005)	Tucannon River	0.886	N/A	N/A
	Lower Granite run	0.994	N/A	N/A
	Snake A run	0.998	N/A	N/A
	Snake B run	0.927	N/A	N/A
	Asotin Creek	N/A	N/A	N/A
	Upper Grande Ronde River	0.967	N/A	N/A
	Joseph Creek	1.069	N/A	N/A
	Imnaha River	1.045	N/A	N/A
	Camp Creek	1.077	N/A	N/A
South-Central California Coast	South-Central California Coast	N/A	N/A	N/A
Southern California	Santa Ynez River	N/A	N/A	N/A
	Ventura River	N/A	N/A	N/A
	Matilija River	N/A	N/A	N/A
	Creek River	N/A	N/A	N/A
	Santa Clara River	N/A	N/A	N/A

Steelhead (continued)

DPS	Population	$\lambda - H=0$	95% CI -lower	95% CI - upper
Upper Columbia River (Good et al., 2005)	Wenatchee / Entiat Rivers	1.067	N/A	N/A
	Methow / Okanogan Rivers	1.086	N/A	N/A
Upper Willamette River (McElhany et al., 2007)	Molalla River	0.988	0.790	1.235
	North Santiam River	0.983	0.789	1.231
	South Santiam River	0.976	0.855	1.114
	Calapooia River	1.023	0.743	1.409

Appendix 2: Abbreviations / Acronyms

7-DADMax	7-day average of the daily maximum
ACA	Alternative Conservation Agreement
AChE	acetylcholinesterase
a.i.	active ingredient
APEs	alkylphenol ethoxylates
APHIS	U.S. Department of Agriculture Animal Plant and Health Inspection Service
BE	Biological Evaluation
BEAD	Biological and Economic Analysis Division
BLM	Bureau of Land Management
BMP	Best Management Practices
BOR	Bureau of Reclamation
BOR	Bureau of Reclamation
BPA	Bonneville Power Administration
BRT	Biological Review Team (NOAA Fisheries)
BY	Brood Years
CAISMP	California Aquatic Invasive Species Management Plan
CALFED	CALFED Bay-Delta Program (California Resource Agency)
CBFWA	Columbia Basin Fish and Wildlife Authority
CBI	Confidential Business Information
CC	California Coastal
CCC	Central California Coast
CCV	Central California Valley
CDPR	California Department of Pesticide Regulation
CHART	Critical Habitat Assessment Review Team
CIDMP	Comprehensive Irrigation District Management Plan
CFR	Code of Federal Regulations
cfs	cubic feet per second

CDFG	California Department of Fish and Game
Corps	U.S. Department of the Army Corps of Engineers
CSOs	combined sewer/stormwater overflows
CSWP	California State Water Project
CURES	Coalition for Urban/Rural Environmental Stewardship
CVP	Central Valley Projects
CVRWQCB	Central Valley Regional Water Quality Control Board
CWA	Clean Water Act
d	day
DCI	Date Call-Ins
DDD	Dichloro Diphenyl Dichloroethane
DDE	Diphenyl Dichlorethylene
DDT	Dichloro Diphenyl Trichloroethane
DER	Data Evaluation Review
DEQ	Oregon Department of Environmental Quality
DIP	Demographically Independent Population
DOE	Washington State Department of Ecology
DPS	Distinct Population Segment
EC	Emulsifiable Concentrate Pesticide Formulation
EC ₅₀	Median Effect Concentration
EEC	Estimated Environmental Concentration
EFED	Environmental Fate and Effects Division
EIM	Environmental Information Management
EPA	U.S. Environmental Protection Agency
ESPP	Endangered Species Protection Program
ESA	Endangered Species Act
ESU	Evolutionarily Significant Unit
EU	European Union
EXAMS	Tier II Surface Water Computer Model
FERC	Federal Energy Regulatory Commission

FCRPS	Federal Columbia River Power System
FFDCA	Federal Food and Drug Cosmetic Act
FIFRA	Federal Insecticide, Fungicide, and Rodenticide Act
FQPA	Food Quality Protection Act
ft	feet
GENEEC	Generic Estimated Exposure Concentration
h	hour
HCP	Habitat Conservation Plan
HSRG	Hatchery Scientific Review Group
HUC	Hydrological Unit Code
IBI	Indices of Biological Integrity
ICTRT	Interior Columbia Technical Recovery Team
ILWP	Irrigated Lands Waiver Program
IPCC	Intergovernmental Panel on Climate Change
IREC	Interim Re-registration Decision
LCFRB	Lower Columbia Fish Recovery Board
ISG	Independent Science Group
ITS	Incidental Take Statement
km	kilometer
Lbs	Pounds
LC ₅₀	Median Lethal Concentration.
LCR	Lower Columbia River
LOAEC	Lowest Observed Adverse Effect Concentration.
LOEL	Lowest Observed Adverse Effect level
LOC	Level of Concern
LOEC	Lowest Observed Effect Concentration
LOQ	Limit of Quantification
LWD	Large Woody Debris
m	meter
MCR	Middle Columbia River

mg/L	milligrams per liter
MOA	Memorandum of Agreement
MPG	Major Population Group
MRID	Master Record Identification Number
MTBE	Methyl tert-butyl ether
NASA	National Aeronautics and Space Administration
NAWQA	U.S. Geological Survey National Water-Quality Assessment
NC	Northern California
NEPA	National Environmental Protection Agency
NLCD	Natural Land Cover Data
NP	Nonylphenol
NPDES	National Pollutant Discharge Elimination System
NPS	National Parks Services
NRCS	Natural Resources Conservation Service
NWS	National Weather Service
NEPA	National Environmental Policy Act
NMA	National Mining Association
NMC	<i>N</i> -methyl carbamates
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOAEC	No Observed Adverse Effect Concentration
NPDES	National Pollution Discharge Eliminating System
NPIRS	National Pesticide Information Retrieval System
NRC	National Research Council
OC	Oregon Coast
ODFW	Oregon Division of Fish and Wildlife
OP	Organophosphates
Opinion	Biological Opinion
OPP	EPA Office of Pesticide Program
PAH	polyaromatic hydrocarbons

PBDEs	polybrominated diphenyl ethers
PCBs	polychlorinated biphenyls
PCEs	primary constituent elements
POP	Persistent Organic Pollutants
ppb	Parts Per Billion
PPE	Personal Protection Equipment
PSP	Pesticide Stewardship Partnerships
PSAMP	Puget Sound Assessment and Monitoring Program
PSAT	Puget Sound Action Team
PRIA	Pesticide Registration Improvement Act
PRZM	Pesticide Root Zone Model
PUR	Pesticide Use Reporting
QA/QC	Quality Assurance/Quality Control
RED	Re-registration Eligibility Decision
REI	Restricted Entry Interval
RPA	Reasonable and Prudent Alternatives
RPM	reasonable and prudent measures
RQ	Risk Quotient
SAP	Scientific Advisory Panel
SAR	smolt-to-adult return rate
SASSI	Salmon and Steelhead Stock Inventory
SC	Southern California
S-CCC	South-Central California Coast
SONCC	Southern Oregon Northern California Coast
SLN	Special Local Need (Registrations under Section 24(c) of FIFRA)
SR	Snake River
TCE	Trichloroethylene
TCP	3,5,6-trichloro-2-pyridinal
TGAI	Technical Grade Active Ingredient
TIE	Toxicity Identification Evaluation

TMDL	Total Maximum Daily Load
TRT	Technical Recovery Team
UCR	Upper Columbia River
USFS	United States Forest Service
USC	United States Code
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
UWR	Upper Willamette River
VOC	Volatile Organic Compounds
VSP	Viable Salmonid Population
WDFW	Washington Department of Fish and Wildlife
WLCRTRT	Willamette/Lower Columbia River Technical Review Team
WQS	Water Quality Standards
WWTIT	Western Washington Treaty Indian Tribes
WWTP	Wastewater Treatment Plant

Appendix 3: Glossary

303(d) waters Section 303 of the federal Clean Water Act requires states to prepare a list of all surface waters in the state for which beneficial uses – such as drinking, recreation, aquatic habitat, and industrial use - are impaired by pollutants. These are water quality limited estuaries, lakes, and streams that do not meet the state’s surface water quality standards and are not expected to improve within the next two years. After water bodies are put on the 303(d) list they enter into a Total Maximum Daily Load Clean Up Plan.

Active ingredient	The component(s) that kills or otherwise affects the pest. A.i.s are always listed on the label (FIFRA 2(a)).
Adulticide	A compound that kills the adult life stage of the pest insect.
Anadromous Fish	Species that are hatched in freshwater migrate to and mature in salt water and return to freshwater to spawn.
Adjuvant	A compound that aides the operation or improves the effectiveness of a pesticide.
Alevin	Life history stage of a salmonid immediately after hatching and before the yolk-sac is absorbed. Alevins usually remain buried in the gravel in or near the egg nest (redd) until their yolk sac is absorbed when they swim up and enter the water column.
Anadromy	The life history pattern that features egg incubation and early juvenile development in freshwater migration to sea water for adult development, and a return to freshwater for spawning.

Assessment Endpoint	Explicit expression of the actual ecological value that is to be protected (<i>e.g.</i> , growth of juvenile salmonids).
Bioaccumulation	Accumulation through the food chain (<i>i.e.</i> , consumption of food, water/sediment) or direct water and/or sediment exposure.
Bioconcentration	Uptake of a chemical across membranes, generally used in reference to waterborne exposures.
Biomagnification	Transfer of chemicals via the food chain through two or more trophic levels as a result of bioconcentration and bioaccumulation.
Degradates	New compounds formed by the transformation of a pesticide by chemical or biological reactions.
Distinct Population Segment	A listable entity under the ESA that meets tests of discreteness and significance according to USFWS and NMFS policy. A population is considered distinct (and hence a “species” for purposes of conservation under the ESA) if it is discrete from an significant to the remainder of its species based n factors such as physical, behavioral, or genetic characteristics, it occupies an unusual or unique ecological setting, or its loss would represent a significant gap in the species’ range.
Escapement	The number of fish that survive to reach the spawning grounds or hatcheries. The escapement plus the number of fish removed by harvest form the total run size.
Evolutionarily Significant Unit	A group of Pacific salmon or steelhead trout that is (1) substantially reproductively isolated from other conspecific

units and (2) represent an important component of the evolutionary legacy of the species.

Fall Chinook Salmon	This salmon stock returns from the ocean in late summer and early fall to head upriver to its spawning grounds, distinguishing it from other stocks which migrate in different seasons.
Fate	Dispersal of a material in various environmental compartments (sediment, water air, biota) as a result of transport, transformation, and degradation.
Flowable	A pesticide formulation that can be mixed with water to form a suspension in a spray tank.
Fry	Stage in salmonid life history when the juvenile has absorbed its yolk sac and leaves the gravel of the redd to swim up into the water column. The fry stage follows the alevin stage and in most salmonid species is followed by the parr, fingerling, and smolt stages. However, chum salmon juveniles share characteristics of both the fry and smolt stages and can enter sea water almost immediately after becoming fry.
Half-pounder	A life history trait of steelhead exhibited in the Rogue, Klamath, Mad, and Eel Rivers of southern Oregon and northern California. Following smoltification, half-pounders spend only 2-4 months in the ocean, then return to fresh water. They overwinter in fresh water and emigrate to salt water again the following spring. This is often termed a false spawning migration, as few half-pounders are sexually mature.
Hatchery	Salmon hatcheries use artificial procedures to spawn adults and raise the resulting progeny in fresh water for release into the natural environment, either directly from the hatchery or by transfer into another area. In some

cases, fertilized eggs are outplanted (usually in “hatch-boxes”), but it is more common to release fry or smolts.

Inert ingredients “an ingredient which is not active” (FIFRA 2(m)). It may be toxic or enhance the toxicity of the active ingredient.

Iteroparous Capable of spawning more than once before death

Jacks Male salmon that return from the ocean to spawn one or more years before full-sized adults return. For coho salmon in California, Oregon, Washington, and southern British Columbia, jacks are 2 years old, having spent only 6 months in the ocean, in contrast to adults, which are 3 years old after spending 1 ½ years in the ocean.

Jills Female salmon that return from the ocean to spawn one or more years before full-sized adult returns. For sockeye salmon in Oregon, Washington, and southern British Columbia, jills are 3 years old (age 1.1), having spent only one winter in the ocean in contrast to more typical sockeye salmon that are age 1.2, 1.32.2, or 2.3 on return.

Kokanee The self-perpetuating, non-anadromous form of *O. nerka* that occurs in balanced sex ration populations and whose parents, for several generations back, have spent their whole lives in freshwater.

Lambda Also known as Population growth rate, or the rate at which the abundance of fish in a population increases or decreases.

LRL Laboratory Reporting Level (USGS NAWQA data)- Generally equal to twice the yearly determined LT-MDL. The LRL controls false negative error. The probability of falsely reporting a non-detection for a sample that contained an

analyte at a concentration equal to or greater than the LRL is predicted to be less than or equal to 1 percent.

Major Population Group (MPG)	A group of salmonid populations that are geographically and genetically cohesive. The MPG is a level of organization between demographically independent populations and the ESU.
Main channel	The stream channel that includes the thalweg (longitudinal continuous deepest portion of the channel).
Metabolite	A transformation product resulting from metabolism.
Mode of Action	A series of key processes that begins with the interaction of a pesticide with a receptor site and proceeds through operational and anatomical changes in an organisms that result in sublethal or lethal effects.
Natural fish	A fish that is produced by parents spawning in a stream or lake bed, as opposed to a controlled environment such as a hatchery.
Nonylphenols	A type of APE and is an example of an adjuvant that may be present as an ingredient of a formulated product or added to a tank mix prior to application.
Off-channel habitat	Water bodies and/or inundated areas that are connected (accessible to salmonid juveniles) seasonally or annually to the main channel of a stream including but not limited to features such as side channels, alcoves, ox bows, ditches, and floodplains.
Parr	The stage in anadromous salmonid development between absorption of the yolk sac and transformation to smolt before migration seaward.

Persistence	The tendency of a compound to remain in its original chemical form in the environment.
Pesticide	Any substance or mixture of substances intended for preventing, destroying, repelling or mitigating any pest.
Reasonable and Prudent Alternative (RPA)	Recommended alternative actions identified during formal consultation that can be implemented in a manner consistent with the scope of the Federal agency's legal authority and jurisdiction, that are economically and technologically feasible, and that the Service believes would avoid the likelihood of jeopardizing the continued existence of the listed species or the destruction or adverse modification of designated critical habitat.
Redd	A nest constructed by female salmonids in streambed gravels where eggs are deposited and fertilization occurs.
Riparian area	Area with distinctive soils and vegetation between a stream or other body of water and the adjacent upland. It includes wetlands and those portions of flood plains and valley bottoms that support riparian vegetation.
Risk	The probability of harm from actual or predicted concentrations of a chemical in the aquatic environment – a scientific judgment.
Salmonid	Fish of the family <i>Salmonidae</i> , including salmon, trout, chars, grayling, and whitefish. In general usage, the term usually refers to salmon, trout, and chars.

SASSI	A cooperative program by WDFW and WWTIT to inventory and evaluate the status of Pacific salmonids in Washington State. The SASSI report is a series of publications from this program.
Semelparous	The condition in an individual organism of reproducing only once in a lifetime.
Smolt	A juvenile salmon or steelhead migrating to the ocean and undergoing physiological changes to adapt from freshwater to a saltwater environment.
Sublethal	Below the concentration that directly causes death. Exposure to sublethal concentrations of a material may produce less obvious effect on behavior, biochemical, and/or physiological function of the organism often leading to indirect death.
Surfactant	A substance that reduces the interfacial or surface tension of a system or a surface-active substance.
Synergism	A phenomenon in which the toxicity of a mixture of chemicals is greater than that which would be expected from a simple summation of the toxicities of the individual chemicals present in the mixture.
Technical Grade Active Ingredient (TGAI)	Pure or almost pure active ingredient. Available to formulators. Most toxicology data are developed with the TGAI. The percent AI is listed on all labels.
Technical Recovery Teams (TRT)	Teams convened by NOAA Fisheries to develop technical products related to recovery planning. TRTs are complemented by planning forums unique to specific states, tribes, or reigns, which use TRT and other technical products to identify recovery actions.

Teratogenic	Effects produced during gestation that evidence themselves as altered structural or functional processes in offspring.
Total Maximum Daily Load (TMDL)	defines how much of a pollutant a water body can tolerate (absorb) daily and remain compliant with applicable water quality standards. All pollutant sources in the watershed combined, including non-point sources, are limited to discharging no more than the TMDL.
Unique Mixture	A specific combination of 2 or more compounds, regardless of the presence of other compounds.
Viable Salmonid Population	An independent population of Pacific salmon or steelhead trout that has a negligible risk of extinction over a 100-year time frame. Viability at the independent population scale is evaluated based on the parameters of abundance, productivity, spatial structure, and diversity.
VSP Parameters	Abundance, productivity, spatial structure, and diversity. These describe characteristics of salmonid populations that are useful in evaluating population viability. See NOAA Technical Memorandum NMFS-NWFSC-, “Viable salmonid populations and the recovery of evolutionarily significant units,” McElhany et al., June 2000.
WDFW	Washington Department of Fish and Wildlife is a co-manager of salmonids and salmonid fisheries in Washington State with WWTIT and other fisheries groups. The agency was formed in the early 1990s by the combination of the Washington Department of Fisheries and the Washington Department of Wildlife.

- WWTIT Western Washington Treaty Indian Tribes is an organization of Native American tribes with treaty fishing rights recognized by the U.S. government. WWTIT is a co-manager of salmonids and salmonid fisheries in western Washington in cooperation with the WDFW and other fisheries groups.
- WQS “A water quality standard defines the water quality goals of a waterbody, or portion thereof, by designating the use or uses to be made of the water and by setting criteria necessary to protect public health or welfare, enhance the quality of water and serve the purposes of the Clean Water

Appendix 5: Co-occurrence Analysis for Integration and Synthesis

Our species viability assessment considers the spatial, temporal, and biological overlap of ESA-listed species with the stressors of the action. Where there is co-occurrence, salmonids may be exposed to and affected by the a.i. and its associated stressors.

Because pesticides are registered for specific uses, we determine where specific portions of the proposed action may be carried out based on the type of use. National Land Cover Database (NLCD) land use categories were used as a surrogate for use sites: cultivated crops or hay/pasture for a specific crop or crops; developed areas for residential and urban uses, pest control, and disease vector control; and managed forests for forestry applications. While cropping patterns may shift or lands may become fallow over a longer period of time, the NLCD dataset is the most relevant method of estimating exposure. As we cannot determine where a certain crop will be cultivated, we assume that any pesticide registered for use on an agricultural crop could be applied in an area defined as agricultural land use. We did consider differences in state regulations and SLN registrations, as well as general cropping trends for different basins.

However, we cannot determine where rights-of-way uses will occur based on land use information. We assume that rights-of-way will be concentrated in urban areas, but will also be present in rural areas as well. In more remote areas, roads and railroads are often situated along river valleys, sometimes in close proximity to the stream or river.

We used the GIS program ArcView to overlay the NLCD data on ESUs/DPSs range and distribution shapefiles to determine areas of potential co-occurrence of pesticide use and ESA-listed salmon. Species range shapefiles were developed by NMFS Northwest Regional Office. These files exist for every ESU and consist of polygons encompassing the hydrologic units where that species can be found. In some cases, these polygons include areas that are not currently occupied, but are accessible and are part of the historic range of the species. We also assessed distribution

data for each ESU/DPS. Distribution files were developed by the Northwest and Southwest regional offices in the process of identifying and designating critical habitat for 19 species in 2005.

The remaining ESUs/DPSs did not have existing distribution layers. They were created for this consultation by overlaying datasets from other sources with the NMFS range polygons. The data is largely presence/absence data collected by governmental agencies and university researchers. Information on Idaho, Oregon, and Washington species was compiled and presented by Streamnet (www.streamnet.org) while California data came from CalFish (www.calfish.org). Streams where fish were present within the range polygon were exported to a new distribution file. This method was used to create files for Snake River Fall-run Chinook salmon, Snake River Spring-run Chinook salmon, Sacramento River Winter-run Chinook salmon, Snake River sockeye salmon, Ozette Lake Sockeye salmon, Lower Columbia River Coho salmon, Southern Oregon Northern California Coho salmon, Central California Coast Coho salmon, and Puget Sound Steelhead salmon.

For all ESUs/DPSs, a 2.5 km “buffer” was created on each side of salmonid aquatic habitat. This distance was selected by the team as it is large enough to account for discrepancies between GIS layers due to channel alteration / migration, but not so large that it would encompass the entire range of an ESU. We expect pesticide applications in these areas are most relevant to concentrations experienced by salmonids via pesticide runoff and drift. If land in any of the relevant NLCD categories was within the buffer we determined that salmon and the a.i. could co-occur. Over the 15-year duration of the proposed action, we expect some individuals within each of the listed ESUs/DPSs in the action area will be exposed to these a.i.s during their life cycle. Given that these pesticides can be used across the landscape, and that temporal and spatial distribution of listed salmonids are both highly variable, we expect exposure is also highly variable among both individuals and populations of listed salmon.

Once co-occurrence is determined via GIS for each a.i., we evaluated the spatial and temporal extent of potential exposure for the ESU/DPS, given the life history of the species. In many cases, fish may be in the system for prolonged periods of time, and there is generally no specific seasonal restriction on application of pesticides. Additionally, species are made up of “runs” which spawn at

different times of the year. Thus, the spatial and biological overlap is of greater importance in analyzing this action than the temporal component.

We further considered the existing environmental mixtures, seasonally elevated water temperatures, and other factors which influence the survival of the species, such as loss of habitat features, hydropower and water management conditions, and invasive species or predators. Other important factors that were taken into consideration include location of federal land, railroad lines, and electrical transmission lines.

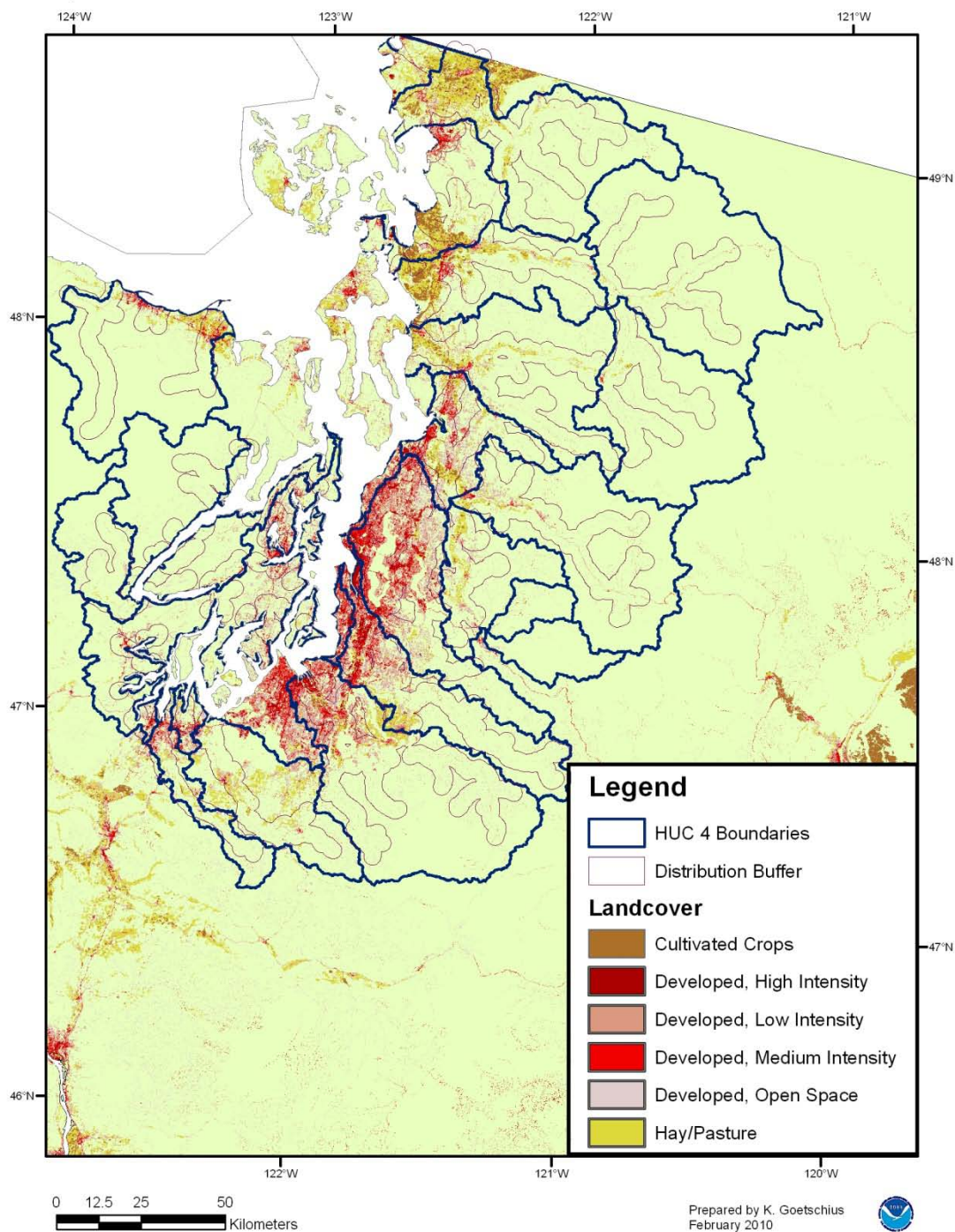
To illustrate the co-occurrence analysis process, this appendix includes two maps for each ESU/DPS. The first map shows the range of the ESU with each HUC 4 outlined in blue, the 2.5 km buffer in burgundy and relevant categories from the NLCD land use layer. This map aided in the Species analyses. The second map was used in the critical habitat analysis. For 19 of the species, conservation values have been assigned to the HUC 5 level units. In Idaho, Oregon, and Washington, these units are referred to as watersheds, while California uses the term “hydrological sub-area” or HSA. The Critical Habitat maps show either, (a) all designated HUC5s and their conservation values, or (b) the species map with the buffer removed. The exceptions to this are Snake River Fall-Run Chinook and Ozette Lake Sockeye, as they cover such small areas, and the two species for which critical habitat has not been designated (Columbia River Coho and Puget Sound Steelhead). These four species each only have one map.

The following species have conservation values assigned by HUC5:

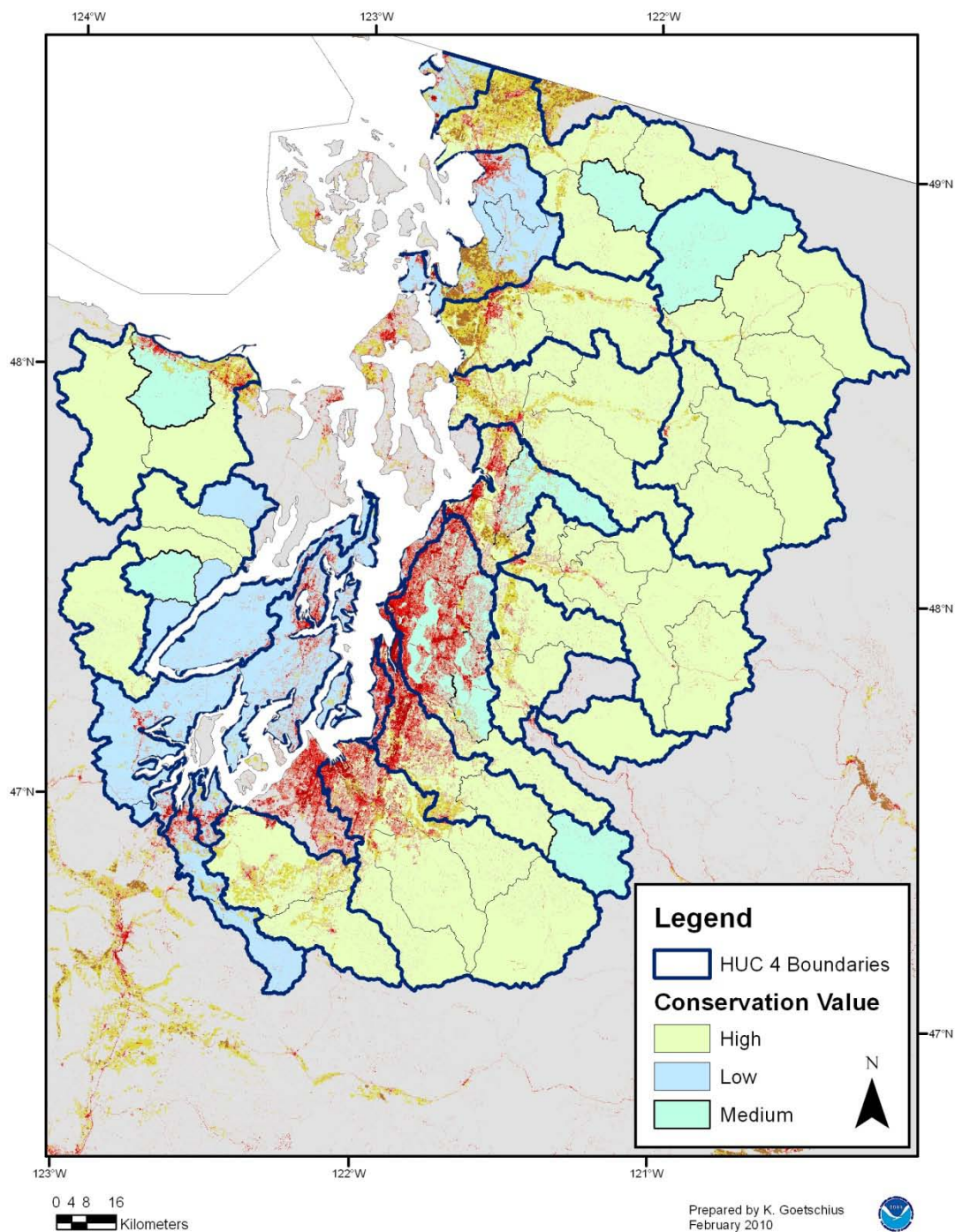
1. Puget Sound Chinook
2. Lower Columbia River Chinook
3. Upper Columbia River Spring Run Chinook
4. Upper Willamette River Chinook
5. California Coastal Chinook
6. Central Valley Spring Run Chinook
7. Columbia River Chum
8. Hood Canal Chum
9. Oregon Coast Coho
10. Lower Columbia River Steelhead
11. Middle Columbia River Steelhead

12. Upper Columbia River Steelhead
13. Upper Willamette River Steelhead
14. Snake River Steelhead
15. Northern California Steelhead
16. Central California Coast Steelhead
17. California Central Valley Steelhead
18. South-Central California Coast Steelhead
19. Southern California Steelhead

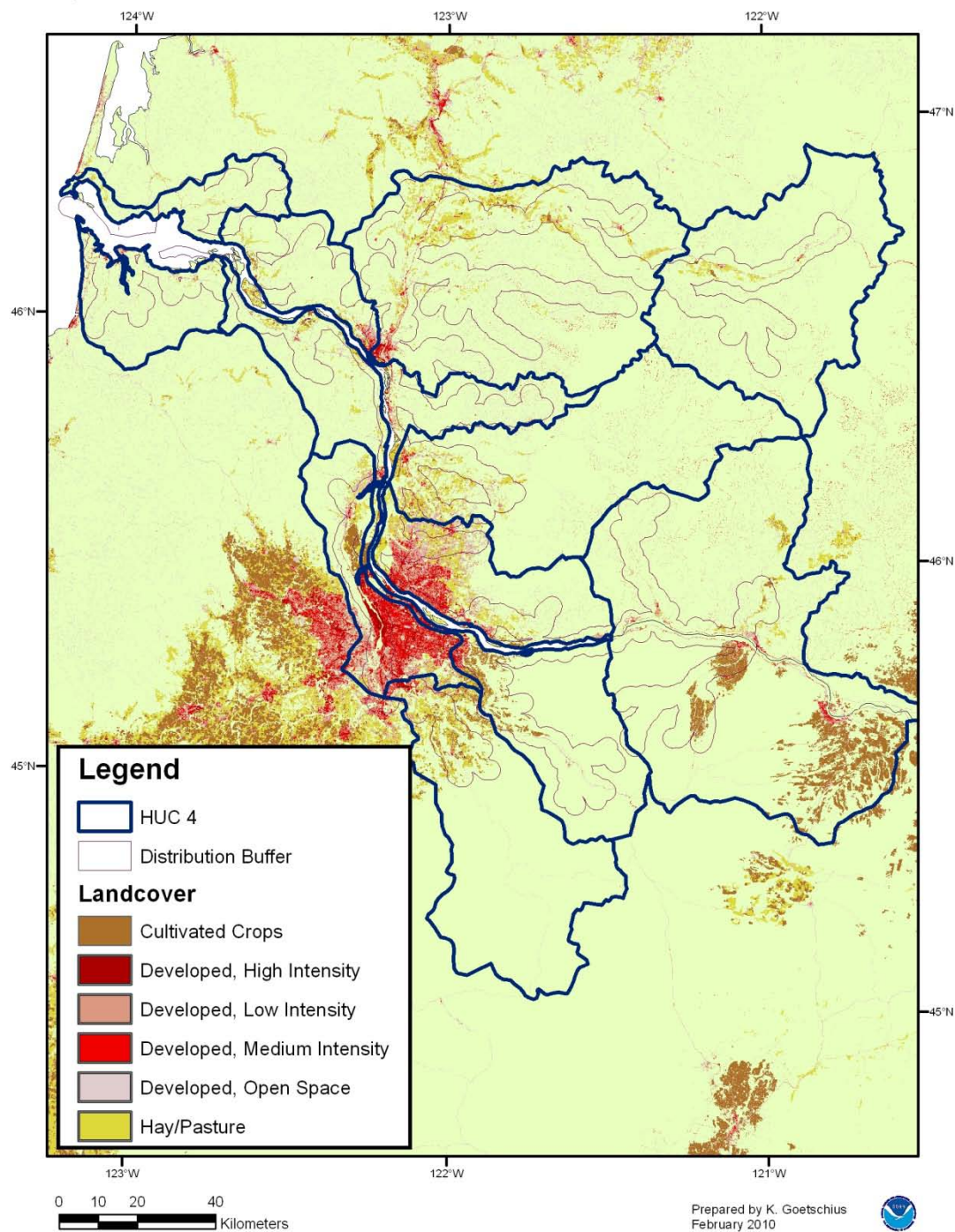
Puget Sound Chinook ESU Species Distribution



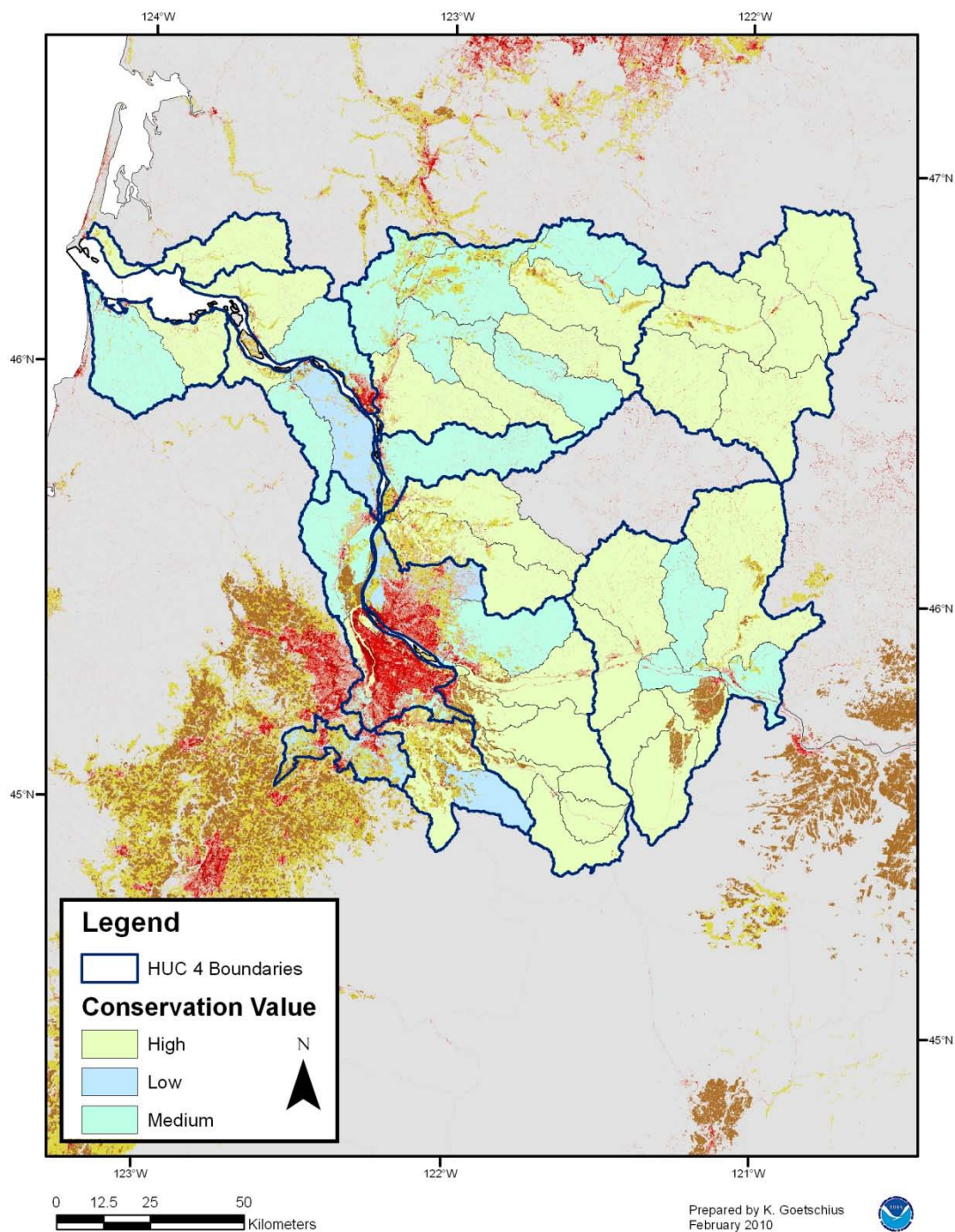
Puget Sound Chinook ESU Critical Habitat



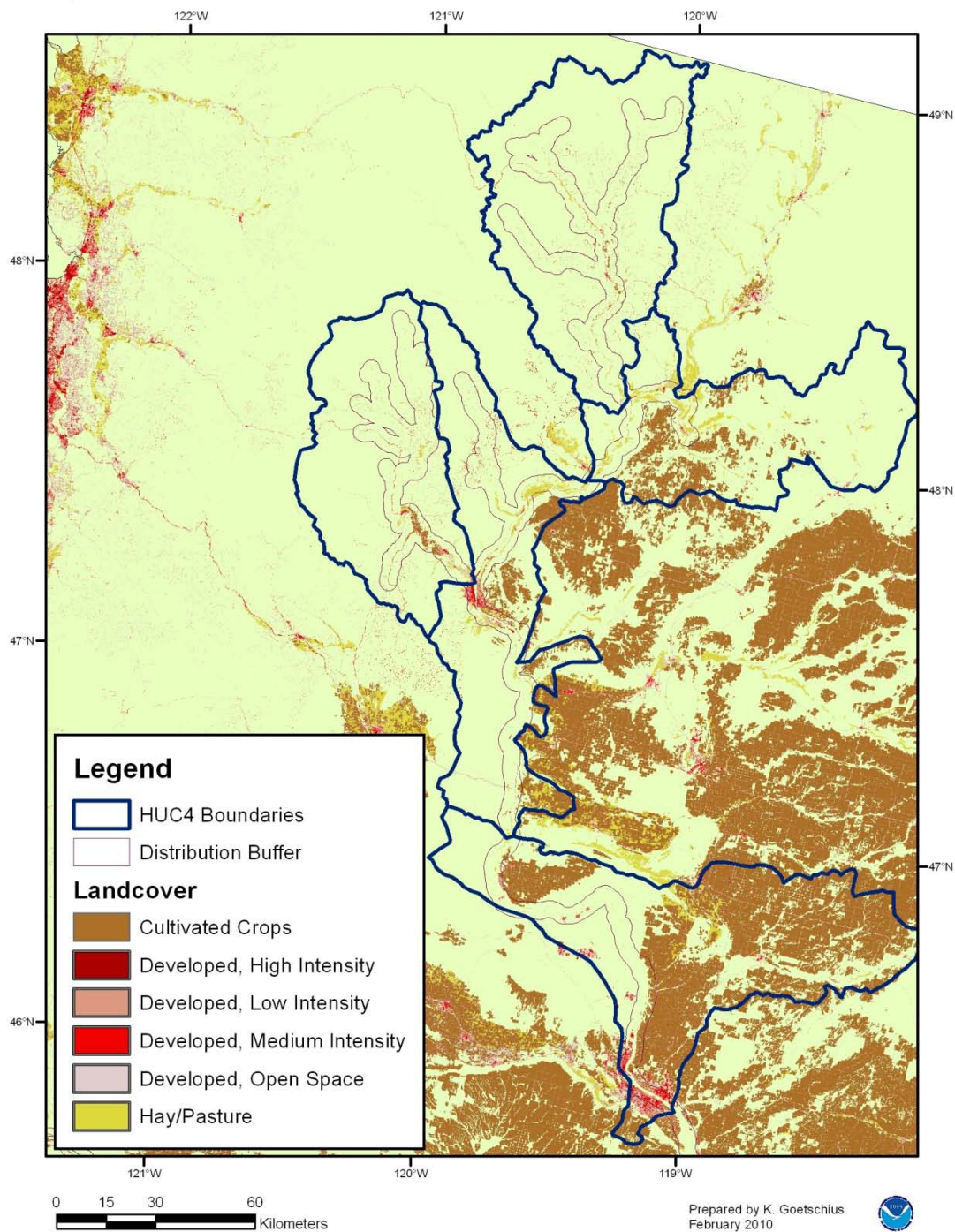
Lower Columbia River Chinook ESU Species Distribution



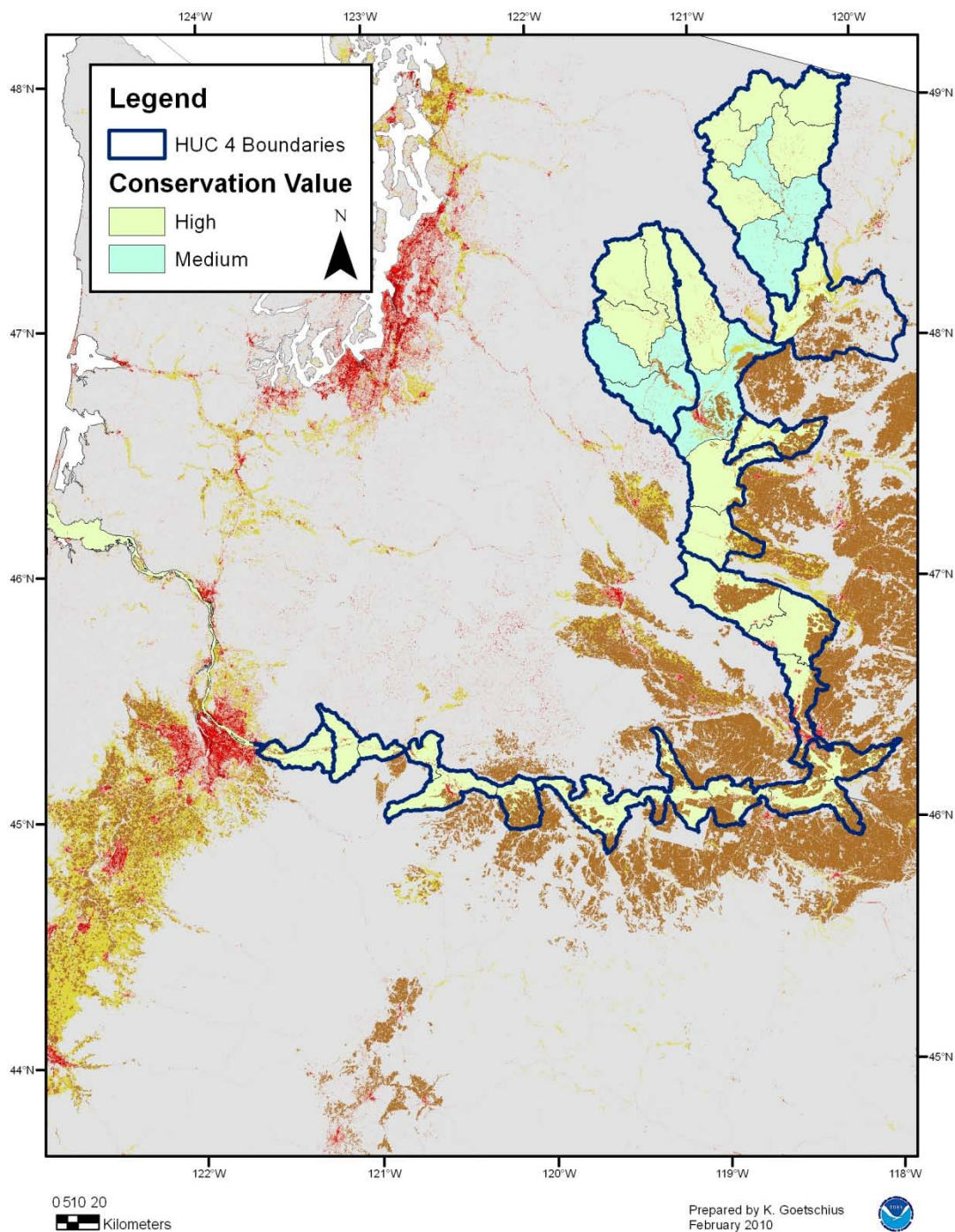
Lower Columbia River Chinook ESU Critical Habitat



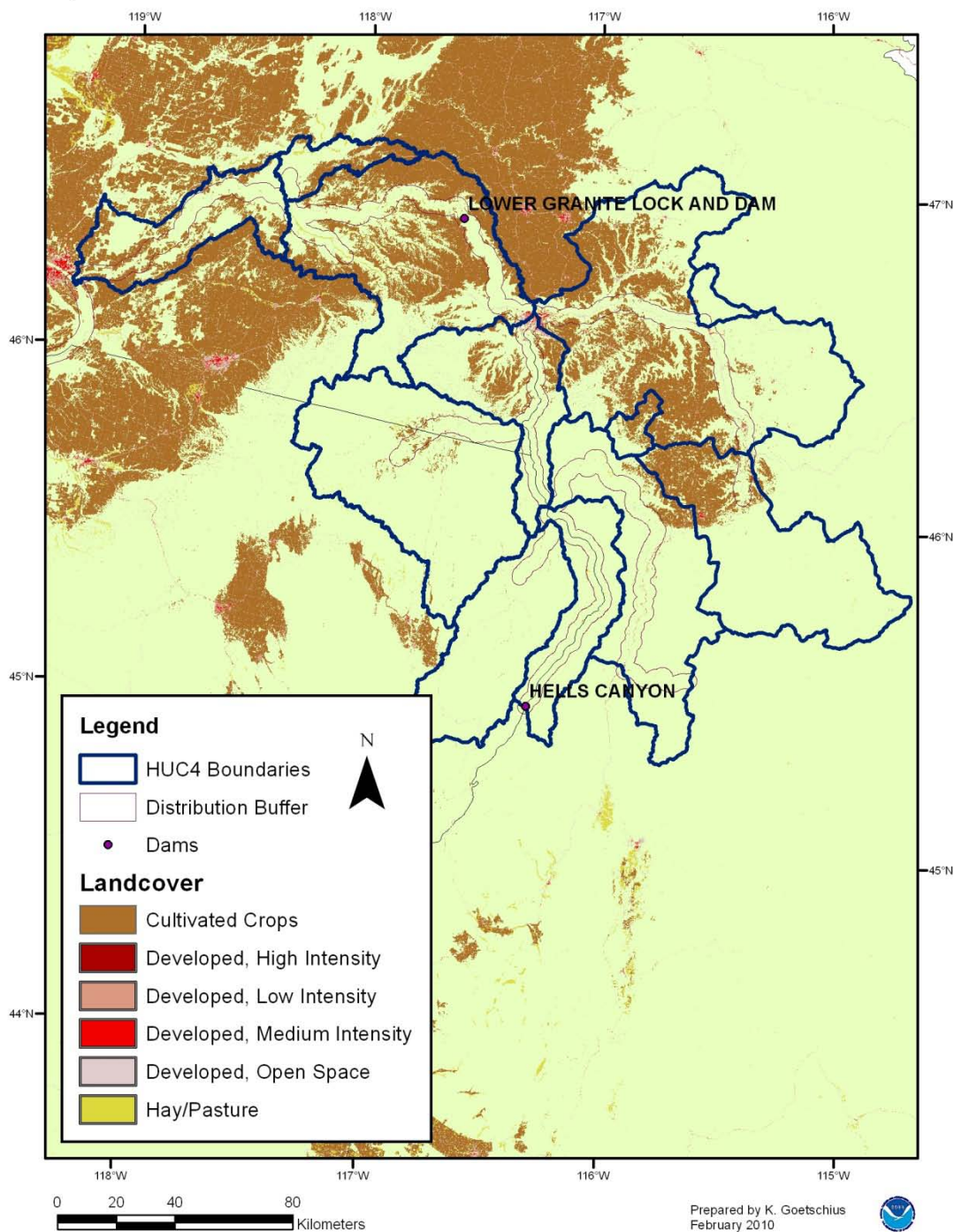
Upper Columbia Spring-Run Chinook ESU Species Distribution



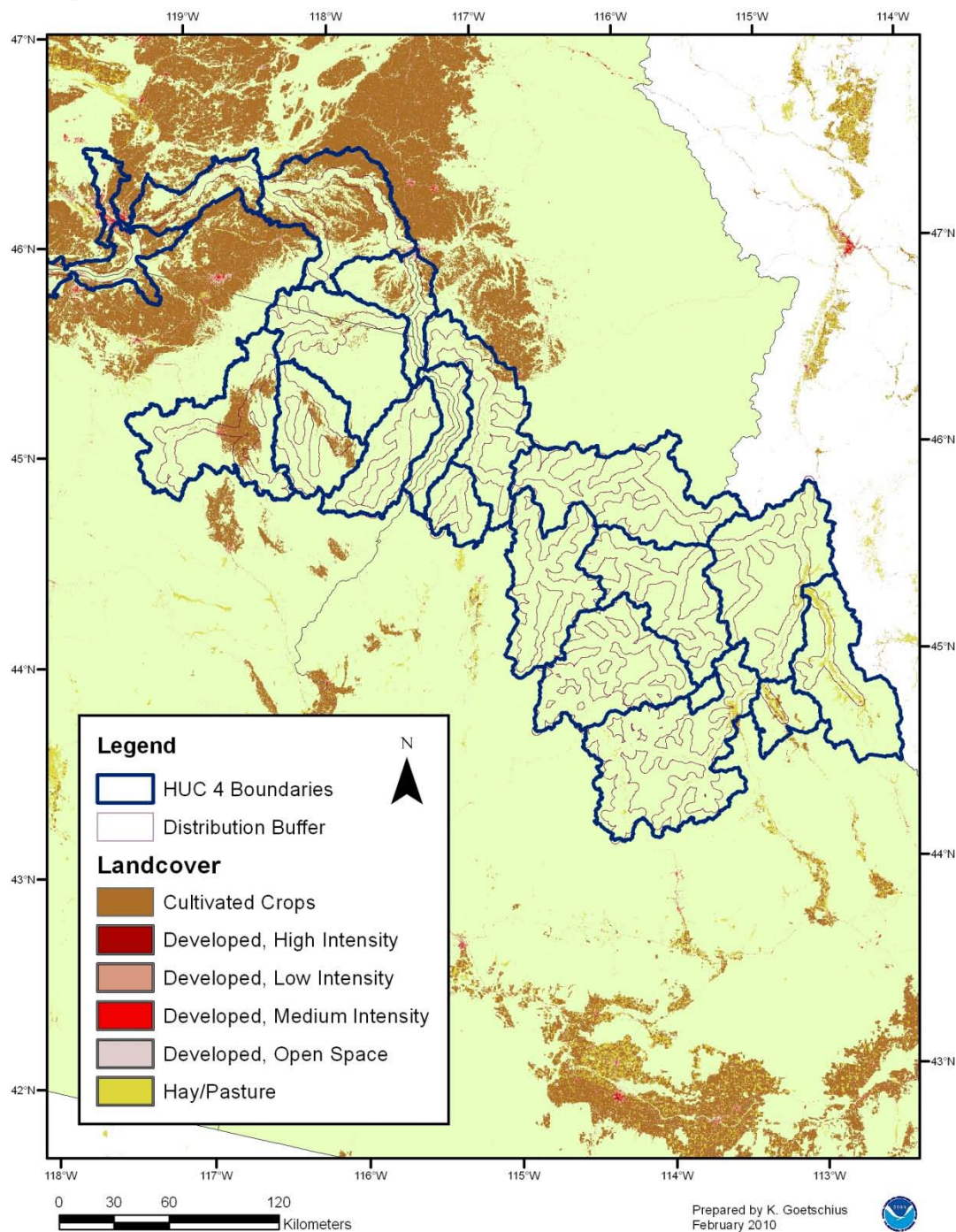
Upper Coulmbia River Chinook ESU Critical Habitat



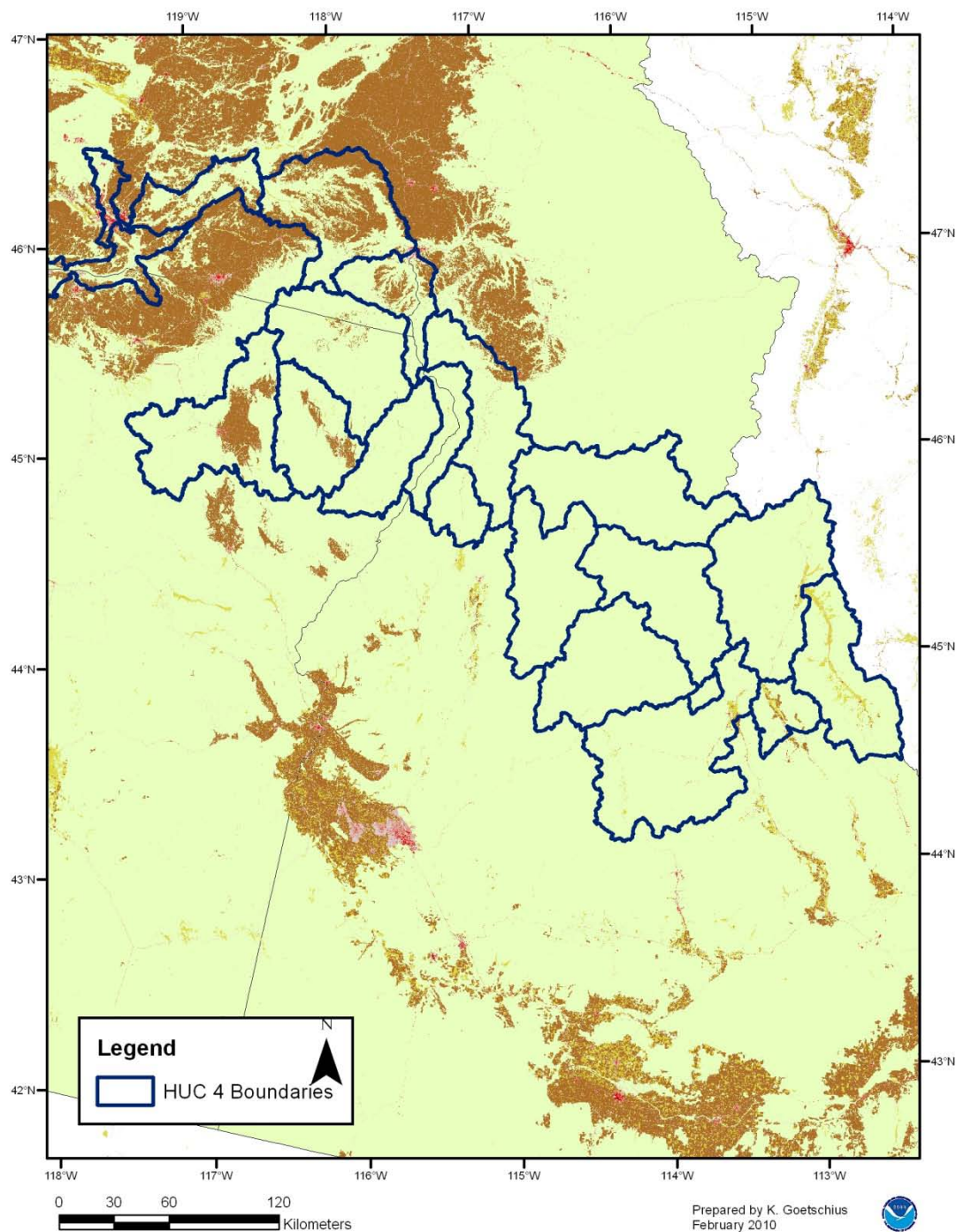
Snake River Fall Run Chinook ESU Species Distribution



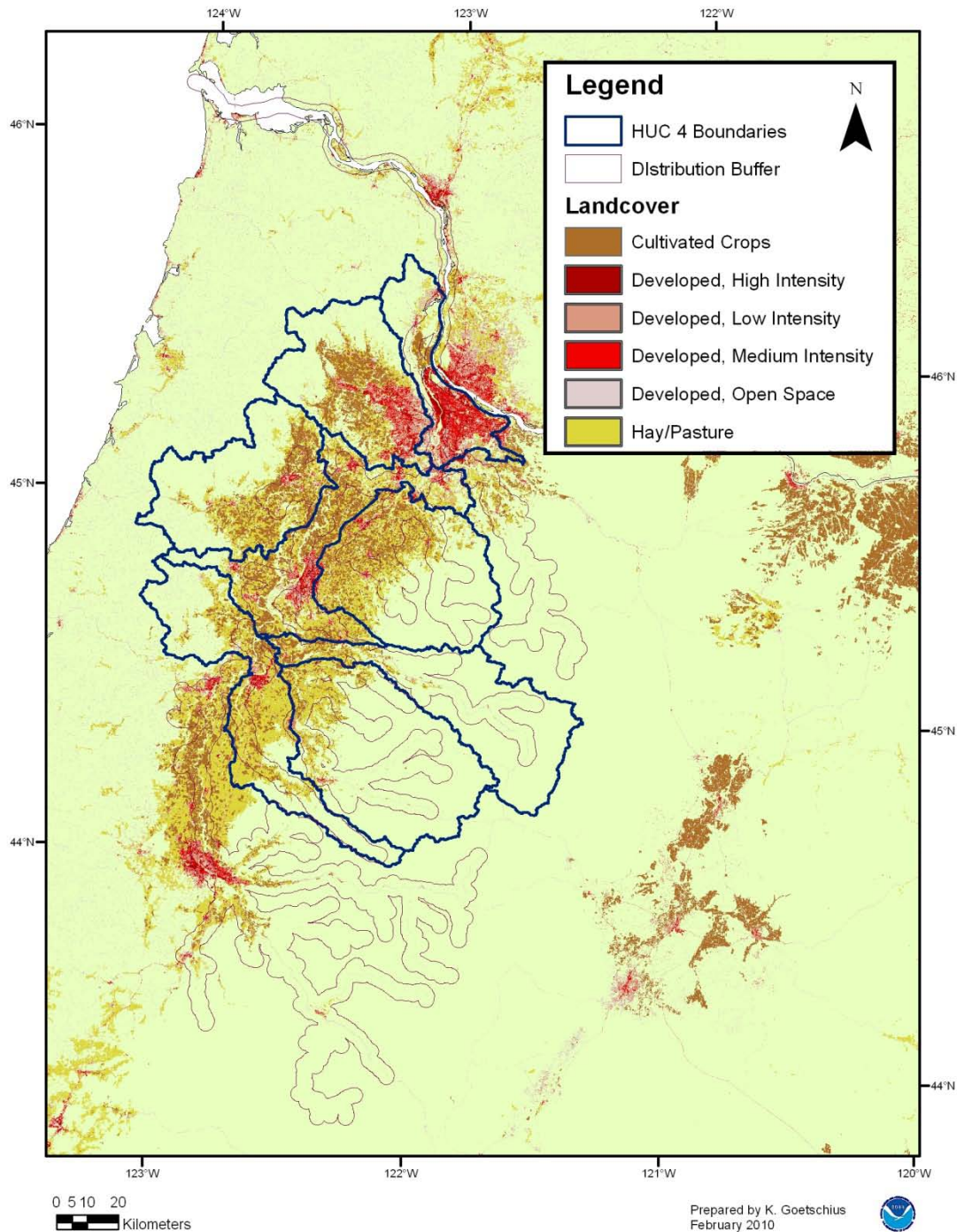
Snake River Spring-Summer Run Chinook Species Distribution



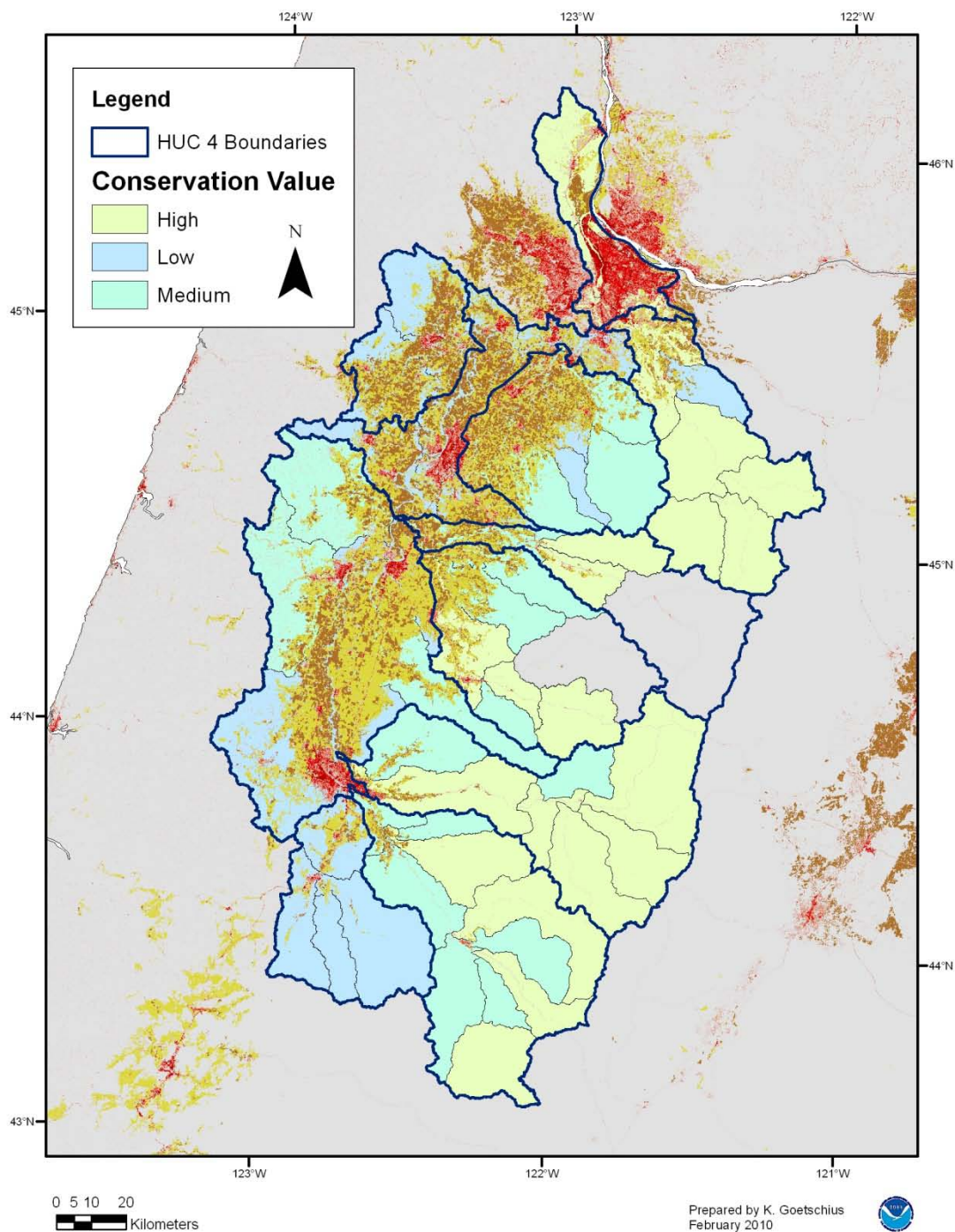
Snake River Spring-Summer Run Chinook Critical Habitat



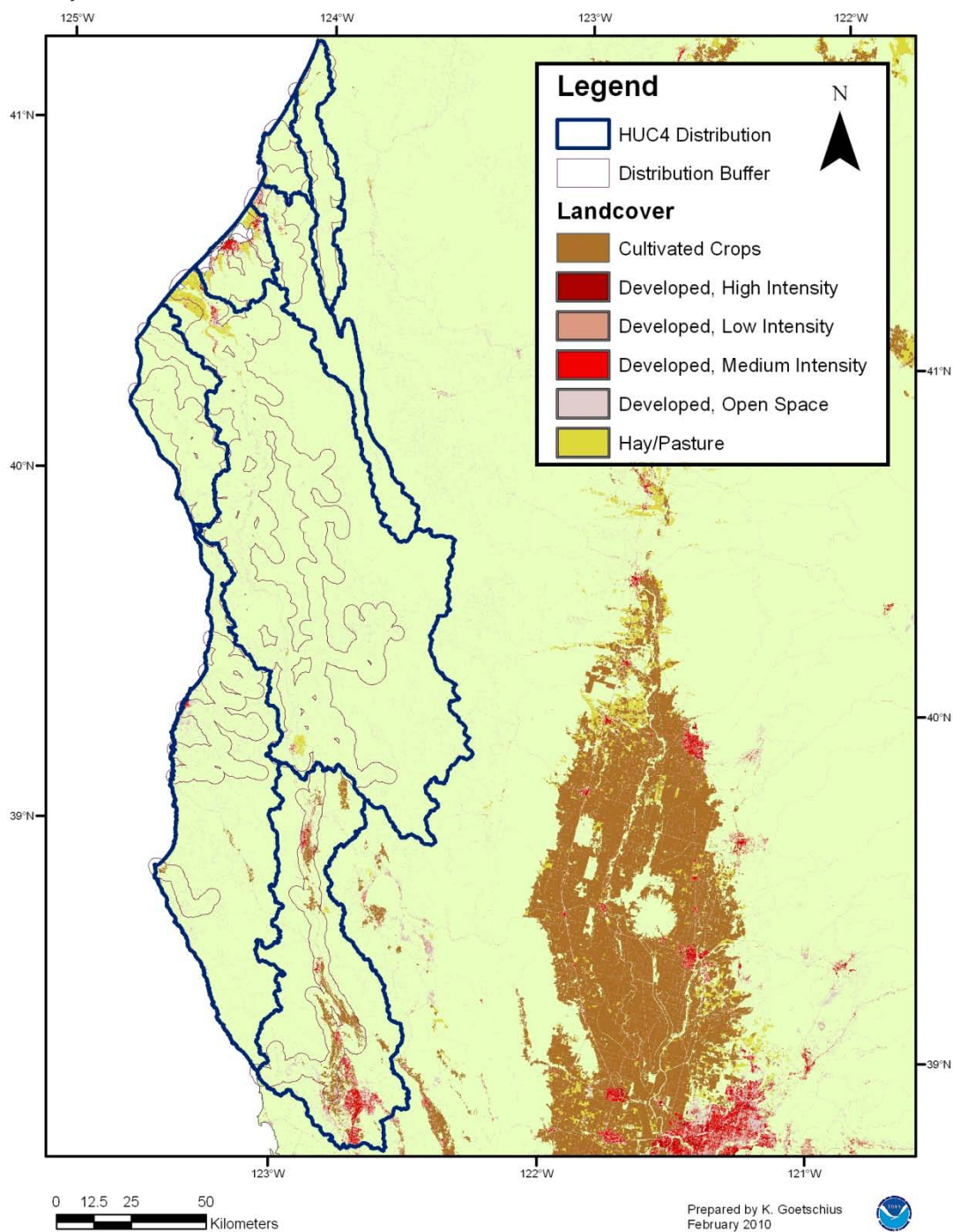
Upper Willamette River Chinook ESU Species Distribution



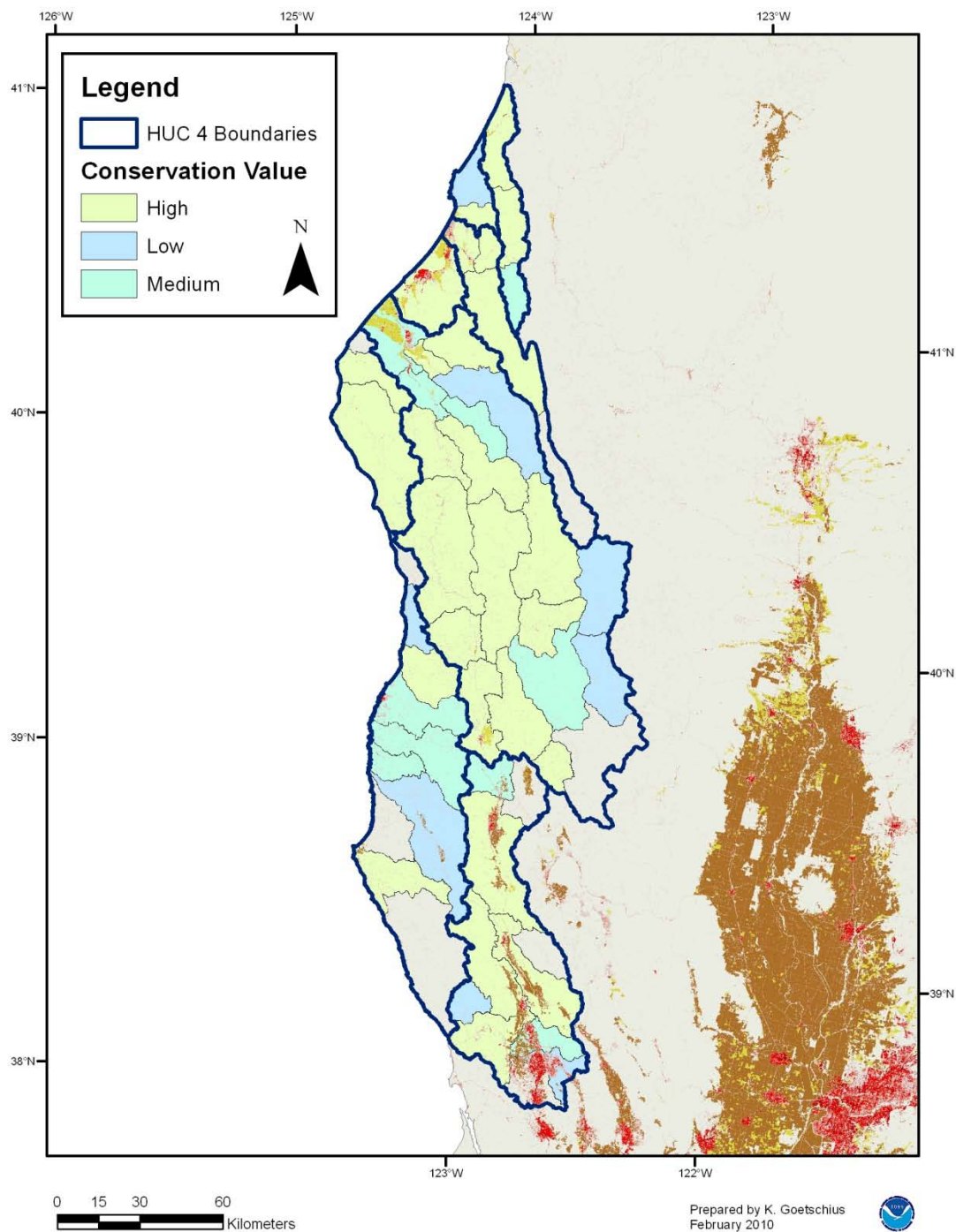
Upper Willamette River Chinook ESU Critical Habitat



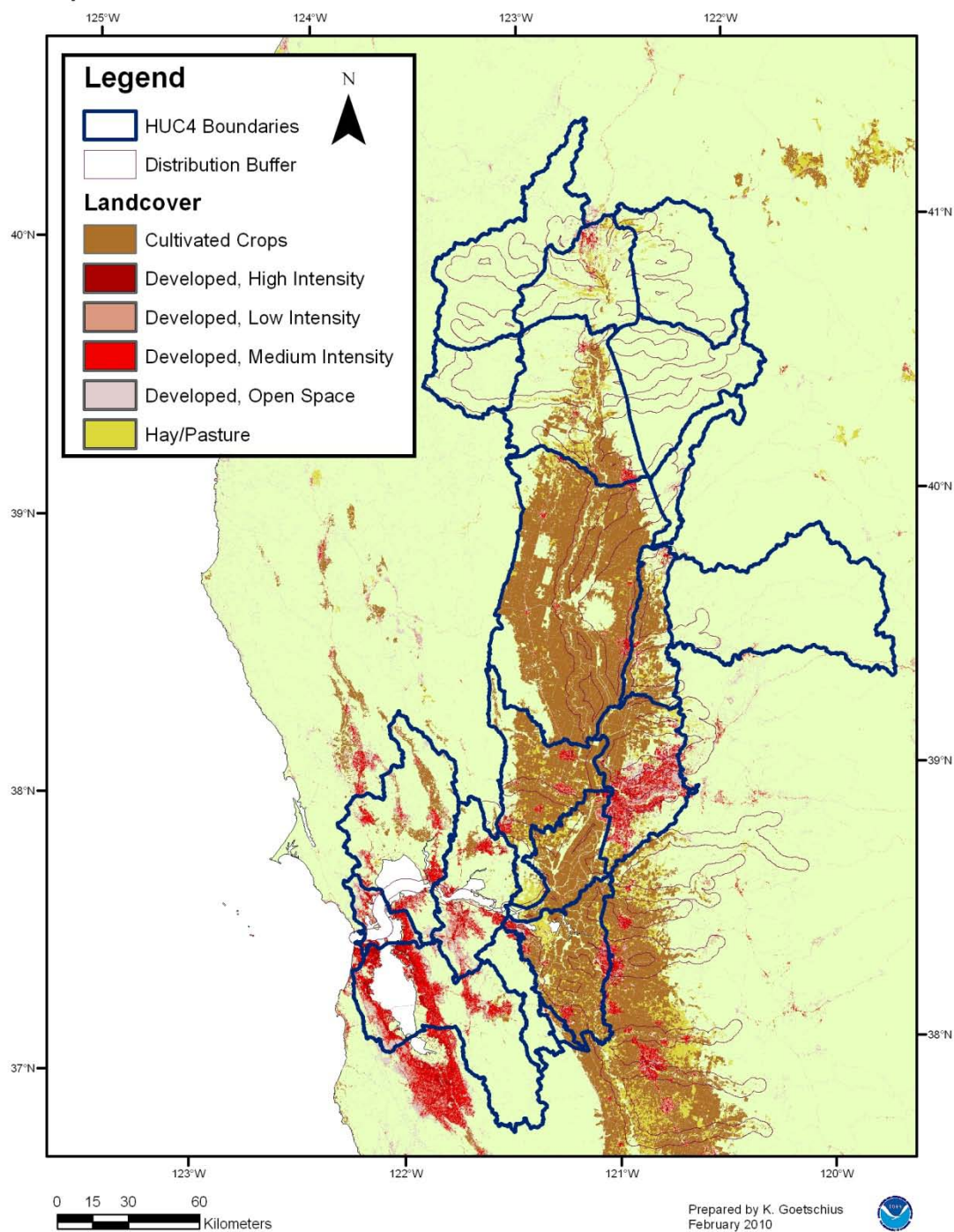
California Coastal Chinook ESU Species Distribution



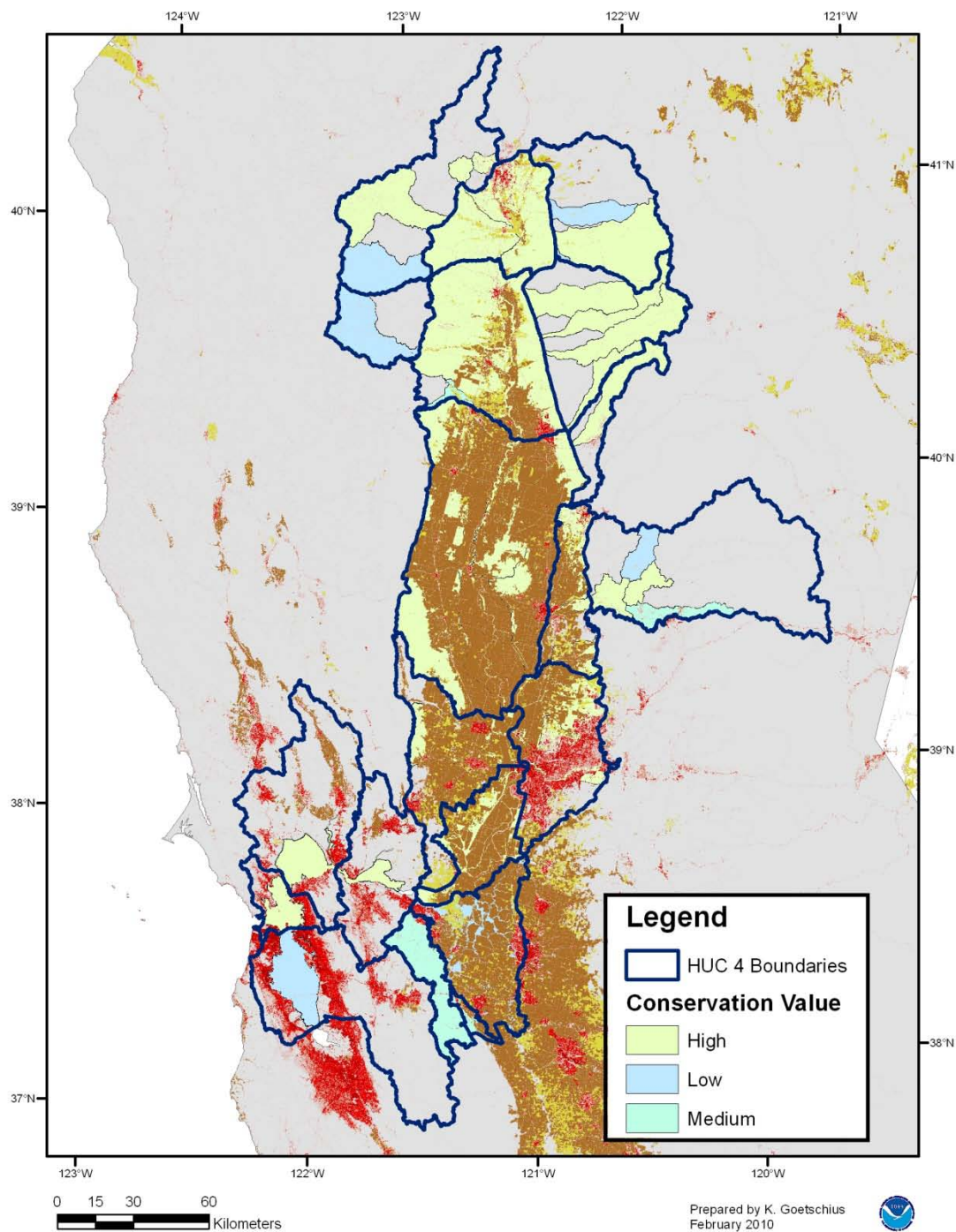
California Coastal Chinook ESU Critical Habitat



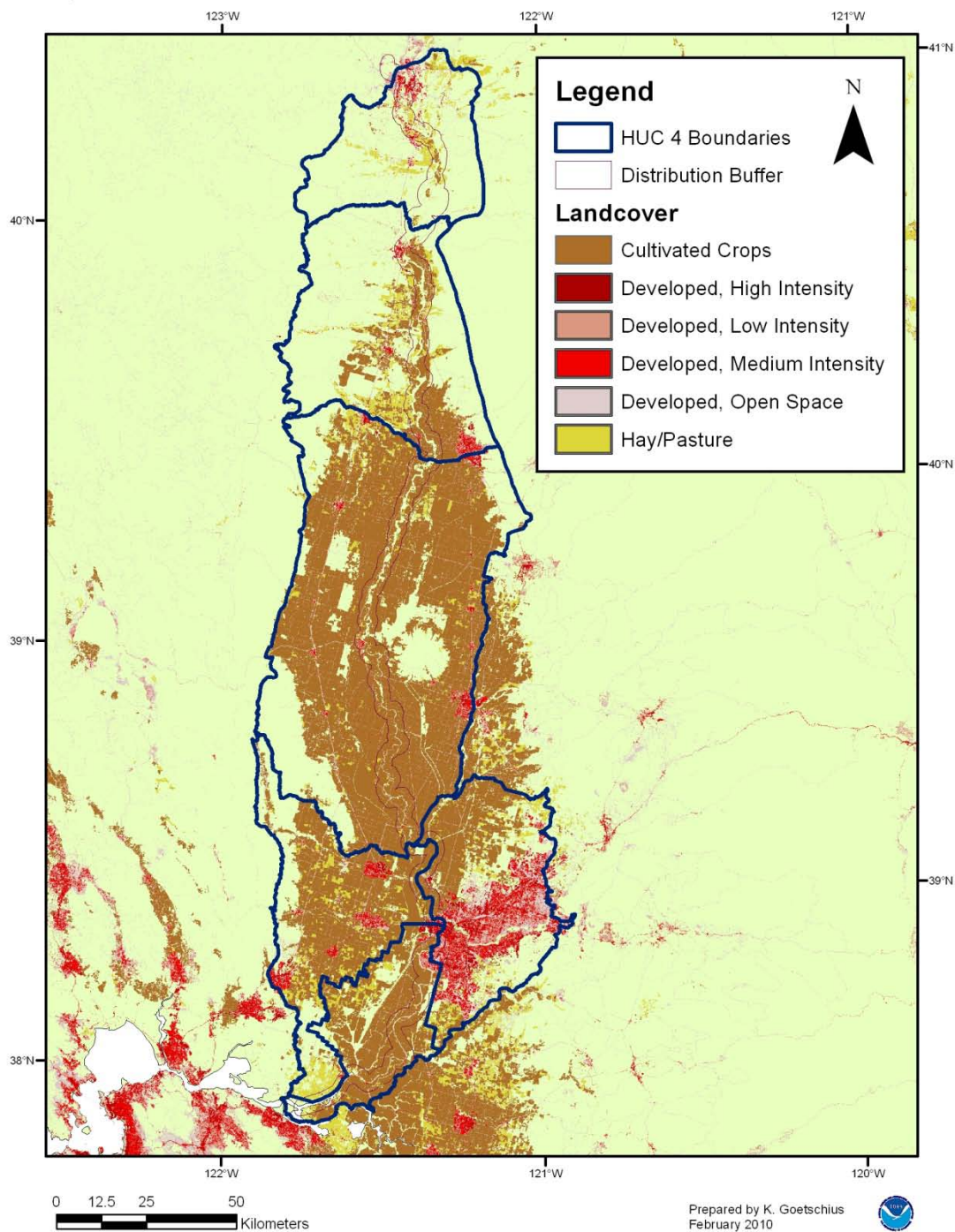
Central Valley Spring-Run Chinook ESU Species Distribution



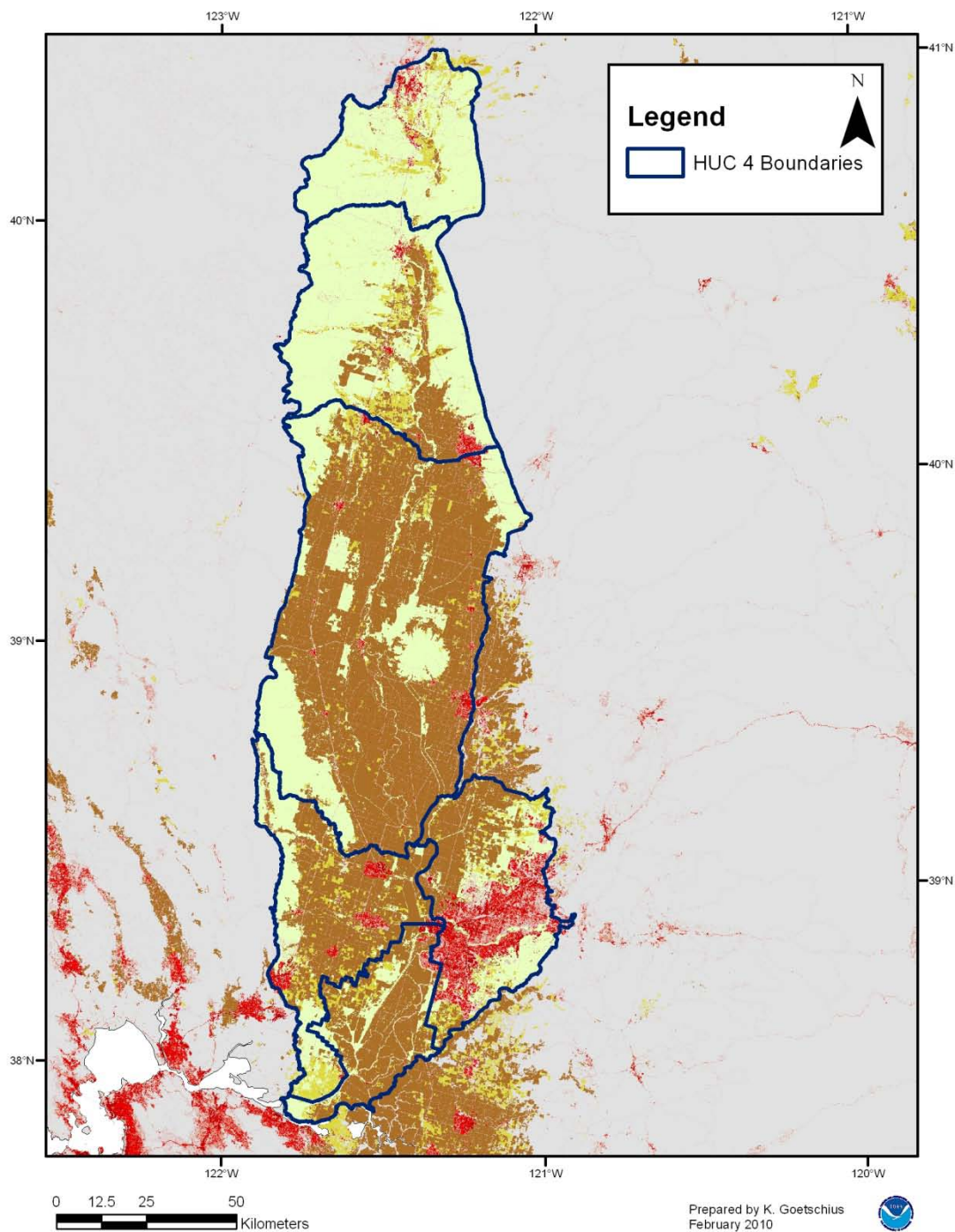
Central Valley Spring-Run Chinook ESU Critical Habitat



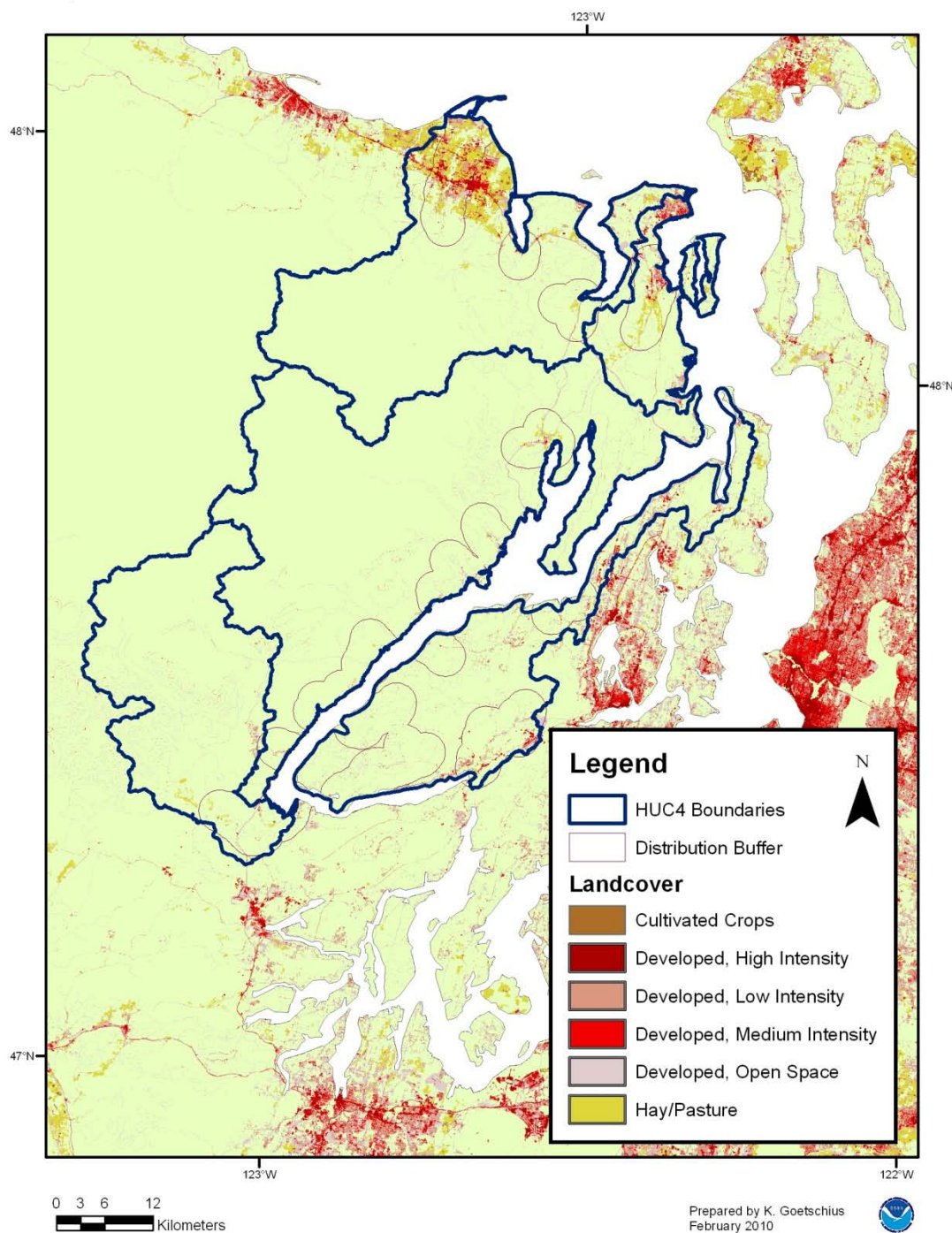
Sacramento River Winter Run Chinook ESU Species Distribution



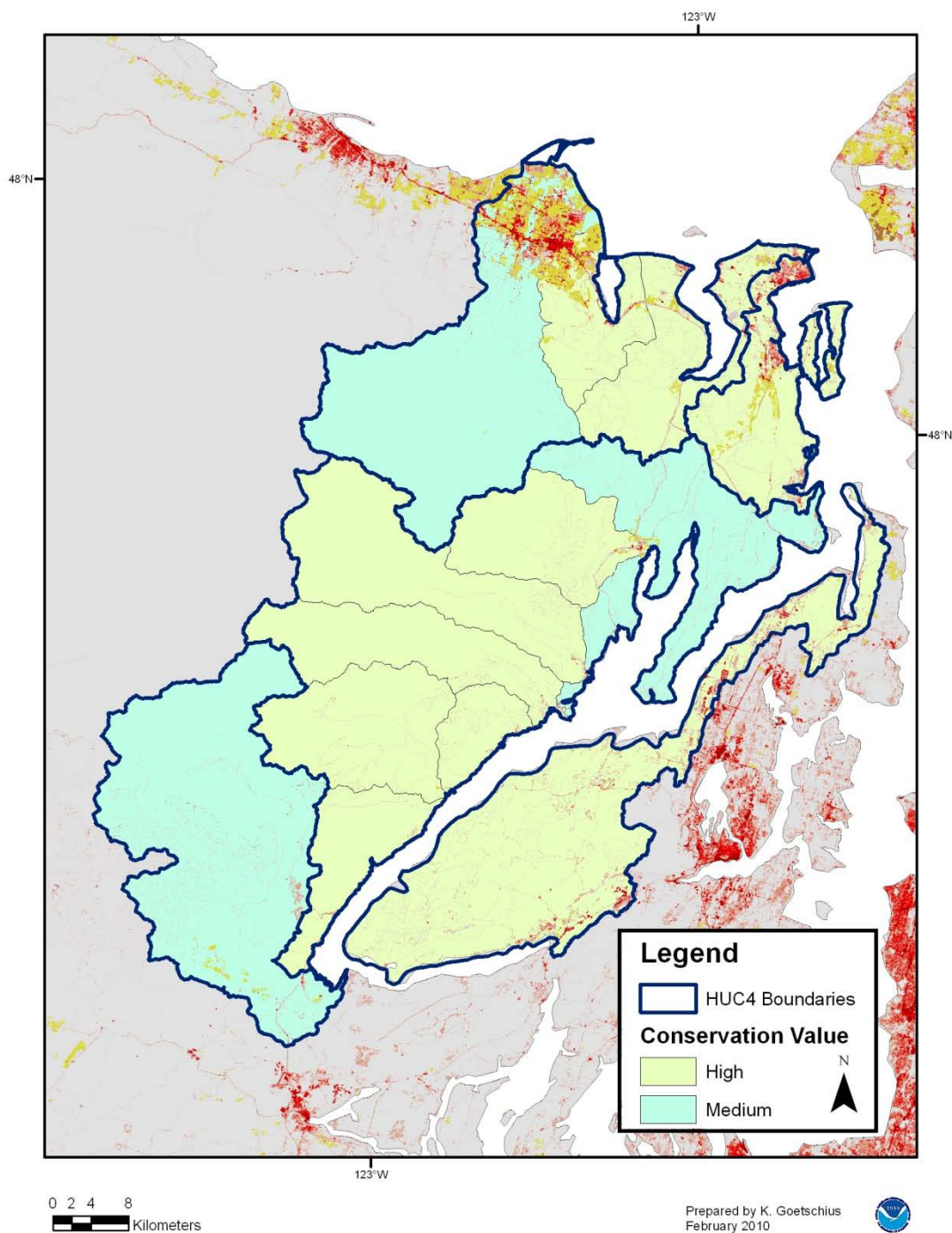
Sacramento River Winter Run Chinook ESU Critical Habitat



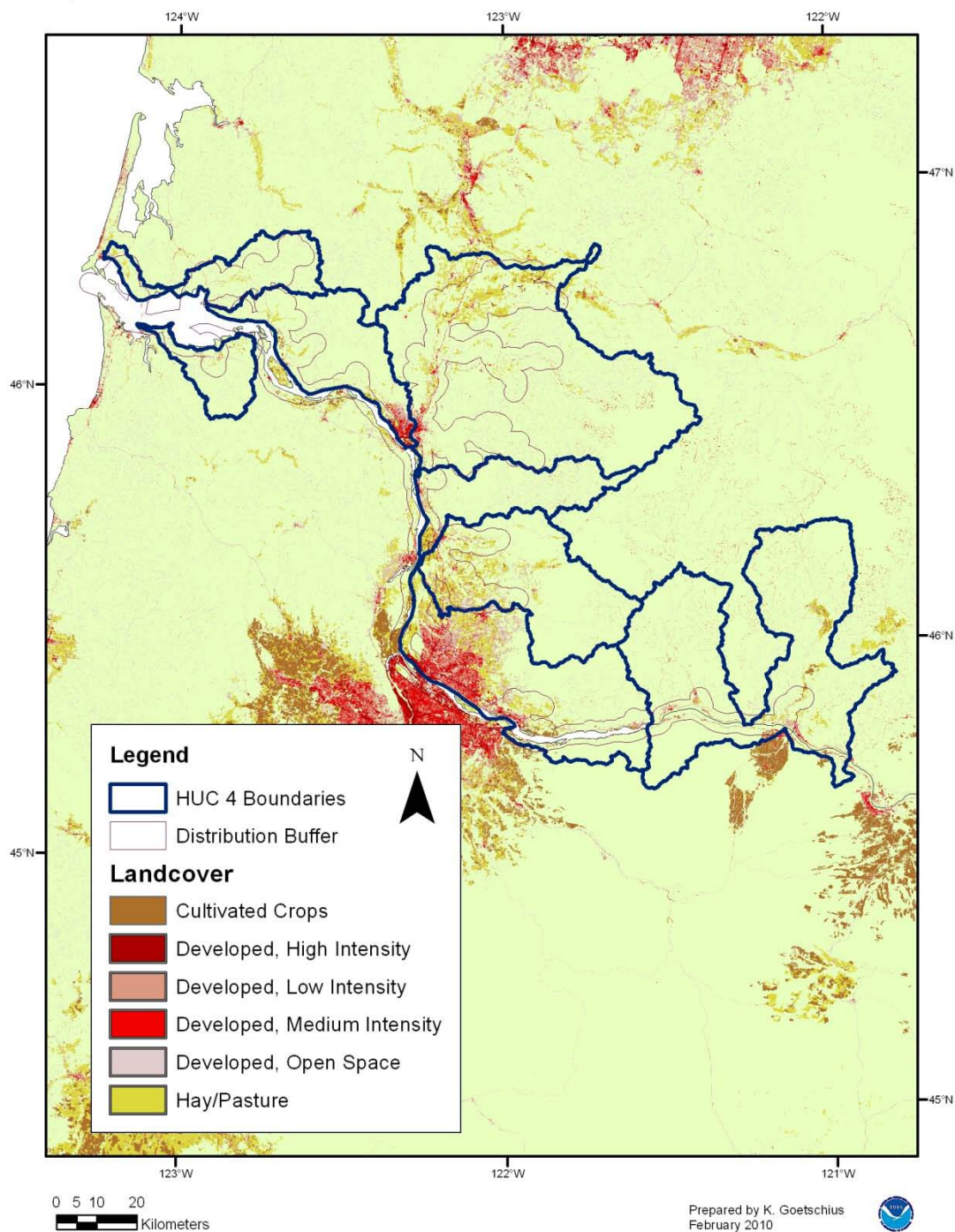
Hood Canal Summer-Run Chum ESU Species Distribution



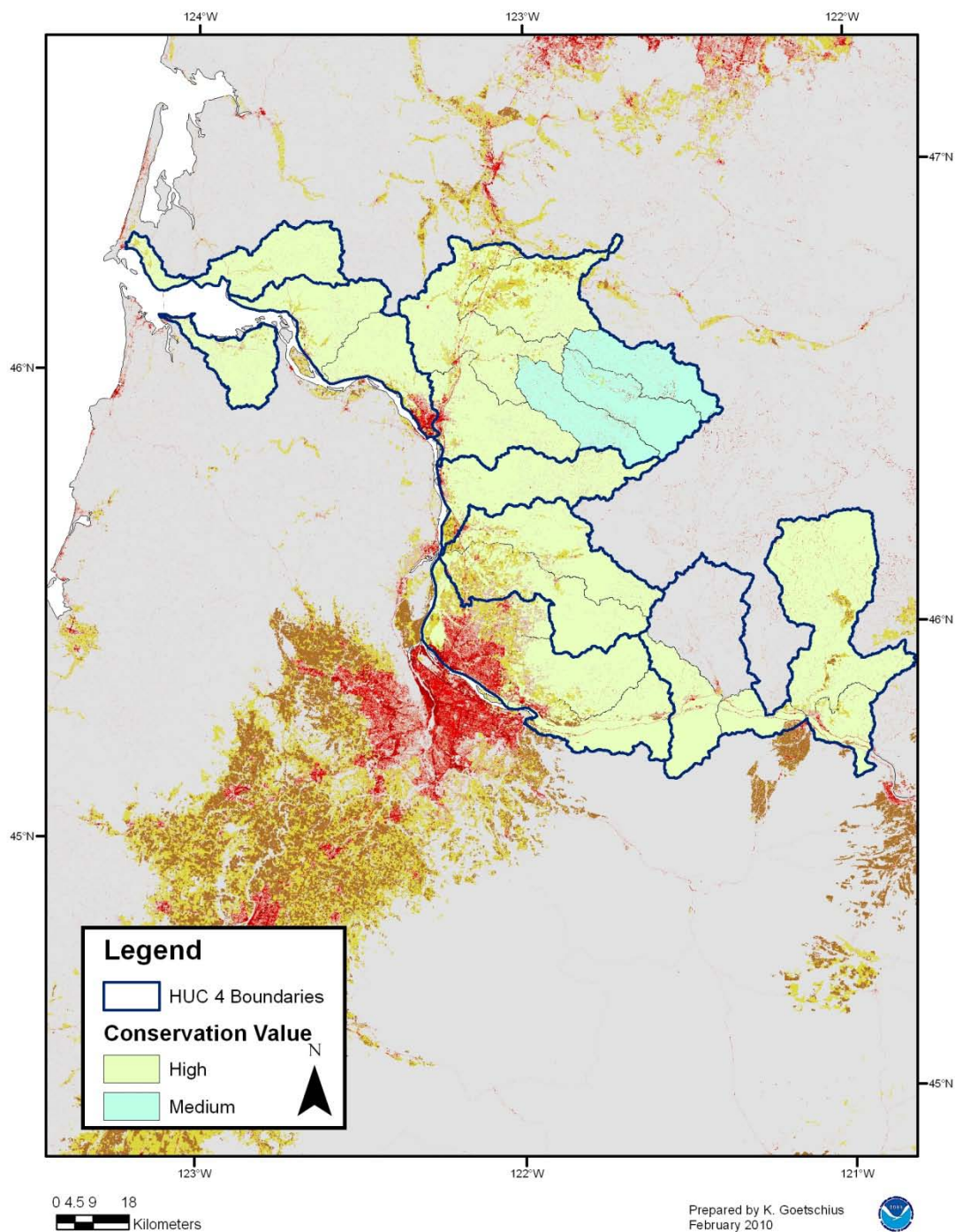
Hood Canal Summer-Run Chum ESU Critical Habitat



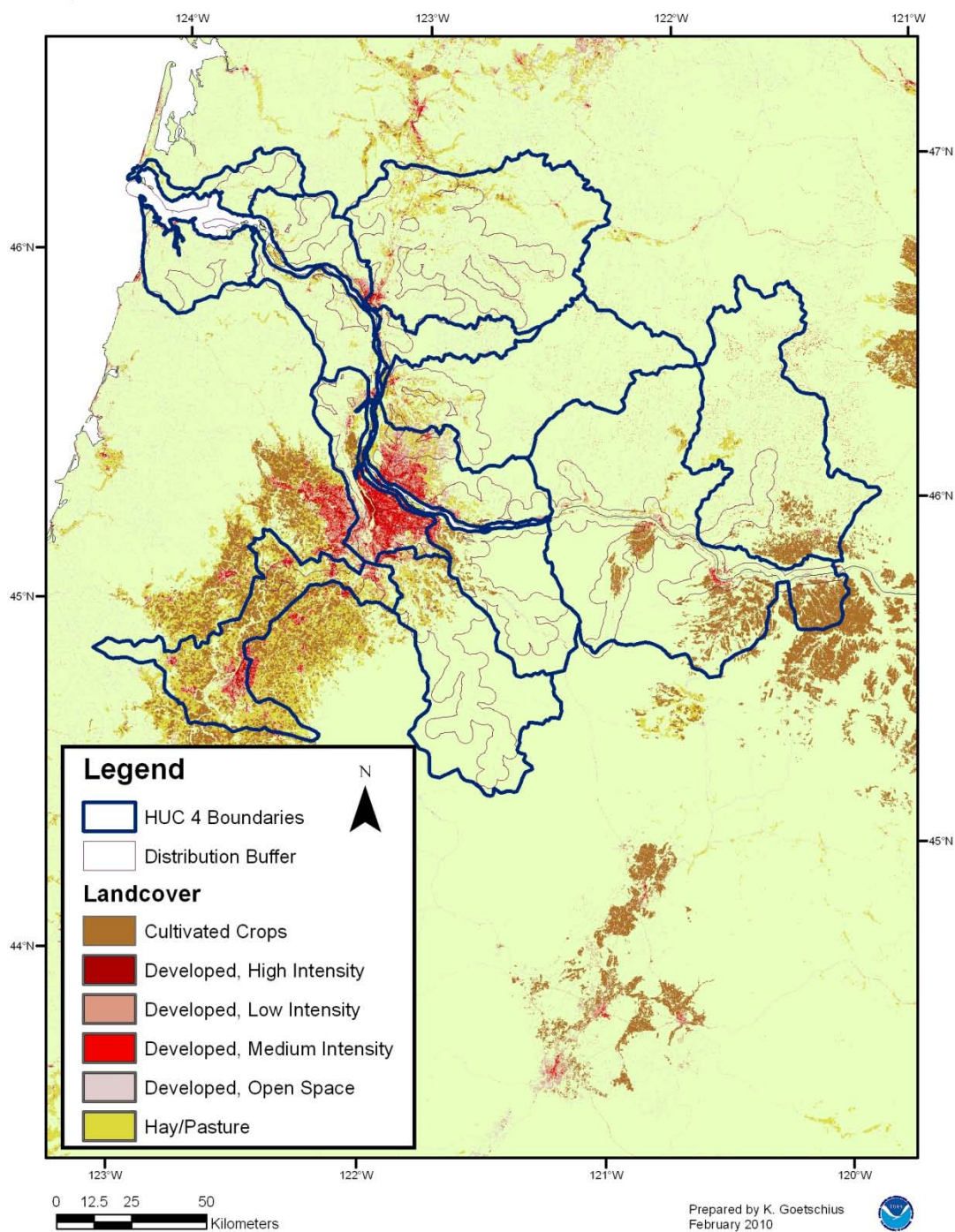
Columbia River Chum ESU Species Distribution



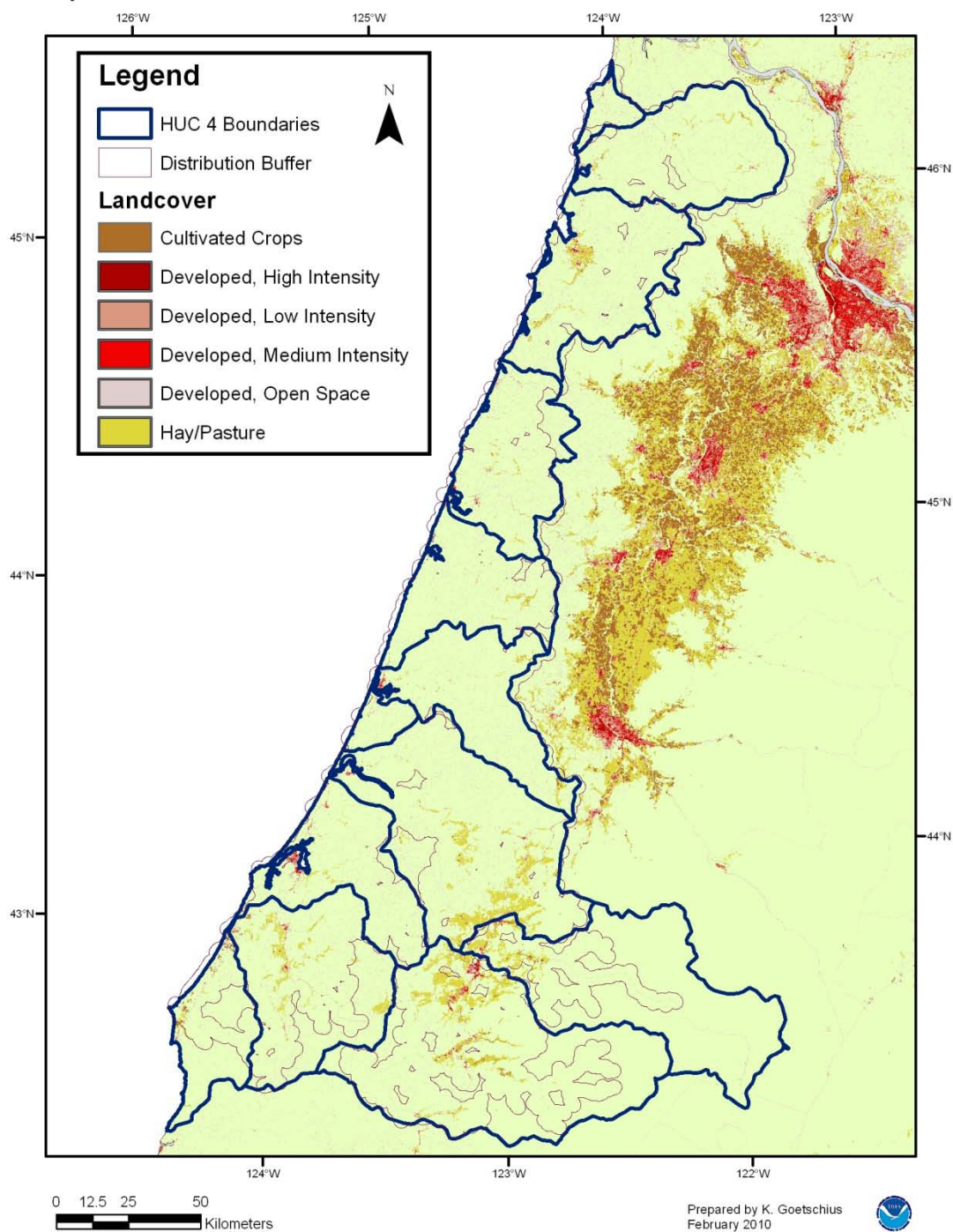
Columbia River Chum ESU Critical Habitat



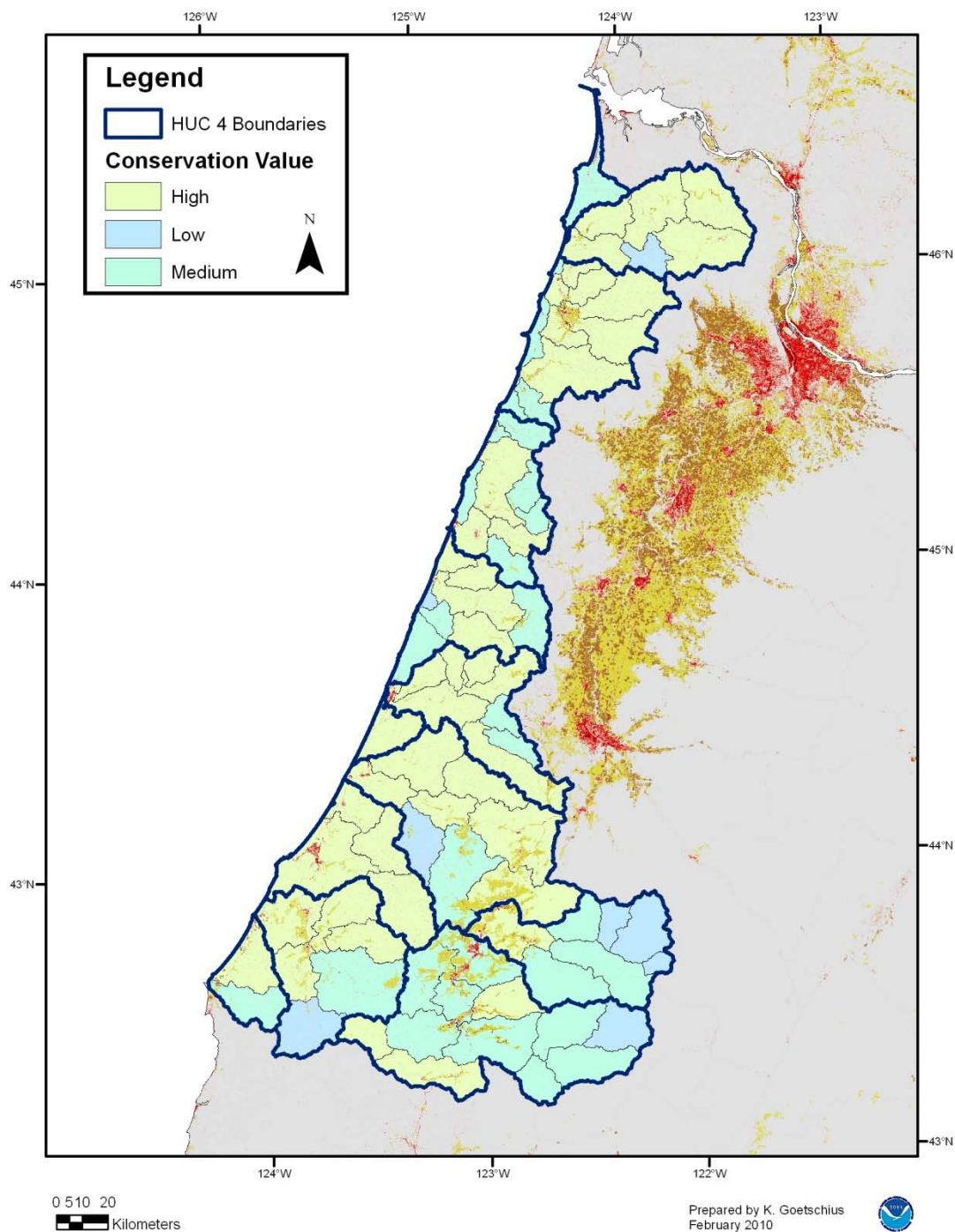
Lower Columbia River Coho ESU Species Distribution



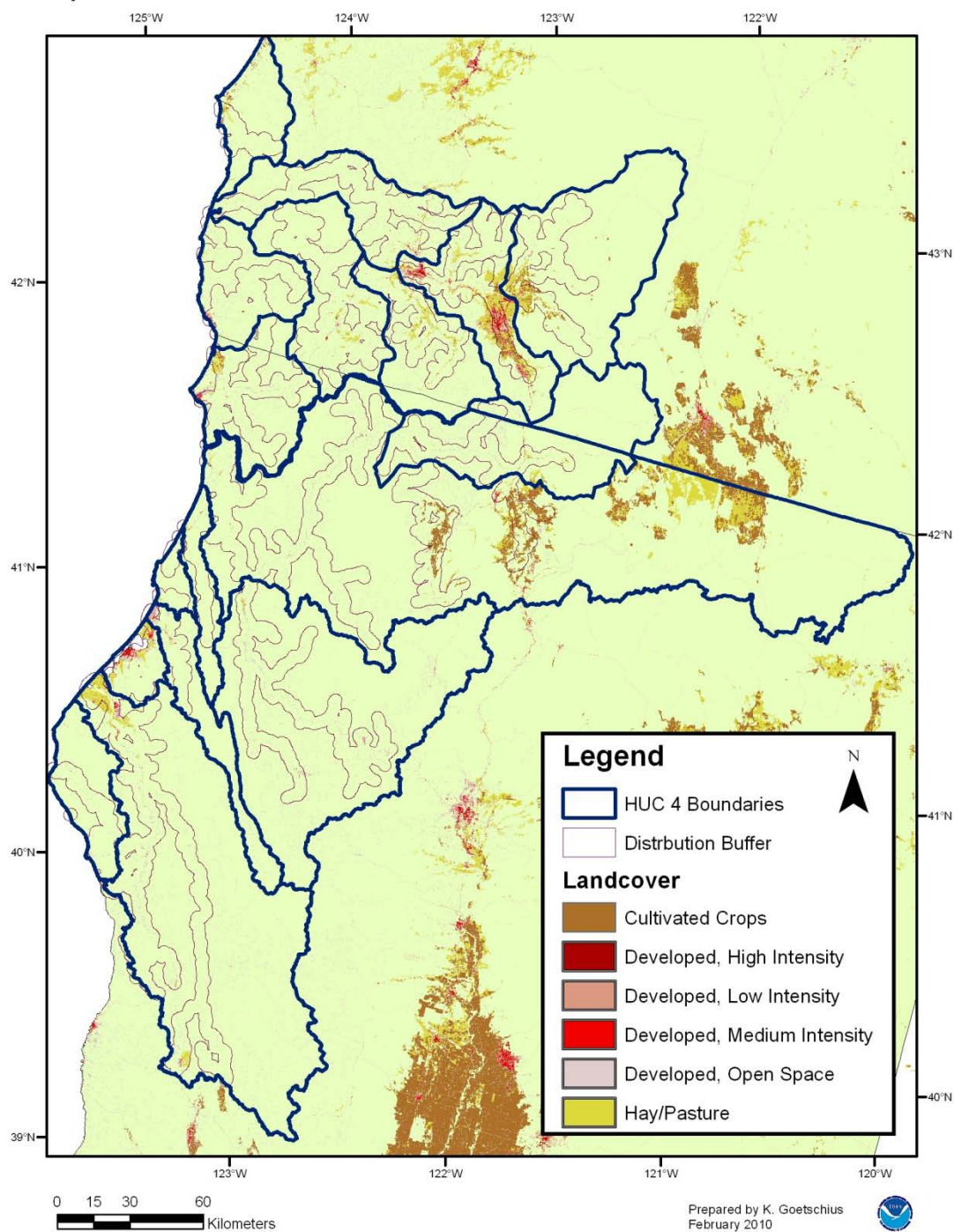
Oregon Coast Coho ESU Species Distribution



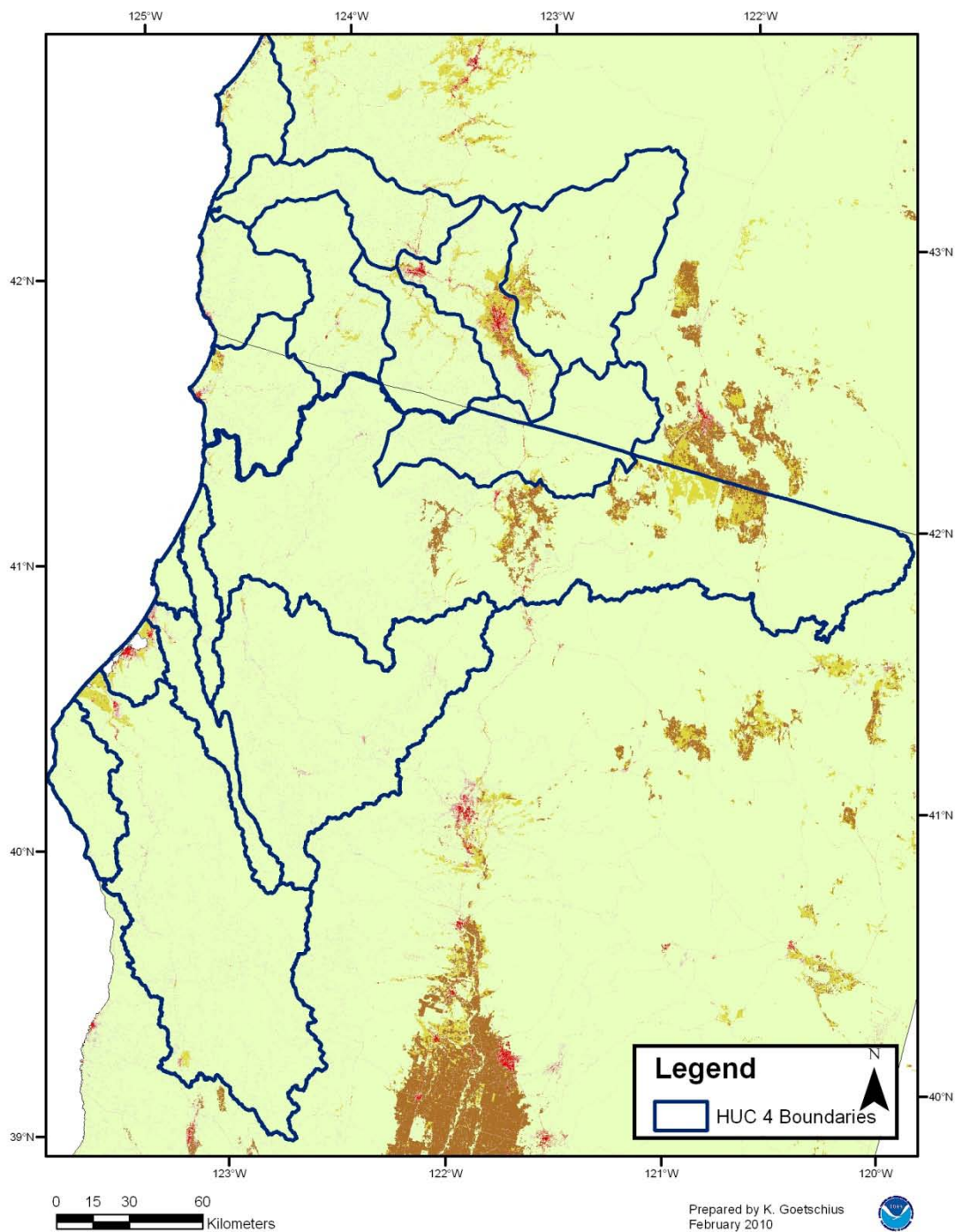
Oregon Coast Coho ESU Critical Habitat



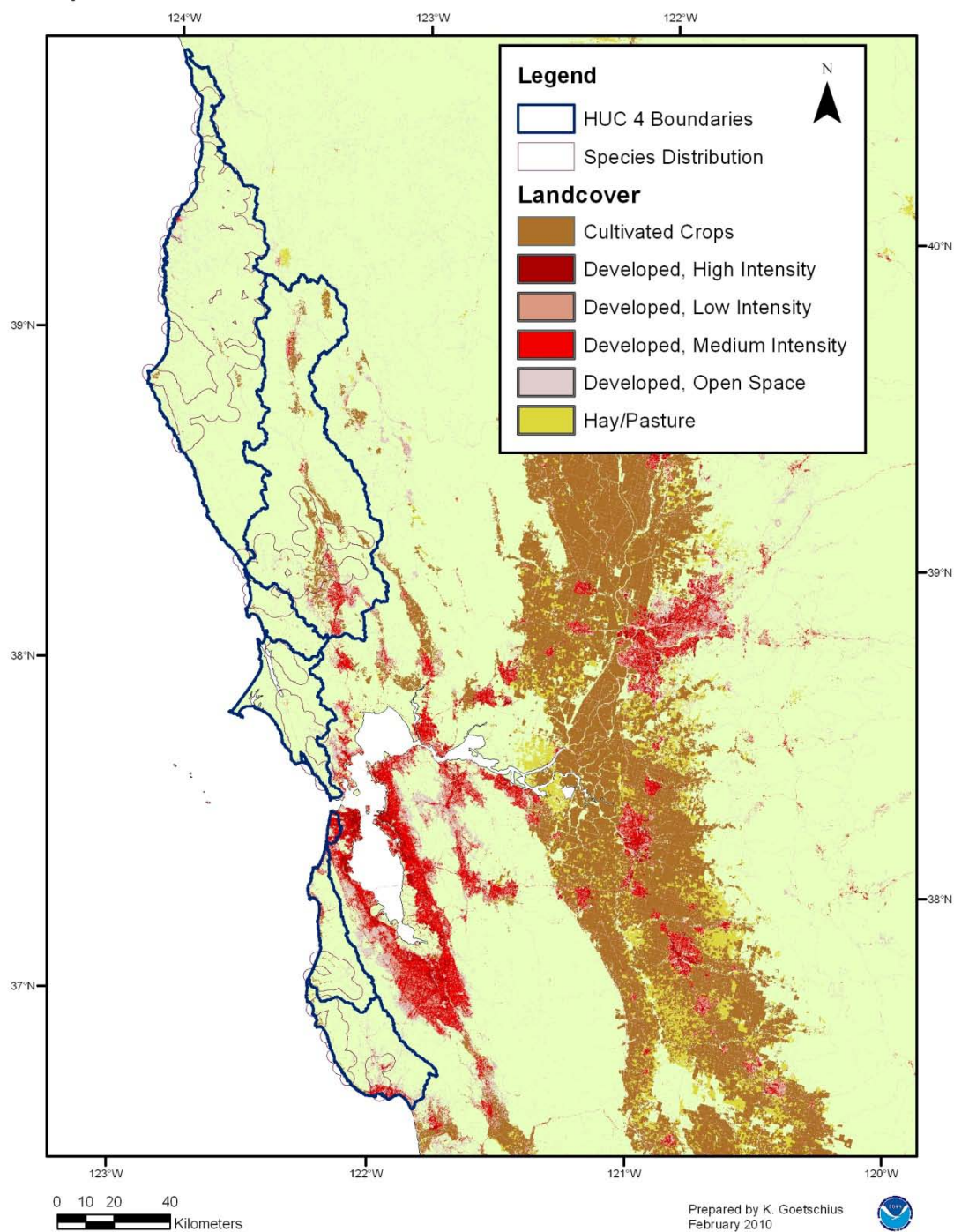
Southern Oregon Northern California Coho ESU Species Distribution



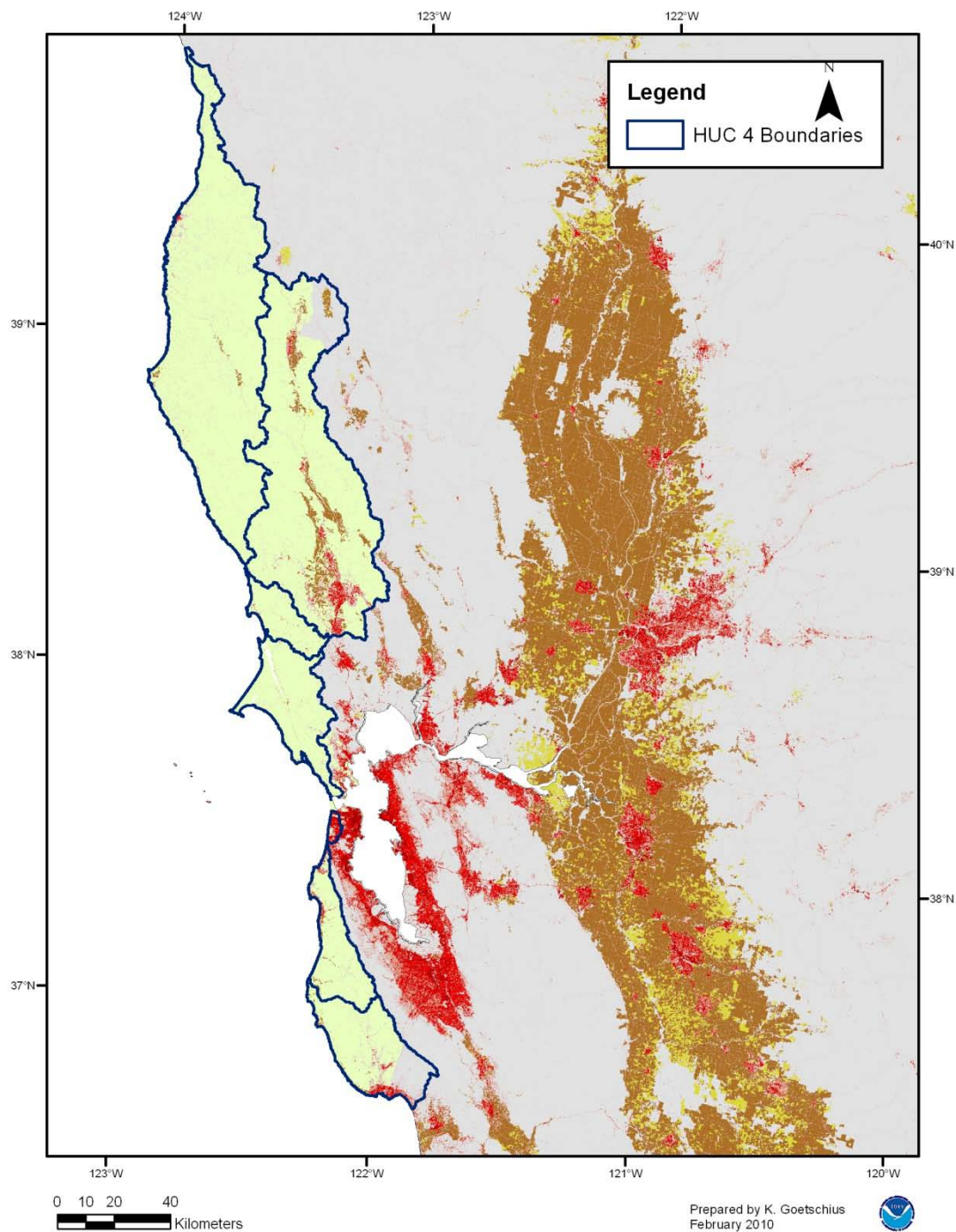
Southern Oregon Northern California Coho ESU Critical Habitat



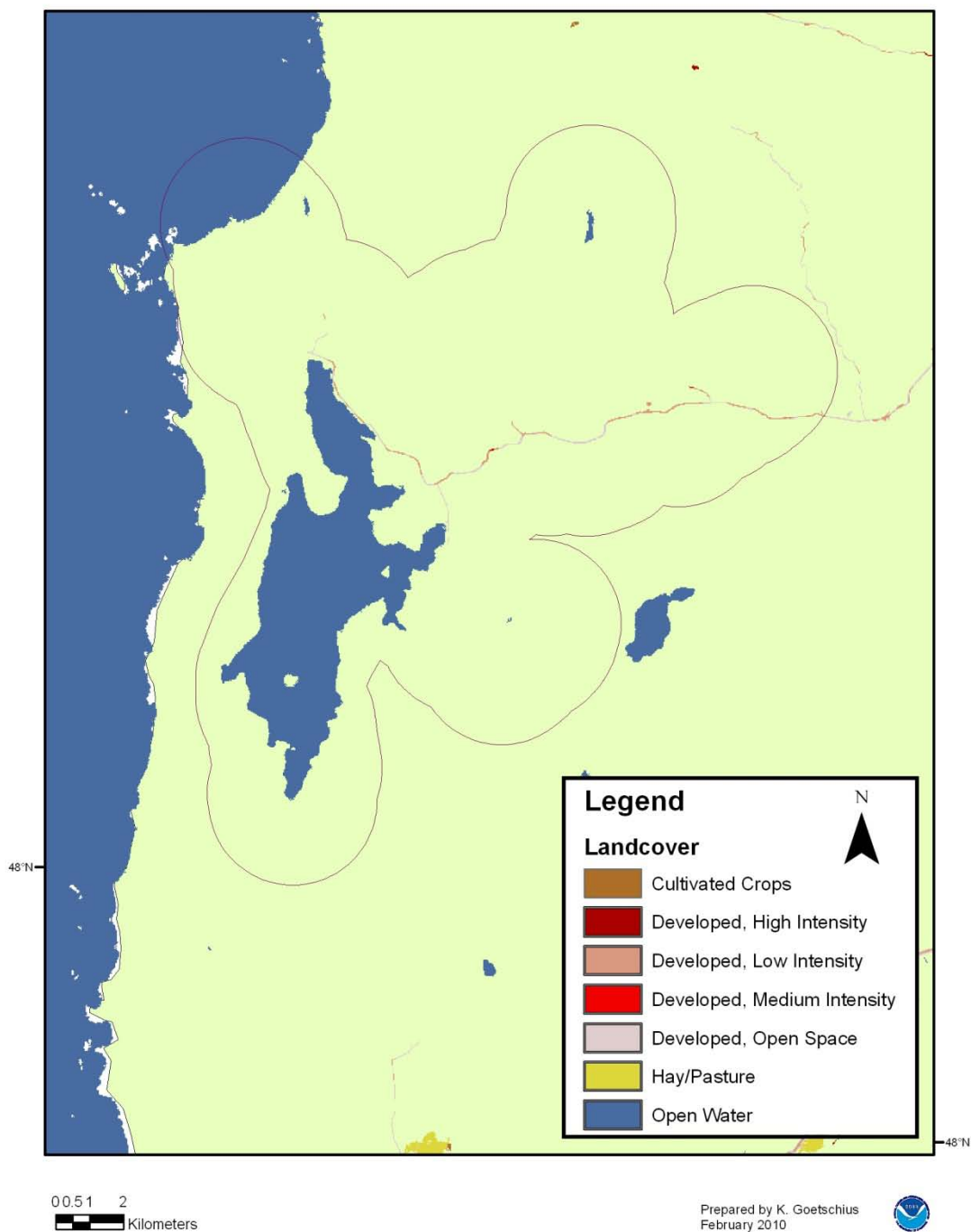
Central California Coastal Coho Species Distribution



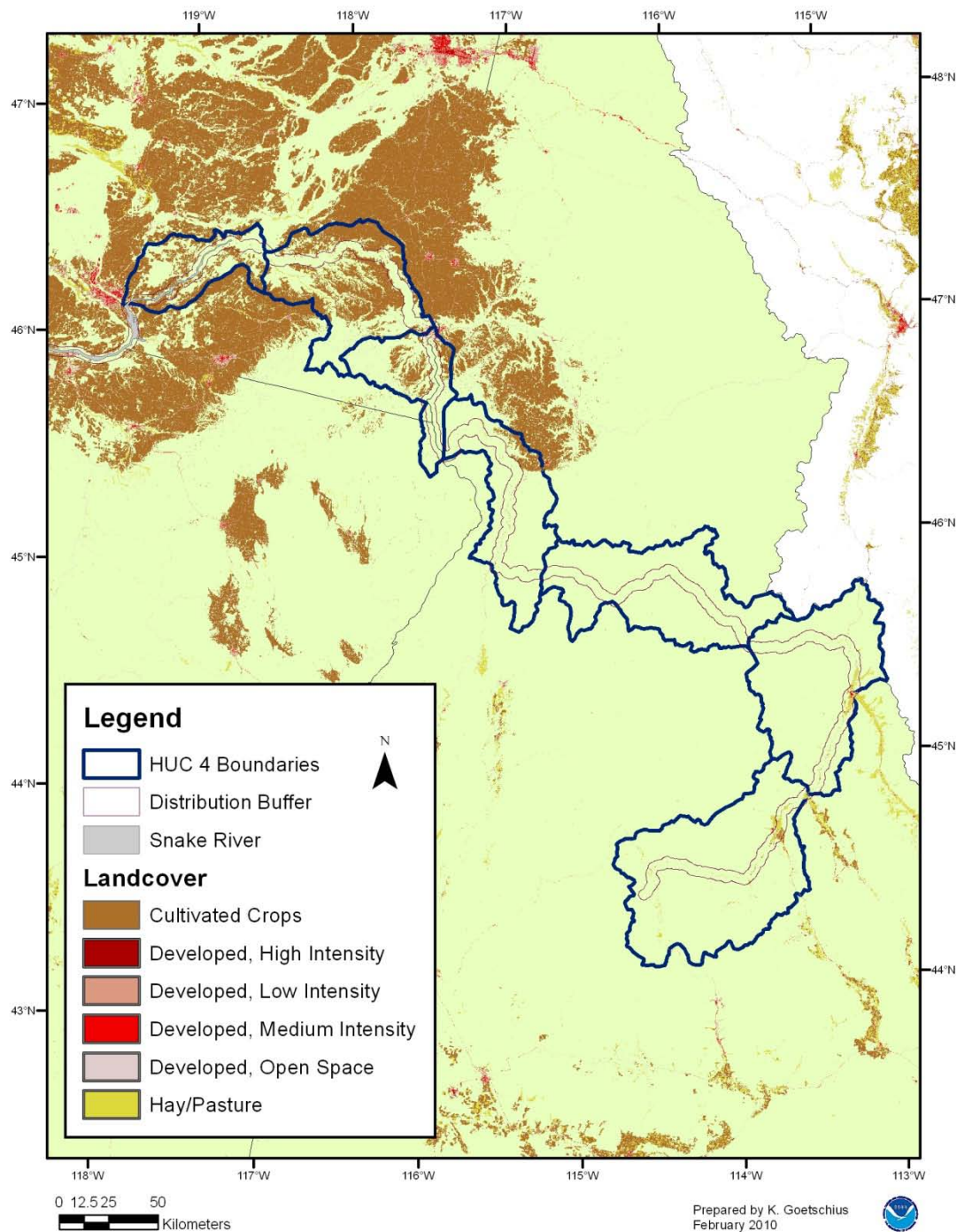
Central California Coastal Coho Critical Habitat



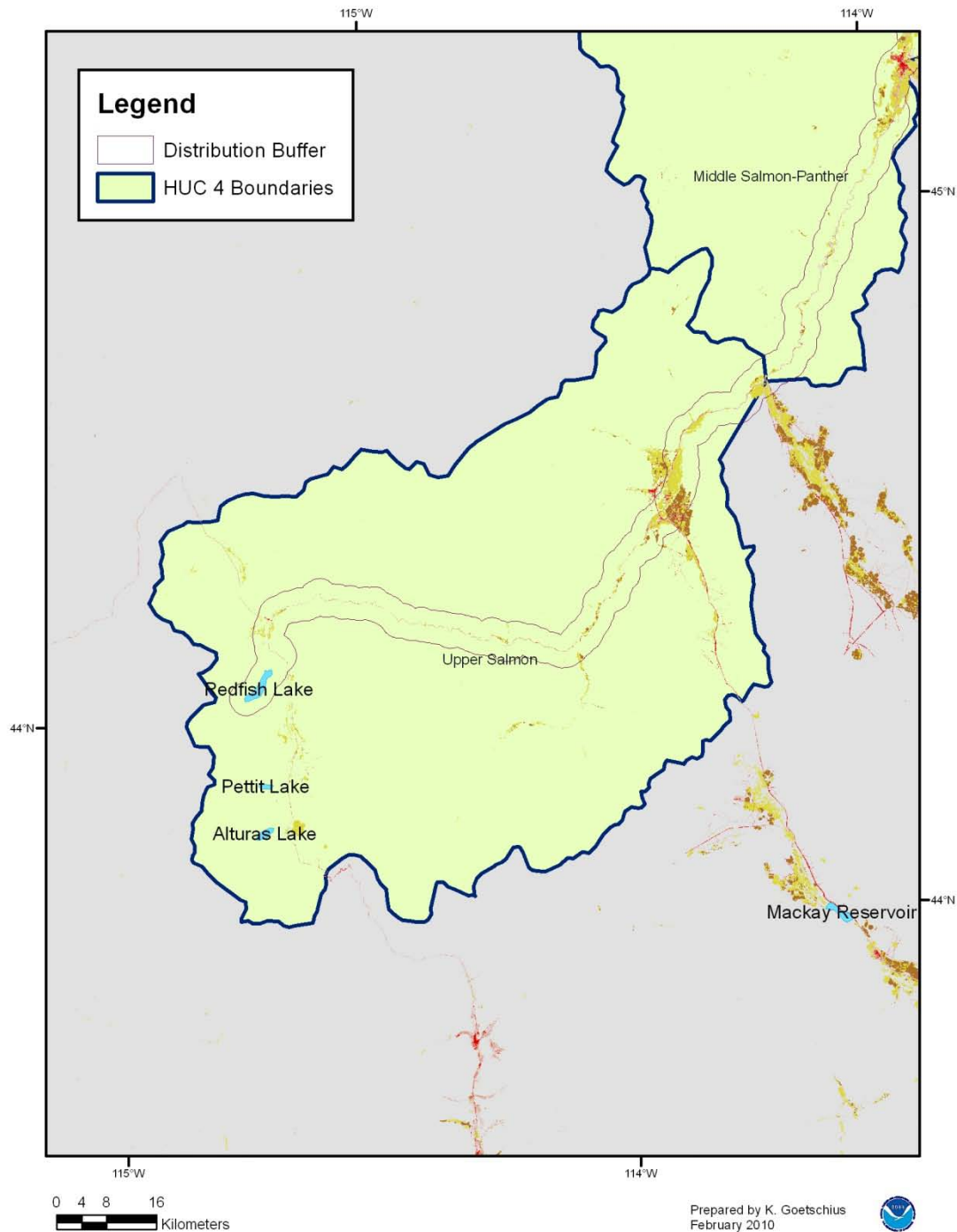
Ozette Lake Sockeye Species Distribution and Critical Habitat



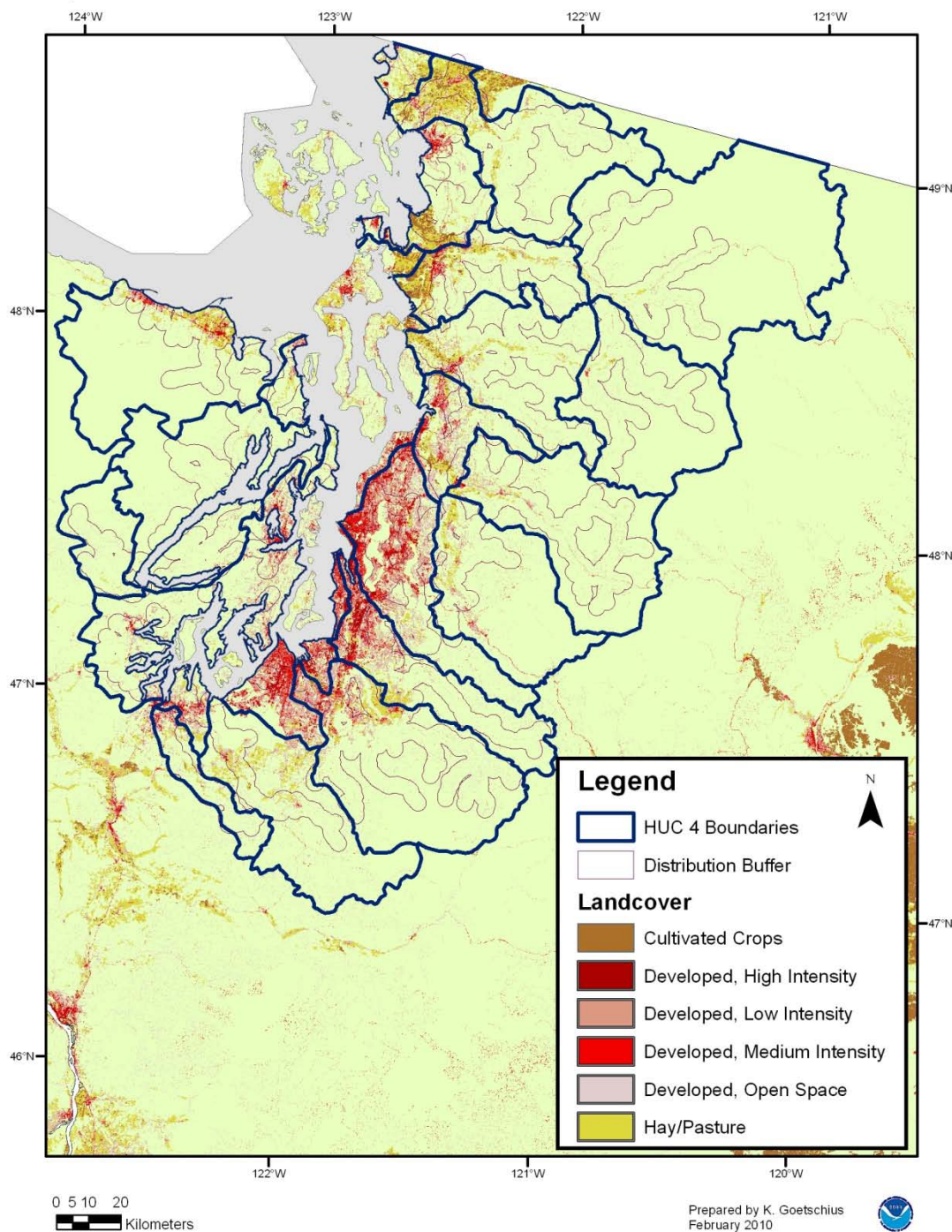
Snake River Sockeye ESU Species Distribution



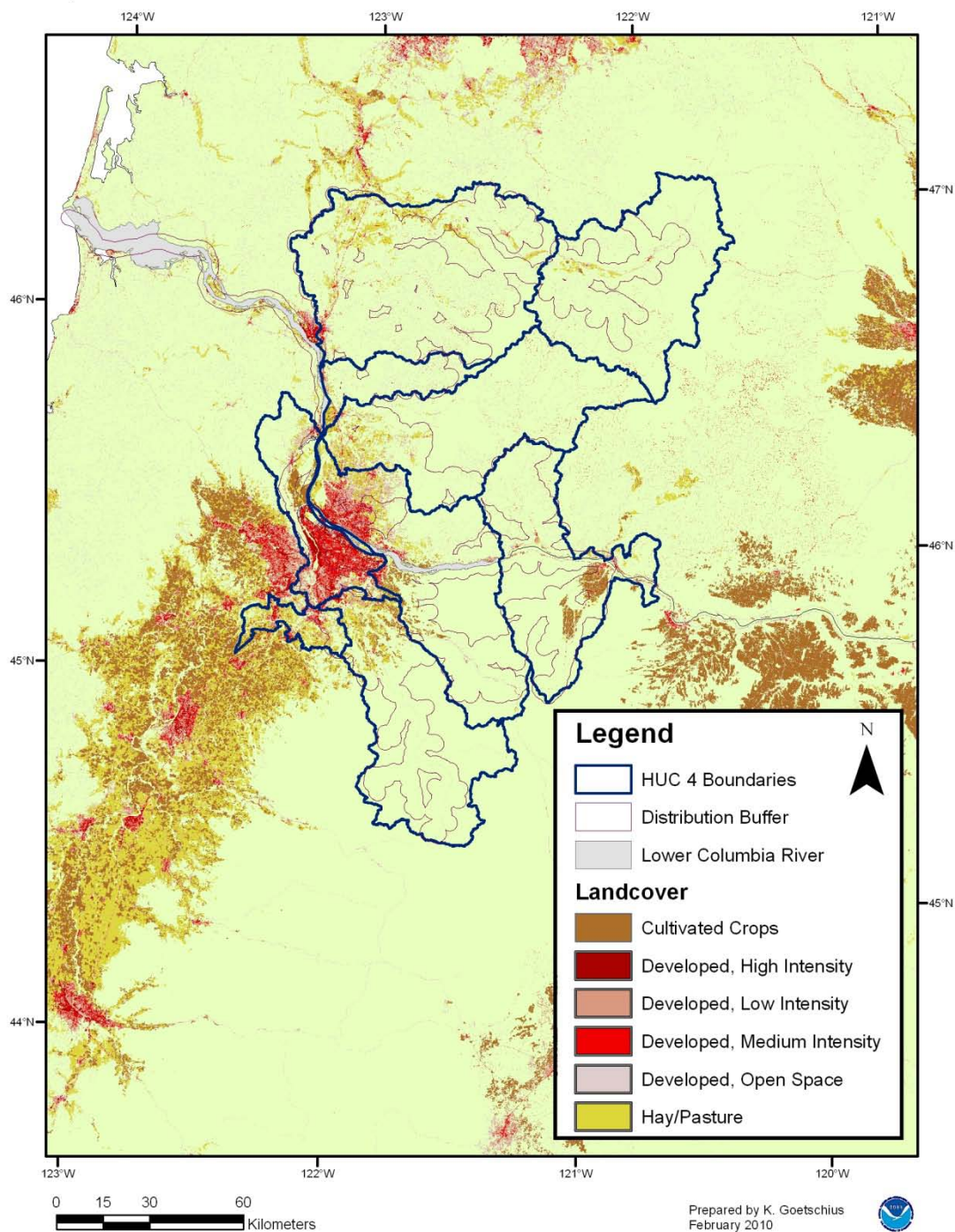
Snake River Sockeye Critical Habitat, close up of lakes



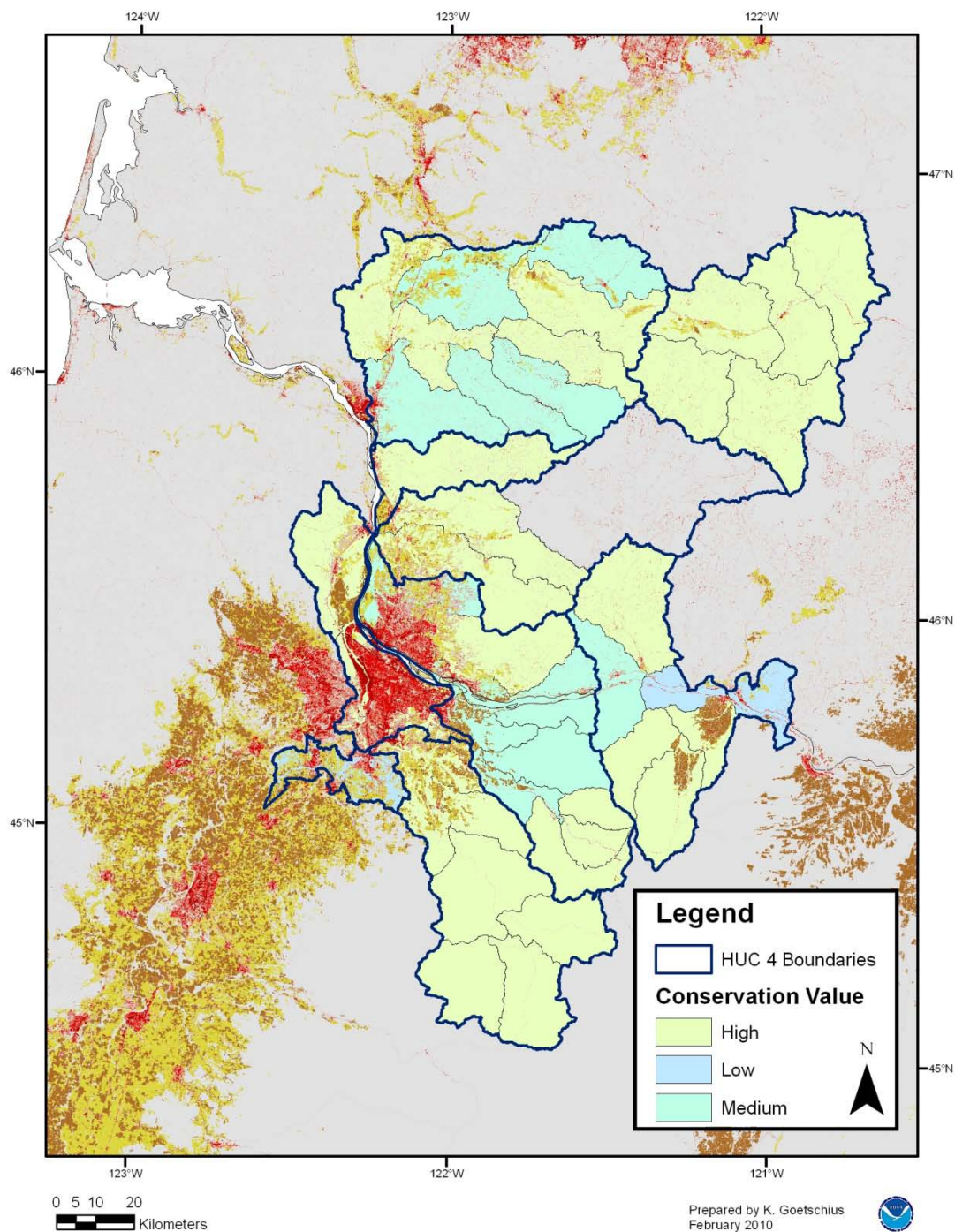
Puget Sound Steelhead DPS Species Distribution



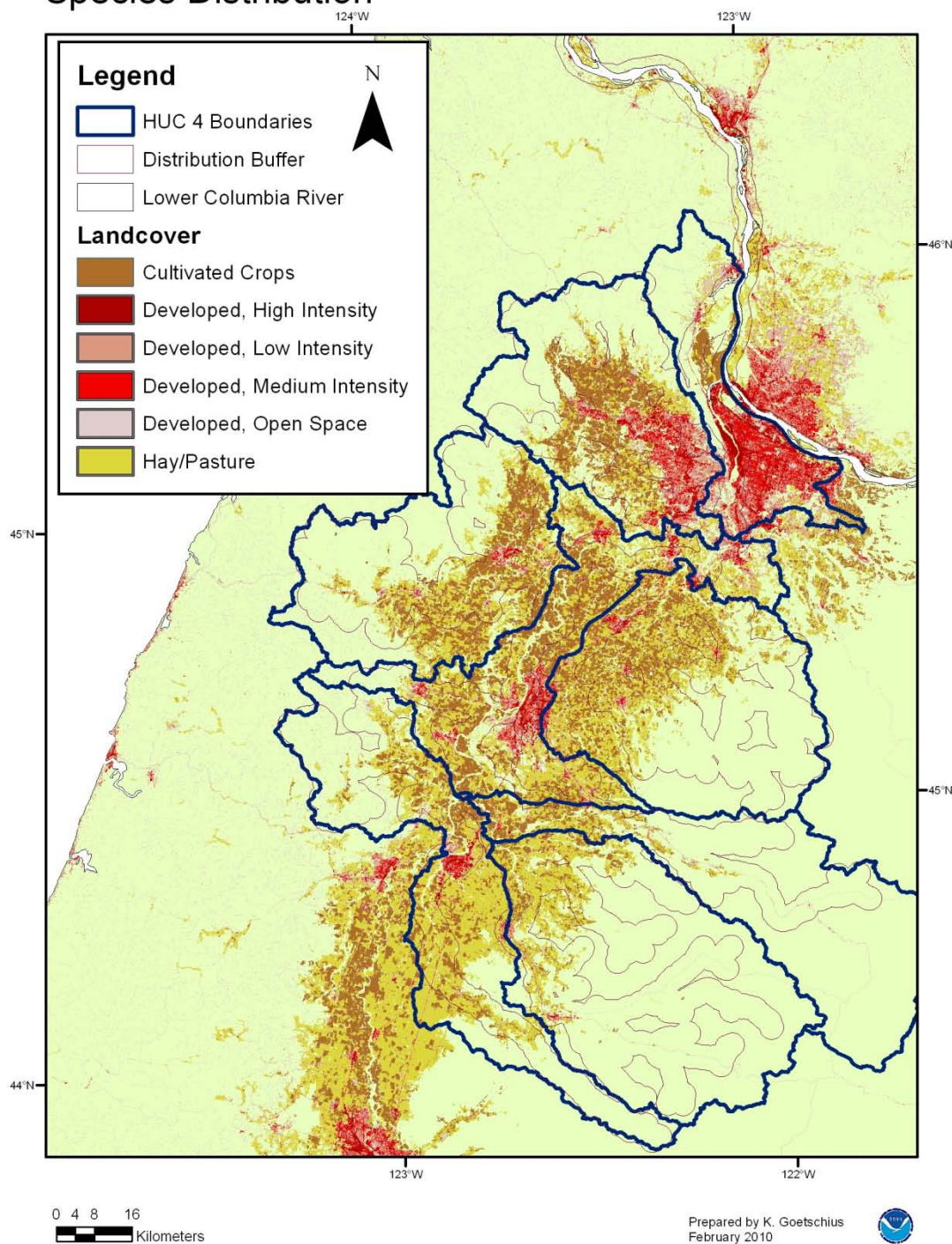
Lower Columbia River Steelhead DPS Species Distribution



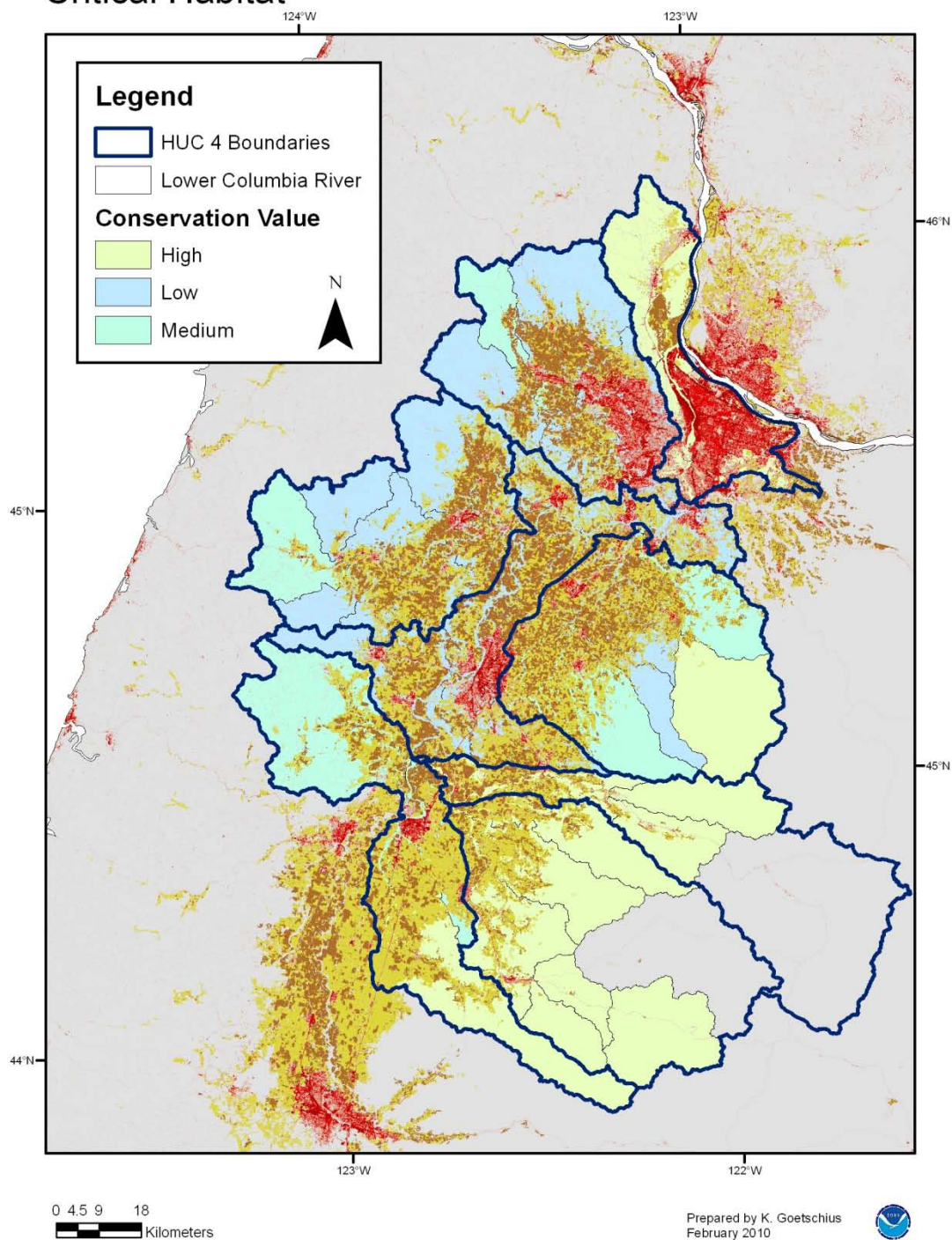
Lower Columbia River Steelhead DPS Critical Habitat



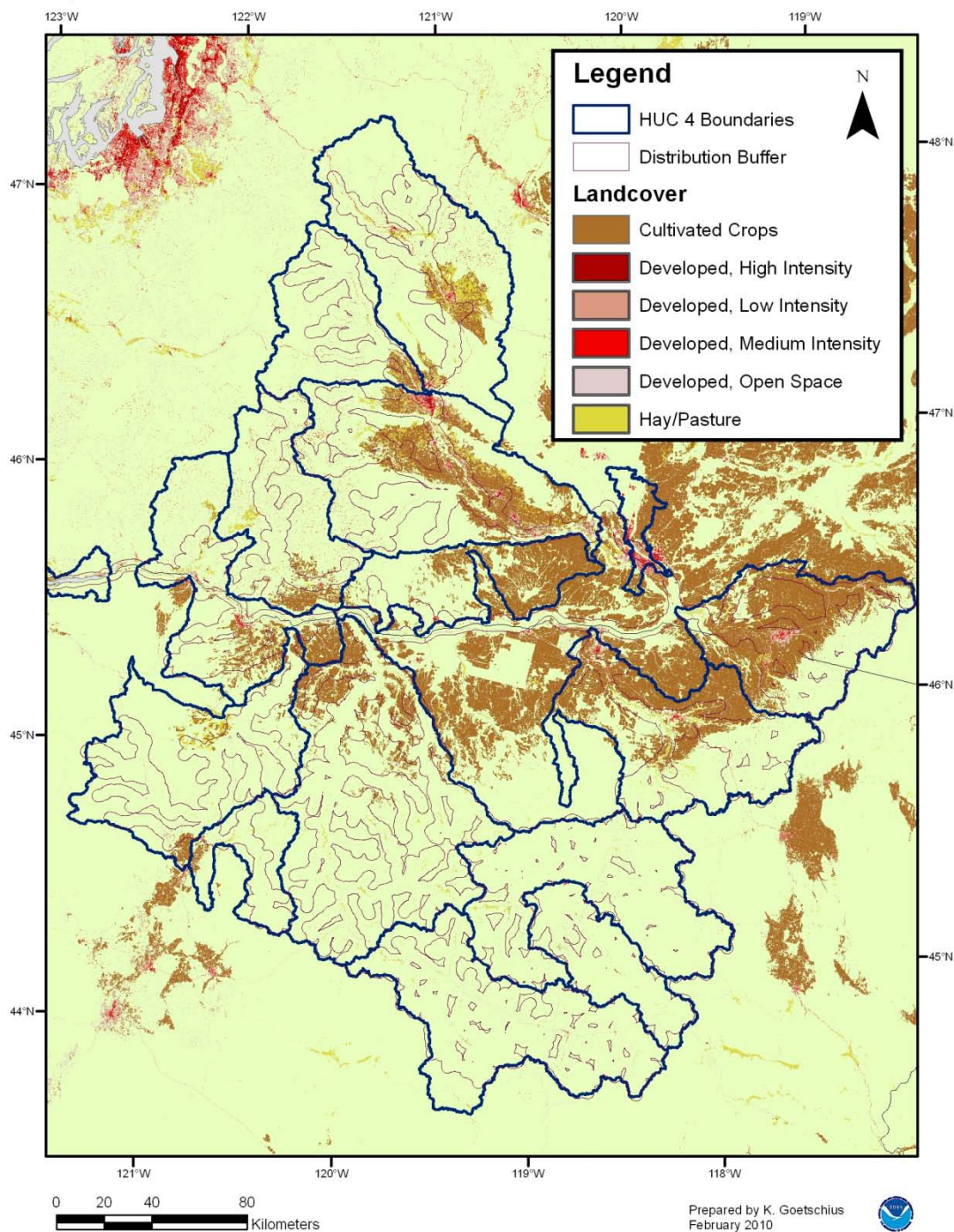
Upper Willamette River Steelhead DPS Species Distribution



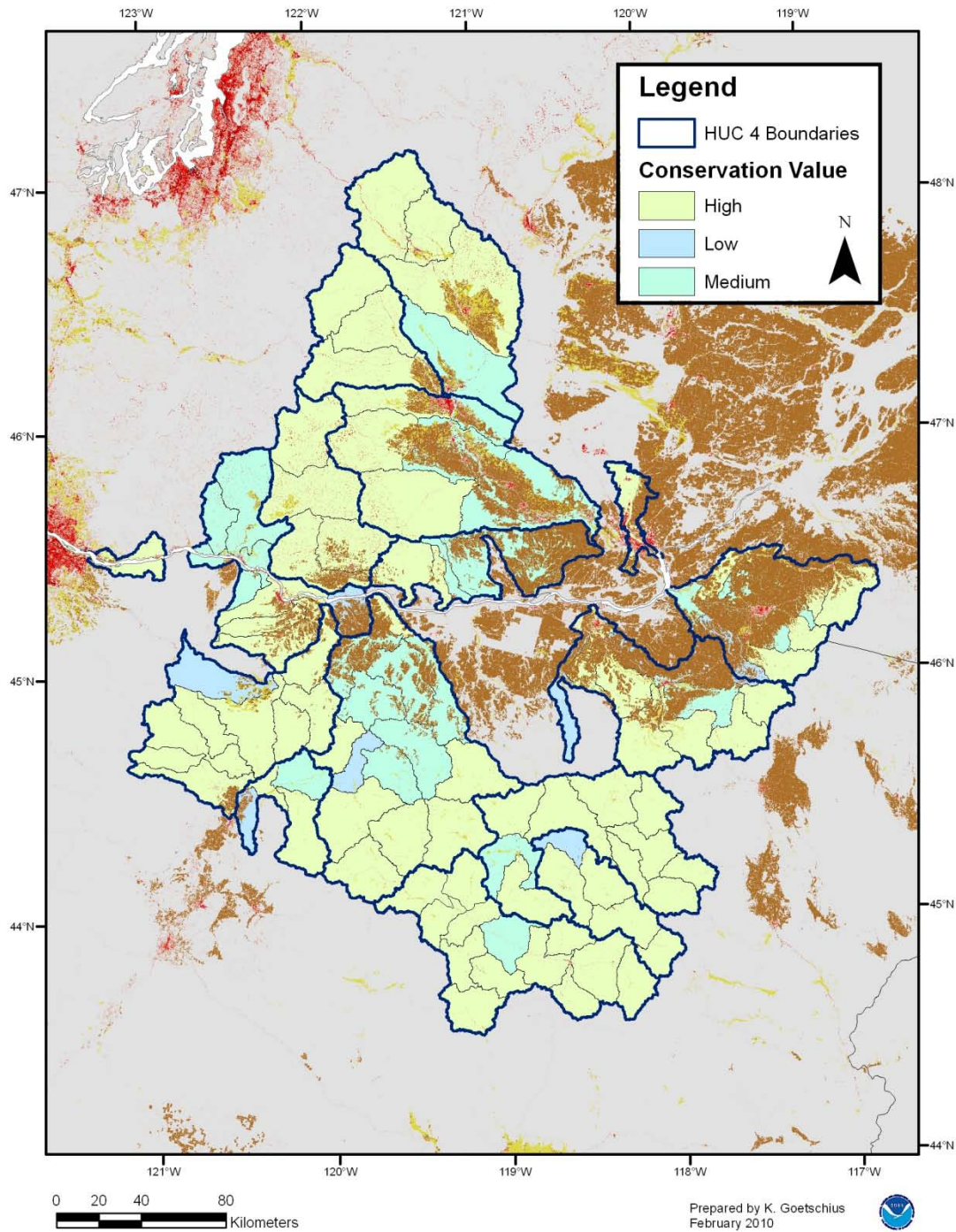
Upper Willamette River Steelhead DPS Critical Habitat



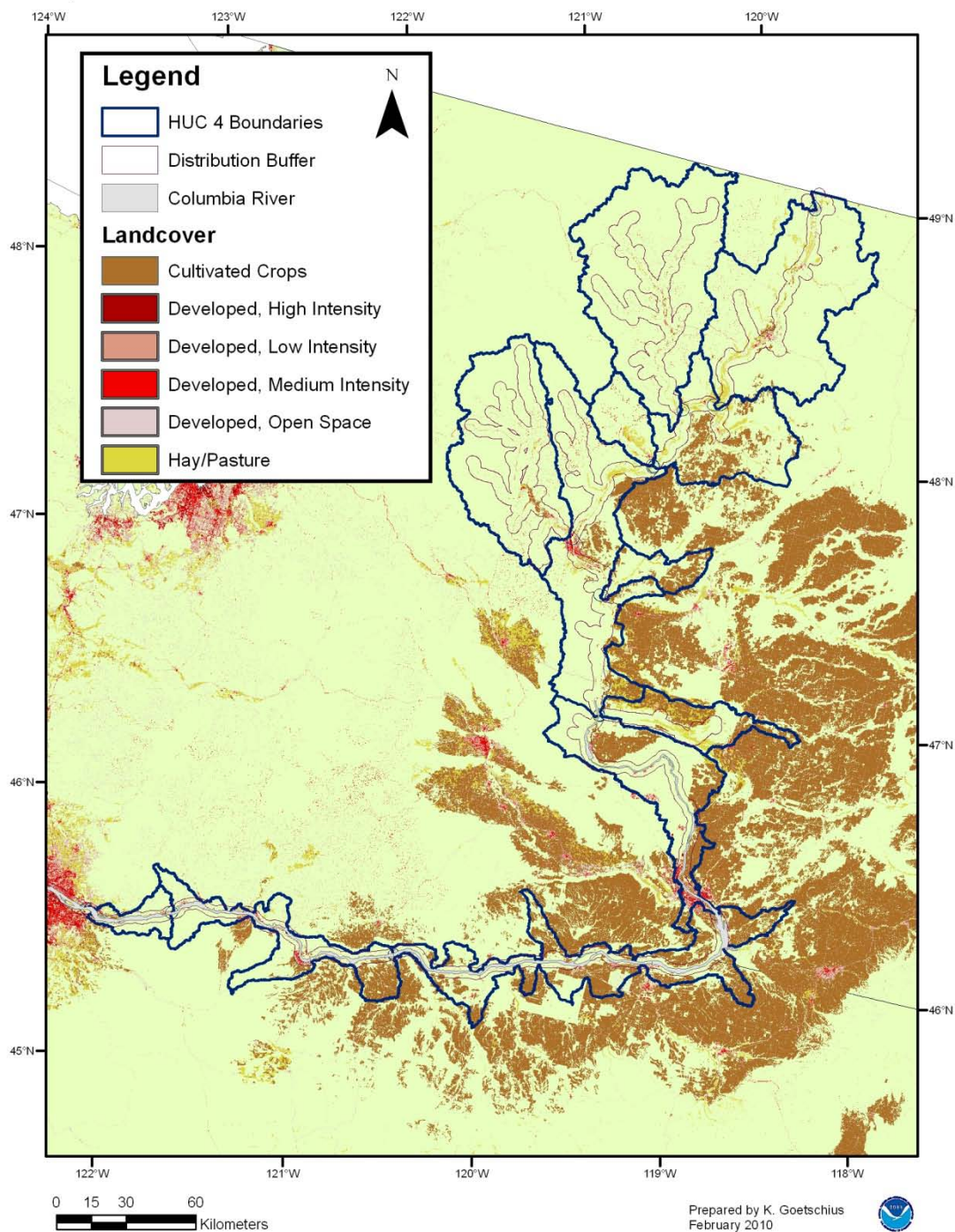
Middle Columbia River Steelhead DPS Species Distribution



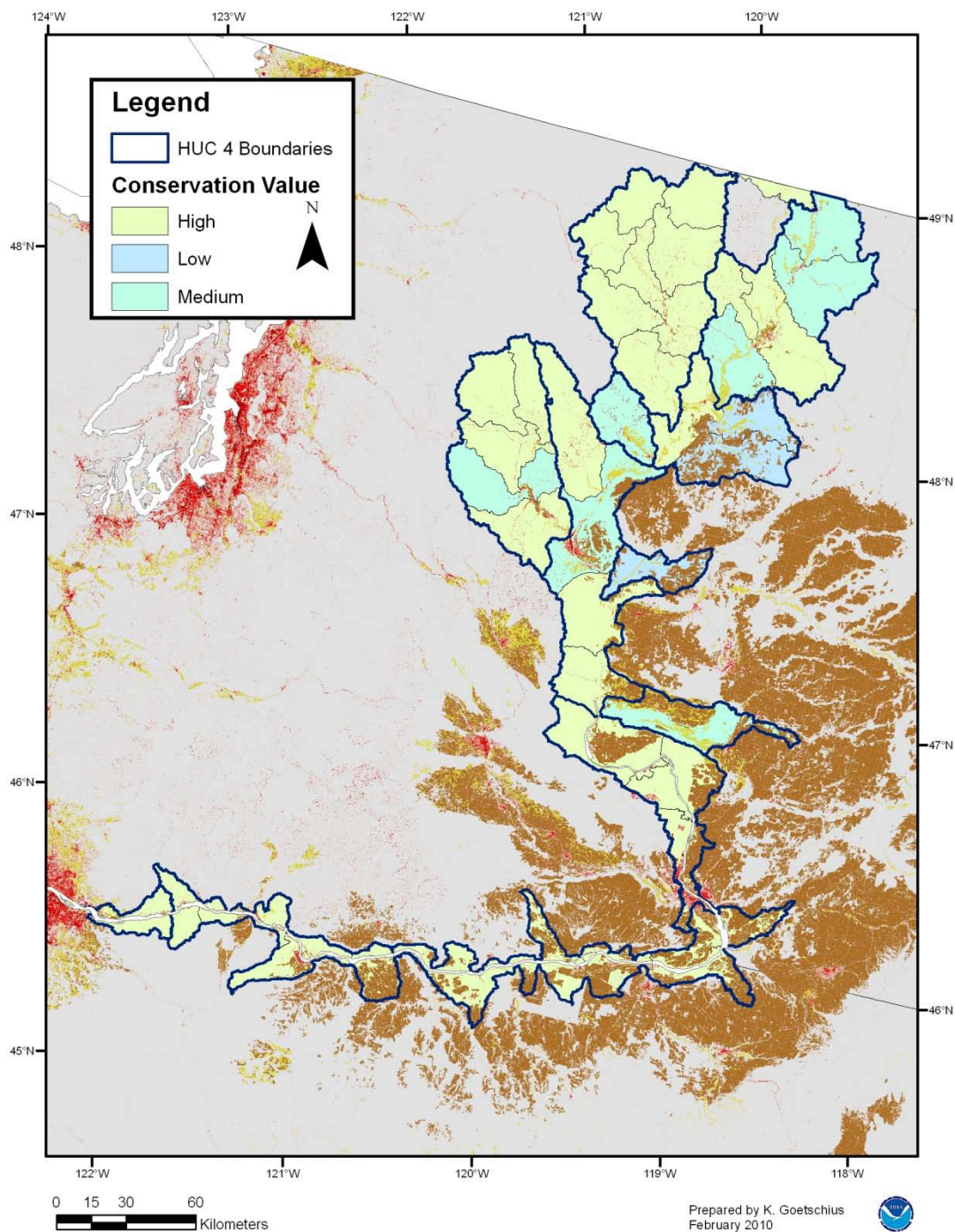
Middle Columbia River Steelhead DPS Critical Habitat



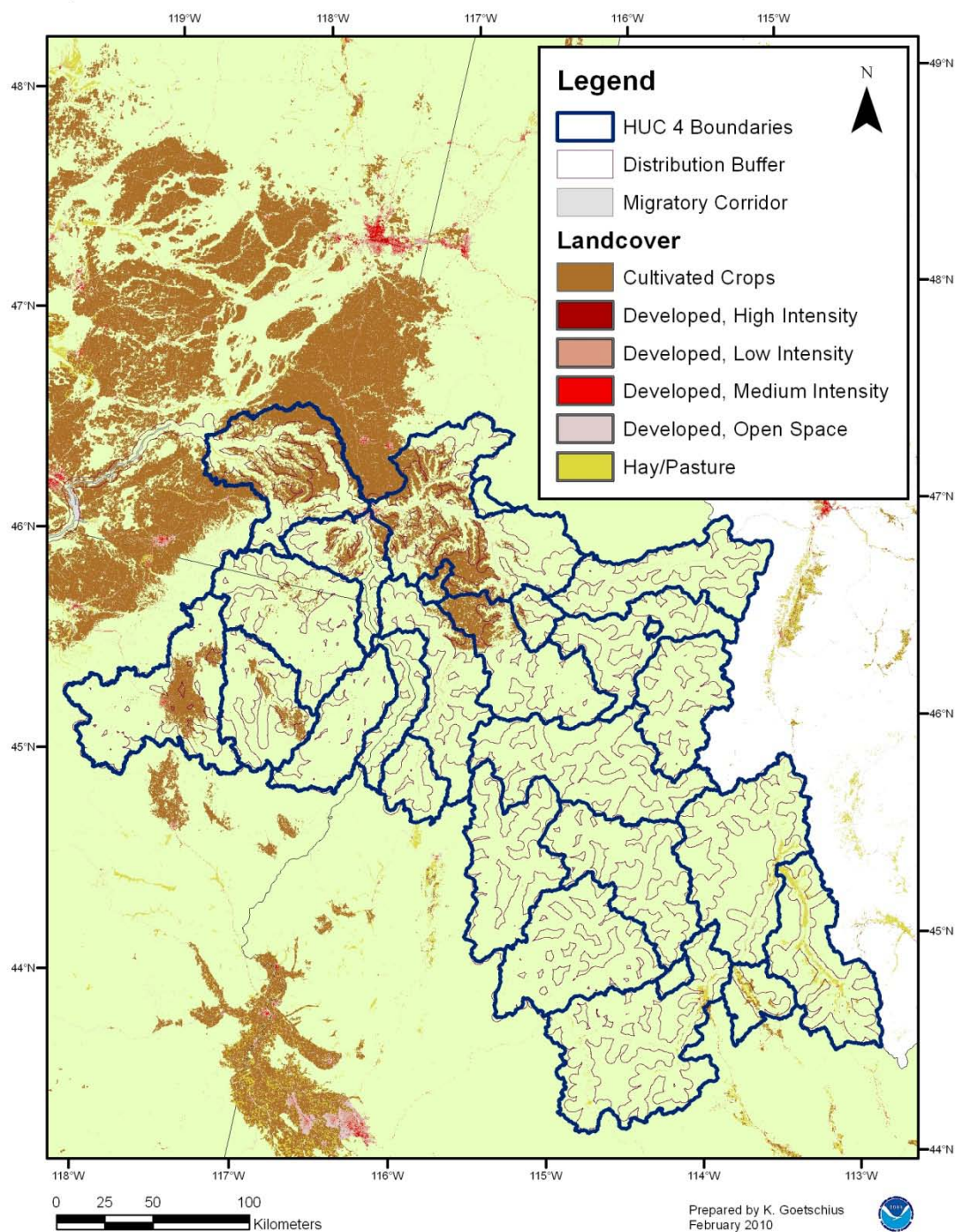
Upper Columbia River Steelhead DPS Species Distribution



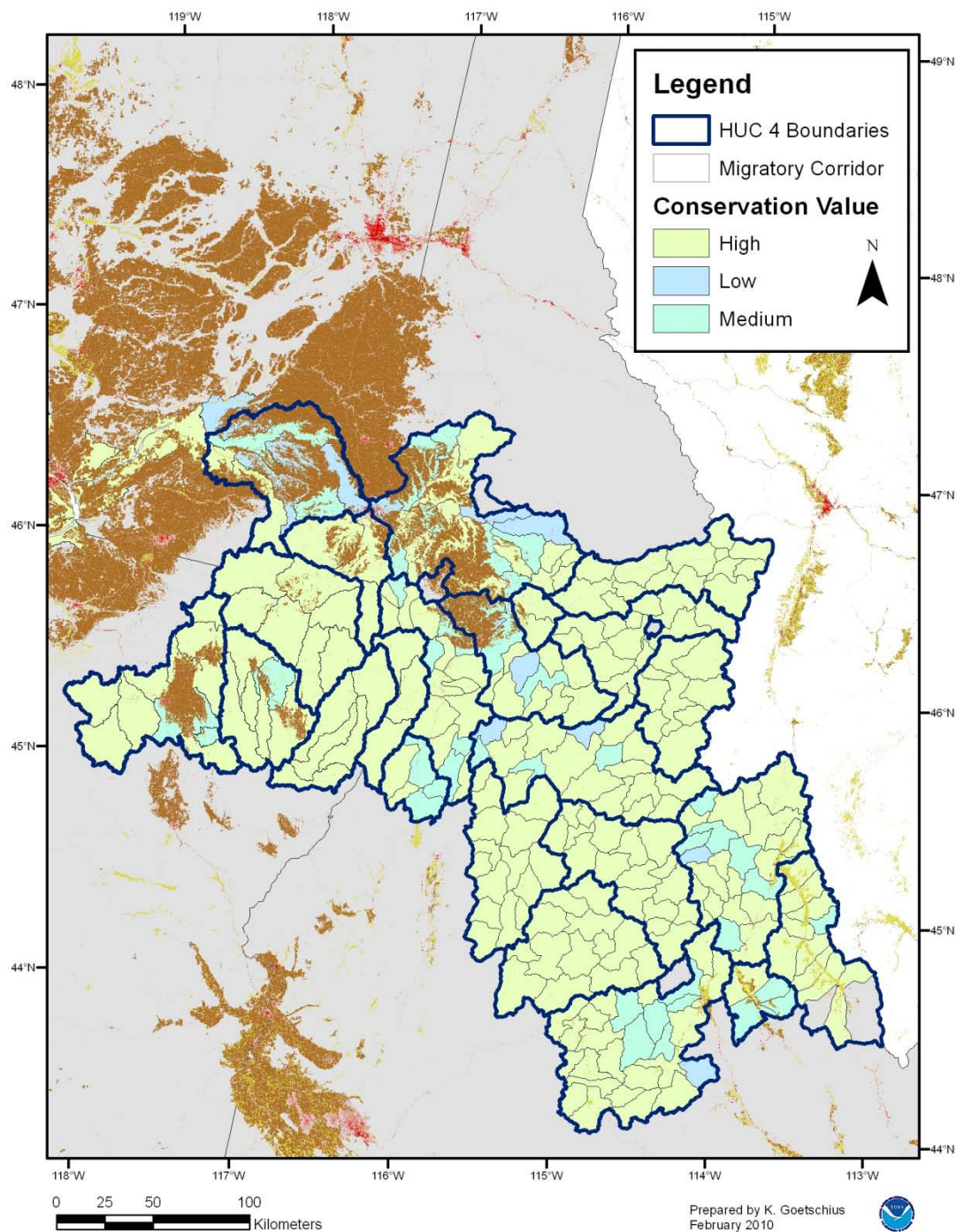
Upper Columbia River Steelhead DPS Critical Habitat



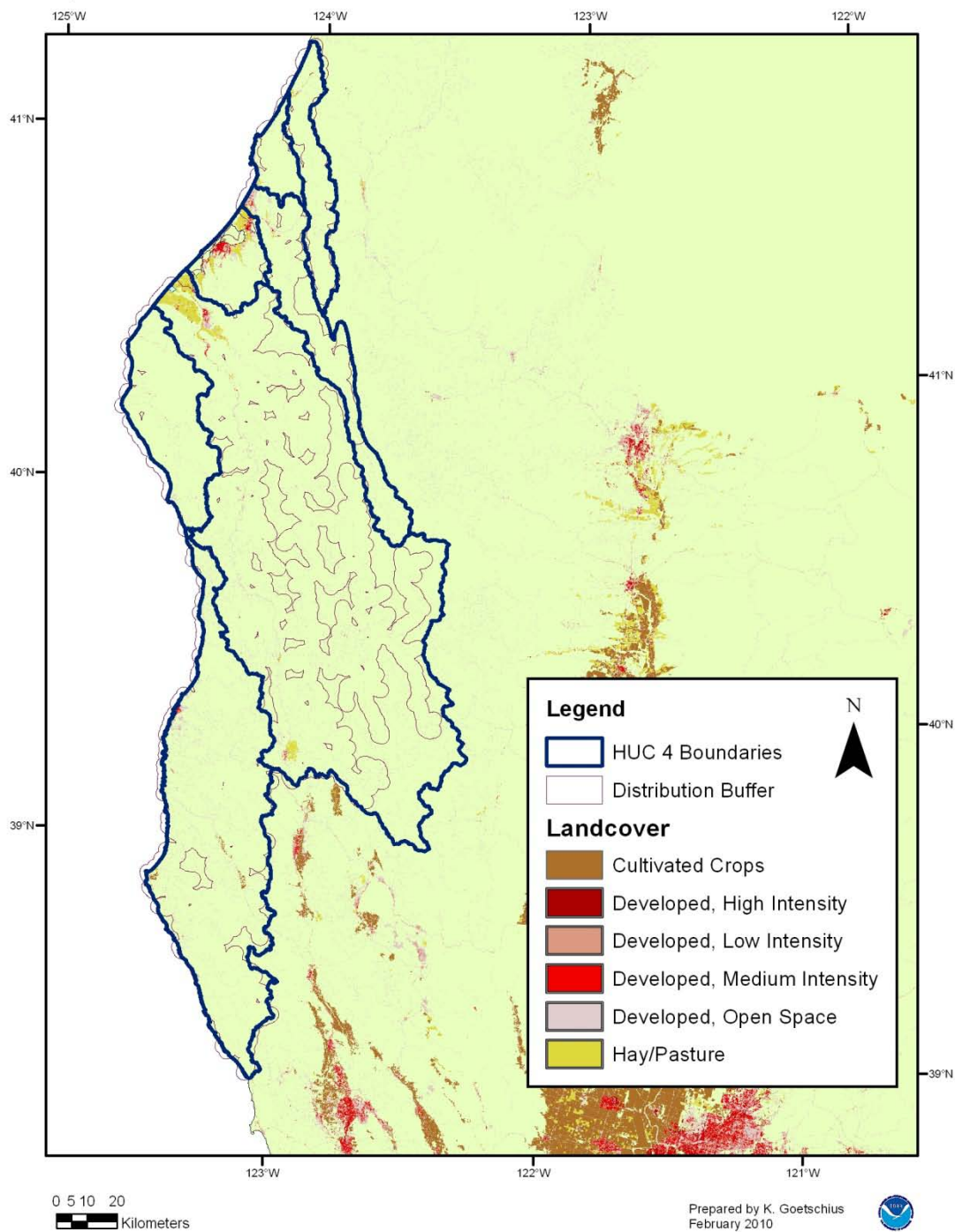
Snake River Steelhead DPS Species Distribution



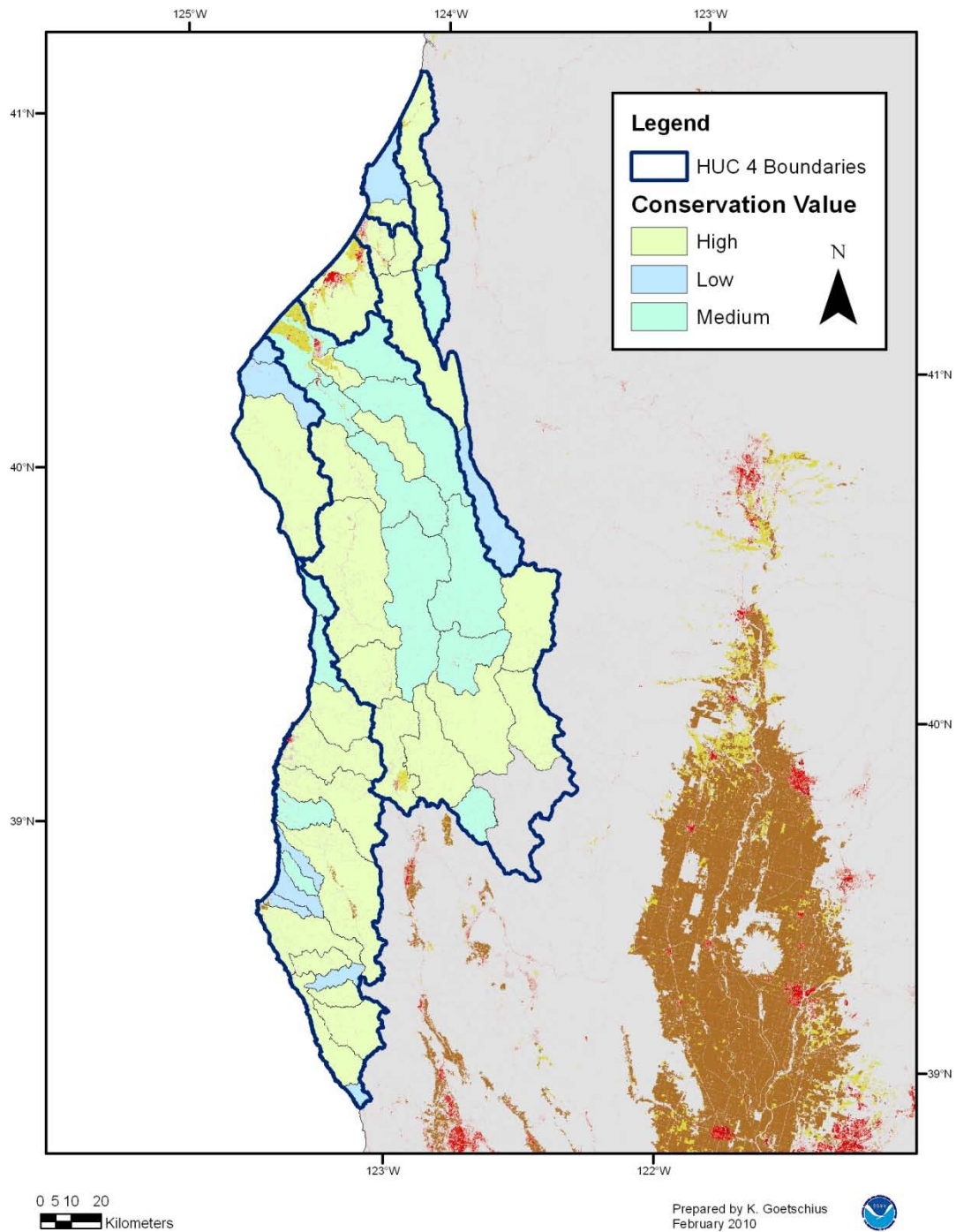
Snake River Steelhead DPS Critical Habitat



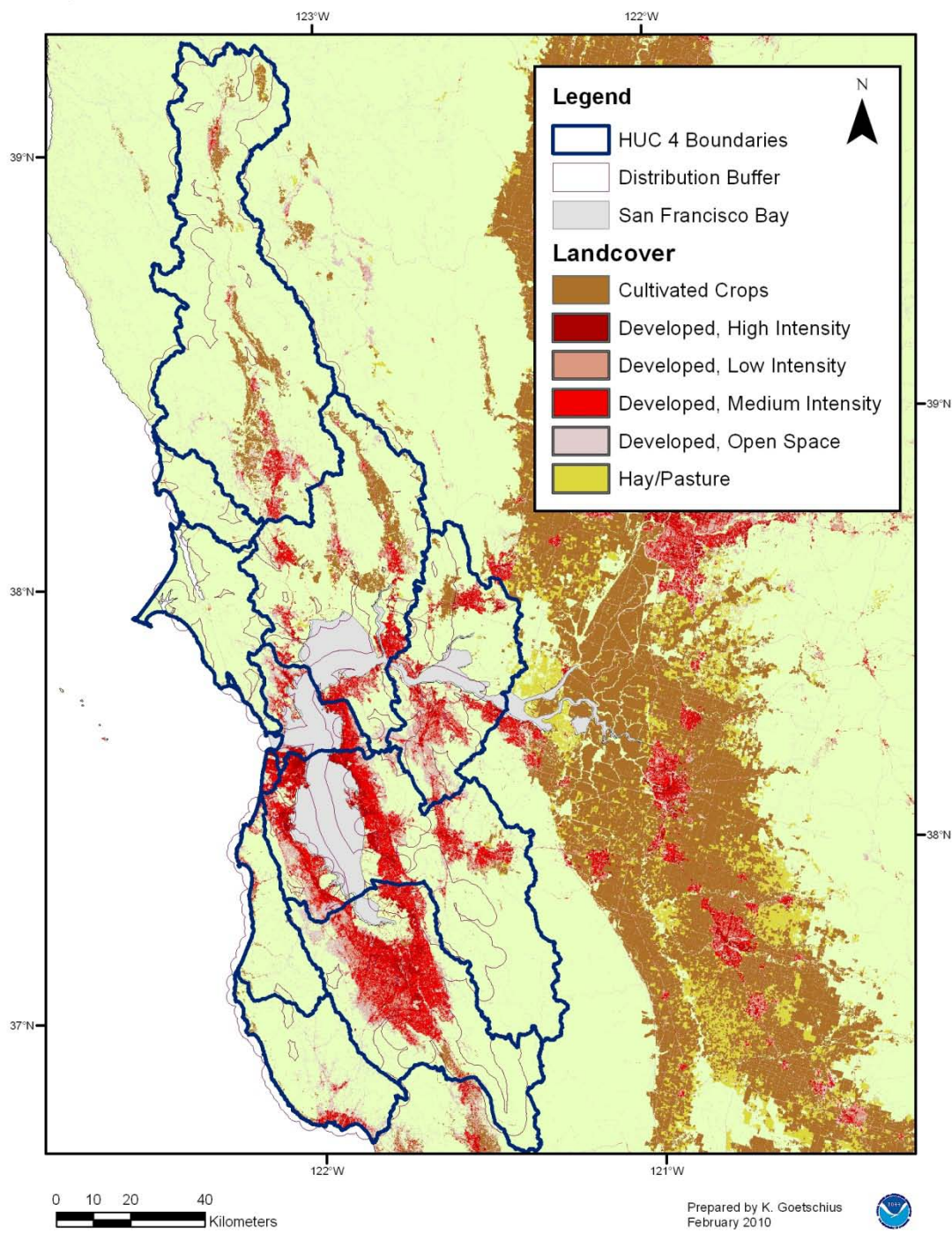
Northern California Steelhead DPS Species Distribution



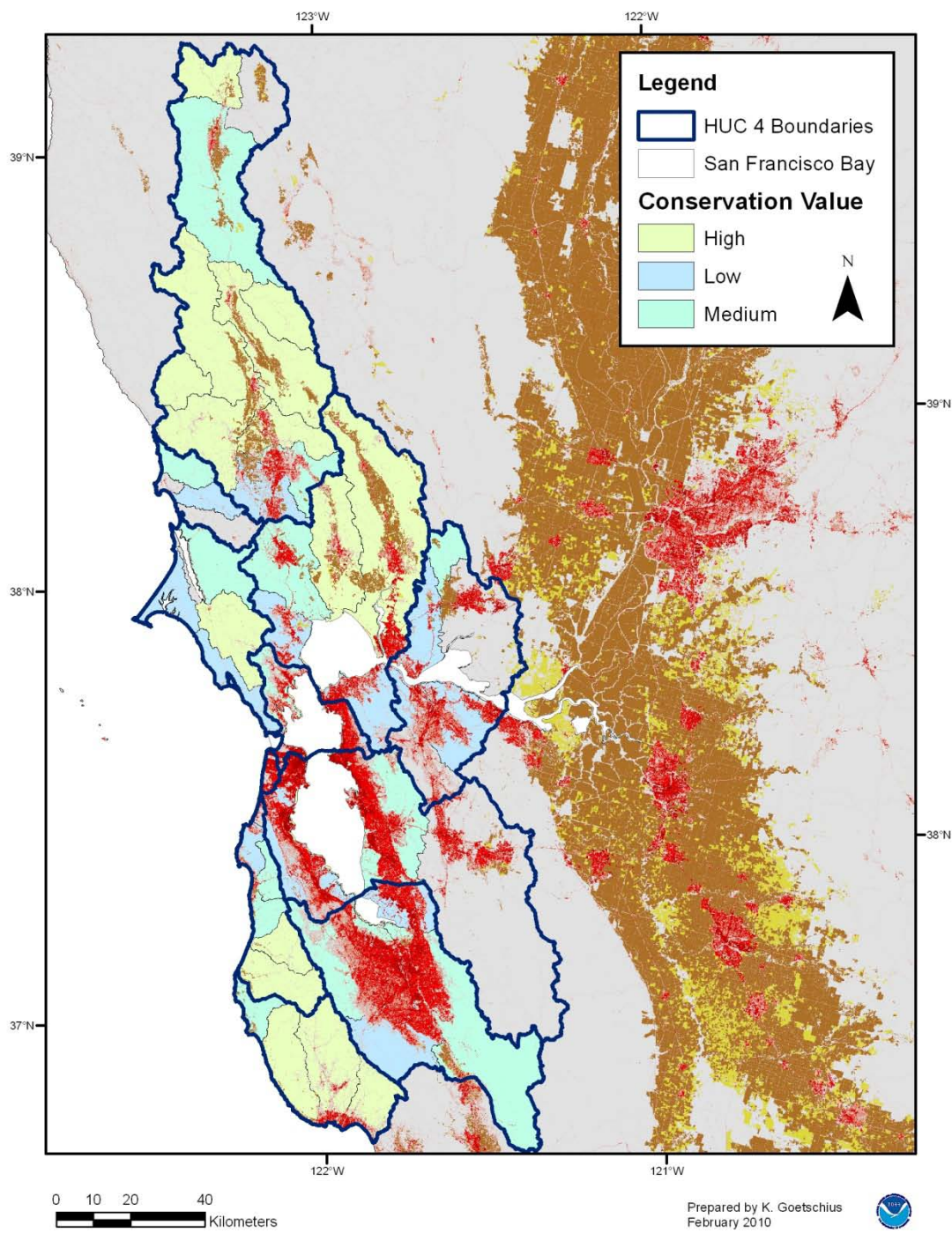
Northern California Steelhead DPS Critical Habitat



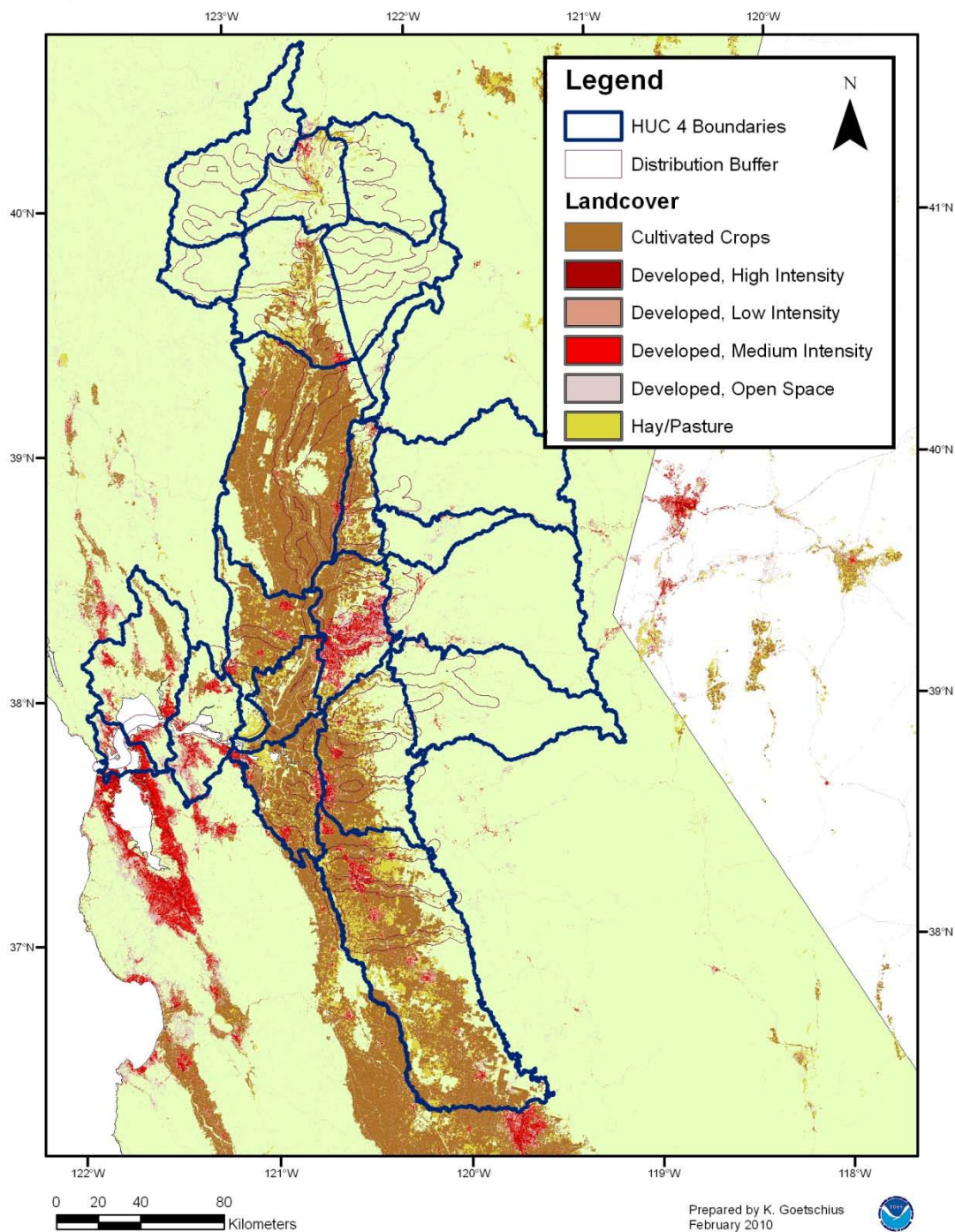
Central California Coast Steelhead DPS Species Distribution



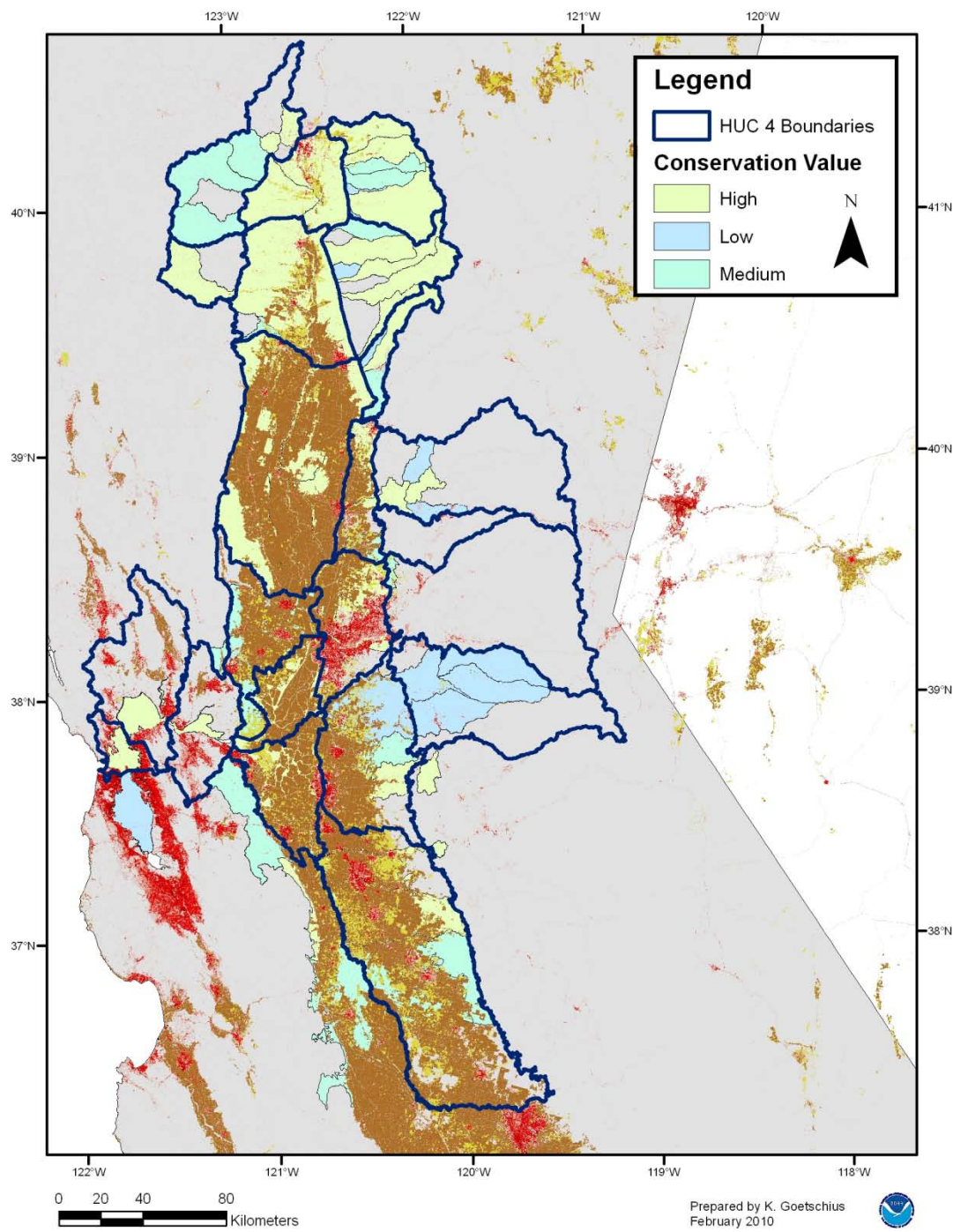
Central California Coast Steelhead DPS Critical Habitat



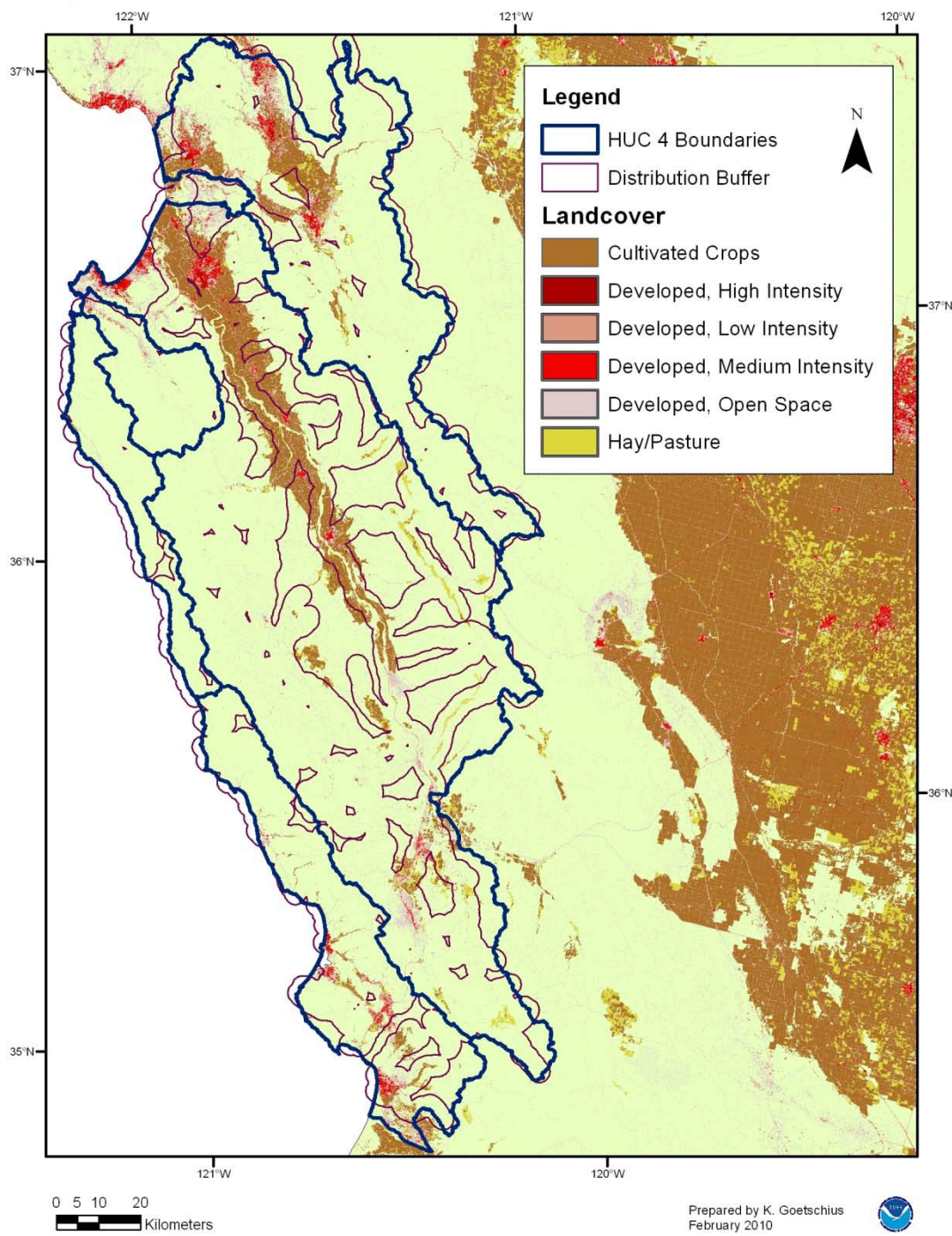
California Central Valley Steelhead DPS Species Distribution



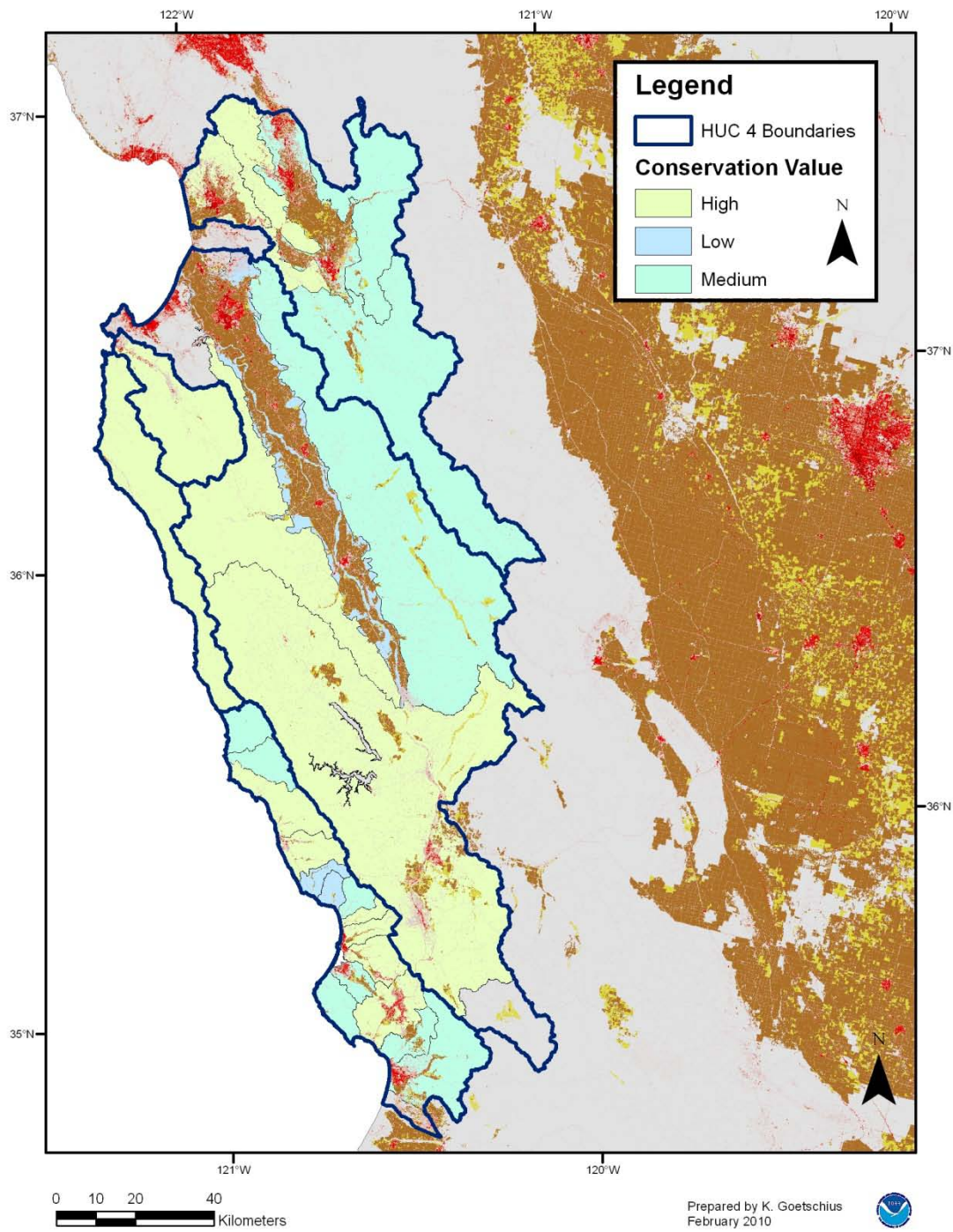
California Central Valley Steelhead DPS Critical Habitat



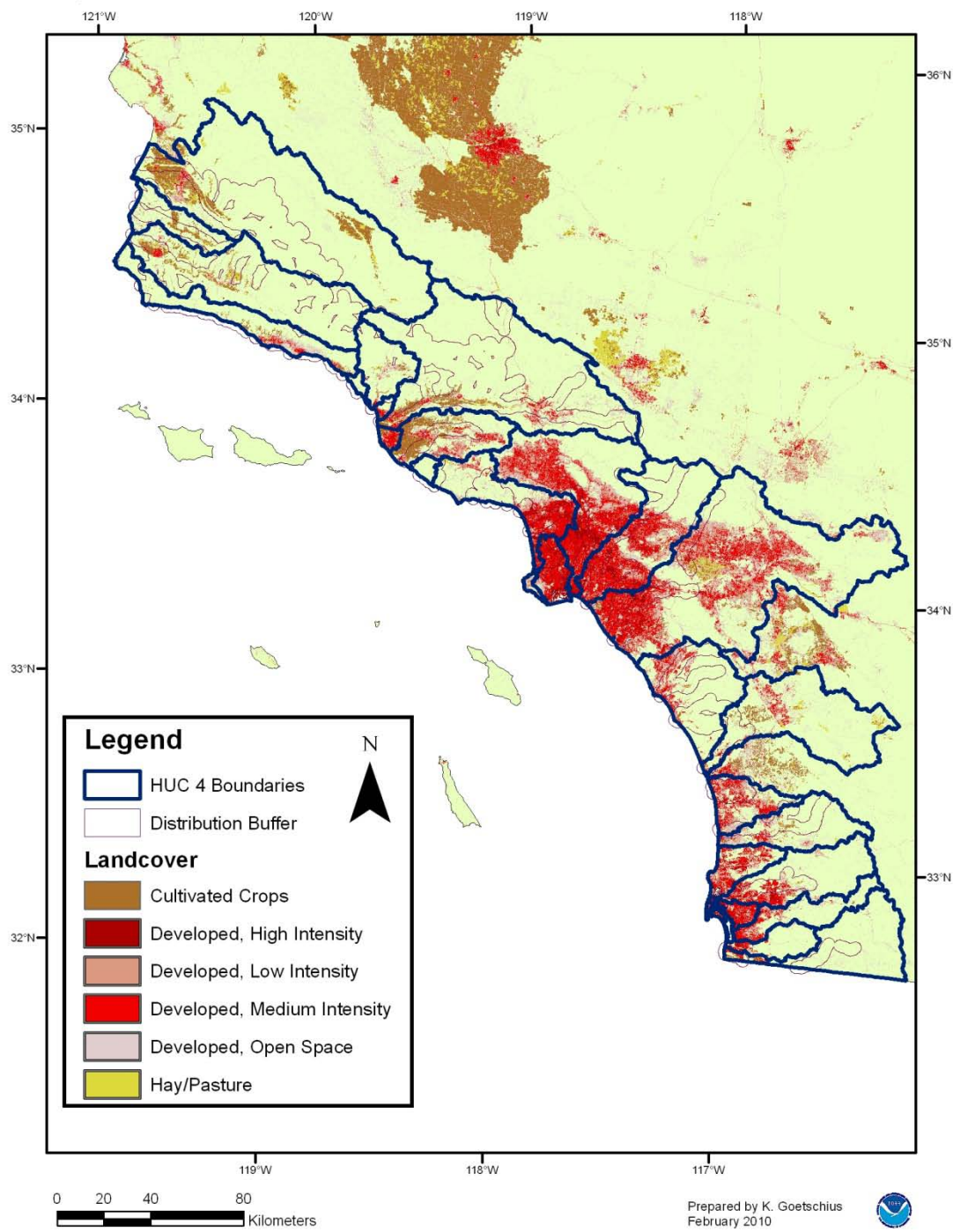
South-Central California Coastal Steelhead DPS Species Distribution



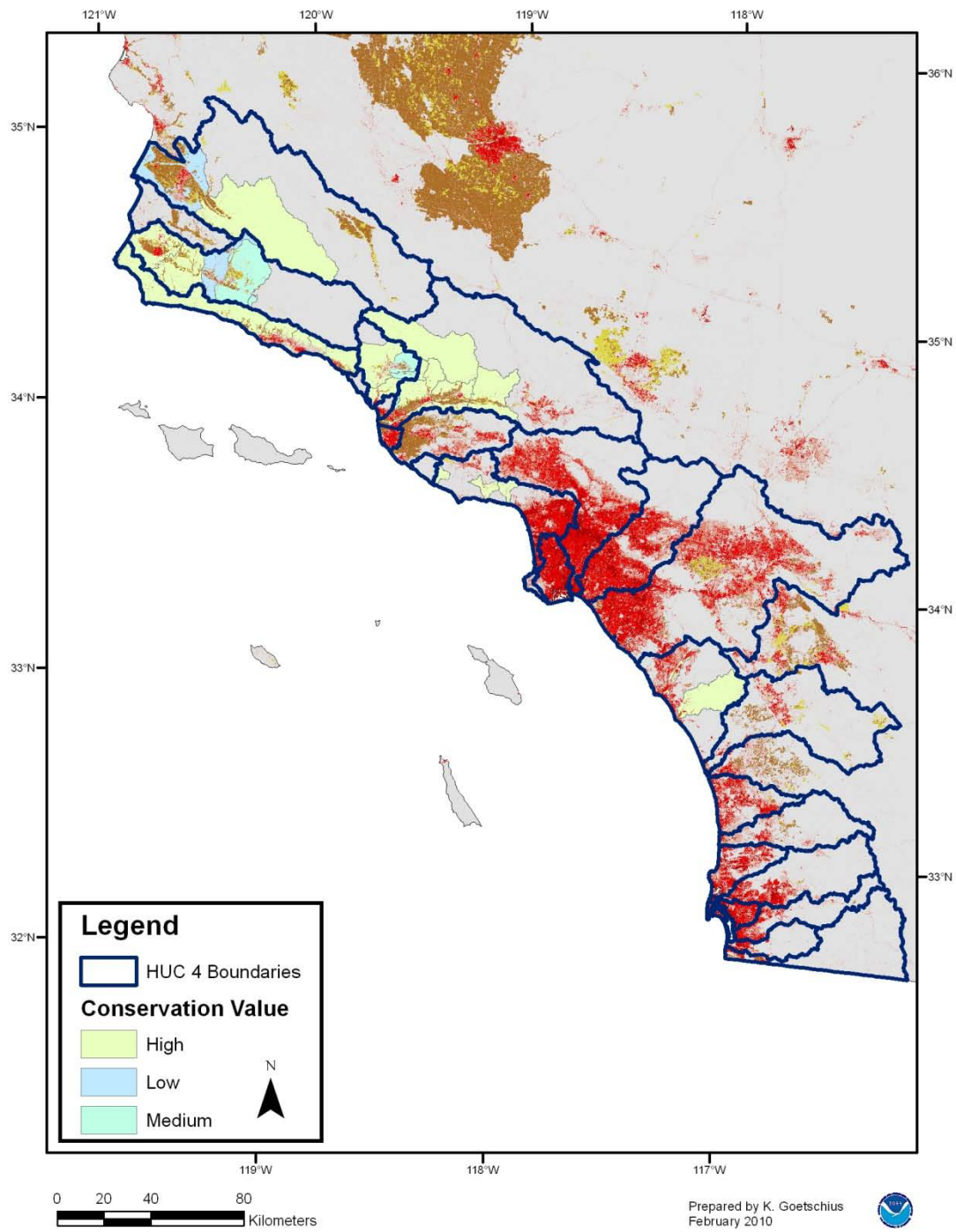
South-Central California Coastal Steelhead DPS Critical Habitat



Southern California Steelhead DPS Species Distribution



Southern California Steelhead DPS Critical Habitat



***Appendix 6: Generalized Average Annual Run-timing for ESA listed
Pacific Coast Salmon and Steelhead***

Washington State Listed ESU / DPS Life Histories Freshwater Phase Presence

Chinook Salmon

Puget Sound Chinook (spring/summer, fall combined)

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Lower Columbia River Chinook

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Upper Columbia River Spring-run Chinook (Endangered)

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Chum Salmon

Hood Canal Summer-run

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Columbia River Chum

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Coho Salmon

Lower Columbia River Coho

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Sockeye Salmon

Ozette Lake Sockeye

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Steelhead

Puget Sound Steelhead (winter/summer runs)

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Lower Columbia River Steelhead (winter/summer runs)

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Middle Columbia River Steelhead

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Upper Columbia River Steelhead

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Oregon / Idaho Listed ESU / DPS Life Histories

Steelhead

Upper Willamette River

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Snake River Basin Steelhead

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Chinook Salmon

Snake River Spring/Summer Run Chinook

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Snake River Fall Run Chinook

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Upper Willamette River Chinook

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Sockeye Salmon

Snake River Sockeye Salmon (Endangered)

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Coho Salmon

Oregon Coast Coho

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

California Listed ESU / DPS Life Histories

Coho

Central California Coast Coho

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Southern Oregon / North California Coast Coho

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Chinook

California Coastal

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases												
Rearing and migration (juveniles)												

Central Valley Spring-run Chinook

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Sacramento River Winter-run Chinook (endangered)

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Steelhead

Northern California

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Central California Coast Steelhead

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

California Central Valley Steelhead

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

South- Central California Coast Steelhead

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

Southern California Steelhead (endangered)

Life History phase	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Entering Fresh Water (adults/jacks)												
Spawning												
Incubation (eggs)												
Emergence (alevin to fry phases)												
Rearing and migration (juveniles)												

References Used:

Busby, P. J., T. C. Wainwright, G. J. Bryant, L. J. Lierheimer, R. S. Waples, F. W. Waknitz, and I. V. Lagomarsino. 1996. Status review of west coast steelhead from Washington, Idaho, Oregon, and California. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-27. 261 p.

Currens, K. P., R. R. Fuerstenberg, W. H. Graeber, K. Rawson, M. H. Ruckelshaus, N. J. Sands, and J. B. Scott. 2009. Identification of an independent population of sockeye salmon in Lake Ozette, Washington. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-96. 18 p.

Good, T.P., R.S. Waples, and P. Adams (editors). 2005. Updated status of federally listed ESUs of West Coast salmon and steelhead. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-66. 598 p.

Haggerty, M., A. Ritchie, J. Shellberg, M. Crewson, J. Jalonon. 2008. Lake Ozette Sockeye Limiting Factors Analysis. Prepared for the Makah Indian Tribe and NOAA NMFS, in cooperation with the Lake Ozette Sockeye Steering Committee.

Johnson, O. W., W. S. Grant, R. G. Kope, K. Neely, F. W. Waknitz, and R. S. Waples. 1997. Status review of chum salmon from Washington, Oregon, and California. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-32. 280 p.

Matthews, G. M., R.S. Waples. 1991. Status review for Snake River spring and summer Chinook salmon. U.S. Dept. Commer., NOAA Technical Memo. NMFS F/NWC-200. 75 p.

Myers, J.M., R.G. Kope, G.J. Bryant, D. Teel, L.J. Lierheimer, T.C. Wainwright, W.S. Grant, F.W. Waknitz, K. Neely, S.T. Lindley, and R.S. Waples. 1998. Status review of chinook salmon from Washington, Idaho, Oregon, and California. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-35. 443 p.

Oregon Department of Fish and Wildlife. Natural Resources Information Management Program.

<http://nrimp.dfw.state.or.us/nrimp/default.aspx?p=259>

NOAA. Northwest Regional Office ESA Salmon Listings. <http://www.nwr.noaa.gov/ESA-Salmon-Listings/Index.cfm>

Steiner Environmental Consulting. August 1996. A History of the Salmonid Decline in the Russian River. Steiner Environmental Consulting.

U.S. Fish and Wildlife Service. August 1987. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Southwest)/ Coho Salmon. Biological Report 82 (11.70).

U.S. Fish and Wildlife Service. April 1986. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Southwest)/ Chinook Salmon. Biological Report 82 (11.49).

U.S. Fish and Wildlife Service. June 1986. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Southwest)/ Steelhead Trout. Biological Report 82 (11.60).

Waknitz, F.W., G. M. Matthews, T. Wainwright, and G. A. Winans. 1995. Status review for mid-Columbia River summer Chinook salmon. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-22. 80 p.

Waples, R. S., R. P. Jones, B. R. Beckman, and G. A. Swan. 1991. Status review for Snake River fall Chinook salmon. U.S. Dept. Commer., NOAA Tech. Memo. NMFS F/NWC-201. 73 p.

Washington Department of Fish and Wildlife. 2010. Salmon stock inventory (SaSI) database. Biological Data Systems Unit. Data set accessed 2010-12-09.

Weitkamp, L. A., T. C. Wainwright, G. J. Bryant, G. B. Milner, D. J. Teel, R. G. Kope, and R. S. Waples. 1995. Status review of coho salmon from Washington, Oregon and California. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-24. 258 p.

West Coast Chinook Salmon Biological Review Team. 1998. Status Review Update for West Coast Chinook Salmon (*Oncorhynchus tshawytscha*) from Puget Sound, Lower Columbia River, Upper Willamette River, and Upper Columbia River Spring-Run ESUs. 62p.

West Coast Chinook Salmon Biological Review Team. 1999. Status Review Update for Deferred ESUs of West Coast Chinook Salmon (*Oncorhynchus tshawytscha*) from Washington, Oregon, California, and Idaho. 115p.

West Coast Chum Salmon Biological Review Team. 1999. Status review update for chum salmon from Hood Canal summer-run and Columbia River ESUs. U.S. Dept. Commer. NOAA Predecisional ESA Document. 27 p.

West Coast Coho Salmon Biological Review Team. 1997. Status Review Update for Coho Salmon from the Oregon and Northern California Coasts. 103p.

Appendix 7 – Ecological Effects Data from EPA's Red Legged Frog BE's

(Referred to as “Appendix F” in those documents)

Appendix 8. AgDrift Tier III aerial assessment input values for mixture calculation

Aircraft: Air Tractor AT-401

Boom length: 76.3%

Boom height: 10 feet

Flight lines: 1

Swath width definition: Fixed, 60 feet

Swath width displacement: 0.3702 fractional displacement

Droplet size distribution: ASAE course

Nonvolatile rate: 13.5 lb/A

Active Rate: 12 lbs diuron/A, 1.5 lbs imazapyr/A

Spray volume rate: 20 gallons/A

Carrier: water

Windspeed: 10 mph

Temperature: 86 degrees F

Relative humidity: 50%

Flux plane: 0 ft

Distance to water body from edge of application area: 0 and 300 feet

Aquatic definition: EPA-defined pond and NMFS-defined floodplain habitat (2 m wide, 0.1 m deep)

Appendix F
Ecological Effects Data

Table F-1. Acute freshwater fish toxicity to 2,4-D							
Species	% ai	96-hour LC50 (95% CI)		Measured/ Nominal Test Concentrations	Toxicity Category (based on ae)	MRID No. Author/Year	Study Classification
		(mg ai/L)	(mg ae/L)				
2,4-D acid (PC Code: 030001)							
Rainbow trout (<i>Oncorhynchus mykiss</i>)	98.7	358	358	Measured	Practically non-toxic	411583-01, Alexander et. al., 1983	Supplemental
Bluegill sunfish (<i>Lepomis macrochirus</i>)	98.7	263	263	Measured	Practically non-toxic	411583-01, Alexander et. al., 1983	Supplemental
Fathead minnow (<i>Pimephales promelas</i>)	98.7	320	320	Measured	Practically non-toxic	411583-01, Alexander et. al., 1983	Supplemental
2,4-D Sodium Salt (PC Code: 030004)							
Rainbow trout (<i>Oncorhynchus mykiss</i>)	80	>100	>91	Nominal	Practically non-toxic	53986, McCann, 1973	Acceptable
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)							
Rainbow trout (<i>Oncorhynchus mykiss</i>)	73.1	>120	>81.6	Measured	Practically non-toxic	419751-05, Graves. et. al., 1991.	Acceptable
Bluegill sunfish (<i>Lepomis macrochirus</i>)	73.1	>121	>82.3	Measured	Practically non-toxic	419751-04, Graves. et. al., 1991.	Acceptable
Fatahead minnow (<i>Pimephales promelas</i>)	73.1	344	234	Measured	Practically non-toxic	419751-04, Graves, et. al, 1991	Acceptable
Bluegill sunfish (<i>Lepomis macrochirus</i>)	Not Reported	149	101	Measured	Practically non-toxic	0073-091-01, Sleight, B., 1971.	Supplemental
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)							
Rainbow trout (<i>Oncorhynchus mykiss</i>)	67.3	>1000	>830	Measured	Practically non-toxic	233350, Vilkas, A,G,, 1977.	Acceptable

Table F-1. Acute freshwater fish toxicity to 2,4-D							
Species	% ai	96-hour LC50 (95% CI)		Measured/ Nominal Test Concentrations	Toxicity Category (based on ae)	MRID No. Author/Year	Study Classification
		(mg ai/L)	(mg ae/L)				
Bluegill sunfish (<i>Lepomis macrochirus</i>)	73.1	>121	>100	Measured	Practically non-toxic	419751-04, Graves, et. al., 1991.	Acceptable
Bluegill sunfish (<i>Lepomis macrochirus</i>)	67.3	250	207.5	Measured	Practically non-toxic	411583-11, Alexander, et. al., 1983.	Acceptable
Bluegill sunfish (<i>Lepomis macrochirus</i>)	51.1 9TEP)	>1000	>830	Measured	Practically non-toxic	234027, Vilkas, A.G., 1978.	Supplemental
Fathead minnow (<i>Pimephales promelas</i>)	67.3	318	264	Measured	Practically non-toxic	419751-04, Graves. et. al., 1991.	Acceptable
2,4-D Isoproylamine (IPA) (PC Code: 030025)							
Rainbow trout (<i>Oncorhynchus mykiss</i>)	48.7 (TEP)	2840	2244	Measured	Practically non-toxic	01338869, 1983.	Acceptable
Bluegill sunfish (<i>Lepomis macrochirus</i>)	48.7 (TEP)	1700	1343	Measured	Practically non-toxic	01338869, 1983.	Acceptable
Fathead minnow (<i>Pimephales promelas</i>)	48.7 (TEP)	2180	1722	Measured	Practically non-toxic	01338869, 1983.	Supplemental
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)							
Rainbow trout (<i>Oncorhynchus mykiss</i>)	69.2	300	162	Measured	Practically non-toxic	413538-03, Mayes, et. al., 1989.	Acceptable
Bluegill sunfish (<i>Lepomis macrochirus</i>)	69.2	401	217	Measured	Practically non-toxic	413538-04, Mayes, et al., 1989	Acceptable
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)							
Rainbow trout (<i>Oncorhynchus mykiss</i>)	97.4	2.09	1.44	Measured	Moderately toxic	413538-01, Alexander, et. al, 1983.	Acceptable

Table F-1. Acute freshwater fish toxicity to 2,4-D							
Species	% ai	96-hour LC50 (95% CI)		Measured/ Nominal Test Concentrations	Toxicity Category (based on ae)	MRID No. Author/Year	Study Classification
		(mg ai/L)	(mg ae/L)				
Rainbow trout (<i>Oncorhynchus mykiss</i>) static	77.5	0.65 (56- hour LC50)	0.45	Measured	Highly toxic	00050674, Pitcher, F.G., 1974.	Supplemental
Bluegill sunfish (<i>Lepomis macrochirus</i>)	Not reported	1.2	0.828	Measured	Moderately toxic	400980-01, Mayer, 1986.	Supplemental
Bluegill sunfish (<i>Lepomis macrochirus</i>)	97.4	0.62	0.428	Measured	Highly toxic	413538-01, Alexander, et. al, 1983.	Acceptable
Bluegill sunfish (<i>Lepomis macrochirus</i>)	29.0 (TEP)	>100	0.69	Measured	Practically non- toxic	400980-01, Mayer, 1986 J.A., 1969	Supplemental
Fathead minnow (<i>Pimephales promelas</i>)	97.4	2.60	1.79	Measured	Moderately toxic	413538-01, Alexander, et. al, 1983.	Acceptable
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)							
Rainbow trout (<i>Oncorhynchus mykiss</i>)	66.9 (TEP)	4.82	3.2	Measured EHE	Moderately toxic	417373-03, Mayes, et. al., 1990.	Acceptable
Rainbow trout (<i>Oncorhynchus mykiss</i>)	92	22	14.5	Nominal	Slightly toxic	45068, Buccafusco, R.J., 1976.	Acceptable
Bluegill sunfish (<i>Lepomis macrochirus</i>)	92	18	11.9	Nominal	Slightly toxic	45069, Buccafusco, R.J., 1976.	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)							
Rainbow trout (<i>Oncorhynchus mykiss</i>) static	98.2	0.69	0.58	Measured	Highly toxic	439331-01, Drottar, et. al., 1996.	Acceptable

Table F-1. Acute freshwater fish toxicity to 2,4-D							
Species	% ai	96-hour LC50 (95% CI)		Measured/ Nominal Test Concentrations	Toxicity Category (based on ae)	MRID No. Author/Year	Study Classification
		(mg ai/L)	(mg ae/L)				
Rainbow trout (<i>Oncorhynchus mykiss</i>) static	45.8 (TEP)	0.78	0.66	Measured	Highly toxic	439332-01, Drottar, et. al., 1996.	Acceptable
Bluegill sunfish (<i>Lepomis macrochirus</i>) static	98.2	0.31	0.26	Measured	Highly toxic	439307-01, Drottar, et. al., 1996.	Acceptable
Bluegill sunfish (<i>Lepomis macrochirus</i>)	45.8 (TEP)	0.31	0.26	Measured	Slightly toxic	439103-01, Drottar, et. al., 1996.	Acceptable

Table F-2. Chronic (early life cycle) freshwater fish toxicity to 2,4-D						
Species	% ai	Toxicity Value (mg ae/L)		Most sensitive endpoint	MRID No. Author/Year	Study Classification
		NOAEC	LOAEC			
2,4-D acid (PC Code: 030001)*						
Fathead minnow (<i>Pimephales promelas</i>)	96.1	63.4	102	Larval survival	417373-04, Mayes, et. al.,1990.	Acceptable
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)*						
Fathead minnow (<i>Pimephales promelas</i>)	73.8	19.8	66.6	Larval survival	420183-04, Graves, et. al., 1991.	Acceptable
2,4-D Dimethylamine (DMA)Salt (PC Code: 030019)*						
Fathead minnow (<i>Pimephales promelas</i>)	66.5	14.2	23.6	Length	417677-01, Dill, et. al., 1990.	Acceptable

* No early life cycle freshwater fish studies using forms of 2,4-D other than acid, DEA, and DMA were submitted to the Agency.

Table F-3. Chronic (full life cycle) freshwater fish toxicity to 2,4-D						
Species	% ai	Toxicity Value (mg ae/L)		Most sensitive endpoint	MRID No. Author/Year	Study Classification
		NOAEC	LOAEC			
2,4-D Ethylhexyl Ester (EHE) (PC code: 030063)*						
Fathead minnow (<i>Pimephales promelas</i>)	94.7	0.0792	0.1452	Larval fish survival	417373-05, Mayes, et. al., 1990.	Supplemental

* No full life cycle freshwater fish studies using forms of 2,4-D other than EHE were submitted to the Agency.

Table F-4. Acute freshwater invertebrate toxicity to 2,4-D							
Species	% ai	96-hour LC50 (95% CI)		Measured/ Nominal Test Concentrations	Toxicity Category (based on ae)	MRID No. Author/Year	Study Classification
		(mg ai/L)	(mg ae/L)				
2,4-D acid (PC Code: 030001)							
Waterflea (<i>Daphnia magna</i>)	98.7	25	25	Measured	Slightly toxic	411583-01, Alexander et. al., 1983	Acceptable
2,4-D Sodium Salt (PC Code: 030004)							
No data							
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)							
Waterflea (<i>Daphnia magna</i>)	71.3	>100	>68	measured	Practically non-toxic	419751-06, Graves, et. al., 1991.	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)							
Waterflea (<i>Daphnia magna</i>)	51.1 (TEP)	774.5	642.8	NA	Practically non-toxic	232630, Vilkas, A.G., 1977.	Acceptable
Waterflea (<i>Daphnia magna</i>)	67.3	184	153	NA	Practically non-toxic	411583-11	Acceptable
2,4-D Isoproylamine (IPA) (PC Code: 030025)							
Waterflea (<i>Daphnia magna</i>)	48.7	583	461	NA	Practically non-toxic	00138869, Alexander et. al., 1983.	Acceptable
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)							
Waterflea (<i>Daphnia magna</i>)	69.2	630	340.2	measured	Practically non-toxic	413538-05, Mayes, 1989	Acceptable
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)							

Table F-4. Acute freshwater invertebrate toxicity to 2,4-D							
Species	% ai	96-hour LC50 (95% CI)		Measured/ Nominal Test Concentrations	Toxicity Category (based on ae)	MRID No. Author/Year	Study Classification
		(mg ai/L)	(mg ae/L)				
Waterflea (<i>Daphnia magna</i>)	97.4	7.2	4.97	measured	Moderately toxic	413538-01, Alexander, et. al, 1983.	Acceptable
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)							
Waterflea (<i>Daphnia magna</i>)	92	18	11.88	Measured	Slightly toxic	67328, Kuc, W.J., 1977.	Acceptable
Waterflea (<i>Daphnia magna</i>)	96.2	5.2	3.4	measured	Moderately toxic	411583-06, Alexander, et. al., 1983.	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)							
Waterflea (<i>Daphnia magna</i>)	98.2	2.6	2.2	Measured	Moderately toxic	439306-01, Drottar, et.al., 1996.	Acceptable
NA = Not available							

Table F-5. Chronic (life cycle) freshwater invertebrate toxicity to 2,4-D						
Species	% ai	Toxicity Value (mg ae/L)		Most sensitive endpoint	MRID No. Author/Year	Study Classification
		NOAEC	LOAEC			
2,4-D acid (PC Code: 030001)*						
Waterflea (<i>Daphnia magna</i>)	91.3	79	151	No of young	418352-11, Ward T.J. et.al., 1991	Acceptable
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)*						
Waterflea (<i>Daphnia magna</i>)	73.8	16.05	25.64	Survival & Reproduction	420183-03, Holmes, et. al., 1991	Acceptable
2,4-D Dimethylamine (DMA)Salt (PC Code: 030019)*						
Waterflea (<i>Daphnia magna</i>)	66.8	LC50=75.7	N/A	Survival (NOAEC not established)	418352-10, Ward, S. C., 1991.	Supplemental
2,4-D Butoxyethyl (BEE) Ester (030053)*						
Waterflea (<i>Daphnia magna</i>)	96	LC50>0.869 NOAEC = 0.20	0.483	Survival and reproduction	413538-02, Gersich, et. al.,, 1989.	Acceptable

* No full life cycle freshwater invertebrate studies using forms of 2,4-D other than acid, DEA, DMA, and BEE were submitted to the Agency.

Table F-6. Acute freshwater amphibian toxicity to 2,4-D							
Species	% ai	96-hour LC50 (95% CI)		Measured/ Nominal Test Concentrations	Toxicity Category (based on ae)	MRID No. Author/Year	Study Classification
		(mg ai/L)	(mg ae/L)				
2,4-D acid (PC Code: 030001)*							
Leopard frog tadpoles (<i>Rana pipiens</i>)	97.5	359	359	Measured	Practically non-toxic	445173-07, Palmer, S.J. et. al., 1997.	Supplemental
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)*							
Leopard frog tadpoles (<i>Rana pipiens</i>)	67.3	337	278	Measured	Practically non-toxic	445173-06, Palmer, S.J. et. al., 1997.	Supplemental

* No acute freshwater amphibian studies using forms of 2,4-D other than acid and DMA were submitted to the Agency.

Table F-7. Non-vascular aquatic plant toxicity to 2,4-D (Tier 1 studies)						
Species	% ai	Tested Concentration		Percent Response (inhibition)	MRID No. Author/Year	Study Classification
		(mg ai/L)	(mg ae/L)			
2,4-D acid (PC Code: 030001)*						
Green algae <i>Selenastrum capricornutum</i>	96.1	26.4	26.4	24	414200-01, Hughes, 1990	Acceptable
Blue-green algae <i>Anabaena flos-aquae</i>	96.9	>2.02	>2.02	0.488	433079-01, Hughes, 1994	Acceptable
Freshwater diatom <i>Navicula pelliculosa</i>	96.9	>2.13	>2.13	24	433079-02, Hughes, 1990	Acceptable
Marine diatom <i>Skeletonema costatum</i>	96.9	2.08	2.08	-10	433079-03, Hughes, 1990	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)*						
Green algae <i>Selenastrum capricornutum</i>	98.2		0.13	-11 %	437680-01, Hughes, et. al., 1995.	Acceptable

* No Tier I non-vascular aquatic plant studies using forms of 2,4-D other than acid and IPA were submitted to the Agency.

Table F-8. Non-vascular aquatic plant toxicity to 2,4-D (Tier II studies)					
Species	% ai	EC50 / NOAEC		MRID No. Author/Year	Study Classification
		(mg ai/L)	(mg ae/L)		
2,4-D acid (PC Code: 030001)					
No data					
2,4-D Sodium Salt (PC Code: 030004)					
No data					
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)					
Green Algae <i>Selnastrum capricornutum</i>	73.8	11/ 0.50	7.48/ 0.34	427122-05, Thompson et. al., 1993.	Acceptable
Marine diatom <i>Skeletonema costatum</i>	73.8	>95/ 95	>64.6/ 64.6	427122-01, Thompson et. al., 1993	Acceptable
Freshwater diatom <i>Navicula pelliculosa</i>	73.8	>97/ 97	>66/ 66	427122-02, Thompson et. al., 1993.	Acceptable
Blue-green algae <i>Anabaena flos-aquae</i>	73.8	>96/ 96	>65.3/ 65.3	427122-03, Thompson et. al., 1993.	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)					
Green Algae <i>Selnastrum capricornutum</i>	66.7	51.2/ 19.2	42.5/ 16	414200-02, Hughes, J.sS, 1990.	Acceptable
Marine diatom <i>Skeletonema costatum</i>	66.7	148.5/ 96.25	123.3/ 79.89	415059-01, Hughes, J.sS, 1990.	Acceptable
Freshwater diatom <i>Navicula pelliculosa</i>	66.7	4.67/ 1.70	3.88/ 1.41	415059-03, Hughes, J.S., 1990.	Acceptable
Blue-green algae <i>Anabaena flos-aquae</i>	66.7	188.5/ 67.86	156.5/ 56.32	415059-02, Hughes, J.S., 1990.	Acceptable

Table F-8. Non-vascular aquatic plant toxicity to 2,4-D (Tier II studies)					
Species	% ai	EC50 / NOAEC		MRID No. Author/Year	Study Classification
		(mg ai/L)	(mg ae/L)		
2,4-D Isoproylamine (IPA) (PC Code: 030025)					
Green Algae <i>Selenastrum capricornutum</i>	51.3	43.4/ 13.9	34.29/ 10.98	417321-02, Hughes, J.S., 1990.	Acceptable
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)					
Green Algae <i>Selenastrum capricornutum</i>	73.8	75.7 55.4	40.88/ 29.92	417321-01, Hughes, J.S., 1990.	Acceptable
Marine diatom <i>Skeletonema costatum</i>	70.9	79.7 50.4	38.29/	434886-03, Hughes, et. al., 1994	Acceptable
Freshwater diatom <i>Navicula pelliculosa</i>	70.9	94.4 5.35	50.98/ 2.89	434886-01, Hughes, et. al., 1994.	Acceptable
Blue-green algae <i>Anabaena flos-aquae</i>	70.9	133 47.9	71.82/ 25.87	434886-04, Hughes, et. al., 1994.	Acceptable
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)					
Green Algae <i>Selenastrum capricornutum</i>	96	24.9/ 12.5	17.14/ 8.6	431882-01, Hughes, J.S., 1990.	Acceptable
Marine diatom <i>Skeletonema costatum</i>	96	1.48/ 0.78	1.02/ 0.538	42-684-04, Hughes,J.S., 1990.	Acceptable
Freshwater diatom <i>Navicula pelliculosa</i>	96	1.86/ 0.86	1.28/ 0.59	420684-03, Hughes, J.S., 1990	Acceptable
Blue-green algae <i>Anabaena flos-aquae</i>	96	6.37/ 3.14	4.4/ 2.2	420684-03, Hughes, J.S., 1990.	Acceptable
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)					

Table F-8. Non-vascular aquatic plant toxicity to 2,4-D (Tier II studies)					
Species	% ai	EC50 / NOAEC		MRID No. Author/Year	Study Classification
		(mg ai/L)	(mg ae/L)		
Green algae <i>Selenastrum capricornutum</i>	94.7 (62.8 a. eq.)	>30 / 3.75	19.8/ 2.48	417352-06, Hughes, J.S., 1990.	Acceptable
Marine diatom <i>Skeletonema costatum</i>	94.7 (62.8 a. eq.)	0.10 / 0.0938	0.066/ 0.062	417352-04, Hughes, J.S., 1990.	Acceptable
Freshwater diatom <i>Navicula pelliculosa</i>	94.7 (62.8 a. eq.)	1.9 / 1.875	1.25/ 1.24	417352-05, Hughes, J.S., 1990.	Acceptable
Blue-green algae <i>Anabaena flos-aquae</i>	94.7 (62.8 a. eq.)	>0.32 / 0.32	>0.21/ 0.21	417352-02, Hughes, J.S., 1990.	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)					
No data					

Table F-9. Vascular aquatic plant toxicity to 2,4-D (Tier II studies)					
Species	% ai	EC50 / NOAEC		MRID No. Author/Year	Study Classification
		(mg ai/L)	(mg ae/L)		
2,4-D acid (PC Code: 030001)					
Duckweed , <i>Lemna gibba</i>	96.2	0.695 / 0.0581	0.695 / 0.0581	442951-01, Hughes et al, 1997	Acceptable
2,4-D Sodium Salt (PC Code: 030004)					
No data					
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)					
Duckweed , <i>Lemna gibba</i>	73.8	0.44/ 0.07	0.2992/ 0.0476	427122-04, Thompson et. al., 1993.	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)					
Duckweed , <i>Lemna gibba</i>	66.7	0.58/ 0.27	0.48/ 0.23	415059-04, Hughes, J.S., 1990.	Acceptable
2,4-D Isoproylamine (IPA) (PC Code: 030025)					
No data					
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)					
Duckweed , <i>Lemna gibba</i>	70.9	2.37/ 2.38	1.28/ 1.28	434886-02, Hughes, et. al., 1994.	Acceptable
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)					
Duckweed, <i>Lemna gibba</i>	96	0.576/ 0.204	0.3974/ 0.141	4206884-02, Hughes, J.S., 1990.	Acceptable
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)					
Duckweed , <i>Lemna gibba</i>	94.7	0.50 / <0.0938	0.33/ 0.062	417352-03, Hughes, J.S., 1990.	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)					
No data					

Table F-10. Acute bird toxicity (gavage) to 2,4-D						
Species	% ai	LD50		Toxicity Category (based on ae)	MRID No. Author/Year	Study Classification
		(mg ai/kg- bwt)	(mg ae/kg- bwt)			
2,4-D acid (PC Code: 030001)						
Mallard duck (<i>Anas platyrhynchos</i>)	96.1	>5620	>5620	Practically non-toxic	415462-02, Culotta et.al., 1990	Acceptable
2,4-D Sodium Salt (PC Code: 030004)						
No data						
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)						
Northern bobwhite quail (<i>Colinus virginianus</i>)	73.1	595	404.6	Moderately toxic	419751-01, Cambell, et. al, 1991.	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)						
Northern bobwhite quail (<i>Colinus virginianus</i>)	66.8	500	415	Moderately toxic	415462-01, Hoxter et. all., 1990.	Acceptable
Mallard duck (<i>Anas platyrhynchos</i>)	100	>4640	>3851.2	Practically non-toxic	233351, Fink, R., 1978	Acceptable
2,4-D Isoproylamine (IPA) (PC Code: 030025)						
Mallard duck (<i>Anas platyrhynchos</i>)	48.7	>398	>314.4	Moderately toxic	00138871, Beavers, et. al., 1983,	Supplemental
Northern bobwhite quail (<i>Colinus virginianus</i>)	49%	377	298	Moderately toxic	442757-01 Beavers 1985	Acceptable
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)						
Northern bobwhite quail (<i>Colinus virginianus</i>)	70.4	>405	>218.7	Moderately toxic	416444-01, Hoxter, K.A., 1990.	Acceptable
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)						

Table F-10. Acute bird toxicity (gavage) to 2,4-D						
Species	% ai	LD50		Toxicity Category (based on ae)	MRID No. Author/Year	Study Classification
		(mg ai/kg- bwt)	(mg ae/kg- bwt)			
Northern bobwhite quail (<i>Colinus virginianus</i>)	96	>2000	>1380	Slightly toxic	414541-01, Lloyd, D., 1989.	Acceptable
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)						
Northern bobwhite quail (<i>Colinus virginianus</i>)	96.2	633	417.78	Moderately toxic	411583-03, Beavers, J.B., 1984.	Acceptable
Mallard duck (<i>Anas platyrhynchos</i>)	92	.>3000	>1980	Slightly toxic	72472, Fink, R., 1976.	Acceptable
Mallard duck (<i>Anas platyrhynchos</i>)	92	.>4640	>3062	Practically non-toxic	226397, Fink, R., 1976.	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)						
Northern bobwhite quail (<i>Colinus virginianus</i>)	98.2	1879	1578	Slightly toxic	439350-01, Palmer, et. al., 1996.	Acceptable

Table F-11. Acute bird toxicity (dietary) to 2,4-D						
Species	% ai	LD50		Toxicity Category (based on ae)	MRID No. Author/Year	Study Classification
		(mg ai/kg- diet)	(mg ae/kg- diet)			
2,4-D acid (PC Code: 030001)						
Northern bobwhite quail (<i>Colinus virginianus</i>)	96.1	>5620	>5620	Practically nontoxic	415861-01, Culotta J., 1989.	Acceptable
Mallard duck (<i>Anas platyrhynchos</i>)	96.1	>5620	>5620	Practically non-toxic	415462-02, Culotta et.al., 1990	Acceptable
2,4-D Sodium Salt (PC Code: 030004)						
No data						
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)						
Northern bobwhite quail (<i>Colinus virginianus</i>)	73.1	>5620	>3821.6	Slightly toxic	419751-02, Hoxter, et. al., 1991.	Acceptable
Mallard duck (<i>Anas platyrhynchos</i>)	73.1	>5620	>3820.6	Slightly toxic	419751-03, Hoxter, et. al., 1991.	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)						
Northern bobwhite quail (<i>Colinus virginianus</i>)	66.8	>5620	>4665	Slightly toxic	417495-01, Long, et. al., 1990.	Acceptable
Northern bobwhite quail (<i>Colinus virginianus</i>)	100	>10,000	>8300	Practically non-toxic	233351, Fink, R., 1978.	Acceptable
Mallard duck (<i>Anas platyrhynchos</i>)	66.8	>5620	>4665	Slightly toxic	417495-02, Long, et. al., 1990.	Acceptable
2,4-D Isoproylamine (IPA) (PC Code: 030025)						
Northern bobwhite quail (<i>Colinus virginianus</i>)	48.7	>5620	>4440	Slightly toxic	00138870, Beavers, J.B., 1983.	Acceptable

Table F-11. Acute bird toxicity (dietary) to 2,4-D						
Species	% ai	LD50		Toxicity Category (based on ae)	MRID No. Author/Year	Study Classification
		(mg ai/kg- diet)	(mg ae/kg- diet)			
Mallard duck (<i>Anas platyrhynchos</i>)	48.7	>5620	>4440	Slightly toxic	00138872, Beavers, J.B., 1983.	Acceptable
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)						
Northern bobwhite quail (<i>Colinus virginianus</i>)	70.4	>5620	>3035	Slightly toxic	416444-02, Driscoll, et., al. 1990.	Acceptable
Mallard duck (<i>Anas platyrhynchos</i>)	70.4	>5620	>3035	Slightly toxic	416444-03, Driscoll, et., al. 1990.	Acceptable
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)						
Northern bobwhite quail (<i>Colinus virginianus</i>)	96	>5620	>3878	Slightly toxic	414484-01, Grimes, J., 1989.	Acceptable
Mallard duck (<i>Anas platyrhynchos</i>)	96	>5620	>3866	Slightly toxic	414290-07, Grimes, J., 1989.	Acceptable
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)						
Northern bobwhite quail (<i>Colinus virginianus</i>)	96	>5620	>3878	Slightly toxic	414484-01, Grimes, J., 1989.	Acceptable
Mallard duck (<i>Anas platyrhynchos</i>)	96	>5620	>3866	Slightly toxic	414290-07, Grimes, J., 1989.	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)						
Northern bobwhite quail (<i>Colinus virginianus</i>)	98.2	>5456	>4583	Slightly toxic	439349-01, Palmer, et. al., 1996.	Acceptable
Mallard duck (<i>Anas platyrhynchos</i>)	98.2	>5218	>4383	Slightly toxic	439352-01, Palmer, et. al., 1996.	Acceptable

Table F-12. Chronic bird toxicity (reproductive) to 2,4-D						
Species	% ai	NOAEC / LOAEC		Most sensitive endpoints	MRID No. Author/Year	Study Classification
		(mg ai/kg-diet)	(mg ae/kg-diet)			
2,4-D acid (PC Code: 030001) *						
Northern bobwhite quail (<i>Colinus virginianus</i>)	96.9	962/>962	962/>962	No effects	415861-01, Culotta J., 1989.	Acceptable

* No avian reproduction studies using forms of 2,4-D other than the acid were submitted to the Agency.

Table F-13. Acute mammal toxicity (gavage) to 2,4-D						
Species	% ai	LD ₅₀ (mg ae/kg-bwt) ¹	Toxicity Category (based on ae)	MRID No.	Study Classification ²	Comments
2,4-D acid (PC Code: 030001)						
Laboratory rat (<i>Rattus norvegicus</i>)	NA	699	Slightly toxic	00101605	Acceptable	
2,4-D Sodium Salt (PC Code: 030004)						
No data						
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)						
Laboratory rat (<i>Rattus norvegicus</i>)	100% (assumed by study author)	619	Slightly toxic	41920901	Acceptable	Study author assumed test material was 100% DEA salt, dose listed in terms of test material administered.
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)						
Laboratory rat (<i>Rattus norvegicus</i>)	57.9% a.e.	>579	Slightly toxic	00157512	Acceptable	Chemical analysis confirmed 57.9% a.e., administered dose listed in terms of test material (an end-use product), total mortality at highest dose was 4/12 (no other mortality)
2,4-D Isopropylamine (IPA) Salt (PC Code: 030025)						
Laboratory rat (<i>Rattus norvegicus</i>)	39.4% a.e.	747	Slightly toxic	00252291	Acceptable	Administered dose listed in terms of test material (technical), technical label indicated 39.4% a.e.
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)						

Table F-13. Acute mammal toxicity (gavage) to 2,4-D						
Species	% ai	LD₅₀ (mg ae/kg- bwt)¹	Toxicity Category (based on ae)	MRID No.	Study Classification²	Comments
Laboratory rat (<i>Rattus norvegicus</i>)	37.7% a.e.	441	Slightly toxic	41413501	Acceptable	Administered dose listed in terms of test material. Study author stated test material was 37.7% ae.
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)						
Laboratory rat (<i>Rattus norvegicus</i>)	NA	573	Slightly toxic	40629801	Acceptable	Administered dose listed in terms of test material, chemical analysis confirmed 66.1% ae
2,4-D 2-Ethylhexyl Ester (EHE) (PC code: 030063)						
Laboratory rat (<i>Rattus norvegicus</i>)	NA	NA	NA	41209001	Acceptable	Study conducted using end-use product EPA Reg. 34704-607. Label stated product contained 33.18% 2,4-D EHE and 32.52% 2,4-DP EHE (PCcode 31464). Because this product is a mixture, it will not be considered.
2,4-D Isopropyl Ester (IPE) (PC code: 030066)						
Laboratory rat (<i>Rattus norvegicus</i>)	37% a.e.	458	Slightly toxic	41709901	Acceptable	Administered dose listed in terms of test material, material was an end-use product, EPA Reg. 400-444. Label stated material 37% ae.

Table F-13. Acute mammal toxicity (gavage) to 2,4-D						
Species	% ai	LD ₅₀ (mg ae/kg- bwt) ¹	Toxicity Category (based on ae)	MRID No.	Study Classification ²	Comments
¹ All LD ₅₀ s were calculated by EFED using data provided in the original study reports. Clarifications regarding the calculations are made in the “Comments” column. ² Classifications determined by HED.						

Table F-14. Chronic mammalian toxicity (reproductive) to 2,4-D							
Species	% ai	Endpoint (mg ae/kg-bwt/day)			Affected parameters (most sensitive)	MRID No.	Study Classification
			NOAEL	LOAEL			
2,4-D acid (PC Code: 030001) *							
Laboratory rat (<i>Rattus norvegicus</i>)		Parental	Target =5 Actual (3.5-13.5)	Target =20 Actual (14-48)	Decreased female body wt gain(F1) and male renal tubule alteration (F0 and F1)	00150557; 00163996	Acceptable, determined by HED
		Reproductive	Target =20 Actual (18-35)	Target =80 Actual (69-114)	increase in gestation length		
		Offspring	Target =5 Actual (7.2-13.5)	Target =20 Actual (26-48)	decreased pup body weight; increase in pup deaths at 80 mg ae/kg-bwt/day		

* No mammalian reproduction studies using forms of 2,4-D other than the acid were submitted to the Agency.

Table F-15. Acute honey bee toxicity (contact) to 2,4-D						
Species	% ai	96-hour LD50 (95% CI)		Toxicity Category (based on ae)	MRID No. Author/Year	Study Classification
		(µg ai/bee)	(µg ae/bee)			
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019) *						
Honey bee (<i>Apis mellifera</i>)	67.3	>100	>83	Practically non-toxic	445173-04, Palmer S. et al., 1997	Acceptable
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063) *						
Honey bee (<i>Apis mellifera</i>)	96.96	>100	>66	Practically non-toxic	445173-01, Palmer S. et al., 1997	Acceptable

* No contact honey bee studies using forms of 2,4-D other than DMAS and EHE were submitted to the Agency.

Table F-16. Terrestrial plant toxicity (seedling emergence, most sensitive monocot) to 2,4-D - all available forms, technical only					
Species	% ai	EC25 (lbs ae/acre)	Most sensitive endpoint	MRID No. Author/Year	Study Classification
2,4-D acid (PC Code: 030001)					
Onion and sorghum	96.7	2.1	Fresh weight	424168-02, Backus, 1992	Acceptable
2,4-D Sodium Salt (PC Code: 030004)					
No data					
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)					
Onion	50.2	0.38	Fresh weight	426091-01, Backus, 1992	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)					
Sorghum	55.5	0.026	Fresh weight	423895-01, Backus, 1992	Acceptable
2,4-D Isopropylamine (IPA) Salt (PC Code: 030025)					
No data					
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)					
No data					
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)					
Onion	65.6	0.36	Survival	431970-01, Narnish, 1994	Supplemental
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)					
Onion	63.5	0.218	Fresh shoot weight	435269-01, Backus, 1995	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)					
Onion	98.2	0.010	Shoot length	439821-01, Hoberg, 1996	Acceptable

Table F-17. Terrestrial plant toxicity (seedling emergence, most sensitive dicots) to 2,4-D - all available forms, technical only					
Species	% ai	EC25 (lbs ae/acre)	Most sensitive endpoint	MRID No. Author/Year	Study Classification
2,4-D acid (PC Code: 030001)					
Mustard	96.7	0.033	Fresh weight	424168-02, Backus, 1992	Acceptable
2,4-D Sodium Salt (PC Code: 030004)					
No data					
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)					
Mustard	50.2	0.045	Fresh weight	426091-01, Backus, 1992	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)					
Mustard	55.5	0.00953	Fresh weight	423895-01, Backus, 1992	Acceptable
2,4-D Isoproylamine (IPA) Salt (PC Code: 030025)					
No data					
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)					
No data					
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)					
Tomato	65.6	0.05	Dry weight	431970-01, Narnish, 1994	Supplemental
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)					
Radish	63.5	0.037	Fresh shoot weight	424492-01, Backus, 1992	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)					
Lettuce	98.2	0.00081	Shoot length	439821-01, Hoberg, 1996	Supplemental

Table F-18. Terrestrial plant toxicity (vegetative vigor, most sensitive monocot) to 2,4-D - all available forms, technical only					
Species	% ai	EC25 (lbs ae/acre)	Most sensitive endpoint	MRID No. Author/Year	Study Classification
2,4-D acid (PC Code: 030001)					
Onion	96.7	<0.0075	Fresh weight	424168-01, Backus, 1991	Acceptable
2,4-D Sodium Salt (PC Code: 030004)					
No data					
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)					
Onion	50.2	0.04	Fresh weight	426091-02 Backus 1992	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)					
No data					
2,4-D Isoproylamine (IPA) Salt (PC Code: 030025)					
No data					
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)					
No data					
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)					
Onion	65.6	0.19	Dry weight	430671-03, Narnish, 1993	Supplemental
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)					
Sorghum	63.5	0.218	Fresh weight	423439-02, Backus, 1992	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)					
Corn	82.7	0.2016	Shoot weight	437882-01, Hoberg, 1995	Acceptable

Table F-19. Terrestrial plant toxicity (vegetative vigor, most sensitive dicots) to 2,4-D - all available forms, technical only					
Species	% ai	EC25 (lbs ae/acre)	Most sensitive endpoint	MRID No. Author/Year	Study Classification
2,4-D acid (PC Code: 030001)					
Tomato	96.7	0.0075	Fresh weight	424168-01, Backus, 1991	Acceptable
2,4-D Sodium Salt (PC Code: 030004)					
No data					
2,4-D Diethanolamine (DEA) Salt (PC Code: 030016)					
Tomato	50.2	0.003	Fresh weight	426091-02 Backus, 1992	Acceptable
2,4-D Dimethylamine (DMA) Salt (PC Code: 030019)					
No data					
2,4-D Isopropylamine (IPA) Salt (PC Code: 030025)					
No data					
2,4-D Triisopropanolamine (TIPA) Salt (PC Code: 030035)					
No data					
2,4-D Butoxyethyl (BEE) Ester (PC code: 030053)					
Radish	65.6	0.02	survival	430671-03, Narnish, 1993	Supplemental
2,4-D 2-Ethylhexyl Ester (2-EHE) (PC code: 030063)					
Soybean	63.5	0.02	Fresh weight	423439-02, Backus, 1992	Acceptable
2,4-D Isopropyl Ester (IPE) (PC code: 030066)					
Radish	82.7	0.0042	Root weight	437882-01, Hoberg, 1995	Acceptable

Table F-20. Summary of seedling emergence study conducted with Gordon's Amine 4000 2,4-D Weed Killer, a TEP of 2,4-D DMAS¹

Species	Most sensitive Endpoint	Value (lbs ae/acre)			
		NOAEC	EC ₀₅	EC ₂₅	EC ₅₀
Corn	Plant Height	4.0	2.4	>4.0	>4.0
Onion	Dry Weight	0.091	<0.0014	0.097	>0.35
Ryegrass	Plant Height	4.0	2.2	>4.0	>4.0
Wheat	Dry Weight	0.35	0.0054	0.20	>0.35
Cabbage	Dry Weight	0.020	0.010	0.043	0.12
Lettuce	Dry Weight	0.020	0.0054	0.026	0.078
Radish	Dry Weight	0.020	0.0087	0.033	0.082
Soybean	Dry Weight	0.26	<0.26	0.37	1.2
Tomato	None	0.34	N.D.	>0.34	>0.34
Turnip	Dry Weight	0.020	0.035	0.13	0.34

¹ MRID 471060-01, Porch et al. 2006, 39.21%ai, Acceptable

Table F-21. Summary of vegetative vigor study conducted with Gordon's Amine 4000 2,4-D Weed Killer, a TEP of 2,4-D DMAS¹

Species	Most sensitive Endpoint	Value (lbs ae/acre)			
		NOAEC	EC ₀₅	EC ₂₅	EC ₅₀
Onion	Dry Weight	0.0335	0.034	0.14	0.37
Ryegrass	Dry Weight	2.07	1.0	>2.07	>2.07
Wheat	Dry Weight	0.133	0.32	1.1	>2.03
Cabbage	Dry Weight	0.0981	0.079	0.18	0.31
Lettuce	Dry Weight	0.0017	<0.0017	0.0038	0.015
Radish	Dry Weight	0.0016	<0.0016	0.0012	>0.0266
Soybean	Dry Weight	0.0072	0.0093	0.039	>0.0998
Tomato	Dry Weight	0.0016	0.0018	0.0074	0.020
Turnip	Dry Weight	0.0015	<0.0015	0.011	0.065

¹ MRID 471060-02, Porch et al. 2006, 39.21%ai, Acceptable

Table F-22. Summary of seedling emergence study conducted with Gordon's LV400 2,4-D Weed Killer, a TEP of 2,4-D EHE¹					
Species	Most sensitive Endpoint	Value (lbs ae/acre)			
		NOAEC	EC₀₅	EC₂₅	EC₅₀
Corn	Dry Weight	1.9	1.7	3.4	>3.8
Onion	Plant height	0.019	0.047	0.17	>0.34
Ryegrass	Dry Weight	0.26	<0.26	0.27	2.8
Wheat	None	4.0	<0.26	>4.0	>4.0
Cabbage	Dry Weight	0.0015	0.0056	0.021	0.053
Lettuce	Dry Weight	0.0058	0.0061	0.018	0.039
Radish	Dry Weight	0.0058	0.0061	0.036	0.12
Soybean	Dry Weight	0.47	<0.25	0.85	2.3
Tomato	Dry Weight	0.0058	<0.0015	0.012	0.29
Turnip	Dry Weight	0.0058	0.021	0.062	0.13
¹ MRID 471060-03, Porch et al. 2006, 44.9%ai, Acceptable					

Table F-23. Summary of vegetative vigor study conducted with Gordon's LV400 2,4-D Weed Killer, a TEP of 2,4-D EHE¹					
Species	Most sensitive Endpoint	Value (lbs ae/acre)			
		NOAEC	EC₀₅	EC₂₅	EC₅₀
Corn	Dry Weight	0.0289	<0.00803	0.17	1.6
Onion	Dry Weight	0.0254	0.025	0.088	0.21
Ryegrass	None	2.03	N.D.	>2.03	>2.03
Wheat	Dry Weight	0.0356	0.043	0.34	1.4
Cabbage	Dry Weight	0.00167	0.012	0.027	0.047
Lettuce	Dry Weight	0.00167	<0.00167	0.0021	0.0076
Radish	Survival	0.00527	0.0026	0.0068	0.013
Soybean	Dry Weight	0.0259	0.014	0.058	>0.0992
Tomato	Dry Weight	<0.00134	<0.00134	0.0044	0.016
Turnip	Dry Weight	<0.00134	<0.00134	0.0021	0.042
¹ MRID 471060-04, Porch et al. 2006, 44.9%ai, Acceptable					

Appendix A. Ecological Effects Data

Comparison of Toxicity of Organisms to different forms of Triclopyr in terms of the acid equivalent (TEA, BEE, & degradate TCP)

Taxa	Effect Type	Endpoint	Triclopyr Acid - TGAi	TEA (ae)	BEE (ae)	TCP (ae)
Freshwater fish Rainbow Trout (<i>Oncorhynchus mykiss</i>)	Acute	96h LC50	117 mg/L	79.2 mg/L	0.47 mg/L	1.9 mg/L
Freshwater fish Bluegill sunfish (<i>Lepomis macrochirus</i>)	Acute	96h LC50	148 mg/L	155.4 mg/L	0.26 mg/L	16.1 mg/L
Freshwater fish Fathead Minnow (<i>Pimephales promelas</i>)	Chronic	NOEC LOEC	No Data	> 32.2 mg/L < 50.2 mg/L	No Data	No Data
Freshwater fish Rainbow Trout (<i>Oncorhynchus mykiss</i>)	Chronic	NOEC LOEC	No Data	No Data	0.019 mg/L 0.034 mg/L	No Data
Freshwater invertebrate Water flea (<i>Daphnia magna</i>)	Acute	48h EC50	132.9 mg/L	346 mg/L	0.25 mg/L	13.4 mg/L
Freshwater invertebrate Water flea (<i>Daphnia magna</i>)	Chronic	NOEC LOEC	No Data	25 mg/L 46.2mg/L	No Data	No Data
Vascular aquatic plant Duckweed (<i>Lemna gibba</i>)	Acute	14d EC50	No Data	6.1 mg/L	0.86 mg/L	Invalid study
Non-vascular aquatic plant Green algae (<i>Kirchneria subcapitata</i>) (Formerly <i>Selenastrum capricortum</i>)	Acute	5d EC50	29.8 mg/L	12.1 mg/L	2.5 mg/L	2.3 mg/L
Non-vascular aquatic plant Blue-green algae (<i>Anabeana flos-aquae</i>)	Acute	5d EC50	No Data	(7d EC50) 4.1 mg/L	1.42 mg/L	2.3 mg/L
Non-vascular aquatic plant Freshwater diatom (<i>Navicula pelliculosa</i>)	Acute	5d EC50	No Data	(4d EC50) 10.6 mg/L	0.07 mg/L	No Data
Avian Mallard duck (<i>Anas platyrhynchos</i>)	Acute oral	LD50	1698 mg/kg bw	(14d LC50) 1418 mg/kg bw	No Data	No Data

Taxa	Effect Type	Endpoint	Triclopyr Acid	TEA (ae)	BEE (ae)	TCP (ae)
Avian Northern bobwhite Quail (<i>Colinus virginianus</i>)	Acute oral	21d LD50	No Data	No Data	529 mg/kg bw	(8d LD50) > 2585 mg/kg bw
Avian Northern bobwhite Quail (<i>Colinus virginianus</i>)	Subacute dietary	8d LC50	2934 ppm	5,189 ppm	3385 ppm	No Data
Avian Mallard duck (<i>Anas platyrhynchos</i>)	Subacute dietary	8d LC50	No Data	> 4,464.8 ppm	> 3885	> 7265 ppm
Avian Mallard duck (<i>Anas platyrhynchos</i>)	Chronic	NOAEC LOAEC	100 ppm 200 ppm # of 14 d old survivors	No Data	No Data	No Data
Rodent Rat	Acute oral	LD50	630 (F) mg/kg	572 (M & F) mg/kg	578 (M & F) mg/kg	1026 (M) mg/kg
Rodent Rat	Chronic	NOAEL LOAEL	5 mg/kg bw 25 mg/kg bw	No Data	No Data	No Data
Honeybee (<i>Apis mellifera</i>)	Acute	Contact 48h LD50	>100 µg/bee	No Data	> 72 µg/bee	No Data
Terrestrial dicot Sunflower (<i>Helianthus annuus</i>) (Vegetative Vigor)	Acute	14d EC25	No Data	0.005 lbs ae/A Parameter: shoot length	0.006 lbs ae/ A Parameter: shoot weight	No Data
Terrestrial monocot Onion (<i>Allium cepa</i>) (Vegetative Vigor)	Acute	14d EC25	No Data	0.114 lbs ae/A Parameter: shoot weight	0.063 lbs ae/A Parameter: shoot weight	No Data
Terrestrial dicot Alfalfa (<i>Medicago sativa</i>) (Seedling Emergence)	Acute	51d EC25	No Data	No Data	0.045 lbs ae/A parameter: emergence	No Data
Terrestrial dicot Soybean (<i>Glycine max</i>) (Seedling Emergence)	Acute	14d EC25	No Data	> 0.23 lbs ae/A Parameter: shoot length	< 8.0 lbs ae/A Parameter: emergence	No Data
Terrestrial monocot Onion (<i>Allium cepa</i>) (Seedling Emergence)	Acute	51d EC25	No Data	> 0.69 lbs as/A Parameter: shoot length	0.053 lbs ae/A Parameter: shoot weight	No Data

Terrestrial monocot Barley (<i>Hordeum vulgare</i>) (Seedling Emergence)	Acute	14d EC25	No Data	> 0.23 lbs ae/A Parameter: shoot length	< 8.0 lbs ae/A Parameter: emergence	No Data
--	-------	----------	---------	---	--	---------

Triclopyr Acid (colored cells are most sensitive for triclopyr acid)

Habitat	Taxa	Effect Type	Endpoint	Value	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author))
Aquatic	Freshwater fish Rainbow Trout (<i>Oncorhynchus mykiss</i>)	Acute	96h LC50	117	mg/L	Practically Non-Toxic	Acceptable	00049637 Dow Chemical 1973
Aquatic	Freshwater fish Bluegill sunfish (<i>Lepomis macrochirus</i>)	Acute	96h LC50	148	mg/L	Practically Non-Toxic	Acceptable	00049637 Dow Chemical 1973
Aquatic	Freshwater invertebrate Water flea (<i>Daphnia magna</i>)	Acute	48h EC50	132.9	mg/L	Practically Non-Toxic	Acceptable	40346504 McCarty 1977
Aquatic plants	Non-vascular aquatic plant Green algae (<i>Kirchneria subcapitata</i>) (Formerly <i>Selenastrum capricornutum</i>)	Acute	5d EC50	29.8	mg/L	Parameter?	Supplemental	41736303 Cowgill 1989
Terrestrial	Avian Mallard duck (<i>Anas platyrhynchos</i>)	Acute oral	LD50	1698	mg/kg bw	Slightly Toxic	Acceptable	40346401 Dow Chemical 1976
Terrestrial	Avian Cortunix Quail	Subacute dietary	LC50	3272	ppm	Slightly Toxic	Supplemental	00049638 Dow Chemical 1973
Terrestrial	Avian Northern bobwhite Quail (<i>Colinus virginianus</i>)	Subacute dietary	LC50	2934	ppm	Slightly Toxic	Acceptable	40346403 Dow Chemical 1976
Terrestrial	Avian Mallard duck (<i>Anas platyrhynchos</i>)	Subacute dietary	LC50	5620	ppm	Practically Non-Toxic	Acceptable	0031249 Wildlife Int'l 1979
Terrestrial	Avian Northern bobwhite Quail (<i>Colinus virginianus</i>)	Chronic	NOAEC LOAEC LOAEC	500 >500 200	ppm ppm ppm	N/A	Acceptable	00031251 Beavers 1979
Terrestrial	Avian Mallard duck (<i>Anas platyrhynchos</i>)	Chronic	NOAEC LOAEC	100 200	ppm	# of 14 d old survivors	Acceptable	00031250 Beavers 1979

Habitat	Taxa	Effect Type	Endpoint	Value	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial	Rodent Rat	Acute oral	LD50	729 (M) 630 (F) (F 95% CI 450-829)	mg/kg mg/kg mg/kg	Slightly Toxic	Acceptable (HED 2002)	00031940 Henck et al. 1979
Terrestrial	Rodent	Chronic	NOAEL	25	mg/kg bw	Reproductive/ Systemic	Acceptable (RED 1998)	43545701 Vedula et al. 1995
Terrestrial	Rodent	Chronic	LOAEL	250	mg/kg bw			
Terrestrial	Rodent	Chronic	NOAEL	5	mg/kg bw	Reproductive/ Offspring	Acceptable (HED 2002)	43545701 Vedula et al. 1995
Terrestrial	Honeybee (<i>Apis mellifera</i>)	Acute	LOAEL	25	mg/kg bw			
Terrestrial	Honeybee (<i>Apis mellifera</i>)	Acute	Contact 48h LD50	>100	µg/bee	Practically Non-Toxic	Acceptable	40356602 Dingledine 1985

TCP (Degradate Toxicity Information)

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic	Freshwater fish Bluegill sunfish (<i>Lepomis macrochirus</i>)	Acute	96h LC50	16.1	mg/L	Slightly Toxic	Acceptable	41829003 Gorzinski et al. 1991
Aquatic	Freshwater fish Rainbow Trout (<i>Oncorhynchus mykiss</i>)	Acute	96h LC50	16.3	mg/L	Slightly Toxic	Acceptable	41829004 Gorzinski et al. 1991
Aquatic	Freshwater fish Rainbow Trout (<i>Oncorhynchus mykiss</i>)	Acute	96h LC50	1.9	mg/L	Moderately Toxic	Acceptable	44585404 Wan et al. 1987
Aquatic	Freshwater fish Coho salmon	Acute	96h LC50	2.3	mg/L	Moderately Toxic	Supplemental	44585404 Wan et al. 1987
Aquatic	Freshwater fish Chum salmon	Acute	96h LC50	2.3	mg/L	Moderately Toxic	Supplemental	44585404 Wan et al. 1987
Aquatic	Freshwater fish Sockeye salmon	Acute	96h LC50	3.2	mg/L	Moderately Toxic	Supplemental	44585404 Wan et al. 1987
Aquatic	Freshwater fish Chinook salmon	Acute	96h LC50	2.7	mg/L	Moderately Toxic	Supplemental	44585404 Wan et al. 1987
Aquatic	Freshwater fish Pink salmon	Acute	96h LC50	3.5	mg/L	Moderately Toxic	Supplemental	44585404 Wan et al. 1987
Aquatic	Freshwater invertebrate Water flea (<i>Daphnia magna</i>)	Acute	48h EC50	13.4	mg/L	Slightly Toxic	Acceptable	41829003 Gorzinski et al. 1991
Aquatic plants	Vascular aquatic plant Duckweed (<i>Lemna gibba</i>)	Acute	----	----	----	Limited critical endpoints measured	Invalid	45312002 Kirk et al. 2000
Aquatic plants	Non-vascular aquatic plant Green algae (<i>Kirchneria subcapitata</i>) (Formerly <i>Selenastrum capricornutum</i>)	Acute	4d EC50	2.3	mg/L	Parameter: Growth (Yield)	Supplemental	45312001 Kirk et al. 1999

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic plants	Non-vascular aquatic plant Green algae (<i>Kirchneria subcapitata</i>) (Formerly <i>Selenastrum capricortum</i>)	Acute	4d NOAEC (as EC05)	0.84	mg/L	Parameter: Growth (Yield)	Supplemental	45312001 Kirk et al. 1999
Aquatic plants	Non-vascular aquatic plant Blue-green algae (<i>Anabeana flos-aquae</i>)	Acute	5d EC50	2.3	mg/L	Parameter: Growth and Reproduction (Yield)	Supplemental	45312003 Kirk et al. 2000
Aquatic plants	Non-vascular aquatic plant Blue-green algae (<i>Anabeana flos-aquae</i>)	Acute	5d NOAEC	0.46	mg/L	Parameter: Growth and Reproduction (Yield)	Supplemental	45312003 Kirk et al. 2000
Terrestrial	Avian Northern bobwhite Quail (<i>Colinus virginianus</i>)	Acute oral	8d LD50	> 2585	mg/kg bw	Practically Non-Toxic	Acceptable	41829001 Campbell et al. 1990
Terrestrial	Avian Mallard duck (<i>Anas platyrhynchos</i>)	Subacute dietary	8d LC50	> 7265	ppm	Practically Non-Toxic	Supplemental	41829002 Long et al. 1990
Terrestrial	Rodent Rat	Acute oral	LD50	1026 (M) 1124 (F)	mg/kg	Slightly Toxic	Acceptable	00064938 Gerbig & Emerson 1970

Triclopyr Triethylamine (TEA) (colored cells are most sensitive for TEA)

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic	Freshwater fish Rainbow Trout (<i>Oncorhynchus mykiss</i>)	Acute	96h LC50	273.7 (flow-through)	mg/L	Practically Non-Toxic	Acceptable	00151956 McCarty 1978
Aquatic	Freshwater fish Rainbow Trout (<i>Oncorhynchus mykiss</i>)	Acute	96h LC50	233.1	mg/L	Practically Non-Toxic	Acceptable	00151956 McCarty 1978
Aquatic	Freshwater fish Rainbow Trout (<i>Oncorhynchus mykiss</i>)	Acute	96h LC50	79.2 (flow-through)	mg/L	Practically Non-Toxic	Acceptable (for formulated product)	00049637 Dow Chemical 1973
Aquatic	Freshwater fish Bluegill sunfish (<i>Lepomis macrochirus</i>)	Acute	96h LC50	398.7 (flow-through)	mg/L	Practically Non-Toxic	Acceptable	00151956 McCarty 1978
Aquatic	Freshwater fish Bluegill sunfish (<i>Lepomis macrochirus</i>)	Acute	96h LC50	155.4 (flow-through)	mg/L	Practically Non-Toxic	Acceptable (for formulated product)	00049637 Dow Chemical 1973
Aquatic	Freshwater fish Fathead Minnow (<i>Pimephales promelas</i>)	Acute	96h LC50	422.8 (flow-through)	mg/L	Practically Non-Toxic	Acceptable	00151956 McCarty 1978
Aquatic	Freshwater fish Fathead Minnow (<i>Pimephales promelas</i>)	Acute	96h LC50	168.5 (static)	mg/L	Practically Non-Toxic	Acceptable	00151958 Mayes 1983
Aquatic	Freshwater fish Fathead Minnow (<i>Pimephales promelas</i>)	Acute	96h LC50	86.4 (flow-through)	mg/L	Practically Non-Toxic	Acceptable	00151958 Mayes 1983
Aquatic	Marine/Estuarine fish Inland silverside (<i>Menidia beryllina</i>)	Acute	LC50	40.1	mg/L	Practically Non-Toxic	Acceptable (for formulated product)	41633703 Ward 1989
Aquatic	Freshwater fish Fathead Minnow (<i>Pimephales promelas</i>)	Chronic	NOEC LOEC	> 32.2 < 50.2	mg/L mg/L	Parameter: Length	Acceptable	00151958 Mayes 1983
Aquatic	Freshwater invertebrate Water flea (<i>Daphnia magna</i>)	Acute	48h EC50	463.5	mg/L	Practically Non-Toxic	Acceptable	00151959 Gerisch 1982

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic	Freshwater invertebrate Water flea (<i>Daphnia magna</i>)	Acute	48h EC50	346	mg/L	Practically Non-Toxic	Acceptable	00151956 McCarty 1978
Aquatic	Freshwater invertebrate Water flea (<i>Daphnia magna</i>)	Chronic	NOEC LOEC	25 46.2	mg/L	total young and mean brood size	Acceptable	00151959 Gerisch 1982
Aquatic	Mollusca Eastern oyster (<i>Crassostrea virginica</i>) (shell deposition)	Acute	LC50/EC50	18.4	mg/L	Slightly Toxic	Acceptable (for formulated product)	42646101 Kowalski 1992
Aquatic	Mollusca Eastern oyster (<i>Crassostrea virginica</i>) (embryo larvae)	Acute	48h EC50	> 16.9 < 26.3	mg/L mg/L	Parameter: 100% abnormal development at 87 ppm	Acceptable (for formulated product)	00062623 EG & G 1975 40346606?
Aquatic	Crustacea Pink shrimp (<i>Penaeus duorarum</i>)	Acute	LC50/EC50	270.5	mg/L	Practically Non-Toxic	Supplemental	00062623 EG & G 1975 40346606?
Aquatic	Crustacea Grass Shrimp (<i>Palaemonetes pugio</i>)	Acute	LC50/EC50	103.7	mg/L	Practically Non-Toxic	Acceptable (for formulated product)	42646102 Kowalski 1992
Aquatic	Crustacea Fiddler crab (<i>Uca pugnator</i>)	Acute	LC50/EC50	> 302.2	mg/L	Practically Non-Toxic	Supplemental	00062623 EG & G 1975 40346606?
Aquatic plants	Vascular aquatic plant Duckweed (<i>Lemna gibba</i>)	Acute	14d EC50	6.06	mg/L	Parameter: Growth and Reproduction	Acceptable	41633709 Cowgill 1987
Aquatic plants	Vascular aquatic plant Duckweed (<i>Lemna gibba</i>)	Acute	14d EC50	7.6	mg/L	Parameter: Growth and Reproduction	Acceptable	41736302 Cowgill 1988
Aquatic plants	Non-vascular aquatic plant Green algae (<i>Kirchneria subcapitata</i>) (Formerly <i>Selenastrum capricornum</i>)	Acute	5d EC50	12.1	mg/L	Parameter: Growth	Acceptable	41736305 Cowgill 1987

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic plants	Non-vascular aquatic plant Blue-green algae (<i>Anabeana flos-aquae</i>)	Acute	7d EC50	4.1	mg/L	Parameter: Growth	Acceptable	41633706 Cowgill 1987
Aquatic plants	Non-vascular aquatic plant Freshwater diatom (<i>Navicula pelliculosa</i>)	Acute	4d EC50	10.6	mg/L	Parameter: Growth	Acceptable	41633708 Cowgill 1987
Aquatic plants	Non-vascular aquatic plant Marine diatom (<i>Skeletonema costatum</i>)	Acute	5d EC50	4.6	mg/L	Parameter: Growth	Acceptable	41633707 Cowgill 1987
Terrestrial	Avian Mallard duck (<i>Anas platyrhynchos</i>)	Acute oral	14d LD50	1418	mg/kg bw	Practically Non-Toxic	Acceptable	40346501 Fink 1978
Terrestrial	Avian Mallard duck (<i>Anas platyrhynchos</i>)	Acute oral	8d LD50	1417.6	mg/kg bw	Practically Non-Toxic	Acceptable	00134178
Terrestrial	Avian Northern bobwhite Quail (<i>Colinus virginianus</i>)	Subacute dietary	8d LC50	5,189	ppm	Practically Non-Toxic	Acceptable	40346503 Fink 1978
Terrestrial	Avian Mallard duck (<i>Anas platyrhynchos</i>)	Subacute dietary	8d LC50	> 4,464.8	ppm	Practically Non-Toxic	Acceptable	40346502 Fink 1977
Terrestrial	Rodent Rat	Acute oral	LD50	572 (M & F)	mg/kg	Slightly Toxic	Acceptable	41443301 Mizell & Lomax 1988
Terrestrial Plant	Terrestrial dicot Sunflower (<i>Helianthus annus</i>) (Vegetative Vigor)	Acute	14d EC25	0.0063	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Sunflower (<i>Helianthus annus</i>) (Vegetative Vigor)	Acute	NOAEC	0.0028	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial Plant	Terrestrial dicot Sunflower (<i>Helianthus annuus</i>) (Vegetative Vigor)	Acute	14d EC25	0.005	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Sunflower (<i>Helianthus annuus</i>) (Vegetative Vigor)	Acute	NOAEC	0.0028	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Sunflower (<i>Helianthus annuus</i>) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801
Terrestrial Plant	Terrestrial dicot Sunflower (<i>Helianthus annuus</i>) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Sugarbeet (<i>Beta vulgaris</i>) (Vegetative Vigor)	Acute	14d EC25	0.030	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Sugarbeet (<i>Beta vulgaris</i>) (Vegetative Vigor)	Acute	14d EC25	0.11	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Sugarbeet (<i>Beta vulgaris</i>) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801
Terrestrial Plant	Terrestrial dicot Sugarbeet (<i>Beta vulgaris</i>) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Tomato (<i>Lycopersicon esculentum</i>) (Vegetative Vigor)	Acute	14d EC25	0.0135	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Tomato (<i>Lycopersicon esculentum</i>) (Vegetative Vigor)	Acute	14d EC25	0.018	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial Plant	Terrestrial dicot Tomato (<i>Lycopersicon esculentum</i>) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Tomato (<i>Lycopersicon esculentum</i>) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial monocot Wheat (<i>Triticum aestivum</i>) (Vegetative Vigor)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Wheat (<i>Triticum aestivum</i>) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Wheat (<i>Triticum aestivum</i>) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Oilseed rape (<i>Brassica napus</i>) (Vegetative Vigor)	Acute	14d EC25	0.051	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Oilseed rape (<i>Brassica napus</i>) (Vegetative Vigor)	Acute	14d EC25	0.064	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Oilseed rape (<i>Brassica napus</i>) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Oilseed rape (<i>Brassica napus</i>) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Radish (<i>Raphanus sativus</i>) (Vegetative Vigor)	Acute	14d EC25	0.125	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial Plant	Terrestrial dicot Radish (<i>Raphanus sativus</i>) (Vegetative Vigor)	Acute	14d EC25	0.132	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Radish (<i>Raphanus sativus</i>) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Radish (<i>Raphanus sativus</i>) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial monocot Corn (<i>Zea mays</i>) (Vegetative Vigor)	Acute	14d EC25	0.121	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Corn (<i>Zea mays</i>) (Vegetative Vigor)	Acute	14d EC25	0.32	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Corn (<i>Zea mays</i>) (Seedling Emergence)	Acute	14d EC25	> 0.23	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Corn (<i>Zea mays</i>) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial monocot Barley (<i>Hordeum vulgare</i>) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Barley (<i>Hordeum vulgare</i>) (Seedling Emergence)	Acute	14d EC25	> 0.23	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Barley (<i>Hordeum vulgare</i>) (Seedling Emergence)	Acute	NOAEC	0.230	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Barley (<i>Hordeum vulgare</i>) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Barley (<i>Hordeum vulgare</i>) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial Plant	Terrestrial monocot Onion (<i>Allium cepa</i>) (Vegetative Vigor)	Acute	14d EC25	0.114	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Onion (<i>Allium cepa</i>) (Vegetative Vigor)	Acute	14d EC25	0.24	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Onion (<i>Allium cepa</i>) (Seedling Emergence)	Acute	14d EC25	> 0.69	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial monocot Onion (<i>Allium cepa</i>) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Soybean (<i>Glycine max</i>) (Vegetative Vigor)	Acute	14d EC25	0.0057	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Soybean (<i>Glycine max</i>) (Vegetative Vigor)	Acute	NOAEC	0.0003	lbs ae/A	Parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Soybean (<i>Glycine max</i>) (Vegetative Vigor)	Acute	14d EC25	0.028	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Soybean (<i>Glycine max</i>) (Seedling Emergence)	Acute	14d EC25	> 0.23	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Soybean (<i>Glycine max</i>) (Seedling Emergence)	Acute	NOAEC	0.0028	lbs ae/A	Parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant	Terrestrial dicot Soybean (<i>Glycine max</i>) (Seedling Emergence)	Acute	21d EC25	< 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Monocots & Dicots Veg.Crops (10 Sp.) (Vegetative Vigor)	Acute	6 wks EC25	> 9.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990

Triclopyr Butoxyethyl Ester (BEE) (colored cells are most sensitive for BEE)

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic	Freshwater fish Rainbow Trout (<i>Oncorhynchus mykiss</i>)	Acute	96h LC50	0.47	mg/L	Highly Toxic	Acceptable	42884501 Woodburn et al. 1993
Aquatic	Freshwater fish Rainbow Trout (<i>Oncorhynchus mykiss</i>)	Acute	96h LC50	1.29 (Propylene glycol butyl ether)	mg/L	Moderately Toxic	Acceptable	00134181 Acc# 229783?
Aquatic	Freshwater fish Rainbow Trout (<i>Oncorhynchus mykiss</i>)	Acute	96h LC50	0.70	mg/L	Highly Toxic	Acceptable	43442602 Weinberg et al. 1994
Aquatic	Freshwater fish Rainbow Trout (<i>Oncorhynchus mykiss</i>)	Acute	24h LC50	0.35-1.2	mg/L	Highly to Moderately Toxic	Supplemental	41971603 Gorzinski 1991
Aquatic	Freshwater fish Bluegill sunfish (<i>Lepomis macrochirus</i>)	Acute	96h LC50	1.46 (Propylene glycol butyl ether)	mg/L	Moderately Toxic	Acceptable	00134181 Acc# 229783?
Aquatic	Freshwater fish Bluegill sunfish (<i>Lepomis macrochirus</i>)	Acute	96h LC50	0.25	mg/L	Highly Toxic	Acceptable	42917901 Woodburn et al. 1993
Aquatic	Freshwater fish Bluegill sunfish (<i>Lepomis macrochirus</i>)	Acute	24h LC50	0.59	mg/L	Moderately Toxic	Supplemental	41971604 Gorzinski 1991
Aquatic	Freshwater fish Bluegill sunfish (<i>Lepomis macrochirus</i>)	Acute	96h LC50	0.31	mg/L	Highly Toxic	Acceptable	43442601 Weinberg et al. 1994
Aquatic	Freshwater fish Coho Salmon (<i>Oncorhynchus kisutch</i>)	Acute	96h LC50	0.32-0.33 (yolk-sac fry) 1.0 (juvenile fry)	mg/L mg/L	Highly Toxic Moderately Toxic	Supplemental	41736304 Barron 1987
Aquatic	Freshwater fish Fathead Minnow (<i>Pimephales promelas</i>)	Acute	24h LC50	1.7	mg/L	Moderately Toxic	Supplemental	00151965 Batchelder 1981
Aquatic	Marine/Estuarine fish Inland silverside (<i>Menidia beryllina</i>)	Acute	96h LC50	0.32	mg/L	Highly Toxic	Acceptable	42053901 Ward 1991

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic	Marine/Estuarine fish Inland silverside (<i>Menidia beryllina</i>)	Acute	96h LC50	0.34	mg/L	Highly Toxic	Acceptable (for formulated product)	41969901 Ward 1991
Aquatic	Freshwater fish Rainbow Trout (<i>Oncorhynchus mykiss</i>)	Chronic	NOEC LOEC	0.019 0.034	mg/L mg/L	Very Highly Toxic Parameter: growth effects	Acceptable	43230201 Weinberg et al. 1994
Aquatic	Freshwater invertebrate Water flea (<i>Daphnia magna</i>)	Acute	48h EC50	1.2	mg/L	Moderately Toxic	Supplemental	00151963 Batchelder 1980
Aquatic	Freshwater invertebrate Water flea (<i>Daphnia magna</i>)	Acute	48h EC50	8.63	mg/L	Slightly Toxic	Acceptable	00151965 47006032 Milazzo 1981
Aquatic	Freshwater invertebrate Water flea (<i>Daphnia magna</i>)	Acute	48h EC50	0.25	mg/L	Highly Toxic	Acceptable	43442603 Weinberg et al. 1994
Aquatic	Mollusca Eastern oyster (<i>Crassostrea virginica</i>) (shell deposition)	Acute	96h EC50	0.33 (Species)	mg/L	Highly Toxic	Acceptable	41971602 Boeri 1991
Aquatic	Mollusca Eastern oyster (<i>Crassostrea virginica</i>) (shell deposition)	Acute	96h EC50	0.23	mg/L	Highly Toxic	Acceptable (for formulated product)	41969903 Boeri 1991
Aquatic	Crustacea Grass Shrimp (<i>Palaemonetes pugio</i>)	Acute	96h LC50	1.8	mg/L	Moderately Toxic	Acceptable	41971601 Boeri 1991
Aquatic	Crustacea Grass Shrimp (<i>Palaemonetes pugio</i>)	Acute	96h LC50	0.77	mg/L	Highly Toxic	Acceptable (for formulated product)	41969902 Ward 1991
Aquatic plants	Vascular aquatic plant Duckweed (<i>Lemna gibba</i>)	Acute	14d EC50	0.86	mg/L	Parameter: Growth and Reproduction	Supplemental (downgraded)	42719101 Milazzo 1993

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic plants	Vascular aquatic plant Duckweed (<i>Lemna gibba</i>)	Chronic	14d NOAEC	< 0.111	mg/L	Parameter: Growth and Reproduction	Supplemental	42719101 Milazzo 1993
Aquatic plants	Non-vascular aquatic plant Green algae (<i>Kirchneria subcapitata</i>) (Formerly <i>Selenastrum capricornutum</i>)	Acute	5d EC50	2.5	mg/L	Parameter: Growth	Acceptable	41633704, 42090422 Cowgill & Millazzo 1989
Aquatic plants	Non-vascular aquatic plant Blue-green algae (<i>Anabeana flos-aquae</i>)	Acute	5d EC50	1.42	mg/L	Parameter: Growth	Acceptable	42721101 Hughes 1993
Aquatic plants	Non-vascular aquatic plant Freshwater diatom (<i>Navicula pelliculosa</i>)	Acute	5d EC50	0.073	mg/L	Parameter: Growth	Acceptable	42721102 Hughes 1993
Aquatic plants (BEE)	Non-vascular aquatic plant Freshwater diatom (<i>Navicula pelliculosa</i>)	Chronic	5d NOAEC	0.0014	mg/L	Parameter: Growth	Acceptable	42721102 Hughes 1993
Aquatic plants	Non-vascular aquatic plant Marine diatom (<i>Skeletonema costatum</i>)	Acute	5d EC50	0.84	mg/L	Parameter: Growth	Acceptable	42721103 Hughes 1993
Terrestrial	Avian Northern bobwhite Quail (<i>Colinus virginianus</i>)	Acute oral	21d LD50	529	mg/kg bw	Practically Non-Toxic	Acceptable	41902002 Campbell & Lynn 1991
Terrestrial	Avian Northern bobwhite Quail (<i>Colinus virginianus</i>)	Acute oral	14d LD50	611	mg/kg bw	Practically Non-Toxic	Acceptable	41902003 Campbell & Lynn 1991
Terrestrial	Avian Northern bobwhite Quail (<i>Colinus virginianus</i>)	Subacute dietary	8d LC50	6038	ppm	Slightly Toxic	Acceptable	00134180 Wildlife Int'l 1978
Terrestrial	Avian Northern bobwhite Quail (<i>Colinus virginianus</i>)	Subacute dietary	8d LC50	3885	ppm	Slightly Toxic	Acceptable	41905501 Lynn et al. 1991
Terrestrial	Avian Mallard duck (<i>Anas platyrhynchos</i>)	Subacute dietary	8d LC50	> 6689	ppm	Practically Non-Toxic	Acceptable	00134179 Wildlife Int'l 1977

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial	Avian Mallard duck (<i>Anas platyrhynchos</i>)	Subacute dietary	8d LC50	> 3885	ppm	Practically Non-Toxic	Acceptable	41905501 Lynn et al. 1992
Terrestrial	Rodent Rat	Acute oral	LD50	578 (M & F)	mg/kg	Slightly Toxic	Acceptable	40557004 Wall et al. 1987
Terrestrial	Honeybee (<i>Apis mellifera</i>)	Acute	Contact 48h LD50	> 72	µg/bee	Practically Non-Toxic	Acceptable	41219109 Dingledine 1985
Terrestrial Plant	Terrestrial dicot Soybean (<i>Glycine max</i>) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Sunflower (<i>Helianthus annus</i>) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Sunflower (<i>Helianthus annus</i>) (Vegetative Vigor)	Acute	51d EC25	0.006	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995
Terrestrial Plant	Terrestrial dicot Sunflower (<i>Helianthus annus</i>) (Vegetative Vigor)	Acute	NOAEC	0.028	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995
Terrestrial Plant	Terrestrial dicot Tomato (<i>Lycopersicon esculentum</i>) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Radish (<i>Raphanus sativus</i>) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial monocot Onion (<i>Allium cepa</i>) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial monocot Onion (<i>Allium cepa</i>) (Seedling Emergence)	Acute	51d EC25	0.053	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995

Habitat	Taxa	Effect Type	Endpoint	Value (expressed as the acid equivalent)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial Plant (BEE)	Terrestrial monocot Onion (<i>Allium cepa</i>) (Seedling Emergence)	Acute	NOAEC	0.0021	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995
Terrestrial Plant	Terrestrial monocot Onion (<i>Allium cepa</i>) (Vegetative Vigor)	Acute	51d EC25	0.063	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995
Terrestrial Plant	Terrestrial monocot Onion (<i>Allium cepa</i>) (Vegetative Vigor)	Acute	NOAEC	< 0.063	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995
Terrestrial Plant	Terrestrial dicot Sugarbeet (<i>Beta vulgaris</i>) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial monocot Barley (<i>Hordeum vulgare</i>) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial monocot Corn (<i>Zea mays</i>) (Seedling Emergence)	Acute	21d EC25	> 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial monocot Wheat (<i>Triticum aestivum</i>) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Oilseed rape (<i>Brassica napus</i>) (Seedling Emergence)	Acute	21d EC25	< 8.0	lbs ae/A	Parameter: emergence	Acceptable	41734301 Weseloh & Stockdale 1990
Terrestrial Plant	Terrestrial dicot Alfalfa (<i>Medicago sativa</i>) (Seedling Emergence)	Acute	51d EC25	0.045	lbs/A	parameter: emergence	Acceptable	43650001 Schwab 1995
Terrestrial Plant (BEE)	Terrestrial dicot Alfalfa (<i>Medicago sativa</i>) (Seedling Emergence)	Acute	NOAEC	0.0026	lbs/A	parameter: emergence	Acceptable	43650001 Schwab 1995

Triclopyr Butoxyethyl Ester (BEE)/Picloram ethyl ester Mixture

Habitat	Taxa	Effect Type	Endpoint	Value (% ai)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic plants	Vascular aquatic plant Duckweed (<i>Lemna gibba</i>)	Acute	14d EC50 NOEC	6.6 (9.2; 99.8) 1.03 (1.43; 15.9) (TBEE = 9.2% ai)	mg/L mg/L	Parameter: Growth (plants)	Acceptable	43230310 Milazzo 1994
Aquatic plants	Non-vascular aquatic plant Blue-green algae (<i>Anabeana flos-aquae</i>)	Acute	5d EC50	0.20 (0.276; 3.0) (TBEE = 9.2% ai)	mg/L	Parameter: Growth	Invalid	43230307 Boeri et al. 1994
Aquatic plants	Non-vascular aquatic plant Green algae (<i>Kirchneria subcapitata</i>) (Formerly <i>Selenastrum capricornutum</i>)	Acute	5d EC50 NOEC	0.32 (0.441; 4.9) 0.211 (0.294; 3.2) (TBEE = 9.2% ai)	mg/L mg/L	Parameter: Growth	Supplemental	42645901 Hughes et al. 1993
Aquatic plants	Non-vascular aquatic plant Marine diatom (<i>Skeletonema costatum</i>)	Acute	5d EC50	0.12 (0.166; 1.8) (TBEE = 9.2% ai)	mg/L	Parameter: Growth	Invalid	43230304 Boeri et al. 1994
Aquatic plants	Non-vascular aquatic plant Freshwater diatom (<i>Navicula pelliculosa</i>)	Acute	5d EC50	0.36 (0.50; 5.4) (TBEE = 9.2% ai)	mg/L	Parameter: Growth	Invalid	43230301 Boeri et al. 1994
Terrestrial	Honeybee (<i>Apis mellifera</i>)	Acute	Contact 44h LD50	> 1.62 (> 2.25; > 25) (TBEE = 9.2% ai)	µg/bee	Practically Non-Toxic	Acceptable	42625901 Hoxter et al.1992
Terrestrial Plant	Terrestrial dicot Radish (<i>Raphanus sativus</i>) (Vegetative Vigor)	Acute	21 d EC25 NOEL	0.008 (0.035) 0.0035 (0.015) (TBEE = 23.5% ai)	lbs ae/A lbs ae/A	Parameter: height	Supplemental	41296501 Weseloh and Stockdale 1989
Terrestrial Plant	Terrestrial dicot Soybean (<i>Glycine max</i>) (Vegetative Vigor)	Acute	55 d EC25 NOEL	0.00005 (0.00021) 0.000026 (0.00011) (TBEE = 23.5% ai)	lbs ae/A lbs ae/A	Parameter: height	Supplemental	41296501 Weseloh and Stockdale 1989

Habitat	Taxa	Effect Type	Endpoint	Value (% ai)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial Plant	Terrestrial monocot Wheat (<i>Triticum aestivum</i>) (Vegetative Vigor)	Acute	42 d EC25 NOEL	0.05 (0.21) 0.0146 (0.062) (TBEE = 23.5% ai)	lbs ae/A lbs ae/A	Parameter: height	Supplemental	41296501 Weseloh and Stockdale 1989
Terrestrial Plant	Terrestrial dicot Drybean (<i>Phaseolus vulgaris</i>) (Seedling Emergence)	Acute	14 d EC25 NOEL	0.0000009 (0.000004) < 0.000007 (< 0.00003) (TBEE = 23.5% ai)	lbs ae/A lbs ae/A	Parameter: emergence	Supplemental	41296501 Weseloh and Stockdale 1989
Terrestrial Plant	Terrestrial monocot Onion (<i>Allium cepa</i>) (Seedling Emergence)	Acute	14 d EC25 NOEL	0.008 (0.035) 0.004 (0.0156) (TBEE = 23.5% ai)	lbs ae/A lbs ae/A	Parameter: emergence	Supplemental	41296501 Weseloh and Stockdale 1989
Terrestrial Plant	Terrestrial monocot Wheat (<i>Triticum aestivum</i>) (Seedling Emergence)	Acute	21d EC25 NOEL	0.006 (0.025) 0.02 (0.08) (TBEE = 23.5% ai)	lbs ae/A lbs ae/A	Parameter: emergence	Supplemental	41296501 Weseloh and Stockdale 1989
Terrestrial Plant	Terrestrial Lima bean (<i>Phaseolus lunatus</i>) (Seedling Emergence)	Acute	21d EC25 NOEL	0.00003 (0.00004; 0.00042) (TBEE = 9.2% ai)	lbs ae/A lbs ae/A	Parameter: shoot height	Supplemental	43276601 Schwab 1994
Terrestrial Plant	Terrestrial monocot Barley (<i>Hordeum vulgare</i>) (Seedling Emergence)	Acute	21d EC25 NOEL	0.57 (0.791; 8.6) (TBEE = 9.2% ai))	lbs ae/A lbs ae/A	Parameter: shoot weight	Supplemental	43276601 Schwab 1994

Combined –Most sensitive when converted to acid equivalent (Acid, TEA or BEE)

Habitat	Taxa	Effect Type	Endpoint	Form Tested	Value (acid equivalent)	Value (Original Value)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic (BEE)	Freshwater fish Bluegill sunfish (<i>Lepomis macrochirus</i>)	Acute	96h LC50	F (Garlon 4) 62.9% ai TBEE	0.32	0.44	mg/L	Highly Toxic	Acceptable	43442601 Weinberg et al. 1994
Aquatic (BEE)	Freshwater fish Bluegill sunfish (<i>Lepomis macrochirus</i>)	Acute	96h LC50	TGAI 96.98% ai TBEE	0.26	0.36	mg/L	Highly Toxic	Acceptable	42917901 Woodburn et al. 1993
Aquatic (BEE)	Freshwater fish Rainbow Trout (<i>Oncorhynchus mykiss</i>)	Acute	96h LC50	F (Garlon 4) 62.9% ai TBEE	0.70	0.98	mg/L	Highly Toxic	Acceptable	43442602 Weinberg et al. 1994
Aquatic (BEE)	Freshwater fish Rainbow Trout (<i>Oncorhynchus mykiss</i>)	Acute	96h LC50	TGAI 96.98% ai TBEE	0.47	0.65	mg/L	Highly Toxic	Acceptable	42884501 Woodburn et al. 1993
Aquatic (BEE)	Freshwater fish Rainbow Trout (<i>Oncorhynchus mykiss</i>)	Chronic	NOEC LOEC	TGAI 96.98% ai TBEE	0.019 0.034	0.0263 0.048	mg/L	Very Highly Toxic Parameter: growth effects	Acceptable	43230201 Weinberg et al. 1994
Aquatic (TEA)	Freshwater fish Fathead Minnow (<i>Pimephales promelas</i>)	Chronic	NOEC LOEC	F 44.9% ai TEA	> 32.2 <50.2	>46.7 (>104) <72.7 (<162)	mg/L	Parameter: Length	Acceptable	00151958 Mayes 1983
Aquatic (BEE)	Freshwater invertebrate Water flea (<i>Daphnia magna</i>)	Acute	48h EC50	F (Garlon 4) 62.2% ai TBEE	0.25	0.35	mg/L	Highly Toxic	Acceptable	43442603 Weinberg et al. 1994
Aquatic (BEE)	Freshwater invertebrate Water flea (<i>Daphnia magna</i>)	Acute	48h EC50	TGAI 96.4% ai TBEE	1.2	1.7 (nominal)	mg/L	Moderately Toxic	Supplemental	00151963 Batchelder 1980
Aquatic (TEA)	Freshwater invertebrate Water flea (<i>Daphnia magna</i>)	Chronic	NOEC LOEC	F 44.9% ai TEA	25 46.2	36.2 (80.7) 66.9 (149.0)	mg/L	total young and mean brood size	Acceptable	00151959, 42090411, 92189013 Gerisch 1982
Aquatic plants (BEE)	Vascular aquatic plant Duckweed (<i>Lemna gibba</i>)	Acute	14d EC50	TGAI 96.98% ai TBEE	0.86	1.2	mg/L	Parameter: Growth and Reproduction	Supplemental	42719101 Milazzo et al. 1993

Habitat	Taxa	Effect Type	Endpoint	Form Tested	Value (acid equivalent)	Value (Original Value)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Aquatic plants (BEE)	Duckweed (<i>Lemna gibba</i>)	Chronic	14d NOAEC	TGAI 96.98% ai TBEE	< 0.111	< 0.155	mg/L	Parameter: Growth and Reproduction	Supplemental	42719101 Milazzo et al. 1993
Aquatic plants (TEA)	Vascular aquatic plant Duckweed (<i>Lemna gibba</i>)	Acute	14d EC50	F 44.9% ai TEA	6.1	8.8 (19.5)	mg/L	Parameter: Growth and Reproduction	Acceptable	41633709 Cowgill 1987
Aquatic plants (BEE)	Non-vascular aquatic plant Freshwater diatom (<i>Navicula pelliculosa</i>)	Acute	5d EC50	TGAI 96.98% ai TBEE	0.073	0.102	mg/L	Parameter: Growth	Acceptable	42721102 Hughes 1993
Aquatic plants (BEE)	Non-vascular aquatic plant Freshwater diatom (<i>Navicula pelliculosa</i>)	Chronic	5d NOAEC	TGAI 96.98% ai TBEE	0.0014	0.002	mg/L	Parameter: Growth	Acceptable	42721102 Hughes 1993
Aquatic plants (TEA)	Non-vascular aquatic plant Blue-green algae (<i>Anabeana flos-aquae</i>)	Acute	7d EC50	F 45% ai TEA	4.1	5.9	mg/L	Parameter: Growth	Acceptable	41633706 Hughes 1987
Aquatic plants (BEE)	Non-vascular aquatic plant Green algae (<i>Kirchneria subcapitata</i>) (Formerly <i>Selenastrum capricortum</i>)	Acute	5d EC50	F 61.3% ai TBEE	2.5	3.4 (5.6)	mg/L	Parameter: Growth	Acceptable	41633704, 42090422 Cowgill & Millazzo 1989
Terrestrial (BEE)	Avian Northern bobwhite Quail (<i>Colinus virginianus</i>)	Acute oral	21d LD50	TGAI 96.1% ai TBEE	529	735	mg/kg bw	Practically Non-Toxic	Acceptable	41902002 Campbell & Lynn 1991
Terrestrial (BEE)	Avian Northern bobwhite Quail (<i>Colinus virginianus</i>)	Acute oral	14d LD50	F (Garlon 4) 62.9% ai TBEE	611	849 (1350)	mg/kg bw	Practically Non-Toxic	Acceptable	41902003 Campbell & Lynn 1991
Terrestrial (Acid)	Avian Northern bobwhite Quail (<i>Colinus virginianus</i>)	Subacute dietary	LC50	TGAI Acid	2934	2934	ppm	Slightly Toxic	Acceptable	40346403 Dow Chemical 1976
Terrestrial (BEE)	Avian Northern bobwhite Quail (<i>Colinus virginianus</i>)	Subacute dietary	8d LC50	TGAI 96.1% ai TBEE	3885	5401	ppm	Slightly Toxic	Supplemental	41905501 Lynn et al. 1991
Terrestrial (BEE)	Avian Mallard duck (<i>Anas platyrhynchos</i>)	Subacute dietary	8d LC50	TGAI 96.1% ai TBEE	> 3,885	> 5,401	ppm	Practically Non-Toxic	Acceptable	41905502 Lynn et al. 1992

Habitat	Taxa	Effect Type	Endpoint	Form Tested	Value (acid equivalent)	Value (Original Value)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial (Acid)	Avian Mallard duck (<i>Anas platyrhynchos</i>)	Chronic	NOAEC LOAEC	TGAI 98.9% ai Acid	100 200	100 200	ppm	# of 14 d old survivors	Acceptable	00031250 Beavers & Fink 1980
Terrestrial (Acid)	Avian Northern bobwhite Quail (<i>Colinus virginianus</i>)	Chronic	NOAEC LOAEC LOAEC	TGAI 98.9% ai Acid	500 >500 200	500 >500 200	ppm ppm ppm	No sign. reproductive impairment	Acceptable	00031251 Beavers & Fink 1979
Terrestrial (Acid)	Rodent Rat	Acute oral	LD50	TGAI Acid	729 (M) 630 (F)	729 (M) 630 (F)	mg/kg	Slightly Toxic	Acceptable	00031940 Henck et al. 1979
Terrestrial (TBEE)	Rodent Rat	Acute oral	LD50	TGAI 96% ai TBEE	578 (M &F)	803 (M & F)	mg/kg	Slightly Toxic	Acceptable	40557004 Wall et al. 1987
Terrestrial (TEA)	Rodent Rat	Acute oral	LD50	TGAI 44.9 % ai TEA	572 (M &F)	1847 (M & F)	mg/kg	Slightly Toxic	Acceptable	41443301 Mizell & Lomax 1988
Terrestrial (Acid)	Rodent	Chronic	NOAEL LOAEL	TGAI 99.4% ai Acid	25 250	25 250	mg/kg bw mg/kg bw	Reproductive/ Systemic	Acceptable (RED 1998)	43545701 Vedula et al 1995
Terrestrial (Acid)	Rodent	Chronic	NOAEL LOAEL	TGAI 99.4% ai Acid	5 25	5 25	mg/kg bw mg/kg bw	Reproductive/ Offspring	Acceptable (HED 2002)	43545701 Vedula et al 1995
Terrestrial (Acid)	Honeybee (<i>Apis mellifera</i>)	Acute	Contact 48h LD50	TGAI 99.2% ai Acid	>100	>100	µg/bee	Practically Non-Toxic	Acceptable	40356602 Dingledine 1985
Terrestrial (BEE)	Honeybee (<i>Apis mellifera</i>)	Acute	Contact 48h LD50	TGAI 97.7% ai TBEE	> 72	> 100	µg/bee	Practically Non-Toxic	Acceptable	41219109 Dingledine 1985
Terrestrial Plant (TEA)	Terrestrial dicot Sunflower (<i>Helianthus annus</i>) (Vegetative Vigor)	Acute	14d EC25	F 46.2% ai TEA	0.005	0.0076	lbs/A	parameter: shoot length	Acceptable	43129801 Schwab 1993
Terrestrial Plant (BEE)	Terrestrial dicot Sunflower (<i>Helianthus annus</i>) (Vegetative Vigor)	Acute	51d EC25	F (Garlon 4) 62.2% ai TBEE	0.006	0.0089	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995

Habitat	Taxa	Effect Type	Endpoint	Form Tested	Value (acid equivalent)	Value (Original Value)	Unit	Toxicity category	Study Classification	Source (MRID or Acc#, author)
Terrestrial Plant (BEE)	Terrestrial monocot Onion (<i>Allium cepa</i>) (Vegetative Vigor)	Acute	51d EC25	F (Garlon 4) 62.2% ai TBEE	0.063	0.0888	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995
Terrestrial Plant (TEA)	Terrestrial monocot Onion (<i>Allium cepa</i>) (Vegetative Vigor)	Acute	14d EC25	F 46.2% ai TEA	0.114	0.166	lbs/A	parameter: shoot weight	Acceptable	43129801 Schwab 1993
Terrestrial Plant (BEE)	Terrestrial dicot Alfalfa (<i>Medicago sativa</i>) (Seedling Emergence)	Acute	51d EC25	F (Garlon 4) 62.2% ai TBEE	0.045	0.0622	lbs/A	parameter: emergence	Acceptable	43650001 Schwab 1995
Terrestrial Plant (BEE)	Terrestrial monocot Onion (<i>Allium cepa</i>) (Seedling Emergence)	Acute	51d EC25	F (Garlon 4) 62.2% ai TBEE	0.053	0.0732	lbs/A	parameter: shoot weight	Acceptable	43650001 Schwab 1995

Appendix L Ecological Effects Data

Aquatic Animals

Coho salmon (<i>Oncorhynchus kisutch</i>), Static, LC50, / Diuron, 95% ai	96 hr LC50 < 2.4 ppm ai (slope = N.R.)	40098001, 1986 /Supplemental
Cutthroat trout (<i>Oncorhynchus clarki</i>), Static, LC50, / Diuron, 95% ai	96 hr LC50 1.4 ppm ai (slope = N.R.)	40094602, 1980 /Core
Cutthroat trout (<i>Oncorhynchus clarki</i>), Static, LC50, / Diuron, 95% ai	96 hr LC50 0.71 ppm ai (slope = N.R.)	40098001, 1986 /Supplemental
Fathead minnow (<i>Pimephales promelas</i>), Static, LC50, / Diuron, 98.6% ai	96 hr LC50 14.2 ppm ai (slope = N.R.)	00141636, 1975 /Supplemental
Fathead minnow (<i>Pimephales promelas</i>), Static, LOEC, / Diuron, 98.6% ai	60day LOEC 61.8 ppb ai (slope = N.R.)	00141636, 1975 /Core

Lake trout (<i>Salvelinus namaycush</i>), Static, LC50, / Diuron, 95% ai	96 hr LC50 2.7 ppm ai (slope = N.R.)	40094602, 1980 /Core
--	--------------------------------------	----------------------

Lake trout (<i>Salvelinus namaycush</i>), Static, LC50, / Diuron, 95% ai	96 hr LC50 1.2 ppm ai (slope = N.R.)	40098001, 1986 /Supplemental
--	--------------------------------------	------------------------------

Rainbow trout (<i>Oncorhynchus mykiss</i>), Static, LC50, / Diuron, 80% ai	96 hr LC50 19.6 ppm ai (slope = N.R.)	42046002, 1991 /Core
--	---------------------------------------	----------------------

Rainbow trout (<i>Oncorhynchus mykiss</i>), Static, LC50, / Diuron, 28% ai	96 hr LC50 23.8 ppm ai (slope = N.R.)	TN 0897, 1975 /Core
--	---------------------------------------	---------------------

Rainbow trout (<i>Oncorhynchus mykiss</i>), Static, LC50, / Diuron, 80WP% ai	96 hr LC50 16 ppm ai (slope = N.R.)	40094602, 1980 /Supplemental
--	-------------------------------------	------------------------------

Rainbow trout (<i>Oncorhynchus mykiss</i>), Static, LC50, / Diuron, 95% ai	96 hr LC50 1.95 ppm ai (slope = N.R.)	TN 1020, 1976 /Core
--	---------------------------------------	---------------------

Sheepshead minnow (<i>Cyprinodon variegatus</i>), Static, LC50, / Diuron, 99% ai	96 hr LC50 6.7 ppm ai (slope = N.R.)	41418803, 1986 /Core
--	--------------------------------------	----------------------

Sheepshead minnow (<i>Cyprinodon variegatus</i>), Flow-through, LOEC, / Diuron, 96.8% ai	38 D LOEC < 0.44 ppm ai (slope = N.A.)	42312901, 1992 /Supplemental
--	--	------------------------------

Striped mullet (<i>Mugil cephalus</i>), Static, LC50, / Diuron, 95% ai	48 hr LC50 6.3 ppm ai (slope = N.R.)	40228401, 1986 /Supplemental
--	--------------------------------------	------------------------------

Brown shrimp (<i>Penaeus aztecus</i>), Flow-through, LC50, / Diuron, 95% ai	48 hr LC50 > 1.0 ppm ai (slope = N.A.)	40228401, 1986 /Supplemental
---	--	------------------------------

Mysid (<i>Americamysis bahia</i>), Static, LC50, / Diuron, 99% ai	96 hr LC50 1.1 ppm ai (slope = 12.35)	41418801, 1987 /Supplemental
Mysid (<i>Americamysis bahia</i>), Static, LOEC, / Diuron, 96.8% ai	28 D LOEC 0.56 ppm ai (slope = N.R.)	42500601, 1992 /Core
Scud (<i>Gammarus fasciatus</i>), Static, LC50, / Diuron, 95% ai	96 hr LC50 0.16 ppm ai (slope = N.R.)	40094602, 1980 /Core
Water flea (<i>Simocephalus</i> sp.), Static, EC50, / Diuron, 95% ai	48 hr EC50 2.0 ppm ai (slope = N.R.)	40094602, 1980 /Core
Water flea (<i>Daphnia magna</i>), Static, EC50, / Diuron, 80% ai	48 hr EC50 8.4 ppm ai (slope = 9.10)	42046003, 1991 /Core
Water flea (<i>Daphnia pulex</i>), Static, EC50, / Diuron, 95% ai	48 hr EC50 1.4 ppm ai (slope = N.R.)	40094602, 1980 /Core

Water flea (<i>Daphnia magna</i>), Static, LOEC, / Diuron, 98.2% ai	28 D LOEC 0.2 ppm ai (slope = N.R.)	TN 2418, 1979 /Supplemental
Stonefly (<i>Pteronarcys</i> sp.), Static, LC50, / Diuron, 95% ai	96 hr LC50 1.2 ppm ai (slope = N.R.)	40094602, 1980 /Core
Eastern oyster (<i>Crassostrea</i> <i>virginica</i>), Flow-through, EC50, / Diuron, 96.8% ai	96 hr EC50 4.8 ppm ai (slope = 4.49)	42217201, 1991 /Core
Eastern oyster (<i>Crassostrea</i> <i>virginica</i>), Flow-through, EC50, / Diuron, 95% ai	96 hr EC50 1.8 ppm ai (slope = N.R.)	40228401, 1986 /Core

Terrestrial Animals

Bobwhite quail (<i>Colinus</i> <i>virginianus</i>), O, LD50, / Diuron, 92.8% ai	21 D LD50 940 mg/kg ai (slope = 4.01)	50150170, 1985 /Core
---	---	-------------------------

Bobwhite quail (<i>Colinus virginianus</i>), Dietary, LC50, / Diuron, >95% ai	8 D LC50 1730 ppm ai (slope = 7.22)	00022923, 1975 /Core
---	-------------------------------------	----------------------

Japanese quail (<i>Coturnix japonica</i>), Dietary, LC50, / Diuron, >95% ai	8 D LC50 > 5000 ppm ai (slope = N.R.)	00022923, 1975 /Supplemental
---	---------------------------------------	------------------------------

Mallard duck (<i>Anas platyrhynchos</i>), Oral, LD50, / Diuron, 95% ai	14 D LD50 > 2000 mg/kg ai (slope = N.R.)	00160000, 1970 /Core
--	--	----------------------

Mallard duck (<i>Anas platyrhynchos</i>), Dietary, LC50, / Diuron, >95% ai	8 D LC50 > 5000 ppm ai (slope = N.R.)	00022923, 1975 /Core
--	---------------------------------------	----------------------

Ring-necked pheasant (<i>Phasianus colchicus</i>), Dietary, LC50, / Diuron, >95% ai	8 D LC50 > 5000 ppm ai (slope = N.R.)	00022923, 1975 /Core
---	---------------------------------------	----------------------

Honey bee (<i>Apis mellifera</i>), Contact, LD50, / Diuron, Tech% ai	48 hr LD50 > 145.03 ug/Bee ai (slope = N.R.)	36935, 1975 /Core
---	--	-------------------

Plants

Algae (<i>Nitzschia closterium</i>), Static, EC50, / Diuron, 95% ai	72 hr EC50 50 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Algae (<i>Amphora exigua</i>), Static, EC50, / Diuron, 95% ai	72 hr EC50 31 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Algae (<i>Stauroneis amphoroides</i>), Static, EC50, / Diuron, 95% ai	72 hr EC50 31 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Algae (<i>Achnanthes brevipes</i>), Static, EC50, / Diuron, 95% ai	72 hr EC50 24 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Algae (<i>Cyclotella nana</i>), Static, EC50, / Diuron, 95% ai	72 hr EC50 39 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Algae (<i>Isochrysis galbana</i>), Static, EC50, / Diuron, 95% ai	240hr EC50 10 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Algae (<i>Monochrysis lutheri</i>), Static, EC50, / Diuron, 95% ai	72 hr EC50 18 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Algae (<i>Porphyridium cruentum</i>), Static, EC50, / Diuron, 95% ai	72 hr EC50 24 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Diatom (<i>Thalassiosira fluviatilis</i>), Static, EC50, / Diuron, 95% ai	72 hr EC50 95 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental

Green algae (<i>Selenastrum capricornutum</i>), Static, EC50, / Diuron, 96.8% ai	96 hr EC50 2.4 ppb ai (slope = 2.96)	42218401, 1991 /Core
Green algae (<i>Chlorella</i> sp.), Static, EC50, / Diuron, 95% ai	72 hr EC50 19 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Green algae (<i>Chlorococcum</i> sp.), Static, EC50, / Diuron, 95% ai	72 hr EC50 10 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Green algae (<i>Chlamydomonas</i> sp.), Static, EC50, / Diuron, 95% ai	72 hr EC50 37 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Green algae (<i>Dunaliella tertiolecta</i>), Static, EC50, / Diuron, 95% ai	240hr EC50 20 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Green algae (<i>Platymonas</i> sp.), Static, EC50, / Diuron, 95% ai	72 hr EC50 17 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Green algae (<i>Neochloris</i> sp.), Static, EC50, / Diuron, 95% ai	72 hr EC50 28 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Marine diatom (<i>Phaeodactylum tricornutum</i>), Static, EC50, / Diuron, 95% ai	240hr EC50 10 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Marine diatom (<i>Navicula incerta</i>), Static, EC50, / Diuron, 95% ai	72 hr EC50 93 ppb ai (slope = N.R.)	40228401, 1986 /Supplemental
Rape (<i>Brassica</i> sp.), Veg. Vigor sw, EC25, / Diuron, 97.3% ai	21 D EC25 0.0331 lb/A ai (slope = N.R.)	44113401, 1996 /Core
Sorghum (<i>Sorghum halepense</i>), Seed Emerg. sh, EC25, / Diuron, 96.8% ai	14 D EC25 < 12 lb/A ai (slope = NA)	42398501, 1992 /Core
Sorghum (<i>Sorghum bicolor</i>), Veg. Vigor sw, EC25, / Diuron, 97.3% ai	21 D EC25 0.0753 lb/A ai (slope = N.R.)	44113401, 1996 /Core

Sorghum (<i>Sorghum bicolor</i>), Seed Emerg. sh, EC25, / Diuron, 97.3% ai	14 D EC25 0.81 lb/A ai (slope = N.R.)	44113401, 1996 /Core
Soybean (<i>Glycine max</i>), Seed Emerg. , EC25, / Diuron, 96.8% ai	14 D EC25 > 12 lb/A ai (slope = NA)	42398501, 1992 /Core
Soybean (<i>Glycine max</i>), Veg. Vigor sw, EC25, / Diuron, 97.3% ai	21 D EC25 0.012 lb/A ai (slope = N.R.)	44113401, 1996 /Core
Soybean (<i>Glycine max</i>), Seed Emerg. sh, EC25, / Diuron, 97.3% ai	14 D EC25 > 12 lb/A ai (slope = N.R.)	44113401, 1996 /Core
Sugarbeet (<i>Beta vulgaris</i>), Seed Emerg. sh, EC25, / Diuron, 96.8% ai	14 D EC25 < 12 lb/A ai (slope = NA)	42398501, 1992 /Core
Sugarbeet (<i>Beta vulgaris</i>), Veg. Vigor sw, EC25, / Diuron, 97.3% ai	21 D EC25 0.0087 lb/A ai (slope = N.R.)	44113401, 1996 /Core
Sugarbeet (<i>Beta vulgaris</i>), Seed Emerg. sw, EC25, / Diuron, 97.3% ai	14 D EC25 0.092 lb/A ai (slope = N.R.)	44113401, 1996 /Core
Tomato (<i>Lycopersicon esculentum</i>), Seed Emerg. , EC25, / Diuron, 96.8% ai	14 D EC25 > 0.038 lb/A ai (slope = NA)	42398501, 1992 /Core
Tomato (<i>Lycopersicon esculentum</i>), Seed Emerg. sw, EC25, / Diuron, 97.3% ai	14 D EC25 0.075 lb/A ai (slope = N.R.)	44113401, 1996 /Core
Tomato (<i>Lycopersicon esculentum</i>), Veg. Vigor sw, EC25, / Diuron, 97.3% ai	21 D EC25 0.0017 lb/A ai (slope = N.R.)	44113401, 1996 /Core
Wheat (<i>Triticum aestivum</i>), Seed Emerg. sh, EC25, / Diuron, 96.8% ai	14 D EC25 < 12 lb/A ai (slope = NA)	42398501, 1992 /Supplemental
Wheat (<i>Triticum aestivum</i>), Veg. Vigor sw, EC25, / Diuron, 97.3% ai	21 D EC25 0.0208 lb/A ai (slope = 0.90)	44113401, 1996 /Core

Wheat (*Triticum aestivum*), Seed Emerg. sw,
EC25, / Diuron, 97.3% ai

14 D EC25 1.05 lb/A ai
(slope = N.R.)

44113401, 1996 /Core

APPENDIX A

Ecological Effects

a. Ecological Effects Data

(1) Terrestrial Animal Data

Avian Acute Toxicity

Avian Acute Oral Toxicity Findings			
Species	% Test Material (TGAI)	LD ₅₀	Conclusion
Bobwhite Quail	92.8	940 mg/kg	slightly toxic

These results show that linuron is slightly toxic to birds on an acute basis. The guideline requirement for the avian acute oral LD₅₀ study is fulfilled. (MRID 00150170)

Avian Subacute Dietary Toxicity

No acceptable avian dietary toxicity studies on technical linuron have been submitted for review. However, the following data from the USFWS (United States Fish and Wildlife Service) using a 50% formulation were considered. Some toxicity in formulation testing may be due to ingredients other than the active ingredient. Other formulations may be more or less toxic, depending on their ingredients. Technical testing allows prediction of the toxicity due to the active ingredient across all formulations. Therefore, tests with the technical material are still required.

Avian Subacute Dietary Toxicity Findings			
Species	% Test Material	LC ₅₀	Conclusions
Mallard Duck	50	1700 ppm	slightly toxic
Japanese Quail	50	>5,000 ppm	practically nontoxic
Ring-necked Pheasant	50	3438 ppm	slightly toxic

The USFWS extrapolation suggests that 100 percent active ingredient material would be considered "slightly toxic" to the mallard and ring-necked pheasant and "practically nontoxic" to the Japanese quail. (MRID 00034769).

Avian Reproductive Toxicity

Avian reproduction studies are required when birds may be exposed repeatedly or continuously through persistence, bioaccumulation, or multiple applications, or if mammalian reproduction tests indicate reproductive hazard. Because linuron is persistent and can be applied more than one time during a season these studies were required.

Avian Reproductive Toxicity		
Species	% Test Material	Results
Mallard Duck	98.4	NOEL = 100 ppm LOEL = 300ppm(1)
Bobwhite Quail	98.4	NOEL = 100 ppm LOEL = 300 ppm(2)

(1) Treatment-related effects in adult body weight, feed consumption, egg production, and eggshell thickness.

(2) Treatment-related effects in egg production, hatchability, and offspring survival.

There are sufficient data to characterize the effects of linuron on avian reproduction. The No Observable Effects Level for the mallard duck is 100 ppm and the Lowest Observable Effects Level is 300 ppm. (MRID 42541802)

The No Observable Effects Level for the bobwhite quail is 100 ppm and the Lowest Observable Effects Level is 300 ppm. (MRID 42541801)

Toxicity to Mammals

Mammalian Acute Oral Toxicity Findings		
Species	LD ₅₀ (mg/kg)	Conclusion
Rat	2600	practically nontoxic

The available data indicate that at a lowest acute oral LD50 of 2600 mg/kg, linuron is practically nontoxic to the rat.

Toxicity to Insects

The minimum data required to establish the acute toxicity to honey bees is an acute contact LD₅₀ study with the technical material.

Acute Toxicity to Insects			
Species	% Test Material	LD ₅₀	Conclusion
<i>Apis mellifera</i>	not reported	120.86 ug/bee	practically nontoxic

There is sufficient information to characterize linuron as practically non-toxic to bees. (MRID 00018842).

(2) Aquatic Animal Data

Freshwater Fish Toxicity

Acute testing with the TGAI

In order to establish the toxicity of a pesticide to freshwater fish, the minimum data required on the technical grade of the active ingredient are two freshwater fish toxicity studies. One study should use a coldwater species (preferably the rainbow trout), and the other should use a warmwater species (preferably the bluegill sunfish).

Freshwater Fish Acute Oral Toxicity			
Species	% Test Material (TGAI)	LC ₅₀	Conclusions
Rainbow trout	96.2	3 ppm	moderately toxic
Bluegill sunfish	96.2	9.6 ppm	moderately toxic

The results of the 96-hour acute toxicity studies indicate that linuron can be characterized as being moderately toxic to both cold and warm water fish. (MRIDs 40445501 and 40354201).

Acute testing with the formulated product

Formulated product testing is specified if there is direct application to an aquatic environment or if EECs are greater than or equal to the LC50. Linuron is registered for use on rights-of-way (ROWs) which can result in a direct application to aquatic environments.

Freshwater Fish Acute Testing with the Formulated Product			
Species	% A.I.	Result LC50	Conclusions
Rainbow trout	Lorox 50 (WP)	16.4 ppm	slightly toxic
Bluegill sunfish	Lorox 50 (WP)	16.2 ppm	slightly toxic
Bluegill sunfish	Lorox 50 (DF)	9.2 ppm	moderately toxic

The results of the 96-hour EC50 studies indicate that Lorox 50 WP (wetttable powder) is slightly toxic to rainbow trout and bluegill sunfish. Lorox 50 DF (dry flowable) is considered moderately toxic to bluegill sunfish. (MRIDs 00018165, 00018165, and 00018198).

Chronic Test-Early Life Stage

The fish early life stage is required to support reregistration of a chemical if exposure is expected to be continuous, recurrent or persistent, and multiple applications of the chemical may occur. The minimum data required to establish chronic toxicity of linuron to fish is the early life stage toxicity test based on survival of fish embryos and post-hatch larvae. *There are no data available.*

Freshwater Invertebrate Toxicity

Acute testing with the TGAI

The minimum testing required to assess the hazard of a pesticide is a freshwater aquatic invertebrate toxicity test, preferably using first instar *Daphnia magna* or early instar amphipods, stoneflies, mayflies, or midges.

Freshwater Invertebrate Toxicity Findings			
Species	% Test Material (TGAI)	EC ₅₀	Conclusion
<i>Daphnia magna</i>	94.4	0.12 ppm	highly toxic

There is sufficient information to characterize linuron as highly toxic to aquatic invertebrates. (MRID 00142932).

Chronic Test-life cycle

The *Daphnia* Life Cycle is required to support reregistration if the chemical's presence in water is likely to be continuous, recurrent or persistent, and multiple applications of the chemical may occur. The minimum data required to establish chronic toxicity of linuron to invertebrates is the *Daphnia* life cycle test based on reproduction, growth and survival.

Chronic Test-Life Cycle		
Species	% A.I.	Results
<i>Daphnia magna</i>	98.4	MATC > 0.13 < 0.24 ppm

Based on the data submitted, the MATC is greater than 0.13 and less than 0.24 ppm. The Agency has chronic invertebrate data that appear inconsistent with acute data: chronic effects were not seen until levels higher than those causing acute effects. Also, invertebrates were more sensitive than fish in acute tests, but appear considerably less sensitive in the chronic test. Therefore, additional testing is required based on inconsistent results with the acute toxicity data. (MRID 42153401)

Estuarine/Marine Toxicity

Acute testing with the TGAI

Acute toxicity testing with estuarine and marine organisms is required when an end-use product is intended for direct application to the marine/estuarine environment or is expected to reach this environment in significant concentrations.

The requirements under this category include a 96-hour LC₅₀ for an estuarine fish, a 96-hour LC₅₀ for shrimp, and either a 48-hour embryo-larvae study or a 96-hour shell deposition study with oysters.

Estuarine/Marine Acute Toxicity Findings			
Species	% Test Material (TGAI)	LC₅₀	Conclusions
Sheepshead minnow	98.4	0.89 ppm	highly toxic
Eastern oyster	98.4	5.4 ppm	moderately toxic
Mysid shrimp	98.4	3.3 ppm	moderately toxic

There is sufficient information to characterize the TGAI of linuron as highly toxic to the sheepshead minnow and moderately toxic to the eastern oyster and mysid shrimp. (MRIDs 42061801, 42061802, and 42061803).

Acute testing with the formulated product

Marine and estuarine testing using the formulated products is required due to the ROW (Rights-of-way) use. ROWs could cross virtually any habitat, including marine aquatic habitat such as salt marshes. Data are not currently available. Testing is required with at least the most sensitive species in acute testing (sheepshead minnow) using the DF (dry flowable) formulation. A DF formulation was found to be more toxic than expected based on active ingredient testing. Because of the ROW (right-of-way) use, there could be direct exposure to the aquatic environment by the formulated product. TEP testing will enable the Agency to assess the risk of specific formulation(s) actually used on ROWs. Additional species and/or formulations may also be required.

Chronic effects

Chronic marine and estuarine testing are indicated based on the same criteria as freshwater species. In the case of linuron, these indications include (1) LC50 value less than 1 mg/l, (2) EEC 0.01 LC50 and (3) aquatic half-life of less than 4 days. Sheepshead minnow and mysid shrimp should be tested.

(3) Non-Target Plants Data

Toxicity to Terrestrial Plants

Data requirements for determining toxicity to terrestrial plants (Tier 2) remain outstanding. These data are required for linuron because it is an herbicide registered for use on terrestrial food and nonfood sites and the vapor pressure is 1.0×10^{-5} . Labeling indicate that aerial application can be used for soybeans, as well as ground boom spray for other crops. However, a plant risk assessment for linuron cannot be performed without the phytotoxicity data.

Toxicity to Aquatic Plants

Only one of the five required species for testing for toxicity to aquatic plants has been submitted. Testing for *Lemna gibba*, *Skeletonema costatum*, *Anabaena flos-aquae*, and a freshwater diatom remain outstanding. These data are required for linuron as it is an herbicide registered for use on terrestrial food/nonfood sites, has a vapor pressure 1.0×10^{-5} mm Hg, and a water solubility greater than 10 ppm. These data are required to conduct the plant risk assessment for linuron.

Aquatic Plant Toxicity		
Species	% A.I.	EC ₅₀
<i>Selenastrum capricornutum</i>	100	5-day = 0.067 mg ai/l

With a 5-day exposure of 0.067 mg active ingredient per liter of linuron, *S. capricornutum* can be expected to sustain a 50% reduction in density or number of cells. (MRID 42086801).

Appendix A. Ecological Effects Data for Captan

Table A.1. Freshwater Fish Data - Captan Parent						
Species	% A.I.	LC50, µg/L (confidence interval)	Measured/ Nominal Flow-through /static	Toxicity Classification	MRID (study year)	Class-ification
Brook Trout	88.4	34 (22 - 52)	Measured, Flow-through 8-day test	Very Highly toxic	00057846 (Hermanutz, 1973)	Supplemental
Fathead Minnow	88.4	65 (59 – 72)	Measured, Flow-through 6-day test	Very Highly toxic	00057846 (Hermanutz, 1973)	Supplemental
Bluegill sunfish	88.4	72 (47 – 111)	Measured, Flow-through 5-day	Very Highly toxic	00057846 (Hermanutz, 1973)	Supplemental
Coho Salmon	90	137 (117-160)	Static	Highly Toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Coho Salmon	90	56.5 (52.3-61)	Flow-through	Very Highly toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Chinook Salmon	90	120 (103-140)	Static	Highly Toxic	40098001	Supplemental
Cutthroat trout	90	56.4 (42.2-75.4)	Static	Very Highly toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Rainbow Trout	90	73.2 (66.6-80.4)	Static	Very Highly toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Brown Trout	90	80 (63.8– 100)	Static	Very Highly toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Brown Trout	90	26.2 (21.9-31.3)	Flow-through	Very Highly toxic	40098001	Supplemental
Lake Trout	90	49 (40.1-59.9)	Static	Very Highly toxic	(Johnson & Finley, 1980)* 40098001	Supplemental

Lake Trout	90	63.2 (49.6-80.5)	Static	Very Highly toxic	40098001	Supplemental
Lake Trout	90	51 (39.2-66.2)	Flow-through	Very Highly toxic	40098001	Supplemental
Fathead Minnow	90	200 (168-238)	Static	Highly Toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Fathead Minnow	90	134 (100-178)	Flow-through	Highly Toxic	40098001	Supplemental
Channel catfish		77.5 (70.5-85.2)	Static	Very Highly toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Bluegill sunfish	90	141 (119 – 167)	Static	Highly Toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Yellow Perch	90	120 (97.3-147)	Flow-through	Highly Toxic	(Johnson & Finley, 1980)* 40098001	Supplemental
Bluegill sunfish	90	310 (280 – 340) Slope = 1.17	Static	Highly Toxic	GS0120-042 (1979)	Supplemental
Harlequin Fish (Rasbora heteromorpha)	89	300	Static	Highly Toxic	00034713 Tooby et al. 1975	Supplemental (26 hr test, daily change of test water, no mortality data, test species)

* In Mayer and Ellersieck (MRID 40098001)

* Original source: Johnson, W. W., and M. T. Finley. 1980. Handbook of acute toxicity of chemicals to fish and aquatic invertebrates. U.S.F.W.S., Resource. Pub. 137.98 pp.

Table A.2 Freshwater Fish Data - Captan Degradates

Species	% A.I.	96-hr LC ₅₀ , µg/L (confidence interval)	Measured/ nominal Flow-through /static	Toxicity Classification	MRID (study year)	Satisfies Guideline/ Comments
Rainbow Trout	96% THPI	> 120,000	Measured, Static Renewal	Practically non-toxic	43869806	Acceptable
Rainbow Trout	96% THPAm	> 126,000	Measured, Static	Practically non-toxic	44738801	Supplemental (used 10 fish, but 30 are required for limit test)

A.3. Chronic Toxicity to Freshwater Fish

Species	Toxicity	Source	Effects
Fathead minnow	NOAEC = 16.5 µg/L LOAEC = 39.5 µg/L	MRID 00057846	Acceptable Reductions in adult and larval survival, growth and overall larval-juvenile development, survival of the juvenile species, a reduction in eggs laid, and an inability for juveniles to reproduce

Table A.4. Aquatic Invertebrate Captan Data

Species	% A.I.	Toxicity	Measured/ nominal Flow-through /static	Toxicity Classification	MRID (study year)	Classification
Daphnia magna	90	48-hr LC ₅₀ = 8400 (7060-9960) µg/L Slope= 1.187	Static	Moderately Toxic	GS0120041	Acceptable
Daphnia magna	96% THPI	48-hr LC ₅₀ >113,000 µg/L	Static	Practically non-toxic	438698-08	Acceptable
Daphnia magna	technical	NOAEC = 560 µg/L LOAEC = 1000 µg/L	Static	--	441488-01	Supplemental (based on nominal concentrations)

Table A.5. Aquatic Plant Captan Data					
Species	% A.I.	EC ₅₀ , µg/L (confidence interval)	Toxicity Classification	MRID (study year)	Classification
<i>Scenedesmus subspicatus</i> Green algae (96-hr)	92.7	320	Highly toxic	00137688	Supplemental (based on nominal concentrations)
<i>Selenastrum capricornutum</i> Green Algae (96 hr)	90	1770 (1550-2030)	Moderately Toxic	438698-09	Acceptable
<i>Anabaena flos-aquae</i> Freshwater Algae (96 hours)	99.8	1200 (830-1600)	Moderately Toxic	448065-01	Acceptable
<i>Lemna gibba</i> Duckweed (7 days)	99.8	> 12,700	Slightly Toxic	448065-03	Acceptable
<i>Selenastrum capricornutum</i> Green Algae (72 hours)	96% THPI	> 180,000	Practically non-toxic	438698-10	Supplemental (short test duration)

Note: *Skeletonema costatum* (marine diatom), *Isochrysis galbana*, *Pavlova gyrans*, *Pavlova lutheria*, and *Dunaliella tertiolecta* (marine algae) are marine species and not applicable to RLF assessment (MRID 40228401).

U.S. EPA. 1986. Acute Toxicity Handbook of Chemicals to Estuarine Organisms., *U.S.EPA, Gulf Breeze, FL* (US EPA MRID 40228401).

Table A. 6. Captan Bird Data						
Species	LD ₅₀ mg/kg bw	Acute Oral Toxicity (MRID)	LC ₅₀ (mg/kg diet)	Subacute Dietary Toxicity (MRID)	NOAEC mg/kg diet MRID	Affected Endpoints
Northern bobwhite Quail <i>Colinus virginianus</i>	> 2150	00151236 Beavers, 1978	> 2400	GS0120 Fiche/Master ID 00022923 Hill, 1975	1000 (00098295 Fink, 1980)	No affected endpoints
Mallard Duck <i>Anas platyrhynchos</i>	> 2000	GS999-001 Hudson, 1984	>5000	GS0120 Fiche/Master ID 00022923 Hill, 1975	1000 (00098296 Fink, 1980)	No affected endpoints

Table A.7. Mammalian Captan Data

Species	Test Type	LC ₅₀ (mg/kg diet)	NOAEL/ LOAEL (mg/kg diet)	Citation (MRID)	Comments
Rat	Acute Oral	> 5000	--	00265785 (1984)	Two males died. One death occurred on day 1 and one on day 12. One female died on day 4. The deaths were treatment related according to necropsy.
Rat	Acute Oral	Male: 5400 (4290-6800) Female: 5500 (4370-6930)	--	ACC# 241805	--
Rat	Acute Oral	9000	--	00054789 (1949)	--
Rat	One generation	--	> 500/ >500	00120315	
Rat	Three Generation	--	250 / 500	00125293 246101 241001	decreases in the mean litter weights of pups and severe sexual organ atrophy in adults and pups, signs of severe changes in liver weights in the adult males as well as abdominal and intestinal atrophy. In females, there were signs of stomach atrophy and esophageal atrophy

Table A. 8. Terrestrial Invertebrate Data				
Species	Test Type	LD ₅₀ (µg/kg bee)	Citation (MRID)	Comments
<i>Apis mellifera</i> Honeybee	Acute Contact	> 10	Fiche/Master ID 05001991 Stevenson, 1978	
<i>Apis mellifera</i> Honeybee	Acute Contact	> 215	Fiche/Master ID 00080871 Atkins, 1972	
<i>Osmia lignaria</i> Bee	72-hr Acute Oral	46.26 (32.75 – 77.44)	Ecotox # 87252 Ladurner et al, 2005	Captan 50WP 48.9% a.i.
	72-hr Acute Contact	269.68 (151.32 – 2841.84)		

Summary of Amphibian Larvae Study

Chemical Name: Captan

PC Code: 081301

ECOTOX Record Number and Citation: 90515. Mouchet, F., Gauthier, L., Mailhes, C., Ferrier, V, and Devaux, A. 2006. Comparative evaluation of genotoxicity of captan in amphibian larvae (*Xenopus laevis* and *Pleurodeles waltl*) using the comet assay and the micronucleus test. Environmental Toxicology 21(3): 264-277.

Purpose of Review: Litigation (California Red-Legged Frog)

Date of Review: October 2007

Brief Summary of Study Findings:

The toxic and genotoxic potentials of captan were evaluated with the micronucleus test (MNT) and the comet assay (CA).

Adult pairs of *Xenopus* and *Pleurodeles* were mated. Viable eggs were maintained until they reached a development stage appropriate for testing (3 weeks for *Xenopus* and 6 weeks for *Pleurodeles*). Experimental conditions generally followed the French Standard AFNOR (French National Organization for Quality Regulation) NF T90-325.

Amphibians were exposed to either reconstituted water (RW) to which nutritive salts were added or mineral water (MW). Nominal captan concentrations were: 2000, 1000, 500, 250, 125, 65.5, 31.25, and 15.60 µg/L. Actual concentrations in water were not measured. Negative controls were either RW or MW. Positive controls were benzo[a]pyrene (B[a]P, [50-32-8], purity: 96.0%, Sigma France) at 0.125 mg/L for MNT and methyl methanesulfonate (MMs, [66-27-3], purity: 99%, Sigma France) at 1.56 mg/L for CA. Captan was dissolved in DMSO at a final concentration of 0.05% before addition to water. Media in all flasks was renewed daily.

Acute toxicity was examined for 12 days by visual inspection (death, abnormal behavior, reduced size, diminished food intake. No signs of toxicity or mortality were observed in any of the negative controls (personal communication with F. Mouchet, October 2007).

Captan flasks containing RW became turbid between 12-24 hours after renewal. The study author hypothesized that this turbidity was probably caused by amphibian residues/excretion or by the suspended captan or the degradation products that may interact with mineral ions, which make up a larger proportion of RW than MW.

Results of acute toxicity to *Xenopus* and *Pleurodeles* larvae exposed to captan (µg/L) in mineral water (MW) and reconstituted water (RW) for 12 days

Conc(µg/L)		2000	1000	500	250	125	62.5	31.25	15.60
Xenopus	MW	++(100%)	++(100%)	++(100%)	++(100%)	++(55%)	-	-	-
	RW	++(100%)	++(100%)	++(100%)	+	-	-	-	-
Pleurodeles	MW	++(100%)	++(100%)	++(75%)	++(45%)	-	-	-	-
	RW	++(100%)	++(100%)	++(50%)	+	-	-	-	-

‰: percent dead (of 20 larvae); - No toxicity of larvae; + weak toxicity; ++ severe toxicity.

Genotoxicity was only assayed in MW at those concentrations where there was no acute toxicity. At 12 days for MNT and 1, 2, 4, 8, or 12 days for CA a blood sample was taken. Genotoxicity was assessed to the highest concentration that did not lead to signs of acute toxicity of the exposed larvae.

The results of the *Xenopus* MNT showed that a captan concentration of 62.50 µg/L induced a significant genotoxic response. The lowest concentrations (15.60 and 31.25 µg/L) were not genotoxic to *Xenopus* larvae. The results of the *Pleurodeles* MNT showed no genotoxicity regardless of the concentration of captan tested: 125, 62.50, 31.25, or 15.60 µg/L.

Results of the *Xenopus* CA showed that captan had genotoxic effects at all concentrations tested (15.60 µg/L after 8 and 12 days; 31.25 and 62.5 µg/L after 1, 2, 4, and 8 days; and 125 µg/L after 1, 2, and 4 days). The results of the *Pleurodeles* CA showed genotoxic effects at captan concentrations of 62.5 and 125 µg/L after 1 and 2 days of exposure, whatever the parameter, except with tail DNA after 2 days of exposure to 62.5 µg/L.

LC₅₀ and slope (when possible) was estimated by the reviewer using TOXANAL software.

RESULTS CALCULATED USING THE PROBIT METHOD

ITERATIONS G H GOODNESS OF FIT PROBABILITY
 16 .1202498 1 .8025137
 SLOPE = 4.58749
 95 PERCENT CONFIDENCE LIMITS = 2.996684 AND 6.178296
 LC50 = 311.0651
 95 PERCENT CONFIDENCE LIMITS = 253.3895 AND 381.8606

		LD50, (µg/L) (confid int)	Method
Xenopus	MW	119.4 (62.5, 250)	Binomial
	RW	353.6 (250, 500)	Binomial
Pleurodeles	MW	311.1 (253.4, 381.9)	Probit
	RW	500 (250, 1000)	binomial

Description of Use in Document: Qualitative

Rationale for Use: This is the only known study evaluating the toxicity of captan to amphibians.

Limitations of Study:

1. Detailed raw data not available.
2. LC₅₀ and slope (using probit model) not estimable for 3 survival curves as there was only one concentration with partial mortality.
3. Captan concentrations not measured.
4. Turbidity in RW flasks containing captan not definitively explained.

Reviewers: Christine Hartless, Wildlife Biologist (ERB1)

Summary of Wheat Study

Chemical Name: Captan

PC Code: 081301

ECOTOX Record Number and Citation: 91168. Mantecon, J. D. (1989). Persistence of Systemic and Non-Systemic Fungicides in the Control of Seedling Blight of Wheat (*Fusarium graminearum*). *Tests Agrochem.Cultiv.* 10: 76-77.

Purpose of Review: Litigation (California Red-Legged Frog)

Date of Review: September 2007

Brief Summary of Study Findings:

This study was conducted in a greenhouse at the Experiment Station INTA Balcarce, Buenos Aires Province, Argentina. Highly infected seeds of a durum wheat (*Triticum durum* Desf.) cv. Buck Patacon were sown in an artificially infested soil. The treatments were arranged in a randomized complete block design with three replications of 100 seeds each. Greenhouse temperatures averaged 20 ± 5 C. Fungicides were applied one day before sowing the seed. Four fungicides (including captan) were evaluated. Captan was applied at a rate of 120 g ai/kg-seed (0.26 lbs ai/cwt) using a wettable powder product from Stauffer Chemicals that was 83% ai.

The measured response variable was number of seedlings present after 7, 14, 21, and 28 days. Data were analyzed for each day by ANOVA and Tukey's test. Only the results from captan and the control are included in the table below.

	Time (days) after sowing			
	7	14	21	28
Captan	67	89	79	69
Control	39	43	28	26

At each time point, there was a statistically significant difference between the mean number of seedlings in the captan and the control groups.

Description of Use in Document: Qualitative

Rationale for Use: One of several seed treatment studies used in lieu of seedling emergence studies.

Limitations of Study:

1. Detailed raw data not available.
2. Only one treatment level evaluated (EC_{25} cannot be determined).
3. Exposure is by seed treatment, rather than by spray on top of soil surface.
4. Watering regime not available.

Reviewers: Christine Hartless, Wildlife Biologist (ERB1)

Summary of Sorghum Study # 91004

Chemical Name: Captan

PC Code: 081301

ECOTOX Record Number and Citation: 91004. Mc Laren, N. W. and Rijkenberg, F. H. J. (1989). Efficacy of Fungicide Seed Dressings in the Control of Pre- and Post-Emergence Damping-Off and Seedling Blight of Sorghum. *S.Afr.J.Plant Soil* 6 : 167-170.

Purpose of Review : Litigation (California Red-Legged Frog)

Date of Review: September 2007

Brief Summary of Study Findings:

This field study was conducted in Potchefstroom, Republic of South Africa in a field in which seedling diseases had been previously recorded. The seed cultivars DC34, DC99, NK283, and PNR8311 were used. A randomized split plot design with five replications was used. Cultivar was the whole plot factor and seed treatment was the sub-plot factor. Captan was applied as a seed dressing at a rate of 135 mg ai/kg-seed (0.30 lbs ai/cwt). Each subplot consisted of three rows, 11 m in length, spaced 1 m apart. After application of 2:3:2 fertilizer (300 kg/ha) seeds were planted to a depth of ± 5 cm and spaced 15 cm apart. A total of 17 fungicide treatments was used, only captan and control results are reported here.

To facilitate recovery of seed from the soil for determination of germination and pre-emergence damping-off, samples of 20 seeds were planted in cocoons, 30 cm in length, folded from single ply cheesecloth. Two cocoons with the relevant seed treatment and cultivar were randomly placed in each subplot row. Cocoons were recovered after 7 days and the percentage germination and pre-emergence damping-off were assessed. Pre-emergence damping-off was measured as the percentage germinated seeds in which rotting was so severe that growth had ceased prior to emergence of seedlings from soil.

Twenty-one days after planting the percentage post-emergence damping-off (as a percentage of emerged seedlings) was determined in each sub-plot. Thereafter, 25 seedlings were removed from each sub-plot row and washed to remove adhering soil particles. Visual assessments of the percentage mesocotyl and primary root discoloration were made. Seedlings were also dried and weighed.

For each of the measured parameters, there were no statistically significant differences between the captan and the control group.

	Pre-emergence damping off (%)	Mesocotyl discoloration (%)	Root discoloration (%)	Post-emergence damping off (%)	Seedling mass (g)
Captan	19.4	58.2	19.8	10.5	3.8
Control	18.0	64.9	21.4	11.3	3.5
Least Significant Difference LSD (0.05)	8.2	16.4	7.5	2.3	0.5

Description of Use in Document: Qualitative

Rationale for Use: One of several seed treatment studies used in lieu of seedling emergence studies.

Limitations of Study:

1. Detailed raw data not available.
2. Only one treatment level evaluated (EC₂₅ cannot be determined).
3. Exposure is by seed treatment, rather than by spray on top of soil surface.
4. Watering regime not available.

Reviewers: Christine Hartless, Wildlife Biologist (ERB1)

Summary of Sorghum Study # 90836

Chemical Name: Captan

PC Code: 081301

ECOTOX Record Number and Citation: 90836. Davis, M. A. and Bockus, W. W. (2001). Evidence for a *Pythium* sp. as a Chronic Yield Reducer in a Continuous Grain Sorghum Field. *Plant Dis.* 85: 780-784.

Purpose of Review (DP Barcode or Litigation): Litigation (California Red-Legged Frog)

Date of Review: September 2007

Brief Summary of Study Findings:

Field experiments

Two field experiments were conducted (planting dates of 11 May 1995 and 7 June 1995) in which there were three treatment groups (control, captan, and metalaxyl). A high vigor commercial hybrid seed (germination rate > 90%, Cargill 618Y) was planted. Prior to planting, seed was treated. A glass canning jar (1 liter) was “seasoned” by adding 2.5 ml water, the correct amount of chemical, and 100 g of seed. Jar was shaken until all liquid was absorbed by seed. This seed was discarded; procedure was repeated to produce treated seed for experiments. Treated seed was placed in paper bags to dry before sowing. Captan 400D at 3.0 fl oz/cwt (73g ai/kg-seed or 0.16 lbs ai/cwt) was used. Stand counts (plants/m²) were taken on 12 and 23 June, vigor ratings (scale of 1 to 5) were taken on 3 July (boot and growing point differentiation growth stages), and grain yields (kg/acre) were measured on 17 and 20 October. There was a statistically significant increase or no difference in the captan treated seed responses relative to the control seeds in all measured parameters for both experiments.

		11 May 1995	7 June 1995
Stand			
	Control	3.8 c	6.0 b
	captan	7.3 a	7.9 a
	Metalaxyl	6.3 b	7.7 a
Vigor			
	Control	2.3 b	4.2 a
	captan	2.9 a	4.0 a
	Metalaxyl	3.1 a	3.6 a
Grain yield			
	Control	2592 b	5651 b
	captan	2754 b	6302 ab
	Metalaxyl	3947 a	6742 a

Values within a column and parameter followed by a common letter are not significantly different according to analysis of variance followed by least significant difference ($P = 0.05$).

Greenhouse experiment

Seed (Cargill 618Y) was treated or not treated with captan or metalaxyl at 0.16 lbs ai/cwt using the same method as described above. The experiment was arranged in a randomized complete block design using 10 plastic tubes 2.5 cm in diameter by 15 cm long. Each treatment had four replications. Soil was collected from the field experiment site above and left nontreated or autoclaved at 121C for 2 hrs and placed in the tubes. One seed was sown per tube and plants were maintained in a greenhouse at 15-27 C. Plant counts (out of 10 seeds planted) and shoot fresh weight per plant were recorded after 28 days. There was a statistically significant increase or no difference in the captan treated seed responses relative to the control seeds in all measured parameters for either naturally infested soil or autoclaved soil.

		Experiment 1		Experiment 2	
Seed trt	Soil trt	Stand	Fresh shoot wt	Stand	Fresh shoot wt
Nontreated	Autoclaved	5.8 a	0.83 a	7.8 a	1.03 a
Captan	Autoclaved	7.5 a	0.85 a	8.0 a	1.18 a
Metalaxyl	Autoclaved	7.0 a	0.87 a	8.0 a	1.15 a
Nontreated	Nonautoclaved	3.0 b	0.57 b	5.5 b	0.52 b
Captan	Nonautoclaved	7.0 a	0.60 b	7.8 a	0.69 b
Metalaxyl	Nonautoclaved	7.8 a	0.90 a	8.0 a	1.07 a

Values within a column followed by a common letter are not significantly different according to analysis of variance followed by least significant difference ($P = 0.05$).

Description of Use in Document: Qualitative

Rationale for Use: One of several seed treatment studies used in lieu of seedling emergence studies.

Limitations of Study:

1. Detailed raw data not available.
2. Only one treatment level evaluated (EC_{25} cannot be determined).
3. Exposure is by seed treatment, rather than by spray on top of soil surface.

Reviewers: Christine Hartless, Wildlife Biologist (ERB1)

Summary of Lupine Study

Chemical Name: Captan

PC Code: 081301

ECOTOX Record Number and Citation: 91007. Fahim, M. M., Osman, A. R., Sahab, A. F., and El-Kader, M. M. A. (1983). Agricultural Practices and Fungicide Treatments for the Control of Fusarium Wilt of Lupine. *Egypt.J.Phytopathol.* 15: 35-46.

Purpose of Review (DP Barcode or Litigation): Litigation (California Red-Legged Frog)

Date of Review: September 2007

Brief Summary of Study Findings:

In vivo experiments were carried out in unsterilized 25-cm diameter clay pots containing clay sand mixture (1:1, w/w), referred to as loamy soil. The seeds were treated with the tested fungicides by shaking them in polyethylene bags until an even dressing was observed. Captan was applied at 0.50 lbs ai/cwt as the enduse product Orthocide (75% captan, recommended rate of 3 g Orthocide/kg-seed). Each treatment had five replicates. A total of eight fungicides and the control were evaluated in the experiment; only the captan results are summarized below.

Soil infestation was conducted by mixing cultures of *Fusarium oxysporum* with the soil at a rate of 5%, w/w. The inoculum was a 2-week-old growth of a virulent isolate, obtained from Alquam, Giza Governorate, on barley/sand (3:1, w/w) medium at 30 C. visual observations were made during the growth season. Macroscopic checks were also carried out at maturity. Seeds were air-dried for several days.

At the end of growing season, average weight of 100 seeds in the treated group was the same or greater than in the control. Percent occurrence of diseased plants was less in treated group than in control group.

		Diseased plants, %			Avg wt of 100 seeds, g.
		Pre-emergence	Post-emergence	total	
Captan	Infested	0	10	10	19.0
	Uninfested	0	7.5	8	20.2
Control	Infested	12.5	68.9	73	14.7
	Uninfested	7.5	19.1	25	16.7
Least Significant Difference (LSD) at P=0.05					
Main effect of fungicide		3.5	6.2	-	3.3
Main effect of infestation		1.6	2.9	-	1.5
Interaction (fungicide x infestation)		4.9	8.6	-	4.7

Description of Use in Document: Qualitative

Rationale for Use: One of several seed treatment studies used in lieu of seedling emergence studies.

Limitations of Study:

1. Detailed raw data not available.
2. Only one treatment level evaluated (EC₂₅ cannot be determined).
3. Exposure is by seed treatment, rather than by spray on top of soil surface.
4. Rainfall/watering regime not available.

Reviewers: Christine Hartless, Wildlife Biologist (ERB1)

Summary of Blueberry Study

Chemical Name: Captan

PC Code: 081301

ECOTOX Record Number and Citation: 63909. Polavarapu, S. (2000). Evaluation of Phytotoxicity of Diazinon and Captan Formulations on Highbush Blueberries. *Horttechnology* 10: 308-314.

Purpose of Review (DP Barcode or Litigation): Litigation (California Red-Legged Frog)

Date of Review: September 2007

Brief Summary of Study Findings:

Experiments were conducted during the 1997 and 1998 growing seasons at Rutgers University Blueberry and Cranberry Research and Extension Center, Chatsworth, NJ, on highbush blueberries planted in 1994. Bushes were 4-5 yrs old, approx 5 ft tall, and spaced 9 x 4 ft apart on light sandy organic matter soil with pH of 4.5. Two formulations of diazinon (Diazinon AG600 and Diazanon 50W) and of captan (Captec 4L and Captan 80WP) as well as an adjuvant, LI-700 were evaluated. Results pertaining to the adjuvant will not be reported here. All experiments described below had a negative control group. Application rates (author stated maximum labeled rates were used) are below:

formulation	Rate/acre	lbs ai/acre
Diazinon AG600	22.5 fl oz	NA
Diazanon 50W	2 lb	NA
Captec 4L	3.12 lb	2.43 lbs ai/acre
Captan 80WP	2.5 qt	2.5 lbs ai/acre

NA – not applicable, reviewer did not calculate as only captan is under review in this summary.

Treatments were arranged in a randomized complete block design. Treatments within a block were separated by at least 4 bushes and blocks were arranged 50-133 ft apart.

Pesticides were applied with a CO₂ pressurized backpack sprayer equipped with a hollowcone nozzle calibrated to deliver 30 gal/acre. At each evaluation, samples of foliage and fruit were collected in polyethylene bags and transported to lab for phytotoxicity evaluations. A fruit or foliage cluster was determined to have phytotoxicity even if only one fruit or leaf was injured.

Phytotoxicity injury occurred within 24 to 36 hrs after application of pesticides. Phytotoxicity on berries ranged from deep purple blotches to circular depressions, especially where residues accumulated. In the most severe cases, fruit had 2 to 3 mm diameter circular depressions filled with apparent pesticide residue. Phytotoxicity on leaves was typically brownish purple spots on the underside of the leaf surface. The

degree of phytotoxicity severity caused by the mixtures of captan and diazinon was much greater than the phytotoxicity when captan or diazinon was applied alone.

Data were analyzed using ANOVA and Duncan's multiple range test ($P=0.05$). Data were transformed before analysis using square root (number of clusters with phytotoxicity, number of berries, and berry weight) or arcsin (percent phytotoxicity) transformations.

Experiment 1

- Conducted in 1997, treated on 11 June 1997
- 5 single bush reps per treatment, variety Ellicot
- single treatment was Diazinon AG600 and Captec 4L
- 5 fruit and 5 foliage clusters collected from each side of each bush - 10 days after treatment
- the combined treatment had a significantly greater proportion of berries exhibiting phytotoxicity and lighter weight berries; although, the number of berries per 10 clusters was not different than the control.

treatment	Berries with phytotoxicity (%)		Number of berries/10 clusters	Wt of 100 berries (g)
	green	Blue		
Diazinon AG600 + Captec 4L	99.6 \pm 0.4 a	97.7 \pm 1.7 a	103 \pm 6.6 a	108.4 \pm 7.2 a
Untreated	0.0 \pm 0 b	1.5 \pm 1.0 b	99.2 \pm 5 a	145.5 \pm 11.2 b

For each response variable, treatment means followed by different letters are significantly different at $P=0.05$.

Listed response is mean \pm standard error

Experiment 2

- Conducted in 1997, treated on 12 June 1997
- Three replications, each consisting of 6 bushes in a row, variety Bluecrop
- Treatments were combinations of the 4 listed pesticides.
- First evaluation 7 days after trt, 10 clusters from 3 randomly selected bushes within each replication
- Second evaluation 13 July with 25 fruit clusters per rep (during harvest)
- For all responses, the single pesticide applications were not significantly different from the control. Responses with no significant differences (means not listed in summary, are available in paper) were number of berries per 30 clusters 7 days after treatment, blue berries with phytotoxicity/25 clusters (%) at harvest, all berries with phytotoxicity/25 clusters (%) at harvest, and number of berries/25 clusters at harvest.

	Clusters with phytotoxicity (no/30 clusters) 7 d after treatment		Green berries with phytotoxicity/25 clusters (%) at harvest
	fruit	leaf	
Diazinon AG600	0.0±0 c	0.0±0 d	0.7±0.7 b
Captec 4L	0.0±0 c	0.3±0.3 d	1.9±0.5 ab
Diazinon AG600 + Captec 4L	9.3±1.9 a	22.7±1.2 a	5.2±1.4 a
Diazinon AG600 + Captan 80WP	7.7±1.3 a	16.0±2.0 b	1.1±0.5 b
Diazinon 50W + Captec 4L	3.0±0 b	3.0±1.0 c	0.4±0.4 b
control	0.0 ±0 c	0.0±0 d	0.6±0.6 b

For each response variable, treatment means followed by different letters are significantly different at P=0.05, Duncan's multiple range test.

Listed response is mean ± standard error

Experiment 3

- Conducted in 1997, treatment applied on 25 June 1997
- Three replications, each consisting of three bushes, variety Ellicott
- Treatments were combinations of the 4 listed pesticides.
- 20 fruit and leaf clusters per rep (10 each from two randomly selected bushes) sampled 8 d after treatment.
- In addition to responses reported below, percent phytotoxicity/20 clusters was also analyzed, results were similar to the number of clusters (reported below). There were no significant differences in the number of berries per 20 clusters among treatments.
- Relative to control, captan alone or with diazinon resulted in no significant change or an increase in the observed phytotoxicity in fruit and leaves.

	Clusters with phytotoxicity (no/20 clusters) 8 d after treatment	
	fruit	leaf
Diazinon AG600	0.0±0 c	4.0±1.5 c
Diazinon 50W	0.0±0 c	0.3±0.3 d
Captec 4L	0.3±0.3 bc	14.0±1.5 b
Captan 80WP	0.7±0.3 bc	0.3±0.3 d
Diazinon AG600 + Captec 4L	4.0±2.5 a	20.0±0 a
Diazinon AG600 + Captan 80WP	1.0±1.0 bc	15.3±0.9 b
Diazinon 50W + Captec 4L	1.0±0.6 bc	4.7±1.7 c
control	0.0±0 c	0.0±0 e

For each response variable, treatment means followed by different letters are significantly different at P=0.05, Duncan's multiple range test.

Listed response is mean ± standard error

Experiment 4

- Conducted in 1998, treatments applied on 18 May. For some trts, diazinon applied first, followed by captan 8 hrs later.
- 4 reps, each consisted of 6 bushes in a single row, variety Weymouth

- 30 fruit and leaf clusters sampled from each rep 9 days after trt.
- In addition to responses reported below, percent phytotoxicity/30 clusters was also analyzed, results were similar to the number of clusters (reported below).
- Relative to control, captan alone or with diazinon resulted in no significant change or an increase in the observed phytotoxicity in fruit and leaves. Applying captan 8 hrs after diazinon did demonstrate a significant reduction in phytotoxicity relative to applying both simultaneously.

	Clusters with phytotoxicity (no/30 clusters) 9 d after treatment	
	fruit	leaf
Diazinon AG600	0.0±0 e	0.0±0 c
Diazinon 50W	0.3±0.3 de	0.3±0.3 c
Captec 4L	1.8±0.5 c	8.0±1.5 b
Diazinon AG600 + Captec 4L	25.8±2.5 a	14.5±2.0 a
Diazinon 50W + Captec 4L	13.5±0.6 b	7.2±0.9 b
Diazinon AG600 first + Captec 4L 8 hrs later	2.8±1.1 c	7.0±1.1 b
Diazinon 50W first + Captec 4L 8 hrs later	1.5±0.6 cd	5.0±1.5 b
control	0.0±0 e	0.0±0.0 c

For each response variable, treatment means followed by different letters are significantly different at P=0.05, Duncan's multiple range test.

Listed response is mean ± standard error

Experiment 5

- Conducted in 1998, treatments applied on 26 May. For some trts, chemicals were applied with an 8 h interval between them.
- 4 reps, each consisted of 6 bushes in a single row, variety Bluecrop
- 30 fruit and leaf clusters sampled from each rep 8 days after trt.
- Relative to control, captan alone resulted in no significant change or an increase in the observed phytotoxicity in fruit and leaves. Using an 8 hr interval between pesticide applications (with either captan or diazinon first) resulted in a significant reduction in phytotoxicity relative to applying both simultaneously.

	Phytotoxicity /30 clusters (%) 8 d after treatment	
	fruit	Leaf
Captan 80WP	0.2±0.2 b	1.4±0.5 b
Diazinon AG600 + Captan 80WP	18.1±3.1 a	9.0±1.3 a
Captec 4L first + Diazinon AG600 8 hrs later	2.2±1.8 b	3.5±1.1 b
Captan 80WP first + Diazinon AG600 8 hrs later	1.5±1.0 b	1.8±0.8 b
Captan 80WP first + Diazinon 50W 8 hrs later	0.1±0.1 b	2.3±1.3 b
control	0.0±1 b	0.0±0 c

For each response variable, treatment means followed by different letters are significantly different at P=0.05, Duncan's multiple range test.

Listed response is mean ± standard error

Experiment 6

- Evaluated effect of repeated applications of captan and diazinon applied together
- Conducted in 1998, treatments applied on 22 May, 26 June, 29 July.
- Variety Ellicott was used

- Samples collected 5 to 8 days after treatment.
- Only one treatment (Diazinon AG600 + Captec 4L, applied at same time) that caused most severe phytotoxicity plus control were used.
- Statistical analysis indicated a time*treatment interaction – a greater percentage of fruit and leaves showed phytotoxicity after 22 May application (immediately following petal fall) than after the other two application dates.

Description of Use in Document: Qualitative

Rationale for Use: Foliar spray study used in lieu of vegetative vigor studies.

Limitations of Study:

1. Detailed raw data not available.
2. Only one treatment level evaluated (EC₂₅ cannot be determined).
4. Watering regime not available.
5. Impact on growth of plants not measured.
6. Plants were established, not young seedlings.

Reviewers: Christine Hartless, Wildlife Biologist (ERB1)

Summary of Bee Study

Chemical Name: Captan

PC Code: 081301

ECOTOX Record Number and Citation: 87252. Ladurner, E., Bosch, J., Kemp, W. P., and Maini, S. (2005). Assessing Delayed and Acute Toxicity of Five Formulated Fungicides to *Osmia lignaria* Say and *Apis mellifera*. *Apidologie* 36: 449-460.

Purpose of Review (DP Barcode or Litigation): Litigation (California Red-Legged Frog)

Date of Review: October 2007

Brief Summary of Study Findings:

Contact and oral toxicity of five formulated pesticides were evaluated in this study. Only the results for captan (Captan 50WP, 49% ai) will be reported here.

In May 2002, wintering *O. lignaria* females, reared at the Bee Biology and Systematics Laboratory, Logan, Utah, were incubated at 25 C until emergence from cocoons. Unfed females were transferred to a screened flight cage to allow them to deposit meconium. Females were then starved overnight and exposed to a specific fungicide treatment the next morning, approximately 24 h after emergence. In June 2002, *A. mellifera* foragers of different ages from a healthy, queen-right colony were captured in a clear plastic jar as they left the hive in the morning. All bees were chilled for a maximum of 30 minutes at 4 C prior to treatment.

In the contact toxicity tests, 1 µL of test solution was applied to the dorsal surface of the thorax with a 50 µL-micro syringe. Test solution was prepared by dissolving fungicide in acetone and purified distilled water (50% v/v) to obtain desired concentrations; fresh test solution was used for all tests.

In the oral toxicity tests known amounts of the fungicide were dissolved in a feeding solution (25% v/v sucrose in purified distilled water) to obtain desired concentrations. *O. lignaria* and *A. mellifera* were fed 10 µL of the test solution using the flower method devised by Ladurner et al (2003). The test solution was pipetted into a plastic ampoule and inserted into the calyx of a flower (cherry for *O. lignaria* and morning glory for *A. mellifera*). Flowers and bees were individually housed in holding cages (waxed cardboard cups, 8 cm diameter x 5 cm height) with a wire mesh screen lid. Flowers and bees in holding cages were kept in an incubator (22 C for *O. lignaria* and 25 C for *A. mellifera*) under artificial light (two 15W Cool White fluorescent tubes 15cm above holding cages) for one hour.

For the contact test, control bees were dosed with the mixture of acetone and purified distilled water (50% v/v). For the oral test, control bees were fed the feeding solution (25% v/v sucrose in purified distilled water).

TEST 1

Three sets of ten bees each were evaluated for delayed toxicity in the form of a single dose (122.5 µg ai/bee) for both oral and contact tests. After exposure, each set of 10 bees was transferred to a holding cage (same as described for oral test) with an artificial feeder. The feeder was a 5 mL-LDPE sample vial containing a sucrose solution (25% v/v sucrose in water) with a soaked cigarette filter inserted through the end of the vial. Fresh solution was provided every 24 hrs. Holding cages for *A. mellifera* were also provided with a piece of wax foundation comb. Holding cages were kept in an incubator (*O. lignaria* – temperature=22 C, relative humidity=60-80%, L:D=12:12hr; *A. mellifera* – temperature=25 C, relative humidity=60-80%, L:D=0:24 hr). Survival was recorded every 24 hrs for 7 days.

In oral exposure trials, 97.7% of *A. mellifera* and 88.2% of *A. mellifera* consumed all the test solution in one hour. Control survival was 100% in the *O. lignaria* studies and was 75-80% in the *A. mellifera* studies. Captan resulted minimal mortality for *A. mellifera* and higher mortality rates for *O. lignaria*.

For *A. mellifera*, survival was not significantly reduced relative to control at the end of 7 days (Wilcoxon test) in either the oral or contact tests. For *O. lignaria*, survival was significantly reduced relative to control at the end of 7 days: in the contact test, survival was approximately 50%; and in the oral test, survival was approximately 35% on day 1 and approximately 0% by day 3.

TEST 2 – Methods of administration and bee maintenance were the same as described above. Only *O. lignaria* bees were used for captan, as there was minimal mortality for *A. mellifera* in the first test.

This test was designed to provide an estimate of an LD50. Five doses were administered; however, the test concentrations were not provided. Probit analysis was used for LD50 estimation.

	24 hr	48 hr	72 hr	7 days
Contact	NA	NA	269.68 (151.32, 2841.84)	95.26 (79.83, 134.59)
oral	NA	100.45 (63.75, 245.23)	46.26 (32.75, 77.44)	10.87 (5.40, 19.28)

Units are in µg ai/bee

NA – not available (LD50 was > than highest dose)

95% confidence interval in parentheses

Reference:

Ladurner, E., Bosch, J., Maini, S., Kemp, W.P. 2003. A method to feed individual bees (Hymenoptera: Apiformes) known amounts of pesticides. *Apidologie* 34: 597-602.

Description of Use in Document: Quantitative

Rationale for Use: This study provides a definitive toxicity endpoint for bees.

Limitations of Study:

1. Detailed raw data not available.
2. Dose concentrations not provided for second test.

Reviewers: Christine Hartless, Wildlife Biologist (ERB1)

Appendix B. Chlorothalonil Ecological Effects Characterization

B.1 Toxicity to Birds / Reptiles	2
B.2 Toxicity to Mammals	6
B.3 Toxicity to Non-Target Terrestrial Invertebrates	9
B.4 Toxicity to Freshwater Animals	10
B.5 Toxicity to Non-target Terrestrial Plants	17
B.6 References	20

This appendix presents additional details on available registrant-submitted and open literature studies available on chlorothalonil and its major degradate of toxicological concern, SDS-3701. Studies submitted to the Agency in support of pesticide registration or re-registration are categorized as either; acceptable, supplemental, or invalid. Acceptable means that all essential information was reported, the data are scientifically valid, and the study was performed according to recommended protocols. Studies in the “acceptable” category fulfill the corresponding data requirement in 40 CFR Part 158 and are appropriate for use in risk assessment. Supplemental studies are also scientifically valid; however, they were either performed under conditions that deviate from recommended guideline protocols or certain data necessary for complete verification are missing. Supplemental studies may be used quantitatively in the risk assessment and can, at the Agency’s discretion, fulfill the corresponding data requirement in 40 CFR Part 158. Invalid studies are not scientifically valid, or deviate substantially from recommended protocols such that they are not useful for risk assessment. Invalid studies do not fulfill the corresponding data requirement in 40 CFR Part 158.

With respect to the open literature, studies may be classified as either; qualitative, quantitative, or invalid. The degree to which open literature data are quantitatively or qualitatively characterized is dependent on whether the information is directly correlated with the assessment endpoints (i.e., maintenance of the survival, reproduction, and growth of the California red-legged frog and PCEs of their designated critical habitat identified in the problem formulation). Open literature studies classified as qualitative are not appropriate for quantitative use but are of good quality, address issues of concern to the risk assessment, and, when appropriate, are discussed qualitatively in the risk characterization discussion. Those open literature studies that are classified as quantitative are appropriate for quantitative use in the risk assessment including calculation of RQs. This appendix includes registrant-submitted studies in addition to studies identified in the open literature. In general, effects data in the open literature that are more conservative than the registrant-submitted data or that add to the weight of evidence on the toxicity to under-represented species or taxa are considered for quantitative use. Open literature studies that were either rejected by ECOTOX or that were not considered in this assessment are in Appendix H. Those appendices also include rationale for rejection of studies that did not pass the ECOTOX screen and those that were not evaluated as part of this endangered species assessment. Further detail on the ECOTOX exclusion categories is provided in the Agency’s *Guidance of the*

Evaluation Criteria for Ecological Toxicity Data in the Open Literature (U.S. EPA, 2004).

B.1 Toxicity to Birds, Reptiles, and Terrestrial Phase Amphibians

No studies in reptiles or terrestrial phase amphibians have been submitted to the Agency or were located in the open literature. Therefore, birds were used as surrogates for terrestrial phase amphibians. Acute oral, subacute dietary, and chronic reproduction toxicity studies for birds are discussed in Sections B.1.1 through B.1.3. All avian studies described below are registrant-submitted studies. No additional information on the acute, subacute, and/or chronic toxicity of chlorothalonil or SDS-3701 to birds was located in the open literature that suggests greater sensitivity than the registrant submitted data. In addition, no relevant information on the toxicity of chlorothalonil and/or SDS-3701 to reptiles and/or terrestrial-phase amphibians was located in the open literature.

B.1.1 Birds: Acute and Subacute Studies

Chlorothalonil

An acute oral toxicity study using the technical grade of the active ingredient (TGAI) is required to establish the toxicity of chlorothalonil to birds. The preferred test species is either mallard duck (*Anas platyrhynchos*; a waterfowl) or bobwhite quail (*Colinus virginianus*; an upland gamebird). Results of these studies are summarized below in Table B.1-1 and B.1-2. These studies suggest that chlorothalonil is practically non-toxic to birds on an acute basis.

Table B.1-1. Avian Acute Oral Toxicity Findings for Chlorothalonil					
Test Species	% a.i.	LD₅₀ mg/kg	Citation (MRID)	Toxicity Category	Fulfills Guideline?
Mallard	96%	> 4640	00068753	Practically non-toxic	Yes
Japanese quail	Tech.	> approx. 2000	40964105	Practically non-toxic	Supplemental

Table B.1-2. Avian Subacute Dietary Toxicity Findings for Chlorothalonil					
Test Species	% a.i.	LC ₅₀ ppm	Citation (MRID)	Toxicity Category	Fulfills Guideline?
Northern Bobwhite	96%	> 10,000	00030388	Practically non-toxic	Yes
Mallard	93.6%	> 21,500	00039146	Practically non-toxic	Yes
Mallard	96%	> 10,000	00030389	Practically non-toxic	Yes

SDS-3701

Acute avian LD₅₀ data for the chlorothalonil degradate, SDS-3701, are summarized in Table B.1-3. These studies show that SDS-3701 is "moderately toxic" on an acute oral basis and "slightly toxic" on a dietary basis to the test birds on an acute basis. Sublethal effects were seen in some birds at the lowest test level with the mallard, including lethargy, depression, lost reaction to stimuli, lost coordination, and wing droop.

In an acute oral study (MRID 00030395), fourteen-day old mallard ducks were dosed with 46, 100, 215, 464, and 1000 mg/kg of technical SDS-3701. No deaths occurred in the control pens. The acute oral LD₅₀ for SDS-3701 was 158 mg/kg, with 95% confidence limits of 125 to 201 mg/kg. The highest dose at which no deaths occurred was 46 mg/kg. All the birds died in the two highest dosing groups.

Table B.1-3. Avian Acute Oral and Subacute Dietary Toxicity Findings for SDS-3701					
Test Species	% a.i.	Results	Citation (MRID #)	Toxicity Category	Fulfills Guideline?
Mallard	SDS-3701 (87%)	LD ₅₀ = 158 mg/kg	00030395	Moderately toxic	Yes
Northern Bobwhite	SDS-3701 (87%)	LC ₅₀ = 1746mg/kg NOEL = 562 mg/kg	00115109	Slightly toxic	Yes
Mallard	SDS-3701 (87%)	LC ₅₀ = 2000 mg/kg	00115108	Slightly toxic	Yes

B.1.3 Birds: Reproduction Studies

Chlorothalonil

Avian reproduction studies using the TGAI are required because chlorothalonil is persistent (i.e., half-life exceeds 4 days in aerobic soils) and has multiple applications per growing season. The preferred test species are mallard duck and bobwhite quail. Results of these tests are summarized in Table B.1-4.

Table B.1-4. Avian Reproduction Findings of Chlorothalonil Exposure						
Test Species	% a.i.	NOEL PPM	LOEL PPM	Endpoints affected	Citation (MRID #)	Fulfills Guideline ?
Bobwhite	Tech.	1000 (reprod.)	5000 (reprod.)	"Overt signs of toxicity and reduced reproduction" cited at 5000 ppm; "overt signs of toxicity, mortalities, and profound effects upon several reproductive parameters related to egg production, hatching success, and survival of hatchlings" cited at 10,000 ppm.	40964104	Yes
Bobwhite	Tech.	153	624	18% reduction in no. of eggs laid per hen.	45710218	Yes
Bobwhite	99.6	50 ppm	Not established. Highest level of 50 ppm did not cause impairment.	None	00041440	Supplemental
Mallard	Tech.	>10,000 (reprod.)	>10,000 (reprod.)	No reproductive effects cited at any test level (1000, 5000, 10,000 ppm)	40964102	Yes
Mallard	99.6	50 ppm	Not established. Highest	None	00041441	Supplemental

Table B.1-4. Avian Reproduction Findings of Chlorothalonil Exposure						
Test Species	% a.i.	NOEL PPM	LOEL PPM	Endpoints affected	Citation (MRID #)	Fulfills Guideline ?
			level of 50 ppm did not cause impairment.			ntal

In the most sensitive avian reproduction study (MRID 45710218) chlorothalonil was administered to bobwhite quail in the diet at nominal concentrations of 0 (negative control), 40, 160, or 640 ppm. Mean-measured concentrations were <1.5 (<LOD, control), 41, 153, and 624 ppm a.i., respectively. A treatment-related reduction in the number of eggs laid/hen and thus in the number of 14-day old survivors/hen were observed at the 624 ppm a.i. level. The number of eggs laid/hen was 62.0 for the control group, and 62.4, 68.9, and 51.0 for the 41, 153, and 624 ppm a.i. test groups, respectively. The number of 14-day old survivors/hen was 37.2 for the control group, and 42.8, 42.2, and 30.4 for the 41, 153, and 624 ppm a.i. test groups, respectively. Although not statistically significant, these findings were considered to be biological significance by the study authors and the study reviewers.

SDS-3701

Avian reproduction studies have also been required for SDS-3701. These studies are summarized in the following table. The most sensitive NOAEC was 50 ppm based on reduction in eggshell thickness. However, the relevance of this endpoint to terrestrial amphibians is questionable. Therefore, the NOAEC of 100 ppm was chosen for use in risk assessment.

Table B.1-5. Avian Reproduction Findings (SDS-3701)						
Test Species	% SDS-3701	NOEL PPM	LOEL PPM	Endpoints affected	Citation (MRID)	Fulfills Guideline?
Mallard	99.6	50	100	Reduction in eggshell thickness seen at 100 ppm; at 250 ppm adult body weight, food consumption, and gonad development affected, as well as effects on numbers of eggs laid, embryonic development, eggshell thickness, hatchability, and hatching survival.	40729402	Yes
Bobwhite	99.6	100	250	Reduction in numbers of eggs laid	40729404	Yes

B.2 Toxicity to Mammals

Wild mammal testing is required on a case-by-case basis, depending on the results of lower tier laboratory mammalian studies, intended use patterns, and pertinent environmental fate characteristics. For this assessment, registrant-submitted reproduction toxicity data obtained from the Agency's Health Effects Division (HED) was used. Acute and chronic toxicity data for mammals is presented in Sections B.2.1 and B.2.2, respectively.

B.2.1 Mammals, Acute

Chlorothalonil

Acute mammalian toxicity studies for chlorothalonil are summarized in Table B.2-2. The available mammalian data indicate that chlorothalonil is "practically non-toxic" to small mammals on an acute oral basis, based on the rat oral LD₅₀.

Table B.2-1 Mammalian Acute Toxicity Findings--Chlorothalonil					
Test Species	% a.i.	LD50	Citation (MRID #)	Toxicity Category	Fulfills guidelines?
Rat (small mammal surrogate)	96%	Oral >10,000	MRID 00094941	practically non-toxic	Yes

SDS-3701

Data on the toxicity of SDS-3701 to mammals are tabulated below. These data indicate that the degradate SDS-3701 is more toxic to mammals than the parent chlorothalonil, and is moderately toxic on an acute oral basis.

Table B.2-2 Mammalian Acute Toxicity Findings--SDS-3701				
Test Species	LD₅₀ mg/kg	Comments	Citation (MRID #)	Toxicity Category
Rat (small mammal surrogate)	242 (females)	The LD50 for males was 422 mg/kg-bw and was 332 for the combined sexes	MRIDs MRID 00047938, 00047939, and 00095783	moderately toxic

B.2.2 Mammals, Reproduction Studies

Chlorothalonil

When available, 2-generation reproduction toxicity studies are used to estimate chronic risk to mammals. In a two-generation study, Sprague Dawley rats were administered chlorothalonil (98%) in the diet at levels of 0, 500, 1500 or 3000 ppm (0, 38, 115 and 234 mg/kg/day). For parental/systemic toxicity, the NOAEL was less than 500 ppm (<38 mg/kg/day). The LOEL was 500 ppm (38 mg/kg/day) based on hyperplasia of renal and forestomach tissues. For offspring toxicity, the NOEL was 1500 ppm (115 mg/kg/day) and the LOEL was 3000 ppm (234 mg/kg/day) based on lower neonatal body weights by day 21 (MRID 41706201).

Test Species	Offspring NOAEL	Offspring LOAEL	Citation (MRID)	Fulfills Guidelines?
Rat (2 generation reproduction)	1500 ppm	3000 ppm decrease in pup weight	41706201C	Yes

SDS-3701

Data on the toxicity of the SDS-3701 degradate to mammalian reproduction are tabulated below. In a 1-generation reproduction study in Sprague-Dawley rats, SDS-3701 was administered at 0, 10, 20, 30, 60, or 120 ppm (approximately 0, 0.5, 1.0, 1.5, 3.0 or 6.0 mg/kg/day). For parental systemic toxicity, the NOEL was 1.5 mg/kg/day and the LOEL was 3.0 mg/kg/day. No ecologically relevant reproductive or offspring toxicity occurred at up to the highest level tested (MRID 00127845).

In a 3-generation reproduction study in Sprague-Dawley rats, SDS-3701 was administered at 0, 10, 60 or 125 ppm (approximately 0, 0.5, 3.0 or 6.25 mg/kg/day). No ecologically relevant reproductive or offspring toxicity occurred at up to the highest level tested, 6.25 mg/kg/day (MRID 00127844).

Test Species	Reproduction NOAEL	Reproduction LOAEL	Citation (MRID #)
Rat (3-generation reproduction)	120 ppm	None	00127844
Rat (1-generation reproduction)	125 ppm	None	00127845

B.2.3 Mammals: Open Literature

Based on a review of the open literature, no additional information on the acute or chronic toxicity of chlorothalonil or SDS-3701 to mammals was located that produced more sensitive endpoints relevant to ecological risk assessment.

B.3 Toxicity to Non-Target Terrestrial Invertebrates

B.3.1 Honey Bee Acute Contact Study

Honey bee acute contact LD₅₀ study is required if the proposed use will likely result in exposure to honey bees. The available acute contact toxicity findings for chlorothalonil are summarized in Table B.3-1. The available data suggests that chlorothalonil is practically non-toxic to honey bees.

Table B.3-1: Non-target Insect Acute Contact Toxicity Findings

Test Species	% a.i.	Results	Citation (MRID)	Toxicity Category	Fulfills Guideline?
Honey bee	Tech.	at 181 ug/bee, 14.28% mortality	00036935	Practically non-toxic	Yes
Honey bee	Tech.	non-toxic at 181 ug/bee	00077759	Practically non-toxic	Yes

In addition, a number of studies were located in the open literature that evaluated the toxicity of chlorothalonil to terrestrial invertebrates (Table B.3-2). The available data suggest that sensitive terrestrial invertebrates exist, but many of the species tested were not sensitive to chlorothalonil at the levels tested, which typically approximated maximum labeled application rates.

Table B.3-2: Nontarget Invertebrate Acute Contact Toxicity Findings from the Open Literature

Test Species	Material Tested	Results Summary	Citation ECOTOX#	Comment
Aphid endoparasitoid wasp (<i>Aphidius rhopalosiphii</i>)	Daconil 500 Flowable (50% a.i.)	NOEL = 1250 g a.i./ hectare (1.1 lbs a.i./Acre;	64665 (Jansen, 1999)	Chlorothalonil was considered “slightly harmful” to <i>A. rhopalosiphii</i> . Endpoints evaluated included mortality and reproduction
Earthworms and arthropods	Daconil 2787	NOAEL = 12.6 kg (AI)/ha (11.25 lbs a.i./acre)	71484 (Potter et. al. 1990)	Endpoint studied was abundance and biomass
Carabidae	Daconil (54% a.i.)	NOAEL = 8.2 kg a.i./ha (7.3 lbs a.i./Acre)	89639 (Smitley and Rothwell, 2003)	Transient reduction in abundance occurred after the first week of treatment.
		No reduction in		--

Test Species	Material Tested	Results Summary	Citation ECOTOX#	Comment
Onion Thrips <i>Thrips tabaci</i>	Bravo 500	abundance occurred at an application rate of 4.5 pints/acre (approx. 2.3 lbs a.i./Acre).	90255 (Al-Dosari et.al 1996)	
Corn earworm; fall armyworm	Bravo 720	Dietary exposure to the fall armyworm and corn earworm at 800 ppm and higher resulted in high mortality rates.	90193 (Lynch, 1996)	--
Mustard beetle (<i>Phaedon cochleariae</i>)	Jupital	No significant effects on survival.	90531 (Cherry et. al. 1992)	--
Rove Beetle (<i>Aleochara bilineata</i>)	Daconil 500 (50%)	No effects occurred at levels reportedly equivalent to maximum application rate.	63488 (Samsoe-Petersen, 1995)	Effects evaluated included reproduction, egg production and viability.
Predatory mite (<i>Amblyseius victoriensis</i>)	Bravo	<10% mortality at up to 10-times the reported maximum field rate	67984 (James et.al. 1995)	--
Aphid Predatory Midge (<i>Aphidoletes aphidimyza</i>)	Repulse	Mortality was approximately 10% in larvae after 72 hours at 1100 mg a.i./L.	89884 (Helyer, 1991)	Test level was reportedly equivalent to the maximum labeled application rate.
Aphareta pallipes E. muscae Delia antique D. platura Coenosio tigrina	Bravo 6 F	LC50 = 7.83% 100% mortality at 0.0054% No effects to these species were observed at any test level	71029 (Carruthers et. al. 1985)	Cups sprayed with various concentrations of chlorothalonil from 0.0054% to 10.8%.

B.4 Toxicity to Freshwater Animals

B.4.1 Freshwater Fish, Acute Submitted Data

In order to establish the toxicity of a pesticide to freshwater fish, the minimum data required on the technical grade of the active ingredient are two (one cold water and one warm water) freshwater fish toxicity studies. The freshwater fish acute toxicity findings for the technical grade of the active ingredient are summarized in the Table B.4-1. The registrant submitted studies are consistent with each other and indicate that chlorothalonil is very highly toxic to freshwater fish.

Table B.4-1: Freshwater Fish Acute Toxicity Findings					
Test Species	% a.i.	LC ₅₀ ppb a.i.	Citation (MRID #)	Toxicity Category	Fulfills Guideline?
Rainbow trout	96	42.3	00056486	very highly toxic	Supplemental
Rainbow Trout	96	18	45710219	very highly toxic	Yes
Bluegill	96	60	00041439	very highly toxic	Yes
Bluegill	99	84	00029410	very highly toxic	Yes
Bluegill	98	51	RIOCHL01 Pitcher (1976)	very highly toxic	Yes
Channel catfish	96	48	00030390	very highly toxic	Yes
Fathead minnow	96	23	00030391	very highly toxic	Yes

The most sensitive LC50 value for registrant-submitted freshwater fish studies was from MRID 45710219. In this study Rainbow Trout (*Oncorhynchus mykiss*) were exposed under flow-through conditions to chlorothalonil at nominal concentrations of 0 (negative and solvent controls), 0.010, 0.018, 0.032, 0.056, and 0.10 ppm. Mean-measured concentrations were ≤0.0004 (≤LOD, solvent control), 0.0085, 0.0177, 0.0357, 0.0491, and 0.0739 ppm a.i., respectively. Following 96 hours of exposure, cumulative mortality was 0% in control groups and 0, 60, 90, 100, and 100% in the 0.0085, 0.0177, 0.0357, 0.0491, and 0.0739 ppm a.i. treatment groups, respectively. The 96-hour LC₅₀ (with 95% C.I.) was 0.018 (0.013-0.023) ppm a.i., which classifies chlorothalonil as very highly toxic to Rainbow Trout (*Oncorhynchus mykiss*) on an acute toxicity basis. Swimming and/or hanging at the surface, lethargy, loss of equilibrium, and/or moribundity were observed in surviving fish from the ≥0.0177 ppm a.i. groups; effects were first observed within 6 hours of exposure and continued through 96 hours in groups with surviving fish. The NOEC (for mortality and sub-lethal effects) was 0.0085 ppm a.i.

B.4.2 Freshwater Fish, Open Literature Data

Acute LC50s reported in the open literature are summarized in Table B.4-2. One study (Davies and White, 1985) located in the open literature reported an LC50 that was more sensitive than the most sensitive registrant submitted studies. Davies and White (1985) reported an LC50 of 10.5 ppb in rainbow trout. The most sensitive rainbow trout LC50 from registrant submitted studies was 18 ppb (MRID 45710219), which is similar to the LC50 reported by Davies and White (1985) of 10.5 ppb. Although the LC50 of 10.5 ppb was from a study that used low oxygen levels, the oxygen levels utilized did not affect

controls and were not outside values commonly found in the environment. Therefore, an LC50 of 10.5 ppb was chosen for use in risk estimation.

Table B.4-2: Freshwater Fish Lethal Effect Studies from the Open Literature

Test Species	LC ₅₀ ppb a.i.	Citation (MRID or Ecotox No.)	Toxicity Category	Comment
Jollytail	16	Ecotox No. 87454	very highly toxic	--
Spotted mountain galaxias	19 – 29	Ecotox No. 87454	very highly toxic	Tested species has not been evaluated in registrant submitted studies.
Threespine stickleback	69	Ecotox No. 7055	very highly toxic	Tested species has not been evaluated in registrant submitted studies.
Rainbow trout	10.5 - 76	Ecotox No. 87454; 7055	very highly toxic	The 10.5 ug/L value is the most sensitive acute LC50 available and is consistent with the range of LC50s in rainbow trout. 10.5 ug/L was chosen for use in risk quotient calculations. Study used an oxygen level of 50%; however, controls were not affected by the low DO, and the oxygen levels are not outside of the range expected to occur in the environment.
Tilapia	100 - 120	Ecotox No. 229772	highly toxic	--

A number of studies were also located in the open literature that evaluated sublethal endpoints such as biochemical endpoints. These studies did not report more sensitive toxicity values on endpoints that are correlated with assessment endpoints (survival and reproduction).

B.4.3. Toxicity of Formulated Products:

Formulated product testing is specified for products with direct application to aquatic habitats and for typical end-use products where the EEC for the active ingredient is \geq LC₅₀. The previous Phase IV Review (1/12/93) specified further testing of a 54% ai flowable concentrate due to a cranberry use. The freshwater fish acute toxicity findings for the 54%, 75%, and Bravo W-75 formulations are summarized in Table B.4-3. The data suggest that studies using chlorothalonil products were of similar toxicity than studies using technical grade chlorothalonil.

Table B.4-3: Freshwater Fish Acute Toxicity Findings—Formulated Products				
Test Species	% a.i.	LC₅₀ ppb formulation	Citation (MRID #)	Toxicity Category (FP)
Rainbow trout	54 (Bravo 720)	61 (33.2 ppb ai)	43302101	very highly toxic
Bluegill	54 (Bravo 720)	49 (26.3 ppb ai)	42433804	very highly toxic
Rainbow trout	75	152 (114 ppb ai; 48-hr study)	00087304	highly toxic
Rainbow trout	75	103 (77.2 ppb ai)	00087303	highly toxic
Bluegill	Bravo W-75	167 (125 ppb ai)	00087258	highly toxic

B.4.4. Toxicity of SDS-3701

Testing using the degradate SDS-3701 has been previously specified due to its persistence in water. Freshwater fish acute toxicity findings for the degradate SDS-3701 are summarized in Table B.4-3. These studies show that SDS-3701 is "slightly toxic" to the bluegill and therefore is significantly less toxic than parent chlorothalonil.

Table B.4-4: Freshwater Fish Acute Toxicity Findings—SDS-3701

Test Species	% SDS-3701	LC₅₀ (ppb)	Citation (MRID #)	Toxicity Category
Bluegill	not avail.	45,000	00029415	slightly toxic
Bluegill	99	15,000	00030393	slightly toxic

B.4.5. Freshwater Fish, Chronic Submitted Data

Data from fish early life-stage testing is required for chlorothalonil since it can be expected to be transported to water from the intended use site, acute LC₅₀ values are less than 1 mg/L, and aquatic EECs are ≥ 0.01 of LC₅₀s. The fish early life stage data are summarized in Table B.4-5.

In an early life-stage study in fathead minnows (MRID 00030391), fish were exposed to chlorothalonil at 0, 0.6, 1.4, 3.0, 6.5, and 16 ppb under flow-through conditions. Exposure at 16 and 6.5 ppb chlorothalonil produced adverse effects on egg production and on the survival. Specifically, the hatching success in the control groups averaged

between 90% and 92.5% compared with 79.5% at 6.5 ppb and 46.5% at 16 ppb. The percent survival of the Fo generation in the control groups averaged 94% compared with 9% at 16 ppb. The results indicate that fathead minnow hatching success and survival were affected between 3 and 6.5 ppb.

Table B.4-5. Chronic Fish Toxicity Findings

Test Species	% a.i.	NOAEL (ppb)	LOAEL (ppb)	Citation (MRID #)	Endpoints Affected
Fathead minnow	96	3	6.5	00030391	Hatching success and survivability

B.4.7. Effects to Amphibians

No relevant data on terrestrial or aquatic phase amphibians were located in the open literature or were submitted by chlorothalonil registrants.

B.4.8. Freshwater Invertebrates, Acute Submitted Data

Registrant submitted freshwater invertebrate toxicity studies for technical grade chlorothalonil are summarized in Table B.4-6. Two studies in daphnids were submitted that reported consistent results. The lowest EC50 was 54 ppb (MRID 45710221).

Table B.4-6: Freshwater Invertebrate Toxicity Findings

Test Species	% a.i.	LC ₅₀ (ppb)	Citation (MRID #)	Toxicity Category	Fulfills Guideline?
<i>Daphnia magna</i>	Tech.	68	00068754	very highly toxic	Yes
<i>Daphnia magna</i>	Tech.	54	45710221	very highly toxic	Supplemental

B.4.9. Freshwater Invertebrates, Open Literature Data

Table B.4-7 summarizes the information available from the open literature. The data found in the open literature are not inconsistent with the registrant submitted studies; however, the only species tested in registrant submitted studies was the daphnid. Acute studies in several additional species were located in the open literature, which are summarized in Table B.4-7.

Table B.4-7: Freshwater Invertebrate Acute Effect Studies from the Open Literature

Study type / Test material	Test Organism (Common and Scientific Name) and Age and/or Size	Test Design	Endpoint Concentration in ppb	Citation (ECOTOX #)	Rationale for Use in Risk Assessment ⁽¹⁾
Acute (48 hr) Bravo 500 (40% a.i.)	Water flea (<i>Daphnia magna</i>) adult egg-bearing	Static Renewal; 4 treatment levels; 2 replicates, one fed and one not fed; algal food source	LC50 = 129 (95% CI 84-174) EC50 = 97 (95% CI 81-113)	Ernst et. al. 1991 (7055)	Qualitative. EC50 is not more sensitive than the lowest registrant-submitted study. Results based on nominal chlorothalonil levels.
Acute: Lethal and Sublethal 4 and 7 day values reported ≥ 98% a.i.	Giant Tasmanian Freshwater Crayfish (<i>Astacopsis gouldi</i>)	Flow through; 4 and 7 day values reported; 5 treatment levels	LC50 = 12.0 (7.9-18.1) 4 Day LC50 = 3.6 (2.1-6.0) 7-Day value	Davies et. al 1994 (64835)	Quantitative (acute lethal studies only)
Acute: Lethal and Sublethal 4 and 7 day values reported ≥ 98% a.i.	Amphipod (<i>Neoniphargus sp. A</i>)	Flow through; 4 and 7 day values reported; 5 treatment levels	LC50 = >40 (4 and 7 Day)	Davies et. al 1994 (64835)	QUAL The LC50 value was above the highest test concentration.
Acute: Lethal and Sublethal 4 and 7 day values reported ≥ 98% a.i.	Isopod (<i>Colubotelson chiltoni minor</i>)	Flow through; 4 and 7 day values reported; 5 treatment levels	LC50 = >40 (4 and 7 Day)	Davies et. al 1994 (64835)	QUAL The LC50 value was above the highest test concentration.
Acute: Lethal and Sublethal 4 and 7 day values reported ≥ 98% a.i.	Freshwater aytid shrimp <i>Parataya australiensis</i>	Flow through; 4 and 7 day values reported; 5 treatment levels	LC50 = 16 (14.4-17.9) 4-Day LC50 = 10.9 (9.1-13.1) 7-Day	Davies et. al 1994 (64835)	Quantitative (acute lethal data). The LC50 value is definitive and bounded by a 95% confidence interval. Study design appears adequate.

Toxicity of Formulated Product: The freshwater invertebrate toxicity findings for formulated product testing are summarized in the following table.

Table B.4-8: Freshwater Invertebrate Toxicity Findings—Formulated Product

Test Species	% a.i.	LC ₅₀ (ppb) formulation	Citation (MRID #)	Toxicity Category (FP)	Fulfills Guideline? (for FP tested)
<i>Daphnia magna</i>	54 (Bravo 720)	180 (97; ai) Probit slope = 7.9	42433806	highly toxic	Yes

Toxicity of SDS-3701: The freshwater invertebrate toxicity findings for the degradate, SDS-3701, are summarized in the following table. The data indicate that SDS-3701 is slightly toxic to aquatic invertebrates.

Table B.4-8: Freshwater Invertebrate Toxicity Findings—SDS-3701

Test Species	% SDS-3701	EC ₅₀ (ppb)	Citation (MRID #)	Toxicity Category	Fulfills Guideline?
<i>Daphnia magna</i>	99	26,000 Probit slope = 5.75	00030394	slightly toxic	Yes (for SDS-3701)

In MRID 00030394, the acute toxicity of SDS-3701 was evaluated in daphnids. Procedures used in this acute toxicity test were based on EPA protocols specified in “Methods for Acute Toxicity Tests with Fish, MACroinvertebrates, and amphibians (U.S.EPA, 1975). Acetone was used as solvent and in a solvent control. Nominal concentrations were 10, 15, 22, 32, 46, 68 and 100 mg/L (ppm) and were tested in triplicate. The 48-hour EC₅₀ for Daphnids exposed to SDS-3701 was calculated to be 26 ppm with 95% confidence limits of 21 to 31 ppm. The lowest concentration at which 100% mortality occurred was 68 ppm, while highest concentration in which there were no deaths was 10 ppm.

B.4.6 Freshwater Invertebrate, Chronic Submitted Data

The aquatic invertebrate life-cycle toxicity findings are summarized in the following table.

Table B.4-9: Aquatic Invertebrate Life-Cycle Toxicity Findings

Test Species	NOEL (ppb)	LOEL (ppb)	Citation (MRID #)	Endpoints Affected	Fulfills Guideline?
<i>Daphnia magna</i>	39	79	00115107	survival, cumulative numbers of offspring/female	Yes
	0.6	1.8	45710222	Survival	

In the most sensitive chronic study in aquatic invertebrates (MRID 45710222), the 21-day-chronic toxicity of chlorothalonil to *Daphnia magna* was studied under static renewal conditions. Nominal concentrations were 0 (negative and solvent controls), 0.0010, 0.0032, 0.010, 0.032, and 0.10 ppm. Mean-measured concentrations were <0.0001 (<LOD, solvent control), 0.00060, 0.0018, 0.0058, 0.019, and 0.075 ppm a.i., respectively. However, chlorothalonil was unstable under the static renewal conditions employed in this test, declining to less than the level of detection to 62% of nominal concentrations in expired test media. Because chlorothalonil concentrations declined to < the level of detection at the lower concentrations, there is considerable uncertainty in the chlorothalonil levels associated with toxic effects in this study.

After 21 days of exposure, cumulative adult mortality/immobility was 7% for both control groups, and 5, 25, 40, 55, and 60% in the 0.00060, 0.0018, 0.0058, 0.019, and 0.075 ppm a.i. treatment groups, respectively. There was a significant inhibitory effect on reproduction (number of live young/adult) at the 0.075 ppm a.i. test level. The total number of live young produced per adult was at least 40 for the control through 0.019 ppm a.i. test groups, and was 31 for the 0.075 ppm a.i. group. No dead young were observed during the study in any test group, and the number of unhatched eggs per adult was <1 for all control and test groups. Terminal growth measurements were not performed. The NOAEC for survival was 0.6 ug/L (0.0006 mg/L).

B.5 Toxicity to Non-target Terrestrial Plants

B.5.1 Non-Target Terrestrial Plants: Submitted Data

The required tier 1 plant toxicity data are summarized in Table B.5-1.

Table B.5-1: Nontarget Terrestrial Plant Toxicity Findings

Study	% a.i.	Results (lb ai/A)	Citation (MRID #)	Fulfills Guideline?
Seed germination/seedling emergence--Tier 1 (122-1A); 10 species	97.9	NOEL \geq 16	42433808	Yes
Vegetative vigor--Tier 1 (122-1B); 10 species	97.9	NOEL \geq 16	42433809	Yes

B.5.2 Non-Target Terrestrial Plants: Open Literature Data

Based on a review of the open literature, no additional information was located that indicates greater non-target terrestrial plant sensitivity to chlorothalonil than the submitted data. Studies located in the open literature were predominantly efficacy studies (i.e., studies that evaluated effects of chlorothalonil on fungal diseases) or were studies that did not elicit adverse effects to plants.

B.5.3 Aquatic Plants: Registrant Submitted Data

Available aquatic plant toxicity data summarized in Table B.5-2. The most sensitive aquatic plant species from registrant studies was the diatom with an EC₅₀ of 14 ppb.

Table B.5-2: Nontarget Aquatic Plant Toxicity Findings

Test Species	% a.i.	Results (ppb)	Citation (MRID #)	Meets Guideline Requirements
Freshwater Vascular Plant Duckweed (<i>Lemna gibba</i>)	TGAI	Number of fronds EC ₅₀ = 730 ppb (670-800ppb) Probit slope = 8.08 Biomass (dry weight) EC ₅₀ = 630 ppb (550-730ppb) Probit Slope = 5.3 NOEC 290ppb	44908102	The only DER available is a contractor version. The contractor classifies this study as core.
Freshwater non-vascular plant. <i>Selenastrum capricornutum</i>	97.9	EC ₅₀ = 190 NOEC = 50 LOEC = 100 Slope = 4.027 95% CI 1.34-6.71	42432801	Yes

Test Species	% a.i.	Results (ppb)	Citation (MRID #)	Meets Guideline Requirements
Freshwater non-vascular plant. (Diatom) <i>Navicula pelliculosa</i>	TGAI	EC50 = 14 ppb (12-17 ppb) Probit slope = 4.49 NOEC = 3.9 ppb	44908105	The only DER available is a contractor version. The contractor classifies this study as core.

B.5.3 Aquatic Plants: Open Literature Data

Aquatic plant studies located in the open literature are summarized in Table B.5-3. The most sensitive aquatic plant study was from Mezcua et al. (2002), which reported a 72-hour EC50 of 6.8 ppb in *Selenastrum capricorotum*. This study reportedly followed OECD 201.

Table B.5-3: Aquatic Plant Toxicity Tests (Laboratory)

Study type/ Test material	Test Organism (Common and Scientific Name)	Test Design	Endpoint Concentration / Results	Citation (ECOTOX)	Study Classification ⁽¹⁾
30% SC	Green Algae <i>Scenedesmus obliquus</i>	Laboratory bioassay	EC50 = 100 ppb	65723 (Ma et. al., 2001)	QUAL
30% SC	Green Algae <i>Chlorella pyrenoidosa</i>	Laboratory bioassay	EC50 = 8069 ppb		QUAL
Tech.	Green Algae <i>Selenastrum capricorotum</i>	Laboratory bioassay using Toxkit Algaltoxkit™ (Creasel Belgium) a growth inhibition assay – study reportedly followed OECD 201.	72-hr EC50 = 6.8 ppb. The EC50 @ 30 h = 42,400ppb	80747 (Fernandez-Alba et. al., 2002) 80359 (Mezcua et. al. 2002)	QUAN.

⁽¹⁾ QUAL = The paper is not appropriate for quantitative use but is of good quality, addresses issues of concern to the risk assessment and is used in the risk characterization discussion.

B.6 References

ECOTOX or MRID	Citation
2478	Kikuchi, M. 1993. Toxicity Evaluation of Selected Pesticides Used in Golf Links by Algal Growth Inhibition Test. <i>Journal of Japanese Society of Water Environment</i> 16(10):704-710
63488	Samsoe-Peterson, L. 1995. Effects of 37 Fungicides on the Rove Beetle <i>Aleochara bilineata</i> (Col. : Staphylinidae) in the Laboratory. <i>Entomophaga</i> 40(2):145-152.
64665	Jansen, J.P. 1999. Effects of Wheat Foliar Fungicides and the Aphid Endoparasitoid <i>Aphidius rhopalosiphi</i> DeStefani-Perez (Hym. Aphididae) on Glass Plates and on Plants. <i>Journal of Applied Entomology</i> . 123:217-223
64835	Davies PE;Cook LSJ;Goenarso D. 1994. Sublethal Responses to Pesticides of Several Species of Australian Freshwater Fish and Crustaceans and Rainbow Trout. <i>Environ Toxicol Chem</i> 13(8): 1341-1354 (OECDG Data File)
65723	MA, J., Zeng, R., Xu, L., Wang, S. 2002. Differential Sensitivity of Two Green Algae, <i>Scenedesmus obliquus</i> and <i>Chlorella pyrenoidosa</i> , to 12 Pesticides. <i>Ecotoxicology and Environmental Safety</i> 52(1):57-61
67894	James, D.G and Rayner M. 1995. Toxicity of viticultural pesticides to the predatory mites <i>Amblyseius victorensis</i> and <i>Typhlodromus doreenae</i> . <i>Plant Protection Quarterly</i> . 10(3):99-102.
7055	Ernst, W., Doe, K., Jonah, P., Young, J., Julien, G., and Hennigar, P. 1991. The Toxicity of Chlorothalonil to Aquatic Fauna and the impact of its operational Use on a Pond Ecosystem. <i>Archives of Environmental Contamination and Toxicology</i> . 21:1-9
71029	Carruthers, R.L., Whitfield, G.H., and Haynes, D.L. 1985. Pesticide-Induced Mortality of Natural Enemies of the Onion Maggot, <i>Delia antiqua</i> . <i>Entomophaga</i> 30(2):151-161
71484	Potter, D.A., Buxton, M.C., Redmond, C.T., Patterson, C.G., Powell, A.J. 1990. Toxicity of Pesticides to Earthworms (Oligochaete: Lumbricidae) and Effect on Thatch Degradation in Kentucky Bluegrass Turf. <i>Journal of Economic Entomology</i> 83(6):2362-2369
80359	Mezcua, M., Hernando, M.D., Piedra, L., Aguera, A., Fernandez-Alba, A.R. 2002. Chromatography-Mass Spectrometry and Toxicity Evaluation of Selected Contaminants in Seawater. <i>Chromatographia</i> 56(3/4):199-205

- 80747 Fernandez-Alba, A.R., Hernando, M.D., Piedra, L., Christi, Y. 2002. Toxicity Evaluation of Single and Mixed Antifouling Biocides Measured with Acute Toxicity Bioassays. *Analytical Chimica* 456:303-312.
- 89639 Smitley, D.R. and Rothwell, N.L. 2003. How the Use of Chlorothalonil on Golf Courses Impacts *Paenibacillus* sp., a Pathogen of *Ataenius spretulus* (Coleoptera: Scarabaeidae) *Journal of Economic Entomology* 96(3): 792-797
- 87454 Davies PE;White RWG. 1985. The Toxicology and Metabolism of Chlorothalonil in Fish. I. Lethal Levels for *Salmo gairdneri*, *Galaxias maculatus*, *G. truttaceus* and *G. auratus* and the Fate of ¹⁴C-TCIN in *S. gairdneri*. *Aquat Toxicol* 7(1/2): 93-105
- 89884 Heyler, N. 1991. Laboratory Pesticide screening Method for the Aphid Predatory Midge *Aphidoletes aphidmyza* (Rondani) (Diptera: Cecidomyiidae) *Biocontrol Science and Technology* 1:53-58
- 89911 Idris, A.B. and Grafius, E. 1993. Differential toxicity of pesticides to *Diadegma insulare* (Hymenoptera: Ichneumonidae) and Its Host, the Diamondback Moth (Lepidoptera:Plutellidae) *Horticultural Entomology* 86(2):529-536
- 90193 Lynch, R. E. 1996. Peanut Fungicides: Effect on survival and Development of the corn Earworm, Fall Armyworm and Velvetbean Caterpillar. *Peanut Science*. 23:116-123
- 90255 Al-Dosari, S.A, Cranshaw, W. S., Schweissing, F.C. 1996. Effects on Control of Onion Thrips from Co-application of Onion Pesticides. *Southwestern Entomologist*. 21(1):49-54
- 90531 Cherry, A.J., Sotherton, N.W., Wratten, S.D. 1992. The sub-lethal effects of foliar fungicides on the mustard beetle *Phaedon cochleariae* (F.) (Col., Chrysomelidae) *Journal of Applied Entomology*. 114:510-519
- 90311 McCarter, S.M. 1992. Effects of Bactericide Treatments on Bacterial Spot Severity and Yield of Different Pepper Genotypes and Populations of Certain Insects. *Plant Disease* 76(10): 1042-1045
- RIOCHL01 Pitcher F. 1976. Chlorothalonil 98% and Bluegill (*Leopomis macrochirus*), 96 hour Acute Toxicity. U.S. Environmental Protection Agency, Pesticide Regulation Division, Agricultural Research Center, Animal Biology Laboratory and fish Toxicity Laboratory; unpublished study.
- 00029410 Szalkowski, M.B.; Stallard, D.E.; Bachand, R.T., Jr. (1979) Acute Toxicity of 2,4,5,6-Tetrachloroisophthalonitrile (Chlorothalonil) to Bluegill Sunfish (μ -*Lepomis macrochirus*- μ): Research Report R-79-0003. (Unpublished

study received Feb 19, 1980 under 677-313; submitted by Diamond Shamrock Agricultural Chemicals, Cleveland, Ohio; CDL:099248-H)

- 00029415 Szalkowski, M.B.; Stallard, D.E.; Bachand, R.T., Jr. (1979) Acute Toxicity of 4-Hydroxy-2,5,6-trichloroisophthalonitrile (DS-3701) to Bluegill Sunfish (μ -*Lepomis macrochirus*- μ): Research Report R-79-0004. (Unpublished study received Feb 19, 1980 under 677-313; submitted by Diamond Shamrock Agricultural Chemicals, Cleveland, Ohio; CDL:099248-M)
- 00030388 Shults, S.K.; Killeen, J.C., Jr.; Heilman, R.D. (1979) Chlorothalonil (Technical) Eight-Day Dietary (LC50) Study in Bobwhite Quail. (Unpublished study received Feb 19, 1980 under 677-313; prepared in cooperation with Wildlife International, Ltd., submitted by Diamond Shamrock Agricultural Chemicals, Cleveland Ohio; CDL:099247-A)
- 00030389 Shults, S.K.; Killeen, J.C., Jr.; Heilman, R.D. (1979) Chlorothalonil (Technical) Eight-Day Dietary (LC50) Study in Mallard Ducks. (Unpublished study received Feb 19, 1980 under 677-313; prepared in cooperation with Wildlife International, Ltd., submitted by Diamond Shamrock Agricultural Chemicals, Cleveland, Ohio; CDL:099247-B)
- 00030390 Shults, S.K.; Killeen, J.C., Jr.; Heilman, R.D.; et al. (1980) Chlorothalonil (Technical) Acute Toxicity (LC50) Study in channel Catfish. (Unpublished study including report # BW-79-6-460, received Feb 19, 1980 under 677-313; prepared in cooperation with EG&G, Bionomics, submitted by Diamond Shamrock Agricultural Chemicals, Cleveland, Ohio; CDL:099247-E)
- 00030391 Shults, S.K.; Killeen, J.C., Jr.; Heilman, R.D.; et al. (1980) A Chronic Study in the Fathead Minnow (*Pimephales promelas*) with Technical Chlorothalonil. (Unpublished study including report # BW-79-6-443, received Feb 19, 1980 under 677-313; prepared in cooperation with EG&G, Bionomics, submitted by Diamond Shamrock Agricultural Chemicals, Cleveland, Ohio; CDL:099247-H)
- 00030393 Buccafusco, R.J. (1977) Acute Toxicity of DTX-77-0070 to Bluegill (μ -*Lepomis macrochirus*- μ). (Unpublished study including submitter summary, received Feb 19, 1980 under 677-313; prepared by EG&G, Bionomics, submitted by Diamond Shamrock Agricultural Chemicals, Cleveland, Ohio; CDL:099247-L)
- 00030394 LeBlanc, G.A. (1977) Acute Toxicity of DTX-77-0071 to the Water Flea (μ -*Daphnia magna*- μ). (Unpublished study including submitter summary, received Feb 19, 1980 under 677-313; prepared by EG&G, Bionomics, submitted by Diamond Shamrock Agricultural Chemicals, Cleveland, Ohio; CDL:099247-M)

- 00030395 Beavers, J.B. 1978. Acute Oral Toxicity of DS-3701 in the Mallard Duck. Received February 19, 1980. An unpublished report prepared by Wildlife International, Ltd. For Diamond Shamrock Corporation. (MRID#)
- 00039146 Dieterich, W.H. (1965) Acute Dietary Administration--Wildfowl: Project No. 200-163. (Unpublished study received Feb 25, 1976 under 6F1749; prepared by Hazleton Laboratories, Inc., submitted by Diamond Shamrock Agricultural Chemicals, Cleveland, Ohio; CDL:096459-B)
- 00041439 Shults, S.K.; Killeen, J.C., Jr.; Heilman, R.D.; et al. (1980) Chlorothalonil (Technical) Acute Toxicity (LC450[^]) Study in Bluegill. (Unpublished study including report # BW-79-6-446, received Feb 19, 1980 under 677-313; prepared in cooperation with EG&G, Bionomics, submitted by Diamond Shamrock Agricultural Chemicals, Cleveland, Ohio; CDL:099247-D)
- 00041440 Fink, R. (1976) Final Report: One-Generation Reproduction Study—Bobwhite Quail: Project No. 111-107. (Unpublished study including submitter summary, received Feb 19, 1980 under 677-313; prepared by Wildlife International, Ltd., submitted by Diamond Shamrock Agricultural Chemicals, Cleveland, Ohio; CDL:099247-F)
- 00041441 Fink, R. (1976) Final Report: One-Generation Reproduction Study--Mallard Duck: Project No. 111-108. (Unpublished study including submitter summary, received Feb 19, 1980 under 677-313; prepared by Wildlife International, Ltd., submitted by Diamond Shamrock Agricultural Chemicals, Cleveland, Ohio; CDL:099247-G)
- 00056486 Shults, S.K.; Killeen, J.C., Jr.; Heilman, R.D. (1980) Chlorothalo- nil (Technical) Acute Toxicity (LC50) Study in Rainbow Trout. (Unpublished study including report # BW-79-6-461, received Feb 19, 1980 under 677-313; prepared in cooperation with EG&G, Bionomics, submitted by Diamond Shamrock Agricultural Chemicals, Cleveland, Ohio; CDL:099247-C)
- 00068754 LeBlanc, G.A. (1977) Acute Toxicity of DTX-77-0072 to the Water Flea (*Daphnia magna*). (Unpublished study, including submitter summary, received Jan 19, 1978 under 677-229; prepared by EG & G, Bionomics, submitted by Diamond Shamrock Agricultural Chemicals, Cleveland, Ohio; CDL:232729-B)
- 00068753 Fink, R.; Beavers, J.B.; Brown, R. (1977) Final Report: Acute Oral LD50—Mallard Duck: Project No. 111-109. (Unpublished study, including submitter summary, received Jan 19, 1978 under 677- 229; prepared by Wildlife International Ltd. And Washington College, submitted by Diamond Shamrock Agricultural Chemicals, Cleveland, Ohio; CDL:232729-A)

- 00087258 McCann, J.A.; Pitcher, F. (1973) [Bravo TM W-75: Bluegill (*Lepomis macrochirus*)]: Test No. 548. (U.S. Agricultural Research Service, Pesticide Regulation Div., Agricultural Research Center, Animal Biology Laboratory and Fish Toxicity Laboratory; unpublished study; CDL:128550-A)
- 00087303 Pitcher, F. (1972) [Tetrachloroisophthalonitrile: Rainbow Trout (*Salmo gairdneri*)]: Test No. 503. (U.S. Agricultural Research Service, Pesticides Regulation Div., Animal Biology Laboratory; unpublished study; CDL:130256-A)
- 00087304 Pitcher, F. (1972) [Tetrachloroisophthalonitrile: Rainbow Trout (*Salmo gairdneri*)]: Test No. 504. (U.S. Agricultural Research Service, Pesticides Regulation Div., Animal Biology Laboratory; unpublished study; CDL:130254-A)
- 00094940 Shults, S.K.; Killeen, J.C.; Jr.; Ignatoski, J.A. (1981) Acute Dermal Toxicity(LD50) Study in Albino Rabbits with Technical Chlorothalonil: Document No. 296-5TX-80-0093-002. (Unpublished study received Feb 22, 1982 under 677-283; submitted by Diamond Shamrock Agricultural Chemicals, Cleveland, Ohio; CDL:246843-B)
- 00115108 Shults, S.; Killeen, J.; Ignatoski, J. (1981) Dietary Study (LC50) in Mallard Ducks with DS-3701 (4-Hydroxy-2,5,6-trichloroisophthalonitrile): Document No. 449-5TX-81-0008-002. (Unpublished study received Sep 21, 1982 under 0F2405; submitted by Diamond Shamrock Corp., Cleveland, OH; CDL:071097-B)
- 00115107 Suprenant, D., Altshal, L., LeBlanc, G. and Mastone J. 1981. The Chronic Toxicity of SDS-3701 (T-117-11) to the water flea (*Daphnia magna*). Research report submitted to Diamond Shamrock Corp., Plainville, OH, by EG&G Bionomics, Wareham, MA. Report #. BW-81-10-1031
- 00115109 Shults, S.; Killeen, J.; Ignatoski, J. (1981) Dietary Study (LC50) in Bobwhite Quail with DS-3701 (4-Hydroxy-2,5,6-trichloroisophthalonitrile): Document No. 448-5TX-81-0007-002. (Unpublished study received Sep 21, 1982 under 0F2405; submitted by Diamond Shamrock Corp., Cleveland, OH; CDL:071097-C)
- 00130733 Rodwell, D.; Mizens, M.; Wilson, N.; et al. (1983) A Teratology Study in Rats with Technical Chlorothalonil: Document No. 517- 5TX-82-0011-003. (Unpublished study received Jul 7, 1983 under 677-313; submitted by Diamond Shamrock Agricultural Chemicals, Cleveland, OH; CDL:251069-A)

- 40729402 Shults, S.; Wilson, N.; Killeen, J. (1988) Reproduction Study in Mallard Ducks with 4-Hydroxy-2,5,6-trichloroisophthalonitrile: Project ID. 230-106. Unpublished study prepared by Wildlife International Ltd. and Ricerca, Inc. 193.
- 40729404 Shults, S.; Wilson, N.; Killeen, J. (1988) Reproduction Study in Bobwhite Quail with 4-Hydroxy-2,5,6-trichloroisophthalonitrile: Project Id. 230-105. Unpublished study prepared by Wildlife International Ltd. and Ricerca, Inc. 195 p.
- 40964102 Shults, S.; Wilson, N.; Killeen, J. (1988) Reproduction Study in Mallard Ducks with Technical Chlorothalonil: Ricerca--Document No. 1469-87-0004-TX-002. Unpublished study prepared by Wildlife International Ltd. in cooperation with Ricerca, Inc. 196 p.
- 40964104 Shults, S.; Wilson, N.; Killeen, J. (1988) Reproduction Study in Bobwhite Quail with Technical Chlorothalonil: Ricerca--Document No. 1469-87/-0006-TX-002. Unpublished study prepared by Wildlife International Ltd. in cooperation with Ricerca, Inc. 185 p.
- 40964105 Shults, S.; Wilson, N.; Killeen, J. (1987) Acute Oral Toxicity (LD50) Study in Japanese Quail with Technical Chlorothalonil: Ricerca--Document No. 1582-87-0041-TX-002. Unpublished study prepared by Wildlife International Ltd. 54 p.
- 41706201 Lucas, F.; Benz, G. (1990) A Two Generation Reproduction Study in Rats with Technical Chlorothalonil: Lab Project Number: 87-0121: 1722-87-0121-TX-003. Unpublished study prepared by Ricerca, Inc., and Experimental Pathology Labs, Inc. 1673 p.
- 42432801 Hughes, J.; Williams, T. (1992) The Toxicity of Technical Chlorothalonil Fungicide to *Selenastrum capricornutum*: Lab Project Number: B038-001-1. Unpublished study prepared by Malcolm Pirnie, Inc. 37 p.
- 42433804 Machado, M.W. 1992. Bravo 720 – Acute Toxicity to Bluegill Sunfish (*Lepomis macrochirus*) Under Flow-Through Conditions. SLI Report No. 92-5-4248. Prepared by Springborn Laboratories, Inc., Wareham, MA. Submitted by ISK Biotech Corporation, Mentor, OH.
- 42433806 Putt, A.E. 1992. Bravo 720—Acute Toxicity to Daphnids (*Daphnia magna*) Under Flow-Through Conditions. SLI Report No. 92-4-4225. Prepared by Springborn Laboratories, Inc, Wareham, MA. Submitted by ISK Biotech Corporation, Mentor, OH.
- 42433808 Backus, P. (1992) Effect of Chlorothalonil on Seed Germination/Seedling

- Emergence (Tier I): Lab Project Number: 92-0119: 5234-92-0119-BE-001.
Unpublished study prepared by Ricerca, Inc. 63 p.
- 42433809 Backus, P. (1992) Effect of Chlorothalonil on Vegetative Vigor of Plants (Tier I): Lab Project Number: 92-0120: 5234-92-0120-BE-001.
Unpublished study prepared by Ricerca, Inc. 50 p.
- 43302101 Shults, S.; Brock, A.; Laveglia, J. (1994) Acute Toxicity to Rainbow Trout (*Oncorhynchus mykiss*) under Flow-through Conditions with BRAVO 720: Final Report: Lab Project Number: 5727/93/0120/TX/002: 93/0120: 94/1/5129. Unpublished study prepared by Ricerca, Inc. and Springborn Laboratories, Inc. 93 p.
- 45710218 Redgrave, V. 1993. Chlorothalonil: Bobwhite Quail Dietary Reproduction and Tolerance Studies. Unpublished study performed by Huntingdon Research Centre Ltd., Huntingdon, Cambridgeshire, England. Laboratory Project No. VCM 11/930496. Study sponsored by Vischim S.r.l. Milano, Italy. Study initiated March 30, 1992 and submitted December 10, 1993.
- 45710219 Douglas, M.T., *et al.* 1992. Chlorothalonil - The Acute Toxicity to Rainbow Trout. Unpublished study performed by Huntingdon Research Centre Ltd., Cambridgeshire, England. Laboratory Project No: VCM 8(b)/920232. Study sponsored by Vischim S.r.l., Milan, Italy. Study initiated November 11, 1991 and completed July 27, 1992.
- 45710222 Douglas, M.T., *et al.* 1992. An Assessment of the Effects of Chlorothalonil on the Reproduction of *Daphnia Magna*. Unpublished study performed by Huntingdon Research Centre Ltd., Cambridgeshire, England. Laboratory Project No: VCM 8(e)/920814. Study sponsored by Vischim S.r.l., Milan, Italy. Study initiated January 10, 1992 and completed September 23, 1992.

CHEMICAL	2,4-D	2,4-D	2,4-D	2,4-D	2,4-D	2,4-D
PERCENT	100	100	100	100	100	100
DESCRIP	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
SPECIES	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT
SIZE						
WEIGHT	0.5	0.8	0.5	0.4	0.6	0.8
STRAIN						
DIET						
TEMP	10	10	10	15	5	10
PH	7.4	7.4	8.5	7.4	7.4	6.5
HARDNESS	300	160	44	44	44	44
TYPE	STATIC	STATIC	STATIC	STATIC	STATIC	STATIC
TEST_UNT	LC	LC	LC	LC	LC	LC
TOX_UNT	MG	MG	MG	MG	MG	MG
LC50_24H	32	41	40	41.5	41.5	43.5
FROM_24H	23.7	32.4	30.7	33.1	32.8	35
TO_24H	43.3	51.9	52.1	52	52.5	54.1
LC50_48H						
FROM_48H						
TO_48H						
LC50_96H	24.5	37	40	41.5	41.5	43.5
FROM_96H	18.4	27.9	30.2	33.1	32.8	35
TO_96H	32.6	49	52.1	42	52.5	54.1

2,4-D	2,4-D	2,4-D	2,4-D	2,4-D	2,4-D	2,4-D
100	100	100	100	100	100	100
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	LAKE TROUT
0.5	0.4	0.6	0.5	0.8	0.8	0.3
10	10	10	10	10	10	10
7.4	7.4	6.5	7.5	7.5	8.5	7.4
44	44	44	44	44	44	44
STATIC	STATIC	STATIC	STATIC	STATIC	STATIC	STATIC
LC	LC	LC	LC	LC	LC	LC
MG	MG	MG	MG	MG	MG	MG
49	64	67	130	169	185	44.5
40.4	57.3	57.5	100	146	165	35.3
59.4	71.5	78	170	195	207	56.1
44	64	67	130	169	172	44.5
35.7	57.3	57.5	100	146	148	35.3
54.2	71.5	78	170	195	199	56.1

2,4-D	2,4-D	2,4-D	2,4-D	2,4-D	2,4-D	2,4-D
100	100	100	100	100	98	98
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
LAKE TROUT	LAKE TROUT	LAKE TROUT	LAKE TROUT	LAKE TROUT	RAINBOW TROUT	BLUEGILL
0.7	0.7	0.7	0.9	0.7	0.3	0.5
10	15	5	10	10	12	22
6.5	7.4	7.4	7.5	8.5	7.3	7.3
44	44	44	44	44	40	40
STATIC	STATIC	STATIC	STATIC	STATIC	STATIC	STATIC
LC	LC	LC	LC	LC	LC	LC
MG	MG	MG	MG	MG	MG	MG
63	64	65.7	105	127.5		
59.3	47.8	47.2	95.6	110		
67	85.7	91.4	115	149		
62	64	65.7	105	120	110	180
54	47.8	47.2	95.6	103	77	
71.2	85.7	91.4	115	140	157	

2,4-D
98
TECHNICAL MATERIAL
FATHEAD MINNOW
0.9
22
7.3
40
STATIC
LC
MG
133
103
171

CHEMICAL		TRICLOPYR		TRICLOPYR
PERCENT		43		43
DESCRIP		LIQUID		LIQUID
SPECIES		BLUEGILL		RAINBOW TROUT
SIZE				
WEIGHT		0.8		0.9
STRAIN				
DIET				
TEMP		22		12
PH		7.4		7.4
HARDNESS		44		44
TYPE		STATIC		STATIC
TEST_UNT		LC		LC
TOX_UNT		MG		MG
LC50_24H		>100.		>100.
FROM_24H				
TO_24H				
LC50_48H				
FROM_48H				
TO_48H				
LC50_96H		>100.		>100.
FROM_96H				
TO_96H				

CHEMICAL	DIURON	DIURON	DIURON	DIURON
PERCENT	95	95	95	95
DESCRIP	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
SPECIES	ASELLUS BREVICAUDUS	BLUEGILL	BLUEGILL	BLUEGILL
SIZE	MATURE			
WEIGHT		0.8	1.1	0.8
STRAIN				
DIET				
TEMP	15	13	12	7
PH	7.1	7.1	8	7.1
HARDNESS	44	44	300	44
TYPE	STATIC	STATIC	STATIC	STATIC
TEST_UNT	LC	LC	LC	LC
TOX_UNT	MG	MG	MG	MG
LC50_24H	>10.	27	>30.0	29.8
FROM_24H		24.8		27.3
TO_24H		29.4		32.5
LC50_48H				
FROM_48H				
TO_48H				
LC50_96H	15.5	9.5	8	9.3
FROM_96H	7.2	8.5	5.8	8.1
TO_96H	33.4	10.6	11.7	10.7

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
BLUEGILL	BLUEGILL	BLUEGILL	BLUEGILL	BLUEGILL
0.8	0.8	0.8	1.1	1.1
18	24	29	12	12
7.1	7.1	7.1	6.5	8
44	44	44	44	44
STATIC	STATIC	STATIC	STATIC	STATIC
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
16.7	8.4	3.6	>30.0	>30.0
14.9	7.7	3		
18.7	9.3	4.2		
8.2	6.4	2.8	8.6	10
7.4	5.9	2.3	7.1	8.5
9.1	7	3.3	10.4	11.8

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
BLUEGILL	BLUEGILL	BLUEGILL	BLUEGILL	COHO SALMON
1.1	1.1	1.1	0.9	1.4
12	12	12	12	13
8.5	9.5	8	7.5	7.1
44	44	170	44	44
STATIC	STATIC	STATIC	STATIC	STATIC
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
>30.0	38.8	>30.0	35	11
	33.2		28.6	
	45.4		42.8	
10.4	7	8.3	10.4	<2.4
7.3	5.4	7	7.6	
14.9	9	9.8	14.2	

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT
0.7	0.7	1	0.8	0.9
10	10	10	10	10
7	7.7	6.5	7.7	7.5
44	165	44	295	44
21 DAY DEGRA	STATIC	STATIC	STATIC	STATIC
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
>30.0	>5.0	>4.5	>5.0	4.2
				3.1
				5.7
12.8	1.9	2.1	1.9	1.4
9	1.5	1.7	1.5	1
18	2.5	2.6	2.5	1.9

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT
1	0.6	0.8	0.4	0.3
10	10	15	5	10
8.5	7.8	7.2	7.4	7.2
44	44	44	44	44
STATIC	STATIC	STATIC	STATIC	STATIC
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
>4.5	>4.0	2.8	>4.0	3.4
		1.9		2.9
		4.2		3.8
2.2	1.7	0.71	1.4	1.4
1.8	1.4	0.53	1.1	1.1
2.7	2.1	0.96	1.7	1.9

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT	CUTTHROAT TROUT
0.7	0.7	0.7	0.7	3.1
10	10	10	10	10
7	7	7	7	7.4
44	44	44	44	162
28 DAY DEGRA	14 DAY DEGRA	7 DAY DEGRA	0 DAY DEGRA	FLOW THROUGH
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
30	>20.0	>10.0	>5.0	>4.0
12.3	13.8	11.5	1.5	1.9
9.5	6.3	6.3	1.1	1.6
15.8	29.9	20.9	2	2.1

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
DAPHNIA PULEX	GAMMARUS FASCIATUS	LAKE TROUT	LAKE TROUT	LAKE TROUT
1ST INSTAR	MATURE			SWIMUP FRY
		1	0.9	
15	21	10	10	10
7.1	7.1	6.5	8	7
44	44	44	175	44
STATIC	STATIC	STATIC	STATIC	STATIC
EC	LC	LC	LC	LC
MG	MG	MG	MG	MG
	0.7	>4.5	3.5	4.2
	0.59		2.8	3.3
	0.83		4.4	5.3
1.4				
1				
1.9				
	0.16	2.5	2.1	1.1
	0.13	1.9	1.5	1
	0.19	3.1	3	1.3

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
LAKE TROUT	LAKE TROUT	LAKE TROUT	LAKE TROUT	LAKE TROUT
5.1	0.3	1.5	1.5	0.4
10	10	10	10	5
7.4	7	7	7	7.5
162	44	44	44	44
FLOW THROUGH	STATIC	21 DAY DEGRA	14 DAY DEGRA	STATIC
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
	3.5	>20.0	17.5	5.3
	2.2		10.8	4
	5.5		28.5	7
1.8	1.8	11.5	3.6	2.2
1.6	1.5	7.9	2.5	1.7
2.1	2	16.8	5	2.7

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
LAKE TROUT	LAKE TROUT	LAKE TROUT	LAKE TROUT	LAKE TROUT
1.2	1.1	1	1	0.4
10	10	10	10	15
8	8	8.5	7.5	7.5
295	44	44	44	44
STATIC	STATIC	STATIC	STATIC	STATIC
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
>3.0	3.3	4.2	>4.0	2.9
	2	2.8		2.2
	5.3	6.2		3.9
2.6	2.2	2.6	2.4	1.2
2	1.6	1.9	1.9	0.9
3.4	2.9	3.1	2.9	1.5

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
LAKE TROUT	LAKE TROUT	PTERONARCYS CALIFORNICA	RAINBOW TROUT	RAINBOW TROUT
		2ND YEAR CLASS		
1.5	1.5		1.5	0.8
10	10	15	12	13
7	7	7.1	7.5	7.1
44	44	44	44	44
STATIC	7 DAY DEGRA	STATIC	21 DAY DEGRA	STATIC
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
>3.5	>5.0	3.6		9
		2.8		7.6
		4.7		11
2.7	3.2	1.2	7.4	4.9
2.4	2.2	0.9	6.2	4.1
3	4.6	1.7	8.7	5.9

DIURON	DIURON	DIURON	DIURON	DIURON
80	95	95	95	95
WETTABLE POWDER	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
RAINBOW TROUT	RAINBOW TROUT	RAINBOW TROUT	RAINBOW TROUT	RAINBOW TROUT
1.2	1.2	1.2	1.2	1.2
13	2	7	13	18
7.1	7.1	7.1	7.1	7.1
44	44	44	44	44
STATIC	STATIC	STATIC	STATIC	STATIC
LC	LC	LC	LC	LC
MG	MG	MG	MG	MG
71	11.5	15.5	12.5	8.4
61.1	10.5	14.1	11.5	7.9
82.5	12.7	17.1	13.6	9.1
16	7.7	7.2	6.2	5.3
11.3	6.8	6.5	5.8	5.1
22.7	8.9	7.9	6.6	5.7

DIURON	DIURON	DIURON	DIURON	DIURON
95	95	95	95	95
TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL	TECHNICAL MATERIAL
RAINBOW TROUT	RAINBOW TROUT	RAINBOW TROUT	RAINBOW TROUT	SIMOCEPHALUS SERRULATUS
				1ST INSTAR
1.5	1.5	1.5	1.5	
12	12	12	12	15
7.5	7.5	7.5	7.5	7.1
44	44	44	44	44
0 DAY DEGRA	14 DAY DEGRA	28 DAY DEGRA	7 DAY DEGRA	STATIC
LC	LC	LC	LC	EC
MG	MG	MG	MG	MG
				2
				1.4
				2.8
3.5	13.4	9.4	4.2	
2.7	10.7	8.3	3.1	
4.4	16.7	10.7	5.6	

CHEMICAL	LINURON	LINURON	LINURON	LINURON	LINURON	LINURON
PERCENT	50	95	50	95	50	95
DESCRIP	WETTABLE POWDER	TECHNICAL MATERIAL	WETTABLE POWDER	TECHNICAL MATERIAL	WETTABLE POWDER	TECHNICAL MATERIAL
SPECIES	CHANNEL CATFISH	CHANNEL CATFISH	CHIRONOMUS PLUMOSUS	CHIRONOMUS PLUMOSUS	DAPHNIA MAGNA	DAPHNIA MAGNA
SIZE			3RD INSTAR	3RD INSTAR	1ST INSTAR	1ST INSTAR
WEIGHT	0.6	0.4				
STRAIN						
DIET						
TEMP	22	22	22	22	17	17
PH	7	7.4	7	7	7	7
HARDNESS	40	40	43	43	43	43
TYPE	STATIC	STATIC	STATIC	STATIC	STATIC	STATIC
TEST_UNT	LC	LC	EC	EC	EC	EC
TOX_UNT	MG	MG	MG	MG	MG	MG
LC50_24H	3.2					
FROM_24H	2.3					
TO_24H	4.4					
LC50_48H			4.3	2.9	0.21	0.27
FROM_48H			3.4	1.9	0.16	0.22
TO_48H			5.4	4.2	0.28	0.33
LC50_96H	1.8	2.9				
FROM_96H	1.2	2				
TO_96H	2.6	4.2				