Water Hyacinth Control Program

Biological Assessment

October 25, 2012

A program for effective control of Water Hyacinth in the Sacramento-San Joaquin Delta and its tributaries.

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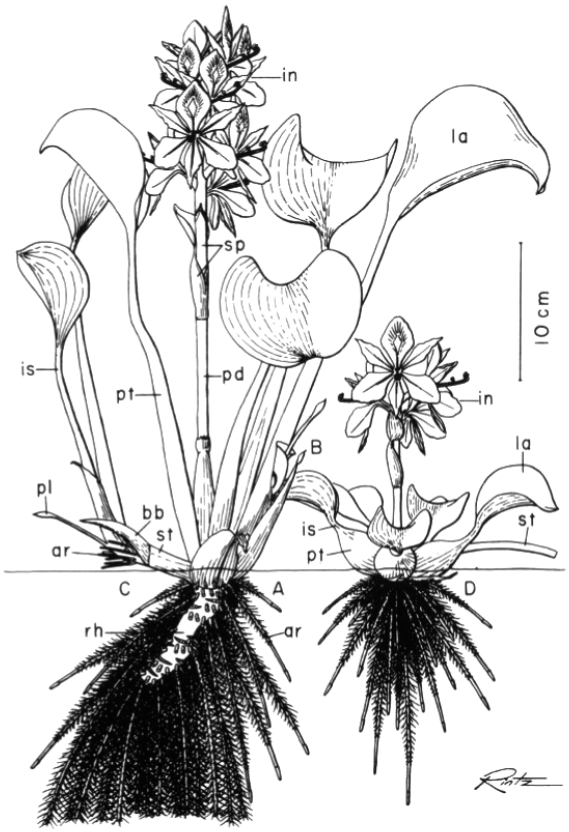

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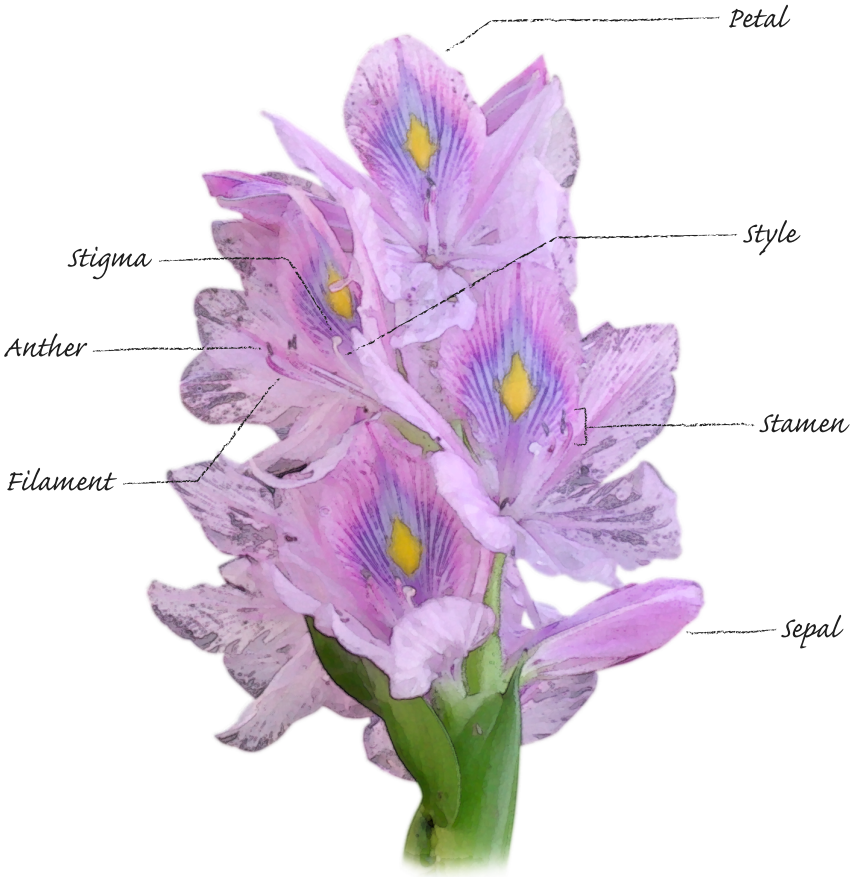
Water hyacinth *(Eichhornia crassipes)*



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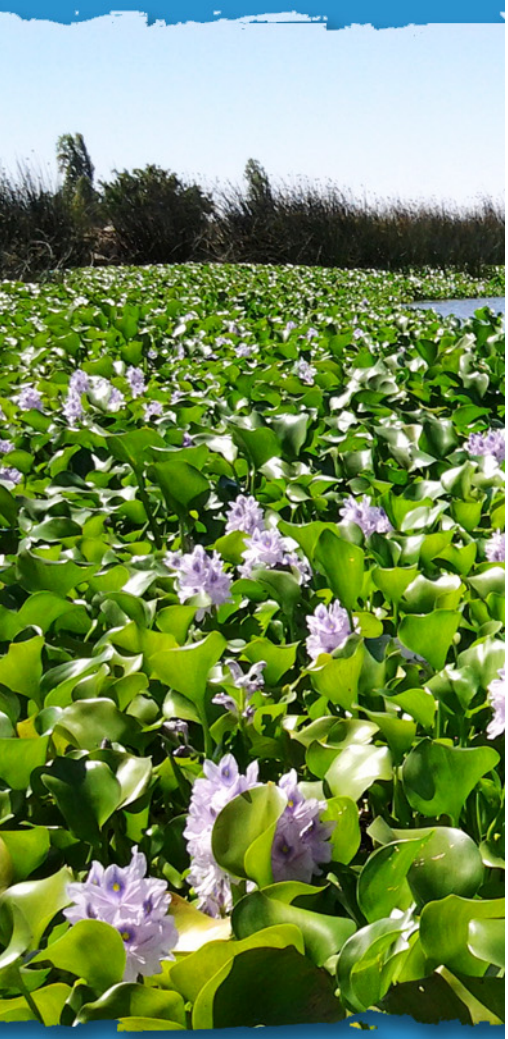

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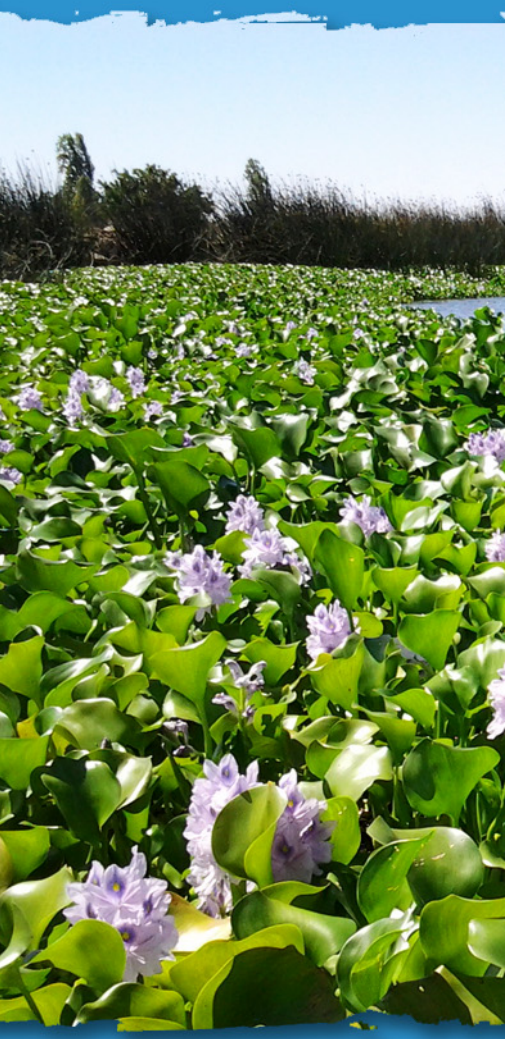
Acronyms and Abbreviations



Acronyms and Abbreviations

1. **2,4-D** – 2,4-dichlorophenoxyacetic acid
2. **a.i.** – Active Ingredient
3. **AGR** – Agricultural Supply (Basin Plan beneficial use)
4. **ALS** – Acetolactate Synthase
5. **AMPA** – Aminomethylphosphonic Acid
6. **APMP** – Aquatic Pesticide Monitoring Program
7. **Bay-Delta Estuary** – San Francisco Bay and Sacramento-San Joaquin Delta
8. **BA** – Biological Assessment
9. **BCF** – Bioconcentration Factor
10. **BDCP** – Bay Delta Conservation Plan
11. **BMP** – Best Management Practices
12. **BO or BiOp** – Biological Opinion
13. **BSMT** – Bay Study Midwater Trawl
14. **BSOT** – Bay Study Otter Trawl
15. **C** – Centigrade/Celsius
16. **CAC** – County Agricultural Commissioner
17. **CALFED** – California-Federal Bay Delta Program
18. **CCF** – Clifton Court Forebay
19. **CCWD** – Contra Costa Water District
20. **CDFA** – California Department of Food and Agriculture
21. **CDFG** – California Department of Fish and Game
22. **CE** – California Endangered
23. **CEC** – Contaminants of Emerging Concern
24. **CEQA** – California Environmental Quality Act
25. **CESA** – California Endangered Species Act
26. **cfs** – Cubic Feet Per Second
27. **CI** – Confidence Interval
28. **COA** – Coordinated Operations Agreement
29. **COMM** – Commercial Sport Fishing (Basin Plan beneficial use)
30. **COLD** – Cold Freshwater Habitat (Basin Plan beneficial use)
31. **CNDDB** – California Natural Diversity Database
32. **CNPS** – California Native Plant Society
33. **CRR** – Cohort Replacement Rate
34. **CSC** – California Species of Special Concern

Acronyms and Abbreviations *(continued)*

35. **CT** – California Threatened
36. **CVP** – Central Valley Project
37. **CVRWQB** – Central Valley Regional Water Quality Control Board
38. **CVTRT** – Central Valley Technical Review Team
39. **CWA** – Clean Water Act
40. **CWT** – Coded-Wire Tag
41. **dBA** – Decibels
42. **DBW** – California Department of Boating and Waterways
43. **DCC** – Delta Cross Channel
44. **Delta** – Sacramento-San Joaquin Delta
45. **DMA** – Dimethylamine Salt
46. **DO** – Dissolved Oxygen (measured in mg/l or ppm)
47. **DOC** – California Department of Conservation
48. **DPR** – California Department of Pesticide Regulation
49. **DPS** – Distinct Population Segment
50. **DRERIP** – Delta Regional Ecosystem Restoration Implementation Plan
51. **DWSP** – Delta Water Supply Project
52. **DWR** – California Department of Water Resources
53. **E:I** – Export to Import
54. **EA** – Environmental Assessment
55. **EC** – Effective Concentration
56. **EC50** – Effective Concentration for 50 Percent of Target
57. **EDCP** – *Egeria densa* Control Program
58. **EEC** – Exposure Estimate Concentration
59. **EFH** – Essential Fish Habitat
60. **EIR** – Environmental Impact Report
61. **EIS** – Environmental Impact Statement
62. **ERP** – Ecosystem Restoration Program
63. **ESA** – Endangered Species Act (federal)
64. **EST** – Estuarine habitat (Basin Plan beneficial use)
65. **ESU** – Evolutionary Significant Unit
66. **EWA** – Environmental Water Account
67. **FC** – Federal Candidate (for consideration of endangered or threatened status)
68. **FCH** – Federal Critical Habitat
69. **FCHP** – Federal Critical Habitat for this Species Proposed
70. **FE** – Federal Endangered

Acronyms and Abbreviations *(continued)*

71. **FETAX** – Frog Embryo Teratogenesis Assay – *Xenopus*
72. **FIFRA** – Federal Insecticide, Fungicide, and Rodenticide Act
73. **FMWT** – Fall Midwater Trawl
74. **FONSI** – Finding of No Significant Impact
75. **FRH** – Feather River Hatchery
76. **FT** – Federal Threatened
77. **GCID** – Glenn Colusa Irrigation District
78. **GGS** – Giant Garter Snake
79. **GWR** – Groundwater Recharge (Basin Plan beneficial use)
80. **HAPC** – Habitat Areas of Particular Concern
81. **HCP** – Habitat Conservation Plan
82. **HQ** – Hazard Quotient
83. **IEP** – Interagency Ecology Program
84. **IND** – Industrial Service Supply (Basin Plan beneficial use)
85. **IPM** – Integrated Pest Management
86. **JPE** – Juvenile Production Estimate
87. **JPI** – Juvenile Production Index
88. $\mathbf{K_{oc}}$ – Soil Adsorption Coefficient (normalized by organic matter)
89. **LC5** – Lethal Concentration for 5 Percent of Subjects
90. **LC10** – Lethal Concentration for 10 Percent of Subjects
91. **LC50** – Lethal Concentration for 50 Percent of Subjects
92. **LD50** – Lethal Dose or Lethal Dietary Dose for 50 Percent of Subjects
93. **LOC** – Level of Concern
94. **LOD** – Limit of Detection
95. **LOAEC** – Lowest Observable Adverse Effect Concentration
96. **LOEC** – Lowest Observable Effect Concentration
97. **LOEL** – Lowest Observable Effect Level
98. **LSNFH** – Livingston Stone National Fish Hatchery
99. **LSZ** – Low Salinity Zone
100. **MAF** – Million Acre Feet
101. **MATC** – Maximum Acceptable Toxicant Concentration
102. **MCL** – Maximum Contaminant Level
103. **MCP** – Maintenance Control Practices
104. **MCPA** – 4-chloro-2-methylphenoxyacetic acid
105. **MIGR** – Migration of Aquatic Organisms (Basin Plan beneficial use)
106. **mM** – Millimolar (a concentration of one thousandth of a mole per liter)

Acronyms and Abbreviations *(continued)*

107. **MOE** – Margin of Error or Margin of Safety
108. **MOU** – Memorandum of Understanding
109. **MRDL** – Maximum Residual Disinfectant Level
110. **MSA** – Magnuson-Stevens Fishery Conservation and Management Act
111. **MSDS** – Material Safety Data Sheet
112. **MUN** – Municipal and Domestic Supply
113. **NAV** – Navigation (Basin Plan beneficial use)
114. **NBA** – North Bay Aqueduct
115. **NCCP** – Natural Community Conservation Plan
116. **ND** – Non-detectable
117. **NIH** – National Institute of Health
118. **NMFS** – National Marine Fisheries Service
119. **NOAA-Fisheries** – National Oceanic and Atmospheric Administration-Fisheries (also previously referred to as NMFS, National Marine Fisheries Service)
120. **NOAEC** – Non-observable Adverse Effect Concentration
121. **NOEC** – Non-observable Effect Concentration
122. **NOEL** – Non-observable Effect Level
123. **NOI** – Notice of Intent
124. **NOP** – Notice of Preparation
125. **NPDES** – National Pollution Discharge Elimination System
126. **NPE** – Nonylphenol Ethoxylates
127. **NRDC** – Natural Resources Defense Council
128. **NTU** – Nephelometric Turbidity Units
129. **OCAP** – Operations Criteria and Plan
130. **OMP** – Operations Management Plan
131. **OMR** – Old and Middle River
132. **PAHs** – Poly aromatic Hydrocarbons
133. **PCA** – Pest Control Advisor
134. **PCE** – Primary Constituent Elements (of critical habitat)
135. **PEIR** – Program Environmental Impact Report
136. **PFMC** – Pacific Fisheries Management Council
137. **POD** – Pelagic Organism Decline
138. **POEA** – Polyethoxylated tallowamine
139. **ppb** – Parts per Billion (μg/l)
140. **ppm** – Parts per Million (mg/l or mg/kg)
141. **ppt** – Parts per Thousand (g/l)

Acronyms and Abbreviations *(continued)*

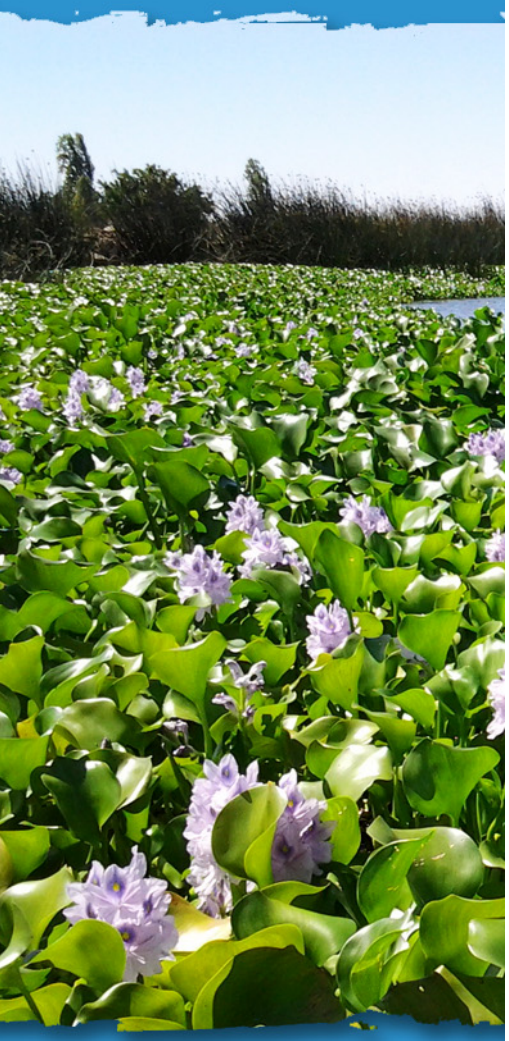
142. **PPE** – Personal Protective Equipment
143. **PRO** – Industrial Process Supply (Basin Plan beneficial use)
144. **psu** – Practical Salinity Unites
145. **PUR** – Pesticide Use Recommendations
146. **PVA** – Population Viability Analysis
147. **QAC** – Qualified Applicator Certificate
148. **QAPP** – Quality Assurance Project Plan
149. **RARE** – Rare, Threatened, or Endangered Species (Basin Plan beneficial use)
150. **RBDD** – Red Bluff Diversion Dam
151. **RCRA** – Resource Conservation and Recovery Act
152. **REC-1** – Water Contact Recreation (Basin Plan beneficial use)
153. **REC-2** – Non-water Contact Recreation (Basin Plan beneficial use)
154. **RfD** – Reference Dose
155. **RM** – River Mile
156. **ROD** – Record of Decision
157. **RPA** – Reasonable and Prudent Alternative
158. **RQ** – Risk Quotient
159. **RR** – Risk Ratio
160. **RTS** – Rotary Screw Traps
161. **RUP** – Restricted Use Permit
162. **SDIP** – South Delta Improvement Program
163. **SF** – San Francisco
164. **SFA** – Seasonally Flooded Agricultural
165. **SFEI** – San Francisco Estuary Institute
166. **SHELL** – Shellfish harvesting (Basin Plan beneficial use)
167. **SJ** – San Joaquin
168. **SJRRP** – San Joaquin River Restoration Program
169. **SL** – Standard Length
170. **SMR** – Standard Mortality Ratio
171. **SPWN** – Spawning, Reproduction, and/or Early Development (Basin Plan beneficial use)
172. **SVWMA** – Sacramento Valley Water Management Agreement
173. **SWB** – State Water Board (Water Resources Control Board)
174. **SWP** – State Water Project
175. **SWRCB** – State Water Resources Control Board
176. **TDF** – Through-Delta Facility
177. **TFE** – Tidal Freshwater Emergent

Acronyms and Abbreviations *(continued)*

178. **THM** – Trihalomethane
179. **TL** – Total Body Length
180. **TNS** – Townet Survey
181. **USBR** – United States Bureau of Reclamation
182. **USDA-ARS** – United States Department of Agriculture – Agricultural Research Service
183. **USFS** – United States Forest Service
184. **USFWS** – United States Fish and Wildlife Service
185. **VAMP** – Vernalis Adaptive Management Plan
186. **WARM** – Warm Freshwater Habitat (Basin Plan beneficial use)
187. **WHCP** – Water Hyacinth Control Program
188. **WILD** – Wildlife Habitat (Basin Plan beneficial use)
189. **WOE** – Weight-of-evidence
190. **WY** – Water Year
191. **X2** – The Line at which 2ppt (parts per thousand) Saline Occurs
192. **YOY** – Young of the Year.

Section 1

Introduction



1. Introduction

The purpose of this initiation package is to review the proposed Water Hyacinth Control Program (WHCP) to determine if this proposed action may affect any of the threatened, endangered, proposed, or sensitive species; and designated or proposed critical habitats listed herein. In addition, the following information is provided to comply with statutory requirements to use the best scientific and commercial information available when assessing risks posed to listed and/or proposed species; and designated and/or proposed critical habitat by proposed federal actions. This initiation package is prepared in accordance with legal requirements set forth under regulations implementing Section 7, of the Endangered Species Act (50 CFR 402; 16 U.S.C. 1536 (c)).

The WHCP was established in 1982 to control water hyacinth, an invasive aquatic weed, in the Sacramento-San Joaquin Delta (Delta) and its major tributaries. The WHCP is managed by the California Department of Boating and Waterways (DBW) with their federal partner, United States Department of Agriculture, Agricultural Research Service (USDA-ARS).

A. Threatened, Endangered, Proposed Threatened, or Proposed Endangered Species

USDA-ARS obtained a list of federal endangered and threatened species that occur in, or may be affected by projects in, the Sacramento/San Joaquin Delta on June 11, 2012, from the USFWS web page (http://www.fws.gov/sacramento). The list was current as of September 18, 2011.[1] The original list included 22 animal species, 13 plant species, and 19 critical habitats. USDA-ARS reviewed the list and identified those species that utilize waterways, channels, and immediate channel banks of the WHCP area. These 7 species are identified below. Species that do not occur in, or utilize waterways, channels, and channel banks of the Delta or its tributaries, are not considered in this biological assessment. These non-impacted species are identified in **Exhibit 1-1,** on page 1-3.

The following three listed and proposed species regulated by the United States Fish and Wildlife Service (USFWS) may be affected by the proposed action:

[1] As of September 14, 2012, the species identified on the June 11, 2012, list are current.

USFWS

1. Delta smelt *(Hypomesus transpacificus)*, **T**[2] [3]
2. Giant garter snake *(Thamnophis gigas)*, **T**
3. Valley elderberry longhorn beetle *(Desmocerus californicus dimporphus)*, **T.**

The following four listed and proposed species regulated by the National Oceanic and Atmospheric Administration's National Marine Fisheries Service (NMFS) may be affected by the proposed action:

NMFS

4. Sacramento River winter-run Chinook salmon *(Oncorhynchus tshawytscha)*, **E**
5. Central Valley spring-run Chinook salmon *(Oncorhynchus tshawytscha)*, **T**
6. Central Valley steelhead *(Oncorhynchus mykiss)*, **T**
7. Southern Distinct Population Segment (DPS) of North American green sturgeon *(Acipenser medirostris)*, **T.**

B. Candidate Species, Sensitive Species, and Species of Concern

There is currently one (1) candidate species, sensitive species, and species of concern that may be affected by the proposed WHCP action:

USFWS

1. Longfin smelt *(Spirinchus thaleichthys)*[4]

C. Critical Habitat

The WHCP action addressed within this document falls within the Critical Habitat for one (1) species regulated by USFWS, and four (4) species regulated by NMFS, as follows:

USFWS

1. Delta smelt *(Hypomesus transpacificus)*

NMFS

2. Sacramento River winter-run Chinook salmon *(Oncorhynchus tshawytscha)*
3. Central Valley spring-run Chinook salmon *(Oncorhynchus tshawytscha)*
4. Central Valley steelhead *(Oncorhynchus mykiss)*
5. Southern Distinct Population Segment (DPS) of North American green sturgeon *(Acipenser medirostris).*

[2] T = threatened species, E = endangered species

[3] On April 7, 2010, USFWS announced a 12-month finding that the reclassification of the delta smelt from threatened to endangered was warranted, but precluded by other higher-priority listing actions. USFWS will develop a proposed rule to reclassify delta smelt as their priorities allow (Federal Register, Volume 75, No. 66, April 7, 2010, page 17667).

[4] On April 2, 2012, USFWS released the results of a 12-month finding on the San Francisco Bay-Delta Distinct Population Segment (DPS) of longfin smelt. The finding was that this longfin smelt DPS warrants protection under the Endangered Species Act (ESA), but that USFWS is precluded at this time from drafting a formal listing rule by the need to address other higher priority species.

Exhibit 1-1

Listed Species and Critical Habitats that Occur in, or May Be Affected by Projects in, the Sacramento-San Joaquin Delta and Not Considered in This Biological Assessment

Invertebrates	*Apodemia mormo langei* – Lange's metalmark butterfly (E) *Branchinecta conservatio* – Conservancy fairy shrimp (E) *Branchinecta longiantenna* – longhorn fairy shrimp (E) *Branchinecta lynchi* – vernal pool fairy shrimp (T) *Elaphrus viridis* – delta green ground beetle (T) *Lepidurus packardi* – vernal pool tadpole shrimp (E)
Amphibians	*Ambystoma californiense* – California tiger salamander, central population (T) *Rana draytonii* – California red-legged frog (T)
Reptiles	*Masticophis lateralis euryxanthus* – Alameda whipsnake [=striped racer] (T)
Birds	*Rallus longirostris obsoletus* – California clapper rail (E) *Sternula antillarum (=Sterna, =albifrons) browni* – California least tern (E)
Mammals	*Neotoma fuscipes riparia* – riparian (San Joaquin Valley) woodrat (E) *Reithrodontomys raviventris* – salt marsh harvest mouse (E) *Sylvilagus bachmani riparius* – riparian brush rabbit (E) *Vulpes macrotis mutica* – San Joaquin kit fox (E)
Plants	*Amsinckia grandiflora* – large-flowered fiddleneck (E) *Castilleja campestris ssp. succulenta* – succulent (=fleshy) owl's-clover (T) *Cirsium hydrophilum var. hydrophilum* – Suisun thistle (E) *Cordylanthus mollis ssp. mollis* – soft bird's-beak (E) *Cordylanthus palmatus* – palmate-bracted bird's-beak (E) *Erysimum capitatum ssp. angustatum* – Contra Costa wallflower (E) *Lasthenia conjugens* – Contra Costa goldfields (E) *Neostapfia colusana* – Colusa grass (T) *Oenothera deltoides ssp. howellii* – Antioch Dunes evening-primrose (E) *Orcuttia tenuis* – slender Orcutt grass (T) *Orcuttia viscida* – Sacramento Orcutt grass (E) *Sidalcea keckii* – Keck's checker-mallow (=checkerbloom) (E) *Tuctoria mucronata* – Solano grass (=Crampton's tuctoria) (E)
Critical Habitats For:	Alameda whipsnake Antioch Dunes evening-primrose California tiger salamander, central population California red-legged frog (No critical habitat within the Delta) Colusa grass Conservancy fairy shrimp Contra Costa goldfields Contra Costa wallflower large-flowered fiddleneck longhorn fairy shrimp soft bird's-beak (proposed) Solano grass (=Crampton's tuctoria) Suisun thistle (proposed) vernal pool fairy shrimp vernal pool tadpole shrimp

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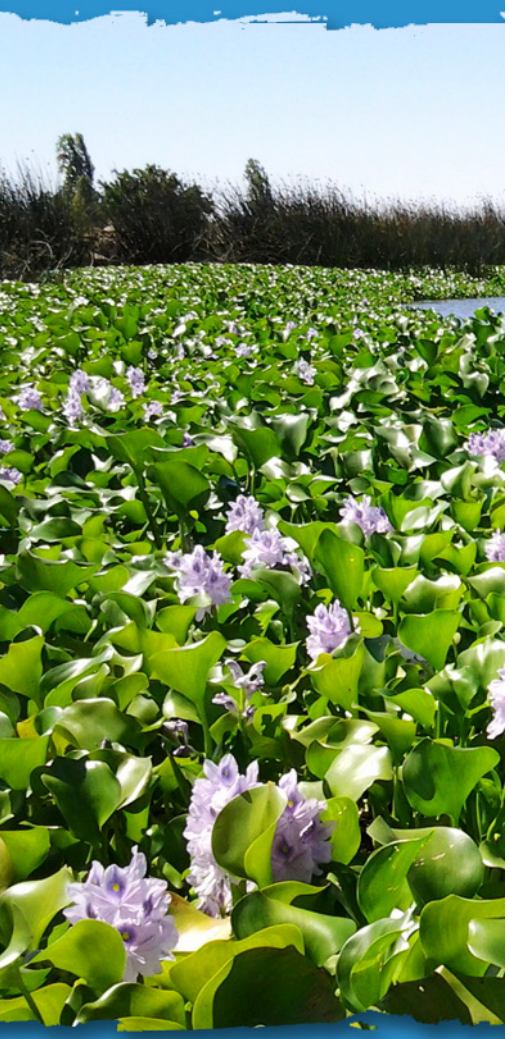
Section 2

Consultation to Date



2. Consultation to Date

This current consultation process for WHCP with USFWS and NMFS follows several prior formal and informal consultations and biological opinions (BOs). The WHCP has operated under BOs from USFWS and NMFS for over ten (10) years since 2001. Below, for each service, we describe the prior and current consultation history.

A. United States Fish and Wildlife Service (USFWS) Consultations to Date

In February, 2001, USDA-ARS initiated consultation with USFWS, submitting a Biological Assessment (BA) of WHCP. On June 1, 2001, USFWS issued a biological opinion (BO) for WHCP (1-1-01-F-0050). This BO was subsequently amended three (3) times (1-1-02-F-0157, 1-1-03-F-0114, and 1-1-04-F-0113).

In February 2004, USDA-ARS submitted a request to USFWS to reinitiate consultation after toxicological studies required in the June 1, 2001, BO had been completed. USDA-ARS requested that a new BO reflect this study information on the toxicity of the herbicides and adjuvants used in WHCP. The USFWS issued a new BO for WHCP in May 2004, that reduced the toxicological testing requirements of the program (1-1-04-F-0149). The WHCP operated under this 2004 BO through the 2010 treatment season.

In 2011, USFWS determined that changes in WHCP's program description and the changed status of delta smelt resulted in the need to re-initiate consultation for WHCP. In April 2011, USDA-ARS requested initiation of formal consultation with USFWS, and provided USFWS with program information.

In June 2011, USFWS determined that elements of the project description had changed enough since issuance of the 2004 BO that the information provided by USDA-ARS was not sufficient to initiate formal consultation. The USFWS provided USDA-ARS with comments and information needs. After discussions with USFWS, the service provided an extension to the 2004 BO for the 2011 treatment season, starting late, in September 2011, and extending longer than a normal treatment season, through the end of November 2011.

On January 18, 2012, USDA-ARS, USFWS, NMFS, and DBW met to discuss consultations for 2012, and later treatment seasons. The two services determined that given the short time before the preferred season start dates,

USDA-ARS and DBW should divide the consultation process into two packages, one for the 2012 treatment season, and a separate package for 2013, and beyond.

On March 5, 2012, USDA-ARS requested initiation of consultation for the 2012 treatment season with submission to USFWS of a package of information on the program. On March 19, 2012, USFWS responded to USDA-ARS, providing a copy of a June, 2011 letter, and requesting that USDA-ARS provide the additional requested information. USDA-ARS submitted a letter to USFWS on March 23, 2012, addressing issues listed in the June, 2011 letter.

USDA-ARS, DBW, USFWS, and NMFS met to discuss USDA-ARS's March 29, 2012 submission on March 30, 2012. At that time, USFWS stated that USDA-ARS should submit additional information requested by NMFS to both agencies to address USFWS's additional information needs. USDA-ARS submitted this information to USFWS, in the form of a biological evaluation, on April 26, 2012. This biological evaluation concluded that the WHCP may affect, but is not likely to adversely affect threatened and endangered species. USDA-ARS requested a concurrence letter from USFWS. On August 3, 2012, USFWS issued a formal response. USFWS concurred that WHCP may affect, but is not likely to adversely affect threatened valley elderberry longhorn beetle and/or its critical habitat. USFWS issued a biological opinion (81410-2011-F-035) on the effects of WHCP on delta smelt and its critical habitat and giant garter snake. The BO is provided in the WHCP Biological Assessment Supplemental Materials Binder.

B. National Oceanic and Atmospheric Administration – National Marine Fisheries Service (NMFS) Consultations to Date

In February, 2001, USDA-ARS initiated consultation with NMFS, submitting a Biological Assessment (BA) for WHCP. NMFS issued a BO for WHCP on June 8, 2001, with two amendments dated June 11, 2002, and August 11, 2003. These biological opinions respectively concluded that WHCP was not likely to jeopardize the continued existence of Sacramento River winter-run Chinook salmon *(Oncorhynchus tshawytscha)*, Central Valley spring-run Chinook salmon *(O. tshawytscha)*, and Central Valley steelhead *(O. mykiss)*, or adversely modify designated critical habitat for the 2001, 2002, and 2003 through 2005 application seasons.

In September, 2005, USDA-ARS requested initiation of formal consultation with NMFS for WHCP for the 2006 through 2010 application seasons. Following submission of additional materials, NMFS formally initiated consultation in November 2005, and released a BO for the 2006 through 2010 treatment seasons on April 4, 2006 (151422SWR2005SA00681:JSS).

In April 2011, USDA-ARS requested initiation of formal consultation with NMFS for WHCP for the 2011 treatment season. Following discussions between USDA-ARS, DBW, and NMFS, NMFS issued a BO on August 26, 2011 for the remainder of the 2011 treatment season. In October 2011, USDA-ARS requested a thirty-day extension to the 2011 WHCP BO. On November 3,

2011, NMFS granted an extension of the treatment season through November30, 2011.

On January 18, 2012, USDA-ARS, USFWS, NMFS, and DBW met to discuss consultations for 2012 and later treatment seasons. The agencies determined that given the short time before the preferred season start dates, USDA-ARS and DBW should divide the consultation process into two packages, one for the 2012 treatment season, and a separate package for 2013 and beyond. On March 5, 2012, USDA-ARS requested initiation of consultation for the 2012 treatment season with submission to NMFS of a package of information on the program. On March 23, 2012, NMFS responded to the USDA, requesting that USDA-ARS provide additional information. USDA-ARS submitted a letter to NMFS on March 30, 2012, addressing issues listed in the March 23, 2011 letter.

USDA-ARS, DBW, USFWS, and NMFS met to discuss USDA-ARS's March 29, 2012 submission on March 30, 2012. At that time, NMFS clarified additional information that USDA-ARS should submit. USDA-ARS submitted this information to NMFS, in the form of a biological evaluation, on April 26, 2012. The biological evaluation concluded that the WHCP may affect, but is not likely to adversely affect threatened and endangered species. USDA-ARS requested a concurrence letter from NMFS, which they received on July 12, 2012. The letter is provided in the Supplemental Materials Binder.

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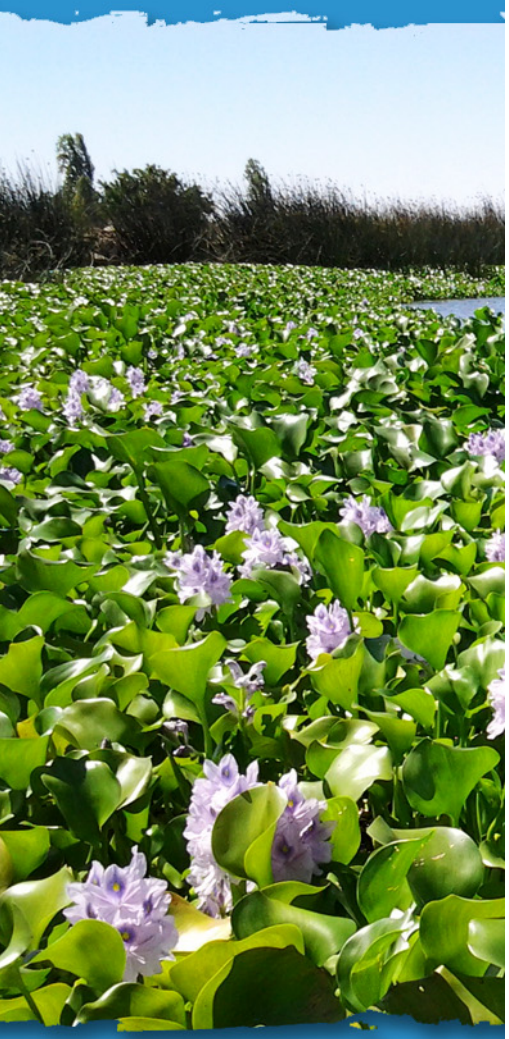
Section 3

Description of the Proposed Action



3. Description of the Proposed Action

The goal of WHCP is to keep waterways safe and navigable by controlling the growth and spread of water hyacinth in the Delta and its surrounding tributaries. Because of the continued survivability and persistence of water hyacinth in the Delta, WHCP legislative mandate is for control, rather than eradication of water hyacinth.

The WHCP seeks to minimize negative impacts of the invasive plant on navigation, public safety, recreation, agricultural activities, and ecosystem services in Delta waterways. The WHCP balances potential impacts of water hyacinth management by working to minimize non-target species impacts and to prevent environmental degradation in Delta waterways and tributaries.

This section of the Biological Assessment provides a detailed description of WHCP. The section is organized as follows:

A. Action Agency and Authority for WHCP
B. Overview and Purpose of WHCP
C. Action Area for WHCP
D. Timing of Activities for WHCP
E. Control Methods for WHCP
F. Estimated Efficacy for WHCP Methods
G. Monitoring Protocols for WHCP
H. Mitigation Measures for WHCP.

A. Action Agency and Authority for WHCP

USDA-ARS and DBW implement WHCP. The WHCP is an aquatic weed program designed to control the growth and spread of water hyacinth in the Delta and its tributaries. The USDA-ARS has served as the federal nexus for WHCP for the last fifteen (15) years, providing research and scientific expertise. USDA-ARS has provided technical and programmatic advice to WHCP for 28 years, since the program's inception.

The WHCP is a well-established program, which has been operating in the Delta for over 28 years. In 1982, in response to concerns about water hyacinth in the Delta, the California Legislature passed Senate Bill 1344 (Garamendi, Chapter 263, Statutes of 1982), designating DBW as the lead agency for controlling water hyacinth in the Delta, its tributaries, and Suisun Marsh. DBW established an interagency water hyacinth Task Force in the first years of WHCP to coordinate the control

activities of federal, state, and local interests. USDA-ARS was a member of this initial task force, and has been actively involved in WHCP since its inception in 1983.

The DBW operated WHCP between 1983 and 1999 with the guidance of a Water Hyacinth Task Force, treating between 166 and 2,743 acres per year. In 2000, DBW halted WHCP in response to legal action from *Delta Keepers*, claiming that DBW should obtain a NPDES permit under the 9th Circuit Court's Headwaters Inc. v. Talent Irrigation District decision. During 2000, DBW and USDA-ARS worked with state and federal regulatory agencies to obtain newly required biological opinions and an NPDES permit. DBW resumed treatment in 2001 under the guidance of new USFWS and NMFS biological opinions and a NPDES permit.

Between 2001 and 2006, WHCP operated under these original, and somewhat more restrictive, biological opinions, and the NPDES permit. Subsequent documents reflected the lower level of environmental impact demonstrated during the first five years of WHCP operation under the original biological opinions (2001 to 2005).

Between 2007 and 2011, WHCP operated under the following biological opinions and a statewide general NPDES permit for aquatic pesticide applications:

- NPDES Statewide General Permit (CAG990005) (will be replaced in November 2012 with new General Permit)
- USFWS Biological Opinion (1-1-02-F-157 and 1-1-03-F-0114) (valid through 2011)
- NOAA-NMFS Biological Opinion (151422SWR2005SA00681:JSS) (valid through 2011).

Photo: Water hyacinth.

During the 2012 treatment season, WHCP operated under the same NPDES Statewide General Permit, an USFWS biological opinion (81410-2011-F-035) issued August 3, 2012, and a NMFS letter of concurrence dated July 12, 2012.

In addition, WHCP now operates under a programmatic environmental impact report (PEIR) prepared by DBW in 2009. The 2009 PEIR added additional mitigation requirements for WHCP. DBW plans to amend the 2009 PEIR, as appropriate and prior to the 2013 treatment season, to reflect revised mitigation measures included in this Biological Assessment. USDA-ARS and DBW, through this biological assessment, are now seeking five-year biological opinions or letters of concurrence, for the 2013 through 2017, treatment seasons.

In addition to DBW treatments, Merced County began a treatment program for water hyacinth on the Merced and San Joaquin Rivers in Merced County in 1986, and in 1996 Fresno County began a similar program. Both counties entered into formal contracts with DBW, with DBW providing

funding, equipment, materials, and technical support. The counties operate under USDA-ARS/DBW biological opinions, PEIR, and NPDES permit.

There are no interrelated or independent actions associated with WHCP.

B. Overview and Purpose of WHCP

In order to provide context to proposed WHCP activities described in this Biological Assessment, this subsection begins by summarizing water hyacinth's invasion and spread in the United States and California, as well as summarizing WHCP activities over the last several years. Finally, this subsection describes the purpose of WHCP and provides an overview of proposed WHCP activities.

1. History of Water Hyacinth Invasion

Water hyacinth (*Eichhornia crassipes*) is a non-native, invasive, free-floating aquatic macrophyte. Aquatic macrophytes are plants that are large enough to be apparent to the naked eye; in other words they are larger than most algae.

Water hyacinth is often noted in the literature as one of the world's most problematic weeds (Gopal 1987, Cohen and Carlton 1995, Batcher 2000, Lancar and Krake 2002). Native to the Amazon region of South America, water hyacinth has spread to more than fifty countries on five continents. Water hyacinth creates significant problems in waterways and irrigation canals in Africa and Southeast Asia (Cohen and Carlton 1995, Lancar and Krake 2002).

Photo: Water hyacinth coverage.

Water hyacinth was introduced into the United States in 1884 at the Cotton States Exposition in New Orleans when display samples were distributed to visitors and extra plants were released into local waterways. By 1895, water hyacinth had spread across the Southeast and was growing in 40-km long mats that blocked navigation in the St. Johns River in Florida (Cohen and Carlton 1995).

The invasion of water hyacinth in California was slower than in the Southeast, probably due to water flows and the more temperate climate in the Delta (Toft 2000). Water hyacinth was first reported in 1904 in a Yolo County, California slough. It spread gradually for many decades, and was reported in Fresno and San Bernardino Counties in 1941, and in the Sacramento-San Joaquin Delta in the late 1940s and early 1950s. There were increased reports of water hyacinth in the Delta region during the 1970s, and by 1981, water hyacinth covered 1,000 acres of the Delta, and 150 miles of the 700 miles of Delta waterways (U.S. Army Corps of Engineers 1985).

Water hyacinth coverage estimates in the Delta since 1981 have ranged from less than

500 acres, up to approximately 2,500 acres. This wide range of annual water hyacinth acreage in the Delta is dependent on many factors including: acres treated, timing of treatments, winter air and water temperatures, summer air and water temperatures, water flows, and rainfall.

2. Summary of Prior WHCP Activities

The WHCP has been an adaptive integrated pest management program (IPM). WHCP activities have emphasized chemical treatment, supported by limited hand-picking, herding, mechanical removal, and evaluation of biological controls.

Selected primary program herbicides were 2,4-Dichlorophenoxyacetic acid, dimethylamine (DMA) salt, or 2,4-D) and glyphosate, with 2,4-D being used for the majority of treatments. Since the inception of WHCP in 1983, the program evolved to utilize less toxic herbicides and adjuvants. After toxicity testing demonstrated greater potential impacts on macroinvertebrates resulting from diquat, one of the original WHCP herbicides, DBW voluntarily stopped utilizing diquat. Similarly, WHCP discontinued use of four adjuvants (Placement, R-11, Bivert, and SurpHtac), utilizing only Agridex® between 2005 and 2012.

Figure 3-1, above, provides a summary of WHCP historical characteristics. The WHCP has been (and will continue to be) a relatively small aquatic weed control program concerned with managing the invasive, and non-native, water hyacinth in a large and complex Delta water environment.

Figure 3-1
Summary of WHCP Prior Activities

- Program Initiated: **1983**
- Annual Budget: **$5.3 to $6.0 million** (for WHCP and EDCP) (2010 to 2011)
- Total Staffing: **19** (for WHCP and EDCP, includes DBW and USDA-ARS) (2012)
- Monitoring Crews/Boats: **one to two 2-person crews** (for WHCP and EDCP) (2012)
- Average* Annual WHCP Sites Treated: **167** (range = 104 to 211 sites)
- Average* Annual WHCP Acres Treated: **815** (range = 421 to 1,137 acres)
- Average* Annual WHCP Treatments: **584** (range = 330 to 941 treatments)
- Average* Annual WHCP Acres per Treatment: **1.39** (range = 0.01 to 3.0 acres per single treatment)

* 2007 to 2011 averages and ranges per WHCP daily logs.

Historical treatment data provides an order of magnitude indication of likely chemical treatment levels in future years. **Table 3-1,** on the next page, summarizes the treatment types, number of sites, gallons used, pounds active ingredient, and acres treated from 2007 to 2011. A primary and unpredictable factor influencing treatment acres in any given year is the extent of water hyacinth infestation. In addition, as compared to the 2007 through 2011 data, future chemical use could decrease due to use of the new, lower volume, herbicides. Future chemical use could also be reduced by treating water hyacinth early in the treatment season. Future treatment acres and chemical use could increase as a result of higher infestation levels, and deployment of increased staff resources and/or improved staff utilization.

Table 3-1
Summary Data for WHCP – 2007 to 2011

			Gallons				Acres		
Year	Number of Sites	Number of Treatments	2,4-D	Glyphosate	Total Herbicide	Agridex	2,4-D	Glyphosate	Total Acres
1. 2007	211	941	938	149	1,087	441	938	199	1,137
2. 2008	146	439	336	64	400	163	336	85	421
3. 2009	177	492	619	64	683	266	619	86	705
4. 2010	199	719	879	109	988	372	879	145	1,024
5. 2011	104	330	449	253	702	286	449	338	787
Total	837	2,921	3,221	639	3,860	1,528	3,221	853	4,074
Average	167	584	644	128	772	306	644	171	815

	Averages			Pounds Active Herbicide Ingredient		
Year	Average Acres/Treatment	Average Herbicide Gallons/Site	Average Pounds/Site	2,4-D	Glyphosate	Total
1. 2007	1.21	5.15	19.90	3,752	447	4,199
2. 2008	0.96	2.74	10.52	1,344	192	1,536
3. 2009	1.43	3.86	15.07	2,476	192	2,668
4. 2010	1.42	4.96	19.31	3,516	327	3,843
5. 2011	2.38	6.75	24.57	1,796	759	2,555
Total	N/A	N/A	N/A	12,884	1,917	14,801
Average	1.39	4.61	17.68	2,577	383	2,960

In a typical year, DBW treated between 150 and 200 sites (the 2011 season started late, thus fewer sites were treated). On average, sites were treated three to four times per season; however, repeat treatments often involved treating different water hyacinth mats within the same treatment site due to the 3 acre treatment limit per site that was in the prior fish passage protocol developed in 2001.

During any treatment season, sites were treated between zero and six times. WHCP only treated those sites that had water hyacinth infestations, and treatments were also limited by time and resource constraints. Treatment sites within the Delta range from 6.5 acres to 1,707 acres in size, with an average of 219 acres. Thus, there were often several different locations within a site that required treatment. Only one approved herbicide was utilized for a given treatment site and time. Repeat treatments sometimes utilized a different herbicide, depending on conditions at the site.

In the 2010 treatment season, DBW treated 199 sites, out of 350 sites, and a total

of 1,448 acres between June 1st and October 15th, as follows:

- 1 treatment per site at 39 sites (20%)
- 2 treatments per site at 40 sites (21%)
- 3 treatments per site at 31 sites (15%)
- 4 treatments per site at 23 sites (11%)
- 5 treatments per site at 18 sites (9%)
- >5 treatments per site at 48 sites (24%).

On any given treatment day DBW historically treated, on average, between 5 and 16 acres (based on data from 2007 through 2011). Treatment acres per day were limited by: (1) the number of crews available; (2) travel time to reach the site; (3) time required to set-up, conduct monitoring, and treat a site; (4) the amount of water hyacinth growing at a particular site; (5) the fish passage protocol; (6) the herbicide label restrictions; and (7) weather and tide conditions.

The average acres per single treatment from 2007 to 2011 was 1.39, and the average gallons of herbicide utilized per site was 4.61. This was equivalent to just under 18 pounds of active ingredient per site (averaging 2,4-D and glyphosate).[1] These data demonstrate that only a small portion of each site was actually treated. The WHCP conducted spot treatments of water hyacinth mats, left buffer strips to protect fish, and did not treat areas beyond the plant mass.

In 2007 through 2011, DBW treated between 421 and 1,137 acres per year, equivalent to between 0.6 percent and 1.7 percent of the project area's 67,779 surface acres of water. The maximum water hyacinth acreage treated over the course of the program to-date was 2,770 acres in 2004, still only 4.1 percent of the project area water acres. The WHCP has only treated more than 2,700 acres in two seasons since 1983 (1994 and 2004).

In 2012, prior to the start of the season, WHCP proposed to treat a maximum of 2,496 acres of water hyacinth using the two approved herbicides, 2,4-D and glyphosate, consisting of 108 high priority sites and 164 medium priority sites; 197 of these sites were within the Delta, and 75 sites were in Stanislaus and Merced counties. DBW implemented a site selection prioritization process to select the high priority and medium priority sites. Summary proposed treatment data from 2012 are provided in **Table 3-2,** on the next page. The actual number of acres treated in 2012 will be determined at the end of the treatment season, and is likely to be less than 2,496 acres.

There is a long history of research and program development associated with the WHCP. Since 2001, USDA-ARS and DBW have conducted or sponsored a number of additional studies to evaluate treatment alternatives, efficacy, and identify new treatment options. Many of these additional studies were requested as part of previous USFWS or NMFS consultations. These studies, provided with this consultation submission in the Supplemental Materials Binder, include the following six reports:

[1] For purposes of comparison, the three new herbicides have application rates of 0.0875 to 0.25 pounds active ingredient per acre, as compared to the three to four pounds active ingredient levels for glyphosate and 2,4-D. Thus, utilizing these new herbicides would significantly reduce the chemical burden of WHCP.

Table 3-2
Summary of WHCP 2012 Treatment Acreage Pre-Season Estimates

High Priority Sites					
Location	Number of Sites	Estimated Treatment Acres	Total Water Acres[a]	Estimated Percent of Total Water Acres Treated*	Estimated Maximum[b] Treated Acres per Site
Delta (Northern Sites)	88	1,056	20,174	5.2%	12
Southern Sites	20	240	1,028	23.3%	12
Total High Priority	108	1,296	21,202	6.1%	
Medium Priority Sites					
Delta (Northern Sites)	109	1,090	23,284	4.7%	10
Southern Sites	55	110	1,879	5.8%	2
Total Medium Priority	164	1,200	25,163	4.8%	
Total Delta	197	2,146	61,619	3.5%	
Total Southern Sites	75	350	6,180	5.7%	
Total (All)	**272**	**2,496**	**67,799**	**3.7%**	

[a] Total water acres in legal Delta is 61,619 acres. Total water acres in the southern sites is 6,180. The sub-total acreages and percentages are the percent of water within the listed sites that will be treated. The Total Delta and Total Southern sites percent of acres treated are based on the total water acres in those regions

[b] The 12 acre maximum was proposed in 2012 only, due to the delayed start of treatments and the existing fish passage protocol. DBW will treat more acres per site in future years.

- *Acute Oral and Dermal Toxicity of Aquatic Herbicides and a Surfactant to Garter Snakes,* Robert C. Hosea, California Department of Fish and Game (2004)
- *Chronic Toxicities of Herbicides Used to Control Water Hyacinth and Brazilian Elodea on Neonate Cladoceran and Larval Fathead Minnow,* Frank Riley and Sandra Finlayson, California Department of Fish and Game (2004)
- *Acute Toxicities of Herbicides Used to Control Water Hyacinth and Brazilian Elodea on Larval Delta Smelt and Sacramento Splittail,* Frank Riley and Sandra Finlayson, California Department of Fish and Game (2004)
- *Ceriodaphnia dubia (water flea) Static Definitive Chronic Toxicity Test Data (7-day) for Exposure to Various Aquatic Herbicides,* California Department of Fish and Game, Aquatic Toxicology Laboratory (2003)
- Pogonichthys macrolepitdotus (Sacramento Splittail) Static Definitive Acute Toxicity Test Data (96-hour) for Exposure to Various Aquatic Herbicides, California Department of Fish and Game, Aquatic Toxicology Laboratory (2003)
- *Mapping Invasive Plant Species in the Sacramento-San Joaquin Delta Region Using Hyperspectral Imagery,* Susan L. Ustin, Ph.D., et al, Center for Spatial Technologies and Remote Sensing (CSTARS), California Space Institute Center of Excellence (CalSpace), UC Davis (2004).

Table 3-3
WHCP Objectives and Performance Measures

Objectives	Performance Measures
1. Limit future growth and spread of water hyacinth in the Delta 2. Improve boat and vessel navigation in the Delta 3. Utilize the most efficacious treatment methods available with the least environmental impacts 4. Prioritize sites so that WHCP activities are focused on sites with a high degree of infestation, as well as navigational, agricultural, environmental, recreational, or public safety importance 5. Employ a combination of control methods to allow maximum program flexibility 6. Improve WHCP as more information is available on appropriate control methods for the Delta 7. Monitor results of WHCP to fully understand its impacts on the environment 8. Improve shallow-water habitat for native species by controlling water hyacinth 9. Decrease WHCP control efforts over time, if sufficient efficacy of water hyacinth treatment is realized 10. Minimize use of control methods that could cause adverse environmental impacts.	■ Reduce total acres infested with water hyacinth ■ Reduce water hyacinth biomass at high priority navigation sites currently infested with water hyacinth ■ Reduce water hyacinth biomass at nursery sites ■ Prevent water hyacinth infestation of new sites ■ Produce fewer incidents of boat navigation, agricultural, recreation, and public safety incidents related to water hyacinth ■ Prepare reports for regulatory agencies and the public summarizing WHCP monitoring results ■ Minimize WHCP environmental impacts, as measured by compliance with program permits ■ Increase efficacy of WHCP, and of each control method over time ■ Increase the number of shallow-water sites suitable for native species ■ Limit the number of, and significance of, environmental impacts resulting from WHCP ■ Limit the number of WHCP acres treated with methods that have the potential for adverse environmental impacts ■ Reduce the quantity (tonnage) of herbicides applied into the Delta over time.

In addition to these special reports, each year USDA-ARS and DBW prepared an annual report for the WHCP. This annual report summarized infestation levels, treatment acreage and types, compliance with biological opinions and the NPDES permit, materials and methods, monitoring results, and daily treatment logs. The annual report fulfilled the reporting requirements of the biological opinions and NPDES permit. The WHCP 2011 Annual Report is provided in the Supplemental Materials Binder.

3. Purpose and Overview of Proposed WHCP

The WHCP will continue to be an adaptive and integrated pest management (IPM) program. The WHCP will utilize treatment protocols that balance the need to control water hyacinth with the need to minimize resulting environmental impacts to Delta waterways. The proposed program consists of an integrated and adaptive approach, emphasizing chemical treatment, supported by hand-picking, herding, mechanical removal, and continued assessment of biological controls, adjusting over time, as treatment methods, technology, and environmental factors change.

Table 3-3, above, identifies ten specific objectives for WHCP. Table 3-3 also identifies performance measures (i.e. expected outcomes) that the USDA-ARS and DBW will use to evaluate success of WHCP in meeting these project objectives.

Selected primary program herbicides will be 2,4-Dichlorophenoxyacetic acid, dimethylamine (DMA) salt, or 2,4-D) and glyphosate, with 2,4-D being used for the majority of treatments. Beginning in 2013, WHCP will add two new herbicides that have recently been approved by the California Department of Pesticide Regulation (CDPR) for water hyacinth treatment in aquatic environments: penoxsulam and imazamox. In addition, WHCP will add a third new herbicide, imazapyr, once it has been approved by CDPR for use on water hyacinth. All five herbicides are described in this Biological Assessment.

DBW applies herbicides with an adjuvant to increase adhesion to water hyacinth leaves. WHCP will utilize the adjuvant Agridex and the vegetable oil-based adjuvant, Competitor.

In addition to herbicide treatments, the WHCP proposes to utilize hand-picking, herding, and mechanical removal. These approaches can help reduce the need for herbicides.

Hand-picking will primarily be utilized to reduce plant biomass in nursery areas. Herding will be used in order to push water hyacinth mats (1) into main channels where it flows naturally out of the Delta and dies in the more saline water of San Francisco Bay; or (2) toward mechanical removal sites.

The WHCP proposes to utilize two mechanical removal methods: (1) use of specialized mechanical equipment with conveyors to physically remove plants, and (2) use of small excavators sited on concrete boat ramps to scoop plants into trucks/trailers for disposal. In addition, the USDA-ARS, DBW, and their partners will continue to evaluate the use of biological controls to reduce the spread of water hyacinth.

The DBW will utilize two-person crews to conduct WHCP treatments. The number of crews may be increased or decreased, depending on available resources and program needs. USDA-ARS and DBW propose a growth-based start-date approach that will minimize potential for impacts on fisheries and maximizes treatment efficacy. This approach will be dependent on fish survey data and field surveys for water hyacinth, within calendar-date windows, as described in Subsection D, below. Chemical treatments will likely begin by mid-March in selected areas. Chemical treatments in some regions of the Delta will continue through the end of November. The amount of herbicide applied in the project area to control water hyacinth can be minimized by treating plants early in the growing season before plants have grown into large mono-specific mats characteristic of this species. Early treatment will also minimize the negative ecosystem impacts of this invasive species.

The DBW crews will conduct hand-picking and herding activities. The WHCP proposes to contract mechanical removal work to qualified private companies. USDA-ARS, the California Department of Food and Agriculture (CDFA), and/or University of California, Davis (UCD) will continue to conduct biological control programs.

The legally defined WHCP region is divided into approximately 350 treatment sites that average between one and two miles in length. These sites may be treated multiple times during a treatment season.

Treatment sites will be prioritized so that nursery areas, and areas where water hyacinth causes negative navigational, agricultural, public safety, environmental, or industrial impacts, are treated first. The WHCP will also consider logistical and operational factors such as prevailing winds, travel time, and weather conditions when selecting treatment locations.

The WHCP will follow an Operations Management Plan that specifies a pre-application planning protocol; an Application/Monitoring Coordination Protocol; "Best Maintenance Practices" for Handling Herbicides; Spray Equipment Maintenance and Calibration; and an Herbicide Spill Contingency Plan. The Operations Management Plan will include requirements related to avoiding threatened or endangered species; conducing habitat evaluation; dissolved oxygen measurement; a fish passage protocol; and other program monitoring requirements. The Operations Management Plan is currently being updated by DBW management, and will be provided to USFWS and NMFS when completed.

Based on NPDES permit requirements, DBW will follow an Annual Monitoring Protocol. This protocol will fulfill monitoring requirements of the Central Valley Regional Water Quality Control Board, NMFS, and USFWS. The State Water Resources Control Board (SWRCB) is updating the NPDES General Permit, with a draft for public comment released on June 27, 2012, and a final version for Board approval expected in November 2012. DBW will revise their monitoring protocol to reflect the new conditions in the final General Permit.

C. Action Area for WHCP

The WHCP action area can be defined at several different levels. First, the overall project area is defined in statute. Within the legislatively-defined project area, WHCP is divided into approximately 350 treatment sites. Only waterways within any given treatment site are actually part of the action area, and in any given treatment season, water hyacinth is growing, and treated in, only a portion of the 350 total treatment sites. The WHCP proposes specified avoidance protocols that may also limit treatments within a given treatment site.

1. Project Area Specified in Enabling Legislation

The project area for WHCP is specified in statute, as follows: "the delta[i], its tributaries, and the marsh" (Harbors and Navigation Code Section 64). The State of California legal definition of the Sacramento-San Joaquin Delta (Delta) includes six counties (San Joaquin, Yolo, Sacramento, Solano, Contra Costa, and Alameda).

The WHCP includes eleven (11) counties (including the six "Delta" counties) that encompass much of the Sacramento-San Joaquin Delta and its upland tributaries. The eleven counties are: (1) Alameda, (2) Contra Costa, (3) Fresno, (4) Madera, (5) Merced, (6) Sacramento, (7) San Joaquin, (8) Solano, (9) Stanislaus, (10) Tuolumne, and (11) Yolo.

The general boundaries for the treatment area in the Delta and its tributaries are as follows:

- West up to, and including, Sherman Island, at the confluence of the Sacramento and San Joaquin Rivers;

- West up to the Sacramento Northern Railroad, to include water bodies north of the southern confluence of the Sacramento River and Sacramento River Deep Water Ship Channel;
- North to the northern confluence of the Sacramento River and Sacramento River Deep Water Ship Channel, plus waters within Lake Natoma;
- South along the San Joaquin River to Mendota, just east of Fresno;
- East along the San Joaquin River to Friant Dam on Millerton Lake;
- East along the Tuolumne River to LaGrange Reservoir below Don Pedro Reservoir; and
- East along the Merced River to Merced Falls, below Lake McClure.

The project area is contained within the following fifty-one (51) United States Geological Service (USGS) quadrants: (1) Antioch North, (2) Rio Vista, (3) Jersey Island, (4) Isleton, (5) Bouldin Island, (6) Clifton Court Forebay, (7) Thornton, (8) Terminous, (9) Holt, (10) Union Island, (11) Lodi North, (12) Lodi South, (13) Stockton West, (14) Lathrop, (15) Woodward Island, (16) Courtland, (17) Gravelly Ford, (18) Mendota Dam, (19) Folsom, (20) Yosemite Lake, (21) Gustine, (22) Stevinson, (23) San Luis Ranch, (24) Turner Ranch, (25) Santa Rita Bridge, (26) Poso Farm, (27) Friant, (28) Lanes Bridge, (29) Vernalis, (30) Ripon, (31) Riverbank, (32) Waterford, (33) Paulsell, (34) Cooperstown, (35) La Grange, (36) Westley, (37) Brush Lake, (38) Ceres, (39) Denair, (40) Turlock Lake, (41) Snelling, (42) Merced Falls, (43) Crows Landing, (44) Hatch, (45) Turlock, (46) Cressey, (47) Winton, (48) Biola, (49) Herndon, (50) Firebaugh, and (51) Fresno North.

Within WHCP project area, there are approximately 350 treatment sites that average between one and two miles in length. The total number of treatment sites may be further defined and refined by WHCP to reflect jurisdictional and operational factors. The primary purpose of these defined treatment sites is to facilitate planning and reporting of WHCP activities. **Exhibit 3-1,** beginning on the next page, provides a map of the WHCP treatment area and current numbered treatment sites. **Section 7** and the Supplemental Materials Binder provide a spreadsheet identifying each current treatment site, county, acres, previous treatment history, and 2012 prioritization criteria scoring.

In any given year, WHCP will treat only a portion of the total treatment sites. Multiple treatments within a treatment site may be necessary because many sites in the Delta cannot be treated during the ideal early-growth phase due to the potential presence of listed fish species. In addition, some larger sites may have more water hyacinth than can be treated at one time in order to reduce dissolved oxygen (DO) impacts). These sites will be treated in more than one application.

Following the prioritization and site selection criteria described below, USDA-ARS and DBW will identify likely treatment sites and acres prior to each treatment season, and provide a list of these sites to USFWS and NMFS. Based on the extent of water hyacinth infestation, only a portion of any given site may be treated to comply with herbicide label requirements. The WHCP will conduct spot treatments of water hyacinth mats, and will not treat beyond the

Exhibit 3-1
Northern Sites Map

Page 1 of 2

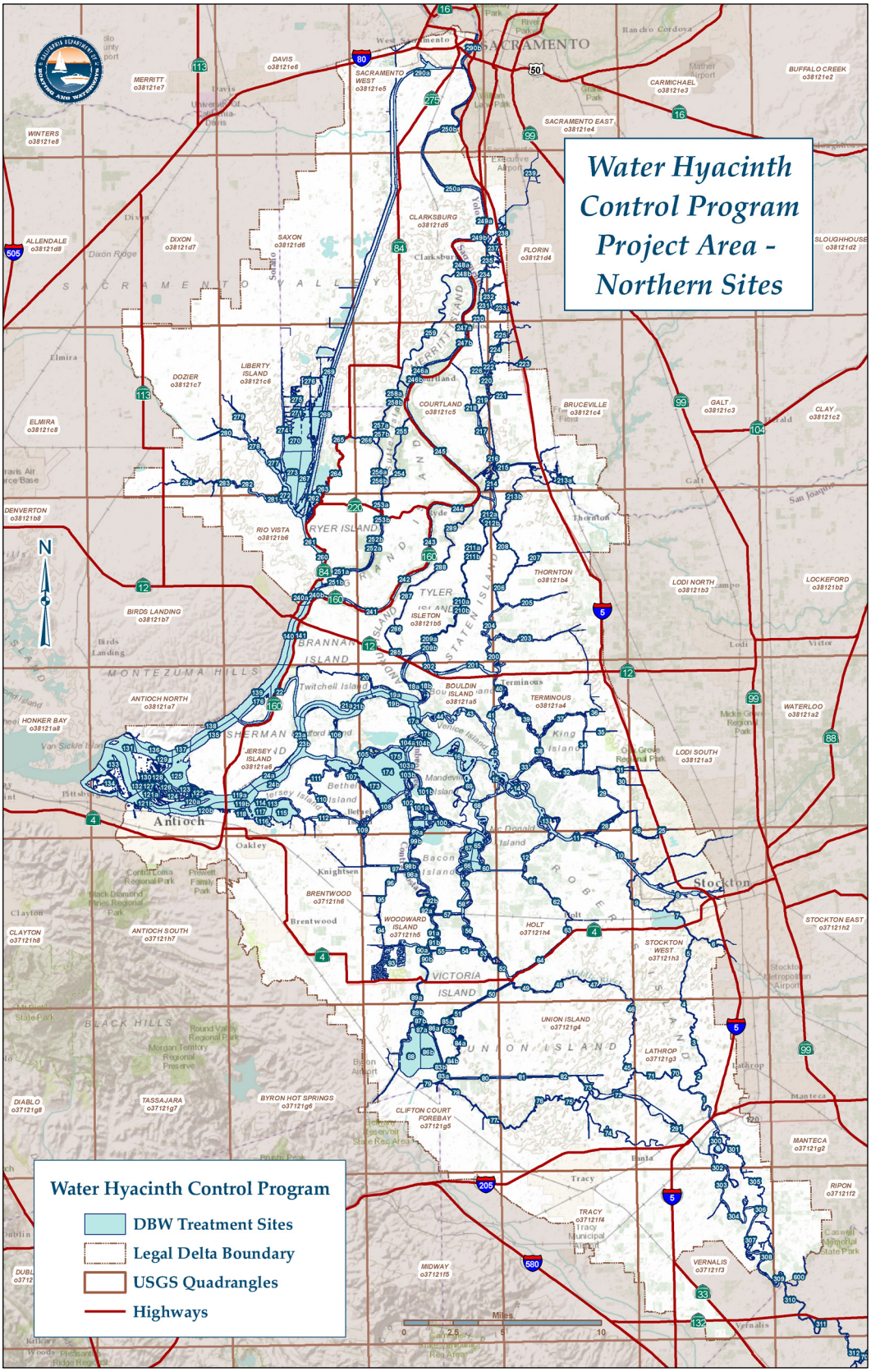
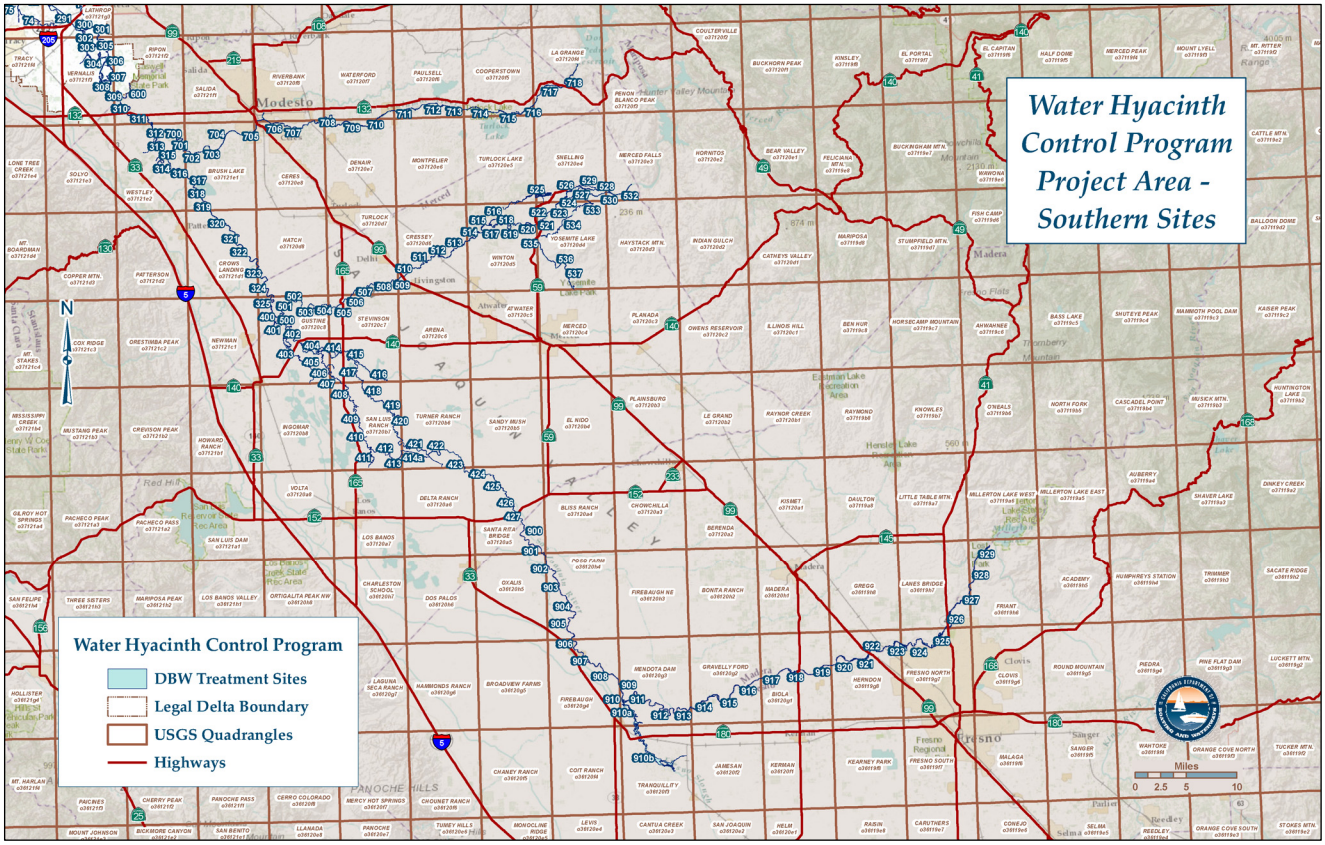


Exhibit 3-1
Southern Sites Map



NOTE: Large-scale versions of the map Exhibits in this Biological Assessment are provided in Tab 20 of the Supplemental Materials Binder.

plant mass. In addition, USDA-ARS and DBW will provide the services with copies of the Notice of Intent (NOIs) before each treatment week. The NOIs identify likely treatment sites for each treatment crew for the following week's application.

Because water hyacinth constantly migrates daily throughout the Delta with winds, tides, and water flow, some treatment sites identified at the beginning of the season may not need to be treated, and there may be additional sites that were not identified at the beginning of the season that do need to be treated. In no case will the program treat water hyacinth beyond the defined treatment sites within the project area described above and specified in statute.

2. Prioritization of Treatment Sites and Methods

Prior to the start of each treatment season, DBW and USDA-ARS will prioritize treatment sites and methods. The prioritization process will be based on results of pre-season field surveys combined with the experience and knowledge of water hyacinth growth patterns of the treatment crews and program environmental scientists.

During pre-season field surveys, treatment crews will survey each treatment site and identify total acres infested. This pre-season infestation figure is only one indicator, as water hyacinth is dormant during the winter, and typically dies back in cold weather. To prioritize sites, experienced treatment crew members, the field supervisor, and environmental scientists will review each site and rank sites on several factors, including: (1) whether or not the site is a nursery area, (2) current infestation levels, (3) potential for infestation, and (4) whether the site is important for navigation, public safety, recreation, and/or commercial use. Sites will be scored on each of these factors, the team will calculate a total priority score for each site, and prepare an initial priority ranking. The DBW may employ aerial surveys or other appropriate remote sensing methods to assist in site prioritization as well as follow-up evaluation. Staff will present the priority ranking to DBW management and USDA-ARS, who will then evaluate and approve a treatment plan for the season.

This initial plan will indicate the general priority for site treatment. The plan may shift during the treatment season, as water hyacinth moves throughout the Delta, and may grow more rapidly in certain areas. Treatment crews will continue to monitor and record total acres infested, by site, throughout the treatment season, in order to provide management with information they need to focus treatments to high priority sites. Wind and weather conditions may also dictate when a particular site will be treated. In addition, treatment crews will return to sites for additional treatments during the season when field surveys indicate presence of persistent or new infestations.

Using the initial prioritization and management plan as a starting point, each field crew will prioritize their assigned sites weekly via a field survey of their area. Based on the management plan, the field supervisor will determine weekly and daily spraying needs and assign crews to sites based on wind, weather, tides, travel times, available

personnel, and equipment resources. The field supervisor will ensure that Notice of Intent requirements are met.

Prior to each treatment week, the field supervisor will report the treatment sites to the respective County Agricultural Commissioner. Prioritized sites are likely to change rapidly depending on the constant growth and movement of water hyacinth, as well as wind and weather conditions.

The WHCP will include hand-picking as necessary. The WHCP will include herding methods when high water flows can push water hyacinth out of the Delta, or to assist with mechanical removal. Herding will occur at locations and times when weather and water conditions are appropriate. The WHCP will include mechanical methods to remove dense mats of water hyacinth in locations where chemical treatment is precluded, for example in sites with large numbers of valley elderberry shrubs on the shoreline.

3. Avoidance Areas within Project Action Area

Within WHCP action area, and in addition to the prioritization process described above, WHCP proposes an additional layer of site selection based on presence of listed species. The intent of these avoidance actions is to minimize the opportunity for treatments to occur when a listed species is present in, or near, a particular site.

The DBW will provide treatment crews with a field guide (Species Identification Deck) for easy identification of special-status species on-site. Prior to treating a site, crews will conduct a visual survey to determine whether special status plants, animals, or sensitive habitats are present. Crews will complete an Environmental Observation Survey for each site to document presence or absence of special status species. If special status species or sensitive habitats are present at the site, field crew will not perform any chemical treatment. Historically, once or twice within a treatment season, crews identify a potential listed species (for example, a snake that could be a giant garter snake). A copy of the Environmental Observation Survey is provided in the Supplemental Materials Binder.

The WHCP will implement a number of additional measures to avoid the potential for impacts on listed species. WHCP will seek to adjust the timing of treatments to avoid periods when juvenile steelhead and salmon may be present. The WHCP will base treatment dates, in part, on Interagency Ecology Program (IEP) monitoring data showing that the salmon pulse has migrated through the system.

DBW environmental scientists will consult the IEP database each week to determine whether salmon are present in any sites scheduled to be treated in the following week. If salmon are present, or likely to be present, based on IEP and NMFS surveys and analyses, then DBW will remove the site from the weekly treatment list until such time as salmon are not likely to be present.

The WHCP will implement delta smelt avoidance measures. Similar to avoidance measures for salmon, WHCP environmental scientists will review IEP monitoring data to avoid treating sites where delta smelt are known to be currently present.

To further minimize potential to impact delta smelt, WHCP will not begin treatments in areas likely to be used as spawning and rearing habitat for delta smelt until after July 1st. These include North Delta sites at Cache Slough, Liberty Island, and Lindsay Slough (sites 262, 267, 272, 277), and sites at the confluence of the Sacramento River and San Joaquin River (Sherman Lake sites 121a, 121b, and 122 to 131).[2] **Exhibit 7-6** provides a map illustrating these avoidance sites.

To avoid potential impacts to the valley elderberry longhorn beetle, the DBW will conduct a survey of treatment sites to prepare a map that identifies locations of *Sambucus ssp*. (elderberry shrub), and provide this map to field crews. **Exhibit 3-2,** starting on the next page, provides the most recent version of a map showing valley elderberry shrub locations and giant garter snake habitat valuations.

DBW crews will maintain a 50 foot buffer between treatment sites and shoreline elderberry shrubs, and will conduct treatments downwind of elderberry shrubs. Given WHCP treatment protocol to treat only when winds are less than 10 mph (7 mph in Contra Costa County) and to utilize a coarse droplet size, a 50 foot buffer provides an extra margin of safety for valley elderberry shrubs. There are several treatment sites with a large number elderberry shrubs along the waterway, potentially limiting chemical treatment. For these sites, WHCP may utilize mechanical methods. Currently numbered treatment sites with relatively large numbers of valley elderberry shrubs include: 10-11, 46, 47, 48, 99, 234, 511, 529, 707, 708, and 710.

At the same time that DBW conducts its survey of valley elderberry within treatment sites, DBW environmental scientists will conduct a survey to evaluate the value of habitat within treatment sites for giant garter snake. Habitat valuations include six habitat value levels: (1) no, (2) low, (3) low-moderate, (4) moderate, (5) moderate-high, and (6) high. DBW will create a map that identifies the habitat valuations (see Exhibit 3-2), as well as identifies locations of giant garter snake sitings. Treatment sites with historically high quality giant garter snake habitat include: 16, 17, 19, 28, 32, 47, 63, 75, 76, 115, 121, 122, 125, 215, 221, and 223. Sites where giant garter snakes have been seen in the past include: 15, 36, 119, 225, 237, 246, 275, and 410.

[2] Based on historical data, WHCP has conducted relatively few treatments in these sites. Between 2007 and 2011, these sites were only treated between July and October. Over the five-year period at the four North Delta sites, WHCP treated 31.7 acres out of 2,463.7 water acres (1.3 percent). At the twelve Sherman Lake sites, WHCP treated 101.6 out of 4,118.4 water acres (2.5 percent) over five years.

Exhibit 3-2

Valley Elderberry Shrub Locations and Giant Garter Snake Habitat Valuation – Northern Sites

Page 1 of 2

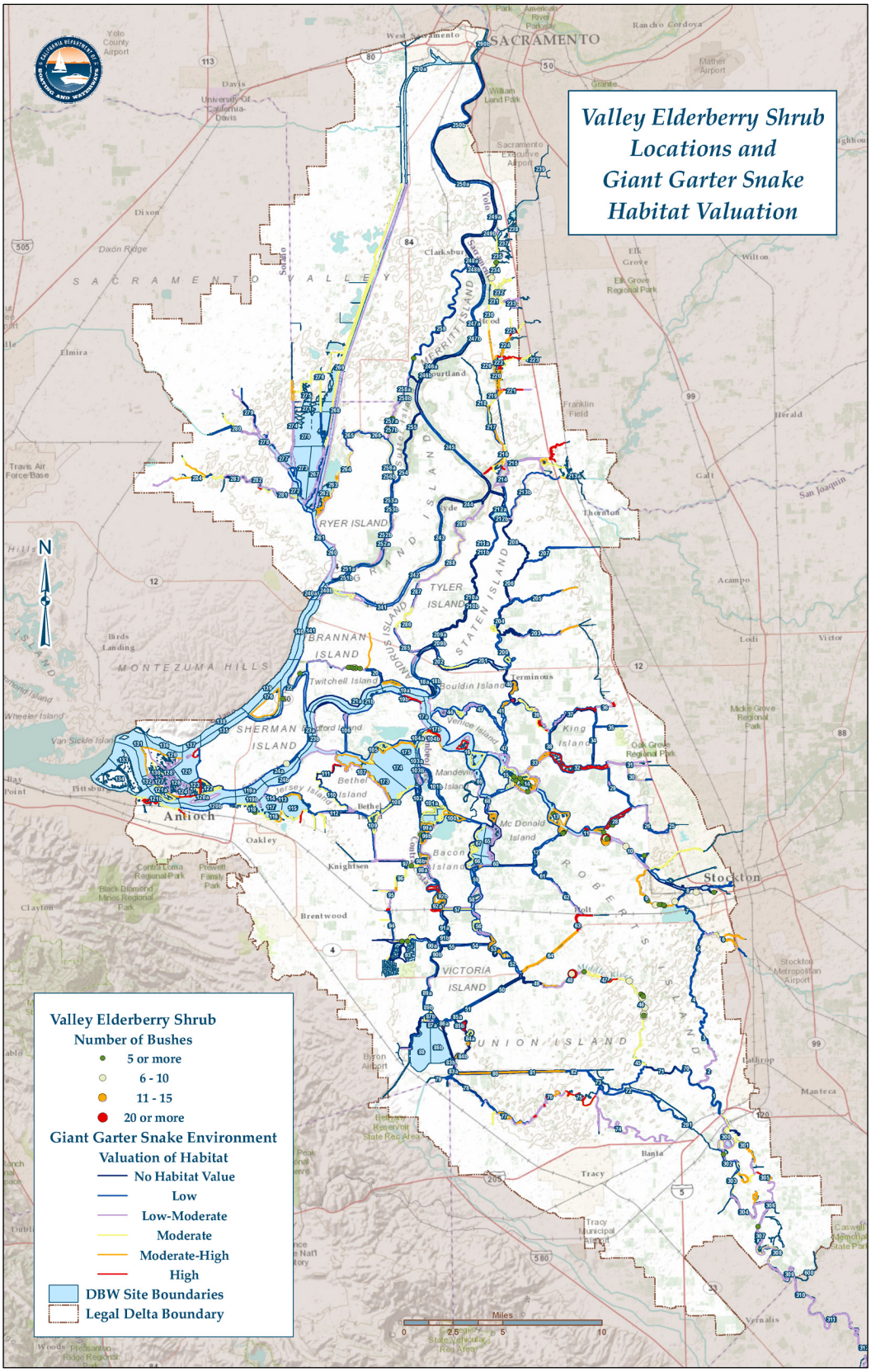
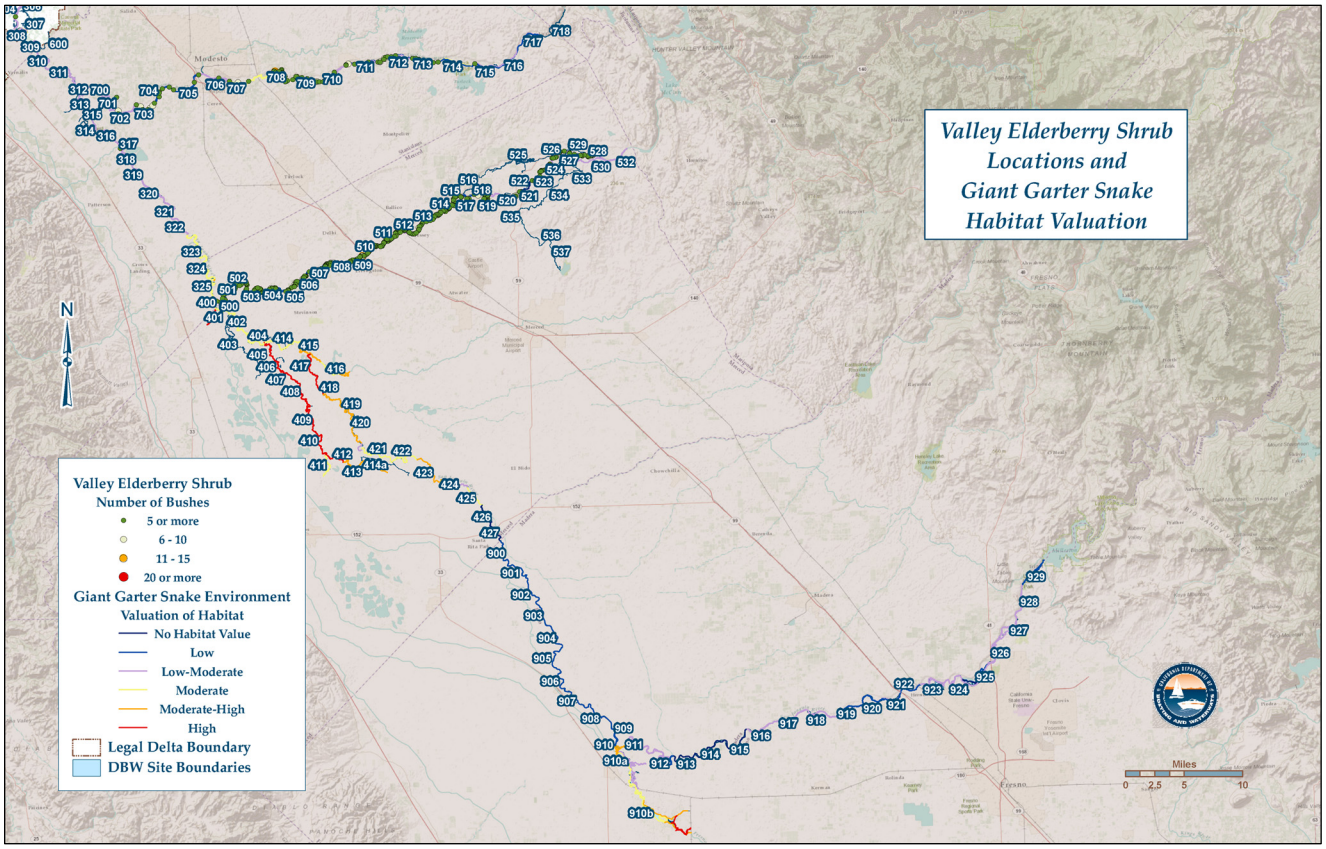


Exhibit 3-2

Valley Elderberry Shrub Locations and Giant Garter Snake Habitat Valuation – Southern Sites *(continued)*

Page 2 of 2



NOTE: Large-scale versions of the map Exhibits in this Biological Assessment are provided in Tab 20 of the Supplemental Materials Binder.

Figure 3-2
Proposed Calendar of Treatment Activities for WHCP

Activity	2012											
	JAN	FEB	MAR[a]	APR	MAY	JUNE	JULY	AUG	SEP	OCT	NOV	DEC
Environmental Surveys												
Surveyor Training												
Coverage Surveys												
2,4-D Treatment[b]												
Glyphosate Treatment												
Penoxsulam Treatment[c]												
Imazamox Treatment[d]												
Imazapyr Treatment[e] (After approval)												
Herbicide Monitoring												
Hand-picking												
Herding												
Mechanical Removal												
Reporting												

[a] The March start date for herbicide treatment would be dependent on temperature and fish surveys
[b] 2,4-D may be used in the legal Delta between June 15th and September 15th, and in southern sites between July 15th and August 15th
[c] Penoxsulam was approved by CDPR for aquatic use in 2009, and will be utilized in the WHCP starting in 2013
[d] Imazamox was approved by CDPR for aquatic use in 2012, and will be utilized in the WHCP starting in 2013
[e] Imazapyr will not be utilized until approved by CDPR for water hyacinth.

D. Timing of Activities of WHCP

Figure 3-2, above, provides a proposed schedule of WHCP treatment activities. For 2013 and beyond, WHCP proposes a start-date approach that utilizes a combination of calendar-dates, field surveys of water hyacinth to evaluate plant growth, and IEP surveys to determine presence of special status fish species. The objective of this approach is to improve WHCP chemical treatment efficacy without negatively impacting special status fish species. Seasonal temperature fluctuations in the Delta impact both water hyacinth growth and migratory fish activity. These weather fluctuations can become relatively extreme, and may make calendar-based start dates less relevant.

Treatment start dates linked only to calendar dates may not necessarily reflect presence or absence of migratory special status fish species. These species migrate through the Delta under conditions of cooler water temperatures. At the same time, water hyacinth growth increases as temperatures rise, particularly in back-water and dead end slough areas where fish are not likely to be present. As air and water temperatures rise during a season (and more particularly at a given site), migratory fish are more likely to move through the Delta earlier, and at the same time water hyacinth will be in its rapid growth phase, when chemical treatments are most effective. In these situations, treating water hyacinth early (as long as listed fish are not present) can help to

maximize effectiveness of chemical treatments. In addition, these early treatments can reduce the spread of water hyacinth in the Delta, leading to a reduction in the total amount of herbicide necessary during the treatment season. This reduction will, in turn, reduce WHCP resources, boat use, and labor.

The 2011 NMFS biological opinion for USEPA registration of 2,4-D (for Pacific salmonids) limits 2,4-D applications within the legal Delta to between June 15th through September 15th, and between July 15th and August 15th in the southern sites (NMFS 2011). Thus, any chemical treatments prior to, or following, the three-month 2,4-D application period will utilize other approved herbicides.

The WHCP start-dates will be determined as outlined below. Generally, this proposed approach will allow WHCP to treat new water hyacinth growth in any approved sites, starting in approximately March of each year. In addition, any sites that have not been approved for early treatments will open up for treatment based on the previously determined calendar dates:

1. The WHCP will begin regular field surveys in known nursery areas (focusing on back-water and dead end locations) in late-February of each season
2. When field surveys show contiguous areas of more than 100 square feet of re-growing water hyacinth (seen as re-greening of winter stunted plants), crews will photograph the sites and document the locations
3. The WHCP will report these locations to USFWS and NMFS, and consult the IEP database to determine whether listed fish species are present. If listed fish species are not present, and USFWS and NMFS concur, treatments will immediately start in these specific approved treatment sites
4. The WHCP will continue conducting field surveys and reporting re-growing water hyacinth to USFWS, NMFS, and IEP until the calendar start dates for particular sites have been reached. Prior to the calendar start dates, sites that show re-growth of over 100 square feet will be evaluated for presence of listed fish, and immediately treated (with USFWS and NMFS approval)
5. When sites have not already been pre-approved and treated (per above), WHCP will maintain the historical April 1st and April 15th start dates in those sites where listed fish are not likely to be present; and the May 15th and July 1st start dates in other areas (see Figure 3-3 and related maps)
6. If IEP data shows that listed fish are not likely to be present at Delta sites, WHCP may begin chemical treatments in those sites
7. If IEP data shows that fish are likely to be present at Delta sites, WHCP will not begin chemical treatments, but will continue to work with IEP to determine when listed fish are not present, and when treatments may begin.

Figure 3-3, on the next page, summarizes proposed WHCP start and end dates. Maps illustrating treatment dates are listed in Section 7 of this biological assessment and provided in the Supplemental Materials Binder. This flexible approach to treatment start dates has the potential to improve WHCP efficacy and reduce chemicals in the Delta. Both of these factors will provide long-term benefits to listed species in the Delta.

Figure 3-3
Summary of Proposed WHCP Chemical Treatment Dates

Proposed WHCP Treatment Dates (2013-2017)		Maps
March 1 to July 1*	Specific approved high priority nursery sites, TBD	
April 1 to November 30	Sites where listed fish are not present (203, 205-207, 214-217, 219, 221-226, 230, 233-234, 236-239, 400-427, 900-929)	Exhibits 7-7 and 7-8
April 15 to November 30	Sites once South Delta barriers in place (45-49, 70-78, 82)	Exhibit 7-7
May 15 to November 30	River sites once temperature is 68.5 F for one week (310-325, 500-537, 700-718)	Exhibit 7-8
June 1 or July 1 to November 30	All remaining sites (July 1 if salmon are present)	Exhibit 7-7
June 15 to September 15	Only dates 2,4-D may be applied in Legal Delta; no 2,4-D applied north of Highway 12	Exhibit 7-4
July 15 to August 15	Only dates 2,4-D may be applied in all other sites	Exhibit 7-5

* WHCP will implement a survey-based approach to conducting treatments that allows for early season treatments in areas with re-growing water hyacinth when listed fish species are not present.

Hand-picking and herding operations will occur as necessary. Mechanical removal will take place primarily when, and where, water hyacinth mats have achieved significant mass, depending on location and site parameters. Mechanical removal of small localized mats may occur as necessary.

On any given treatment day, actual start of treatments depends on the distance from DBW's boat dock to the treatment site. Field crews begin their work day at 6:30 am, thus treatment activities generally occur in mid-morning, and again in early-afternoon.

E. Control Methods for WHCP

The WHCP applies Integrated Pest Management (IPM) and Maintenance Control Practices (MCP). IPM denotes the coordinated use of available control methods for a particular pest. The State of California defines IPM as: a pest management strategy that focuses on long-term prevention or suppression of pest problems through a combination of techniques such as monitoring for pest presence and establishing treatment threshold levels, using non-chemical practices to make the habitat less conducive to pest development, improving sanitation, and employing mechanical and physical controls. Herbicides that pose the least possible hazard and are effective in a manner that minimizes risks to people, property, and the environment, are used only after careful monitoring indicates they are needed according to pre-established guidelines and treatment thresholds.

MCP refers to practices that minimize plant biomass through regular, low-level, control treatments applied at times during a plant's

life cycle when treatments are most effective. Herbicide treatments are most effective when plants are rapidly growing. **Figure 3-4,** on the next page, illustrates typical aquatic macrophyte growth phases. The shaded area illustrates the most effective treatment period. Ideally, under a maintenance control program, the acres of water hyacinth required to be treated could be reduced each year.

The WHCP will continue to follow IPM and MCP. The WHCP balances IPM and MCP approaches in order to simultaneously reduce impacts and increase effectiveness. For example, in order to avoid impacts to migrating special status fish, treatments may occur as early in the growing season as possible, but later in a plant's lifecycle than would be ideal.

The WHCP follows an adaptive management approach in which DBW will seek to improve efficacy and reduce environmental impacts over time, as new and better information is available about the program. Within their adaptive management approach, WHCP will:

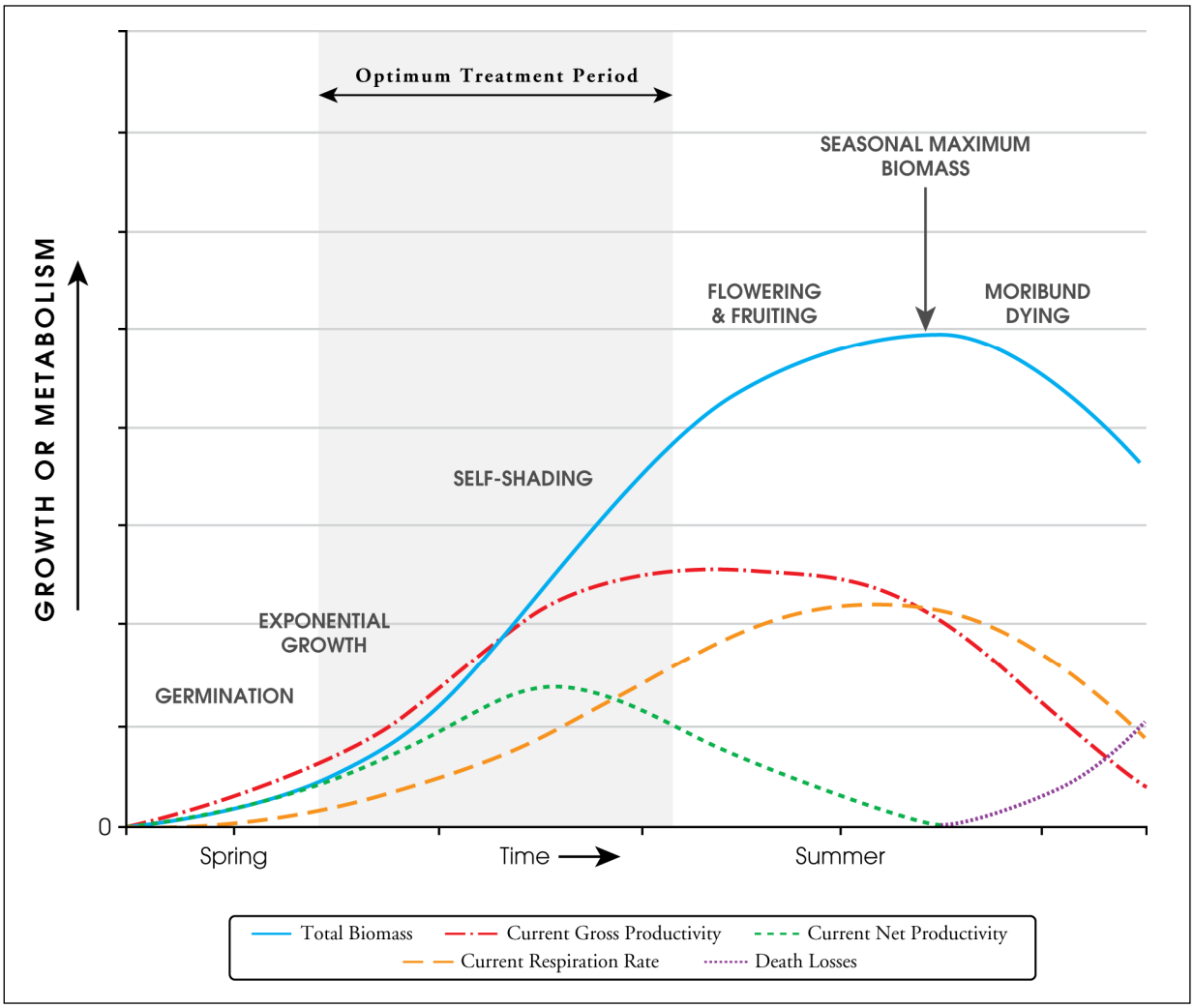
- Evaluate the need for control measures on a site-by-site basis
- Follow NPDES general permit pre- and post-treatment monitoring protocols and evaluate data to determine environmental impacts
- Support ongoing research to explore the impacts of WHCP and alternative control methodologies, including biological controls and herbicides and adjuvants with reduced environmental impacts
- Report findings from monitoring evaluations and research to regulatory agencies and stakeholders.

The WHCP will utilize four herbicides in 2013, and will add a fifth herbicide once it receives CDPR approval for use on water hyacinth. The four herbicides are 2,4-D, glyphosate, penoxsulam, and imazamox. Penoxsulam and imazamox are newer, potentially less toxic herbicides. Imazapyr, described in this biological assessment, is also a lower-toxicity herbicide that will be added once it receives CDPR approval for water hyacinth. As alternatives to chemical treatment, the WHCP will also utilize hand-picking, herding, mechanical removal, and evaluation of biological controls.

To minimize potential environmental impacts, WHCP will select the most appropriate control methods for a given site in the Delta based on the season and that site's conditions. The WHCP will also monitor results of the WHCP, and base future control methods on these results. The selected treatment alternative will be chosen to provide the greatest reduction in water hyacinth biomass while avoiding or minimizing environmental impacts. The WHCP will adjust program actions, as necessary, in response to recommendations and evaluations by regulatory agencies and stakeholders.

The WHCP emphasizes chemical treatment, with limited handpicking and herding, mechanical removal, and continued assessment of biological controls. Selected herbicides are 2,4-D, glyphosate, penoxsulam, and imazamox. WHCP may incorporate one additional new herbicide: imazapyr. All herbicides will be applied with an adjuvant, either Agridex or Competitor.

Figure 3-4
Generalized Growth and Metabolic Patterns for a Typical Annual Aquatic Macrophyte



Source: After Westlake, 1965b, in Aquatic Weed School 2012, Anderson.

The WHCP will include handpicking as necessary. The WHCP will include herding, in some cases to support mechanical removal, and in other cases to move obstructive water hyacinth mats. Herding will be conducted when weather and water conditions are appropriate. The WHCP will include mechanical methods to remove dense mats of water hyacinth in locations where chemical treatment is precluded and/or mechanical removal is likely to be more successful.

The DBW, USDA-ARS, University of California, Davis, and the California Department of Food and Agriculture (CDFA) will also evaluate viable biological control methods for water hyacinth. These research efforts currently focus on a study of plant hoppers.

For each particular season and treatment site, WHCP will evaluate characteristics of the site and select the most appropriate treatment option(s). Prior to the start of each treatment season, WHCP will provide USFWS and NMFS

with a treatment plan that outlines and prioritizes likely treatment sites, acres, and treatment methods. The WHCP will provide updated plans during the season, in the likely event that natural movement of water hyacinth within the Delta changes treatment site priorities.

1. General Program Activities

There will be a number of management activities within WHCP that support the program. USDA-ARS staffing for the WHCP and EDCP will include a managing supervisor, administrative support, and scientific staff. Within DBW, employees that work directly on the WHCP and EDCP will include a manager, a senior environmental scientist, field environmental scientists, a field supervisor, a GIS mapping specialist, and field crew members. DBW may add or reduce staff to support program needs over time. The WHCP also receives management and administrative support from within DBW.

Prior to the start of each treatment season, DBW will conduct environmental awareness training for all field crew members. The training includes: species identification and impact avoidance guidelines; protocol for identification and protection of valley elderberry shrubs; protocol for identification and protection of delta smelt, Chinook salmon, steelhead, green sturgeon, and associated protected habitats; and protocol for take of protected species. In addition, field crew members also will be trained on use and calibration of spray equipment and the WHCP Operations Management Plan.

The WHCP will implement pre- and post-season surveys to identify locations and coverage of water hyacinth, and supplement these formal surveys with mid-season evaluations of water hyacinth coverage. Starting in February, and again in October and November, field crews will conduct visual surveys of all treatment sites. For each site, crews will record the extent of water hyacinth coverage (acres and percent coverage), and status of water hyacinth at the site.

In the early season survey, field crews will identify problem areas such as those with the greatest impact on navigation, public safety, nursery areas, and sites close to pumps or other structures. Treatment crews will also identify crops adjacent to treatment sites in order to help select the appropriate herbicide for treatment. Crews will validate field survey information with data from the prioritization process and note any changes. This survey information will be used to help prioritize treatment locations at the start of the treatment season, and to measure efficacy of water hyacinth treatments at the end of the season.

During the treatment season, as crews are working throughout the Delta, they will continue to monitor and record water hyacinth coverage, by site. This ongoing survey will assist the management team in identifying mid-season adjustments to prioritizing treatment sites and determining treatment effectiveness.

Each year USDA-ARS and DBW will prepare an annual report for the WHCP. This annual report will summarize infestation levels, treatment acreage and types, compliance with biological opinions and the NPDES permit, materials and methods, monitoring results, and daily treatment logs. The annual report will fulfill the reporting requirements of the federal agencies and NPDES permit.

Photo: Mid-season evaluation of treatments at Stockton Marina.

Photo: Water hyacinth spraying

2. Herbicide Treatments

The WHCP proposes to utilize five different herbicide active ingredients: 2,4-D, glyphosate, penoxsulam, imazamox, and imazapyr. Two of these herbicides, 2,4-D and glyphosate, have been used since the program's inception. The remaining three herbicides will be new to the program. These three herbicides have received United States Environmental Protection Agency (USEPA) approval for water hyacinth. Currently, two of the herbicides, penoxsulam and imazamox, have been approved by CDPR for water hyacinth treatment in California. The third herbicide, imazapyr, has been approved for aquatic use in California by CDPR, but not specifically for water hyacinth. The WHCP will not utilize imazapyr until it receives CDPR approval, potentially within the next two years. All five herbicides are included in the program description and biological impact assessment in the event that they will be incorporated into WHCP. The new herbicides are intended to have low toxicity profiles, and would thus reduce the potential for negative impacts.

There are several reasons why WHCP is adding new herbicides to the program. First, new lower-toxicity profile herbicides have the potential to reduce the environmental impact of WHCP. Second, new herbicides may reduce the amount of herbicide applied to Delta waterways to treat water hyacinth. Third, timing and crop restrictions currently limit the application of 2,4-D, which has been the primary and most effective WHCP herbicide. Thus, expanding the number of herbicides beyond 2,4-D and glyphosate expands treatment options. Fourth, utilizing herbicides with varying modes of action reduces the potential for target species to develop resistance. While there are no indications of water hyacinth resistance to date, some terrestrial species of weeds have developed resistance to glyphosate (Powles 2008) or acetolactate synthase (ALS) inhibitors (Wisconsin Department of Natural Resources 2012), and the aquatic weed hydrilla may develop resistance to fluridone (Richardson 2008).

Resistance is an important consideration in use of any herbicide over a long period of time. In terrestrial applications, some plants have become resistant to glyphosate or the ALS inhibitors after many (over ten) years of use.

Resistance is not necessarily the same across terrestrial and aquatic plants, and generally is species specific. However, because WHCP is a long-term control program, it will be prudent to increase the portfolio of herbicide active ingredients and of non-herbicide treatment options in order to reduce the potential for resistance. Rotating treatments after several years among herbicides with different modes of action reduces the potential for a plant to develop resistance. USDA-ARS, WHCP environmental scientists and Pest Control Advisors will evaluate water hyacinth response to program herbicides over time to identify potential resistance problems.

The two new WHCP herbicides (penoxsulam and imazamox) are part of the USEPA's Office of Pesticide Program's Conventional Reduced Risk Program. This program expedites the review and regulatory decision-making process of conventional pesticides that pose less risk to human health and the environment than existing conventional alternatives (Washington DOE 2012). Pesticides are typically included in the reduced risk program because they have advantages over existing pesticides such as low impact on human health, lower toxicity to non-target organisms, low potential for groundwater contamination, lower use rates, low pest resistance potential, and/or compatibility with integrated pest management practices.

Crews will conduct treatments with hand-held sprayers applied from aluminum airboats or aluminum outboard motor boats. The work boats will be equipped with direct metering of herbicides, adjuvants, and water pump systems. The crews will spray the chemical mixture directly onto the plants utilizing pump-driven hand-held spray nozzles. The pump will mix calibrated amounts of herbicide, adjuvant, and water. The WHCP will apply the chemicals at the herbicide label-specified rates. Treatment crews will follow specific requirements, as described, to account for wind, dissolved oxygen, drinking water intakes, agricultural intakes, and total acres treated. Treatment crews will follow all label requirements, and implement a new fish passage protocol to ensure that migratory fish are not impacted by the WHCP.

DBW and USDA-ARS developed a fish passage protocol in 2001 that was based, in part, on herbicide labels at that time. Since 2001, program knowledge has increased, program herbicides have changed, and herbicide labels have been revised to be less restrictive. As a result, DBW and USDA-ARS have developed a new fish passage protocol that is based on knowledge of dissolved oxygen data, current herbicide label requirements, and the herbicides that will be used in the program today. This new fish passage protocol is provided in the Supplemental Materials Binder. Prior to implementing the new fish passage protocol, DBW will amend the 2009 WHCP PEIR to reflect the new requirements. The 2009 WHCP PEIR incorporated the 2001 fish passage protocol, and thus DBW will seek to amend the document prior to the start of the 2013 treatment season.

The amount of herbicide used and number of acres treated in a given year can reflect the magnitude of infestation. However, there are several other factors that will affect the amount of treatment that WHCP conducts (regulatory limits, local water conditions, weather, staff levels, etc.).

Photo: Water hyacinth spraying.

Herbicide use in future years is impacted by weather conditions. A high rainfall winter could potentially result in significant increases in water hyacinth in the following season. This is because riverbeds and shorelines exposed by drought conditions in prior years act as nursery areas. When nursery areas become inundated again after heavy rains, water hyacinth seeds germinate, and new plants move downriver into the Delta. In other cases, heavy storms can push water hyacinth plants through the Delta and into saline waters, where these plants die. High spring and summer temperatures increase water hyacinth growth, increasing infestation levels, and the amount of herbicide required.

The ideal herbicide treatment time for water hyacinth is when the plant is in the early growth phases, between 5 percent and 25 percent of maximum size (Spencer and Ksander 2005). In much of the Delta, this has historically occurred between early May and the end of June (Spencer and Ksander 2005); however, early growth in quiet nursery waters may occur as early as mid-March (Anderson, 2012). Treating water hyacinth during the early growth phase will increase herbicide efficacy and reduce the total amount of herbicide required, in addition to reducing program resource needs. In recent years, water hyacinth treatments in the early season have been limited due to the potential for presence of listed species. The proposed WHCP timing approach will help optimize the balance between improved herbicide efficacy and presence of listed species.

WHCP will only treat those sites that have water hyacinth infestations, treating only the water hyacinth plants within those sites. WHCP may also be limited by time and resource constraints. Within a given treatment location, WHCP will treat according to current herbicide label requirements to limit potential for decaying plants to result in low dissolved oxygen levels. **Table 3-4,** on the next page, summarizes current requirements related to dissolved oxygen and number of treatments. In Table 3-4, number of treatments refers to repeat treatments of a given mat of water hyacinth and the untreated strips between. These restrictions do not apply to different water hyacinth mats within a larger WHCP-defined numbered treatment site.

Treatment sites within the Delta range from 6.5 acres to 1,707 acres in size, with an average of 219 acres. Thus, there may be several different water hyacinth infestations spread out within a site that require treatment. In these cases, WHCP will treat all water hyacinth mats in the site as time and resources allow. Repeat treatments may utilize a different herbicide, depending on conditions at the site.

Table 3-4
Summary of Herbicide Label Requirements Related to Dissolved Oxygen and Repeat Treatments

Herbicide	Dissolved Oxygen Requirements	Number of Treatments	Time Between Treatments
2,4-D	It may be appropriate to treat only part of the infestation at one time. For example, apply the product in lanes separated by untreated strips that can be treated after the vegetation in treated lanes has disintegrated (2-3 weeks in growing season). Begin treatment along the shore and move outward in bands to allow fish to move into untreated areas.	Two applications per season	21 days between applications
Glyphosate	When infestations require treatment of the total surface area of impounded water*, treating the area in strips may avoid oxygen depletion due to decaying vegetation.	May require retreatment	24 hours between applications
Penoxsulam	None	Not specified	Not specified
Imazamox	None	Up to 4 applications per season at 32 ounces per acre application rate	Not specified
Imazapyr	When infestations require treatment of the total surface area of impounded water*, treating the area in strips may avoid oxygen depletion due to decaying vegetation. Do not treat more than one-half of the surface area of the water in a single operation. Begin treatment along the shore and move outward in bands to allow fish to move into untreated areas.	Up to 3 applications per season at 32 ounces per acre application rate	10 to 14 days between treatments

* The WHCP project area encompasses tidal and riverine waters, not impounded waters.

WHCP will follow these guidelines when determining whether a given mat of water hyacinth will be treated again.

a. DBW will treat a location once if, after the herbicide has had time to take effect, the initial treatment was effective in killing the majority of water hyacinth plants at that site.

b. DBW will treat a given water hyacinth mat a second time if buffer strips for fish passage were left untreated. In this case, DBW will return to treat the remainder of the site after the specified time between treatments (per herbicide label requirements). In this case, DBW is treating new plants within a given water hyacinth mat, not the previously treated plants.

c. DBW will treat previously treated water hyacinth plants a second time in a given site if the first treatment was not effective in killing the plants. In this case, DBW will not conduct the second treatment until the specified time period, per label requirements.

d. The actual number of locations and numbered treatment sites that will be treated more than once depends on factors such as herbicide efficacy, growth of the water hyacinth plants and tidal movement that cannot be easily predicted. WHCP will seek to

minimize the number of times that a given water hyacinth mat will be treated, and will follow herbicide labels regarding total number of applications allowed.

On any given treatment day, treatment acres per day are limited by: (1) the number of crews available; (2) travel time to reach the site; (3) time required to set-up, conduct monitoring, and treat a site; (4) the amount of water hyacinth growing at a particular site; (5) the herbicide label restrictions; (6) fish passage protocols; and (7) weather and tide conditions.

Prior treatment acres provide an order of magnitude indication of future treatment acres. Allowing for earlier herbicide treatments could reduce the number of acres requiring treatment, while warm weather and good growing conditions could increase the number of acres requiring treatment. Improvements in WHCP resource allocation and efficiency could allow treatment crews to treat more acres, if necessary.

In 2007 through 2011, DBW treated between 421 and 1,137 acres per year, equivalent to between 0.6 percent and 1.7 percent of the project areas 67,779 surface acres of water. The maximum water hyacinth acreage treated over the course of the program to-date was 2,770 acres in 2004, still only 4.1 percent of the project area water acres. The WHCP has only treated more than 2,700 acres in two seasons since 1983 (1994 and 2004).

The remainder of this subsection describes the mode of action; chemical characteristics; environmental fate; application rates and frequency; label requirements; and concentrations in water for each of the four approved and one pending WHCP herbicides, and for the two adjuvants, used in conjunction with WHCP herbicides. **Exhibit 3-3,** on the next page, provides a comparison of WHCP herbicides.

The WHCP does not have access to the compounds of the inert ingredients in herbicides, as explained by USEPA (http://www.epa.gov/opprd001/inerts/inertsdisclosure.html):

> "USEPA does not currently identify inert ingredients on pesticide labels. Pesticide manufacturers often claim as confidential the identities of inert ingredients in their products. Federal confidentiality regulations (40 CRF part 2, subpart B) require USEPA to protect information claimed as confidential by companies. One exception is when USEPA provides inert ingredient information to medical professionals treating persons in connection with exposure to a pesticide (FIFRA § 12(a)(2)(D)).
>
> USEPA requires registrants to identify to the Agency all ingredients in their pesticide products. A challenge for inerts disclosure by registrants is that their pesticides may include proprietary products whose contents are held confidential by the manufacturer. EPA knows the composition of those products, but does not disclose it to registrants."

Photo: Boat operator logging location on computer tablet prior to starting treatment at Stockton Marina.

Exhibit 3-3
Summary Comparison of Current and New WHCP Treatment Herbicides

	2,4-D	Glyphosate	Penoxsulam	Imazamox	Imazapyr
Status	CDPR approved In use	CDPR approved In use	CDPR approved New	CPDR approved New	CDPR approved (not for water hyacinth) Future use
Application Rate	64 to 128 ounces/acre 1.9 to 3.8 lb. a.i./acre	96 ounces/acre 3 lb. a.i./acre	2 to 5.6 ounces/acre 0.03125 to 0.0875 lb. a.i./acre	16 to 64 ounces/acre 0.125 to 0.50 lb. a.i./acre	16 to 32 ounces/acre 0.125 to 0.25 lb. a.i./acre
Calculated Concentration in 1 Meter Deep Water with 20% Overspray	85 ppb	67 ppb	2 ppb	11.2 ppb	5.6 ppb
NPDES Maximum Limitation in Receiving Waters	70 ppb	700 ppb	10.1 ppm	To Be Determined	11.2 ppm
USEPA Fish Toxicity Classification	Practically non-toxic	Slightly toxic to practically non-toxic	Practically non-toxic	Practically non-toxic	Practically non-toxic
Pros	Proven effective; lower cost; selective broadleaf herbicide	Proven effective	Requires less herbicide; lower toxicity; good WH control in studies and excellent results in Florida operations; less DO impact; low cost per acre	Requires less herbicide; lower toxicity; good WH control in studies; less DO impact; relatively fast acting (same browning time as glyphosate); quick drying; no irrigation restrictions	Requires less herbicide; lower toxicity; good WH control in studies
Cons	Limited application period; can't be used near grapes, tomatoes; higher concentrations required than new herbicides	Slower acting than 2,4-D; binds to sediment; higher concentrations required than new herbicides; non-selective; increased cases of terrestrial weed resistance	Potential for groundwater pollution, although low potential at application rates; 1ppb irrigation water restriction	No NPDES receiving water maximum limitation yet determined due to recent CPDR approval	Not as selective as penoxsulam and imazamox; toxic to woody plants; greater potential for off-target plant impacts

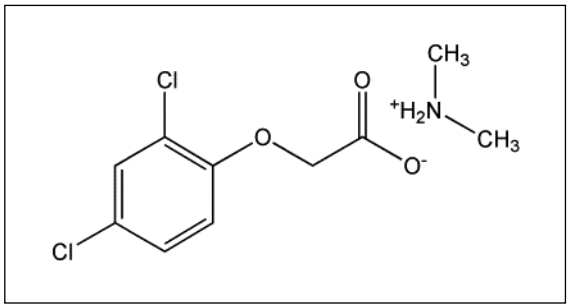
California Department of Pesticide Regulation (CDPR) follows a similar policy regarding inert ingredients. However, any toxic ingredients in a pesticide formulation must be identified on Material Safety Data Sheets (MSDS) for the product; none of the WHCP herbicides identify toxic inert ingredients. MSDSs for each of the WHCP herbicides and adjuvants are provided in the Supplemental Materials Binder.

2,4-D

2,4-Dichlorophenoxyacetic acid, dimethylamine (DMA) salt, or 2,4-D is a post-emergent systemic herbicide active ingredient specific to broadleaf plants and is most effective in plants with a large enough leaf area to absorb sufficient quantities. The chemical structure of 2,4-D is illustrated in **Figure 3-5,** above. This herbicide has been the primary WHCP treatment option since the program's inception in 1983, and has been used in the United States since the 1940s. The WHCP has utilized different 2,4-D products, most recently Weedar® 64. This herbicide active ingredient mimics the plant hormone auxin, causing rapid cell division and abnormal growth. Injuries to plants include impacts to growth and reproduction, with symptoms occurring immediately and plant death taking several weeks (NMFS, 2011). 2,4-D can be absorbed by both foliage and roots.

2,4-D is water soluble and chemically stable. The organic carbon sorption coefficient, K_{oc} of 2,4-Dimethylamine salt is between 72 and 136, indicating weak adsorption (Gervais, et al. 2008).

Figure 3-5
2,4-D Chemical Structure



Decomposition of herbicides in water depends on a number of characteristics, including: water quality, sediments in the water, temperature, and chemical properties of the herbicide. A review of 34 research papers concerning the persistence of 2,4-D in water under both laboratory and field conditions concluded that (1) under laboratory conditions, 2,4-D in water decomposed in periods of hours to days; and (2) under some warm water field conditions, 2,4-D has consistently been shown to be reduced to non-detectable levels in closed water bodies in approximately one month; and (3) persistence of 2,4-D at extremely low levels may be encouraged by water movements in lakes, reservoirs, and streams (Gren 1983).

The chemical 2,4-D breaks down due to photodecomposition or by algal or bacterial decomposition (ESA/Madrone 1984). Westerdahl et al., (1983) found that the disappearance of 2,4-D in aquaria containing both plants and hydrosoil, and only hydrosoil, suggested that macrophytes, algae, fungi, and organic debris were the most likely sinks for 2,4-D. The aqueous half-life of 2,4-D (time in which one-half of the material is degraded) in a set of pools was

10 to 11 days. In a study with natural waters, 2,4-D half-life ranged from 0.5 to 6.6 days (HSDB 2001). Walters (1999) reported an aqueous photolysis half-life for 2,4-D, at 25C, of 13.0 days, and an aqueous aerobic half-life of 15.0 days.

Results of prior WHCP follow-up monitoring typically showed declining 2,4-D concentrations (often to non-detectable levels) between two and seven days after treatment. Breakdown products of 2,4-D detected in laboratory experiments included 1,2,4-benzenetriol, 2,4-dichlorophenol (2,4-DCP), 2,4-dichloroanisole (2,4-DCA), 4-chlorophenol, chlorohydroquinone (CHQ), volatile organics, bound residues, and carbon dioxide. These degradates are expected to be of low occurrence in the environment and of low toxicity, or both (Gervais et al. 2008).

For treating water hyacinth, 2, 4-D will applied at a rate of between two and four quarts per acre, per label specifications. This is equivalent to 1.9 to 3.8 pounds of active ingredient per acre. It will be applied using a broadcast spray method.

For the majority of sites treated with 2,4-D, it will be preferable to conduct spot treatments directly onto water hyacinth leaves. For sites that are heavily vegetated, buffer strips will be created and another treatment will occur, if needed, after the treated vegetation has decayed. Treatment crews may return to a site to spray locations within a site that were not previously treated, or to retreat regrowth in previously treated plants only after plants killed in the initial treatment have decayed or floated away, no sooner than 21 days.

Table 3-5
Calculated* Maximum Concentrations of 2,4-D, Immediately Following WHCP Treatment

Concentration of:	2,4-D *(Active Ingredient)*
1. Chemical directly out of spray nozzle	2,300 ppm
2. Chemical in 1 meter deep water, @ 100% water contact	0.43 ppm
3. Chemical in 2 meter deep water, @ 100% water contact	0.21 ppm
4. Chemical in 1 meter deep water, @ 20% water contact	85 ppb
5. Chemical in 2 meter deep water, @ 20% water contact	43 ppb

* The concentrations above are based on the pounds of active ingredient in maximum specified application rate per acre, and an appropriate dilution factor based on the volume of water in the tank mix, or within one or two meter-acres.

Label requirements for treating water hyacinth with 2,4-D include a number of specifications to reduce potential for drift, contamination of water, and low dissolved oxygen following treatment. These include: leaving buffer strips when treating large areas; delaying retreatment for 21 days; no spraying when wind is greater than 10 miles per hour; and delaying the use of treated water for irrigation or domestic purposes for three weeks or until 2,4-D is no more than 0.1 ppm (Nufarm 2006).

Table 3-5, above, summarizes expected instantaneous concentrations of 2,4-D at the spray nozzle, and in the water. Table 3-5 provides conservative estimates assuming that 100 percent of the herbicide reaches one or two meters deep of water, and a more realistic estimate assuming 20 percent of the herbicide reaches one or two meters deep of water. Early WHCP tests by Anderson (1982), found that

only 10 to 20 percent of 2,4-D moved through the water hyacinth mat and into the water, thus 20 percent water contact is conservative. The amount of 2,4-D applied in the project area to control water hyacinth, and the resulting glyphosate concentrations in Delta waters, can be minimized by treating plants early in the growing season before plants have grown into the large mono-specific mats characteristic of this species. Early treatment will also minimize the negative ecosystem impacts of this invasive species.

The calculated maximum concentrations in Table 3-5 reflect potential chemical concentrations immediately after (or during) spraying. In reality, mixing could occur through the entire depth of water at the site, and tidal movement and through water Delta flow dilute herbicides even further. The Delta is not a stationary water environment, thus, the concentration of herbicide immediately after treatment is not stable, but rather readily dilutes (in addition to degradation pathways). There are two tidal cycles in the Delta every day, with typical water fluctuations of three to five feet in each cycle. In addition, the Delta functions in a complex hydrological system consisting of inflows from rivers and reservoirs, Delta exports, and tidal fluctuations. Approximately 30 km^3 of freshwater enter the Delta (and then San Francisco Bay) annually, with peak flows in early March (Knowles 2000). Freshwater inflows and Delta exports are the major influences of salinity in the Delta. Illustrating the movement of water within the Delta, the X2 salinity line (distance of the near-bottom 2 practical salinity units (psu) isohaline line from the Golden Gate) varies by up to 30 km during the course of a year (Knowles 2000).

Historical water quality monitoring data demonstrates that actual 2,4-D concentrations decrease rapidly in the Delta following treatment. Water samples taken downstream of the treatment site at two to three feet depth one-hour post treatment show actual herbicide levels that are at least an order of magnitude below the calculated concentrations in 1 meter of water in Table 3-5. Note that Table 3-5 includes both ppm and ppb concentrations.

In 1982, prior to the start of WHCP, USDA-ARS (Anderson 1982) conducted field tests of 2,4-D levels following herbicide applications at Coney Island, in the Delta. Anderson collected samples in float samplers (open-top vessels on top of the water containing 500 mls Delta water), inside the spray plot, upstream of the spray plot, and downstream of the spray plot, at 15 to 30 minute intervals post-treatment. This simulated the actual concentration reaching the water hyacinth plant, and the instantaneous concentration on the surface of the water if the herbicide reached the water, rather than the plant, prior to the herbicide mixing and diluting with the water. In addition, Anderson (1982) utilized 2,4-D levels 25 percent higher than current herbicide application rates. Both of these factors resulted in a higher concentration than if the samples had been collected in the water, as illustrated by the lower historical 2,4-D levels taken in actual water samples. The data in **Table 3-6,** on the next page, provides the range and average for test measurements, illustrate the above-maximum immediate 2,4-D concentrations and the drop in concentrations within the first 90 minutes post-treatment. Anderson also utilized this study to estimate herbicide overspray.

Table 3-6
Results of Delta Coney Island Field Test, Concentrations of 2,4-D Following Treatment

Time and Location of Samples *(Number of Samples)*	Range	Average
1. Float samplers in spray plot (5)	51 ppb to 3,150 ppb	1,047 ppb
2. Water samples in spray plot @ 15 minutes post (6)	107 ppb to 8,420 ppb	2,262 ppb
3. Water samples in spray plot @ 60 minutes post (3)	593 ppb to 1,398 ppb	895 ppb
4. Water samples in spray plot @ 90 minutes post (3)	100 ppb to 157 ppb	119 ppb
5. Water samples upstream of spray plot @ 15 minutes post (3)	17 ppb to 59 ppb	32 ppb
6. Water samples downstream of spray plot @ 30 minutes post (3)	3 ppb to 5 ppb	4 ppb
7. Water samples downstream of spray plot @ 60 minutes post (3)	0 ppb to 50 ppb	17 ppb
8. Water samples downstream of spray plot @ 90 minutes post (3)	3 ppb to 23 ppb	10 ppb

WHCP environmental monitoring results since 2001 provide additional data on actual herbicide residue levels following treatments. From 2001 to 2005, DBW obtained chemical residue tests on 110 water samples collected at two to three feet depth one hour after treatment, inside the treatment areas. Samples were obtained from 48 different sites, and throughout the treatment season (for both chemicals at some sites). The average concentration at each of the 2,4-D sites ranged from non-detectable (ND), to 390 ppb. The 390 ppb measure was an outlier, representing one of over 100 sampling events between 2001 and 2005. The highest measured 2,4-D level since 2005 was 30 ppb, and this measure was also an outlier, representing one of 62 sampling events. **Figure 3-6,** on the next page, summarizes herbicide concentrations of the in-treatment-site samples for 2001 to 2005.

Over six years of environmental monitoring (2006 to 2011), DBW monitored receiving waters directly downstream of the treatment sites, one-hour after treatment. As in previous years, environmental scientists also returned to each site two to seven days later to sample upstream, within, and downstream of the treatment site. All samples were taken at two to three feet depth. Over the six year period, DBW conducted 62 sampling events for 2,4-D. DBW also monitored Agridex at all the 97 sampling events. In every case, Agridex concentrations were non-detectable.

Figure 3-7, on the next page, illustrates the 2006 to 2011 sampling results from immediately downstream of treatment sites, in WHCP receiving waters, for 2,4-D. This is a slightly different location than the 2001 to 2005 results illustrated in Figure 3-6. While both sets of samples were taken one-hour post-treatment, we would expect the downstream location to have lower chemical concentrations than the in-treatment-site location, due to dilution as herbicide flows out of the treatment site.

Figure 3-6
Number of Sites at Various 2,4-D Concentrations (IN Treatment Site) (2001 to 2005)

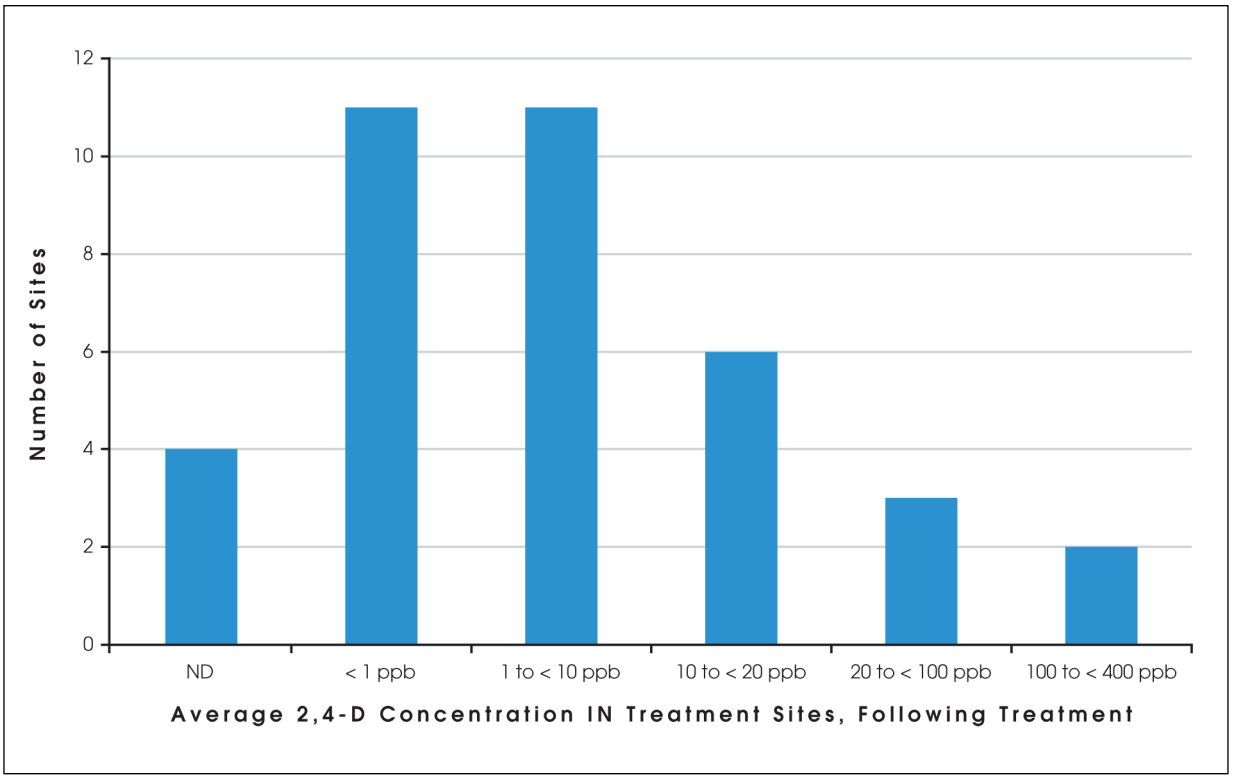


Figure 3-7
Concentrations of 2,4-D Downstream of Treatment (2006 to 2011)

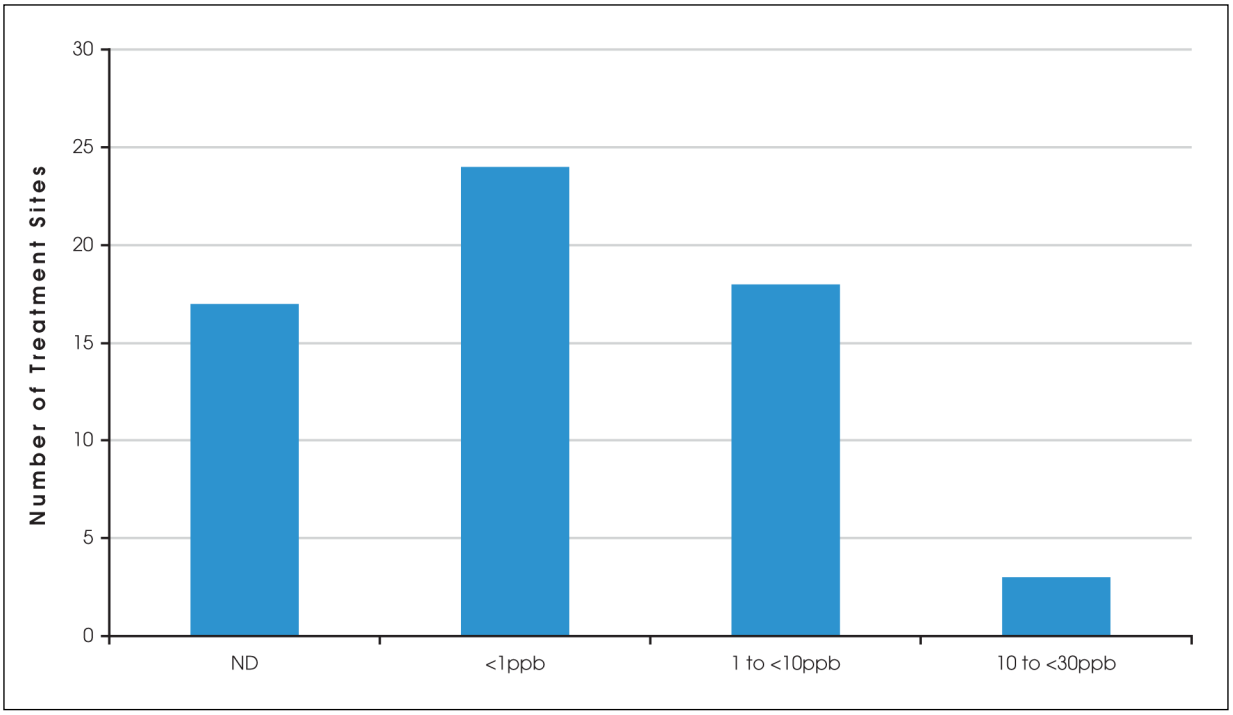


Table 3-7
Concentrations of 2,4-D Downstream of WHCP Treatments, 1 Hour Post-Treatment (2006 to 2011)

Concentration (ppb or ug/l)	Number of Sites
No Detect (ND)	17
<1 ppb	24
1 to <10 ppb	18
10 to <30 ppb	3
Total	62

Figure 3-8
Glyphosate Chemical Structure

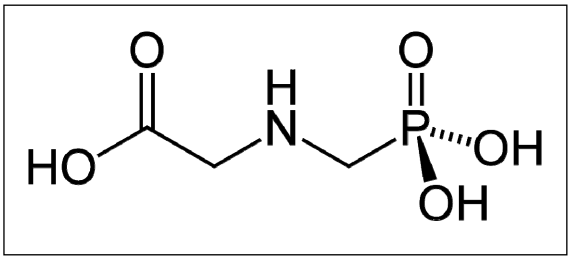


Table 3-7, above, provides a tabular summary of the sampling data presented in Figure 3-7. For 2,4-D, the maximum post-treatment concentration one hour after treatment was 30 ppb, and 66 percent of the samples over the 5 year period had levels of less than 1 ppb or non-detectable.

In the 2006 to 2011 follow-up sampling results (two to seven days after treatment), there were a few cases where 2,4-D levels were slightly higher in than immediately post treatment, although still low (a maximum of 16.3 ppb at one site). Typically, 2,4-D levels declined to very low or non-detectable levels in the follow-up sampling taken between two days and seven days after treatment showed very low herbicide levels in waters in and downstream of the treatment site. 2,4-D levels four to six days following treatment at seven 2,4-D samples taken in 2011 ranged from non-detectable to 0.2 ppb. Between 2006 and 2010, the maximum 2,4-D level found between one and four days following treatment was 2.5 ppb.

The calculated, test plot, and actual WHCP herbicide levels indicate that 2,4-D concentrations in the Delta following herbicide treatment are likely to be low. Maximum 2,4-D levels immediately after spraying within a treatment site have reached levels as high as 390 ppb (0.4 ppm, rounded), although this occurred one time in monitoring conducted immediately after treatment, under a water hyacinth mat, out of over 100 similar samples taken between 2001 and 2005. Maximum 2,4-D levels immediately downstream of the site were less than 1 ppb in 39 percent of samples, between 1 ppb and 10 ppb in 29 percent of samples, and have never been measured at levels higher than 30 ppb (30 ppb was measured once out of 62 samples). Based on historical data, herbicides remain at these maximum levels for a short period of time (for example, the downstream sampling typically occurs within one hour of treatment).

Glyphosate

Glyphosate is a broad spectrum, non-selective, systemic herbicide active ingredient. The chemical structure of glyphosate is illustrated in **Figure 3-8,** above. Glyphosate has been a secondary WHCP treatment option (behind 2,4-D) since the program's inception, and glyphosate was first approved for use in the United States in 1973. DBW

currently utilizes the glyphosate product AquaMaster®, and has also utilized Rodeo®, a similar glyphosate herbicide also approved for aquatic use.

Glyphosate is water soluble, and is absorbed across the plant surface and translocated throughout the plant. Glyphosate inhibits activity of the shikimic acid pathway enzymes, found only in plants and microorganisms. Glyphosate is not metabolized by plants (Schuette 1998). The organic carbon sorption coefficient, K_{oc}, of glyphosate is between 300 and 20,100, indicating strong adsorption to soil (Miller et al. 2010).

Studies show that glyphosate is not persistent in the water column. Glyphosate binds tightly to sediment, removing the active ingredient from water. The half-life of glyphosate in pond water ranges from 12 days to 10 weeks (EXTONET 1996). At two Delta test plots, researchers applied 100 gallons of 6 pounds per acre glyphosate solution, double the labeled rate. The highest concentration of glyphosate was found after 4 hours (60 ppb), in a test spray area not subject to tidal flow (Corcoran et al. 1984). At a test site with tidal flow, the highest concentration of glyphosate (40 ppb) was found one-half hour after treatment (Corcoran et al. 1984). When glyphosate was sprayed aerially at a rate of 5 pints per acre (also higher than the labeled rate), glyphosate was at its maximum concentration one-half day after treatment (0.28 ppm to 0.60 ppm). After six to eight days, glyphosate levels ranged from undetectable (<0.001 ppm) to 0.49 ppm (Henry et al. 1994). In turbid water, glyphosate is degraded by microorganisms (Siepmann 1995).

Studies in Canada suggest that sediment adsorption and microbial degradation are responsible for glyphosate's loss from water (Schuette 1998). Glyphosate degradation in soil yields aminomethylphosphonic acid (AMPA) and glyoxylic acid. Both products are further degraded to carbon dioxide (Miller et al., 2010).

For treating water hyacinth, glyphosate will be applied at a rate of three quarts per acre, per label requirements. This will be equivalent to 3 pounds active ingredient per acre. Glyphosate will be applied via a broadcast sprayer.

The majority of the sites treated with glyphosate will be spot treatments. For the sites that are heavily vegetated, buffer strips will be created, and another treatment will occur, if needed.

The herbicide label requirements for glyphosate have no restrictions for use of treated water for irrigation, recreation, or domestic purposes. The herbicide label specifies that glyphosate is not to be applied within 0.5 miles of an active potable water intake; or intakes must be turned off for a minimum of 48 hours after the application, or until glyphosate concentrations are less than 0.7 ppm. When treating large infestations, the label recommends treating the area in strips to avoid oxygen depletion.

Table 3-8, on the next page, summarizes expected instantaneous concentrations of active ingredients at the spray nozzle, and in the water. Table 3-8 provides conservative estimates assuming that 100 percent of the herbicide reaches one or two meters depth of water, and a more realistic (but still conservative) estimate assuming 20 percent

of the herbicide reaches one or two meters depth of water. Note that Table 3-8 includes both ppm and ppb concentrations. The amount of glyphosate applied in the project area to control water hyacinth, and the resulting glyphosate concentrations in Delta waters, can be minimized by treating plants early in the growing season before plants have grown into the large mono-specific mats characteristic of this species. Early treatment will also minimize the negative ecosystem impacts of this invasive species.

The calculated maximum concentrations in Table 3-8 reflect potential chemical concentrations immediately after (or during) spraying. However, herbicides dissipate over time, as the Delta is subject to tidal action and water flow. Thus, the concentration of chemicals will be further diluted as water moves within the Delta.

The historical WHCP environmental monitoring results provide additional data on actual herbicide residue levels following treatments. From 2001 to 2005, the DBW obtained chemical residue tests on 110 water samples collected one-hour after treatment, inside the treatment areas at two to three feet depth. Samples were obtained from 48 different sites, and throughout the treatment season (for both chemicals at some sites). The average concentration at each of the 14 glyphosate sites ranged from non-detectable to 158 ppb. The 158 ppb measure was an outlier, accounting for one of over 100 sampling events between 2001 and 2005. **Figure 3-9,** on the next page, summarizes glyphosate concentrations of the in-treatment-site samples for 2001 to 2005.

Table 3-8
Calculated* Maximum Concentrations of Glyphosate Immediately Following WHCP Treatment

Concentration of:	Glyphosate *(Active Ingredient)*
1. Chemical directly out of spray nozzle	3,600 ppm
2. Chemical in 1 meter deep water, @ 100% water contact	0.34 ppm
3. Chemical in 2 meter deep water, @ 100% water contact	0.17 ppm
4. Chemical in 1 meter deep water, @ 20% water contact	67 ppb
5. Chemical in 2 meter deep water, @ 20% water contact	34 ppb

* The concentrations above are based on the pounds of active ingredient in maximum specified application rate per acre, and an appropriate dilution factor based on the volume of water in the tank mix, or within one or two meter-acres.

Over six years of environmental monitoring (2006 to 2011), DBW has monitored receiving waters directly downstream of the treatment sites, one-hour after treatment. As in previous years, environmental scientists also returned to each site two to seven days later to sample upstream, within, and downstream of the treatment site. Over the six year period, DBW conducted 35 sampling events for glyphosate. All samples were taken at a depth of two to three feet.

Figure 3-10, on the next page, illustrates the 2006 to 2011 sampling results from immediately downstream of treatment sites, in WHCP receiving waters. This is a slightly different location than the 2001 to 2005 results illustrated in Figure 3-9. While both sets of samples were taken immediately post-treatment, we would expect the downstream location to have lower chemical concentrations

Figure 3-9
Number of Sites at Various Glyphosate Concentrations (IN Treatment Site) (2001 to 2005)

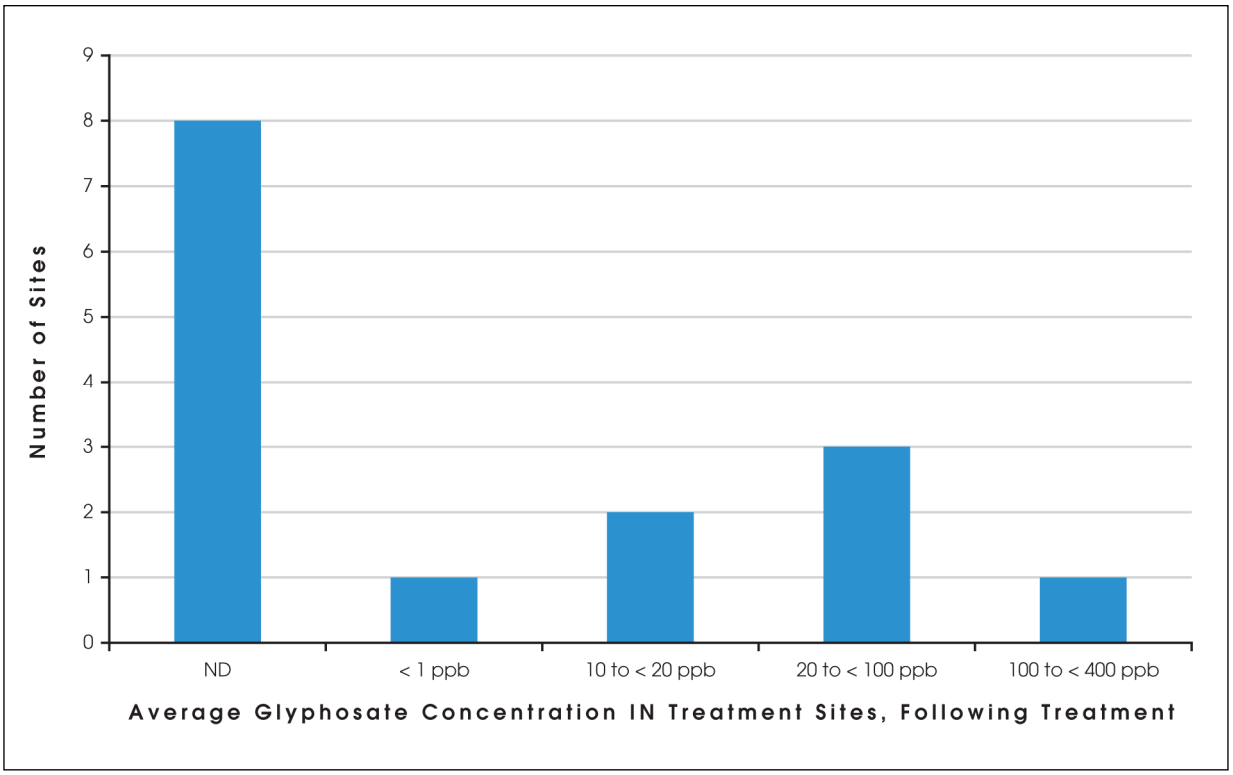
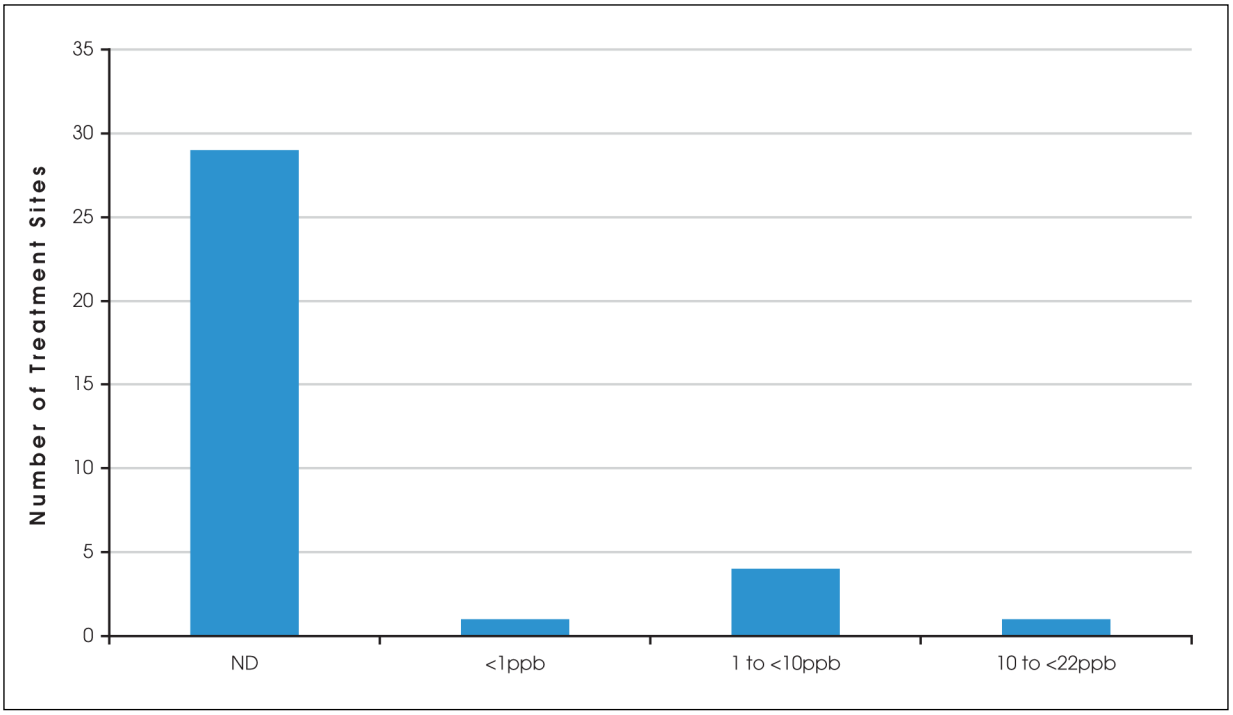


Figure 3-10
Concentrations of Glyphosate Downstream of Treatment (2006 to 2011)



than the in-treatment-site location, due to dilution as herbicide flows out of the treatment site. **Table 3-9,** right, provides a tabular summary of the sampling data presented in Figure 3-10. For glyphosate, the maximum post-treatment concentration one hour after treatment was 22 ppb, and 86% of the samples had levels of less than 1 ppb or non-detectable.

Glyphosate levels in follow-up sampling taken between one day and seven days after treatment show even lower herbicide levels in waters in and downstream of the treatment site. Glyphosate was non-detectable in samples taken five to seven days after treatment. Between 2006 and 2011, all glyphosate samples taken one or more days post-treatment had non-detectable levels of the herbicide.

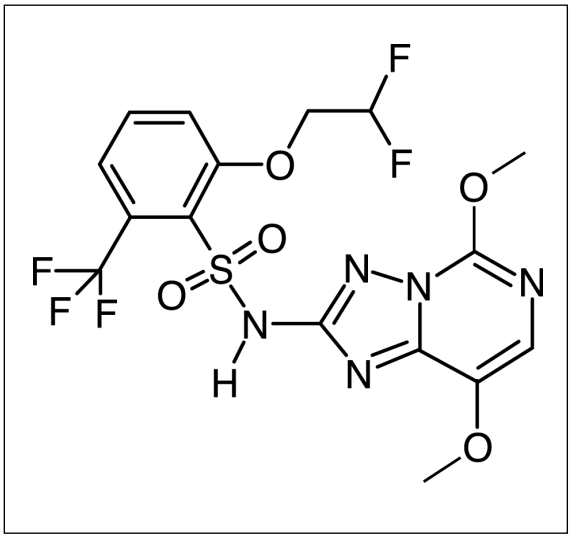
In prior years, glyphosate was tested fewer times than 2,4-D, because this herbicide was used less frequently during the 2006 to 2011 treatment seasons. Glyphosate levels decreased in the follow-up visits, however there were a few cases in which glyphosate levels were higher in the pre-treatment samples (up to 21 ppb), indicating the herbicide was present in Delta waters from other sources.

The calculated, test plot, and actual WHCP herbicide levels indicate that glyphosate levels in the Delta following herbicide treatment will be low. Maximum glyphosate levels within a treatment site, immediately after spraying, may reach as high as 158 ppb (0.158 ppm), but are likely to be less than 30 ppb. Maximum glyphosate levels immediately downstream are likely to be less than 2 ppb. Herbicides may remain at these maximum levels for a relatively short period of time (for example, the downstream sampling typically occurs within one hour of treatment).

Table 3-9
Concentrations of Glyphosate Downstream of WHCP Treatments, 1 Hour Post-Treatment (2006 to 2011)

Concentration (ppb or ug/l)	Number of Sites
ND	29
<1 ppb	1
1 to <10 ppb	4
10 to <22 ppb	1
Total	35

Figure 3-11
Penoxsulam Chemical Structure



Penoxsulam

Penoxsulam received USEPA approval through the reduced risk program for use on aquatic weeds from the USEPA in 2007 and from the California DPR in 2009. Penoxsulam was initially approved for use on rice crops by USEPA in 2004. The chemical structure of penoxsulam is illustrated in **Figure 3-11,** above. The WHCP currently plans to utilize the penoxsulam product Galleon*.

Penoxsulam (2-(2,2-difluoroethoxy)-N-(5,8-dimethoxyl[1,2,4] triazolo[1,5-c]pyrimidin-2-yl)-6-trifluoromethyl)benzenesulfonamide), is a broad spectrum systemic herbicide in the triazolopyrimidine sulfonamide family. This herbicide inhibits the enzyme acetolactate synthase (ALS), which regulates the production of three essential amino acids: valine, leucine, and isoleucine (Washington DOE 2012). ALS inhibitors such as penoxsulam slowly starve plants of these amino acids, eventually killing the plants by halting DNA synthesis. These biochemical pathways are not present in animals.

Plants absorb penoxsulam through leaves, shoots, and roots. The herbicide affects new growth more rapidly than older plant tissue. Symptoms following treatment with penoxsulam include immediate growth inhibition, a chlorotic growing point with reddening, and slow plant death over a period of 60 to 120 days (Washington DOE 2012). Madsen and Wersal (2008) found that four weeks after treatment with 1.4 oz/acre, up to the maximum rate of 5.6 oz/acre, penoxsulam (with a surfactant) provided 95 percent control of water hyacinth in 100-gallon outdoor tanks. Langeland et al. (2009) identified penoxsulam as providing excellent control for water hyacinth in Florida.

Penoxsulam has low to moderate water solubility, and is very mobile in soil. The organic carbon sorption coefficient, K_{oc}, of penoxsulam is between 13 and 305 in soil (indicating weak adsorption), with higher adsorption in sediment, K_{oc} = 1,130 (USEPA 2007).

Penoxsulam follows two complex degradation pathways, and degrades into eleven major and two minor degradates, listed in **Table 3-10,** on the next page (USEPA 2007). None of these metabolites or degradates have been identified as having a higher toxicity potential than penoxsulam (Washington DOE 2012).

There was some concern in the first review of penoxsulam (USEPA 2004) that some of the major degradates of penoxsulam might pose phytoxicity concerns; however, additional testing found no observable injury by the eleven metabolites to pre-emergent seeds, and that only two caused injury to seedlings at high-levels (USEPA 2007).

In water, penoxsulam breaks down primarily by photolysis, with some microbial degradation. Water depth, water clarity, plant density, and season of application can influence photolytic degradation. Penoxsulam breaks down faster in higher water clarity and lower plant density. The water solubility of penoxsulam increases in more alkaline conditions. The half-life of penoxsulam in water ranges from 1.5 to 14 days (USEPA 2007). The total system half-life of penoxsulam is 16 to 38 days (Washington DOE 2012). In sediment, penoxsulam is expected to degrade rapidly through anaerobic degradation (USEPA 2007). Penoxsulam is adsorbed by soil and has low to moderate leaching potential in most soil types, where it is broken down by microbial degradation (The Dow Chemical Company 2008). However, California DPR has identified penoxsulam (along with many other herbicides including 2,4-D and glyphosate) as having the potential to pollute ground water. Penoxsulam has low vapor pressure, and will not dissipate by volatization.

Table 3-10
Major and Minor Transformation Products of Penoxsulam

Abbreviation	Chemical Name
1. BSTCA	3-[[[2-(2,2-difluroethoxy)-6-(trifluromethyl)phenyl]-sulfonyl]amino]-1H-1,2,4-triazole-5-carboxylic acid
2. TPSA	5,8-dimethoxyl[1,2-4]triazolo-[1,5-c]pyrimidin-2-yl-sulfamic acid
3. 2-Amino TP	5,8-dimethoxyl[1,2-4]triazolo-[1,5-c]pyrimidin-2-amine
4. BSA	2-(2,2-difluoroethoxy(-5-(trifluoromethyl) benzenesulphonic acid
5. 2-Amino TCA	2-amino-1,2-4-triazole carboxylic acid
6. 5-OH, 2-Amino TP	2-amino-8-methoxy-[1,2,4]triazolo[1,5-c]pyrimidin-5-(6H)-one
7. BSTCA methyl	methyl 5-[[[2-(2,2-difluoroethoxy)-6-(trifluoromethyl) phenyl]sulphony]amino]-1H-1,2,4-triazole-5carboxylate
8. BST	2-(2,2-difluoroethyoxy)-N-1H-1,2-4-triazol-3-yl-6-(trifluoromethyl)benzenesulphonamide
9. Di-FESA	3-(2,2-difluoroethoxy)-2-hydroxybenzoic acid
10. 5-OH-penoxsulam	2-(2,2-difluoroethoxy)-N-(5,6-dihydro-8-methoxy-5-oxo[1,2,4]triazolo[1,5-c]pyrimidin-2-yl)-6-(trifluoromethyl)benzenesulfonamide
11. Sulfonyl-formamidine	2-(2,2-difluoroethoxy)-N-(iminomethyl-6-(trifluoromethyl)-benzenesulfonamide
1. Sulfonamide	2-(2,2-difluoroethoxy)-6-(trifluoromethyl)-benzenesulfonamide
2. 5-OH XDE638	2-(2,2-difluoroethoxy)-6-trifluoromethyl-N-(5,8-dihyroxy-[1,2,4]triazolo[1,5-c]pyrimidin-2-yl) benzenesulfonamide

For treating water hyacinth, penoxsulam will be applied at between 2.0 to 5.6 ounces per acre, per label requirements, with higher rates for denser plants and plants not at their peak growing phase. This will be equivalent to between 0.03125 and 0.0875 pounds of active ingredient per acre. Penoxsulam will be applied with a surfactant (at concentrations on the surfactant label), with a spray volume in accordance to label specifications.

There are no label restrictions for penoxsulam regarding dissolved oxygen, as the slow-acting nature of this herbicide should have minimal impact on dissolved oxygen levels (Washington DOE 2012). However, WHCP will maintain existing monitoring measures related to dissolved oxygen to evaluate potential reductions in DO.

Waters treated with penoxsulam will not to be used for food crop irrigation until concentrations are determined to be equal to, or less than, 1 ppb. Water samples will be collected using Enzyme-Linked Immunoassay (ELISA) or other approved analytical methods. There are no restrictions on consumption of treated water for potable use or by livestock, pets, or other animals, and no restrictions on the use of treated water for recreational use, including swimming and fishing. Penoxsulam will be used with a surfactant, and applied with a course high flow spray nozzle to avoid drift. Penoxsulam will not be applied when wind speeds are below 2 mph, or above 10 mph.

Table 3-11
Calculated* Maximum Concentrations of Penoxsulam Immediately Following WHCP Treatment

Concentration of:	Penoxsulam *(active ingredient)*
1. Chemical directly out of spray nozzle	105 ppm
2. Chemical in 1 meter deep water, @ 100% water contact	9.8 ppb
3. Chemical in 2 meter deep water, @ 100% water contact	4.9 ppb
4. Chemical in 1 meter deep water, @ 20% water contact	2 ppb
5. Chemical in 2 meter deep water, @ 20% water contact	1 ppb

* The concentrations above are based on the pounds of active ingredient in maximum specified application rate per acre, and an appropriate dilution factor based on the volume of water in the tank mix, or within one or two meter-acres.

Figure 3-12
Imazamox Chemical Structure

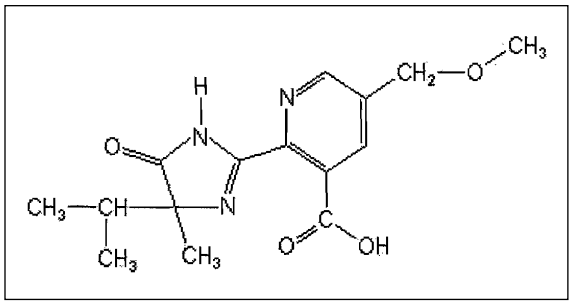


Table 3-11, above, provides the estimates for penoxsulam concentration immediately out of the spray nozzle, and under various water contact assumptions. For comparison, the maximum allowable label concentration for in-water application of penoxsulam for treating submerged weeds is 150 ppb. As penoxsulam will be a new WHCP herbicide, there are no prior program data regarding actual herbicide concentrations following water hyacinth treatment. The WHCP will conduct monitoring at the initial penoxsulam treatment sites to develop a baseline for expected herbicide concentrations in treatment sites and receiving waters following treatment. Note that Table 3-11 includes both ppm and ppb concentrations. The amount of penoxsulam applied in the project area to control water hyacinth, and the resulting penoxsulam concentrations in Delta waters, can be minimized by treating plants early in the growing season before plants have grown into the large mono-specific mats characteristic of this species. Early treatment will also minimize the negative ecosystem impacts of this invasive species.

Imazamox

Imazamox is a relatively new aquatic herbicide active ingredient. The chemical structure of imazamox is illustrated in **Figure 3-12,** left. The aquatic formulation of imazamox, Clearcast®, received USEPA approval through the reduced risk program in 2008 (SERA 2010). The WHCP will initially utilize this imazamox active ingredient product.

CDPR approved imazamox for aquatic use in August, 2012. Imazamox was approved for terrestrial use by the USEPA in 1997, and by the California DPR, in 2002. Clearcast consists of 12.1 percent solution of the ammonium salt of imazamox (2-[4,5-dihydro-4-methyl-4-(1-methylethyl)-5-oxo-1H-imidazol-2-yl]-5-(methyoxymethyl)-3-pyridinecarboxylic acid). It is in the imidazolinone herbicide family, along with imazapyr. The mode of action is similar to penoxsulam and imazapyr, inhibiting the

acetolactate synthase (ALS) enzyme, blocking the synthesis of three essential amino acids, leucine, isoleucine, and valine (Washington DOE 2012).

Imazamox is a relatively fast-acting systemic herbicide. It is rapidly absorbed into the foliage and translocated throughout the plant by phloem and xylem tissues (Washington DOE 2012). Imazamox inhibits plant growth within the first 24 hours, with visual symptoms appearing about one week after treatment. Symptoms include yellowing leaves and general discoloration. Water hyacinth plants are dead within six weeks after treatment (Burns 2009). In one greenhouse study, Clearcast was more effective at controlling water hyacinth within five weeks (94 percent control) than Habitat® (imazapyr) (79 percent control), but slightly less effective than glyphosate (99 percent control). However, Clearcast and Habitat required less than 25 percent as much active ingredient as glyphosate treatment (Emerine et al. 2010). Langeland et al. (2009) identify imazamox as excellent in controlling water hyacinth in Florida.

Imazamox is highly soluble in water, and is mobile to highly mobile in soil (Washington DOE 2012; USEPA 2008). The organic carbon sorption coefficient, K_{oc}, of imazamox is between 5 and 143 (indicating weak adsorption). Volatization of imazamox is not significant (USEPA 1997). Imazamox has a low potential for bioaccumulation (Washington DOE 2012).

The primary method of degradation of imazamox in surface water is photolytic (Washington DOE 2012). Photolytic degradation is influenced by water depth, water clarity, and season, and continues via microbial action to carbon dioxide. The half-life in water ranges from five to fifteen days (Washington DOE 2012). CDPR identified imazamox as having the potential to pollute groundwater due to its high water solubility; however, in well-lit waters, imazamox breaks down quickly (Washington DOE 2012). US EPA concluded that even if imazamox persists in dark or turbid waters it is unlikely to present a risk to fish, invertebrates, birds, or mammals (Washington DOE 2012).

Imazamox is moderately persistent in soil, degrading aerobically to a non-herbicidal metabolite which is immobile or moderately mobile in soil (USEPA 1997). The primary metabolite is a demethylated parent chemical with intact ring structures and two carboxylic acid groups. A secondary metabolite is a demethylated, decarboxylated parent with intact rings and one carboxylic acid group (USEPA 2008). Leaching of imazamox in field studies was very limited, and microbial breakdown products under aerobic soil conditions are not herbicidal. The range of half-lives in terrestrial field dissipation studies was fifteen to 130 days, with typical half-lives ranging from 35 to 50 days (USEPA 1997; USEPA 2008). Imazamox is unlikely to accumulate in sediments.

For treating water hyacinth, imazamox will be applied at a rate of 16 to 64 ounces per acre, per label requirements. This is equivalent to 0.125 to 0.5 pounds active ingredient per acre. Imazamox is most effective when applied to actively growing plants. Imazamox will be applied with an adjuvant at rate of one quart per 100 gallons of solution.

Table 3-12
Calculated* Maximum Concentrations of Imazamox Immediately Following WHCP Treatment

Concentration of:	Imazamox *(active ingredient)*
1. Chemical directly out of spray nozzle	600 ppm
2. Chemical in 1 meter deep water, @ 100% water contact	56 ppb
3. Chemical in 2 meter deep water, @ 100% water contact	28 ppb
4. Chemical in 1 meter deep water, @ 20% water contact	11.2 ppb
5. Chemical in 2 meter deep water, @ 20% water contact	5.6 ppb

* The concentrations above are based on the pounds of active ingredient in maximum specified application rate per acre, and an appropriate dilution factor based on the volume of water in the tank mix, or within one or two meter-acres.

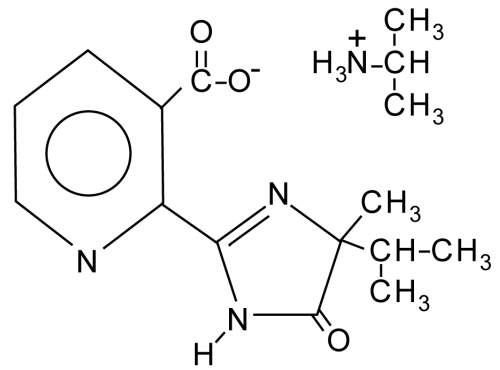
There are no label restrictions regarding dissolved oxygen; however, DBW will follow the same monitoring approaches as for other herbicides to evaluate potential for low DO levels to impact endangered species. Waters treated with imazamox will not be used for irrigation until concentrations are less than 50 ppb. The label requires a 24 hour period after treatment to irrigate from still and quiescent waters. There are no wait restrictions for irrigation when imazamox is applied to flowing waters at a rate of less than or equal to 4 quarts (64 ounces) per acre to waters with an average depth of at least four feet. There are no restrictions on livestock watering, swimming, fishing, domestic use, or use of treated water for agricultural sprays (SePRO 2010). To reduce drift, imazamox will be used with a surfactant, and applied in a course spray with the nozzle height at approximately no more than four feet above the plant canopy. Imazamox will not to be applied in a temperature inversion, or when wind speeds are less than 2 miles per hour or greater than 10 miles per hour.

Table 3-12, left, provides estimates for imazamox concentrations immediately out of the spray gun, and under various contact assumptions. For comparison, the maximum allowable concentration for in-water application of imazamox for submerged weeds is 500 ppb, which is greater than the calculated concentration from the spray nozzle for water hyacinth treatment. As imazamox will be a new WHCP herbicide, there are no prior test data regarding actual herbicide concentrations following water hyacinth treatment. The WHCP will conduct monitoring at the initial imazamox treatment sites to develop a baseline for expected herbicide concentrations in treatment sites and receiving waters following treatment. Note that Table 3-12 includes both ppm and ppb concentrations. The amount of imazamox applied in the project area to control water hyacinth, and the resulting imazamox concentrations in Delta waters, can be minimized by treating plants early in the growing season before plants have grown into the large mono-specific mats characteristic of this species. Early treatment will also minimize the negative ecosystem impacts of this invasive species.

Imazapyr

Imazapyr is an herbicide active ingredient approved for aquatic use. The chemical structure of imazapyr is illustrated in **Figure 3-13,** on the next page. The imazapyr product Habitat® received USEPA approval for use in non-crop aquatic sites in 2003, and from the CDPR in 2005 (Pless 2005).

Figure 3-13
Imazapyr Chemical Structure



Imazapyr isopropylamine salt

Imazapyr products have been used in terrestrial applications since 1983. Imazapyr has been used to control non-native spartina in San Francisco Bay since 2006, but has not yet been approved for water hyacinth control in California. Approval by CDPR for water hyacinth control could occur in the next two years, at which point the WHCP will incorporate imazapyr as an additional herbicide.

Like penoxsulam and imazamox, imazapyr is an ALS inhibitor, although it is in the imidazolinone chemical class. Habitat consists of 28.7 percent of the isopropylamine salt of imazapyr (2-[4,5-dihydro-4-methyl-4(1-methylethyl)-5-oxo-1H-imidazol-2-yl]-3-pyridinecarboxylic acid). Imazapyr is a systemic, broad-spectrum, pre- and post-emergent herbicide. Imazapyr inhibits the enzyme acetolactate synthase in plants, blocking the production of three essential amino acids (valine, leucine, and isoleucine) (AMEC Geometrix 2009). This enzyme is not present in animals.

Imazapyr is absorbed by leaves and roots, and accumulates in the meristem region of the plant. Imazapyr is most effective when target plants are growing rapidly. The rate of plant death is slow, and it may take several weeks or months for complete plant death. Treated plants stop growing soon after spray application, and chlorosis appears first in the newest leaves, with necrosis spreading from this point (BASF 2008). Langeland et. al. (2009) identified imazapyr as providing excellent control for water hyacinth in Florida.

Imazapyr is highly soluble in water and is both very mobile and persistent in soil. Imazapyr isopropylamine salt is primarily in the anionic form at typical environmental pH values (AMEC Geometrix 2009). The organic carbon sorption coefficient, K_{oc}, of imazapyr is between 8 and 150, depending on the type of soil, indicating weak adsorption (AMEC Geomatrix 2009; SERA 2004).

In water, imazapyr degrades by photolysis, with a half-life of three to five days (USEPA 2006). The three major metabolites are pyridine hydroxyl-carboxylic acid, pyridine dicarboxylic acid, and nicotinic acid (Niacin, or Vitamin B3) (USEPA 2006). Under laboratory conditions, the half-lives of the two pyridine acids are three to eight days in two different sediment-water systems. These metabolites are more polar than imazapyr, no more toxic than the parent compound, and more rapidly excreted (USEPA 2006). Nicotinic acid is a possible neurotoxin at high doses, but there is no concern for low dose exposures of this metabolite (USEPA 2006).

In soil, the degradation of imazapyr is essentially stable to hydrolysis, and aerobic and anaerobic soil degradation (AMEC Geomatrix 2009). In soil, imazapyr degrades primarily through microbial degradation. The

soil half-life of imazapyr ranges from 210 days to 5.9 years, depending on climate, temperature, precipitation, wind, hydrology, soil characteristics, microbial activity, and chemical degradation (AMEC Geomatrix 2009). At annual rainfall rates of 10 inches or more (which includes the Delta), imazapyr will be removed from soil by runoff and/or percolation. Imazapyr is most persistent in loamy soils, slightly persistent in clay soils, and less persistent in sandy soils (AMEC Geomatrix 2009). Field dissipation half-life of imazapyr is 25 to 180 days (SERA 2004). In field studies using the isopropylamine salt of imazapyr herbicide Arsenal, which is similar to Habitat, imazapyr dissipated in ponds and sediment with half-lives of 2 to 4 days when applied at the maximum label rate of 1.5 pounds active ingredient per acre (three times the rate for water hyacinth) (AMEC Geomatrix 2009). Imazapyr is not expected to bioaccumulate in aquatic species (USEPA 2006). The plant half-life of imazapyr is 15 to 37 days (AMEC Geometrix 2009). Imazapyr is non-volatile (USEPA 2009).

For treating water hyacinth, imazapyr will be applied at a rate of one to two pints per acre, per label requirements, with the higher application rate for dense plant mats and when the weed is not at its peak growing rate. Two pints per acre is equivalent to 0.25 pounds of active ingredient per acre. Imazapyr will be applied with a surfactant at concentrations of 1.5 to 2 pints surfactant per acre, at a spray volume as specified on the label.

To avoid potential for low dissolved oxygen from decaying vegetation, the imazapyr label requires applications be made in strips when target vegetation covers a large percentage of the surface area of impounded water (BASF 2008). The label also restricts treatment to no more than one half of the surface area of the water in a single operation, with the remainder not treated for at least ten to fourteen days. Treatment will begin along the shore and proceed outward in bands to allow aquatic organisms to move to untreated areas. The label also requires imazapyr to be applied as large droplets and with high flow rates to avoid potential for drift. Imazapyr should be applied at the lowest possible (above the plant) height, at wind speeds of between three and ten miles per hour, and not during temperature inversions.

Water treated with imazapyr may not be used for irrigation purposes for 120 days after application or until residue levels are 1.0 ppb or less. Given the likely concentrations of imazapyr following treatment, and Delta water flows, the 1.0 ppb threshold is likely to occur much sooner than 120 days. In quiescent or slow moving waters, imazapyr will not be applied within one mile of an active irrigation intake. In moving water, imazapyr will not be applied within one-half mile downstream of an active irrigation water intake, and should only be applied upstream of the intake when the intake is turned off with sufficient time for treated water to flow past the intake. There are no restrictions for recreational use of treated waters for swimming and fishing, and no restrictions on livestock consumption of treated water. Imazapyr will not be applied directly to water within one-half mile upstream of an active potable water intake. Potable water intakes must be turned off for a minimum of 48 hours after imazapyr application.

Table 3-13
Calculated* Maximum Concentrations of Imazapyr Immediately Following WHCP Treatment

Concentration of:	Imazapyr *(active ingredient)*
1. Chemical directly out of spray nozzle	300 ppm
2. Chemical in 1 meter deep water, @ 100% water contact	28 ppb
3. Chemical in 2 meter deep water, @ 100% water contact	14 ppb
4. Chemical in 1 meter deep water, @ 20% water contact	5.6 ppb
5. Chemical in 2 meter deep water, @ 20% water contact	2.8 ppb

* The concentrations above are based on the pounds of active ingredient in maximum specified application rate per acre, and an appropriate dilution factor based on the volume of water in the tank mix, or within one or two meter-acres.

Table 3-13, above, provides estimates for imazapyr concentration immediately out of the spray gun, and under various water contact assumptions. As imazapyr will be a new WHCP herbicide (once it is approved) there are no prior data regarding actual herbicide concentrations following water hyacinth treatment. The WHCP will conduct monitoring at the initial imazapyr treatment sites to develop a baseline for expected herbicide concentrations in treatment sites and receiving waters following treatment. Note that Table 3-13 includes both ppm and ppb concentrations. The amount of imazapyr applied in the project area to control water hyacinth, and the resulting imazapyr concentrations in Delta waters, can be minimized by treating plants early in the growing season before plants have grown into the large mono-specific mats characteristic of this species. Early treatment will also minimize the negative ecosystem impacts of this invasive species.

Adjuvants

The WHCP will utilize adjuvants with herbicides to ensure contact and translocation of herbicides. The WHCP will not utilize polyethoxylated tallow amine (POEA) surfactants, which are known to be toxic to amphibians, or nonylphenoloethoxylate (NPE) surfactants, which are known to be toxic to fish and some invertebrates. The WHCP will utilize two adjuvants. Agridex®, a crop oil concentrate adjuvant, has been used for several years by WHCP. Competitor®, a vegetable oil based adjuvant, will be incorporated into WHCP.

Agridex

Agridex is a non-ionic blend of surfactants and spray oil that is designed for use with a broad range of pesticides where an oil concentration adjuvant is recommended. Agridex improves pesticide application by modifying the wetting and deposition characteristics of the spray solution, resulting in a more even and uniform spray deposit. The active ingredients in Agridex are paraffin base petroleum oil and polyoxyethylate polyol fatty acid esters. It will be used with WHCP herbicides at a rate of approximately one to four pints per 100 gallons of spray solution.

Over six years of environmental monitoring (2006 to 2011), DBW has monitored receiving waters directly downstream of the treatment sites, immediately after treatment. Environmental scientists also returned to each site two to seven days later to sample upstream, within, and downstream of the treatment site. DBW also monitored Agridex at all the 97 herbicide sampling events. In every case, Agridex concentrations were non-detectable. **Table 3-14,** on the next page, provides

Table 3-14
Calculated* Maximum Concentrations of Agridex Immediately Following WHCP Treatment

Concentration of:	Agridex *(total adjuvant)*
1. Chemical directly out of spray nozzle	5,000 ppm
2. Chemical in 1 meter deep water, @ 100% water contact	1.24 ppb
3. Chemical in 2 meter deep water, @ 100% water contact	0.62 ppb
4. Chemical in 1 meter deep water, @ 20% water contact	0.25 ppb
5. Chemical in 2 meter deep water, @ 20% water contact	0.12 ppb

* The concentrations above are based on the pounds of active ingredient in maximum specified application rate per acre, and an appropriate dilution factor based on the volume of water in the tank mix, or within one or two meter-acres.

Table 3-15
Calculated* Maximum Concentrations of Competitor Immediately Following WHCP Treatment

Concentration of:	Competitor *(total adjuvant)*
1. Chemical directly out of spray nozzle	5,000 ppm
2. Chemical in 1 meter deep water, @ 100% water contact	1.24 ppb
3. Chemical in 2 meter deep water, @ 100% water contact	0.62 ppb
4. Chemical in 1 meter deep water, @ 20% water contact	0.25 ppb
5. Chemical in 2 meter deep water, @ 20% water contact	0.12 ppb

* The concentrations above are based on the pounds of active ingredient in maximum specified application rate per acre, and an appropriate dilution factor based on the volume of water in the tank mix, or within one or two meter-acres.

estimates of Agridex concentrations directly out of the spray nozzle and under different dilution assumptions. Note that Table 3-14 includes both ppm and ppb concentrations.

Competitor

Competitor is a modified vegetable oil containing a non-ionic emulsifier system. It may be used as an adjuvant with aquatically labeled pesticides. Competitor has not been used previously in the WHCP; however, USFWS utilizes Competitor when treating water hyacinth in Stone Lakes National Wildlife Refuge. The active ingredients in Competitor are ethyl oleate, sorbitan alkylpolyethoxylate ester, and dialkyl polyoxyethylene glycol. These ingredients make up 98 percent by weight, with the remaining 2 percent constituents that are ineffective as spray adjuvant. Competitor will be used at a rate of one to four pints per acre (to a maximum of 1 percent volume/ volume ratio). **Table 3-15,** left, provides estimates of Competitor concentration immediately out of the spray nozzle and at different dilution assumptions. Note that Table 3-15 includes both ppm and ppb concentrations.

3. Hand-Picking

Hand-picking of water hyacinth will be conducted primarily when or where chemical treatment cannot be made, and may occur throughout the year. As treatment crews survey for water hyacinth, they will conduct hand-picking in selected areas. The goals of the hand-picking aspect of the program are to aid in the control of water hyacinth and reduce impacts of chemical application by clearing areas that are not accessible to chemical

treatment, subject to high infestation, nurseries, and within emergent vegetation.

Crews will follow specific hand-picking protocols to ensure the protection of water quality and special status species. Reflecting a typical season of hand-picking, between October 15, 2007, and April 1, 2008, treatment crews collected over 4,000 thirty-gallon barrels of water hyacinth. Once collected, water hyacinth will be deposited on at authorized disposal sites, to decompose.

4. Herding

Herding refers to the moving of water hyacinth mats by pushing or pulling mats from one location to another. Mats will be moved to removal locations or to the main channel. Once in a main channel, the water hyacinth will flow out of the Delta, into saline waters and die. Water hyacinth cannot survive in waters of greater than 2 ppt to 2.5 ppt saline water (brackish water).

For herding water hyacinth out of the Delta, field supervisors will take into account tides, storm events, and dam releases to select appropriate days and times for herding to take place. Crews will not herd in areas where physical damage to emergent, native vegetation is likely to occur such as among stands of cattails (*Typha* spp.), *Phragmites* spp., bulrushes (*Scirpus* spp.), or native cordgrass (*Spartina foliosa*). In addition, the total amount of water hyacinth herded in one area will be limited to avoid impeding navigation. Due to timing and logistical limitations of herding activities, this method may not be used as frequently as handpicking.

The WHCP will also utilize herding in conjunction with mechanical removal, as described below. Crews will push mats or sections of mats toward an excavator located on a boat ramp. This will maximize the amount of water hyacinth that can be removed by the stationary excavator.

5. Mechanical Removal

The WHCP will utilize two different mechanical removal approaches. The first approach will be to park a small excavator and dump truck on a concrete boat ramp and mechanically lift water hyacinth from the waterway surrounding the ramp. Crews will support the excavation by herding water hyacinth that is outside of the excavator's reach closer to the equipment. This mechanical removal approach will be used only in limited locations when water hyacinth growth is concentrated near a boat ramp. There may be relatively few locations within the Delta that are appropriate for excavation.

Photo: Mechanical removal with excavator.

Photo: Mechanical cutter and conveyor equipment.

The second approach will utilize mechanical equipment designed specifically to safely remove aquatic weeds from waterways. This mechanical equipment utilizes cutters and conveyors to physically remove the plant from the water, and onto the bed of the equipment. The equipment will collect and unload vegetation using a conveyor system on a boom, adjustable to the appropriate cutting height (two to three feet below the surface for water hyacinth). Cutter bars will collect material and bring it aboard the vessel using the conveyor; when the vessel has reached capacity (between 2,000 and 15,000 pounds of plant material), the cut plant material will be offloaded to a dump truck parked at a nearby boat ramp to offload water hyacinth. Water hyacinth will be disposed of at an authorized location, typically utilizing nearby farm fields.

Mechanical removal can be costly, it will be used to supplement chemical treatment and when immediate removal of weeds is required. Mechanical removal will primarily be utilized to remove dense mats of water hyacinth in locations where chemical treatment must be avoided, such as sites with many valley elderberry shrubs along the shoreline. WHCP environmental scientists will consult the IEP database and survey mechanical removal sites immediately prior to weed removal to ensure that no listed species are present. If listed species are present, mechanical removal operations at that site will be postponed. Similar mechanical equipment is regularly used to control water hyacinth in Florida and other Southeastern states.

The WHCP will implement an operation protocol similar to the protocol for chemical treatment prior to conducting mechanical removal. WHCP environmental scientists will check IEP monitoring data to ensure that salmon species are not present at the removal site. In addition, the equipment operator will utilize the same Environmental Checklist to evaluate presence of listed species or sensitive habitats. If listed species or sensitive habitats are present, the operator will not conduct mechanical removal at that site.

The WHCP has not utilized this method of mechanical removal in prior years. Studies of mechanical removal conducted during 2003 and 2004 in the Delta by the San Francisco Estuary Institute (SFEI) (Greenfield et al 2005; Spencer at al 2005; Greenfield and McNabb, 2005) raised concerns about the potential for water hyacinth plant cuttings from mechanical removal to grow and spread within the Delta. However, current removal approaches reduce the potential for plant fragments to lead to increased infestations (Fowler (personal communication), 2012).

6. Biological Controls

Biological control is the use of biological agents, typically insects or pathogens, to control undesirable plants. The WHCP has experimented with biological controls, with limited success, since inception of the program.

In 1982, the USDA-ARS first released the water hyacinth-eating weevil, *Neochetina bruchi*, in the Delta. Following the initial releases of *Neochetina bruchi*, USDA-ARS released other host-specific species (*Neochetina eichhorniae* and *Sameodes albiguttailis*).

Recent surveys have shown that *Neochetina bruchi* is the only species to have survived and spread throughout the Delta. However, the small size of *Neochetina bruchi* populations have failed to effectively control water hyacinth. From 2003 to 2006, the DBW contracted with the California Department of Food and Agriculture (CDFA) to examine populations of *Neochetina bruchi* in an effort to understand the impacts and dynamics of *Neochetina bruchi* populations in the Delta.

A CDFA study demonstrated the challenge of biological control in the Delta (Akers and Pitcairn 2006). The study found a mismatch between the life cycle of the weevil, and the climate and growing cycle of water hyacinth in the Delta. Weevils have limited survival during the winter, because the 7°C average temperature in the Delta (Akers and Pitcairn 2006) is well below *Neochetina bruchi* optimum feeding and oviposition temperatures, at 30°C (Julien 2001).

In the spring, when water hyacinth starts to grow rapidly, weevil populations are too low to effectively damage the plant. In October, when the weevil population has increased to a level where it might provide some control, the plant is starting to decline. In addition, perhaps because of low humidity in the Delta, plant weevil populations that provide effective control in other regions (at least 5 weevils per plant), do not provide control in the Delta. Akers and Pitcairn summarize, "the weevils do not exert a level of damage consistent enough to bring the weed under control" (Akers and Pitcairn 2006).

These findings are consistent with evaluations of success and failure factors related to biological control of water hyacinth. Factors that may reduce the effectiveness of biological controls include: temperate climates, high nutrient status of the water, periodic flooding or drought conditions, and uptake of heavy metals by water hyacinth (Julien 2001). All of these factors are present in the Delta.

When it is effective, biological control of water hyacinth could be attractive because of low potential environmental impacts, long-term sustainability, and low cost. In the Delta, biological control has been shown to have severely limited effectiveness. In addition, researchers and waterway managers recommend that biological control alone is not a solution, and it should be part of an integrated management approach (Labrada 1995, Julien 2001, Center et al 1999). The DBW will continue to evaluate and incorporate biological control as part of WHCP, but will not rely on biological agents to control water hyacinth in the Delta.

While successful implementation of biological control for water hyacinth is challenging in the Delta, WHCP continues to evaluate and consider new alternatives. In

the past, DBW funded research at UC Davis to identify plant pathogens in the Delta with potential for controlling water hyacinth. However, to date there have been no practical applications of plant pathogens to control water hyacinth in the Delta. Plant pathogens, in combination with other mechanisms, may be a promising future alternative for water hyacinth control (Charudattan 2001).

The most recent biological control utilized in the Delta is the plant hopper (*Megamelus scutellaris*). The California Department of Food and Agriculture (CDFA) released over 5,000 leafhoppers in three locations (Whisky Slough, Willow Creek (in Folsom), and Seven Mile Slough) in July 2011. *Megamelus scutellaris* is native to Peru, Brazil, Uruguay, and Argentina. It was approved for use in the United States for control of water hyacinth in 2010 after extensive testing and quarantine (USDA 2010).

After the first winter (2011/2012), plant hopper survival results were mixed (Pitcairn, personal communication, 2012). Plant hoppers released on ponds in the Folsom/Lake Natoma region had good survival rates. The Whiskey Slough release site was chemically treated later in the same season, and plant hoppers did not remain on the dead plants. The Seven Mile Slough site appears to have over-wintered plant hoppers into 2012. CDFA funding for the plant hopper program was eliminated, and the plant hopper colony is now being managed by USDA-ARS. USDA-ARS may place colonies in specific sites that will not be otherwise treated. Although plant hoppers cause significant damage to water hyacinth in greenhouses, establishment of colonies of in the Delta is much slower, resulting in limited efficacy to-date.

In addition to plant hoppers, USDA-ARS is working with the Argentinian government to collect and export other natural water hyacinth biological control agents. Any new biological controls will not be introduced until they have gone through USDA's extensive testing, quarantine, and permitting procedures.

F. Estimated Efficacy for WHCP Methods

In any given year, the extent of water hyacinth growth in the Delta is dependent on the interactions between many different factors, including:

- Winter air and water temperatures (colder temperatures kill back plants in the winter, although water hyacinth in the Delta can withstand short periods of below-freezing temperatures (Santos et al. 2009))
- Spring temperatures (warm temperatures increase water hyacinth growth during periods when treatment is restricted)
- Summer temperatures (water hyacinth grows faster in warmer temperatures)
- Heavy rainfall (increases water hyacinth when previously dry riverbeds and shorelines act as nursery areas, but decreases water hyacinth when plants are flushed out of the Delta by heavy flows)
- Light rainfall (minimizes the extent to which water hyacinth is flushed out of the Delta in the winter)

- Fall season infestation levels (high fall infestation levels may result in more water hyacinth over-wintering)
- Nutrient load (higher nutrient loads may promote water hyacinth growth)
- Nursery infestations (high infestations in nurseries result in more plants moving into Delta waters during the growth season)
- Salinity levels (salinity of 2.0 to 2.5 ppt is toxic to water hyacinth (Haller et al. 1974); this is at the low end of brackish water, thus intrusion of saltwater into the Delta could reduce infestations; Khanna et al. (2009) found that seasonal variability of salinity levels in the Delta allows water hyacinth to occur throughout the Delta, even with extreme sensitivity to salinity).

Most of these factors are beyond the influence of WHCP. To the extent that water hyacinth is successfully controlled by the end of the treatment season, particularly in nursery areas, infestations in the following season will be less than they would otherwise. Unfortunately, even a successful treatment season can be offset by weather conditions that support heavy hyacinth growth in the following year. Center and Spencer (1981) note that the ability of water hyacinth to rapidly re-establish populations following extreme perturbations makes eradication difficult.

The WHCP will continuously monitor program effectiveness through field surveys (pre-, mid-, and post-season). At the end of each treatment season, WHCP will prepare a report summarizing the extent of control achieved during the treatment season. The report will discuss the impact of external factors such as temperature and rainfall, in addition to specific control mechanisms, on water hyacinth infestation levels.

To support field surveys, the USDA-ARS and DBW may implement aerial color and color infrared surveys to assess late season water hyacinth (and other macrophyte species) coverage. These surveys could help document efficacy and improve understanding of ecosystem impacts of water hyacinth treatment, for example the extent of growth of pennywort and water primrose in areas previously infested with water hyacinth.

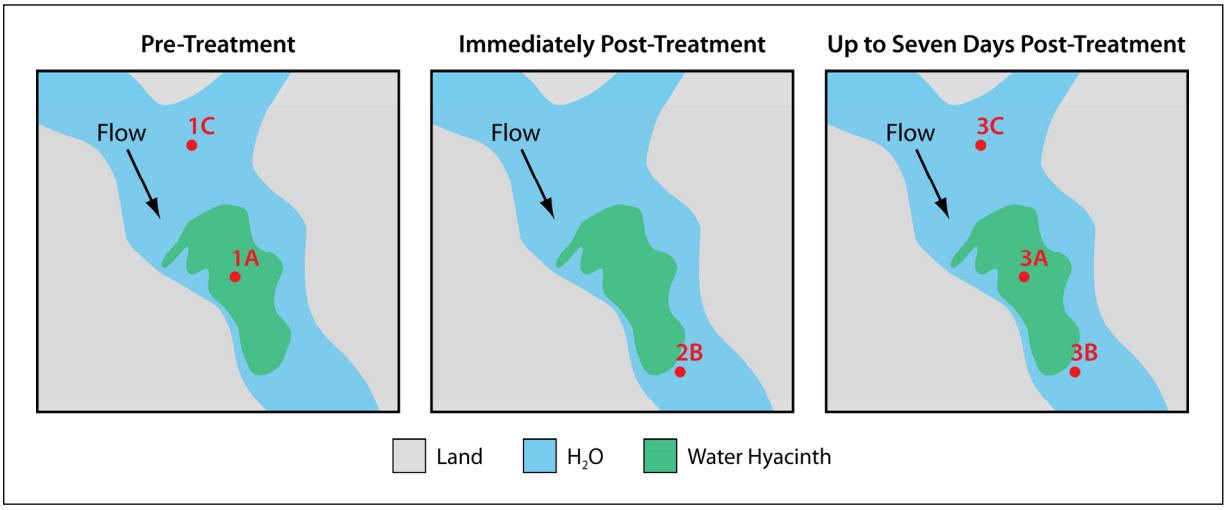
Another measure of effectiveness of the WHCP is the number of public complaints about water hyacinth infestations. While this method is subjective, it provides an indication of the extent to which the WHCP is effective in reducing the impact of water hyacinth on navigation and recreation in the Delta. For example, the number of citizens that called to complain about water hyacinth increased significantly in 2011, as compared to prior years, when treatments did not start until September.

G. Monitoring Protocols for WHCP

The WHCP will conduct extensive monitoring for the program. The WHCP will be responsible for collecting water quality monitoring data, as well as collecting water samples for chemical residue testing.

Based on NPDES permit requirements, WHCP will follow a monitoring protocol. This protocol has historically fulfilled requirements of the Regional Water Quality

Figure 3-14
WHCP Monitor Sites



Control Board, NOAA Fisheries, and USFWS. At each monitoring site, WHCP's environmental scientists will take samples immediately pre-application (upstream and adjacent to the water hyacinth mat), and immediately post-application (downstream of the treatment area). WHCP environmental scientists will also take samples one week following treatment (upstream, adjacent to, and downstream of the treatment area). At each sampling event, environmental scientists will take samples from the following six locations, illustrated in **Figure 3-14,** above:

- 1A – Pre-treatment, in site
- 1C – Pre-treatment, control
- 2B – Immediately post-treatment, downstream
- 3A – Within 7 days, in site
- 3B – Within 7 days, downstream
- 3C – Within 7 days, control.

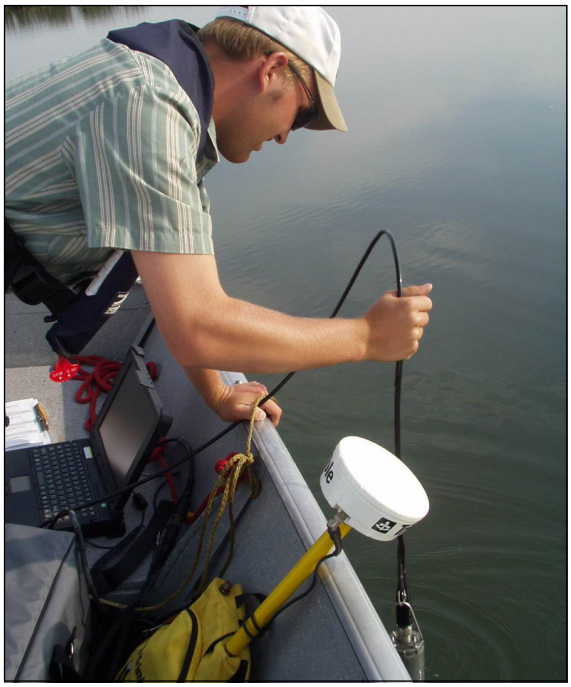


Photo: Water hyacinth monitoring.

The WHCP will select monitoring sites that reflect a mix of water types (tidal, riverine, and tidal dead-end), herbicides, and different habitat types. The WHCP will revise the monitoring approach to comply with the new NPDES General Permit, as described below.

Table 3-16
WHCP Environmental Monitoring Requirements

Treatment Crews (for each site treated)	Environmental Scientists (for each sample event)
1. Water temperature (°C) 2. Dissolved oxygen (DO, mg/L or parts per million (ppm)) 3. Wind speed (mph) 4. Coordinates of treatment location 5. Presence of elderberry shrubs 6. Presence of species of concern 7. Acres treated 8. Quantity of herbicide and adjuvant	1. Water temperature (°C) 2. Dissolved oxygen (DO, mg/L or ppm) 3. Turbidity (NTU) 4. pH 5. Salinity (ppt) 6. Specific conductance (mS/cm) 7. Water depth (feet) 8. Tide cycle 9. Water samples (pre-treatment, post-treatment, control; submitted to a Certified Analytical Laboratory)

At each monitoring site, WHCP environmental scientists will monitor dissolved oxygen, turbidity, pH, and several other water quality measures. WHCP environmental scientists will collect water in bottles, packed in ice, and submit them to a Certified Analytical Laboratory to measure chemical residue levels.

Coordination between treatment crews and monitoring crews will be very structured. Treatment and monitoring plans will be established in advance. Before any treatment or monitoring, crews will confer to make sure both crews know what sites will be treated and monitored on that day. The treatment crew will stand by until the monitoring crew completes the pre-treatment sampling, at which time the monitoring crew will give the treatment crew the "all clear" to begin treatment. The treatment crew will contact the monitoring crew as soon as treatment is complete so post-treatment monitoring can begin as required. Treatment and monitoring crews will be in separate vessels. Monitoring vessels will not carry herbicide to minimize any contamination that might occur.

Environmental scientists plan to also conduct special monitoring of dissolved oxygen to determine the impact of water hyacinth and the WHCP on DO levels. For this study, crews will measure DO to evaluate the impact of water hyacinth and water hyacinth treatments on DO.

WHCP treatment crews will conduct daily monitoring, in addition to the extensive monitoring to be conducted by WHCP environmental scientists. Treatment crews will monitor and report pre- and post-treatment dissolved oxygen, wind speed, temperature, acres treated, quantity of herbicide and adjuvant, presence of elderberry shrubs or other species of concern, and coordinates of treatment location. **Table 3-16,** above, lists monitoring requirements for WHCP environmental scientists and WHCP treatment crews.

Table 3-17
General Permit Receiving Water Limits or Monitoring Triggers for WHCP Herbicides

Herbicide Active Ingredient	Maximum Limitation
2,4-D	70 ppb
Glyphosate	700 ppb
Penoxsulam	10.1 ppm
Imazapyr (isopropylamine salt)	11.2 ppm

The State Water Quality Control Board is updating the NPDES General Permit, with a draft for public comment released on June 27, 2012, and a final version for Board approval expected in November 2012. A copy of the draft NPDES General Permit is provided in the Supplemental Materials Binder. The June 27, 2012 preliminary version of the General Permit maintains a similar monitoring protocol as described in Figure 3-14. However, the new General Permit requires a sampling frequency of six application events per year for each environmental setting (flowing water and non-flowing water), per herbicide. Glyphosate will require sampling for only one application event per year, based on the low herbicide levels found in prior year sampling. Once WHCP has provided the SWRCB with results from six consecutive application events showing concentrations that are less than the receiving water limitation/trigger for an active ingredient in a specific environmental setting, WHCP sampling shall be reduced to one application event per year for that active ingredient in that environmental setting. **Table 3-17,** left, provides the receiving water limits and monitoring triggers for the four potential WHCP herbicides. These maximum limitations are all above the calculated maximum concentrations for 2,4-D, glyphosate, penoxsulam, and imazapyr in Tables 3-5, 3-8, 3-11, and 3-13. The SWRCB will add imazamox to the General Permit now that it is approved for use in California. The WHCP will revise monitoring protocols, as appropriate, to comply with the new NPDES General Permit.

H. Mitigation Measures for WHCP

The WHCP will implement a number of mitigation measures to minimize or reduce potential impacts of the program. These mitigation measures have been developed over time, working with USFWS, NMFS, the State Water Board, and local Agricultural Commissioners. **Exhibit 3-4,** starting on the next page, describes twenty (20) WHCP mitigation measures that WHCP will regularly implement to reduce or eliminate potential impacts of the WHCP. The WHCP plans to revise any mitigation measures that have changed since the 2009 WHCP PEIR, to reflect the mitigation measures in Exhibit 3-4, prior to the start of the 2013 treatment season.

Exhibit 3-4
WHCP Mitigation Measures Summary

Mitigation Measures Summary	Mitigation Measures Description
1. Avoid herbicide application and mechanical removal near special status species, and sensitive riparian and wetland habitat; and other biologically important resources	Each year, prior to the start of the treatment season, WHCP will conduct field crew environmental awareness training. Under this training, crews will be informed about the presence and life histories of special status species; habitats associated with species; sensitive habitats and wetlands; the terms and conditions of the program's biological opinions; incidental take procedures; and that unlawful take of an animal or destruction of its habitat is a violation of the Endangered Species Act. WHCP also will provide crews with a field guide (Species Identification Deck) for easy identification of special status species on-site. Prior to treating a site, crews will conduct a visual survey to determine whether special status plants, animals, or sensitive habitats are present. Crews will complete an Environmental Observations Checklist for each site to document the presence or absence of special status species. If any special status species or sensitive habits are present at the site, the field crew will not perform any treatment.
2. Provide a 50 foot buffer between treatment sites and shoreline elderberry shrubs (*Sambucus* ssp.), host plant for the valley elderberry longhorn beetle *(Desmocerus californicus dimorphus)*	WHCP will conduct a survey of treatment sites to prepare a map that identifies locations of elderberry shrubs, and provide this map to field crews. WHCP crews will maintain the 50 buffer zone for herbicide treatments when elderberry shrubs are present. Crews will also conduct treatments downwind of elderberry shrubs. In addition, WHCP environmental scientists will survey a sample of elderberry shrubs which could be potentially impacted by application activities at the beginning of the treatment season, and at the end of the treatment season. The environmental scientists will compare the health of elderberry shrubs at control sites (i.e. not adjacent to treatments) with elderberry shrubs located adjacent to treated sites. If elderberry shrubs located near treated sites show signs of adverse effects from treatment, WHCP will develop additional mitigation measures to protect elderberry shrubs (for example, increasing the size of the buffer zone).
3. Conduct herbicide treatments in order to minimize potential for drift	In addition to complying with the label application requirements, WHCP will, to the degree possible, schedule herbicide applications to occur at high tide, or at a point in the tidal cycle determined by the field supervisor to provide the least non-target impact at a particular site. In general, treatment at high tide will allow for better spray accuracy and access, and will provide for greater dilution volume of herbicides. WHCP crews will change nozzle type and spray pressures whenever conditions warrant, limiting the amount of herbicide which may inadvertently contact non-target species or enter the water.
4. Operate program vessels in a manner that causes the least amount of disturbance to the habitat	Operational procedures for WHCP vessels will minimize boat wakes and propeller wash. These procedures will be particularly important in shallow water, or other sensitive habitats.
5. Implement temporal and spatial limitations and restrictions on herbicide treatments and mechanical removal to minimize treatments during times, and at locations, where larval and/or migratory fish are likely to be present	The specific locations and times followed in the WHCP in the past have been guided by the prior biological opinions. In the future, WHCP will implement a survey-based approach to conducting treatments that allows for early season treatments in areas with re-growing water hyacinth when listed fish species are <u>not present</u>. In addition, WHCP will follow calendar year treatment dates when listed fish species may be present (although treatments will not be conducted in sites where IEP data shows listed fish are present). These treatment time restrictions minimize potential exposure of migratory salmonids and sensitive juvenile fish to WHCP herbicides. Figure 3-3 summarizes treatment timing.

Exhibit 3-4

WHCP Mitigation Measures Summary *(continued)*

Mitigation Measures Summary	Mitigation Measures Description
6. Monitor herbicide and adjuvant levels to ensure that WHCP does not result in potentially toxic concentrations of chemicals in Delta waters	WHCP will conduct comprehensive monitoring. This monitoring is in compliance with the general NPDES permit, and prior NMFS and USFWS Biological Opinions. WHCP will collect samples prior to treatment, immediately after treatment, and post-treatment within one week of spraying. WHCP will conduct water quality monitoring for visual parameters, physical parameters, and chemical parameters at a specified number of sites it treats for each pesticide, per water body type. Water samples will be submitted to a certified analytical laboratory to measure 2,4-D, glyphosate, and penoxsulam, imazamox, and imazapyr, as appropriate, and adjuvant levels. Should these levels exceed allowable limits, WHCP will take immediate measures to reduce chemical levels at future treatment sites.
7. Implement an adaptive management approach to minimize the use of herbicides	Under an adaptive management approach, WHCP will seek to improve efficacy and reduce environmental impacts over time as new and better information is available. Specifically, WHCP will evaluate the need for control measures on a site by site, month-to-month, basis; select appropriate indicators for pre-treatment monitoring; monitor indicators following treatment and evaluate data to determine program efficacy and environmental impacts; support ongoing research to explore impacts of the WHCP and alternative control methodologies; report findings to regulatory agencies; and adjust program actions, as necessary, in response to recommendations and evaluations by DBW staff, USDA-ARS, regulatory agencies and stakeholders. In addition to this adaptive management approach, WHCP will follow maintenance control practices that from a program standpoint seek to reduce the number of acres of water hyacinth to be treated each year, until treatment acreage reaches a minimal level. This will reduce the volume of herbicide utilized by the WHCP.
8. Provide treatment crews with electronic mapping that identifies previously surveyed areas for giant garter snake habitat	WHCP application crews will use this map as a tool for performing pre-application visual inspections for the presence of giant garter snakes. If giant garter snakes are present, treatment crews will not treat at that location.
9. Monitor dissolved oxygen levels pre- and post-treatment for all WHCP treatments, and at selected locations in the Delta over time	Based on the pre-treatment DO levels, the WHCP application crew will determine whether to conduct treatment at that site. No treatment will be performed when dissolved oxygen levels are between 3 ppm (the level below which DO is considered to be detrimental to fish species) and the basin plan limits established by the Central Valley Regional Water Quality Control Board (CVRWQCB). The basin plan limits depend on location and time of year, and range from 5 ppm to 8 ppm. DBW will maintain written and map summaries of specific DO numeric limits. When pre-treatment levels are below 3 ppm, fish species are not likely to be present due to the extremely low oxygen levels. When pre-treatment levels are above the basin plan limit, WHCP treatments, following label guidelines and mitigation measures, are not expected to adversely affect special status fish, resident native or migratory fish, or sensitive riparian or wetland habitats. To further monitor the impact of water hyacinth and the WHCP on DO levels, field crews will measure DO at several representative sites to evaluate the impact of water hyacinth and water hyacinth treatments on DO.

Exhibit 3-4
WHCP Mitigation Measures Summary *(continued)*

Mitigation Measures Summary	Mitigation Measures Description
10. Follow the new fish passage protocol to reduce the potential for low dissolved oxygen levels from decaying water hyacinth to negatively impact listed fish species	WHCP will follow current label requirements regarding dissolved oxygen impacts for each WHCP herbicide in order to avoid impacts to listed fish species. These requirements are detailed in the fish passage protocol provided in the Supplemental Materials Binder. Depending on the herbicide, these requirements include treating in strips, and specific wait times between treatments
11. Collect plant fragments during and immediately following treatment	To maximize containment of plant fragments, WHCP crews will collect water hyacinth fragments. Crews will also be trained on the importance of minimizing fragment escape.
12. Conduct handpicking and herding only as required	WHCP will limit handpicking and herding activities. In the unlikely event that water hyacinth fragments escape the raking and/or nets, the dormant plants are more likely to be washed out of the Delta, and less likely to become established, than if they had escaped during the growing season.
13. Identify and utilize disposal areas for mechanically removed and hand-picked water hyacinth that have no and/or low habitat value for the federal and State listed giant garter snake *(Thamnophis gigas)*	WHCP will provide crews electronic mapping that identifies previously surveyed areas for giant garter snake habitat. Crews also will conduct surveys to ensure that there are no other special status plant or animal species located within 50 feet of disposal sites. Mechanically removed water hyacinth will be disposed of only in pre-approved disposal sites.
14. Identify and utilize disposal areas for mechanically removed and hand-picked water hyacinth that are at least 50 feet away from elderberry shrubs (*Sambucus* ssp.) and 50 feet away from aquatic giant garter snake habitat	WHCP disposal will not occur near elderberry shrubs, which are potential habitat for the federally threatened valley elderberry longhorn beetle (*Desmocerus californicus dimorphus*). WHCP disposal will not occur near aquatic giant garter snake habitat. Mechanically removed water hyacinth will be disposed of only in pre-approved disposal sites.
15. Minimize public exposure to herbicide treated water	Prior to the treatment season, WHCP will release a public notice announcing the program. WHCP treatments generally take place in heavily infested waterways, which are usually unsuitable for water recreation. However, if recreationists are present when treatment occurs, treatments crews will inform recreationists about the treatment, asking them to move to a different location, or move treatments to a different location.
16. Require treatment crews to participate in training on herbicide and heat hazards	WHCP will provide training to ensure that treatment crews have the knowledge and tools necessary to conduct the program in a safe manner. Training will include reading, understanding, and following herbicide label requirements; purpose and proper use of Personal Protective Equipment; symptoms of herbicide poisoning and minimization of exposure; avoidance, symptoms, and treatment of heat exposure; and emergency medical procedures.

Exhibit 3-4

WHCP Mitigation Measures Summary *(continued)*

Mitigation Measures Summary	Mitigation Measures Description
17. Follow best management practices to minimize the risk of spill and to minimize the impact of a spill, should one occur	The WHCP best management practices include several provisions to reduce the potential for spill, such as: fastening herbicide containers securely in boats in original, watertight containers; carrying a marker buoy and anchor line to mark any spills in water; reporting spills immediately to appropriate State and local agencies; immediately stopping movement of land spills using absorbing materials; marking and monitoring spills in water for herbicide residues and environmental impacts, if appropriate. Treatment crews will include at least one person with a Qualified Applicators Certificate (QAC), and all crew members will participate in annual training on herbicide handling procedures.
18. Implement safety precautions on hot days to prevent heat illness	In addition to annual training on heat illness prevention, and compliance with CalOSHA's California Heat Illness Prevention Standard, WHCP Field Supervisors will conduct special training sessions on days when weather is expected to be hot. This training will cover the symptoms of heat illness, and immediate actions to take should any symptoms occur. Field Supervisors will cancel treatments if the weather is exceptionally hot. WHCP will also provide bimini tops (shade covers) for WHCP treatment boats.
19. Follow the Memorandum of Understanding (MOU) protocol for herbicide applications within one (1) mile of Contra Costa Water District (CCWD) drinking water intake facilities	The MOU is an agreement between CCWD and DBW. Generally, no applications shall occur within Rock Slough, or within one mile of the confluence of Rock Slough and Old River, or within one mile of CCWD's Old River or Mallard Slough intake pumps without consensual agreement between CCWD and DBW. Herbicide applications within one mile of CCWD's water intakes may only occur with prior consent of CCWD. In order to treat within one mile of an intake, WHCP must notify CCWD at least two weeks in advance, and make every reasonable attempt to schedule applications during periods when CCWD's intakes are shut down for environmental or maintenance reasons, allowing at least two complete tidal cycles between application and restart. This measure is primarily aimed at reducing the potential for drinking water contamination from the WHCP.
20. Notify County Agricultural Commissioners about WHCP activities	Before an application may occur, WHCP shall file Pesticide Use Recommendations (PUR) and a Notice of Intent (NOI) with the appropriate County Agricultural Commissioner (CAC) office, when required for restricted materials or as requested by each county. Each NOI will include the site number, spray dates, locations, and herbicides and adjuvants to be used. NOIs will be submitted before the upcoming treatment week. Based on information in the NOIs, CAC's could inform land owners of particular periods of time during which irrigation should not occur. If necessary, WHCP shall also obtain a Restricted Use Permit (RUP) from all appropriate CACs.

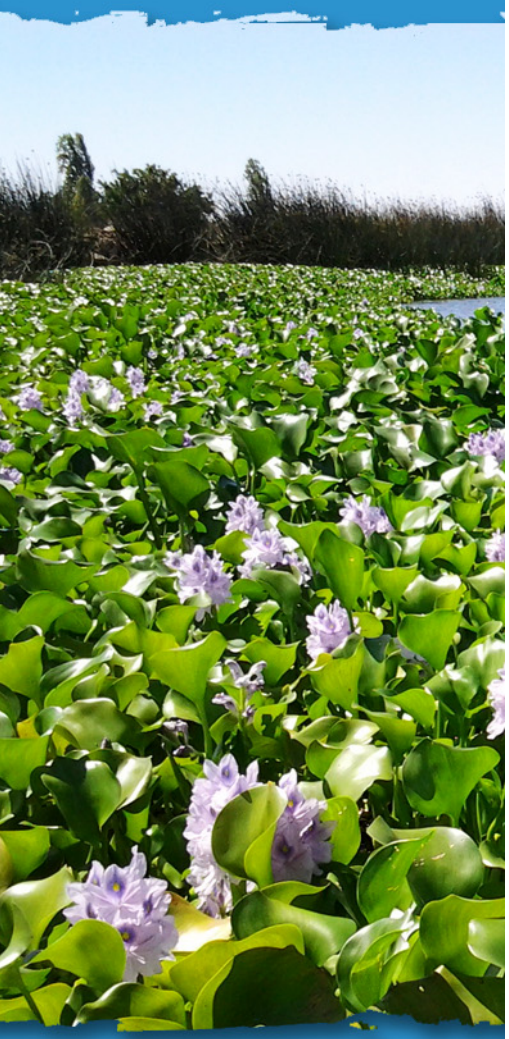
[i] The legal definition of the Sacramento-San Joaquin Delta is as follows. These boundaries are reflected in Exhibit 1-2. 12220. The Sacramento-San Joaquin Delta shall include all the lands within the area bounded as follows, and as shown on the attached map prepared by the Department of Water Resources titled "Sacramento-San Joaquin Delta," dated May 26, 1959:

Beginning at the Sacramento River at the I Street bridge proceeding westerly along the Southern Pacific Railroad to its intersection with the west levee of the Yolo By-Pass; southerly along the west levee to an intersection with Putah Creek, then westerly along the left bank of Putah Creek to an intersection with the north-south section line dividing sections 29 and 28, T8N, R6E; south along this section line to the northeast corner of section 5, T7N, R3E; west to the northwest corner of said section; south along west boundary of said section to intersection of Reclamation District No. 2068 boundary at northeast corner of SE 1/4 of section 7, T7N, R3E; southwesterly along Reclamation District No. 2068 boundary to southeast corner of SW 1/4 of section 8, T6N, R2E; west to intersection of Maine Prairie Water Association boundary at southeast corner of SW 1/4 of section 7, T6N, R2E; along the Maine Prairie Water Association boundary around the northern and western sides to an intersection with the southeast corner of section 6, T5N, R2E; west to the southwest corner of the SE 1/4 of said section; south to the southwest corner of the NE 1/4 of section 7, T5N, R2E; east to the southeast corner of the NE 1/4 of said section; south to the southeast corner of said section; west to the northeast corner of section 13, T5N, R1E; south to the southeast corner of said section; west to the northwest corner of the NE 1/4 of section 23, T5N, R1E; south to the southwest corner of the NE 1/4 of said section; west to the northwest corner of the SW 1/4 of said section; south to the southwest corner of the NW 1/4 of section 26, T5N, R1E; east to the northeast corner of the SE 1/4 of section 25, T5N, R1E; south to the southeast corner of said section; east to the northeast corner of section 31, T5N, R2E; south to the southeast corner of the NE 1/4 of said section; east to the northeast corner of the SE 1/4 of section 32, T5N, R2E; south to the northwest corner of section 4, T4N, R2E; east to the northeast corner of said section; south to the southwest corner of the NW 1/4 of section 3, T4N, R2E; east to the northeast corner of the SE1/4 of said section; south to the southwest corner of the NW 1/4 of the NW 1/4 of section 11, T4N, R2E; east to the southeast corner of the NE 1/4 of the NE 1/4 of said section; south along the east line of section 11, T4N, R2E to a road intersection approximately 1000 feet south of the southeast corner of said section; southeasterly along an unnamed road to its intersection with the right bank of the Sacramento River about 0.7 mile upstream from the Rio Vista bridge; southwesterly along the right bank of the Sacramento River to the northern boundary of section 28, T3N, R2E; westerly along the northern boundary of sections 28, 29, and 30, T3N, R2E and sections 25 and extended 26, T3N, R1E to the northwest corner of extended section 26, T3N, R1E; northerly along the west boundary of section 23, T3N, R1E to the northwest corner of said section; westerly along the northern boundary of sections 22 and 21, T3N, R1E to the Sacramento Northern Railroad; southerly along the Sacramento Northern Railroad; southerly along the Sacramento Northern Railroad to the ferry slip on Chipps Island; across the Sacramento River to the Mallard Slough pumping plant intake channel of the California Water Service Company; southward along the west bank of the intake channel and along an unnamed creek flowing from Lawler Ravine to the southern boundary of the Contra Costa County Water District; easterly along the southern boundary of the Contra Costa County Water District to the East Contra Costa Irrigation District boundary; southeasterly along the southwestern boundaries of the East Contra Costa Irrigation District, Byron-Bethany Irrigation District, West Side Irrigation District and Banta-Carbona Irrigation District to the northeast corner of the NW 1/4 of section 9, T3S, R6E; east along Linne Road to Kasson Road; southeasterly along Kasson Road to Durham Ferry Road; easterly along Durham Ferry Road to its intersection with the right bank of the San Joaquin River at Reclamation District No. 2064; southeasterly along Reclamation District No. 2064 boundary, around its eastern side to Reclamation District No. 2075 and along the eastern and northern sides of Reclamation District No. 2075 to its intersection with the Durham Ferry Road; north along the Durham Ferry Road to its intersection with Reclamation District No. 17; along the eastern side of Reclamation District No. 17 to French Camp Slough; northerly along French Camp Turnpike to Center Street; north along Center Street to Weber Avenue; east along Weber Avenue to El Dorado Street; north along El Dorado Street to Harding Way; west along Harding Way to Pacific Avenue; north along Pacific Avenue to the Calaveras River; easterly along the left bank of the Calaveras River to a point approximately 1,600 feet west of the intersection of the Western Pacific Railroad and the left bank of said river; across the Calaveras River and then north 18* 26' 36 west a distance of approximately 2,870 feet; south 72* 50' west a distance of approximately 4,500 feet to Pacific Avenue (Thornton Road); north along Pacific Avenue continuing onto Thornton Road to its intersection with the boundary line dividing Woodbridge Irrigation District and Reclamation District No. 348; east along this boundary line to its intersection with the Mokelumne River; continuing easterly along the right bank of the Mokelumne River to an intersection with the range line dividing R5E and R6E; north along this range line to the Sacramento-San Joaquin County line; west along the county line to an intersection with Reclamation District No. 1609; northerly along the eastern boundary of Reclamation District No. 1609 to the Cosumnes River, upstream along the right bank of the Cosumnes River to an intersection with the eastern boundary of extended section 23, T5N, R5E; north along the eastern boundary of said extended section to the southeast corner of the NE 1/4 of the NE 1/4 of said extended section; west to the southeast corner of the NE 1/4 of the NW 1/4 of extended section 14, T5N, R5E; west to an intersection with Desmond Road; north along Desmond Road to Wilder-Ferguson Road; west along Wilder-Ferguson Road to the Western Pacific Railroad; north along the Western Pacific Railroad to the boundary of the Elk Grove Irrigation District on the southerly boundary of the N 1/2 of section 4, T5N, R5E; northerly along the western boundary of the Elk Grove Irrigation District to Florin Road; west on Florin Road to the eastern boundary of Reclamation District No. 673; northerly around Reclamation District No. 673 to an intersection with the Sacramento River and then north along the left bank of the Sacramento River to I Street bridge. Section, range, and township locations are referenced to the Mount Diablo Base Line and Meridian. Road names and locations are as shown on the following United States Geological Survey Quadrangles, 7.5 minute series: Rio Vista, 1953; Clayton, 1953; Vernalis, 1952; Ripon, 1952; Bruceville, 1953; Florin, 1953; and Stockton West, 1952.

Section 4

Status of Species and Critical Habitat in the Action Area

4. Status of Species and Critical Habitat in the Action Area



This section of the biological assessment summarizes relevant information on the biological requirements of the species, population trends, abundance, viability, distribution, and condition of critical habitat. This section draws heavily on USFWS and NMFS documents, as well as the WHCP PEIR, and other relevant resources.

This section addresses <u>four species and one critical habitat</u> under the jurisdiction of USFWS, and <u>four species and four critical habitats</u> under the jurisdiction of NMFS. This section is organized as follows:

A. USFWS Listed Species (5) and Critical Habitats (1)

1. *Threatened delta smelt*
   - *Threatened delta smelt Critical Habitat*
2. *Threatened giant garter snake*
3. *Threatened valley elderberry longhorn beetle*
4. *Candidate Threatened San Francisco Bay-Delta Distinct Population Segment (DPS) of longfin smelt*

B. NMFS Listed Species (4) and Critical Habitats (4)

1. *Endangered Sacramento River winter-run Chinook salmon*
   - *Endangered Sacramento River winter-run Chinook salmon Critical Habitat*
2. *Threatened Central Valley spring-run Chinook salmon*
   - *Threatened Central Valley spring-run Chinook salmon and Threatened Central Valley steelhead Critical Habitats*
3. *Threatened Central Valley steelhead*
4. *Threatened Southern Distinct Population Segment (DPS) of North American green sturgeon*
   - *Threatened Southern Distinct Population Segment (DPS) of North American green sturgeon Critical Habitat*

Source: www.fws.gov.

Photo: Delta smelt.

A. USFWS Listed Species and Critical Habitats

1. Threatened delta smelt *(Hypomesus transpacificus)*

The delta smelt *(Hypomesus transpacificus)* is State listed as endangered, and federally listed as threatened, with a recent decision to reclassify the federal listing from threatened to endangered. Delta smelt was first listed as threatened in 1993, with critical habitat designated in 1994.

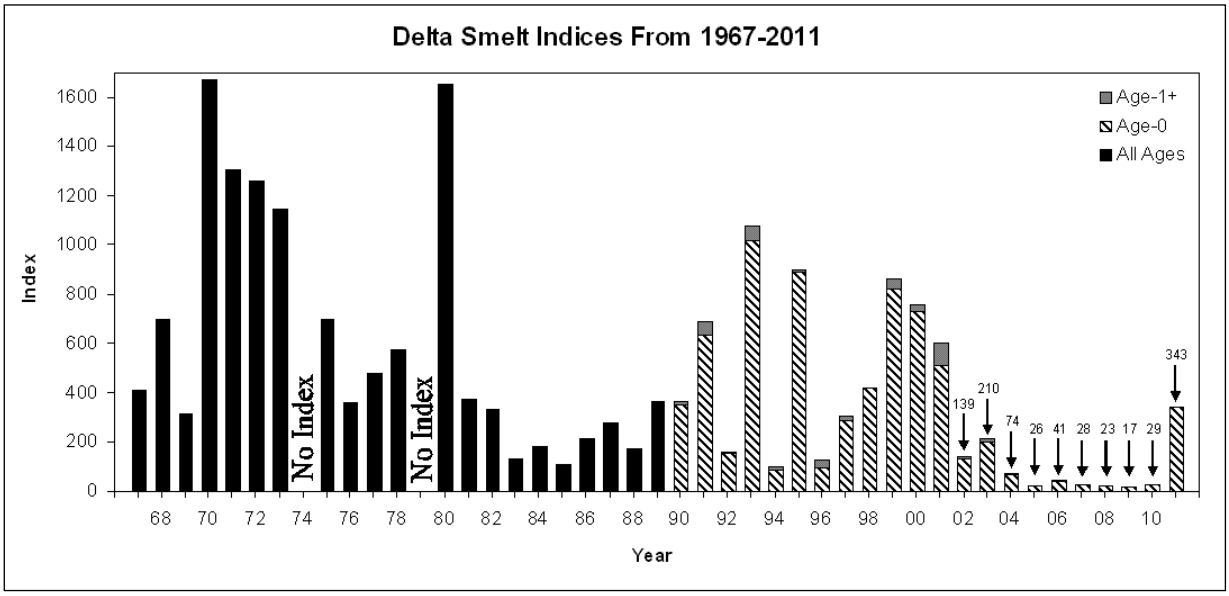
Delta smelt was one of eight fish species addressed in a 1996 *Recovery Plan for the Sacramento-San Joaquin Delta Native Fishes*. The federal threatened status was maintained following the 5-Year Review (USFWS March 2004). USFWS developed a Spotlight Species 5-Year Action Plan for delta smelt in 2009 (USFWS September 13, 2010). On April 7, 2010, USFWS issued a 12-month finding to reclassify delta smelt from threatened to endangered. USFWS completed a 5-Year Review of delta smelt on September 13, 2010, that confirmed the reclassification from threatened to endangered. While the reclassification is documented, it is precluded by other higher priority listing actions. The future reclassification will not impact WHCP activities that already seek to avoid impacts to delta smelt.

Delta smelt are threatened by factors such as: alterations to salinity and turbidity, entrainment, modified river flows, invasive species, and water export facility operations. Delta smelt abundance indices declined significantly in the early 2000s, as part of the broader Pelagic Organism Decline (POD). In 2011, following a high water year, delta smelt indices increased to their highest level since 2001. However, the indicators are still far below historical levels. **Figure 4-1,** on the next page, provides a summary of Fall Midwater Trawl (FMWT) surveys for delta smelt, from 1967 to 2011.

Critical habitat for delta smelt includes Suisun Bay (including contiguous Grizzly and Honker Bays); the length of Goodyear, Suisun, Cutoff, First Mallard, and Montezuma sloughs; and existing continuous waters within the Sacramento-San Joaquin Delta. Delta smelt are native to the Sacramento-San Joaquin estuary. They are found primarily in the lower Sacramento and San Joaquin Rivers, in the Delta above their confluence, in Suisun Marsh water channels, and in Suisun Bay. Delta smelt are endemic to low-salinity and freshwater habitats of the Delta (Bennett 2005).

Delta smelt spawning occurs within the Delta, overlapping with some WHCP treatment site locations, primarily during April through mid-May. Some juvenile rearing occurs within the Delta into early July (USFWS 2008); however, juveniles and adult delta smelt spend time in the low salinity zone (LSZ), at salinity levels that are not conducive to water hyacinth growth.

Figure 4-1
Delta smelt Fall Midwater Trawl Indices (1967-2011)



Source: California Department of Fish and Game (http://www.dfg.ca.gov/delta/data/fmwt/Indices/sld002.asp)

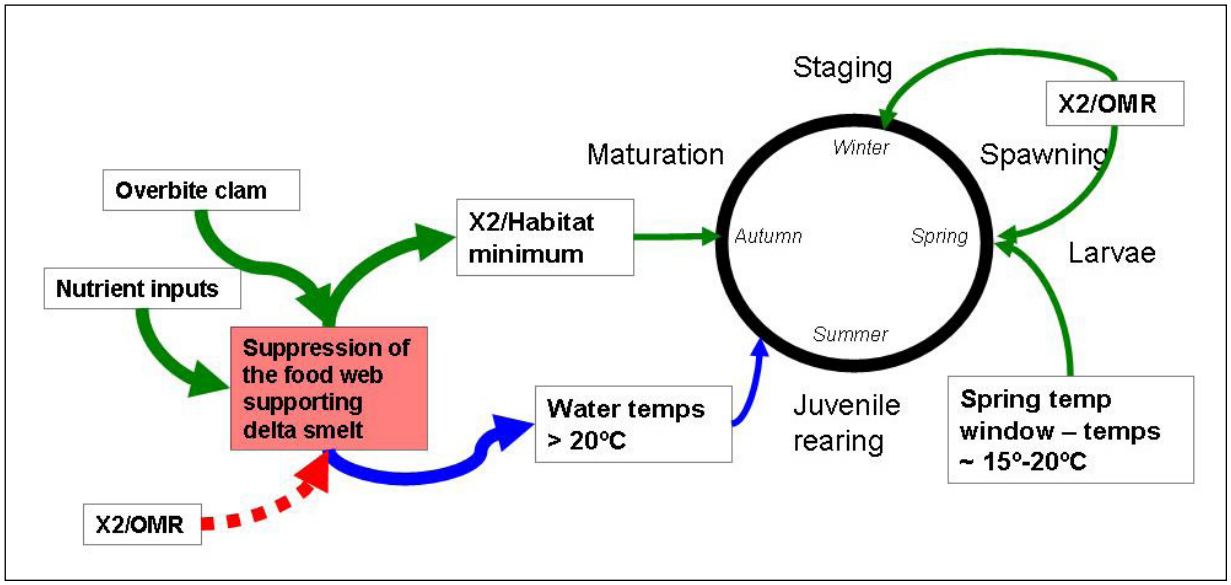
The description below of delta smelt biology, abundance, and habitat requirements is drawn from the USFWS Operations Criteria and Plan (OCAP) Biological Opinion of December 15, 2008.

Delta smelt are a member of the Osmeridae family (northern smelts) (Moyle 2002) and is one of six species currently recognized in the *Hypomesus* genus (Bennett 2005). Delta smelt are endemic to the San Francisco Bay/ Sacramento-San Joaquin Delta Estuary (Bay-Delta) in California, and are restricted to the area from San Pablo Bay, upstream through the Delta in Contra Costa, Sacramento, San Joaquin, Solano, and Yolo counties (Moyle 2002). Their range extends from San Pablo Bay upstream to Verona on the Sacramento River; and Mossdale on the San Joaquin River. Delta smelt were formerly considered to be one of the most common pelagic fish in the upper Sacramento-San Joaquin estuary.

Delta smelt are a slender-bodied fish, generally about 60 to 70 millimeters (mm) long, although they can reach lengths of up to 120 mm (Moyle 2002). Delta smelt are nearly translucent and have a steely blue sheen to their sides. Delta smelt usually aggregate but do not appear to be a strongly schooling species.

Genetic analyses have confirmed that *H. transpacificus* presently exists as a single intermixing population (Stanley et al. 1995; Trenham et al. 1998). The most closely related species is the Surf smelt (*H. pretiosis*), a marine species common along the western coast of North America. Despite its morphological similarity, delta smelt are less-closely related to wakasagi (*H. nipponensis*), an anadromous western Pacific species introduced into California Central Valley reservoirs in 1959 and now distributed in the historic range of delta smelt (Trenham et al. 1998). Genetic introgression among *H. transpacificus* and *H. nipponensis* is low.

Figure 4-2
Lifecycle Conceptual Model For delta smelt
(The Larger the Arrow Size, the Stronger the Influence on the Process Box)



(Source: USFWS December 15, 2008)

The delta smelt life cycle is completed within the freshwater and brackish low salinity zone (LSZ) of the Bay-Delta. **Figure 4-2,** above, portrays the conceptual model used for delta smelt. Delta smelt are moderately euryhaline (Moyle 2002). However, salinity requirements vary by life stage. Delta smelt are a pelagic species, inhabiting open waters away from the bottom and shore-associated structural features (Nobriga and Herbold, 2008). Although delta smelt spawning has never been observed in the wild, clues from the spawning behavior of related osmerids suggests delta smelt use bottom substrate and nearshore features during spawning. However, apart from spawning and egg-embryo development, the distribution and movements of all life stages are influenced by transport processes associated with water flows in the estuary, which also affect the quality and location of suitable openwater habitat (Dege and Brown 2004; Feyrer et al. 2007; Nobriga et al. 2008).

Delta smelt are weakly anadromous and undergo a spawning migration from brackish water to freshwater annually (Moyle 2002). In early winter, mature delta smelt migrate from brackish, downstream rearing areas, in and around Suisun Bay and the confluence of the Sacramento and San Joaquin rivers, upstream to freshwater spawning areas in the Delta. Delta smelt historically have also spawned in the freshwater reaches of Suisun Marsh. In winters featuring high Delta outflow, the spawning range of delta smelt shifts west to include the Napa River (Hobbs et al. 2007).

The upstream migration of delta smelt, which ends with their dispersal into river channels and sloughs in the Delta (Radtke 1966; Moyle 1976, 2002; Wang 1991), seems to be triggered or cued by abrupt

changes in flow and turbidity associated with the first flush of winter precipitation (Grimaldo et al. 2009) but can also occur after very high flood flows have receded. Grimaldo et al. (2009) noted salvage often occurred when total inflows exceeded over 25,000 cfs, or when turbidity elevated above 12 Nephelometric Turbidity Units (NTU) (at Clifton Court Forebay (CCF) station).

Delta smelt spawning may occur from mid-winter through spring; most spawning occurs when water temperatures range from about 12°C to 18°C (Moyle 2002). Most adult delta smelt die after spawning (Moyle 2002). However, some fraction of the population may hold over as two-year-old fish and spawn in the subsequent year.

During and after a variable period of larval development, the young fish migrate downstream until they reach the LSZ (indexed as X2) where they reside until the following winter (Moyle 2002). The location of the delta smelt population follows changes in the location of the LSZ, which depends primarily on Delta outflow.

Spawning

Adult delta smelt spawn during the late winter and spring months, with most spawning occurring during April through mid-May (Moyle 2002). Spawning occurs primarily in sloughs and shallow edge areas in the Delta. Delta smelt spawning has also been recorded in Suisun Marsh and the Napa River (Moyle 2002). Most spawning occurs at temperatures between 12°C to 18°C. Although spawning may occur at temperatures up to 22°C, hatching success of the larvae is very low (Bennett 2005).

Fecundity of females ranges from about 1,200 to 2,600 eggs, and is correlated with female size (Moyle 2002). Moyle et al. (1992) considered delta smelt fecundity to be "relatively low." However, based on Winemiller and Rose (1992), delta smelt fecundity is fairly high for a fish its size. In captivity, females survive after spawning and develop a second clutch of eggs (Mager et al. 2004); field collections of ovaries containing eggs of different size and stage indicate that this also occurs in the wild (Adib-samii 2008).

Captive delta smelt can spawn up to four to five times. While most adults do not survive to spawn a second season, a few (less than five percent) do (Moyle 2002; Bennett 2005). Those that do survive are typically larger (90 to 110 mm standard length (SL)) females that may contribute disproportionately to the population's egg supply (Moyle 2002 and references therein)). Two-year-old females may have three to six times as many ova as first year spawners.

Most of what is known about delta smelt spawning habitat in the wild is inferred from the location of spent females and young larvae captured in the Spring Kodiak Trawl (SKT) and 20-mm survey, respectively. In the laboratory, delta smelt spawned at night (Baskerville-Bridges et al. 2000; Mager et al. 2004). Other smelts, including marine beach spawning species and estuarine populations and the landlocked Lake Washington longfin smelt, are secretive spawners, entering spawning areas during the night and leaving before dawn. If this behavior is exhibited by delta smelt, then delta smelt distribution based on the SKT, which is conducted during daylight hours in offshore habitats, may reflect general regions of spawning activity, but not actual spawning sites.

Delta smelt spawning has only been directly observed in the laboratory and eggs have not been found in the wild. Consequently, what is known about the mechanics of delta smelt spawning is derived from laboratory observations and observations of related smelt species. Delta smelt eggs are 1 mm diameter and are adhesive and negatively buoyant (Moyle 1976, 2002; Mager et al. 2004; Wang 1986, 2007). Laboratory observations indicate that delta smelt are broadcast spawners, discharging eggs and milt close to the bottom over substrates of sand and/or pebble in current (DWR and Reclamation 1994; Brown and Kimmerer 2002; Lindberg et al. 2003; Wang 2007).

Presence of newly hatched larvae likely indicates regions where spawning has occurred. The 20-mm trawl has captured small (~5 mm Standard Length [SL]) larvae in Cache Slough, the lower Sacramento River, San Joaquin River, and at the confluence of these two rivers (e.g., 20-mm trawl survey 1 in 2005). Larger larvae and juveniles (size > 23 mm SL), which are more efficiently sampled by the 20-mm trawl gear, have been captured in Cache Slough (Sacramento River) and the Sacramento Deep Water Channel in July (e.g. 20-mm trawl survey 9 in 2008).

Because they are small fish inhabiting pelagic habitats with strong tidal and river currents, delta smelt larval distribution depends on both the spawning area from which they originate and the effect of transport processes caused by flows. Larval distribution is further affected by water salinity and temperature. Hydrodynamic simulations reveal that tidal action and other factors may cause substantial mixing of water with variable salinity and temperature among regions of the Delta (Monson et al 2007). This could result in rapid dispersion of larvae away from spawning sites.

Sampling of larval delta smelt in the Bay-Delta in 1989 and 1990 suggested that spawning occurred in the Sacramento River; in Georgiana, Prospect, Beaver, Hog, and Sycamore sloughs; in the San Joaquin River adjacent to Bradford Island and Fisherman's Cut; and possibly other areas (Wang 1991). However, in recent years, the densest concentrations of both spawners and larvae have been recorded in the Cache Slough/Sacramento Deepwater Ship Channel complex in the North Delta. Some delta smelt spawning occurs in Napa River, Suisun Bay and Suisun Marsh during wetter years (Sweetnam 1999; Wang 1991; Hobbs et al. 2007). Early stage larval delta smelt have also been recorded in Montezuma Slough near Suisun Bay (Wang 1986).

Larval Development

Mager et al. (2004) reported that embryonic development to hatching takes 11 to13 days, at 14°C to 16°C for delta smelt, and Baskerville-Bridges et al. (2000) reported hatching of delta smelt eggs after 8 to 10 days at temperatures between 15°C to 17°C. Lindberg et al. (2003) reported high hatching rates of delta smelt eggs in the laboratory at 15°C, and Wang (2007) reported high hatching rates at temperatures between 14°C to 17°C. Bennett (2005) showed hatching success peaks near 15°C. Swim bladder inflation occurs at 60 to 70 days, post-hatch, at 16°to 17°C (Mager et al. 2004). At hatching and during the succeeding three

days, larvae are buoyant, swim actively near the water surface, and do not react to bright, direct light (Mager et al. 2004).

As development continues, newly hatched delta smelt become semi-buoyant and sink in stagnant water. However, larvae are unlikely to encounter stagnant water in the wild. In the laboratory, a turbid environment (>25 NTU) was necessary to elicit a first feeding response (Baskerville-Bridges et al. 2000; Baskerville-Bridges 2004). Successful feeding seems to depend on a high density of food organisms and turbidity, and increases with stronger light conditions (Baskerville-Bridges *et al.* 2000; Mager et al. 2004; Baskerville-Bridges *et al.* 2004).

Growth rates of wild-caught delta smelt larvae are faster than laboratory-cultured individuals. Mager et al. (2004) reported growth rates of captive-raised delta smelt reared at near-optimum temperatures (16°C to 17°C). Their fish were about 12 mm long after 40 days, and about 20 mm long after 70 days. In contrast, analyses of otoliths indicated that wild delta smelt larvae were 15 to 25 mm, or nearly twice as long at 40 days of age (Bennett 2005). By 70 days, most wild fish were 30 to 40 mm long, and beyond the larval stage. This suggests there is strong selective pressure for rapid larval growth in nature, a situation that is typical for fish in general (Houde 1987).

Laboratory-cultured delta smelt larvae have generally been fed rotifers at first-feeding (Baskerville-Bridges et al. 2004; Mager et al. 2004). However, rotifers rarely occur in the guts of wild delta smelt larvae (Nobriga 2002). The most common first prey of wild delta smelt larvae is the larval stages of several copepod species. These copepod 'nauplii' are larger, and have more calories, than rotifers. This difference in diet may enable the faster growth rates observed in wild-caught larvae.

The food available to larval fishes is constrained by mouth gape and status of fin development. Larval delta smelt cannot capture as many kinds of prey as larger individuals, but all life stages have small gapes that limit their range of potential prey. Prey availability is also constrained by habitat use, which affects what types of prey are encountered. Larval delta smelt are visual feeders. They find and select individual prey organisms and their ability to see prey in the water is enhanced by turbidity (Baskerville- Bridges et al. 2004). Thus, delta smelt diets are largely comprised of small crustacea that inhabit the estuary's turbid, low-salinity, open-water habitats (i.e., zooplankton). Larval delta smelt have particularly restricted diets (Nobriga 2002). They do not feed on the full array of zooplankton with which they co-occur; they mainly consume three copepods, *Eurytemora affinis*, *Pseudodiaptomus forbesi*, and freshwater species of the family Cyclopidae. Further, the diets of first-feeding delta smelt larvae are largely restricted to the larval stages of these copepods; older, larger life stages of the copepods are increasingly targeted as the delta smelt larvae grow, their gape increases, and they become stronger swimmers.

The triggers for and duration of delta smelt larval movement from spawning areas to rearing areas are not known. Hay (2007) noted that eulachon larvae are probably flushed into estuaries from upstream spawning areas within the first day after hatching, but downstream movement of

delta smelt larvae occurs much later. Most larvae gradually move downstream toward the two parts per thousand (ppt) isohaline (X2). X2 is scaled as the distance in kilometers from the Golden Gate Bridge (Jassby et al. 1995). It is a physical attribute of the Bay-Delta that is used as a habitat indicator and as a regulatory standard.

At all life stages, delta smelt are found in greatest abundance in the water column and usually not in close association with the shoreline. They inhabit open, surface waters of the Delta and Suisun Bay, where they presumably aggregate in loose schools where conditions are favorable (Moyle 2002). In years of moderate to high Delta outflow (above normal to wet water years (WYs)), delta smelt larvae are abundant in the Napa River, Suisun Bay, and Montezuma Slough, but the degree to which these larvae are produced by locally spawning fish or the degree to which they originate upstream and are transported by tidal currents to the bay and marsh is uncertain.

Juveniles

Young-of-the-year delta smelt rear in the LSZ from late spring through fall and early winter. Once in the rearing area growth is rapid, and juvenile fish are 40 to 50 mm SL long by early August (Erkkila *et al.* 1950; Ganssle 1966; Radtke 1966). They reach adult size (55 to 70 mm SL) by early fall (Moyle 2002). Delta smelt growth during the fall months slows considerably (only 3 to 9 mm total), presumably because most of the energy ingested is being directed towards gonadal development (Erkkila *et al.* 1950; Radtke 1966).

Nobriga et al. (2008) found that delta smelt capture probabilities in the Townet Survey (TNS) are highest at specific conductance levels of 1,000 to 5,000 µS cm-1 (approximately 0.6 to 3.0 practical salinity unit [psu]). Water hyacinth grows at salinity levels of 2 psu or less. Similarly, Feyrer et al. (2007) found a decreasing relationship between abundance of delta smelt in the Fall Midwater Trawl (FMWT) and specific conductance during September through December. The location of the LSZ and changes in delta smelt habitat quality in the San Francisco Estuary can be indexed by changes in X2. The LSZ historically had the highest primary productivity and is where zooplankton populations (on which delta smelt feed) were historically most dense (Knutson and Orsi 1983; Orsi and Mecum 1986). However, this has not always been true since the invasion of the overbite clam (Kimmerer and Orsi 1996). The abundance of many local aquatic species has tended to increase in years when winter-spring outflow was high and X2 was pushed seaward (Jassby et al. 1995), implying that the quantity and quality (overall suitability) of estuarine habitat increases in years when outflows are high. However, delta smelt is not one of the species whose abundance has statistically covaried with winter-spring freshwater flows (Stevens and Miller 1983; Moyle et al. 1992; Kimmerer 2002; Bennett 2005). There is evidence that X2 in the fall influences delta smelt population dynamics.

Delta smelt seem to prefer water with high turbidity, based on a negative correlation between the frequency of delta smelt occurrence in survey trawls during summer, fall, and early winter; and water clarity. For

example, the likelihood of delta smelt occurrence in trawls at a given sampling station decreases with increasing Secchi depth at the stations (Feyrer et al. 2007, Nobriga et al. 2008). This is very consistent with behavioral observations of captive delta smelt (Nobriga and Herbold 2008). Few daylight trawls catch delta smelt at Secchi depths over one half meter and capture probabilities for delta smelt are highest at 0.40 m depth or less. The delta smelt's preference for turbid water may be related to increased foraging efficiency (Baskerville-Bridges et al. 2004) and reduced risk of predation.

Temperature also affects delta smelt distribution. Swanson and Cech (1995) and Swanson et al. (2000) indicate delta smelt tolerate temperatures (<8°C to >25°C), however warmer water temperatures >25°C restrict their distribution more than colder water temperatures (Nobriga and Herbold 2008). Delta smelt of all sizes are found in the main channels of the Delta and Suisun Marsh and the open waters of Suisun Bay where the waters are well oxygenated and temperatures are usually less than 25°C in summer (Nobriga et al. 2008).

Foraging Ecology

Delta smelt feed primarily on small planktonic crustaceans, and occasionally on insect larvae (Moyle 2002). Juvenile-stage delta smelt prey upon copepods, cladocerans, amphipods, and insect larvae (Moyle 2002). Historically, the main prey of delta smelt was the euryhaline copepod *Eurytemora affinis* and the euryhaline mysid *Neomysis mercedis*. The slightly larger *Pseudodiaptomus forbesi* has replaced *E. affinis* as a major prey source of delta smelt since its introduction into the Bay-Delta, especially in summer, when it replaces *E. affinis* in the plankton community (Moyle 2002). Another smaller copepod, *Limnoithona tetraspina*, which was introduced into the Bay-Delta in the mid-1990s, is now one of the most abundant copepods in the LSZ, but not abundant in delta smelt diets. *Acartiella sinensis*, a calanoid copepod species that invaded the Delta at the same time as *L. tetraspina*, also occurs at high densities in Suisun Bay and in the western Delta over the last decade. Delta smelt eat these newer copepods, but *Pseudodiaptomus* remains a dominant prey (Baxter et al. 2008).

River flows influence estuarine salinity gradients and water residence times and thereby affect both habitat suitability for benthos and the transport of pelagic plankton upon which delta smelt feed. High tributary flow leads to lower residence time of water in the Delta, which generally results in lower plankton biomass (Kimmerer 2004). In contrast, higher residence times, which result from low tributary flows, can result in higher plankton biomass but water diversions, overbite clam grazing (Jassby et al. 2002), and possibly contaminants (Baxter et al. 2008) remove a lot of plankton biomass when residence times are high. These factors all affect food availability for planktivorous fishes that utilize the zooplankton in Delta channels. Delta smelt cannot occupy much of the Delta anymore during the summer (Nobriga et al. 2008). Thus, there is the potential for mismatches between regions of high zooplankton abundance in the Delta and delta smelt distribution now that the overbite clam has decimated LSZ zooplankton densities.

The delta smelt compete with and are prey for several native and introduced fish species in the Delta. The introduced inland silverside may prey on delta smelt eggs and/or larvae and compete for copepod prey (Bennett and Moyle 1996; Bennett 2005). Young striped bass also use the LSZ for rearing and may compete for copepod prey and eat delta smelt. Centrarchid fishes and coded wire tagged Chinook salmon smolts released in the Delta for survival experiments since the early 1980s may potentially also prey on larval delta smelt (Brandes and McLain 2001; Nobriga and Chotkowski 2000). Studies during the early 1960s found delta smelt were only an occasional prey fish for striped bass, black crappie and white catfish (Turner and Kelley 1966). However, delta smelt were a comparatively rare fish even then, so it is not surprising they were a rare prey. Striped bass appear to have switched to piscivorous feeding habits at smaller sizes than they historically did, following severe declines in the abundance of mysid shrimp (Feyrer et al. 2003). Nobriga and Feyrer (2008) showed that inland silverside, which is similar in size to delta smelt, was only eaten by subadult striped bass less than 400 mm fork length. While largemouth bass are not pelagic, they have been shown to consume some pelagic fishes (Nobriga and Feyrer 2007).

Habitat

The existing physical appearance and hydrodynamics of the Delta have changed substantially from the environment in which native fish species like delta smelt evolved. The Delta once consisted of tidal marshes with networks of diffuse dendritic channels connected to floodplains of wetlands and upland areas (Moyle 2002). The in-Delta channels were further connected to drainages of larger and smaller rivers and creeks entering the Delta from the upland areas. In the absence of upstream reservoirs, freshwater inflow from smaller rivers and creeks and the Sacramento and San Joaquin Rivers were highly seasonal and more strongly and reliably affected by precipitation patterns than they are today. Consequently, variation in hydrology, salinity, turbidity, and other characteristics of the Delta aquatic ecosystem was greater in the past than it is today (Kimmerer 2002b). For instance, in the early 1900s, the location of maximum salinity intrusion into the Delta during dry periods varied from Chipps Island, in the lower Delta, to Stockton, along the San Joaquin River, and Merritt Island in the Sacramento River. Operations of upstream reservoirs have reduced spring flows while releases of water for Delta water export and increased flood control storage have increased late summer and fall inflows (Knowles 2002), though Delta outflows have been tightly constrained during late summer-fall for several decades.

Channelization, conversion of Delta islands to agriculture, and water operations have substantially changed the physical appearance, water salinity, water clarity, and hydrology of the Delta. As a consequence of these changes, most life stages of the delta smelt are now distributed across a smaller area than historically (Arthur et al. 1996, Feyrer et al. 2007). Wang (1991) noted in a 1989 and 1990 study of delta smelt larval distribution that, in general, the San Joaquin River was used more intensively for spawning than the Sacramento River. Though not restricting spawning *per se*, based on particle tracking modeling, export of water by the

Central Valley Project (CVP) and State Water Project (SWP) would usually restrict reproductive success of spawners in the San Joaquin River by entraining most larvae during downstream transport from spawning sites to rearing areas (Kimmerer and Nobriga 2008). There is one, non-wet year exception to this generalization: in 2008, delta smelt entrainment was managed under a unique system of restrictions imposed by the Court in Natural Resources Defense Council (NRDC) v Kempthorne. In 2008, CVP/SWP operations were constrained in accordance with recommendations formulated by the Service expressly to limit entrainment of delta smelt from the Central Delta.

Persistent confinement of the spawning population of delta smelt to the Sacramento River increases the likelihood that a substantial portion of the spawners will be affected by a catastrophic event or localized chronic threat. For instance, large volumes of highly concentrated ammonia released into the Sacramento River from the Sacramento Regional County Sanitation District may affect embryo survival or inhibit prey production. Further, agricultural fields in the Yolo Bypass and surrounding areas are regularly sprayed by pesticides, and water samples taken from Cache Slough sometimes exhibited toxicity to Hyalella azteca (Werner et al. 2008). The thresholds of toxicity for delta smelt for most of the known contaminants have not been determined, but the exposure to a combination of different compounds increases the likelihood of adverse effects. The extent to which delta smelt larvae are exposed to contaminants varies with flow entering the Delta. Flow pulses during spawning increase exposure to many pesticides (Kuivila and Moon 2004) but decrease ammonia concentrations entering the Delta from wastewater treatment plants.

The distribution of juvenile delta smelt has also changed over the last several decades. During the years 1970 through 1978, delta smelt catches in the TNS survey declined rapidly to zero in the Central and South Delta and have remained near zero since. A similar shift in FMWT catches occurred after 1981 (Arthur et al. 1996). This portion of the Delta has also had a long-term trend increase in water clarity during July through December (Arthur et al. 1996; Feyrer et al. 2007; Nobriga et al. 2008).

The position of the LSZ where delta smelt rear has also changed over the years. Summer and fall environmental quality has decreased overall in the Delta because outflows are lower and water transparency is higher. These changes may be due to increased upstream water diversions for flooding rice fields (Kawakami et. al. 2008). The confluence of the Sacramento and San Joaquin rivers has, as a result, become increasingly important as a rearing location for delta smelt, with physical environmental conditions constricting the species range to a relatively narrow area (Feyrer et al. 2007; Nobriga et al. 2008). This condition has increased the likelihood that most of the juvenile population is exposed to chronic and cyclic environmental stressors, or catastrophic events. For instance, all seven delta smelt collected during the September 2007 FMWT survey were captured at statistically significantly higher salinities than what would be expected based upon historical distribution data generated by Feyrer et al. (2007). During the same year, the annual bloom of toxic cyanobacteria (*Microcystis*

aeruginosa) spread far downstream to the west Delta and beyond during the summer (Peggy Lehman, pers comm). This has been suggested as an explanation for the anomaly in the distribution of delta smelt relative to water salinity levels (Reclamation 2008).

Delta Smelt Population Dynamics and Abundance Trends

The FMWT provides the best available long-term index of the relative abundance of delta smelt (Moyle et al. 1992; Sweetnam 1999). The indices derived from these surveys closely mirror trends in catch per unit effort (Kimmerer and Nobriga 2005), but do not at present support statistically reliable population abundance estimates, though substantial progress has recently been made (Newman 2008). FMWT derived data are generally accepted as providing a reasonable basis for detecting and roughly scaling interannual trends in delta smelt abundance.

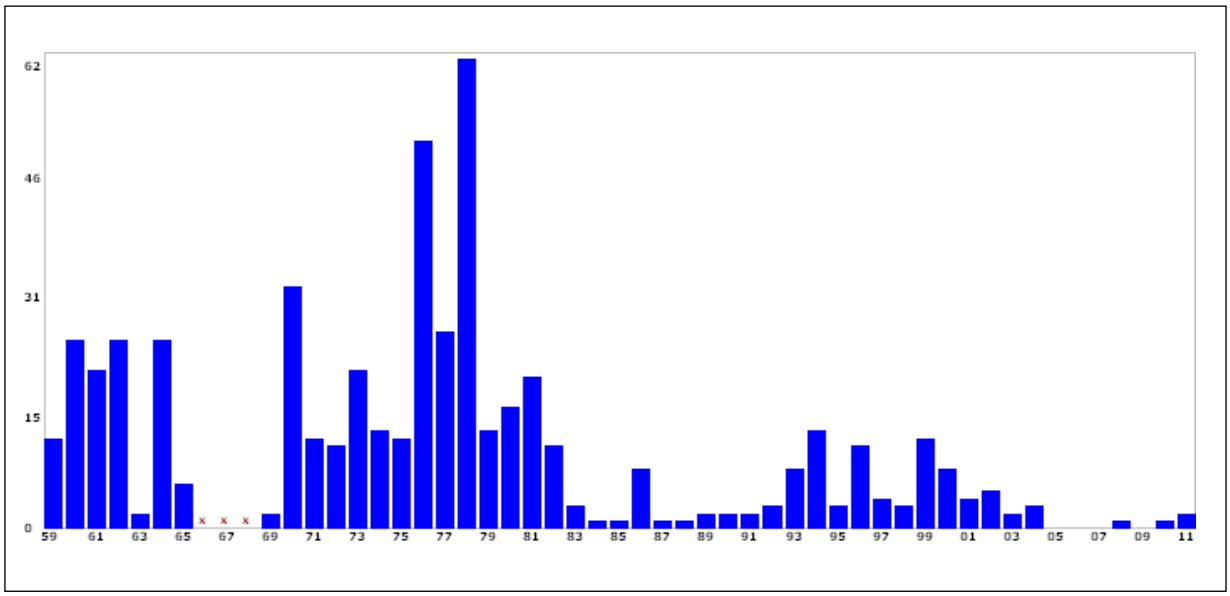
The FMWT derived indices have ranged from a low of 17 in 2009, to 1,653 in 1970 (Figure 4-1). For comparison, TNS indices, shown in **Figure 4-3,** on the next page, have ranged from a low of 0.3 in 2005 and 2009, to a high of 62.5 in 1978. From 1969 to 1981, the mean delta smelt TNS and FMWT indices were 22.5 and 894, respectively. Both the FMWT and TNS indices suggest the delta smelt population declined abruptly in the early 1980s (Moyle et al. 1992). From 1982 to 1992, the mean delta smelt TNS and FMWT indices dropped to 3.2 and 272 respectively. The population rebounded somewhat in the mid-1990s (Sweetnam 1999); the mean TNS and FMWT indices were 7.1 and 529, respectively, during the 1993 to 2002 period. However, delta smelt numbers have trended precipitously downward since about 2000, with a slight increase in 2011.

Even with the 2011 increase, the delta smelt population indices are two orders of magnitude smaller than historical highs and recent population abundance estimates are up to three orders of magnitude below historical highs (Newman 2008). After 1999 both the FMWT and the TNS population indices showed declines, and from 2000 through 2007 the median FMWT index was 106.5. The lowest FMWT abundance indices ever obtained were recorded during 2004 to 2007 (74, 27, 41, and 28, respectively).

The median TNS index during the period from 2000 through 2008, fell similarly to 1.6, and has also dropped to its lowest levels during the last four years with indexes of 0.3, 0.4, 0.4, and 0.6 during 2005 through 2008, respectively. It is highly unlikely that the indices from 2004 to 2007 can be considered statistically different from one another (see Sommer et al. 2007), but they are very likely lower than at any time prior in the period of record.

Since about 2002, delta smelt is one of four pelagic fish species subject to what has been termed the Pelagic Organism Decline or POD (Sommer et al. 2007). The POD denotes the sudden, overlapping declines of San Francisco Estuary pelagic fishes first recognized in data collected from 2002 to 2004. The POD species include delta smelt, longfin smelt, Threadfin shad (*Dorosoma petenense*), and (age-0) Striped bass (*Morone saxatillis*), which together account for the bulk of resident pelagic fish biomass in the tidal water upstream of X2.

Figure 4-3
Townet Survey delta smelt Indices (1959-2011)



Source: California Department of Fish and Game (http://www.dfg.ca.gov/delta/data/townet/indices.asp?species=3)

The year 2002 is often recognized as the start of the POD because of the striking declines of three of the four POD species between 2001 and 2002; however, statistical review of the data (e.g., Manly and Chotkowski 2006) has revealed that for at least delta smelt, the POD downtrend really began earlier (around 1999). Post-2001 abundance indices for the POD species have included record lows for all but Threadfin shad. The causes of the POD and earlier declines are not fully understood, but appear to be layered and multifactorial (Baxter et al. 2008). Several analyses have concluded that the shift in pelagic fish species abundance in the early 1980s was caused by a decrease in habitat carrying capacity or production potential (Moyle et al. 1992, Bennett 2005; Feyrer et al. 2007).

There is some evidence that the recruitment of delta smelt may have sometimes responded to springtime flow variation (Herbold et al. 1992; Kimmerer 2002). However, the weight of evidence suggests that delta smelt abundance does not (statistically) respond to springtime flow like the abundance of the species mentioned above (Stevens and Miller 1983; Jassby et al. 1995; Bennett 2005). The number of days of suitable spawning temperature during spring is correlated with subsequent abundance indices in the autumn (Bennett 2005). This is evidence that cool springs, which allow for multiple larval cohorts, can contribute to population resilience. However, these relationships do not explain a large proportion of variance in autumn abundance. Depending on which abundance index is used, the r^2 are 0.24 to 0.29.

The relationship between numbers of spawning fish and the numbers of young subsequently recruiting to the adult population is known as a stock-recruit relationship. Analysis of stock-recruit relationships using delta smelt survey data indicate that a weak density dependent effect has occurred during late summer/fall (Bennett 2005, Reclamation 2008), suggesting that delta smelt year-class strength has often been set during late summer and fall. This finding is supported by studies suggesting that the delta smelt is food limited (Bennett 2005; IEP 2005) and evidence for density dependent mortality has been presented by Brown and Kimmerer (2001). However, the number of days during the spring that water temperature remained between 15°C and 20°C, with a density-dependence term to correct for the saturating TNS-FMWT relationship, predicts FMWT indices fairly well ($r^2 \approx 0.70$; $p < 0.05$; Bennett, unpublished presentation at the 2003 CALFED Science Conference). This result shows that the quantity of young delta smelt produced also contributes to future spawner abundance.

Bennett (2005) analyzed the relationship between delta smelt spawner population and spawner recruits using data before, and after, the 1980s decline. He concluded that density dependence pre-1982 may have occurred at FMWT values of 600 to 800 and at FMWT values of 400 to 500 for the period 1982 through 2002. Bennett (2005) also conducted extensive stock-recruit analyses using the TNS and FMWT indices. He provided statistical evidence that survival from summer to fall is nonlinear (= density-dependent). He also noted that carrying capacity had declined. Bennett (2005) surmised that density-dependence and lower carrying capacity during the summer and fall could happen in a small population if habitat space was smaller than it was historically. This hypothesis was recently demonstrated to be true (Feyrer et al. 2007). Reduced Delta outflow during autumn has led to higher salinity in Suisun Bay and the Western Delta while the proliferation of submerged vegetation has reduced turbidity in the South Delta. Together, these mechanisms have led to a long-term decline in habitat suitability for delta smelt. High summer water temperatures also limit delta smelt distribution (Nobriga et al. 2008) and impair health (Bennett et al. 2008).

A minimum amount of suitable habitat during summer-autumn may interact with a suppressed pelagic food web to create a bottleneck for delta smelt (Bennett 2005; Feyrer et al. 2007; Bennett et al. 2008). Prior to the overbite clam invasion, the relative abundance of maturing adults collected during autumn was unrelated to the relative abundance of juveniles recruiting the following summer (i.e., the stock-recruit relationship was density-vague). Since the overbite clam became established, autumn relative abundance explains 40 percent of the variability in subsequent juvenile abundance (Feyrer et al. 2007). When autumn salinity is factored in, 60 percent of the variance in subsequent juvenile abundance is accounted for statistically.

Since 2000, the stock-recruit relationship for delta smelt has been stronger still ($r^2 = 0.88$ without autumn habitat metrics factored in; Baxter et al. 2008). This has led to speculation about Allee effects. Allee effects occur when reproductive output per fish declines at low

population levels (Allee 1931, Berec et al. 2006). Below a certain threshold the individuals in a population can no longer reproduce rapidly enough to replace themselves and the population spirals to extinction. For delta smelt, possible mechanisms for Allee effects include mechanisms directly related to reproduction and genetic fitness such as difficulty finding enough males to maximize egg fertilization during spawning (e.g., Purchase et al. 2007). Genetic problems arising from small population sizes like inbreeding and genetic drift also can contribute to Allee effects, but genetic bottlenecks occur after demographic problems like the example of finding enough mates (Lande 1988). Other mechanisms related to survival such as increased vulnerability to predation are also possible based on studies of other species.

These data provide evidence that factors affecting juvenile delta smelt during summer and autumn are also impairing delta smelt reproductive success. Thus, the interaction of warm summer water temperatures, suppression of the food web supporting delta smelt, and spatially restricted suitable habitat during autumn affect delta smelt health and ultimately survival and realized fecundity.

Another possible contributing driver of reduced delta smelt survival, health, fecundity, and resilience that occurs during winter is the "Big Mama Hypothesis" (Bill Bennett, UC Davis, pers. comm. and various oral presentations). As a result of his synthesis of a variety of studies, Bennett proposed that the largest delta smelt (whether the fastest growing age-1 fish or fish that manage to spawn at age-2) could have a large influence on population trends. Delta smelt larvae spawned in the South Delta have high risk of entrainment under most hydrologic conditions (Kimmerer 2008), but water temperatures often warm earlier in the South Delta than the Sacramento River (Nobriga and Herbold 2008). Thus, delta smelt spawning often starts and ends earlier in the Central and South Delta than elsewhere. This differential warming may contribute to the "Big Mama Hypothesis" by causing the earliest ripening females to spawn disproportionately in the South Delta, putting their offspring at high risk of entrainment. Although water diversion strategies have been changed to better protect the 'average' larva, the resilience historically provided by variable spawn timing may be reduced by water diversions and other factors that covary with Delta inflows and outflows.

Substantial increases in winter salvage at Banks and Jones that occurred contemporaneously with recent declines in delta smelt and other POD species (Kimmerer 2008, Grimaldo et al. 2009) support the interpretation that entrainment played a role in the POD-era depression of delta smelt numbers. Increased winter entrainment of delta smelt represents a loss of pre-spawning adults and all their potential progeny (Sommer et al. 2007). Note that winter salvage levels subsequently decreased to very low levels for all POD species during the winters of 2005 to 2006 and 2006 to 2007, possibly due to the very low population sizes during those periods. Reduced pumping for protection of delta smelt also substantially reduced Old and Middle River (OMR) flow towards the pumps and subsequently reduced number of delta smelt entrained during the winters of 2006 to 2007 and 2007 to 2008.

The hydrologic and statistical analyses of relationships between OMR flows and salvage suggest a reasonable mechanism by which winter entrainment increased with increased exports during the POD years; however, entrainment is not a substantial source of mortality every year. Manly and Chotkowski (2006; IEP 2005) found that monthly or semi-monthly measures of exports or Old and Middle rivers flow had a reliable, statistically significant effect on delta smelt abundance; however, individually they explained a small portion (no more than a few percent) of the variability in the fall abundance index of delta smelt across the entire survey area and time period. Kimmerer (2008) addressed delta smelt entrainment by means of particle tracking, and estimated historical entrainment rates for larvae and juvenile delta smelt to be as high as 40 percent; however, he concluded that non-entrainment mortality in the summer had effects on FMWT delta smelt numbers. Hence, there are other factors that often mask the effect of entrainment loss on delta smelt fall abundance in these analyses. Among them, availability and quality of summer and fall habitat are clearly affected by CVP/SWP operations.

We conclude that entrainment and habitat availability/quality jointly contribute to downward pressure on spawner recruitment and one or both of these general mechanisms is operating throughout the year. The intensity of constraints of the other threats affecting the delta smelt carrying capacity varies between years, and the importance of contributing stressors changes as outflow, export operations, weather, and the abundances of other ecosystem elements vary. For instance, Bennett (2005) noted that seasonally low outflow and warmer water temperatures may concentrate delta smelt and other planktivorous fishes into relatively small patches of habitat during late summer. This would increase competition and limit food availability during low outflow. Higher outflow that expands and moves delta smelt habitat downstream of the Delta is expected to improve conditions for delta smelt (Feyrer et al. 2007). The high proportion of the delta smelt population that has been entrained during some years (Kimmerer 2008) would be expected to reduce the ability of delta smelt to respond to the improved conditions, thereby limiting the potential for increased spawner recruitment. Further, the smaller sizes of maturing adults during fall may have affected delta smelt fecundity (Bennett, 2005). This would further reduce the species' ability to respond to years with improved conditions.

Seasonal Life History of delta smelt

The following discussion, also from USFWS (December 15, 2008), describes the life stage and location of delta smelt by season. This is relevant to WHCP activities, which occur within the Delta during limited time periods. **Table 4-1,** on the next page, summarizes the life cycle of delta smelt. Spawning and some juvenile rearing occurs within potential WHCP treatment locations.

Winter (December to February)

Adult delta smelt are generally distributed in low salinity habitats of the greater Suisun Bay region and the Sacramento and San Joaquin River confluence during fall. Variation in outflow appears to initiate their

migration from Suisun Bay upstream to freshwater habitats for spawning. This is because initial catches upstream normally occur in close association with increased turbidity associated with the first strong flow pulse of the winter (Grimaldo et al. 2009). As a result, entrainment of adult delta smelt at Banks and Jones is also closely associated with factors controlled by outflow or X2 (Grimaldo et al. 2009). Specifically, salvage of adult delta smelt is significantly negatively associated with flows in OMR, and when the flows are highly negative the starting location of the fish indexed by X2 the month prior to entrainment also has an effect (Grimaldo et al. 2009).

Outflow during winter also affects the entrainment of early-spawned larvae when their distribution is within the hydrodynamic zone affected by pumping operations (Kimmerer 2008). Winter outflow also affects the distribution of spawning fish in major regions. For example, the Napa River is used for spawning only in years when outflow is sufficient to connect the Napa River with low salinity habitat in the estuary (Hobbs et al. 2007).

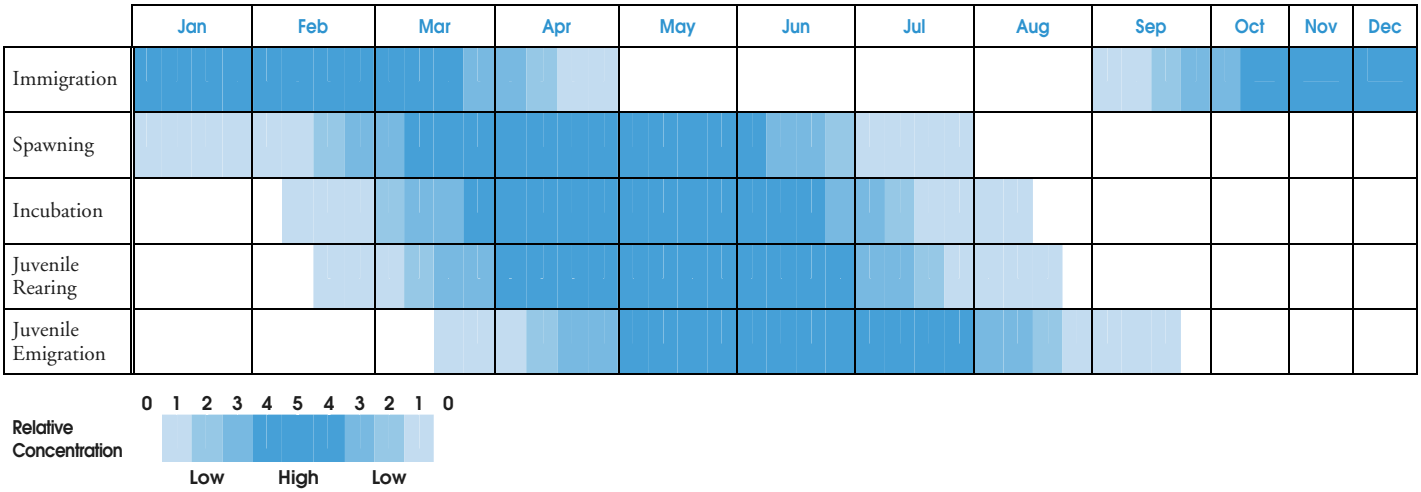
Spring (March to May)

During spring, young of the year (YOY) delta smelt generally move from upstream spawning locations downstream into low salinity rearing habitats. There is some evidence that recruitment variability of delta smelt may have sometimes responded to springtime flow variation (Herbold et al. 1992; Kimmerer 2002). For example, the number of days X2 is in Suisun Bay during spring is weakly positively correlated with abundance as measured by the FMWT index. However, the weight of evidence suggests that delta smelt abundance does not statistically respond to springtime flow in a similar manner to other species for which the spring X2 requirements were developed (Stevens and Miller 1983; Jassby et al. 1995; Bennett 2005).

However, studies have demonstrated that outflow has a strong effect on the distribution of YOY delta smelt (Dege and Brown 2004) and that it therefore also ultimately influences entrainment at Jones and Banks pumping facilities (Kimmerer 2008). Dege and Brown (2004) found that X2 had a strong influence on the geographic distribution of delta smelt, but distribution with respect to X2 was not affected, indicating that distribution is closely associated with habitat conditions proximal to X2. YOY delta smelt are consistently located just upstream of X2 in freshwater until they become juveniles and enter the low salinity habitats of Suisun Bay later in the year.

Outflow affects the entrainment of YOY delta smelt at the Jones and Banks facilities in several ways. First, because outflow affects adult spawning migration and juvenile distribution, it affects their position relative to the hydrodynamic influence of the diversions (Kimmerer 2008). Second, Old and Middle River (OMR) is the best predictor of salvage and entrainment for adult delta smelt and it is also relevant to larval and juvenile entrainment when considered in the context of X2. In general, the more water that is exported relative to that which is dedicated to outflow enhances negative flows in OMR flow towards the diversions, which in turn increases salvage (Baxter et al. 2008; Kimmerer 2008; Grimaldo et al. 2009).

Table 4-1
Life-cycle of the delta smelt *(Hypomesus transpacificus)*



Source: Hoogeweg et al. 2011 p.147

Summer (June to August)

Summer represents a primary growing season for delta smelt while they are distributed in low salinity habitats of the estuary. X2 affects delta smelt distribution during summer (Sweetnam 1999). Food supply and habitat suitability are currently believed to be important factors for delta smelt during summer (Bennett 2005; Baxter et al. 2008; Nobriga and Herbold 2008). The CVP/SWP affect summer habitat suitability and might affect summer prey co-occurrence through their effect on Delta hydrodynamics.

Fall (September to November)

During fall, delta smelt are typically fully distributed in low salinity rearing habitats located around the confluence of the Sacramento and San Joaquin Rivers. Suitable abiotic habitat for delta smelt during fall has been defined as relatively turbid water (Secchi depths < 1.0 m) with a salinity of approximately 0.6 to 3.0 psu (Feyrer et al. 2007). The amount of suitable abiotic habitat available for delta smelt, measured as hectares of surface area, is negatively related to X2. The average X2 during fall has exhibited a long-term increasing trend (movement further upstream), which has resulted in a corresponding reduction in the amount and location of suitable abiotic habitat (Feyrer et al. 2007, 2008).

The available data provide evidence to suggest that the amount of suitable abiotic habitat available for delta smelt during fall affects the population in a measurable way. There is a statistically significant stock-recruit relationship for delta smelt in which pre-adult abundance measured by the FMWT positively affects the abundance of juveniles the following year in the TNS (Bennett 2005; Feyrer et al. 2007). Incorporating suitable abiotic habitat into the stock-recruit model as a covariate improves the model by increasing the amount of variability explained by 43

percent, r-squared values improved from 46 percent to 66 percent (Feyrer et al. 2007).

It is likely that changes in X2 and the corresponding amount of suitable abiotic habitat are important to the long-term decline of delta smelt but may have been of lesser importance in the more recent POD. Over the long-term, the amount of suitable abiotic habitat for delta smelt during fall has decreased anywhere from 28 percent to 78 percent, depending on the specific habitat definitions that are considered (Feyrer et al. 2008). The majority of this habitat loss has occurred along the periphery, limiting the distribution of delta smelt mainly to a core region in the vicinity of the confluence of the Sacramento and San Joaquin Rivers (Feyrer et al. 2007). Concurrently, delta smelt abundance as measured by the FMWT decreased by 63 percent. This correspondence and the significant stock-recruit relationship with the habitat covariate strongly suggest that delta smelt have been negatively affected by long-term changes in X2 and habitat. However, at the onset of the POD, delta smelt abundance and suitable abiotic habitat had already declined to a point where it was unlikely that Feyrer's two variable definition of habitat was the primary limiting factor constraining the population.

Nevertheless, X2 and inflow-corrected X2 during fall in the years following the POD (2000 to 2005) was several km upstream compared to that for the pre-pod years (1995 to 1999). This suggests that operations in the Delta have exported more water relative to inflow, which has had a negative effect on X2 by moving it upstream. This is confirmed by a long-term positive trend in the export to inflow (E:I) ratio for all months from June through December. In fact, long-term trends in X2, inflow-corrected X2, and the E:I ratio indicate this pattern has been in effect for many years and likely one of the factors responsible for the long-term decline in habitat suitability for delta smelt.

Threatened delta smelt Critical Habitat

USFWS designated critical habitat for the delta smelt on December 19, 1994 (59 FR 65256). The geographic area encompassed by the designation includes all water and all submerged lands below ordinary high water and the entire water column bounded by and contained in Suisun Bay (including the contiguous Grizzly and Honker Bays); the length of Goodyear, Suisun, Cutoff, First Mallard (Spring Branch), and Montezuma sloughs; and the existing contiguous waters contained within the legal Delta (as defined in section 12220 of the California Water Code) (USFWS 1994).

Description of the Primary Constituent Elements (PCEs)

In designating critical habitat for the delta smelt, USFWS identified the following primary constituent elements essential to the conservation of the species:

1. "Physical habitat" is defined as the structural components of habitat. Because delta smelt is a pelagic fish, spawning substrate is the only known important structural component of habitat. It is possible that depth variation is an important structural characteristic of pelagic habitat that helps fish maintain position within the estuary's LSZ (Bennett et al. 2002).

2. "Water" is defined as water of suitable quality to support various delta smelt life stages with the abiotic elements that allow for survival and reproduction. Delta smelt inhabit open waters of the Delta and Suisun Bay. Certain conditions of temperature, turbidity, and food availability characterize suitable pelagic habitat for delta smelt. Factors such as high entrainment risk and contaminant exposure can degrade this PCE even when the basic water quality is consistent with suitable habitat.
3. "River flow" is defined as transport flow to facilitate spawning migrations and transport of offspring to LSZ rearing habitats. River flow includes both inflow to and outflow from the Delta, both of which influence the movement of migrating adult, larval, and juvenile delta smelt. Inflow, outflow, and OMR influence the vulnerability of delta smelt larvae, juveniles, and adults to entrainment at Banks and Jones pumping facilities. River flow interacts with the fourth primary constituent element, salinity, by influencing the extent and location of the highly productive LSZ where delta smelt rear.
4. "Salinity" is defined as the LSZ nursery habitat. The LSZ is where freshwater transitions into brackish water; the LSZ is defined as 0.5 to 6.0 psu (parts per thousand salinity; Kimmerer 2004). The 2 psu isohaline is a specific point within the LSZ where the average daily salinity at the bottom of the water is 2 psu (Jassby et al. 1995). By local convention the location of the LSZ is described in terms of the distance from the 2 psu isohaline to the Golden Gate Bridge (X2); X2 is an indicator of habitat suitability for many San Francisco Estuary organisms and is associated with variance in abundance of diverse components of the ecosystem (Jassby et al. 1995; Kimmerer 2002). The LSZ expands and moves downstream when river flows into the estuary are high. Similarly, it contracts and moves upstream when river flows are low.

During the past 40 years, monthly average X2 has varied from as far downstream as San Pablo Bay (45 km) to as far upstream as Rio Vista on the Sacramento River (95 km). At all times of year, the location of X2 influences both the area and quality of habitat available for delta smelt to successfully complete their life cycle. In general, delta smelt habitat quality and surface area are greater when X2 is located in Suisun Bay. Both habitat quality and quantity diminish the more frequently and further the LSZ moves upstream, toward the confluence.

Conservation Role of delta smelt Critical Habitat

USFWS's primary objective in designating critical habitat is to identify the key components of delta smelt habitat that support successful spawning, larval and juvenile transport, rearing, and adult migration. Delta smelt are endemic to the Bay-Delta and the vast majority only live one year. Thus, regardless of annual hydrology, the Delta must provide suitable habitat all year, every year. Different regions of the Delta provide different habitat conditions for different life stages, but those habitat conditions must be present when needed, and have sufficient connectivity to provide migratory pathways and the flow of energy, materials and organisms among the habitat components. The entire Delta and Suisun Bay

are designated as critical habitat; over the course of a year, the entire habitat is occupied.

Delta smelt live their entire lives in the tidally-influenced fresh- and brackish waters of the San Francisco Estuary (Moyle 2002). Delta smelt are an open-water, or pelagic, species. They do not associate strongly with structure. They may use nearshore habitats for spawning (PCE #1), but free-swimming life stages mainly occupy offshore waters (PCE #2). Thus, the distribution of the population is strongly influenced by river flows through the estuary (PCE #3) because the quantity of fresh water flowing through the estuary changes the amount and location of suitable low-salinity, open-water habitat (PCE #4). This is true for all life stages. During periods of high river flow into the estuary, delta smelt distribution can transiently extend as far west as the Napa River and San Pablo Bay. Delta smelt distribution is highly constricted near the Sacramento-San Joaquin river confluence during periods of low river flow into the estuary (Feyrer et al. 2007).

In the 1994 designation of critical habitat, the best available science held that the delta smelt population was responding to variation in spring X2. In the intervening years, the scientific understanding of delta smelt habitat has improved. The current understanding is that X2 and OMR both must be considered to manage entrainment and that X2 indexes important habitat characteristics throughout the year.

Source: www.californiaherps.com.

Photo: giant garter snake.

2. Threatened giant garter snake *(Thamnophis gigas)*

The giant garter snake *(Thamnophis gigas)* is listed as State and federal threatened. Giant garter snakes are the largest garter snake in North America and are endemic to the valley floor wetlands in the Sacramento and San Joaquin Valleys. They inhabit sloughs, ponds, small lakes, and other low-gradient waterways, including irrigation canals where water is present throughout the summer. Giant garter snakes are rarely found away from water, forage in the water for food, and will retreat to water to escape predators and disturbance (USFWS May 2004). These snakes typically avoid larger waterways with predatory fish, and woodland streams with excessive cover.

The description below of giant garter snake range, habitat, biology, and status is drawn from USFWS Biological Opinion for WHCP, 81410-2011-F-0035 (USFWS August 2012).

Giant garter snake (GGS) reach a total length of approximately 160 cm. Females tend to be slightly longer and proportionately heavier than males. Generally, GGS has a dark dorsal background color with pale dorsal and lateral stripes, although coloration and pattern

prominence are geographically and individually variable (Hansen 1980). The weight of adult female GGS is typically 500 to 700 grams. Dorsal background coloration varies from brownish to olive with a checkered pattern of black spots, separated by a yellow dorsal stripe and two light colored lateral stripes. Background coloration and prominence of black checkered pattern and the three yellow stripes are geographically and individually variable (Hansen 1980). The ventral surface is cream to olive or brown and sometimes infused with orange, especially in northern populations.

The historical range of the snake is thought to have extended from the vicinity of Chico in Butte County, southward to Buena Vista Lake, near Bakersfield, in Kern County (Fitch 1940, Fox 1948, Hansen and Brode 1980). Early collecting localities of the GGS coincide with the distribution of large flood basins, particularly riparian marsh or slough habitats and associated tributary streams (Hansen and Brode 1980).

The known range of GGS has changed little since the time of listing. In 2005 and 2006, GGS have been seen northward in Chico, and southward at the Mendota Wildlife Area in Fresno County. Habitat has been lost to urban development in the Natomas Basin in Sacramento and Sutter Counties. Two known population clusters south of Stockton are small, fragmented, and unstable.

Endemic to wetlands in the Sacramento and San Joaquin valleys, GGS inhabit marshes, sloughs, ponds, small lakes, low gradient streams, and other waterways and agricultural wetlands, such as irrigation and drainage canals, rice fields, and adjacent uplands (USFWS 1999). Essential habitat consists of: (1) wetlands with adequate water during the GGS's active season (early-spring through mid-fall) to provide food and cover; (2) emergent, herbaceous wetland vegetation, such as cattails and bulrushes, for escape cover and foraging habitat during the active season; (3) upland habitat with grassy banks and openings in waterside vegetation for basking; and (4) higher elevation uplands for over-wintering habitat with escape cover (vegetation, burrows) and underground refugia (crevices and small mammal burrows) (Hansen 1988). GGS are typically absent from larger rivers and other bodies of water that support introduced populations of large, predatory fish, and from wetlands with sand, gravel, or rock substrates (Hansen 1988, Hansen and Brode 1980). Riparian woodlands do not provide suitable habitat because of excessive shade, lack of basking sites, and absence of prey populations (Hansen 1988).

GGS are extremely aquatic, are rarely found away from water, forage in the water for food, and will retreat to water to escape predators and disturbance. GGS are active foragers, feeding primarily on aquatic prey such as fish and amphibians. Historically, prey likely consisted of Sacramento blackfish *(Orthodon microlepidotus)*, thick-tailed chub *(Gila crassicauda)*, and red-legged frog *(Rana draytonii)*. Because these species are no longer available (chub extinct, red-legged frog extirpated from the Central Valley, blackfish declining/in low numbers), the predominant food items are now introduced species such as carp *(Cyprinus carpio)*, mosquito-fish *(Gambusia affinis)*, bullfrogs *(Rana catesbiana)*, and Pacific chorus frog

(Pseudacris regilla) (Fitch 1941, Rossman et al. 1996).

Rice fields have become important habitat for GGS. In particular, the associated canals and their banks are important for both spring and summer active behavior and winter hibernation (Hansen 2004, Wylie 1998). While within the rice fields, GGS forage in the shallow water for prey, utilizing rice plants and vegetated berms dividing rice checks for shelter and basking sites (Hansen and Brode 1993).

The breeding season for GGS extends through March and April, and females give birth to live young from late July through early September (Hansen and Hansen 1990). Brood size is variable, ranging from 10 to 46 young, with a mean of 23 (Hansen and Hansen 1990). At birth young average about 20.6 cm snout to vent length and three to five grams. Young immediately scatter into dense cover and absorb their yolk sacs, after which they begin to feed on their own. Although growth rates are variable, young typically more than double in size by one year of age (USFWS 1999a). Sexual maturity averages three years in males and five years for females (USFWS 1999a).

The GGS typically inhabits small mammal burrows and other soil crevices throughout its winter dormancy period (November to mid-March). Although these areas are generally thought to be above prevailing flood elevations, snakes may not always utilize high ground during their winter dormancy period. The Biological Resources Division of the United States Geological Survey has documented GGS at the Colusa National Wildlife Refuge overwintering in areas with few high ground retreat sites (Wylie et al. 1997). GGS in another study population in Gilsizer Slough overwintered at a low elevation wetland area, even though higher ground was present nearby. Both of these populations survived flooding and were not displaced from the area. GGS also use burrows as refuge from extreme heat during their active period. Wylie et al. (1997) documented GGS using burrows in the summer as much as 165 feet away from the marsh edge. Overwintering GGS have been documented using burrows as far as 820 feet from the edge of marsh habitat (Wylie et al. 1997).

During radio-telemetry studies, GGS typically moved little from day to day. However, total activity varied widely between individuals. Snakes have been documented moving up to 5 miles (8 kilometers) over the period of a few days (Wylie et al. 1997). In agricultural areas, GGS were documented using rice fields in 19 to 20 percent of the observations, marsh habitat in 20 to 23 percent of observations, and canal and agricultural waterway habitats in 50 to 56 percent of the observations (Wylie et al. 1997).

At the time of the listing, GGS was known from 13 populations. Populations 4 through 13 included the San Joaquin Valley, portions of the eastern fringe of the Delta, and the southern Sacramento Valley; an area encompassing about 75 percent of the species' known geographical range (USFWS 1993). Several of these populations are within the WHCP project action area.

Habitat loss is a primary threat to this species (USFWS 1999). Prior to Bureau of Reclamation activities beginning in the mid- to late-1800s, about 60 percent of the Sacramento Valley was subject to seasonal

overflow flooding providing expansive areas of GGS habitat (Hinds 1952). Now, less than 10 percent, or approximately 319,000 acres, of the historic 4.5 million acres of Central Valley wetlands remain (US Department of Interior 1994), of which very little provides habitat suitable for the GGS. Loss of habitat due to agricultural activities and flood control have extirpated the GGS from the southern one-third of its range in former wetlands associated with the historic Buena Vista, Tulare, and Kern lakebeds (Hansen 1980, Hansen and Brode 1980).

Other threats include ongoing maintenance of aquatic habitats for flood control and agricultural purposes, which can fragment and isolate available habitat, prevent dispersal of snakes among habitat units, and adversely affect the snake's availability of food items (Hansen 1988, Brode and Hansen 1992). Other threats include application of herbicides to control aquatic vegetation (Wylie et al. 1995), rodent control activities within upland aestivation (warm weather dormancy) habitat for the GGS (Wylie et al. 1995 and Wylie et al. 1997), and livestock grazing along the edges of water sources which may degrade water quality (Hansen 1988).

Currently, GGS is only known from a small number of populations. The status of these populations and the threats to these snakes and their habitats are detailed in the final rule that listed the GGS as threatened (USFWS 1993). A number of land use practices and other human activities currently threaten the survival of the GGS throughout the remainder of its range.

Source: www.fws.gov.

Photo: valley elderberry longhorn beetle.

3. Threatened valley elderberry longhorn beetle *(Desmocerus californicus dimporphus)*

Valley elderberry longhorn beetle is classified as federally threatened. The most recent 5-year review of valley elderberry longhorn beetle, completed in September 2006, recommended delisting the beetle, primarily due to the fact that conservation actions have resulted in protection of 50,000 acres of riparian habitat and the restoration of 1,500 acres of beetle habitat. In addition, the number of occurrences increased from 10 locations in 1980, to 190 known locations in 2006 (USFWS 2009).

On September 10, 2010, USFWS received a petition from the Pacific Legal Foundation requesting that USFWS delist the valley elderberry longhorn beetle. USFWS initiated a 12-month status review on August 19, 2011, to determine if delisting is warranted (Federal Register, August 19, 2012). USFWS's Spotlight Species 5-Year Action Plan (2010 to 2014) for the valley elderberry longhorn beetle recommends post-delisting monitoring of status, patch occupancy, and local turnover, should the species be delisted (USFWS 2009).

Valley elderberry longhorn beetle is a dimorphic species strictly tied to its host plant, the elderberry *(Sambucus* ssp.) during its entire

life cycle. Adults emerge from pupation inside the wood of the elderberry in the spring as the trees begin to flower. The exit holes made by the emerging adults are distinctive small oval openings. Often these holes are the only clue that beetles occur in an area. Adults eat elderberry foliage until approximately June when they mate. Females lay eggs in crevices on the bark. Upon hatching, larvae begin to tunnel into the shrub, where they will spend one to two years eating interior wood, which is their sole food source.

Valley elderberry longhorn beetle historically occurred throughout the Sacramento and San Joaquin valleys and into the foothills of the Coast Ranges and the Sierra Nevada to 2,200-foot in elevation. Elderberry shrub is a common component of riparian forests and savannah areas (USFWS 2004). Recent surveys have found beetles in only scattered localities along the Sacramento, American, San Joaquin, Kings, Kaweah, and Tuolumne rivers and their tributaries. Valley elderberry shrubs with evidence of beetles have been spotted in WHCP treatment sites along the Sacramento and Cosumnes Rivers (CNDDB 2006).

Over the last 150 years, agricultural and urban development has destroyed 90 percent of Central Valley riparian vegetation, which included the elderberry host plant, resulting in extreme fragmentation of the beetle's habitat.

The valley elderberry longhorn beetle is threatened by habitat loss and fragmentation, invasion by Argentine ants, agricultural conversion, levee construction, removal of riparian vegetation, riprapping of shoreline, and possibly other factors such as pesticide drift, exotic plant invasion, and grazing (USFWS 2004).

Source: swr.nmfs.noaa.gov.

Photo: Longfin Smelt.

4. Candidate Threatened San Francisco Bay-Delta Distinct Population Segment (DPS) of longfin smelt *(Spirinchus thaleichthys)*

The San Francisco Bay-Delta DPS of longfin smelt was classified by USFWS as threatened on April 2, 2012. However, listing of the species is currently precluded by other higher-priority actions, thus longfin smelt is a candidate species. USFWS will list the San Francisco Bay-Delta DPS of longfin smelt as priorities allow, and will review its status annually. The 2012 finding supersedes a 2009 finding that the species did not warrant listing under the ESA because the Bay-Delta population did not meet discreteness criteria. However, in the most recent 12-month finding, USFWS evaluated new information and determined that the Bay-Delta population is distinct from other longfin smelt populations on the West Coast (Federal Register April 2, 2012). Longfin smelt is listed as a threatened species by the State of California. Threats to longfin smelt in the Bay-Delta include reduced freshwater outflow, a food web altered by the invasive overbite clam, and ammonium contamination (USFWS March 29, 2012). The following description of longfin smelt status and abundance is extracted from the April 2, 2012 12-month finding published in the Federal Register.

Longfin smelt measure 9 to 11 centimeters (cm) standard length, although third-year females may grow up to 15 cm. The sides and lining of the gut cavity appear translucent silver, the back has an olive to iridescent pinkish hue, and mature males are usually darker in color than females. Longfin smelt can be distinguished from other smelts by their long pectoral fins, weak or absent striations on their opercular (covering the gills) bones, incomplete lateral line, low numbers of scales in the lateral series (54 to 65), long maxillary bones (in adults, these bones extend past mideye, just short of the posterior margin of the eye), and lower jaw extending anterior of the upper jaw (Mcallister 1963, p. 10; Miller and Lea 1972, pp. 158–160; Moyle 2002, pp. 234–236).

The longfin smelt belongs to the true smelt family Osmeridae and is one of three species in the *Spirinchus* genus; the night smelt (*Spirinchus starksi*) also occurs in California, and the shishamo (*Spirinchus lanceolatus*) occurs in northern Japan (McAllister 1963, pp. 10, 15). Because of its distinctive physical characteristics, the Bay-Delta population of longfin smelt was once described as a species separate from more northern populations (Moyle 2002, p. 235). McAllister (1963, p. 12) merged the two species *S. thaleichthys* and *S. dilates* because the difference in morphological characters represented a gradual change along the north-south distribution rather than a discrete set. Stanley *et al.* (1995, p. 395) found that individuals from the Bay-Delta population and Lake Washington (Washington State) population differed significantly in allele (proteins used as genetic markers) frequencies at several loci (gene locations), although the authors also stated that the overall genetic dissimilarity was within the range of other conspecific fish species. They concluded that longfin smelt from Lake Washington and the Bay-Delta are conspecific (of the same species) despite the large geographic separation.

Delta smelt and longfin smelt hybrids have been observed in the Bay-Delta estuary, although these offspring are not thought to be fertile because delta smelt and longfin smelt are not closely related taxonomically or genetically (California Department of Fish and Game (CDFG) 2001, p. 473).

Biology

Nearly all information available on longfin smelt biology comes from either the Bay-Delta population or the Lake Washington population. Longfin smelt generally spawn in freshwater and then move downstream to brackish water to rear. The life cycle of most longfin smelt generally requires estuarine conditions (CDFG 2009, p. 1).

Longfin smelt are considered pelagic and anadromous (Moyle 2002, p. 236), although anadromy in longfin smelt is poorly understood, and certain populations are not anadromous and complete their entire life cycle in freshwater lakes and streams. Within the Bay-Delta, the term pelagic refers to organisms that occur in open water away from the bottom of the water column and away from the shore.

Juvenile and adult longfin smelt have been found throughout the year in salinities ranging from pure freshwater to pure seawater, although once past the juvenile stage, they are typically collected in waters with salinities ranging from 14 to 28 parts

per thousand (ppt) (Baxter 1999, pp. 189–192). Longfin smelt are thought to be restricted by high water temperatures, generally greater than 22 degrees C (71 degrees F) (Baxter *et. al.* 2010, p. 68), and will move down the estuary (seaward) and into deeper water during the summer months, when water temperatures in the Bay-Delta are higher. Within the Bay-Delta, adult longfin smelt occupy water at temperatures from 16 to 20 C (61 to 68 F), with spawning occurring in water with temperatures from 5.6 to 14.5 C (41 to 58 F) (Wang 1986, pp. 6–9).

Longfin smelt usually live for 2 years, spawn, and then die, although some individuals may spawn as 1- or 3-year old fish before dying (Moyle 2002, p. 36). In the Bay-Delta, longfin smelt are believed to spawn primarily in freshwater in the lower reaches of the Sacramento River and San Joaquin River. Longfin smelt congregate in deep waters in the vicinity of the LSZ near X2 during the spawning period, and it is thought that they make short runs upstream, possibly at night, to spawn from these locations (CDFG 2009, p. 12; Rosenfield 2010, p. 8).

Salinity in psu is determined by electrical conductivity of a solution, whereas salinity in parts per thousand (ppt) is determined as the weight of salts in a solution. For use in this document, the two measurements are essentially equivalent. X2 is defined as the distance in kilometers up the axis of the estuary (to the east) from the Golden Gate Bridge to the location where the daily average near-bottom salinity is 2 psu (Jassby *et al.* 1995, p. 274; Dege and Brown 2004, p. 51).

Longfin smelt in the Bay-Delta may spawn as early as November and as late as June, although spawning typically occurs from January to April (CDFG 2009, p. 10; Moyle 2002, p. 36). Longfin smelt have been observed in their winter and spring spawning period as far upstream as Isleton in the Sacramento River, Santa Clara shoal in the San Joaquin system, Hog Slough off the South-Fork Mokelumne River, and in Old River south of Indian Slough (CDFG 2009a, p. 7; Radtke 1966, pp. 115–119). As **Table 4-2,** on the next page, illustrates, longfin smelt are most likely to be found in the Delta between November and March.

Exact spawning locations in the Delta are unknown and may vary from year to year in location, depending on environmental conditions. However, it seems likely that spawning locations consist of the overlap of appropriate conditions of flow, temperature, and salinity with appropriate substrate (Rosenfield 2010, p. 8). Longfin smelt are known to spawn over sandy substrates in Lake Washington and likely prefer similar substrates for spawning in the Delta (Baxter *et. al.* 2010, p. 62; Sibley and Brocksmith 1995, pp. 32–74). Baxter found that female longfin smelt produced between 1,900 and 18,000 eggs, with fecundity greater in fish with greater lengths (CDFG 2009, p. 11). At 7 C (44.6 F), embryos hatch in 40 days (Dryfoos 1965, p. 42); however, incubation time decreases with increased water temperature. At 8 to 9.5 C (46.4 to 49.1 F), embryos hatch at 29 days (Sibley and Brocksmith 1995, pp. 32–74).

Table 4-2
Summary of Longfin Smelt Life History Within the Bay-Delta, and Generalized Coastal Ocean Circulation

Month	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov
First Year		Peak Hatching – freshwater, upstream										
			Larval Rearing – San Pablo and San Francisco Bays – salinities < 8 psu									
	Juveniles Rearing				Juvenile Rearing – Primarily San Pablo, San Francisco Bays							
Second Year			Juvenile Rearing									
					Juvenile Rearing – Movement to the coastal ocean begins in the summer, mass movement to coastal ocean begins in July and August							
	Spawning Migration											Spawning Migration
	Peak Spawning – freshwater, Delta											
Coastal Current	Storm Season (Northward Flow				Upwelling Season (Predominate Southward and Offshore Flow)				Relaxation Season (Weak Northward Flow)			

Source: Federal Register, April 2, 2012, p.19779. Shaded areas indicate peak periods within the Delta.

Larval longfin smelt less than 12 millimeters (mm) in length are buoyant because they have not yet developed an air bladder; as a result, they occupy the upper one-third of the water column. After hatching, they quickly make their way to the LSZ via river currents (CDFG 2009, p. 8; Baxter 2011a, pers comm.). Longfin smelt develop an air bladder at approximately 12 to 15 mm (0.5 to 0.6 in.) in length and are able to migrate vertically in the water column. At this time, they shift habitat and begin living in the bottom two thirds of the water column (CDFG 2009, p. 8; Baxter 2008, p. 1).

Longfin smelt larvae can tolerate salinities of 2 to 6 psu within days of hatching, and can tolerate salinities up to 8 psu within weeks of hatching. Very few larvae (individuals less than 20 mm in length) are found in salinities greater than 8 psu, and it takes almost 3 months for longfin smelt to reach juvenile stage. A fraction of juvenile longfin smelt individuals are believed to tolerate full marine salinities (greater than 8 psu) (Baxter 2011a, pers. comm.).

Longfin smelt are dispersed broadly in the Bay-Delta by high flows and currents, which facilitate transport of larvae and juveniles long distances. Longfin smelt larvae are dispersed farther downstream during high freshwater flows (Dege and Brown 2004, p. 59). They spend approximately 21 months of their 24-month life cycle in brackish or marine waters (Baxter 1999, pp. 2–14; Dege and Brown 2004, pp. 58–60).

In the Bay-Delta, most longfin smelt spend their first year in Suisun Bay and Marsh, although surveys conducted by the City of San Francisco collected some first-year longfin in coastal waters (Baxter 2011c, pers. comm.; City of San Francisco 1995, no

pagination). The remainder of their life is spent in the San Francisco Bay or the Gulf of Farallones (Moyle 2008, p. 366; City of San Francisco 1995, no pagination). Rosenfield and Baxter (2007, pp. 1587, 1590) inferred based on monthly survey results that the majority of longfin smelt from the Bay-Delta were migrating out of the estuary after the first winter of their life cycle and returning during late fall to winter of their second year. They noted that migration out of the estuary into nearby coastal waters is consistent with captures of longfin smelt in the coastal waters of the Gulf of Farallones.

It is possible that some longfin smelt may stay in the ocean and not re-enter freshwater to spawn until the end of their third year of life (Baxter 2011d, pers. comm.). Moyle (2010, p. 8) states that longfin smelt that migrate out of and back into the Bay-Delta estuary may primarily be feeding on the rich planktonic food supply in the Gulf of Farallones. Rosenfield and Baxter (2007, p. 1290) hypothesize that the movement of longfin smelt into the ocean or deeper water habitat in summer months is at least partly a behavioral response to warm water temperatures found during summer and early fall in the shallows of south San Francisco Bay and San Pablo Bay (Rosenfield and Baxter 2007, p. 1590).

In the Bay-Delta, calanoid copepods such as *Pseudodiatomus forbesi* and *Eurytemora sp.,* as well as the cyclopoid copepod *Acanthocyclops vernali* (no common names), are the primary prey of longfin smelt during the first few months of their lives (approximately January through May) (Slater 2009b, slide 45). Copepods are a type of zooplankton (organisms drifting in the water column of oceans, seas, and bodies of fresh water). The longfin smelt's diet shifts to include mysids such as opossum shrimp (*Neomysis mercedis*) and other small crustaceans (*Acanthomysis* sp.) as soon as they are large enough (20 to 30 mm (0.78 to 1.18 in)) to consume these larger prey items, sometime during the summer months of the first year of their lives (CDFG 2009, p. 12). Upstream of San Pablo Bay, mysids and amphipods form 80 to 95 percent or more of the juvenile longfin smelt diet by weight from July through September (Slater 2009, unpublished data). Longfin smelt occurrence is likely associated with the occurrence of their prey, and both of these invertebrate groups occur near the bottom of the water column during the day under clear water marine conditions.

Abundance

In most locations throughout their range, longfin smelt populations have not been monitored. Within the Bay- Delta, longfin smelt are consistently collected in the monitoring surveys that have been conducted by CDFG as far back as the late 1960s. USFWS knows of no similar monitoring data for other longfin smelt populations. CDFG did report catches of longfin smelt in Humboldt Bay from surveys conducted between 2003 and 2009; small numbers of longfin were collected each of the years except 2004 (CDFG 2010, unpublished data). Moyle (2002, p. 237; 2010, p. 4) noted that the longfin smelt population in Humboldt Bay appeared to have declined between the 1970s and 2002, but survey data are not available from that time.

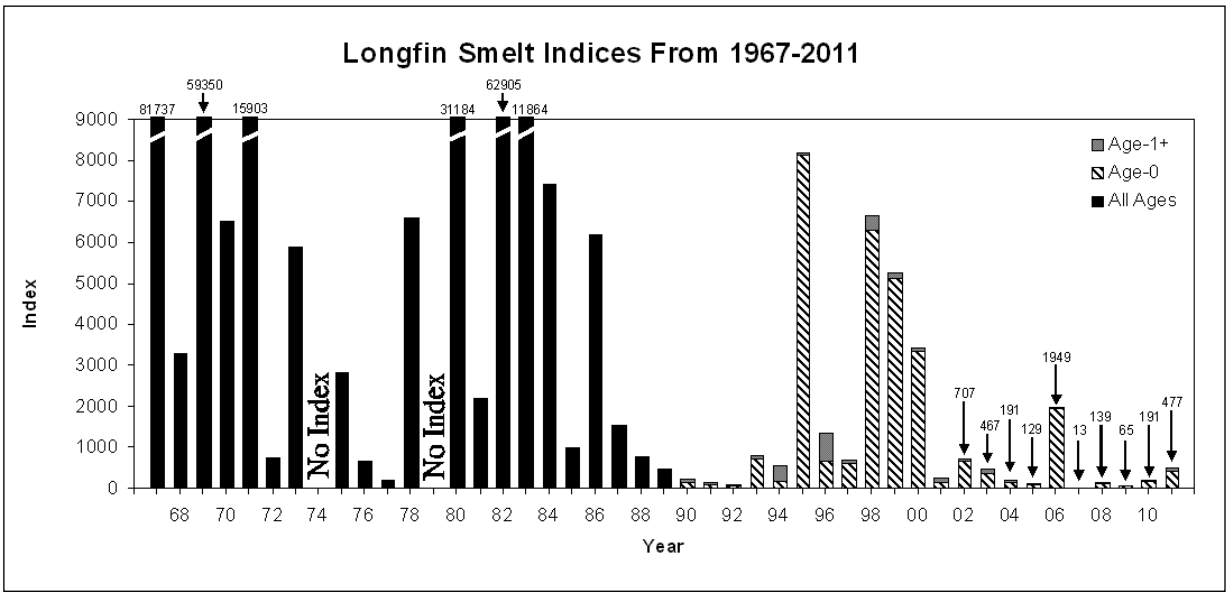
Longfin smelt numbers in the Bay-Delta have declined significantly since the 1980s (Moyle 2002, p. 237; Rosenfield and Baxter 2007, p. 1590; Baxter *et. al.* 2010, pp. 61–64). Rosenfield and Baxter (2007, pp. 1577–1592) examined abundance trends in longfin smelt using three long-term data sets (1980 to 2004) and detected a significant decline in the Bay-Delta longfin smelt population. They confirmed the positive correlation between longfin smelt abundance and freshwater flow that had been previously documented by others (Stevens and Miller 1983, p. 432; Baxter *et al.* 1999, p. 185; Kimmerer 2002b, p. 47), noting that abundances of both adults and juveniles were significantly lower during the 1987 to 1994 drought than during either the pre- or postdrought periods (Rosenfield and Baxter 2007, pp. 1583–1584).

Despite the correlation between drought and low population in the 1980s and 90s, the declines in the first decade of this century appear to be caused in part by additional factors. Abundance of longfin smelt has remained very low since 2000, even though freshwater flows increased during several of these years (Baxter *et al.* 2010, p. 62). Abundance indices derived

from the Fall Midwater Trawl (FMWT), Bay Study Midwater Trawl (BSMT), and Bay Study Otter Trawl (BSOT) all show marked declines in Bay-Delta longfin smelt populations from 2002 to 2009 (Messineo *et al.* 2010, p. 57). Longfin smelt abundance over the last decade is the lowest recorded in the 40-year history of CDFG's FMWT monitoring surveys. Scientists became concerned over the simultaneous population declines since the early 2000s of longfin smelt and three other Bay-Delta pelagic fish species—delta smelt (*Hypomesus transpacificus*), striped bass (*Morone saxatilis*), and threadfin shad (*Dorosoma petenense*) (Sommer *et al.* 2007, p. 273). The declines of longfin smelt and these other pelagic fish species in the Bay-Delta since the early 2000s has come to be known as the Pelagic Organism Decline, and considerable research efforts have been initiated since 2005, to better understand causal mechanisms underlying the declines (Sommer *et al.* 2007, pp. 270–277; MacNally *et al.* 2010, pp. 1417–1430; Thomson *et al.* 2010, pp. 1431–1448). The population did increase in the 2011 FMWT index to 477 (Contreras 2011, p. 2), probably a response to an exceptionally wet year.

The FMWT index of abundance in the Bay-Delta shows great annual variation in abundance but a severe decline over the past 40 years (**Figure 4-4,** on the next page). The establishment of the overbite clam (*Corbula amurensis*) in the Bay-Delta in 1987 is believed to have contributed to the population decline of longfin smelt, as well as to the declining abundance of other pelagic fish species in the Bay-Delta (Sommer *et al.* 2007, p. 274). Figure 4-4 shows low values of the abundance index for longfin smelt during drought years (1976–1977 and 1986–1992) and low values overall since the time that the overbite clam became established in the estuary.

Figure 4-4
Longfin Smelt Fall Midwater Trawl Indices (1967-2011)



Source: CDFG (http://www.dfg.ca.gov/delta/data/fmwt/Indices/sld003.asp)

Using data from 1975 to 2004 from the FMWT survey, Rosenfield and Baxter 2007 (p. 1589) found that longfin smelt exhibit a significant stock-recruitment relationship—abundance of juvenile (age-0) fish is directly related to the abundance of adult (age-1) fish from the previous year. They found that the abundance of juvenile fish declined by 90 percent during the time period analyzed. Rosenfield and Baxter (2007, p. 1589) also found a decline in age-1 individuals that was significant even after accounting for the decline in the age-0 population. If unfavorable environmental conditions persist for one or more years, recruitment into the population could be suppressed, affecting the species' ability to recover to their previous abundance. The current low abundance of adult longfin smelt within the Bay-Delta could reduce the ability of the species to persist in the presence of various threats.

B. NMFS Listed Species and Critical Habitats

Source: U.S. Geological Survey.

Photo: Chinook Salmon.

1. Endangered Sacramento River winter-run Chinook salmon *(Oncorhynchus tshawytsha)*

In 1989, the Sacramento River winter-run Chinook salmon was listed as threatened under the federal ESA by NMFS (54 FR 32085). NMFS reclassified the winter-run as endangered in 1994 (59 FR 440), and

reaffirmed this classification in 2005 (NMFS 2005). Winter-run Chinook salmon were classified by the State as endangered in 1989. In 1993, NMFS designated critical habitat for the winter-run Chinook from Keswick Dam (Sacramento river mile 302) to the Golden Gate Bridge (58 FR 33212) (Federal Register 2004). NMFS developed a draft recovery plan in 1997 that was never finalized. In the 2005 5-Year Review, NMFS determined that the endangered classification for winter-run Chinook salmon was still warranted. NMFS completed another 5-Year Review of Sacramento River winter-run Chinook salmon in August 2011, and again recommended maintaining the endangered classification (NMFS August 2011a). The 2011 review also recommended increasing the recovery priority number from 3 to 1 (based on a scale of 1 to 12 with 1 the highest priority).

Adult Sacramento River winter-run Chinook salmon migrate through the Delta from November through June. Juveniles spend approximately 40 days migrating through the Delta, and are primarily present from November through early May (NMFS March 2012). The major concerns related to the status of winter-run Chinook are that there is only one remaining extant population, and it is spawning outside of its historical range (below Keswick Dam and above the Red Bluff Diversion Dam (RBDD)) in artificially maintained habitat (cold water releases from Shasta Dam)that is vulnerable to drought and catastrophe (NMFS August 2011a).

The text below describing Sacramento River winter-run Chinook salmon in more detail is drawn from the March 8, 2012 Biological Opinion of the South Delta Temporary Barriers Program (NMFS March 2012). The initial discussion of general life history also relates to the Central Valley spring-run Chinook salmon.

General Chinook salmon Life History

Chinook salmon exhibit two generalized freshwater life history types (Healey 1991). “Streamtype” Chinook salmon, enter freshwater months before spawning and reside in freshwater for a year or more following emergence, whereas “ocean-type” Chinook salmon spawn soon after entering freshwater and migrate to the ocean as fry or parr within their first year. Spring-run Chinook salmon can exhibit a stream-type life history. Adults enter freshwater in the spring, hold over summer, spawn in the fall, and some of the juveniles may spend a year or more in freshwater before emigrating.

The remaining fraction of the juvenile spring-run population may also emigrate to the ocean as young-of-the-year in spring. Winter-run Chinook salmon are somewhat anomalous in that they have characteristics of both stream- and ocean-type races (Healey 1991). Adults enter freshwater in winter or early spring, and delay spawning until spring or early summer (stream-type). However, juvenile winter-run Chinook salmon migrate to sea after only 4 to 7 months of river life (ocean-type). Adequate instream flows and cool water temperatures are more critical for the survival of Chinook salmon exhibiting a stream-type life history due to over summering by adults and/or juveniles.

Chinook salmon typically mature between 2 and 6 years of age (Myers *et al.* 1998). Freshwater entry and spawning timing generally are thought to be related to local

water temperature and flow regimes. Runs are designated on the basis of adult migration timing; however, distinct runs also differ in the degree of maturation at the time of river entry, thermal regime and flow characteristics of their spawning site, and the actual time of spawning (Myers *et al.* 1998). Both spring-run and winter-run Chinook salmon tend to enter freshwater as immature fish, migrate far upriver, and delay spawning for weeks or months. For comparison, fall-run Chinook salmon enter freshwater at an advanced stage of maturity, move rapidly to their spawning areas on the main stem or lower tributaries of the rivers, and spawn within a few days or weeks of freshwater entry (Healey 1991).

During their upstream migration, adult Chinook salmon require stream flows sufficient to provide olfactory and other orientation cues used to locate their natal streams. Adequate stream flows are necessary to allow adult passage to upstream holding habitat. The preferred temperature range for upstream migration is 38 F to 56 F (Bell 1991, CDFG 1998). Boles (1988) recommends water temperatures below 65 F for adult Chinook salmon migration, and Lindley *et al.* (2004) report that adult migration is blocked when temperatures reach 70 F, and that fish can become stressed as temperatures approach 70 F. Reclamation reports that spring-run Chinook salmon holding in upper watershed locations prefer water temperatures below 60 F; although salmon can tolerate temperatures up to 65 F before they experience an increased susceptibility to disease (Williams 2006).

Information on the migration rates of Chinook salmon in freshwater is scant and primarily comes from the Columbia River basin where information regarding migration behavior is needed to assess the effects of dams on travel times and passage (Matter *et al.* 2003). Keefer *et al.* (2004) found migration rates of Chinook salmon ranging from approximately 10 kilometers (km) per day to greater than 35 km per day and to be primarily correlated with date, and secondarily with discharge, year, and reach, in the Columbia River basin. Matter *et al.* (2003) documented migration rates of adult Chinook salmon ranging from 29 to 32 km per day in the Snake River.

Adult Chinook salmon inserted with sonic tags and tracked throughout the Delta and lower Sacramento and San Joaquin rivers were observed exhibiting substantial upstream and downstream movement in a random fashion while migrating upstream over the course of several days at a time (CALFED 2001). Adult salmonids migrating upstream are assumed to make greater use of pool and mid-channel habitat than channel margins (Stillwater Sciences 2004), particularly larger salmon such as Chinook salmon, as described by Hughes (2004). Adults are thought to exhibit crepuscular behavior during their upstream migrations; meaning that they primarily are active during twilight hours. Recent hydroacoustic monitoring showed peak upstream movement of adult Central Valley spring-run Chinook salmon in lower Mill Creek, a tributary to the Sacramento River, occurring in the 4-hour period before sunrise and again after sunset.

Spawning Chinook salmon require clean, loose gravel in swift, relatively shallow riffles or along the margins of deeper runs, and suitable water temperatures, depths, and velocities for red construction and adequate

oxygenation of incubating eggs. Chinook salmon spawning typically occurs in gravel beds that are located at the tails of holding pools (USFWS 1995a). The range of water depths and velocities in spawning beds that Chinook salmon find acceptable is very broad.

The upper preferred water temperature for spawning Chinook salmon is 55 F to 57 F (Chambers 1956, Smith 1973, Bjornn and Reiser 1991, and Snider 2001).Incubating eggs are vulnerable to adverse effects from floods, siltation, desiccation, disease, predation, poor gravel percolation, and poor water quality. Studies of Chinook salmon egg survival to hatching conducted by Shelton (1995) indicated 87 percent of fry emerged successfully from large gravel with adequate subgravel flow. The optimal water temperature for egg incubation ranges from 41 F to 56 F (44 F to 54 F [Rich 1997], 46 F to 56 F [NMFS 1997 Winter-run Chinook salmon Recovery Plan], and 41 F to 55.4 F [Moyle 2002]). A significant reduction in egg viability occurs at water temperatures above 57.5 F and total embryo mortality can occur at temperatures above 62 F (NMFS 1997). Alderdice and Velsen (1978) found that the upper and lower temperatures resulting in 50 percent pre-hatch mortality were 61 F and 37 F, respectively, when the incubation temperature was held constant. As water temperatures increase, the rate of embryo malformations also increases, as well as the susceptibility to fungus and bacterial infestations. The length of development for Chinook salmon embryos is dependent on the ambient water temperature surrounding the egg pocket in the redd. Colder water necessitates longer development times as metabolic processes are slowed. Within the appropriate water temperature range for embryo incubation, embryos hatch in 40 to 60 days, and the alevins (yolk-sac fry) remain in the gravel for an additional 4 to 6 weeks before emerging from the gravel.

During the 4 to 6 week period when alevins remain in the gravel, they utilize their yolk-sac to nourish their bodies. As their yolk-sac is depleted, fry begin to emerge from the gravel to begin exogenous feeding in their natal stream. The post-emergent fry disperse to the margins of their natal stream, seeking out shallow waters with slower currents, finer sediments, and bank cover such as overhanging and submerged vegetation, root wads, and fallen woody debris, and begin feeding on zooplankton, small insects, and small invertebrates. As they switch from endogenous nourishment to exogenous feeding, the fry's yolk-sac is reabsorbed, and the belly suture closes over the former location of the yolk-sac (button-up fry). Fry typically range from 25 mm to 40 mm during this stage. Some fry may take up residence in their natal stream for several weeks to a year or more, while others are displaced downstream by the stream's current. Once started downstream, fry may continue downstream to the estuary and rear, or may take up residence in river reaches farther downstream for a period of time ranging from weeks to a year (Healey 1991).

Fry then seek nearshore habitats containing beneficial aspects such as riparian vegetation and associated substrates important for providing aquatic and terrestrial invertebrates, predator avoidance, and slower velocities for resting (NMFS 1996a). The benefits of shallow water habitats for salmonid rearing also have recently been realized as shallow water habitat

has been found to be more productive than the main river channels, supporting higher growth rates, partially due to higher prey consumption rates, as well as favorable environmental temperatures (Sommer *et al.* 2001).

When juvenile Chinook salmon reach a length of 50 mm to 57 mm, they move into deeper water with higher current velocities, but still seek shelter and velocity refugia to minimize energy expenditures. In the mainstems of larger rivers, juveniles tend to migrate along the margins and avoid the elevated water velocities found in the thalweg of the channel. When the channel of the river is greater than 9 feet to 10 feet in depth, juvenile salmon tend to inhabit the surface waters (Healey 1982). Migrational cues, such as increasing turbidity from runoff, increased flows, changes in day length, or intraspecific competition from other fish in their natal streams may spur outmigration of juveniles when they have reached the appropriate stage of maturation (Kjelson *et al.* 1982, Brandes and McLain 2001).

As fish begin their emigration, they are displaced by the river's current downstream of their natal reaches. Similar to adult movement, juvenile salmonid downstream movement is crepuscular. Documents and data provided to NMFS in support of ESA section 10 research permit applications depicts that the daily migration of juveniles passing RBDD is highest in the four hour period prior to sunrise (Martin *et al.* 2001). Juvenile Chinook salmon migration rates vary considerably presumably depending on the physiological stage of the juvenile and hydrologic conditions. Kjelson *et al*. (1982) found fry Chinook salmon to travel as fast as 30 km per day in the Sacramento River and Sommer *et al.* (2001) found rates ranging from approximately 0.5 miles up to more than 6 miles per day in the Yolo Bypass. As Chinook salmon begin the smoltification stage, they prefer to rear further downstream where ambient salinity is up to 1.5 to 2.5 parts per thousand (Healey 1980, Levy and Northcote 1981).

Fry and parr may rear within riverine or estuarine habitats of the Sacramento River, the Delta, and their tributaries. In addition, Central Valley spring-run Chinook salmon juveniles have been observed rearing in the lower reaches of non-natal tributaries and intermittent streams in the Sacramento Valley during the winter months (Maslin *et al.* 1997, Snider 2001). Within the Delta, juvenile Chinook salmon forage in shallow areas with protective cover, such as intertidal and subtidal mudflats, marshes, channels, and sloughs (McDonald 1960, Dunford 1975). Cladocerans, copepods, amphipods, and larvae of diptera, as well as small arachnids and ants are common prey items (Kjelson *et al.* 1982, Sommer *et al.* 2001, MacFarlane and Norton 2002).

Shallow water habitats are more productive than the main river channels, supporting higher growth rates, partially due to higher prey consumption rates, as well as favorable environmental temperatures (Sommer *et al.* 2001). Optimal water temperatures for the growth of juvenile Chinook salmon in the Delta are between 54 F to 57 F (Brett 1952). In Suisun and San Pablo Bays water temperatures can reach 54 F by February in a typical year. Other portions of the Delta (*i.e.*, south Delta and central Delta) can reach 70 F by February in a dry year. However, cooler temperatures are usually the norm until after the spring runoff has ended.

Within the estuarine habitat, juvenile Chinook salmon movements are dictated by the tidal cycles, following the rising tide into shallow water habitats from the deeper main channels, and returning to the main channels when the tide recedes (Levy and Northcote 1982, Levings 1982, Levings *et al.* 1986, Healey 1991). As juvenile Chinook salmon increase in length, they tend to school in the surface waters of the main and secondary channels and sloughs, following the tides into shallow water habitats to feed (Allen and Hassler 1986). In Suisun Marsh, Moyle *et al.* (1989) reported that Chinook salmon fry tend to remain close to the banks and vegetation, near protective cover, and in dead-end tidal channels. Kjelson *et al.* (1982) reported that juvenile Chinook salmon demonstrated a diel migration pattern, orienting themselves to nearshore cover and structure during the day, but moving into more open, offshore waters at night. The fish also distributed themselves vertically in relation to ambient light. During the night, juveniles were distributed randomly in the water column, but would school up during the day into the upper 3 meters of the water column. Available data indicates that juvenile Chinook salmon use Suisun Marsh extensively both as a migratory pathway and rearing area as they move downstream to the Pacific Ocean. Juvenile Chinook salmon were found to spend about 40 days migrating through the Delta to the mouth of San Francisco Bay and grew little in length or weight until they reached the Gulf of the Farallones (MacFarlane and Norton 2002). Based on the mainly oceantype life history observed (*i.e.*, fall-run Chinook salmon) MacFarlane and Norton (2002) concluded that unlike other salmonid populations in the Pacific Northwest, Central Valley Chinook salmon show little estuarine dependence and may benefit from expedited ocean entry.

Sacramento River winter-run Chinook salmon

The distribution of winter-run Chinook salmon spawning and rearing historically was limited to the upper Sacramento River and its tributaries, where spring-fed streams provided cold water throughout the summer, allowing for spawning, egg incubation, and rearing during the midsummer period (Slater 1963, Yoshiyama *et al.* 1998). The headwaters of the McCloud, Pit, and Little Sacramento rivers, and Hat and Battle creeks, historically provided clean, loose gravel; cold, well-oxygenated water; and optimal stream flow in riffle habitats for spawning and incubation. These areas also provided the cold, productive waters necessary for egg and fry development and survival, and juvenile rearing over the summer. The construction of Shasta Dam in 1943 blocked access to all of these waters except Battle Creek, which has its own impediments to upstream migration (*i.e.*, the fish weir at the Coleman National Fish Hatchery and other small hydroelectric facilities situated upstream of the weir) (Moyle *et al.* 1989, NMFS 1997, 1998a,b). Approximately 299 miles of tributary spawning habitat in the upper Sacramento River is now inaccessible to winter-run Chinook salmon. Yoshiyama *et al.* (2001) estimated that in 1938, the Upper Sacramento had a "potential spawning capacity" of 14,303 redds. Most components of the winter-run Chinook salmon life history (*e.g.*, spawning, incubation, freshwater rearing) have been compromised by the habitat blockage in the upper Sacramento River.

Table 4-3
The Temporal Occurrence of Adult (a) and Juvenile (b) Sacramento River winter-run Chinook salmon in the Sacramento River
(Darker shades indicate months of greatest relative abundance)

(a) Adult migration/holding

Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. River basin[a]												
Sac. River[b]												

(b) Juvenile migration

Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. River @ Red Bluff[c]												
Sac. River @ Red Bluff[b]												
Sac. River @ Knights Landing[d]												
Lower Sac. River (seine)[e]												
West Sac. River (trawl)[e]												

Relative Abundance: ■ = High ■ = Medium ■ = Low

Sources: [a]Yoshiyama *et al.* (1998); Moyle (2002); [b]Myers *et al.* (1998); Vogel and Marine (1991); [c]Martin *et all.* (2001); [d]Snider and Titus (2000); [e]USFWS (2001a, 2001b)

Source: NMFS, March 8, 2012, p. 15.

Adult winter-run Chinook salmon enter San Francisco Bay from November through June (Hallock and Fisher 1985) and migrate past the RBDD from mid-December through early August (NMFS 1997). The majority of the run passes RBDD from January through May, with the peak passage occurring in mid-March (Hallock and Fisher 1985). The timing of migration may vary somewhat due to changes in river flows, dam operations, and water year type Yoshiyama *et al.* 1998, Moyle 2002). **Table 4-3,** above, illustrates winter-run Chinook salmon location and timing. Spawning occurs primarily from mid-April to mid-August, with the peak activity occurring in May and June in the Sacramento River reach between Keswick Dam and RBDD (Vogel and Marine 1991). The majority of Sacramento River winter-run Chinook salmon spawners are 3 years old.

Sacramento River winter-run Chinook salmon fry begin to emerge from the gravel in late June to early July and continue through October (Fisher 1994). Emigration of juvenile Sacramento River winter-run Chinook salmon past RBDD may begin as early as mid-July, typically peaks in September, and can continue through March in dry years (Vogel and Marine 1991, NMFS 1997). Juvenile Sacramento River winter-run Chinook salmon occur in the Delta primarily from November through early May based on data collected from trawls in the Sacramento River at West Sacramento (RM 57; USFWS 2001a,b). The timing of migration may vary somewhat due to changes in river flows, dam operations, and water year type. Winter-run Chinook salmon juveniles remain in the Delta until they reach a fork length of approximately 118 millimeters (mm) and 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 *et al.* 1998).

Historical Sacramento River winter-run Chinook salmon population estimates, which included males and females, were as high as near 100,000 fish in the 1960s, but declined to under 200 fish in the 1990s (Good *et al.* 2005). Population estimates in 2003 (8,218), 2004 (7,869), 2005 (15,875) and 2006 (17,304) show a recent increase in the population size (CDFG GrandTab, February 2011) and a 4-year average of 12,316. **Table 4-4,** on the next page, summarizes winter-run Chinook salmon population data from 1986 to 2011. The 2006 run was the highest since the 1994 listing. Abundance measures over the last decade suggest that the abundance was initially increasing (Good *et al.* 2005). However, escapement estimates for 2007, 2008, 2009, and 2010 show a precipitous decline in escapement numbers based on redd counts and carcass counts. Estimates place the adult escapement numbers for 2007 at 2,542 fish, 2,830 fish for 2008, and 4,658 fish for 2009 (CDFG Grand Tab 2010) and 1,596 fish for 2010 (NMFS 2011[JPE letter]).

Two current methods are utilized to estimate the juvenile production of Sacramento River winter-run Chinook salmon: the Juvenile Production Estimate (JPE) method, and the Juvenile Production Index (JPI) method (Gaines and Poytress 2004). Gaines and Poytress (2004) estimated the juvenile population of Sacramento River winter-run Chinook salmon exiting the upper Sacramento River at RBDD to be 3,707,916 juveniles per year using the JPI method between the years 1995 and 2003 (excluding 2000 and 2001). Using the JPE method, they estimated an average of 3,857,036 juveniles exiting the upper Sacramento River at RBDD between the years of 1996 and 2003. Averaging these two estimates yields an estimated population size of 3,782,476.

Based on the RBDD counts, the population has been growing rapidly since the 1990s with positive short-term trends (excluding the 2007-2010 escapement numbers). An age-structured density-independent model of spawning escapement by Botsford and Brittnacker (1998 as referenced in Good *et al.* 2005) assessing the viability of Sacramento River winter-run Chinook salmon found the species was certain to fall below the quasi-extinction threshold of 3 consecutive spawning runs with fewer than 50 females (Good *et al.* 2005). Lindley *et al.* (2003) assessed the viability of the population using a Bayesian model based on spawning escapement that allowed for density dependence and a change in population growth rate in response to conservation measures found a biologically significant expected quasi-extinction probability of 28 percent.

Although the status of the Sacramento River winter-run Chinook salmon population had been improving until as recently as 2006, there is only one population, and it depends on cold-water releases from Shasta Dam, which could be vulnerable to a prolonged drought (Good *et al.* 2005). Recent population trends in the previous 4 years have indicated that the status of the winter-run Chinook salmon population may be changing as reflected in the diminished abundance during this period. The 2011 winter-run Chinook salmon JPE in Table 4-4 is only 162,051 fish entering the Delta, a substantial decline from the previous JPE values seen in the last decade.

Table 4-4

Winter-run Chinook salmon Population Estimates from RBDD Counts (1986 to 2001) and Carcass Counts (2001 to 2011), and Corresponding Cohort Replacement Rates for the Years Since 1986 (CDFG Grand Tab February 2011)

Year	Population Estimate[a]	5-Year Moving Average of Population Estimate	Cohort Replacement Rate[b]	5-Year Moving Average of Cohort Replacement Rate	NMFS-Calculated Juvenile Production Estimate (JPE)[c]
1986	2,596				
1987	2,185				
1988	2,878				
1989	696		0.27		
1990	430	1,757	0.20		
1991	211	1,280	0.07		40,100
1992	1,240	1,091	1.78		273,100
1993	387	593	0.90	0.64	90,500
1994	186	491	0.88	0.77	74,500
1995	1,297	664	1.05	0.94	338,107
1996	1,337	889	3.45	1.61	165,069
1997	880	817	4.73	2.20	138,316
1998	2,992	1,338	2.31	2.48	454,792
1999	3,288	1,959	2.46	2.80	289,724
2000	1,352	1,970	1.54	2.90	370,221
2001	8,224	3,347	2.75	2.76	1,864,802
2002	7,441	4,659	2.26	2.26	2,136,747
2003	8,218	5,705	6.08	3.02	1,896,649
2004	7,869	6,621	0.96	2.72	881,719
2005	15,839	9,518	2.13	2.84	3,556,995
2006	17,296	11,333	2.10	2.71	3,890,534
2007	2,542	10,353	0.32	2.32	1,100,067
2008	2,830	9,275	0.18	1.14	1,152,043
2009	4,537	8,609	0.26	1.00	1,144,860
2010	1,596	5,760	0.63	0.70	332,012
2011	824	2,466	0.29	0.34	162,051
Median	2,364	2,218	1.05	2.26	370,221
Mean	3,814	4,113	1.63	1.90	969,186

[a] NMFS included both the escapement numbers from the Feather River Fish Hatchery (FRFH) and the Sacramento River and its tributaries in this table. Sacramento River Basin run size is the sum of the escapement numbers from the FRFH and the tributaries.

[b] Abbreviations: CRR = Cohort Replacement Rate, Trib = tributary

Source: NMFS, March 8, 2012, p. 22.

Recently, Lindley *et al.* (2007) determined that the Sacramento River winter-run Chinook salmon population that spawns below Keswick Dam is at a moderate extinction risk according to population viability analysis (PVA), and at a low risk according to other criteria (*i.e.*, population size, population decline, and the risk of wide ranging catastrophe). However, concerns of genetic introgression with hatchery populations are increasing. Hatchery-origin winter-run Chinook salmon from Livingston Stone National Fish Hatchery (LSNFH) have made up more than 5 percent of the natural spawning run in recent years and in 2005, it exceeded 18 percent of the natural run. If the proportion of hatchery origin fish from the LSNFH exceeded 15 percent in 2006-2007, Lindley *et al*. (2007) recommended reclassifying the winter-run Chinook population extinction risk as moderate, rather than low, based on the impacts of the hatchery fish over multiple generations of spawners. However, since 2005, the percentage of hatchery fish recovered at the LSNFH has been consistently below 15 percent. Furthermore, Lindley's assessment in 2007 did not include the recent declines in adult escapement abundance which may modify the conclusion reached in 2007.

Lindley *et al.* (2007) also states that the winter-run Chinook salmon population fails the "representation and redundancy rule" because it has only one population, and that population spawns outside of the ecoregion in which it evolved. In order to satisfy the "representation and redundancy rule," at least two populations of winter-run Chinook salmon would have to be reestablished in the basalt- and porous-lava region of its origin. An ESU represented by only one spawning population at moderate risk of extinction is at a high risk of extinction over an extended period of time (Lindley *et al*. 2007).

Viable Salmonid Population Summary for Sacramento River winter-run Chinook salmon

Abundance

During the first part of this decade, redd and carcass surveys as well as fish counts, suggested that the abundance of winter-run Chinook salmon was increasing since its listing. However, the depressed abundance estimates over the past five years are an exception to this trend and may represent a combination of a new cycle of poor ocean productivity (Lindley *et al.* 2009) and recent drought conditions in the Central Valley. Population growth is estimated to be positive in the short-term trend at 0.26; however, the long-term trend is negative, averaging - 0.14. Recent winter-run Chinook salmon abundance represents only 3 percent of the maximum post-1967, 5-year geometric mean, and is not yet well established (Good *et al.* 2005). The current annual and five year averaged cohort replacement rates (CRR) are both below 0.5. The annual CRR has been below 1.0 for the past five years and indicates that the winter-run population is not replacing itself.

Productivity

ESU productivity has been positive over the short term, and adult escapement and juvenile production had been increasing annually (Good *et al.* 2005) until recently, with declining escapement estimates for the years 2007 through 2011. However, the long-term trend for the ESU remains negative, as it consists of only one population that is subject to possible impacts from environmental and

artificial conditions. The most recent CRR estimates suggest a reduction in productivity for the three separate cohorts starting in 2007.

Spatial Structure

The greatest risk factor for winter-run Chinook salmon lies with their spatial structure (Good *et al.* 2005). The remnant population cannot access historical winter-run Chinook salmon habitat and must be artificially maintained in the Sacramento River by a regulated, finite cold-water pool behind Shasta Dam. Winter-run Chinook salmon require cold water temperatures in summer that simulate their upper basin habitat, and they are more likely to be exposed to the impacts of drought in a lower basin environment. Battle Creek remains the most feasible opportunity for the ESU to expand its spatial structure, which currently is limited to the upper 25-mile reach of the mainstem Sacramento River below Keswick Dam. Based on Reasonable and Prudent Alternative actions described in the 2009 OCAP Biological Opinion (BiOp), passage of winter-run Chinook salmon above Keswick and Shasta Dams is being considered as one of the actions. This would reintroduce winter-run Chinook salmon into regions they had historically occupied and significantly benefit the spatial structure of the ESU.

Diversity

The second highest risk factor for the Sacramento River winter-run Chinook salmon ESU has been the detrimental effects on its diversity. The present winter-run Chinook salmon population has resulted from the introgression of several stocks that occurred when Shasta Dam blocked access to the upper watershed. A second genetic bottleneck occurred with the construction of Keswick Dam; and there may have been several others within the recent past (Good *et al.* 2005). Concerns of genetic introgression with hatchery populations are also increasing. Hatchery-origin winter-run Chinook salmon from LSNFH have made up more than 5 percent of the natural spawning run in recent years and in 2005, it exceeded 18 percent of the natural run. The average over the last 10 years (approximately 3 generations) has been 8 percent, still below the low-risk threshold for hatchery influence. Since 2005, the percentage of hatchery fish in the river has been consistently below 15 percent.

Endangered Sacramento River winter-run Chinook salmon Critical Habitat

The designated critical habitat for Sacramento River winter-run Chinook salmon includes the Sacramento River from Keswick Dam (RM 302) to Chipps Island (RM 0) at the westward margin of the Delta; all waters from Chipps Island westward to Carquinez Bridge, including Honker Bay, Grizzly Bay, Suisun Bay, and Carquinez Strait; all waters of San Pablo Bay westward of the Carquinez Bridge; and all waters of San Francisco Estuary to the Golden Gate Bridge north of the San Francisco/Oakland Bay Bridge. In the Sacramento River, critical habitat includes the river water column, river bottom, and adjacent riparian zone used by fry and juveniles for rearing. The portion of the Sacramento River within the Legal Delta includes potential WHCP treatment sites. In the areas westward of Chipps Island, critical habitat includes the estuarine water column

and essential foraging habitat and food resources used by Sacramento River winter-run Chinook salmon as part of their juvenile emigration or adult spawning migration. Critical habitat primary constituent elements (PCEs) for winter-run Chinook salmon are described below under Central Valley spring-run Chinook salmon.

2. Threatened Central Valley spring-run Chinook salmon *(Oncorhynchus tshawytsha)*

Central Valley spring-run salmon was listed as threatened by both the State and federal governments in 1999, and reaffirmed as threatened by the federal government in 2005. Critical habitat for Central Valley spring-run Chinook salmon was designated in September 2005. Critical habitat within the Delta includes portions of three hydrologic units: Sacramento Delta, Valley Putah-Cache, and Valley-American. Unlike winter-run Chinook, which utilize only the Sacramento River, spring-run Chinook utilize primarily the Feather and Yuba Rivers, with smaller populations likely in the Sacramento River and Big Chico Creek (NMFS 2005).

NMFS developed a draft recovery plan in 1997 that was never finalized. In the 2005 5-Year Review, NMFS determined that the threatened classification for spring-run Chinook salmon was still warranted. NMFS completed another 5-Year Review of Central Valley spring-run Chinook salmon in August 2011. At this time, NMFS determined that the status of this ESU has probably deteriorated since 2005, and again recommended maintaining the threatened classification (NMFS August 2011b). The 2011 review placed the Mill and Deer creek populations of spring-run Chinook salmon in the high extinction risk category, and the Butte Creek population in the low risk category (NMFS August 2011b). The 2011 review also recommended no change in the recovery priority number of 7 (based on a scale of 1 to 12 with 1 the highest priority), but that the status be reevaluated in two to three years. Major concerns related to the status of Central Valley spring-run Chinook salmon include: low diversity, poor spatial structure, risk of catastrophic disturbance, and low abundance resulting from loss of historical spawning habitat, degradation of remaining habitat, and genetic threats from the Feather River Hatchery (NMFS August 2011b).

Adult Central Valley spring-run Chinook salmon migrate through the Delta primarily from January through June. Most juveniles emigrate through the Delta from November through early May (NMFS March 2012). The text below describing the status of Central Valley spring-run Chinook salmon is drawn from the March 8, 2012 Biological Opinion of the South Delta Temporary Barriers Program (NMFS March 2012).

Historically the spring-run Chinook salmon were the second most abundant salmon run in the Central Valley (CDFG 1998). These fish occupied the upper and middle reaches (1,000 to 6,000 feet) of the San Joaquin, American, Yuba, Feather, Sacramento, McCloud and Pit rivers, with smaller populations in most tributaries with sufficient habitat for over-summering adults (Stone 1874, Rutter 1904, Clark 1929). The Central Valley Technical Review Team

(CVTRT) estimated that historically there were 18 or 19 independent populations of Central Valley spring-run Chinook salmon, along with a number of dependent populations and four diversity groups (Lindley *et al.* 2004). Of these 18 populations, only three extant populations currently exist (Mill, Deer, and Butte creeks on the upper Sacramento River) and they represent only the northern Sierra Diversity group. All populations in the Basalt and Porous Lava group and the Southern Sierra Nevada Group have been extirpated.

The Central Valley drainage as a whole is estimated to have supported spring-run Chinook salmon runs as large as 600,000 fish between the late 1880s and 1940s (CDFG 1998). Before the construction of Friant Dam, nearly 50,000 adults were counted in the San Joaquin River alone (Fry 1961). Construction of other low elevation dams in the foothills of the Sierras on the American, Mokelumne, Stanislaus, Tuolumne, and Merced rivers extirpated Central Valley spring-run Chinook salmon from these watersheds. Naturally-spawning populations of Central Valley spring-run Chinook salmon currently are restricted to accessible reaches of the upper Sacramento River, Antelope Creek, Battle Creek, Beegum Creek, Big Chico Creek, Butte Creek, Clear Creek, Deer Creek, Feather River, Mill Creek, and Yuba River (CDFG 1998).

Adult Central Valley spring-run Chinook salmon leave the ocean to begin their upstream migration in late January and early February (CDFG 1998) and enter the Sacramento River between March and September, primarily in May and June Yoshiyama *et al.* 1998, Moyle 2002). **Table 4-5,** on the next page, summarizes Central Vallley spring-run location and timing. Lindley *et al.* (2007) indicates adult Central Valley spring-run Chinook salmon enter native tributaries from the Sacramento River primarily between mid-April and mid-June. Typically, spring-run Chinook salmon utilize mid- to high-elevation streams that provide appropriate temperatures and sufficient flow, cover, and pool depth to allow over-summering while conserving energy and allowing their gonadal tissue to mature (Yoshiyama *et al.* 1998). Spring-run Chinook salmon spawning occurs between September and October depending on water temperatures. Between 56 and 87 percent of adult spring-run Chinook salmon that enter the Sacramento River basin to spawn are 3 years old (Calkins *et al.* 1940, Fisher 1994).

Spring-run Chinook salmon fry emerge from the gravel from November to March (Moyle 2002) and the emigration timing is highly variable, as they may migrate downstream as young-of-the-year or as juveniles or yearlings. The modal size of fry migrants at approximately 40 mm between December and April in Mill, Butte, and Deer creeks reflects a prolonged emergence of fry from the gravel (Lindley *et al.* 2007). Studies in Butte Creek (Ward *et al.* 2002, 2003, McReynolds *et al.* 2005) found the majority of Central Valley spring-run Chinook salmon migrants to be fry occurring primarily during December, January, and February; and that these movements appeared to be influenced by flow. Small numbers of Central Valley spring-run Chinook salmon remained in Butte Creek to rear and migrated as yearlings later in the spring.

Table 4-5
The Temporal Occurrence of Adult (a) and Juvenile (b) Central Valley spring-run Chinook salmon in the Sacramento River
(Darker shades indicate months of greatest relative abundance)

(a) Adult migration

Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. River basin[a,b]												
Sac. River mainstem[c]												
Mill Creek[d]												
Deer Creek[d]												
Butte Creek[d]												
(b) Adult holding												
(c) Adult Spawning												

(d) Juvenile migration

Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sac. River Tribs[e]												
Upper Butte Creek[f]												
Mill, Deer, Butte Creeks[d]												
Sac. River at RBDD[c]												
Sac. River at Knights Landing												

Relative Abundance: ■ = High ■ = Medium ■ = Low

Note: Yearling spring-run Chinook salmon rear in their natal streams through the first summer following their birth. Downstream emigration generally occurs the following fall and winter. Young of the year spring-run Chinook salmon emigrate during the first spring after they hatch.

Sources: [a]Yoshiyama *et al.* (1998); [b]Moyle (2002); [c]Myers *et al.* (1998); [d]Lindley *et al.* (2007). [e]CDFG (21998); [f]McReynolds *et al.* (2005); Ward *et al.* (2002, 2003); [g]Snider and Titus (2000)

Source: NMFS, March 8, 2012, p. 20.

Juvenile emigration patterns in Mill and Deer creeks are very similar to patterns observed in Butte Creek, with the exception that Mill and Deer creek juveniles typically exhibit a later young-of-the-year migration and an earlier yearling migration (Lindley *et al.* 2007). Once juveniles emerge from the gravel they initially seek areas of shallow water and low velocities while they finish absorbing the yolk sac and transition to exogenous feeding (Moyle 2002). Many also will disperse downstream during high-flow events. As is the case in other salmonids, there is a shift in microhabitat use by juveniles to deeper faster water as they grow larger. Microhabitat use can be influenced by the presence of predators which can force fish to select areas of heavy cover and suppress foraging in open areas (Moyle 2002). The emigration period for spring-run Chinook salmon extends from

November to early May, with up to 69 percent of the young-of-the-year fish outmigrating through the lower Sacramento River and Delta during this period (CDFG 1998). Peak movement of juvenile Central Valley spring-run Chinook salmon in the Sacramento River at Knights Landing occurs in December, and again in March and April. However, juveniles also are observed between November and the end of May (Snider and Titus 2000). Based on the available information, the emigration timing of Central Valley spring-run Chinook salmon appears highly variable (CDFG 1998). Some fish may begin emigrating soon after emergence from the gravel, whereas others over-summer and emigrate as yearlings with the onset of intense fall storms (CDFG 1998).

On the Feather River, significant numbers of spring-run Chinook salmon, as identified by run timing, return to the Feather River Hatchery (FRH). In 2002, the FRH reported 4,189 returning spring-run Chinook salmon, which is 22 percent below the 10-year average of 4,727 fish. However, coded-wire tag (CWT) information from these hatchery returns indicates substantial introgression has occurred between fall-run and spring-run Chinook salmon populations within the Feather River system due to hatchery practices. Because Chinook salmon have not always been temporally separated in the hatchery, spring-run and fall-run Chinook salmon have been spawned together, thus compromising the genetic integrity of the spring-run Chinook salmon stock. The number of naturally spawning spring-run Chinook salmon in the Feather River has been estimated only periodically since the 1960s, with estimates ranging from two fish in 1978 to 2,908 in 1964. However, the genetic integrity of this population is questionable because of the significant temporal and spatial overlap between spawning populations of spring-run and fall-run Chinook salmon (Good *et al.* 2005). For the reasons discussed above, the Feather River spring-run Chinook population numbers are not included in the following discussion of ESU abundance.

The Central Valley spring-run Chinook salmon ESU has displayed broad fluctuations in adult abundance, ranging from 1,404 in 1993 to 24,903 in 1998. **Table 4-6,** on the next page, summarizes Central Valley spring-run Chinook salmon population data from 1986 through 2011. Sacramento River tributary populations in Mill, Deer, and Butte creeks are probably the best trend indicators for the Central Valley spring-run Chinook salmon ESU as a whole because these streams contain the primary independent populations within the ESU. Generally, these streams have shown a positive escapement trend since 1991. Escapement numbers are dominated by Butte Creek returns, which have averaged over 7,000 fish since 1995. During this same period, adult returns on Mill Creek have averaged 778 fish, and 1,463 fish on Deer Creek.

Although trends through the first half of the past decade were generally positive, annual abundance estimates display a high level of fluctuation, and the overall number of Central Valley spring-run Chinook salmon remains well below estimates of historic abundance. The past several years (since 2005) have shown declining abundance numbers in most of the tributaries. Additionally, in 2002 and 2003, mean water temperatures in Butte Creek exceeded 21 C for ten or more days in July (reviewed by Williams 2006). These persistent

Table 4-6
Central Valley Spring-run Chinook salmon Population Estimates from CDFG Grand Tab (February 2011) with Corresponding Cohort Replacement Rates for Years Since 1986

Year	Sacramento River Basin Escapement Run Size[a]	FRFH Population	Tributary Populations	5-Year Moving Average of Tributary Population Estimate	Trib CRR[b]	5-Year Moving Average of Trib CRR	5-Year Moving Average of Basin Population Estimate	Basin CRR	5-Year Moving Average of Basin CRR
1986	25,696	1,433	24,263						
1987	13,888	1,213	12,675						
1988	18,933	6,833	12,100						
1989	12,163	5,078	7,085		0.29			0.47	
1990	7,683	1,893	5,790	12,383	0.46		15,673	0.55	
1991	5,926	4,303	1,623	7,855	0.13		11,719	0.31	
1992	3,044	1,497	1,547	5,629	0.22		9,550	0.25	
1993	6,076	4,672	1,404	3,490	0.24	0.27	6,978	0.79	0.48
1994	6,187	3,641	2,546	2,582	1.57	0.52	5,783	1.04	0.59
1995	15,238	5,414	9,824	3,389	6.35	1.70	7,294	5.01	1.48
1996	9,083	6,381	2,702	3,605	1.92	2.06	7,926	1.49	1.72
1997	5,193	3,653	1,540	3,603	0.60	2.14	8,355	0.84	1.84
1998	31,649	6,746	24,903	8,303	2.53	2.60	13,470	2.08	2.09
1999	10,100	3,731	6,369	9,068	2.36	2.75	14,253	1.11	2.11
2000	9,244	3,657	5,587	8,220	3.63	2.21	13,054	1.78	1.46
2001	17,598	4,135	13,463	10,372	0.54	1.93	14,757	0.56	1.27
2002	17,419	4,189	13,230	12,710	2.08	2.23	17,202	1.72	1.45
2003	17,691	8,662	9,029	9,536	1.62	2.04	14,410	1.91	1.42
2004	13,982	4,212	9,770	10,216	0.73	1.72	15,187	0.79	1.35
2005	16,126	1,774	14,352	11,969	1.08	1.21	16,563	0.93	1.18
2006	10,948	2,181	8,767	11,030	0.97	1.29	15,233	0.62	1.20
2007	9,974	2,674	7,300	9,844	0.75	1.03	13,744	0.71	0.99
2008	6,420	1,624	4,796	8,997	0.33	0.77	11,490	0.40	0.69
2009	3,801	989	2,812	7,605	0.32	0.69	9,454	0.35	0.60
2010	3,792	1,661	2,131	5,161	0.29	0.53	6,987	0.38	0.49
2011	4,967	1,900	3,067	4,021	0.64	0.47	5,791	0.77	0.52
Median	10,037	3,655	6,727	8,262	0.73	1.70	12,386	0.79	1.27
Mean	11,647	3,621	8,026	7,708	1.29	1.48	11,585	1.08	1.21

[a] NMFS included both the escapement numbers from the Feather River Fish Hatchery (FRFH) and the Sacramento River and its tributaries in this table. Sacramento River Basin run size is the sum of the escapement numbers from the FRFH and the tributaries.

[b] Abbreviations: CRR = Cohort Replacement Rate, Trib = tributary

Source: NMFS, March 8, 2012, p. 22.

high water temperatures, coupled with high fish densities, precipitated an outbreak of Columnaris Disease (*Flexibacter columnaris*) and Ichthyophthiriasis (*Ichthyophthirius multifiis*) in the adult spring-run Chinook salmon over-summering in Butte Creek. In 2002, this contributed to the pre-spawning mortality of approximately 20 to 30 percent of the adults. In 2003, approximately 65 percent of the adults succumbed, resulting in a loss of an estimated 11,231 adult spring-run Chinook salmon in Butte Creek.

Lindley *et al.* (2007) indicated that the spring-run population of Chinook salmon in the Central Valley had a low risk of extinction in Butte and Deer creeks, according to their population viability analysis (PVA) model and the other population viability criteria (*i.e.*, population size, population decline, catastrophic events, and hatchery influence). The Mill Creek population of spring-run Chinook salmon is at moderate extinction risk according to the PVA model, but appears to satisfy the other viability criteria for low-risk status. However, like the winter-run Chinook salmon population, the Central Valley spring-run Chinook salmon population fails to meet the "representation and redundancy rule" since there is only one demonstrably viable population out of the three diversity groups that historically contained them. The spring-run population is only represented by the group that currently occurs in the northern Sierra Nevada.

The spring-run Chinook salmon populations that formerly occurred in the basalt and porous-lava region and southern Sierra Nevada region have been extirpated. The northwestern California region contains a few ephemeral populations (*e.g.*, Clear, Cottonwood, and Thomes creeks) of spring-run Chinook salmon that are likely dependent on the Northern Sierra populations for their continued existence. Over the long term, these remaining populations are considered to be vulnerable to catastrophic events, such as volcanic eruptions from Mount Lassen or large forest fires due to the close proximity of their headwaters to each other. Drought is also considered to pose a significant threat to the viability of the spring-run Chinook salmon populations in these three watersheds due to their close proximity to each other. One large event could eliminate all three populations.

Viable Salmonid Population Summary for Central Valley spring-run Chinook salmon

Abundance

Over the first half of the past decade, the Central Valley spring-run Chinook salmon ESU has experienced a trend of increasing abundance in some natural populations, most dramatically in the Butte Creek population (Good *et al.* 2005). There has been more opportunistic utilization of migration-dependent streams overall. The FRH spring-run Chinook salmon stock has been included in the ESU based on its genetic linkage to the natural population and the potential development of a conservation strategy for the hatchery program. In contrast to the first half of the decade, the last 5 years of adult returns indicate that population abundance is declining from the peaks seen in the 5 years prior (2001 to 2005) for the entire Sacramento River basin. The recent declines in abundance place the Mill and

Deer creek populations in the high extinction risk category due to the rate of decline, and in the case of Deer Creek, also the level of escapement. Butte Creek has sufficient abundance to retain its low extinction risk classification, but the rate of population decline in the past several years is nearly sufficient to classify it as a high extinction risk based on this criteria. Some tributaries, such as Clear Creek and Battle Creek have seen population gains, but the overall abundance numbers are still low.

Productivity

The 5-year geometric mean for the extant Butte, Deer, and Mill Creek spring-run Chinook salmon populations ranges from 491 to 4,513 fish (Good *et al.* 2005), indicating increasing productivity over the short-term and was projected to likely continue into the future (Good *et al.* 2005). However, as mentioned in the previous paragraph, the last 5 years of adult escapement to these tributaries has seen a cumulative decline in fish numbers and the cohort replacement rate (CRR) has declined in concert with the population declines. The productivity of the Feather River and Yuba River populations and contribution to the Central Valley spring-run ESU currently is unknown.

Spatial Structure

Spring-run Chinook salmon presence has been reported more frequently in several upper Central Valley creeks, but the sustainability of these runs is unknown. Butte Creek spring-run Chinook salmon cohorts have recently utilized all currently available habitat in the creek; and it is unknown if individuals have opportunistically migrated to other systems. The spatial structure of the spring-run Chinook salmon ESU has been reduced with the extirpation of all San Joaquin River basin spring-run Chinook salmon populations. In the near future, an experimental population of Central Valley spring-run Chinook salmon will likely be reintroduced into the San Joaquin River below Friant Dam as part of the San Joaquin River Settlement Agreement if NMFS finds that a permit can be issued to do so. Its long term contribution to the Central Valley spring-run Chinook salmon ESU is uncertain. The populations in Clear Creek and Battle Creek may add to the spatial structure of the Central Valley spring-run population if they can persist by colonizing waterways in the Basalt and Porous and Northwestern California Coastal Range diversity group areas.

Diversity

The Central Valley spring-run Chinook salmon ESU includes two genetic complexes. Analyses of natural and hatchery spring-run Chinook salmon stocks in the Central Valley indicates that the Northern Sierra Nevada spring-run Chinook salmon population complex (Mill, Deer, and Butte creeks) retains genetic integrity. The genetic integrity of the Northern Sierra Nevada spring-run Chinook salmon population complex in the Feather River has been somewhat compromised. The Feather River spring-run Chinook salmon have introgressed with the fall-run Chinook salmon, and it appears that the Yuba River population may have been impacted by FRH fish straying into the Yuba River. Additionally, the diversity of the

spring-run Chinook salmon ESU has been further reduced with the loss of the San Joaquin River basin spring-run Chinook salmon populations.

Threatened Central Valley spring-run Chinook Salmon and Threatened Central Valley Steelhead Critical Habitat

Critical habitat was designated for Central Valley spring-run Chinook salmon and California Central Valley steelhead on September 2, 2005, (70 FR 52488). Critical habitat for Central Valley spring-run Chinook salmon includes stream reaches such as those of the Feather and Yuba rivers, Big Chico, Butte, Deer, Mill, Battle, Antelope, and Clear creeks, the Sacramento River, as well as portions of the northern Delta. Critical habitat for California Central Valley steelhead 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 San Joaquin River, including its tributaries, and the waterways of the Delta. Critical habitat includes the stream channels in the designated stream reaches and the lateral extent as defined by the ordinary high-water line. In areas where the ordinary high-water line has not been defined, the lateral extent will be defined by the bankfull elevation (defined as the level at which water begins to leave the channel and move into the floodplain; it is reached at a discharge that generally has a recurrence interval of 1 to 2 years on the annual flood series) (Bain and Stevenson 1999; 70 FR 52488). Critical habitat for Central Valley spring-run Chinook salmon and steelhead is defined as specific areas that contain the primary constituent elements (PCE) and physical habitat elements essential to the conservation of the species. Following are the inland habitat types used as PCEs for Central Valley spring-run Chinook salmon and California Central Valley steelhead, and as physical habitat elements for Sacramento River winter-run Chinook salmon.

PCEs for Sacramento River winter-run Chinook salmon, Central Valley spring-run Chinook salmon and California Central Valley steelhead include:

Spawning Habitat

Freshwater spawning sites are those with water quantity and quality conditions and substrate supporting spawning, incubation, and larval development. Most spawning habitat in the Central Valley for Chinook salmon and steelhead is located in areas directly downstream of dams containing suitable environmental conditions for spawning and incubation. Spawning habitat for Sacramento River winter-run Chinook salmon is restricted to the Sacramento River primarily between RBDD and Keswick Dam. Central Valley spring-run Chinook salmon also spawn on the mainstem Sacramento River between RBDD and Keswick Dam and in tributaries such as Mill, Deer, and Butte creeks (however, little spawning activity has been recorded in recent years on the Sacramento River mainstem for spring-run Chinook salmon). Spawning habitat for California Central Valley steelhead is similar in nature to the requirements of Chinook salmon, primarily occurring in reaches directly below dams (*i.e.*, above RBDD on the Sacramento River) on perennial watersheds throughout the

Central Valley. These reaches can be subjected to variations in flows and temperatures, particularly over the summer months, which can have adverse effects upon salmonids spawning below them. Even in degraded reaches, spawning habitat has a high conservation value as its function directly affects the spawning success and reproductive potential of listed salmonids.

Freshwater Rearing Habitat

Freshwater rearing sites are those with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; water quality and forage supporting juvenile development; and natural cover such as shade, submerged and overhanging large woody material, log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks. 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. Rearing habitat condition is strongly affected by habitat complexity, food supply, and the presence of predators of juvenile salmonids. Some complex, productive habitats with floodplains remain in the system (*e.g.*, the lower Cosumnes River, Sacramento River reaches with setback levees [*i.e.*, primarily located upstream of the City of Colusa]) and flood bypasses (*i.e.*, Yolo and Sutter bypasses). However, the channelized, leveed, and riprapped river reaches and sloughs that are common in the Sacramento-San Joaquin system typically have low habitat complexity, low abundance of food organisms, and offer little protection from either fish or avian predators. Freshwater rearing habitat also has a high conservation value even if the current conditions are significantly degraded from their natural state. Juvenile life stages of salmonids are dependent on the function of this habitat for successful survival and recruitment.

Freshwater Migration Corridors

Ideal freshwater migration corridors are free of migratory obstructions, with water quantity and quality conditions that enhance migratory movements. They contain natural cover such as riparian canopy structure, submerged and overhanging large woody objects, aquatic vegetation, large rocks, and boulders, side channels, and undercut banks which augment juvenile and adult mobility, survival, and food supply. Migratory corridors are downstream of the spawning areas and include the lower mainstems of the Sacramento and San Joaquin rivers and the Delta. These corridors allow the upstream passage of adults, and the downstream emigration of outmigrant juveniles. Migratory habitat condition is strongly affected by the presence of barriers, which can include dams (*i.e.*, hydropower, flood control, and irrigation flashboard dams), unscreened or poorly screened diversions, degraded water quality, or behavioral impediments to migration. For successful survival and recruitment of salmonids, freshwater migration corridors must function sufficiently to provide adequate passage. For this reason, freshwater migration corridors are considered to have a high conservation value even if the migration corridors are

significantly degraded compared to their natural state.

Estuarine Areas

Estuarine areas free of migratory obstructions with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh and salt water are included as a PCE. Natural cover such as submerged and overhanging large woody material, aquatic vegetation, and side channels, are suitable for juvenile and adult foraging. Estuarine areas are considered to have a high conservation value as they provide factors which function to provide predator avoidance and as a transitional zone to the ocean environment.

Source: www.fs.fed.us.

Photo: Central Valley steelhead.

3. Threatened Central Valley steelhead *(Oncorhynchus mykiss)*

Central Valley steelhead *(Oncorhynchus mykiss),* which are the anadromous form of rainbow trout, were federally listed threatened on March 19, 1998. The DPS consists of steelhead populations in the Sacramento and San Joaquin River basins. Steelhead populations from the Coleman National Fish Hatchery and Feather River Hatchery were included in the DPS in January 2006. The threatened status of Central Valley steelhead was confirmed in 2005 (NMFS 2005), and again in August 2011 (NMFS August 2011c). In August 2011, NMFS noted that the biological status of Central Valley steelhead has worsened since 2005, and recommended that its status be reassessed in two to three years. NMFS noted that the original threats due to loss and degradation of habitat remained, and that hatcheries, drought, poor ocean conditions, and climate change posed additional threats to the species. Critical habitat was designated on September 2, 2005, and is described above. Critical habitat includes potential WHCP treatment sites. NMFS developed a draft recovery plan for Central Valley steelhead in 2009, which has not yet been finalized.

Central Valley steelhead migrate to the ocean as juveniles and return to fresh water to spawn when they are 2 to 4 years old. Spawning migration (through the Delta) can be anytime from August through March. Steelhead usually do not die after spawning. Survivors return to the ocean between April and June, and some make several more spawning migrations. Juvenile steelhead usually remain in fresh water for the first year, then migrate to the ocean between November and May. Steelhead are found in the Delta predominantly during migration.

Steelhead are primarily threatened by loss of the vast majority of historical spawning habitats above impassable dams, and mixing with hatchery fish (NMFS 2005). California began implementing measures to protect steelhead in 1998, including 100 percent marking of all hatchery steelhead, zero bag limits for unmarked

steelhead, gear restrictions, closures, and designation of size limits to protect smolts (NMFS 2007). The text below describing the status of Central Valley steelhead is drawn from the March 8, 2012 Biological Opinion of the South Delta Temporary Barriers Program (NMFS March 2012).

Steelhead can be divided into two life history types, summer-run steelhead and winter-run steelhead, based on their state of sexual maturity at the time of river entry and the duration of their spawning migration, stream-maturing and ocean-maturing. Only winter-run steelhead currently are found in Central Valley rivers and streams (McEwan and Jackson 1996), although there are indications that summer-run steelhead were present in the Sacramento river system prior to the commencement of large-scale dam construction in the 1940s [Interagency Ecological Program (IEP) Steelhead Project Work Team 1999]. At present, summer-run steelhead are found only in North Coast drainages, mostly in tributaries of the Eel, Klamath, and Trinity River systems (McEwan and Jackson 1996).

California Central Valley steelhead generally leave the ocean from August through April (Busby *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 *et al.* 1961, McEwan and Jackson 1996. **Table 4-7,** on the next page, summarizes Central Valley steelhead location and timing. Timing of upstream migration is correlated with higher flow events, such as freshets or sand bar breaches at river mouths, and associated lower water temperatures. Unlike Pacific salmon, steelhead are iteroparous, or capable of spawning more than once before death (Barnhart *et al.* 1986, Busby *et al.* 1996). However, it is rare for steelhead to spawn more than twice before dying; most that do so are females (Busby *et al.* 1996). Iteroparity is more common among southern steelhead populations than northern populations (Busby *et al.* 1996). Although one-time spawners are the great majority, Shapovalov and Taft (1954) reported that repeat spawners are relatively numerous (17.2 percent) in California streams.

Spawning occurs during winter and spring months. The length of time it takes for eggs to hatch depends mostly on water temperature. Hatching of steelhead eggs in hatcheries takes about 30 days at 51 F. Fry emerge from the gravel usually about 4 to 6 weeks after hatching, but factors such as redd depth, gravel size, siltation, and temperature can speed or retard this time (Shapovalov and Taft 1954). Newly emerged fry move to the shallow, protected areas associated with the stream margin (McEwan and Jackson 1996) and they soon move to other areas of the stream and establish feeding locations, which they defend (Shapovalov and Taft 1954).

Steelhead rearing during the summer takes place primarily in higher velocity areas in pools, although young-of-year also are abundant in glides and riffles. Productive steelhead habitat is characterized by complexity, primarily in the form of large and small woody debris. Cover is an important habitat component for juvenile steelhead both as velocity refugia and as a means of avoiding predation (Meehan and Bjornn 1991).

Table 4-7
The Temporal Occurrence of Adult (a) and Juvenile (b) California Central Valley steelhead in the Central Valley
(Darker shades indicate months of greatest relative abundance)

(a) Adult migration/holding

Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
[1,3]Sac. River												
[2,3]Sac R at Red Bluff												
[4]Mill, Deer Creeks												
[6]Sac. R. at Fremont Weir												
[6]Sac. R. at Fremont Weir												
[7]San Joaquin River												

(b) Juvenile migration

Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
[1,2]Sacramento River												
[2,8]Sac. R at Knights Landing												
[9]Sac. River @ Knights Landing												
[10]Chipps Island (wild)												
[8]Mossdale												
[11]Woodbridge Dam												
[12]Stan R. at Caswell												
[13]Sac R. at Hood												

Relative Abundance: ■ = High ■ = Medium □ = Low

Sources: [1]Hallock 1961; [2]McEwan 2001; [3]USFWS unpublished data; [4]CDFG 1995; [5]Hallock *et al.* 1957; [6]Bailey 1954; [7]CDFG Steelhead Report Card Data; [8]CDFG unpublished data; [9]Snider and Titus 2000; [10]Nobriga and Cadrett 2003; [11]Jones & Stokes Associates, Inc., 2002; [12]S.P. Cramer and Associates, Inc. 2000 and 2001; [13]Schaffter 1980, 1997.

Source: NMFS, March 8, 2012, p. 25.

Juvenile steelhead emigrate episodically from natal streams during fall, winter, and spring high flows. Emigrating California Central Valley steelhead use the lower reaches of the Sacramento River and the Delta for rearing and as a migration corridor to the ocean. Juvenile California Central Valley steelhead feed mostly on drifting aquatic organisms and terrestrial insects and will also take active bottom invertebrates (Moyle 2002). Some may utilize 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 *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. Nobriga

and Cadrett (2003) also have verified these temporal findings based on analysis of captures at Chipps Island.

Historic California Central Valley steelhead run sizes are difficult to estimate given the paucity of data, but may have approached 1 to 2 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 through the 1960s in the Sacramento River, upstream of the Feather River. Steelhead counts at the RBDD 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, with an estimated total annual run size for the entire Sacramento-San Joaquin system, based on RBDD counts, to be no more than 10,000 adults (McEwan and Jackson 1996, McEwan 2001). Steelhead escapement surveys at RBDD ended in 1993 due to changes in dam operations. Nobriga and Cadrett (2003) compared CWT and untagged (wild) steelhead smolt catch ratios at Chipps Island trawl from 1998 through 2001 to estimate that about 100,000 to 300,000 steelhead juveniles are produced naturally each year in the Central Valley. In the *Updated Status Review of West Coast Salmon and Steelhead* (Good *et al.* 2005), the Biological Review Team (BRT) made the following conclusion based on the Chipps Island data:

> *"If we make the fairly generous assumptions (in the sense of generating large estimates of spawners) that average fecundity is 5,000 eggs per female, 1 percent of eggs survive to reach Chipps Island, and 181,000 smolts are produced (the 1998-2000 average), about 3,628 female steelhead spawn naturally in the entire Central Valley. This can be compared with McEwan's (2001) estimate of 1 million to 2 million spawners before 1850, and 40,000 spawners in the 1960s".*

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 and a few wild steelhead are produced in the American and Feather rivers (McEwan and Jackson 1996). Recent snorkel surveys (1999 to 2002) indicate that steelhead are present in Clear Creek (J. Newton, USFWS, pers. comm. 2002, as reported in Good *et al*. 2005). Because of the large resident *O. mykiss* population in Clear Creek, steelhead spawner abundance has not been estimated.

Until recently, California Central Valley steelhead were thought to be extirpated from the San Joaquin River system. Recent monitoring has detected small self-sustaining populations of steelhead in the Stanislaus, Mokelumne, and Calaveras rivers, and other streams previously thought to be devoid of steelhead (McEwan 2001). On the Stanislaus River, steelhead smolts have been captured in rotary screw traps at Caswell State Park and Oakdale each year since 1995 (S.P. Cramer and Associates Inc. 2000, 2001). Zimmerman *et al.* (2008) has documented Central Valley steelhead in the Stanislaus, Tuolumne and Merced rivers based on otolith (inner ear) microchemistry.

It is possible that naturally-spawning populations exist in many other streams but are undetected due to lack of monitoring programs (IEP Steelhead Project Work Team 1999). Incidental catches and observations of steelhead juveniles also have occurred on the Tuolumne and Merced Rivers during fall-run Chinook salmon monitoring activities, indicating that steelhead are widespread, throughout accessible streams and rivers in the Central Valley (Good *et al*. 2005). California Department of Fish and Game (CDFG) staff have prepared catch summaries for juvenile migrant California Central Valley steelhead on the San Joaquin River near Mossdale which represents migrants from the Stanislaus, Tuolumne, and Merced rivers. Based on trawl recoveries at Mossdale between 1988 and 2002, as well as rotary screw trap efforts in all three tributaries, CDFG staff stated that it is "clear from this data that rainbow trout do occur in all the tributaries as migrants and that the vast majority of them occur on the Stanislaus River" (Letter from Dean Marston, CDFG, to Michael Aceituno, NMFS, 2004). The documented returns on the order of single fish in these tributaries suggest that existing populations of California Central Valley steelhead on the Tuolumne, Merced, and lower San Joaquin rivers are severely depressed.

Lindley *et al*. (2006) indicated that prior population census estimates completed in the 1990s found the California Central Valley steelhead spawning population above RBDD had a fairly strong negative population growth rate and small population size. Good *et al.* (2005) indicated the decline was continuing as evidenced by new information (Chipps Island trawl data). California Central Valley steelhead populations generally show a continuing decline, an overall low abundance, and fluctuating return rates. The future of California Central Valley steelhead is uncertain due to limited data concerning their status. However, Lindley *et al.* (2007), citing evidence presented by Yoshiyama *et al.* (1996); McEwan (2001); and Lindley *et al.* (2006), concluded that there is sufficient evidence to suggest that the DPS is at moderate to high risk of extinction.

Viable Salmonid Population Summary for Central Valley steelhead

Abundance

All indications are that natural California Central Valley steelhead have continued to decrease in abundance and in the proportion of natural fish over the past 25 years (Good *et al.* 2005); the long-term trend remains negative. There has been little steelhead population monitoring, despite 100 percent marking of hatchery steelhead since 1998. Hatchery production and returns are dominant over natural fish and include significant numbers of non-DPS-origin Eel River steelhead stock. Continued decline in the ratio between wild juvenile steelhead to hatchery juvenile steelhead in fish monitoring efforts indicates that the wild population abundance is declining. Hatchery releases (100 percent adipose fin clipped fish since 1998) have remained relatively constant over the past decade, yet the proportion of ad-clipped fish to wild adipose fin bearing fish has steadily increased over the past several years.

Productivity

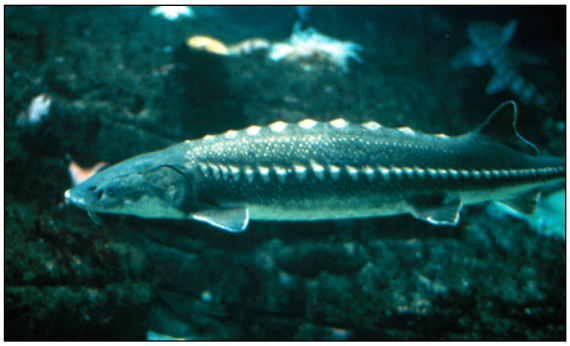
Approximatley 100,000 to 300,000 natural juvenile steelhead are estimated to leave the Central Valley annually, based on rough calculations from sporadic catches in trawl gear (Good *et al.* 2005). Concurrently, one million in-DPS hatchery steelhead smolts and another half million out-of-DPS hatchery steelhead smolts are released annually in the Central Valley. The estimated ratio of nonclipped to clipped steelhead has decreased from 0.3 percent to less than 0.1 percent, with a net decrease to one-third of wild female spawners from 1998 to 2000 (Good *et al.* 2005). Recent data from the Chipps Island fish monitoring trawls indicates that in recent years over 90 percent of captured steelhead smolts have been of hatchery origin. In 2010, the data indicated hatchery fish made up 95 percent of the catch.

Spatial Structure

Steelhead appear to be well-distributed where found throughout the Central Valley (Good *et al.* 2005). Until recently, there was very little documented evidence of steelhead due to the lack of monitoring efforts. Since 2000, steelhead have been confirmed in the Stanislaus and Calaveras rivers. The efforts to provide passage of salmonids over impassable dams may increase the spatial diversity of California Central Valley steelhead populations if the passage programs are implemented for steelhead.

Diversity

Analysis of natural and hatchery steelhead stocks in the Central Valley reveal genetic structure remaining in the DPS (Nielsen *et al.* 2003). There appears to be a great amount of gene flow among upper Sacramento River basin stocks, due to the post-dam, lower basin distribution of steelhead and management of stocks. Recent reductions in natural population sizes have created genetic bottlenecks in several California Central Valley steelhead stocks (Good *et al.* 2005; Nielsen *et al.* 2003). The out-of-basin steelhead stocks of the Nimbus and Mokelumne River hatcheries are not included in the California Central Valley steelhead DPS.



Source: www.delta.dfg.ca.gov.

Photo: Green Sturgeon.

4. Threatened Southern Distinct Population Segment (DPS) of North American green sturgeon *(Acipenser medirostris)*

Green sturgeon *(Acipenser meditrostris)* southern population (south of the Eel River), found in San Francisco Bay and the Delta, and spawning in the Sacramento River, was designated as a federal threatened species by NMFS in July 2006. Critical habitat was designated in October 2009. Take prohibitions were established in June 2010. The Southern DPS is separate from green sturgeon found at the Eel River and north to

British Columbia (NMFS February 2005). The green sturgeon is also listed as a California species of concern by CDFG. Relatively little is known about the biology and population characteristics of green sturgeon. There are many studies currently underway by a number of universities and state and federal agencies to better understand the distribution, migration, spawning habitat utilization, and population genetics of green sturgeon (NMFS 2011).

Green sturgeon appear to inhabit the Delta during their second and third years, although adult sturgeon migrate through the Delta to spawning grounds on the Upper Sacramento River between mid-February and May. Adults spend most of their time in the ocean, and may migrate as far north as British Columbia. Adults spawn every three to five years.

In June 2011, NMFS and DFG biologists rescued over 200 fish that had been trapped in bypass channels of the Sacramento River by high water (NMFS June 2011). There were 25 green sturgeon among the rescued fish, including one with a total length of over seven feet, a girth of 36 inches, and weight of at least 250 pounds. The rescued green sturgeon were implanted with tracking devices to help gain more information on sturgeon movement.

The text below describing the green sturgeon in more detail is drawn from the March 8, 2012 Biological Opinion of the South Delta Temporary Barriers Program (NMFS March 2012).

In North America, spawning populations of green sturgeon are currently found in only three river systems: the Sacramento and Klamath rivers in California and the Rogue River in southern Oregon. Green sturgeon are known to range from Baja California to the Bering Sea along the North American continental shelf. Data from commercial trawl fisheries and tagging studies indicate that the green sturgeon occupy waters within the 110 meter contour (Erickson and Hightower 2007). During the late summer and early fall, subadults and nonspawning adult green sturgeon frequently can be found aggregating in estuaries along the Pacific coast (Emmett *et al.* 1991, Moser and Lindley 2007). Particularly large concentrations of green sturgeon from both the northern and southern populations occur in the Columbia River estuary, Willapa Bay, Grays Harbor and Winchester Bay, with smaller aggregations in Humboldt Bay, Tillamook Bay, Nehalem Bay, and San Francisco and San Pablo Bays (Emmett *et al* 1991, Moyle *et al.* 1992, and Beamesderfer *et al.* 2007). Lindley *et al.* (2008) reported that green sturgeon make seasonal migratory movements along the west coast of North America, overwintering north of Vancouver Island and south of Cape Spencer, Alaska.

Individual fish from the Southern DPS of green sturgeon have been detected in these seasonal aggregations. Information regarding the migration and habitat use of the Southern DPS of green sturgeon has recently emerged. Lindley (2006) presented preliminary results of large-scale green sturgeon migration studies, and verified past population structure delineations based on genetic work and found frequent large-scale migrations of green sturgeon along the Pacific Coast. This work was further expanded by recent tagging studies of green sturgeon conducted by Erickson and

Hightower (2007) and Lindley *et al.* (2008). To date, the data indicates that North American green sturgeon are migrating considerable distances up the Pacific Coast into other estuaries, particularly the Columbia River estuary. This information also agrees with the results of previous green sturgeon tagging studies (CDFG 2002), where CDFG tagged a total of 233 green sturgeon in the San Pablo Bay estuary between 1954 and 2001. A total of 17 tagged fish were recovered: 3 in the Sacramento-San Joaquin Estuary, 2 in the Pacific Ocean off of California, and 12 from commercial fisheries off of the Oregon and Washington coasts. Eight of the 12 recoveries were in the Columbia River estuary (CDFG 2002).

The Southern DPS of green sturgeon includes all green sturgeon populations south of the Eel River, with the only known spawning population being in the Sacramento River. Green sturgeon life history can be broken down into four main stages: eggs and larvae, juveniles, sub-adults, and sexually mature adults. Sexually mature adults are those fish that have fully developed gonads and are capable of spawning. Female green sturgeon are typically 13 to 27 years old when sexually mature and have a total body length (TL) ranging between 145 and 205 cm at sexual maturity (Nakamoto *et al. 1995*, Van Eenennaam *et al.* 2006). Male green sturgeon become sexually mature at a younger age and smaller size than females. Typically, male green sturgeon reach sexual maturity between 8 and 18 years of age and have a TL ranging between 120 cm to 185 cm (Nakamoto *et al. 1995*, Van Eenennaam *et al.* 2006). The variation in the size and age of fish upon reaching sexual maturity is a reflection of their growth and nutritional history, genetics, and the environmental conditions they were exposed to during their early growth years. Adult green sturgeon are believed to feed primarily upon benthic invertebrates such as clams, mysid shrimp, grass shrimp, and amphipods (Radtke 1966). Adult sturgeon caught in Washington state waters were found to have fed on Pacific sand lance (*Ammodytes hexapterus*) and callianassid shrimp (Moyle *et al.* 1992). It is unknown what forage species are consumed by adults in the Sacramento River upstream of the Delta.

Adult green sturgeon are gonochoristic (sex genetically fixed), oviparous (egg laying) and iteroparous (bare repeat offspring). They are believed to spawn every 2 to 5 years (Beamesderfer *et al.* 2007). Upon maturation of their gonadal tissue, but prior to ovulation or spermiation, the sexually mature fish enter freshwater and migrate upriver to their spawning grounds. The remainder of the adult's life is generally spent in the ocean or near-shore environment (bays and estuaries) without venturing upriver into freshwater. Younger females may not spawn the first time they undergo oogenesis and subsequently they reabsorb their gametes without spawning. Adult female green sturgeon produce between 60,000 and 140,000 eggs, depending on body size, with a mean egg diameter of 4.3 mm (Moyle *et al.* 1992, Van Eenennaam *et al.* 2001). They have the largest egg size of any sturgeon, and the volume of yolk ensures an ample supply of energy for the developing embryo. The outside of the eggs are adhesive, and are more dense than those of white sturgeon (Kynard *et al.* 2005, Van Eenennaam *et al.* 2009).

Adults begin their upstream spawning migrations into the Sacramento River in late February with spawning occurring between March and July (CDFG 2002. Heublin 2006, Heublin *et al.* 2009, Vogel 2008). Peak spawning is believed to occur between April and June in deep, turbulent, mainstem channels over large cobble and rocky substrates with crevices and interstices. Females broadcast spawn their eggs over this substrate, while the male releases its milt (sperm) into the water column. Fertilization occurs externally in the water column and the fertilized eggs sink into the interstices of the substrate where they develop further (Kynard *et al.* 2005, Heublin *et al.* 2009). Known historic and current spawning occurs in the Sacramento River (Adams *et al.* 2002, Beamesderfer *et al.* 2004, Adams *et al.* 2007). Currently, Keswick and Shasta dams on the mainstem of the Sacramento River block passage to the upper river. Although no historical accounts exist for identified green sturgeon spawning occuring above the current dam sites, suitable spawning habitat existed and the geographic extent of spawning has been reduced due to the impassable barriers constructed on the river.

Spawning on the Feather River is suspected to have occurred in the past due to the continued presence of adult green sturgeon in the river below Oroville Dam. This continued presence of adults below the dam suggests that fish are trying to migrate to upstream spawning areas now blocked by the dam, which was constructed in 1968. In 2011, fertilized green sturgeon eggs were recovered during monitoring activities by DWR on the Feather River and several adult green sturgeon were recorded on video congregating below Daguerre Dam on the Yuba River. Spawning in the San Joaquin River system has not been recorded historically or observed recently.

Kelly *et al.* (2007) indicated that green sturgeon enter the San Francisco Estuary during the spring and remain until autumn. **Table 4-8,** on the next page, summarizes green sturgeon location and timing. The authors studied the movement of adults in the San Francisco Estuary and found them to make significant long-distance movements with distinct directionality. The movements were not found to be related to salinity, current, or temperature, and Kelly *et al.* (2007) surmised that they are related to resource availability and foraging behavior. Recent acoustical tagging studies on the Rogue River (Erickson *et al.* 2002) have shown that adult green sturgeon will hold for as much as 6 months in deep (> 5m), low gradient reaches or off channel sloughs or coves of the river during summer months when water temperatures were between 15 C and 23 C. When ambient temperatures in the river dropped in autumn and early winter (<10 C) and flows increased, fish moved downstream and into the ocean. Erickson *et al.* (2002) surmised that this holding in deep pools was to conserve energy and utilize abundant food resources. Benson *et al.* (2007) found similar behavior on the Klamath and Trinity River systems with adult sturgeon acoustically tagged during their spawning migrations. Most fish held over the summer in discrete locations characterized by deep, low velocity pools until late fall or early winter when river flows increased with the first storms of the rainy season. Fish then moved rapidly downstream and out of the system.

Table 4-8

The Temporal Occurrence of (a) Adult, (b) Larva, (c) Juvenile, and (d) Subadult Coastal Migrant Southern DPS of green sturgeon. (Locations emphasize the Central Valley of California)
(Darker shades indicate months of greatest relative abundance)

(a) Adult-sexually mature (≥145 – 205 cm TL for females and ≥ 120 – 185 cm TL old for males

Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Upper Sac. River[a,b,c,i]												
SF Bay Estuary[d,h,i]												

(b) Larval and juvenile (≤ 10 months old)

Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
RBDD, Sac River[c]												
GCID, Sac River[c]												

(c) Older Juvenile (> 10 months old and ≤ 3 years old)

Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
South Delta*[f]												
Sac-SJ Delta[f]												
Sac-SJ Delta[e]												
Suisun Bay[e]												

(d) Sub-Adult/non-sexually mature (approx. 75 cm to 145 cm for females and 75 to 120 cm for males)

Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Pacific Coast[c,g]												

Relative Abundance: ■ = High ■ = Medium □ = Low

* Fish Facility salvage operations

Sources: [a]USFWS (2002); [b]Moyle *et al.* (1992); [c]Adams *et al.* (2002) and NMFS (2005a); [d]Kelly *et al.* (2007); [e]CDFG (2002); [f]IEP Relational Database, fall midwater trawl green sturgeon captures from 1969 to 2003; [g]Nakamoto *et al.* (1995); [h]Heublein (2006); [i]CDFG Draft Sturgeon Report Card (2007)

Source: NMFS, March 8, 2012, p. 33.

Recent data gathered from acoustically tagged adult green sturgeon revealed comparable behavior by adult fish on the Sacramento River based on the positioning of adult green sturgeon in holding pools on the Sacramento River above the Glenn Colusa Irrigation District (GCID) diversion (river mile (RM) 205). Studies by Heublin (2006, 2009) and Vogel (2008) have documented the presence of adults in the Sacramento River during the spring and through the fall into the early winter months. These fish hold in upstream locations prior to their emigration from the system later in the year. Like the Rogue and Klamath river systems, downstream migration appears to be triggered by increased flows, decreasing water temperatures, and occurs rapidly once initiated. It should also be noted that some adults rapidly leave the system following

their suspected spawning activity and enter the ocean only in early summer (Heublin 2006). This behavior has also been observed on the other spawning rivers (Benson *et al.* 2007*)* but may have been an artifact of the stress of the tagging procedure in that study.

Eggs and Larvae

Currently spawning appears to occur primarily above Red Bluff Diversion Dam (RBDD), based on the recovery of eggs and larvae at the dam in monitoring studies (Gaines and Martin 2002, Brown 2007). Green sturgeon larvae hatch from fertilized eggs after approximately 169 hours at a water temperature of 59 F (Van Eenennaam *et al.* 2001, Deng *et al.* 2002), which is similar to the sympatric white sturgeon development rate (176 hours). Studies conducted at the University of California, Davis by Van Eenennaam *et al.* (2005) indicated that an optimum range of water temperature for egg development ranged between 57.2 F and 62.6 F. Temperatures over 23 C (73.4 F) resulted in 100 percent mortality of fertilized eggs before hatching. Eggs incubated at water temperatures between 63.5 F and 71.6 F resulted in elevated mortalities and an increased occurrence of morphological abnormalities in those eggs that did hatch. At incubation temperatures below 57.2 F, hatching mortality also increased significantly, and morphological abnormalities increased slightly, but not statistically so.

Newly hatched green sturgeon are approximately 12.5mm to 14.5 mm in length and have a large ovoid yolk sac that supplies nutritional energy until exogenous feeding occurs. These yolksac larvae are less developed in their morphology than older juveniles and external morphology resembles a "tadpole" with a continuous fin fold on both the dorsal and ventral sides of the caudal trunk. At 10 days of age, the yolk sac has become greatly reduced in size and the larvae initiates exogenous feeding through a functional mouth. The fin folds have become more developed and formation of fin rays begins to occur in all fin tissues. By 45 days of age, the green sturgeon larvae have completed their metamorphosis, which is characterized by the development of dorsal, lateral, and ventral scutes, elongation of the barbels, rostrum, and caudal peduncle, reabsorption of the caudal and ventral fin folds, and the development of fin rays.

The juvenile fish resembles the adult form, including the dark olive coloring, with a dark mid-ventral stripe (Deng *et al.* 2002) and are approximately 75 mm TL. At this stage of development, the fish are considered juveniles and are no longer larvae. Juvenile fish continue to exhibit nocturnal behavioral beyond the metamorphosis from larvae to juvenile stages. Kynard *et al.*'s (2005) laboratory studies indicated that juvenile fish continued to migrate downstream at night for the first 6 months of life. When ambient water temperatures reached 46.4 F, downstream migrational behavior diminished and holding behavior increased. This data suggests that 9 to 10 month old fish would hold over in their natal rivers during the ensuing winter following hatching, but at a location downstream of their spawning grounds.

Green sturgeon juveniles tested under laboratory conditions had optimal bioenergetics performance (*i.e.* growth, food conversion, swimming ability) between 59 F

and 66.2 F under either full or reduced rations (Mayfield and Cech 2004). This temperature range overlaps the egg incubation temperature range for peak hatching success previously discussed. Ambient water temperature conditions in the Rogue and Klamath River systems range from 39 F to approximately 75.2 F. The Sacramento River has similar temperature profiles, and, like the previous two rivers, is a regulated system with several dams controlling flows on its mainstem (Shasta and Keswick dams), and its tributaries (Whiskeytown, Oroville, Folsom, and Nimbus dams).

Larval and juvenile green sturgeon are subject to predation by both native and introduced fish species. Prickly sculpin (*Cottus asper*) have been shown to be an effective predator on the larvae of sympatric white sturgeon (Gadomski and Parsley 2005). This study also indicated that the lowered turbidity found in tailwater streams and rivers due to dams increased the effectiveness of sculpin predation on sturgeon larvae under laboratory conditions.

Larval and juvenile sturgeons have been caught in traps at two sites in the upper Sacramento River: below the RBDD (RM 243) and from the GCID pumping plant (RM 205) (CDFG 2002). Larvae captured at the RBDD site are typically only a few days to a few weeks old, with lengths ranging from 24 mm to 31 mm. This body length is equivalent to 15 to 28 days post hatch as determined by Deng *et al.* (2002). Recoveries of larvae at the RBDD rotary screw traps (RSTs) occur between late April/early May and late August with the peak of recoveries occurring in June (1995 to 1999 and 2003 to 2008 data). The mean yearly total length of post-larval green sturgeon captured in the GCID rotary screw trap, approximately 30 miles downstream of RBDD, ranged from 33 mm to 44 mm between 1997 and 2005 (CDFG, 2002) indicating they are approximately 3 to 4 weeks old (Van Eenennaam *et al.* 2001, Deng *et al.* 2002). Taken together, the average length of larvae captured at the two monitoring sites indicate that fish were hatched upriver of the monitoring site and drifted downstream over the course of 2 to 4 weeks of growth.

According to the CDFG document commenting on the NMFS proposal to list the southern DPS (CDFG 2002), some green sturgeon rear to larger sizes above RBDD, or move back to this location after spending time downstream. Two sturgeon between 180 mm and 400 mm TL were captured in the rotary-screw trap during 1999 and green sturgeon within this size range have been impinged on diffuser screens associated with a fish ladder at RBDD (K. Brown, USFWS, pers. comm. as cited in CDFG 2002). Juvenile green sturgeon have been salvaged at the Harvey O. Banks Pumping Plant and the John E. Skinner Fish Collection Facility in the south Delta, and captured in trawling studies by CDFG during all months of the year (CDFG 2002). The majority of these fish were between 200 mm and 500 mm, indicating they were from 2 to 3 years of age based on Klamath River age distribution work by Nakamoto *et al.* (1995). The lack of a significant proportion of juveniles smaller than approximately 200 mm in Delta captures indicates that juveniles of the Southern DPS of green sturgeon likely

hold in the mainstem Sacramento River, as suggested by Kynard *et al.* (2005).

Population abundance information concerning the Southern DPS green sturgeon is described in the NMFS status reviews (Adams *et al*. 2002, NMFS 2005a). Limited population abundance information comes from incidental captures of North American green sturgeon from the white sturgeon monitoring program by the CDFG sturgeon tagging program (CDFG 2002). By comparing ratios of white sturgeon to green sturgeon captures, CDFG provides estimates of adult and sub-adult North American green sturgeon abundance. Estimated abundance between 1954 and 2001 ranged from 175 fish to more than 8,000 per year and averaged 1,509 fish per year. Unfortunately, there are many biases and errors associated with these data, and CDFG does not consider these estimates reliable. Fish monitoring efforts at RBDD and GCID on the upper Sacramento River have captured between 0 and 2,068 juvenile North American green sturgeon per year (Adams *et al*. 2002). The only existing information regarding changes in the abundance of the Southern DPS of green sturgeon includes changes in abundance at the John E. Skinner Fish Facility between 1968 and 2001. The average number of North American green sturgeon taken per year at the State Facility prior to 1986 was 732; from 1986 on, the average per year was 47 (70 FR 17386, April 6, 2005). For the Harvey O. Banks Pumping Plant, the average number prior to 1986 was 889; from 1986 to 2001 the average was 32 (70 FR 17386, April 6, 2005). In light of the increased exports, particularly during the previous 10 years, it is clear that the abundance of the Southern DPS green sturgeon is dropping. Additional analysis of North American green and white sturgeon taken at the Fish Facilities indicates that take of both North American green and white sturgeon per acre-foot of water exported has decreased substantially since the 1960s (70 FR 17386, April 6, 2005). No green sturgeon were recovered at either the CVP or SWP in 2010. Catches of subadult and adult North American green sturgeon by the IEP between 1996 and 2004 ranged from 1 to 212 green sturgeon per year (212 occurred in 2001), however, the portion of the Southern DPS of North American green sturgeon is unknown as these captures were primarily located in San Pablo Bay which is known to consist of a mixture of Northern and Southern DPS North American green sturgeon. Recent spawning population estimates using sibling based genetics by Israel (2006b) indicates spawning populations of 32 spawners in 2002, 64 in 2003, 44 in 2004, 92 in 2005, and 124 in 2006 above RBDD (with an average of 71).

As described previously, the majority of spawning by green sturgeon in the Sacramento River system appears to take place above the location of RBDD. This is based on the length and estimated age of larvae captured at RBDD (approximately 2–3 weeks of age) and GCID (downstream, approximately 3–4 weeks of age) indicating that hatching occurred above the sampling location. Note that there are many assumptions with this interpretation (*i.e.*, equal sampling efficiency and distribution of larvae across channels) and this information should be considered cautiously.

Available information on green sturgeon indicates that, as with winter-run Chinook salmon, the mainstem Sacramento River may be the last viable spawning habitat (Good *et al.* 2005) for the Southern DPS of green sturgeon. Lindley *et al.* (2007) pointed out that an ESU represented by a single population at moderate risk is at a high risk of extinction over the long term. Although the extinction risk of the Southern DPS of green sturgeon has not been assessed, NMFS believes that the extinction risk has increased because there is only one known population, that which is spawning within the mainstem Sacramento River.

Population Viability Summary for the Southern DPS of North American green sturgeon

The Southern DPS of North American green sturgeon has not been analyzed to characterize the status and viability as has been done in recent efforts for Central Valley salmonid populations (Lindley *et al*. 2006, Good *et al.* 2005). NMFS assumes that the general categories for assessing salmonid population viability will also be useful in assessing the viability of the Southern DPS of green sturgeon. The following summary has been compiled from the best available data and information on North American green sturgeon to provide a general synopsis of the viability parameters for this DPS.

Abundance

Currently, there are no reliable data on population sizes, and data on population trends is also lacking. Fishery data collected at Federal and State pumping facilities in the Delta indicate a decreasing trend in abundance between 1968 and 2006 (70 FR 17386). Captures of larval green sturgeon in the RBDD rotary screw traps have shown variable trends in spawning success in the upper river over the past several years and have been complicated by the operations of the RBDD gates during the green sturgeon spawning season in previous years. In 2011, a wet year in the Sacramento River, captures in the rotary screw trap have been substantially higher than in previous years. The last strong year class, based on captures of larval sturgeon, was in 1995. This would suggest that the 2011 year class for green sturgeon will be a strong year class.

Productivity

There is insufficient information to evaluate the productivity of green sturgeon. However, as indicated above, there appears to be a declining trend in abundance, which indicates low to negative productivity.

Spatial Structure

Current data indicates that the Southern DPS of North American green sturgeon is made up of a single spawning population in the Sacramento River. Although some individuals have been observed in the Feather and Yuba rivers, it is not yet known if these fish represent separate spawning populations or are strays from the mainstem Sacramento River. Therefore, the apparent presence of a single reproducing population puts the DPS at risk, due to the limited spatial structure. As mentioned previously, the confirmed presence of fertilized green sturgeon eggs in the Feather River suggests that spawning can occur in that river, at least during wet years

with sustained high flows. Likewise, observations of several adult green sturgeons congregating below Daguerre Dam on the Yuba River suggests another potential spawning area.

Consistent use of these two different river areas by green sturgeon exhibiting spawning behavior or by the collection of fertilized eggs and/or larval green sturgeon would indicate that a second spawning population of green sturgeon may exist in the Sacramento River basin besides that which has been identified in the upper reaches of the Sacramento River below Keswick Dam.

Diversity

Green sturgeon genetic analyses shows strong differentiation between northern and southern populations, and therefore, the species was divided into Northern and Southern DPSs. However, the genetic diversity of the Southern DPS is not well understood.

Threatened Southern Distinct Population Segment (DPS) of North American green sturgeon Critical Habitat

Critical habitat was designated for the Southern DPS of North American green sturgeon on October 9, 2009 (74 FR 52300). Critical habitat for Southern DPS green sturgeon includes the stream channels and waterways in the Sacramento – San Joaquin River Delta to the ordinary high water line except for certain excluded areas. Critical habitat also includes the main stem Sacramento River upstream from the I Street Bridge to Keswick Dam, and the Feather River upstream to the fish barrier dam adjacent to the Feather River Fish Hatchery. Coastal marine areas include waters out to a depth of 60 meters from Monterey Bay, California, to the Juan De Fuca Straits in Washington. Coastal estuaries designated as critical habitat include San Francisco Bay, Suisun Bay, San Pablo Bay, and the lower Columbia River estuary. Certain coastal bays and estuaries in California (Humboldt Bay), Oregon (Coos Bay, Winchester Bay, Yaquina Bay, and Nehalem Bay), and Washington (Willapa Bay and Grays Harbor) are also included as critical habitat for Southern DPS green sturgeon. Only the critical habitat within the Delta fall within the WHCP treatment area.

Critical habitat for the Southern DPS of North American green sturgeon includes the estuarine waters of the Delta, which contain the following elements:

Food Resources

Abundant food items within estuarine habitats and substrates for juvenile, subadult, and adult life stages are required for the proper functioning of this PCE for green sturgeon. Prey species for juvenile, subadult, and adult green sturgeon within bays and estuaries primarily consist of benthic invertebrates and fish, including crangonid shrimp, callianassid shrimp, burrowing thalassinidean shrimp, amphipods, isopods, clams, annelid worms, crabs, sand lances, and anchovies. These prey species are critical for the rearing, foraging, growth, and development of juvenile, subadult, and adult green sturgeon within the bays and estuaries.

Water Flow

Within bays and estuaries adjacent to the Sacramento River (*i.e.*, the Sacramento-San Joaquin Delta and the Suisun, San Pablo, and San Francisco bays), sufficient flow into the bay and estuary to allow adults to successfully orient to the incoming flow and migrate upstream to spawning grounds is required. Sufficient flows are needed to attract adult green sturgeon to the Sacramento River from the bay and to initiate the upstream spawning migration into the upper river.

Water Quality

Adequate water quality, including temperature, salinity, oxygen content, and other chemical characteristics, is necessary for normal behavior, growth, and viability of all life stages. Suitable water temperatures for juvenile green sturgeon should be below 24 C (75 F). At temperatures above 24 C, juvenile green sturgeon exhibit decreased swimming performance (Mayfield and Cech 2004) and increased cellular stress (Allen *et al.* 2006). Suitable salinities in the estuary range from brackish water (10 parts per thousand - ppt) to salt water (33 ppt). Juveniles transitioning from brackish to salt water can tolerate prolonged exposure to salt water salinities, but may exhibit decreased growth and activity levels (Allen and Cech 2007), whereas subadults and adults tolerate a wide range of salinities (Kelly *et al.* 2007). Subadult and adult green sturgeon occupy a wide range of dissolved oxygen (DO) levels (Kelly *et al.* 2007, Moser and Lindley 2007). Adequate levels of DO are also required to support oxygen consumption by juveniles (ranging from 61.78 to 76.06 mg O2/hr^{-1} kg^{-1}, Allen and Cech 2007). Suitable water quality also includes water free of contaminants (*e.g.*, organochlorine pesticides, poly aromatic hydrocarbons (PAHs), or elevated levels of heavy metals) that may disrupt the normal development of juvenile life stages, or the growth, survival, or reproduction of subadult or adult stages. Green sturgeon have recently been identified by UC Davis researchers as being highly sensitive to selenium levels.

Migratory Corridor

Safe and unobstructed migratory pathways are necessary for the safe and timely passage of adult, sub-adult, and juvenile fish within the region's different estuarine habitats and between the upstream riverine habitat and the marine habitats. Within the waterways comprising the Delta, and bays downstream of the Sacramento River, safe and unobstructed passage is needed for juvenile green sturgeon during the rearing phase of their life cycle. Rearing fish need the ability to freely migrate from the river through the estuarine waterways of the delta and bays and eventually out into the ocean. Passage within the bays and the Delta is also critical for adults and subadults for feeding and summer holding, as well as to access the Sacramento River for their upstream spawning migrations and to make their outmigration back into the ocean. Within bays and estuaries outside of the Delta and the areas comprised by Suisun, San Pablo, and San Francisco bays, safe and unobstructed passage is necessary for adult and subadult green sturgeon to access feeding areas, holding areas, and thermal refugia, and to ensure passage back out into the ocean.

Water Depth

A diversity of depths is necessary for shelter, foraging, and migration of juvenile, subadult, and adult life stages. Tagged adults and subadults within the San Francisco Bay estuary primarily occupied waters over shallow depths of less than 10 m, either swimming near the surface or foraging along the bottom (Kelly *et al*. 2007). In a study of juvenile green sturgeon in the Delta, relatively large numbers of juveniles were captured primarily in shallow waters from 3 to 8 feet deep, indicating juveniles may require shallower depths for rearing and foraging (Radtke 1966). Thus, a diversity of depths is important to support different life stages and habitat uses for green sturgeon within estuarine areas.

Sediment Quality

Sediment quality (*i.e.*, chemical characteristics) is necessary for normal behavior, growth, and viability of all life stages. This includes sediments free of contaminants (*e.g.*, elevated levels of selenium, polyaromatic hydrocarbons (PAHs), and organochlorine pesticides) that can cause negative effects on all life stages of green sturgeon.

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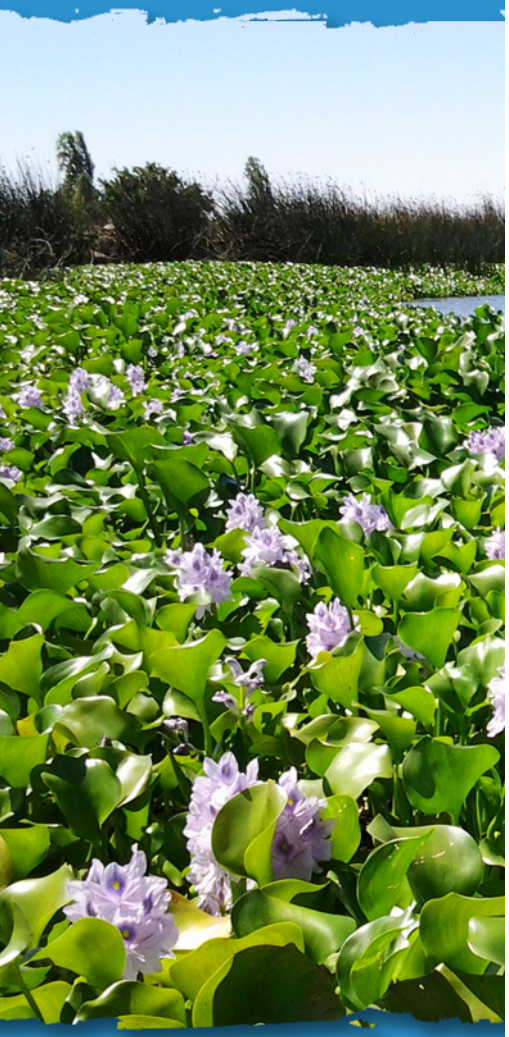
Section 5

Environmental Baseline and Cumulative Effects



5. Environmental Baseline and Cumulative Effects

This section of the biological assessment discusses environmental baseline conditions of WHCP treatment area and cumulative effects. The section is organized as follows:

A. Environmental Baseline

B. Cumulative Effects.

A. Environmental Baseline

The Endangered Species Act (ESA) environmental baseline includes the past and present impacts of all Federal, State, or private actions and other human activities in the action area, anticipated impacts of all proposed Federal projects in an action area that have already undergone Formal, or Early Section 7 Consultation, and the impact of State or private actions that are contemporaneous with WHCP (50 CFR §402.02).

This section describes five specific actions that are related to WHCP and potential WHCP impacts on listed species in order to provide a general overview of the Delta project area environmental baseline. The five Delta actions are as follows: (1) Delta water hyacinth, (2) *Egeria densa* Control Program (EDCP), (3) Delta invasive species, (4) Delta agriculture, and (5) Delta water quality.

These five impacts should be taken within the context of numerous large and small-scale actions in the Delta related to resource conservation, endangered species, restoration, water conveyance, water quality, and water use that affect listed species in the project area. Many of these larger actions have been in operation for decades, while others are in the early stages of planning, environmental permitting, and/or operation. This section will not attempt to describe in detail these large-scale projects, such as the Central Valley Project (CVP) and State Water Project (SWP), that are extensively documented elsewhere (for example documents related to the water project Operational Criteria and Plan (OCAP): USBR 2008, NMFS 2009, and USFWS 2008). These large scale water projects have known impacts on listed species, particularly fish.

This section provides some background context on the baseline condition of the Delta. The Delta is possibly the most environmentally sensitive region in California today. The Delta also has been described as “heavily modified” (Sommer et al. 2007). Starting in the mid-1800’s, the Delta has been subject to

hydraulic gold mining, channelization and wetland reclamation, fish and other non-native species introductions, dams controlling water inflows, and water exports (Sommer et al. 2007).

Concerns about the Delta environment gained momentum in the early 1990s. In establishing the Delta Protection Commission in 1992, the California legislature recognized that the Delta is "a natural resource of statewide, national, and international significance, containing irreplaceable resources." In the twenty years since the Delta Protection Commission was established, and particularly over the last few years, concerns about water quality, land subsidence, flooding, climate change, salinity, invasive species, risk of catastrophic earthquake, and declining fish populations have increased.

The CALFED Bay-Delta Program was created in 1994 to reduce conflicts between interest groups in the Delta and move toward restoring the Delta ecosystem. CALFED produced a number of planning and environmental documents between 1994 and 2000 (California Department of Fish and Game 2011). The first phase of CALFED efforts were completed in 2007.

There is widespread acknowledgement among California policymakers that the Delta is in crisis. As the Governor's Delta Vision Blue Ribbon Task Force stated, "ecosystems have eroded, levees have deteriorated, fish populations have collapsed, and our system of delivering water has become ever more precarious (Isenberg et al. 2008). There are numerous efforts at the federal, state, and local level, to improve conditions in the Delta.

In 2006, Governor Schwarzenegger established the Delta Vision Blue Ribbon Task Force to identify a sustainable strategy for managing the Delta. The Governor's Executive Order recognized that "failure to act to address identified Delta challenges and threats will result in potentially devastating environmental and economic consequences of statewide and national significance" (Executive Order S-17-06).

The Delta Vision Blue Ribbon Task Force established a strategic plan to meet twelve objectives, the first objective being: "The Delta ecosystem and a reliable water supply for California are the primary co-equal goals of a sustainable Delta" (Delta Vision Blue Ribbon Task Force 2008).

In early 2008, Governor Schwarzenegger initiated another major collaborative planning effort, the Bay Delta Conservation Plan (BDCP). This initiative is led by the California Department of Water Resources (DWR), California Department of Fish and Game (CDFG), U.S. Bureau of Reclamation (USBR), USFWS, and NMFS. The "purpose of the BDCP is to help recover endangered and sensitive species and their habitats in the Delta in a way that will also provide for sufficient and reliable water supplies" (DWR 2008). The BDCP will examine four water conveyance and physical habitat restoration alternatives for the Delta, including a peripheral aqueduct or tunnel from the Sacramento River to the south Delta. On July 25, 2012, California Governor Edmund G. Brown Jr., Secretary of the Interior Ken Salazar, and NOAA Assistant Administrator for Fisheries Eric Schwaab outlined revisions to the proposed BDCP that, along with a

range of alternatives, will undergo public environmental review. The revised proposal for a peripheral tunnel includes fewer water intake facilities (three versus five), and lower total water capacity (9,000 cfs versus 15,000 cfs) than earlier proposals (California Natural Resources Agency July 2012). The draft BDCP and corresponding EIR/EIS is to be released for public review in Fall, 2012.

The Sacramento-San Joaquin Delta Reform Act of 2009 (SBX7 1) enacted by the California legislature in November 2009, established a number of additional Delta-wide initiatives. The Delta Reform Act again established State policy coequal goals of a more reliable water supply for California and protecting, restoring, and enhancing the Delta ecosystem. The Delta Conservancy was created as a primary state agency to implement ecosystem restoration in the Delta and to support efforts that advance environmental protection and the economic well-being of Delta residents. The Delta Conservancy released a Draft Delta Conservancy Strategic Plan in May 2012.

The Delta Stewardship Council was established to develop and implement a legally enforceable, long-term management plan for the Delta. The Delta Stewardship Council released a draft plan for review in May 2012, with a final plan expected in November 2012.

The CDFG released a draft Conservation Strategy for Restoration of the Sacramento-San Joaquin Delta Ecological Management Zone and the Sacramento and San Joaquin Valley Regions (CDFG 2011) as part of the CALFED process. After passage of the Delta Reform Act, CDFG coordinated their ongoing planning efforts with the Delta Conservancy and Delta Stewardship Council, as well as the BDCP. The challenge of meeting water supply and ecosystem needs in the Delta has also been the subject of three National Academy of Sciences studies since 2010.

The WHCP is a minor element of this complex and dynamic Delta environment. The WHCP seeks to control only one of the hundreds of invasive species in the Delta. The relatively small WHCP operates within the much larger context of an environment that has been managed and materially manipulated since the mid-1800s.

The challenge in today's Delta is to support gradual restoration of natural Delta ecosystems, where possible, while preventing further environmental deterioration. The specific challenge of WHCP is to control the growth of water hyacinth within this highly modified Delta environment. Water hyacinth, left to grow unchecked, has significant negative environmental impacts. At the same time, WHCP also must minimize potential negative impacts of water hyacinth treatment.

1. Delta Water Hyacinth

Water hyacinth, native to South America, was first reported in California in 1904 in a Yolo County slough. It spread gradually for many decades, and was reported in Fresno and San Bernardino Counties in 1941 and in the Delta in the late 1940s and early 1950s. There were increased reports of water hyacinth in the Delta region during the 1970s, and by 1981, water hyacinth covered 1,000 acres of the Delta, and 150 of the 700 miles of waterways (U.S. Army Corps of Engineers 1985).

Water hyacinth coverage estimates in the Delta since 1981 have ranged from approximately less than 500 acres up to approximately 2,500 acres. This wide range of annual water hyacinth acreage in the Delta is dependent on many factors including: acres treated, timing of treatments, winter air and water temperatures, summer air and water temperatures, water flow, and rainfall.

Water hyacinth grows in wetlands, marshes, shallow ponds, sluggish flowing waters, large lakes, reservoirs, and rivers (Batcher 2000). Water hyacinth often forms monospecific mats across sloughs and other waterways (Batcher 2000, Cohen and Carlton 1995). The mats are dispersed by winds and currents (Batcher 2000). In the Delta, water hyacinth is found in sloughs, connecting waterways, and tributary rivers. The growing season for water hyacinth in the Delta is typically from March to early December. Water hyacinth dies back or reduces growth during the cold winter months. However, the majority of plants do not die, and carry-over plants begin to grow in spring as the weather warms. Plants can tolerate extremes of water level fluctuation and seasonal variations in flow velocity, extremes of nutrient availability, pH, temperature, and toxic substances (Gopal 1987).

Water hyacinth requires freshwater. Water hyacinth will not survive in salinities greater than 2.0 to 2.5 parts per thousand (ppt) (Wilson et al., 2001). Thus, water hyacinth infestations occur in those areas within the Delta with very low salinity. (Freshwater is defined as less than 3ppt, drinking water is less than 1ppt, brackish water is typically defined as between 3ppt and 35ppt, and seawater is 35ppt.) In the Delta, the line at which 2ppt salinity occurs, the X2, fluctuates with tidal levels and water outflow. The X2 line is typically located around Suisun Bay. As a result, water hyacinth generally does not grow in the far western portions of the Delta, beyond this zone.

Over the long-term, water management practices in the Delta have reduced the natural variability in Delta salinity. Water exports and releases during the summer months reduce the inflow of San Francisco Bay waters, and maintain low levels of salinity suitable for drinking water and agriculture. This also improves growing conditions and habitat for water hyacinth and other invasive species.

Water hyacinth reproduces both vegetatively and sexually, although most reproduction is thought to be vegetative. In sexual reproduction, seeds may remain viable for up to twenty years, often sprouting along the muddy shorelines after a dry period, and dropping into the water with high tides (Batcher 2000). In vegetative reproduction, short runner stems (stolons) radiate from the base of the plant to form daughter plants (Batcher 2000).

Water hyacinth nursery areas include slow moving waterways, temporarily isolated oxbow lakes, tule stands along channel margins, and stagnant, dead-end sloughs. Small colonies of plants separate and form floating mats that drift downstream, infesting new areas. When water hyacinth extends into faster channels, or when higher flows occur, plants are torn away from their mats and moved by currents and wind until they encounter obstructions such as marinas, irrigation pumps, or backwater areas (U.S. Army Corps of Engineers 1985).

Water hyacinth spreads and grows rapidly under favorable temperature and nutrient conditions (warmer temperatures and higher nutrient levels). The growing range for water hyacinth is between 10 C and 40 C (Gopal 1987). Water hyacinth mats weigh up to 200 tons per acre and its surface area may double in size in just six to fifteen days (Harley et al. 1996). Water hyacinth follows three growth phases: (1) reapportioning of biomass to emergent shoots following winter freezes, (2) increased branching, ramet production, high leaf densities, and foliar diversity, and (3) increased leaf size (versus numbers), loss of smaller plants, lower absolute density, but maximum standing crop values (Center and Spencer 1981).

In a study comparing water hyacinth growth and temperature in the Sacramento San Joaquin Delta, Spencer and Ksander found that water hyacinth achieved maximum biomass in October (Spencer and Ksander 2005). This was later than expected, and later than in other regions of the country. Water hyacinth in the Delta increased in height from less than 10 cm in winter and early spring, to more than 80 cm in later summer (Spencer and Ksander 2005). New leaves began growing in March, and by August 7, leaves had reached 50 percent of their maximum leaf area (Spencer and Ksander 2005).

Ecosystem Effects of Water Hyacinth

The presence of water hyacinth in the Delta impacts both native species and human uses. Water hyacinth displaces native aquatic plant and animal communities, causes economic hardships, and interferes with water uses (Batcher 2000). Water hyacinth clogging Delta waterways and impeding navigation were an impetus for legislation in 1982 to establish WHCP.

The negative impacts of water hyacinth have been widely recognized and documented in the scientific literature. A 1967 article in the *Hyacinth Control Journal* notes the following:

> "The problems created by water hyacinth are many and varied. First, it constitutes a health hazard by providing mosquito larvae with an ideal breeding place. Small fish that ordinarily feed on these larvae are kept from doing so by the thick mat of vegetation. Water hyacinth pollutes water supplies through growth and decomposition. The oxygen-depleting pollutional load imposed by one acre of growing water hyacinth is estimated to equal the sewage created by 40 people. Second, fish are killed by oxygen starvation and pollution, and native aquatic plants are replaced in areas completely covered by water hyacinth. Third, it interferes with navigation. Fourth, dense growth limits water sports recreation. Fifth, water hyacinth obstructs drainage and flow of water in canals. Sixth, it utilizes water through evapotranspiration" (Timmer and Weldon 1967). [emphasis added]

Like other invasive species control programs, WHCP must balance the cost of control, the impacts of control, and the benefits resulting from control. Below, we describe problems resulting from the spread of water hyacinth in the Delta. These problems are part of the environmental baseline within which WHCP will operate.

The Delta ecosystem is a critically important part of California's natural environment and the ecological hub of the Central Valley. Water hyacinth is labeled as an invasive habitat modifier. It provides a structurally complex canopy, with roots in the water column and leaves above water providing

habitat for both native and non-native species. The CALFED Ecosystem Restoration Program Plan states that "these weeds [water hyacinth] are extremely dangerous because of their ability to displace native plant species, harm fish and wildlife, reduce foodweb productivity, or interfere with water conveyance and flood control systems" (CALFED Vol. 1 2000, p. 462). Similarly, U.S. Fish and Wildlife Service (USFWS) notes that excessive water hyacinth growth outcompetes native vegetation and clogs waterways, impeding and impairing aquatic life (USFWS 1995). In the Stone Lakes National Wildlife Refuge in Sacramento County, USFWS found that fish and wildlife habitat would be "greatly degraded or lost completely on shorelines, shallow water, and deepwater areas" if water hyacinth was allowed to grow unchecked (USFWS 1995).

Water Quality, Including Dissolved Oxygen

The dense water hyacinth mats block sunlight, inhibiting photosynthesis in algae and submersed vascular plants (CALFED Vol. 1 ERP 2000, USFWS 1995). Water hyacinth increases sedimentation and accretion of organic matter, inhibits gaseous interchange with the air, reduces water flow, and depletes oxygen, all of which harm other aquatic organisms (CALFED Vol. 1 ERP 2000). In addition, organic fallout can influence the benthic zone (Toft 2000) and alter ecosystem processes such as nutrient cycling, hydrologic conditions, and water chemistry (CALFED Vol. 1 ERP 2000).

Toft and others have found lower levels of dissolved oxygen under water hyacinth canopies. Average spot measures were below 5 mg/L in water hyacinth (the minimum level for fish survival) and above 5 mg/L in pennywort (Toft 2000). These results were supported by a study in Texas which found lower dissolved oxygen in water hyacinth compared to other aquatic weeds, and a University of California Davis study which found dissolved oxygen levels of as low as 0 mg/L below a solid water hyacinth mat (Toft 2000). Toft hypothesizes that the lower dissolved oxygen levels explain the absence of epibenthic amphipods and isopods beneath the water hyacinth canopy at one test site (Toft 2000, Toft 2003).

Dissolved oxygen levels under the roots of water hyacinth floating islands in the Parana River floodplain in Argentina were a maximum 2.3 mg/l within the first meter, and typically only 1 mg/l , with even lower DO at deeper in the river (Petr 2000). DO levels measured in the Sudd River in Sudan were 1.8 mg/l at 30 cm below the water hyacinth mat (Petr 2000).

Phytoplankton and Zooplankton

Water hyacinth can act to reduce phytoplankton and zooplankton production. A 1975 study by McVea and Boyd found reduced phytoplankton production, due to shading and removal of phosphorus from water, on ponds with 10 percent and 25 percent water hyacinth cover as compared to ponds with 0 percent and 5 percent cover (McVea and Boyd 1975). A recent review of the ecological impacts of water hyacinth by Villamagna and Murphy (2010) summarized that overall, water hyacinth limits productivity of phytoplankton under hyacinth mats. In addition, phytoplankton levels increased when water hyacinth was removed from reservoirs (Villamagna and Murphy 2010).

Shanab et al. (2010) note that water hyacinth is known for its negative effects on microbes, including phytoplankton. Shanab et al. studied the allelopathic effects of water hyacinth, isolating complex and potent antialgal, antifungal, and antibacterial compounds from extracts of water hyacinth.

The impact of water hyacinth on zooplankton abundance is more complex than phytoplankton, with responses varying by taxa and geographic location (Villamagna and Murphy 2010). Zooplankton decrease in response to reduced phytoplankton density and reduced dissolved oxygen (Villamagna and Murphy 2010). In a study simulating shading and anoxia due to free-floating plants such as water hyacinth, Fontanarosa et al. (2010) found that anoxia impaired zooplankton development. However, the structural environment created by water hyacinth mats can also create microhabitats for epiphytic zooplankton (Villamagna and Murphy 2010). Movement of zooplankton within an ecosystem and factors such as turbulence, temperature, phytoplankton, light intensity, chlorophyll-a, and dissolved oxygen may have greater impact on zooplankton than water hyacinth.

Macroinvertebrates

Water hyacinth generally provide habitat for macroinvertebrates, increasing abundance and species richness, particularly at the outer edge of water hyacinth mats (Villamagna and Murphy 2010). However, in the Delta, Toft found significant differences in insect densities in water hyacinth and pennywort (a native aquatic plant), with increased taxa richness and diversity of invertebrates in pennywort in the early summer. While there were a greater number of species present in water hyacinth later in the summer, there were fewer native species (Toft 2000, Toft 2003).

Many of the macroinvertebrates supported by water hyacinth are disease vectors. Water hyacinth increases mosquito habitat by providing larval breeding sites where mosquito predators cannot reach (CALFED Vol. 1 2000), creating microhabitats for the vectors of malaria, encephalitis, schistosomiasis (USFWS 1995), and of more recent concern, West Nile virus. The link between mosquitos and water hyacinth was identified as early as the 1920s, and has been verified in the literature in the decades that follow (Mack and Smith 2011). Blair (2011) notes that water hyacinth is among the invasive weeds that reduce water circulation and inhibit predators of mosquito larvae, and that in general, waterways degraded by invasive weeds promote mosquito breeding. The University of California Mosquito Research Program identifies water hyacinth as one of the key habitats for mosquitos, noting that some species of mosquitos transmit organisms that cause malaria, encephalitis, canine heartworm, and West Nile virus (O'Connor-Marer and Garvey 2001). Water hyacinth also provides habitat for freshwater snails that carry schistosomiasis (Mack and Smith 2011).

Plants

Water hyacinth competes with native plants, including Mason's lilaeopsis, a special status species (CALFED Vol. 1 ERP 2000). Villamagna and Murphy (2010) note that water hyacinth out-competes submerged vegetation, acting similar to a forest canopy by restricting vegetative growth below.

Fish

Numerous studies of the effects of water hyacinth on fish have had variable results (Villamagna and Murphy 2010). Generally, fish (invasive and/or native species) are more abundant on the edges of water hyacinth mats, but avoid large mats where dissolved oxygen levels are low. McVea and Boyd (1975) examined effects of water hyacinth on tilapia productivity in ponds, and found that fish production decreased at 10 percent and 25 percent water hyacinth coverage due to reduced phytoplankton abundance. Petr (2000) notes that low dissolved oxygen resulting from thick water hyacinth mats can result in fish kills. In a lake in the Philippines, a reduction in fish population was figured to be due to low dissolved oxygen levels and increased carbon dioxide levels due to extensive water hyacinth coverage (Petr 2000).

Birds and Mammals

Villamagna and Murphy (2010) summarize that dense water hyacinth mats may physically prevent water bird access to prey, and that homogenous water hyacinth mats could reduce diversity of waterbird species. Even smaller infestations of water hyacinth along shorelines can prevent ducks, turtles, snakes, and frogs from seeking shelter (USFWS 1995).

2. *Egeria densa* Control Program (EDCP)

The DBW, in collaboration with the USDA-ARS, implements the EDCP. The EDCP is an aquatic weed control program designed to minimize the extent of *Egeria densa* in the Delta. The USDA-ARS acts as the nexus for federal regulatory processes, as well as providing research, expertise, and decision-making input for EDCP planning. In 1996, in response to growing concerns about the spread of an aquatic invasive weed, *Egeria densa*, the California Legislature passed Assembly Bill (AB) 2193, authorizing the DBW to develop a control program for this invasive species. The DBW began treating *Egeria densa* in the Delta in 2001, in collaboration with USDA-ARS, after completing an Environmental Impact Report (EIR) and obtaining the required NPDES permit and NMFS and USFWS biological opinions. DBW and USDA-ARS are preparing ESA consultation documents for EDCP concurrent with the WHCP process.

Egeria densa (Brazilian Elodea) is a submerged non-native aquatic plant, introduced into the Delta approximately fifty years ago. This fast growing weed obstructs waterways, crowds out native plants, impedes anadromous fish migration and boat navigation, slows water flows, entraps sediments, and clogs agricultural and municipal water intakes. *Egeria densa* negatively impacts delta smelt by reducing turbidity and overwhelming littoral (near shore) habitats (USFWS 2008). *Egeria densa* infests almost twenty percent of the Delta's 55,000 surface acres, and is spreading at approximately 100 acres per year.

In addition to the BOs and NPDES permit, listed above, the EDCP also operates under three key environmental documents: (1) 2001 EDCP Environmental Impact Report, (2) 2003 First Addendum to EDCP EIR, and (3) 2006 Second Addendum to EDCP EIR (with five year program review and future program operations plan). The

Second Addendum, and the regulatory agency documents have guided the EDCP over the last six years (2007 through 2012).

Prior to 2006, the EDCP operated under the original, and somewhat more restrictive, NPDES permit and Biological Opinions. These more recent documents reflect the lower level of environmental impact demonstrated during the first five years of the EDCP. On July 2, 2012, the USDA-ARS and the DBW received a letter of concurrence from NMFS agreeing with the USDA-ARS and the DBW's determination that the proposed use of fluridone-based herbicide products for the 2012 treatment season is not likely to adversely affect federally listed salmonids, green sturgeon, or critical habitat.

The EDCP generally utilizes trained two-person teams to conduct treatments in the Delta between approximately April 1st to October 15th. Start dates have been limited by the terms of the Biological Opinions, and vary by location. In the last five seasons, the DBW has conducted the majority of EDCP treatments in the first several months of the season (April through June). These same trained crews also implement WHCP.

The EDCP is permitted to utilize two aquatic herbicides, Reward (diquat)[1] and Sonar (fluridone) for control of *Egeria densa*. Over the last five years, the DBW has not utilized Reward, but has utilized up to three formulations of fluridone: (1) Sonar PR Granular, (2) Sonar Q Pellets, and (3) Sonar AS Aqueous. Treatment crews use injection hoses to apply aqueous herbicide into treatment areas, and a broadcast method to apply pellets or granules.

Fluridone is a selective systemic herbicide. Fluridone inhibits formation of carotene, resulting in the degradation of chlorophyll when exposed to sunlight. Because carotene is formed primarily during new growth, fluridone is most effective when the plant is growing rapidly (i.e. in spring, and sometimes in fall during a final growth spurt). This plant growth stage is why the DBW focuses treatments in the early part of the season, with some follow-up at the end of the season. Exposure to sunlight breaks fluridone down into naturally occurring elements in the environment.

As a condition of its permits, the EDCP also conducts an extensive monitoring program to measure herbicide residue and water quality parameters. The DBW is required to conduct both site-specific and daily monitoring. DBW environmental scientists take water samples immediately pre-application, and post-application, at specified time intervals. This monitoring occurs at a specified percentage of total treatment sites. In addition, treatment crews conduct daily monitoring, reporting dissolved oxygen, wind speed, temperature, acres treated, quantity of herbicide, presence of species of concern, and coordinates of the treatment location.

At the completion of each treatment season, the DBW and the USDA-ARS report program results to USFWS, NMFS, and CVRWQB. The DBW conducted toxicity testing in the first several years of operation, but the regulatory agencies eliminated this requirement when test results were negative.

[1] Diquat has higher toxicity, and thus may only be used between June 1 and July 31. The DBW has avoided the use of Diquat over the last several years.

Table 5-1
EDCP Areas Treated (by Site Name), Net Acres, and Pounds Herbicide Active Ingredient (Fluridone)

Site Name	2007	2008	2009	2010	2011
1. Franks Tract	✓	✓		✓	✓
2. White Slough			✓	✓	
3. Disappointment Slough			✓	✓	
4. Fourteen Mile Slough			✓		
5. Pipers Slough					✓
6. Taylor Slough					✓
7. Sandmound Slough					✓
8. Discovery Bay					✓
Net Acres	2,571	2,571	228	641	3,195
Pounds Active Ingredient	126,400	90,339	11,242	39,482	150,708

For the first five (5) years of the EDCP (2001 to 2005), the DBW treated 19 different sites within the Delta, covering between 268 and 622 acres per year. A study of the first five-years of operation found that the EDCP was likely restraining *Egeria densa* from spreading even more than it already had, but that the EDCP was "not keeping up" with the Delta-wide *Egeria densa* infestation. Following this initial five-year program evaluation in 2006, the DBW implemented a new, more focused, approach.

In 2007, renewal of the NMFS and the USFWS Biological Opinions allowed an early April 1st start date and a new treatment regime. In 2007 and 2008, the DBW focused all EDCP treatments within three treatment sites in Franks Tract, a known *Egeria densa* nursery area. This focused treatment approach was highly effective, and after two years of treatment, boats could navigate within Franks Tract.

DBW measures reductions in bio-volume and bio-cover of *Egeria densa* to determine efficacy of EDCP treatments. The current treatment protocols have been effective, as compared to untreated control sites. For example, in October 2011, following the 2011 treatment season, between zero and 16 percent of treated sites had a mean bio-volume of over 50 percent, while 65 to 94 percent of untreated sites had a mean bio-volume of over 50 percent. This reduction in *Egeria densa* bio-volume helps reduce the negative impact of *Egeria* on the Delta ecosystem.

Due to the success of the Franks Tract treatment regime, the DBW continued the focused treatment approach, expanding to new areas in 2009, 2010, and 2011. **Table 5-1,** above, identifies EDCP treatment areas, net acreage treated, and pounds of active ingredient (fluridone) for the last five years of operation, the period covered by the most recent Biological Opinions. In 2012, under the letter of concurrence, the DBW started EDCP treatments on July 9, 2012. The DBW will treat as many as nine different sites, and up to 9,929 acres in 2012.

During the last five years of EDCP operation, there was no known take or harassment of federally endangered or threatened species. To minimize the occurrence of take, the DBW checked Interagency Ecological Program (IEP) surveys, as well as California Department of Fish and Game trawls, prior to, and during, the treatment season to monitor the presence of Chinook salmon, steelhead trout, and delta smelt during early treatment season months when these species may be present in treatment areas. Treatment crews also conducted surveys to evaluate the presence of valley elderberry longhorn beetle and giant garter snake habitats throughout each treatment season.

3. Delta Invasive Species

Invasive species are generally defined as non-indigenous species that adversely affect economics, environments, ecological relationships, and/or habitats where they have been introduced (Masters and Norgrove 2010, USEPA 2008). The Delta is among the most invaded ecosystem worldwide, with over 200 invasive, non-native species (Cohen and Carlton 1995). Cohen and Carlton found that non-native species accounted for 40 to 100 percent of common species at many sites (Cohen and Carlton 1995). Invasive species that have adapted to, and inhabited, the Delta include: non-native Centrarchids (various bass, bluegill, sunfish, crappie), overbite clam (*Corbula amurensis*), Asian clam (*Corbicula fluminea*), several zooplankton species, and invasive plants such as *Egeria densa* and water hyacinth. The presence of invasive species is linked to other changes in the Delta, such as water quality and water flows. In addition, changes in nutrient concentrations over time (i.e. increased nitrogen-to-phosphorous ratios) may be a significant driver of food web changes that are favorable to non-native species (Glibert et al. 2011).

While some non-native species have relatively little effect on the environment, others result in negative ecological and economic impacts in the Delta. Invasive species have altered food webs and habitats, compete with native species for resources, and directly prey upon native species (CDFG 2011). "Problem" invasive species are often grouped into one of two categories: ecosystem engineers or food-web disruptors (Mount et al. 2012). Ecosystem engineers physically alter ecosystem processes, degrading habitat for native species (Mount et al. 2012). *Egeria densa* and water hyacinth are commonly categorized as ecosystem engineers due to their impact on sediment, water clarity, ecosystem diversity, and dissolved oxygen.

Food-web disruptors are species that significantly alter food webs, reducing the quantity or quality of food available for native species (Mount et al. 2012). The presence of invasive clams, and changes in replacement of native zooplankton with non-native zooplankton are examples of food web disruptors in the Delta. The Asian and overbite clams have significantly altered the Delta food web by filtering most phytoplankton from the water, particularly in the western Delta and Suisun Bay. This in turn diminishes food supplies for zooplankton and mysid shrimp, which become scarcer, thus diminishing food supply for fish such as the delta smelt and salmonids that rely on them (Mount et al. 2012). Several studies have found that invasive species (including macrophytes) are the second greatest threat to listed fish species behind

habitat loss, impacting 63 percent to 70 percent of listed species (Schultz and Dibble 2012).

4. Delta Agriculture

The Delta is an important agricultural area. Farming in the Delta region began in the 1850s, following passage of the Swamp and Overflow Act, and Reclamation District Act, which provided for the sale of swamp and overflow lands for reclamation (DPC January 2001). Early farmers built a system of levees and irrigation ditches, and began growing a variety of vegetables, fruits, and grains. Over time, most farms have shifted from growing diverse crops, to growing a few crops, which are rotated (DPC January 2001). Crops that have been important at various times in the Delta include potatoes, asparagus, pears, and sugar beets. Characteristics that make the Delta well-suited to agriculture include: rich soil, ample water, a long growing season, mild climate, and proximity to end markets (DPC May 2001).

California is the fifth largest agricultural economy in the world, producing over 400 plant and animal commodities worth $37.5 billion in 2010 (CDFA 2011). There were over 25 million acres of agricultural land (including grazing land) in California in 2010 (CDFA 2011). In 2010, the Delta region had about 500,000 acres available for agriculture, with 461,000 acres in use (DPC 2011), just over 2 percent of the total agricultural acreage statewide, and approximately 67 percent of Delta land acreage. Of the Delta's 500,000 agricultural acres, approximately 80 percent is classified as prime farmland (DPC 2011).The average annual gross value of the agricultural output of the Delta is typically about two percent of the statewide agricultural output, and was $800 million in 2009. **Table 5-2,** on the next page, summarizes total and Delta agricultural land use in the six Delta counties.

Tables 5-3 and **5-4,** on the next page, identify the top ten Delta agricultural crops in 2009, based on annual average gross value, and acreage. These tables illustrate the diversity of agriculture in the Delta, with no single product dominating either acreage or economic output.

While agriculture is an important component of the Delta's economic infrastructure, it is also one of the many factors that negatively impacts listed species. These negative impacts are the result of several different factors, including the landscape, water diversions, and pesticides. The most significant, and long-term, implications of agriculture in the Delta are the structural changes that began in the 1850s. The levees and islands created to support agriculture are now part of the current Delta landscape.

There are approximately 1,800 agricultural water diversions in the Delta. During the peak summer irrigation season, diversions from these facilities collectively exceed 5,000 cubic feet per second (URS Corporation May 2007). Most of the irrigation diversions in the Delta are small (30 to 60 cm in diameter) and lack fish screens (Nobriga et al. 2004). Nobriga et al. found that fish entrainment was 99 percent higher in unscreened agricultural diversions than in screened diversions (2004). The overall impact of agricultural diversions on fish depends on a number of factors, including location, size, timing, and operation (Moyle and Bennett 2008).

Table 5-2
Total and Agricultural Acres in Delta Counties

County	Total County Acres	Total County Agricultural Acres (2010)	Approximate Country Delta Acres	Delta Total Agricultural Delta Acres (in production) (2010)
1. San Joaquin	912,602	737,503	317,778	214,053
2. Yolo	653,452	479,858	91,861	54,986
3. Sacramento	636,083	328,593	118,717	66,428
4. Solano	582,373	358,225	88,071	72,499
5. Contra Costa	514,019	146,933	104,751	48,062
6. Alameda	525,338	204,233	6,422	5,352
Total	3,823,867	2,255,345	727,600	461,380

Sources: USDA Census of Agriculture (www.agcensus.usda.gov); DOC, http://www.consrv.ca.gov; Delta Protection Commission 2011.DBW.

Table 5-3
Top Ten Delta Agricultural Crops, Based on 2009 Value

Agricultural Product	Annual Gross Value (in millions of dollars)
1. Processing tomatoes	$117.2
2. Wine grapes	105.0
3. Corn	93.0
4. Alfalfa	66.0
5. Asparagus	50.1
6. Pears	36.7
7. Turf	31.6
8. Potato	28.6
9. Almond	8.8
10. Watermelon	8.0

Source: Delta Protection Commission 2011

Table 5-4
Top Ten Delta Agricultural Products, Based on 2009 Acreage

Agricultural Product	Delta Irrigated Acres
1. Corn	105,362
2. Alfalfa	91,978
3. Processing tomatoes	38,123
4. Wheat	34,151
5. Wine grapes	30,148
6. Oats	15,847
7. Safflower	8,874
8. Asparagus	7,217
9. Pear	5,912
10. Bean, dried	5,493

Source: Delta Protection Commission 2011

Table 5-5
Comparison of Total and WHCP Herbicide Use in Six Counties with WHCP Treatments in 2010

Herbicide	Pesticide[a] Use in Six WHCP Counties in 2010	WHCP Herbicide[b] Use In Six WHCP Counties Treated in 2010	WHCP Herbicide Use as Percent of County Total Use
2,4-D dimethylamine salt	137,552	3,516	2.56%
Glyphosate isopropylamine salt	1,508,427	327	0.02%
Imazamox ammonium salt	522	–	–
Imazapyr isoproplyamine salt	266	–	–
Penoxsulam	252	–	–
Total	1,647,019	3,843	**0.23%**

[a] 2,4-D, glyphosate, imazamox, imazapyr, and penoxsulam

[b] 2,4-D and glyphosate

Sources: California Department of Pesticide Regulation and DBW. This total use of herbicides includes all reported uses, including WHCP applications.

Water hyacinth interferes with water pumping at irrigation intakes throughout the Delta with the potential for clogging by water hyacinth, resulting in inefficient pumping, increased pumping costs, and possible mechanical failure of pumps. In a letter to the U.S. Army Corps of Engineers in 1981, the San Joaquin Farm Bureau Federation stated that growers were facing increased costs from efforts to open clogged channels where water hyacinth was decreasing the flow of water to pumps and clogging screens. Water hyacinth also spreads into irrigation and drainage systems (U.S. Army Corps of Engineers 1985), and impairs the use of fish protective devices such as fish screens (CALFED Vol. 1 ERP 2000).

Increased sedimentation resulting from agriculture and urban practices within the Central Valley is one of the primary causes of salmonid and delta smelt habitat degradation (NMFS 1996). Sedimentation can adversely affect all freshwater stages of listed species by clogging or abrading gill surfaces, adhering to eggs, hampering fry emergence, burying eggs or alevins, reducing primary productivity and photosynthesis, and affecting DO levels (California Department of Water Resources 2007).

Agriculture in the Delta results in significant pesticide use that far outweighs WHCP and EDCP herbicide applications in the project area. In 2010, WHCP conducted treatments in only six of the eleven WHCP counties: Contra Costa, Merced, Sacramento, San Joaquin, Solano, and Stanislaus. **Table 5-5,** above, summarizes 2010 pesticide use report data for 2,4-D, glyphosate, imazamox, imazapyr, and penoxsulam for the six counties where WHCP treatments were conducted in 2010. Combined, these six counties utilized 1.6 million pounds active ingredient of the five WHCP herbicides (California DPR 2011). The WHCP utilized 3,843 pounds of herbicide active ingredient in 2010 (2,4-D and glyphosate only). Thus, WHCP herbicide utilization accounted for only 0.23 percent of total herbicide utilization for the five potential WHCP herbicides in 2010.

A recently released study funded through CALFED (Hoogeweg et al. 2011) developed a comprehensive simulation model to evaluate pesticides in the Delta as compared to co-occurrence of species of concern between 2000 and 2009. The study evaluated 40 pesticides identified by the CVRWQB as those of highest risk to aquatic life, focusing on pyrethrins and organophosphates. None of the 40 pesticides evaluated are utilized by WHCP or EDCP. Using a broader watershed approach covering the Sacramento River, San Joaquin River, and Bay-Delta, Hoogeweg et al. estimated that of the approximately 10 million pounds of active ingredient of the 40 selected pesticides applied per year, 14 percent, or 1.4 million pounds, reach the surface water. The study quantified toxicity thresholds (using risk quotients) for the 40 pesticides, and identified time and location of likely incidents (i.e. when estimated pesticide levels exceeded toxicity thresholds). The areas with greatest potential for concern within the Delta were the southern Delta estuary in San Joaquin County, and the confluence of the Cosumnes River, Dry Creek, and the Mokelumne River. Hoogeweg et al. also evaluated 30,000 water quality testing records from the same 2000 to 2009 time period, and found that approximately 75 percent of the 250 testing locations had exceeded toxicity thresholds at least once, and as many as 185 times. This study illustrates the high degree of pesticide loading to Delta waters, with significant quantities of higher-toxicity pesticides, far exceeding the herbicide risk and use of WHCP.

5. Delta Water Quality

The water quality of the Delta has been negatively impacted over the last 150 years (NMFS 2012). The State Water Resources Control Board (SWB) regulates water quality in California, through the federal Clean Water Act (CWA), and the Porter-Cologne Water Quality Control Act. The State Water Code gives Regional Water Boards primary responsibility for formulating and adopting water quality control plans in each of the State's nine regions.

There are two plans that jointly specify water quality controls for the Delta: the Water Quality Control Plan for the San Francisco Bay/Sacramento-San Joaquin Delta Estuary (Bay-Delta Plan), and the Water Quality Control Plan (Basin Plan) for the Sacramento River and San Joaquin River Basins. The Bay- Delta Plan, developed by the SWB, is complementary to the Basin Plan developed by the CVRWQCB. Water quality plans must also be approved by the USEPA. In addition, in February 2011, the USEPA initiated an advance notice of proposed rulemaking to seek comments from interested parties on possible USEPA actions to address water quality conditions affecting aquatic resources in the Delta (USEPA 2011). As of mid-2012, the USEPA had received over fifty comments, but taken no further action.

Both plans consist of beneficial uses to be protected, water quality objectives, and a program for implementation of the water quality objectives. A primary goal of the water quality planning process is to identify and protect beneficial uses for surface and groundwater in a given region. **Table 5-6,** on the next page, summarizes seventeen of the beneficial uses for Delta waters.

Table 5-6
Beneficial Uses in Delta Waters

Beneficial Use	Abbreviation
1. Municipal and domestic supply	MUN
2. Industrial service supply	IND
3. Industrial process supply	PRO
4. Agricultural supply	AGR
5. Groundwater recharge	GWR
6. Navigation	NAV
7. Water contact recreation	REC-1
8. Non-contact water recreation	REC-2
9. Shellfish harvesting	SHELL
10. Commercial and sport fishing	COMM
11. Warm freshwater habitat	WARM
12. Cold freshwater habitat	COLD
13. Migration of aquatic organisms	MIGR
14. Spawning, reproduction, and/or early development	SPWN
15. Estuarine habitat	EST
16. Wildlife habitat	WILD
17. Rare, threatened, or endangered species	RARE

Water quality objectives are "the limits or levels of water quality constituents or characteristics which are established for the reasonable protection of beneficial uses of water or the prevention of nuisance within a specific area" (Water Code Section 13050(h), in CVRWQCB 2007). In establishing water quality objectives, the Regional Water Boards must consider the following:

- Past, present, and probable future beneficial uses;
- Environmental characteristics of the hydrographic unit under consideration, including the quality of water available thereto;
- Water quality conditions that could reasonably be achieved through the coordinated control of all factors which affect water quality in the area;
- Economic considerations;
- The need for developing housing within the region;
- The need to develop and use recycled water (Water Code Section 13241).

The SWB and Regional Water Boards refine their respective plans over time to take into account new water quality issues. The most recent Bay-Delta Plan was published in December 2006. The CVRWQCB is currently revising the Bay-Delta plan, with proposed adoption in February 2013. The revised plan will include new flow objectives for protection of fish and wildlife beneficial uses, salinity, and other objectives for protection of agricultural beneficial uses, and a program of implementation, monitoring, and special studies. The most recent Basin Plan was revised in October 2011. These plans specify surface water quality objectives for a range of categories, including: bacteria, biostimulatory substances, chemical constituents, color, dissolved oxygen, floating material, methylmercury, oil and grease, pH, pesticides, radioactivity, salinity, sediment, settleable material, suspended material, tastes and odors, temperature, toxicity, and turbidity. The Bay-Delta Plan identifies additional requirements for chloride, salinity, dissolved oxygen, delta outflow, river flows, and export limits. These Bay-Delta Plan water quality objectives are intended to protect municipal, industrial, agricultural, and fish and wildlife beneficial uses. The Bay-Delta Plan requirements supersede those of the Basin Plan.

One mechanism that the CVRWQCB uses to implement the Bay-Delta and Basin Plans is a National Pollutant Discharge Elimination System (NPDES) permit. NPDES permits are issued to entities that discharge to waterways, known as point source dischargers. In the 2001, Headwaters, Inc. v. Talent Irrigation case, the Ninth Circuit Court of Appeals held that discharges of pollutants from the use of aquatic pesticides to waters of the United States required coverage under a NPDES permit (CVRWQCB 2006).

The DBW obtained an individual NPDES permit in March 2001, and operated under this permit until April 2006. In April 2006, the DBW applied to operate under the General NPDES Permit for the Discharge of Aquatic Pesticides for Aquatic Weed Control in Waters of the United States – General Permit No. CAG990005 (General Permit).

Following the Talent decision, there was some confusion regarding the need to obtain an NPDES permit for aquatic pesticide use. In November 2006, USEPA issued a regulation stating that application of a pesticide in compliance with relevant requirements of the Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA) does not require a NPDES permit when the application is made directly in waters to control pests in the water, or when the application of the pesticide is made to control pests that are over (or near) waters (Federal Register 2006). The rulemaking was based on USEPA's interpretation of the term "pollutant" under the Clean Water Act.

In theory, this regulation eliminated the need for a NPDES permit for WHCP. However, there were at least two legal challenges to this regulation, and SWB legal counsel recommended that SWB not rescind their general NPDES permits related to aquatic pesticides (SWB 2007). The USEPA ruling did mean that agencies operating under the General Permit had the option to terminate their coverage by the General Permit. The DBW elected to maintain coverage under the General Permit until legal challenges to the ruling were resolved. In January 2009, an appeals court vacated the USEPA rule that had allowed pesticides to be applied to U.S. waters without a NPDES permit. This ruling does not change WHCP operations because DBW maintained permit coverage.

The Bay-Delta Plan notes that "the Bay-Delta Estuary itself is one of the largest ecosystems for fish and wildlife habitat and production in the United States. Historical and current human activities (e.g. water development, land use, wastewater discharges, introduced species, and harvesting), exacerbated by variations in natural conditions, have degraded the beneficial uses of the Bay-Delta Estuary, as evidenced by the declines in populations of many biological resources of the Estuary" (SWB 2006).

Pollutants in Delta waterways include: pesticides (chlorpyrifos, DDT, diazinon, furan compounds, and Group A pesticides[2]), exotic species, mercury, salinity, dissolved oxygen, pathogens, and PCBs (CVRWQCB 2006). Potential sources of these pollutants include: agriculture, municipal point sources, urban runoff, storm sewers, resource extraction, and

[2] Group A pesticides include: aldrin, dieldrin, chlordane, endrin, heptachlor, heptachlor epoxide, hexachlorocyclohexane, endosulfan, and toxaphene.

hydromodification. More recently, concerns have been raised about ammonia levels in the Delta. CVRWQCB is working with researchers at San Francisco State University and University of California, Davis, to evaluate the impact of ammonia in the Delta (CVRWQCB 2008). All of the waters within the Delta are listed as impaired by at least one factor, either due to the presence of unacceptable levels of pollutants or lack of maintaining conditions such as adequate dissolved oxygen levels (USEPA 2011).

While evidence of gross pollution in the Delta has been largely eliminated, the recent rapid growth in population and industrial activity in tributary areas has left some problems unsolved and has created new ones. Existing water quality problems may be categorized as (1) eutrophication and associated dissolved oxygen fluctuations, (2) suspended sediments and turbidity, (3) salinity, (4) toxic material, and (5) bacteria.

Pesticides

Pesticides are found in the water and bottom sediments throughout the Delta. The more persistent chlorinated hydrocarbon pesticides are consistently found at higher levels than the less persistent organophosphate compounds. Sediments in the western Delta have the highest pesticide content. Pesticides have concentrated in aquatic life, but long-term effects and the effects of intermittent exposure are not known. There are now concerns about the aquatic toxicity of pyrethroid-based pesticides (bifenthrin, cyfluthrin, cypermethrin, and permethrin), which have replaced organophosphorus pesticides such as diazinon and chlorpyrifos. Pesticides are applied during specific seasons for specific crops, and there are five flushes of pesticides entering Delta waters: (1) first flush or dormant spray insecticides from orchards in December and January, (2) first flush of herbicides following the first winter storm, (3) spring insecticides applied in March and April, (4) spring and summer detection of rice pesticides, and (5) summer detection of pesticides applied to truck crops (Kuivila and Hladik 2008). Little is known about the potential for interactive toxicity from complex pesticide mixtures and/or pesticides interacting with other chemical, physical, or biological stressors (USEPA 2011).

Contaminants of Emerging Concern

Contaminants of emerging concern (CECs) include pharmaceuticals, personal care products, solvent stabilizers, flame retardants, pesticides, and other commonly used commercial and industrial compounds (USEPA 2011). There is growing concern, but little data, to adequately assess the ecological implications of CECs in the Delta. Some of these contaminants may be endocrine disrupting chemicals. Endocrine disrupting effects have been seen in silversides, delta smelt, and striped bass in the Delta; however, there are no specific linkages to contaminants in the Delta.

Eutrophication and Turbidity

Bacteriological quality, as measured by the presence of coliform bacteria, varies depending on the proximity to waste discharges and significant runoff. The highest concentration of coliform organisms is generally in the western Delta and near major municipal waste discharges.

The most serious enrichment in the Delta is due to a high influx of nutrients. Enrichment problems in the Delta occur along the lower San Joaquin River and in certain areas receiving waste discharges but having little or no net freshwater flow. These problems occur mainly in the late summer and coincide with low streamflow, high temperature, and the harvest season when fruit and vegetable canneries are in full operation. In addition to enrichment, deepening channels for navigation has further depressed dissolved oxygen levels to the point that at times levels are insufficient to support aquatic life. In the fall, these circumstances, combined with reverse flows due to export pumping, have created conditions unsuitable for salmon passage through the Delta to spawning areas in the San Joaquin Valley.

Warm, shallow, dead-end sloughs of the eastern Delta support populations of potentially toxic planktonic blue-green algae during the summer. Floating, semi-attached and attached aquatic plants such as water primrose *(Ludwigia peploides),* water hyacinth *(Eichhornia crassipes),* hornwort or coontail *(Ceratophyllum demersum),* eurasian milfoil *(Myriophylum spicatum), an*d *Egeria densa* frequently clog Delta waterways during summer. Extensive growth of these plants interferes with small boat traffic and contributes to the total organic load as these plants break loose and move downstream in the fall and winter.

Most Delta waters are turbid as a result of suspended silt, clay, and organic matter. Most of these sediments enter the Delta system with flow from major tributaries. Some enriched areas are turbid as a result of planktonic algal populations, but inorganic turbidity tends to suppress nuisance algal populations in much of the Delta. Continuous dredging to maintain deep channels for shipping also has contributed to turbidity and has been a significant factor in the temporary destruction of bottom organisms through displacement and suffocation.

Salinity

Salinity control is necessary in the Delta because it is contiguous with the ocean and its channels are at, or below, sea level. Unless repelled by continuous seaward flow of fresh water, ocean water will advance up the estuary and degrade water quality. During winter and early spring, flows through the Delta are usually above the minimum required to control salinity (described as "excess water conditions"). At least for a few months in summer and during the fall of most years, however, salinity must be carefully monitored and controlled for "balanced water conditions". The Central Valley Project and State Water Project monitor and control salinity, and salinity levels are regulated by the State Water Resources Control Board under its water right authority (through the Bay-Delta and Basin Plans). There are concerns that Delta salinity is increasing as more water is diverted through the SWP and CVP.

Salinity intrusion is a problem mainly during years of below-normal runoff, although in recent years with higher export levels, salinity has also been a concern. The degree of seawater intrusion into the Delta, and thus one source of salinity, is a result of

daily tidal fluctuations, freshwater inflow to the Delta from the Sacramento and San Joaquin Rivers, the rate of export at SWP and CVP intake pumps, and the operation of various control structures such as the Delta Cross-Channel Gates and Suisun Marsh Salinity Control System (USBR 2003).

In the eastern Delta salinity is largely associated with agricultural drainage and the high concentration of salts carried by the San Joaquin River. The Banks and Jones pumping plant operations draw high quality Sacramento River water across the Delta and restrict the low quality area to the southeastern corner. In areas such as dead-end sloughs, irrigation returns cause localized problems. In the western Delta, incursion of saline water from San Francisco Bay is one of the main water quality problems.

Trihalomethane Precursors

Another concern is that Delta water contains trihalomethane (THM) precursors. THMs are suspected carcinogens produced when chlorine used for disinfection reacts with natural substances during the water treatment process. Dissolved organic compounds that originate from decayed vegetation act as precursors by providing a source of carbon in THM formation reactions. During periods of reverse Delta flow, bromides from the ocean mix with Delta water at the western edge of Sherman Island. When bromides occur in water along with organic THM precursors, THMs are formed that contain bromine as well as chlorine. Drinking water supplies taken from the Delta are treated to meet THM standards, set at 0.080 mg/l, MRDL (maximum residual disinfectant level (USBR 2003). Contra Costa Water District (CCWD) reports that bromide in the Delta is 6.5 times above the national average (Taugher 2005). To reduce THM formation, CCWD has reduced the amount of chlorine used in their treatment process.

Sediment and Dissolved Oxygen

Sediment can either act as a sink or as a source of contamination depending on hydrological conditions and the type of habitat the sediment occurs in. Sediment provides habitat for many aquatic organisms and is a major repository for many of the more persistent chemicals introduced in surface waters. In the aquatic environment, most anthropogenic chemicals and waste materials including toxic organic and inorganic chemicals eventually accumulate in sediment (Ingersoll 1995). A more likely source of exposure of listed species to toxins in sediment is through the food chain, when fish feed on organisms that are contaminated with toxic compounds.

A sufficient level of DO is critical to the health and survival of aquatic species (CDFG 2011). Oxygen depletion can be caused by high water temperatures, the occurrence of decomposing aquatic vegetation, poor channel geometry, low streamflow, poor mixing of stream water with the atmosphere, and the presence of oxygen-depleting substances such as sewage, animal wastes, ammonia, organic nitrogen, and algae (CDFG 2011). Low DO levels in Delta locations, particularly the San Joaquin River and Stockton deep water shipping channel, impact migration of listed salmon. Over a

five-year period starting in August 2000, a DO meter recorded channel DO levels at Rough and Ready Island and found 297 days with violations of the 5 mg/l DO criteria for protection of aquatic life (NMFS 2012). Levels of DO below 5 mg/l have been reported as delaying or blocking fall-run Chinook salmon in studies conducted by Hallock et al. (1970).

As noted in DWR (2007) and NMFS (2012), water degradation or contamination can lead to either acute toxicity resulting in death when concentrations are sufficiently elevated, or more typically, when concentrations are lower, to chronic sublethal effects that reduce the physical health of the organism, and lessens it survival over an extended period of time. Mortality may become a secondary effect due to compromised physiology or behavioral changes that lessen the organism's ability to carry out its normal activities. For example, increased levels of heavy metals are detrimental to the health of an organism because they interfere with metabolic functions by inhibiting key enzyme activity in metabolic pathways, decrease neurological function, degrade cardiovascular output, and act as mutagens, teratogens or carcinogens in exposed organisms (Rand et al. 1995; Goyer 1996). For listed species, these effects may occur directly to listed fish or its prey base, which reduces the forage base available to listed species.

B. Cumulative Effects

Cumulative effects within the ESA include effects of future State or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the WHCP (50 CRF §402.2). This biological assessment discusses three areas with potential for cumulative effects.

1. Spongeplant Control Program

AB 1540 (Buchanan, Chapter 188, Statutes of 2012) was approved by the Legislature on August 15, 2012, and was signed by the Governor on August 27, 2012. AB 1540 adds responsibility for an additional invasive plant to DBW and USDA-ARS existing WHCP and EDCP programs. Spongeplant (*Limnobium laevigatum*) is a native of South America, Central America, and Central Mexico. It is a prolific, floating, flowering plant in the "frogbit" family (Hydrocharitacaea) (Anderson 2011). Spongeplant was first found in California in 2003 in small ponds near Arcata and Redding, and was discovered in the San Joaquin River in 2007.

Spongeplant is smaller than water hyacinth (leaves are 1 to 3 cm in diameter), but has several common characteristics. Spongeplant develops best in slow or still waters (including at the edge of fast-moving rivers); it is normally floating, but can also root in the mud (Akers 2010). Mats are extremely dense (2,500 plants per square meter) and can increase rapidly in size through vegetative reproduction, like water hyacinth. Spongeplant also reproduces by seed, increasing the potential for invasion.

Within the Delta, spongeplant is being found within water hyacinth mats. In these cases, treatment will occur concurrently with water hyacinth treatment (either by mechanical removal or chemical treatment). Glyphosate, 2,4-D, and penoxsulam are effective in treating spongeplant (Akers 2010). Spongeplant has also been found in irrigation canals where

water hyacinth is not typically found. Currently, a seven-person CDFA crew has been physically removing spongeplant infestations with excavation equipment.

The environmental impacts of spongeplant are expected to be similar, or perhaps potentially worse, than water hyacinth (Akers 2010). The density of the mats may seal the water's surface, lowering DO and denying open water to waterfowl. Spongeplant may also out-compete native plants. USDA-ARS is currently studying the characteristics of spongeplant in their Rapid Response program in order to better understand how to address this new invasive species.

The addition of spongeplant to DBW's aquatic weed program will still not provide the comprehensive and systematic approach to aquatic invasive plants in the Delta that would be preferred, but it will significantly expand DBW's responsibilities. Because spongeplant is closely aligned with water hyacinth, it is likely that these programs would at some point be integrated.

The CDFA's current approach of physically scooping spongeplant out of irrigation canals and ponds will not work in most of the Delta, thus a program that includes chemical treatment will likely be necessary. The DBW and USDA-ARS envision that a chemical approach could require an amendment to the WHCP Programmatic EIR and a similar consultation process as for the WHCP and EDCP. For this current WHCP consultation processes, the DBW and USDA-ARS are not considering spongeplant. However, there may be new program needs related to spongeplant as soon as January 1, 2013.

2. Climate Change

There have been numerous studies and modeling efforts to evaluate the impacts of climate change on the San Francisco Bay and Delta regions (Cloern et al. 2011, and Cayan et al. 2000, Climate Action Team 2010, DWR 2010, Field et al. 1999, Knowles 2008, Wagner et al. 2011, and USBR 2008, in National Research Council 2011). The Delta may be particularly affected by climate change because it is influenced by shifts in ocean conditions and changes in upland watersheds. The region has already seen patterns of increasing winter and spring air temperatures, decreasing contributions of snowmelt to annual precipitation, and a 2.2 cm per decade increase in mean sea level at the entrance of San Francisco Bay since the 1930s (Cloern et al. 2011). Projected changes, as modeled by Cloern et al. 2011, include:

- Increase in air temperature between 0.14 C to 0.42 C per decade
- Increase in sea level between 9.9 cm and 12.3 cm per decade
- Variable impacts on precipitation and unimpaired runoff, depending on the modeling tools
- Decrease in snow melt of between 0.4 percent and 1.1 percent per decade
- Increase in water temperature between 0.1 C and 0.3 C per decade in the Sacramento River and the Delta
- Increase in salinity of 0.33 to 0.46 psu per decade.

A number of other environmental indicators are also expected to change, increasing stress on native species. These include: an increase in number of days when

projected water temperature in the Delta exceeds 25 C (the threshold for high mortality of delta smelt), an increase in extreme water heights at the Golden Gate, an increase in number of days when temperatures in the Sacramento River exceed 16 C (a threshold for high mortality of salmonid eggs and pre-emergent fry), and a decrease in flood inundation of the Yolo Bypass (inundation is beneficial to Sacramento splittail and salmonids). Some climate change models indicate a drying trend in California, as well as an increase in frequency and magnitude of severe weather events (ICF International 2012).

These climate changes in the Delta will increase existing conflicts and policy debates regarding resource management, water supply, and land use. Cloern et al. predict that protection of native species will be even more challenging as environmental conditions in the Delta diverge from those to which native species are adapted. It is likely that sustaining populations of delta smelt will become increasingly difficult as Delta water temperatures, clarity, and salinity increase (Cloern et al. 2011). In an analysis of forty years of sampling data, Feyrer et al. (2011) found that climate change posed a serious threat to delta smelt.

Winter-run Chinook salmon may be particularly impacted because spawning occurs in the summer, when temperature increases will be greatest. Salmonids in California already experience temperature conditions at the edge of their tolerance because they are at the southern of their range (Katz et al. 2012). At the base of the food web, changes in temperature, flow, and salinity will further alter plankton and zooplankton communities, which may result in impacts at all levels of the food web. Changes in water supply and diversions will negatively impact listed fish species. Katz et al. (2012) suggest a lag effect whereby the cumulative impact of past actions such as clams habitat deterioration, and hatcheries will be amplified by climate change.

There is a growing body of literature assessing the relationship between climate change and invasive species (Hellmann et al. 2008, USEPA 2008, Masters and Norgrove 2010, Rahel and Olden 2008). Hellmann et al. identify five mechanisms by which invasive species impacts may be influenced by climate change:

- Altered mechanisms of transport and introduction
- Altered climatic constraints on native species
- Altered distribution of existing invasive species
- Altered impact of existing invasive species
- Altered effectiveness of management strategies for invasive species.

The specific impacts of climate change on a particular invasive species will vary, depending on the location and the characteristics of the species. However, generally, climate change is expected to improve conditions for invasive species (Masters and Norgrove 2010, USEPA 2008, Rahel and Olden 2008). This will further negatively impact native species. Katz et al. (2012) predict that climate change will increase non-native fish that prey on juvenile salmonids.

The impacts of climate change on water hyacinth are likely to be mixed. Increased air and water temperatures will improve the ability of water hyacinth plants to over-winter, and lengthen the growing season. This stimulatory effect is likely to outweigh the effect of higher salinity, which could reduce the range of water hyacinth within the Delta. In addition, biological controls such as the weevils *Neochetina bruchi* and *Neochetina eichhorniae* could become more effective. These weevils have not previously been effective controls in the Delta due to their inability to survive colder winter temperatures.

3. Increased Urbanization

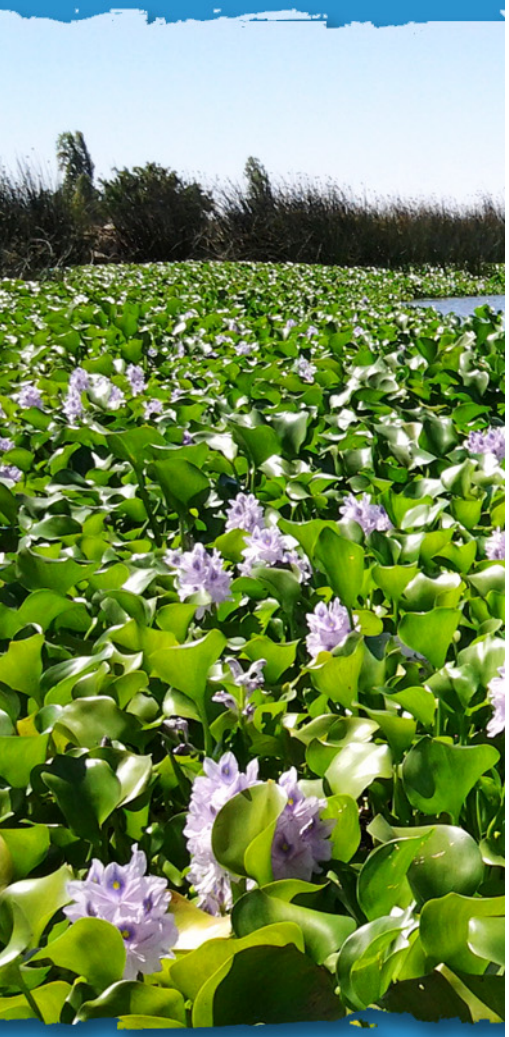
The Delta, East Bay, and Sacramento regions, which include portions of Contra Costa, Alameda, Sacramento, San Joaquin, Solano, Stanislaus, and Yolo counties, are expected to increase in population by nearly three (3) million people between 2007 and 2020. Increases in urbanization and housing developments can impact habitat by altering watershed characteristics, and changing both water use and stormwater runoff patterns. For example, the General Plans for the cities of Stockton, Brentwood, Lathrop, Tracy and Manteca and their surrounding communities anticipate rapid growth for several years to come. The anticipated growth will occur along both the I-5 and US-99 transit corridors in the east and Highway 205/120 in the south and west.

Increased growth will place additional burdens on resource allocations, including natural gas, electricity, and water, as well as on infrastructure such as wastewater sanitation plants, roads and highways, and public utilities. Some of these actions, particularly those which are situated away from waterbodies, will not require Federal permits, and thus will not undergo review through the ESA Section 7 consultation process (NMFS 2012). Feyrer's analysis of fish sampling data noted that increased water demand due to urbanization was a serious threat to delta smelt (Feyrer et al. 2011).

Increased urbanization is also expected to result in increased recreational activities in the region. Among the activities expected to increase in volume and frequency is recreational boating. Boating activities typically result in increased wave action and propeller wash in waterways. Boating activities can degrade riparian and wetland habitat by eroding channel banks and mid-channel islands, thereby causing an increase in siltation and turbidity. Wakes and propeller wash also churn up benthic sediments thereby potentially resuspending contaminated sediments and degrading areas of submerged vegetation. This in turn could reduce habitat quality for the invertebrate forage base required for the survival of juvenile salmonids and green sturgeon moving through the system. Increased recreational boat operation in the Delta is also anticipated to result in more contamination from the operation of gasoline and diesel powered engines on watercraft entering bodies of the Delta (NMFS 2012).

Section 6

Effects of the Action



6. Effects of the Action

The purpose of this section is to analyze the potential impacts of WHCP on listed species and/or critical habitats. Effects refer to both stressors (negative impacts) and subsidies (positive impacts) of the action. This analysis includes an assessment of: direct and indirect effects, including conservation and minimization measures, and the effects of the action on species when added to the environmental baseline and cumulative effects in the action area.

The effects of an action depend on the potential impact of the particular stressors of the action, and the presence of listed species and/or critical habitat within the action area. A particular species may be affected by an action, but if they are not present when the action occurs, there is no potential for an adverse effect. Conversely, a species may be present, but the effect of the action may be insignificant, so again there is no potential for an adverse effect. When the species is present within the action area, and the stressor affects the species, there is potential for an adverse effect.

Thus, to analyze the potential impacts of WHCP on listed species and/or critical habitats, this section first summarizes the presence of listed species within the action area, and then considers the potential direct and indirect effects of WHCP actions. This section is organized as follows:

A. Listed Species in the Action Area
B. Overview of WHCP Stressors
C. Direct Effects of WHCP
D. Indirect Effects of WHCP
E. Direct and Indirect Effects of Interrelated or Interdependent Actions
F. Effects Considering Environmental Baseline and Cumulative Effects
G. Subsidies of WHCP
H. Alternative Actions.

A. Listed Species in the Action Area

The potential exists for impacts to occur to native and listed fish species under the WHCP, since these fish do occur in the general project area, whether or not they occur in water hyacinth beds specifically. This section discusses the potential for exposure of special status and other fish to WHCP treatments.

Although not specific to water hyacinth beds, the Stockton Fish and Wildlife Office of the USFWS conducts an annual monitoring program for juvenile

Delta fisheries. The focus is on Chinook salmon, however the program identifies, tracks, and monitors all fish species sampled at several beach seine and trawl locations. These studies provide time-series data on fish abundance and assemblages in six Delta regions, and support previous findings that the most abundant fish species captured in the Delta are non-indigenous (Hanni, 2005).

Recently published studies analyzing historical fish survey data (Grimaldo et al. 2012, Sommer et al. 2011) have evaluated native fish abundance and location within the Delta. Grimaldo et al. (2012) evaluated spatial and temporal distribution of fish at a reference location and three restored marshes between April 1998 and July 1999. Only 2 percent of the 47,000 fish found were native species, including only 202 Chinook salmon and ten delta smelt. Introduced fish, especially centrarchid fishes, were abundant in submerged aquatic vegetation, while native fish were more abundant in tidal sloughs.

Study results in 2005 (the last year for which summary reports are available), for the monitoring period May 1 through August 31 (coinciding with WHCP activity) captured a total of 56,793 fish and 51 different species (Hanni, 2005). Although over fifty different species were captured in total, a small number of species made up the majority of fish. Between one and six species made up at least 75 percent of the sample in each region (Hanni, 2005). The most abundant fish captured were introduced inland silversides and red shiners, each 27 percent of the total. The most commonly captured native fish were Sacramento suckers (8 percent), and Sacramento splittail (2 percent). Fish assemblage stability measured between May and August from 1995 to 2005 was moderately stable in most regions, and most stable in the Lower Sacramento River region (Hanni, 2005). Fish diversity during the same time period showed a declining trend, except in the South Delta, although data is highly variable and it is difficult to make definitive inferences (Hanni, 2005).

Consistent with these trends, 2000 to 2006 beach seine surveys found an increase in non-native fish and a decrease in native fish (Hanni and Chapman 2006). More recent studies show a significant shift in the make-up of fish species in the Delta, with fewer native species. California Department of Fish and Game's (CDFG) Resident Fish Survey in 1981/1982 found 18 percent natives and 35 percent bass and sunfish, while a University of California, Davis study in 2009/2010 found only 4 percent natives and 74 percent bass and sunfish (Conrad et al. 2010b).

Location of fish species within the Delta also influences the potential for exposure to WHCP treatment herbicides. Sommer's evaluation of migration patterns of delta smelt (Sommer et al. 2011) found that delta smelt are present year-round at Cache Slough (a site in which only six (6) of 308 acres has been treated for water hyacinth over the last five years). Delta smelt migration from Suisun Marsh and the western edge of the Delta to upstream spawning sites typically begins after the first winter storms bring the "first flush" of freshwater. While migrating smelt reach upstream spawning areas within a month, they typically hold in those areas until they begin spawning between late February and May. Delta smelt were not found in the southern

Delta during summer months (Sommer et al. 2011), and were more abundant in Sacramento River fish survey sites than San Joaquin River survey sites (Nobriga et al. 2005).

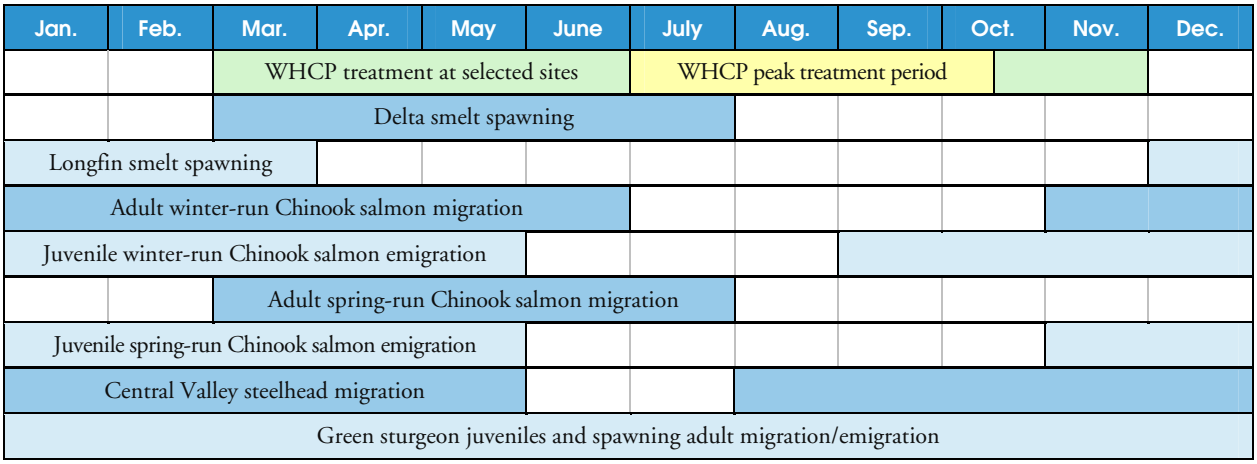
Special status and native fish species may not commonly be present near water hyacinth, further reducing risk of exposure to WHCP chemicals. Toft et al., (2003) sampled fish adjacent to water hyacinth, and found that most of the fish were juveniles, and non-indigenous to the Delta. Three native species, Sacramento splittail, tule perch, and prickly sculpin accounted for only 8.2 percent of the fish captured at one Delta site (Toft 2003). Turbidity (as measured by Secchi depth) and specific conductivity are good predictors of delta smelt occurrence (Feyrer et al. 2007). Delta smelt are more likely to be found in areas with higher turbidity levels, and relatively lower specific conductivity (lower saline levels). At any particular fish survey time and location, delta smelt are primarily found in offshore habitats (Nobriga et al. 2004, Nobriga et al. 2005). Similarly, Chinook salmon are most common in open water shoals (Grimaldo et al. 2012).

DBW will conduct WHCP treatments between March (selected areas only) and November, with the majority of treatments likely in June through September. The WHCP treatment period also coincides temporarily with the migration and emigration of certain runs of Chinook salmon through the Delta, and the presence of delta smelt, longfin smelt, and green sturgeon in the Delta. **Figure 6-1,** on the next page, summarizes the timing of listed fish presence in the Delta based on descriptions in Section 4. Survey data indicate that it is not likely that fish will be directly under WHCP mats. Given the locations of WHCP treatments and listed fish spawning and migratory patterns, it is possible that some listed fish may be present near WHCP treatment sites. Thus, individual listed fish may, on occasion, be present near WHCP treatment sites, with potential for short-term exposure to WHCP treatment chemicals in receiving waters or from overspray.

In early WHCP documentation, the USFWS considered the potential impact of WHCP treatments on special status reptiles:

> *"The concentration of Weedar or Rodeo [equivalent to AquaMaster] used on water hyacinth is not known to be toxic to reptiles (Van Way 1995), and direct exposure of giant garter snakes to these herbicides is unlikely. Giant garter snakes bask on grassy banks and on branches over the water's edge where herbicide applications will not occur. The giant garter snake is extremely shy and snakes in the water or on top of water hyacinth mats would probably move out of the area as the boat crews approach in motor driven boats. Emergent vegetation is used by adults for escape cover and for foraging habitat, and young use dense emergent vegetation for cover while absorbing their yolk sacks. Water hyacinth herbicides are applied only to water hyacinth and will not affect emergent vegetation or snakes utilizing emergent vegetation. The small potential adverse effect herbicide application could have on any giant garter snakes present is likely to be greatly outweighed by the benefit of water hyacinth removal on the species' habitat. Open water surface is a habitat requisite for this species (USFWS 1993). Water hyacinth infestations inhibit giant garter snakes from foraging and are reducing the numbers of prey species.*

Figure 6-1
Proposed Period of WHCP Treatments; Periods of Peak Spawning in the Delta; and Migration and Emigration of Special Status Fish Species through the Sacramento-San Joaquin River System



It is unlikely that northwestern pond turtles would be directly exposed to the herbicides applied on the water hyacinth. Pond turtles are wary, and will quickly leave basking sites when approached. Water hyacinth control would benefit northwestern pond turtles on the Refuge by increasing food availability. Removal of water hyacinth mats would lead to an increase in the abundance and diversity of macroinvertebrates, tadpoles, small fish, and both submergent and floating native plant species" (USFWS 1995, 5-6).

B. Overview of WHCP Stressors

The WHCP consists of an integrated management approach to control invasive water hyacinth in the Delta and its tributaries. The program consists primarily of chemical treatment, supported by limited handpicking, herding, and mechanical removal, and assessment of biological controls. Most of the potential WHCP stressors result from chemical treatments. Below, we briefly discuss other potential stressors and describe potential for adverse effects.

- Handpicking and herding consists of treatment crews manually removing the water hyacinth. The small scale and low-impact nature of these activities will not adversely affect listed species or critical habitat
- Biological controls consist of small experimental application of approved biological control species that are placed in select locations within the Delta. Approved control species selectively affect water hyacinth, and will not adversely affect listed species or critical habitat
- Mechanical removal with small excavators at boating ramps consists of infrequent removal of dense water hyacinth mats that are located at or near boat ramps. Equipment is only placed on concrete boat ramps, and the activity will not adversely affect listed species or critical habitat
- Mechanical removal with specialized aquatic equipment consists of specialized cutting boats and conveyor equipment to remove dense water hyacinth mats. This approach has the potential for direct effects on listed species due to the mechanics of the conveyor belt systems. The potential effects of mechanical removal with specialized aquatic equipment will be evaluated below

- Chemical treatment consists of treatment crews spraying approved herbicides and adjuvants directly on water hyacinth plants. This approach has the potential for direct effects to listed species due to toxic effects on fish, effects on water quality, bioaccumulation of herbicides, and disturbance by treatment crew boats. Chemical treatment has the potential for indirect effects to listed species and critical habitats due to loss of native plants, food web effects, and low dissolved oxygen. The potential effects of chemical treatment will be evaluated below.

C. Direct Effects of WHCP

Direct effects are caused by the action and occur at the time of the action.

1. Toxicity of Herbicides and Adjuvants to Listed Species

The potential for direct toxic affects to listed species depends on the proximity of listed species, concentrations of herbicides, and length of exposure.

Concentrations of WHCP Herbicides and Adjuvants in, and Adjacent, to Treatment Sites and Water Quality Effects

There are two factors to evaluate related to herbicide concentrations following WHCP treatments: (1) the concentration as it relates to NPDES guidelines and Basin Plan limits to maintain water quality, and (2) the concentration as it relates to toxic levels. The WHCP has been monitoring 2,4-D, glyphosate, and adjuvant concentrations following WHCP treatments since the program's inception in 1983. Section 3 summarizes recent monitoring data for 2,4-D, glyphosate, and Agridex. Because the three potential new WHCP herbicides have not been utilized in the Delta, to date, there is no prior data. Section 3 also provides calculated concentrations of 2,4-D, glyphosate, and the three new WHCP herbicides, penoxsulam, imazapyr, and imazamox, as well as the two adjuvants. These calculations are based on the highest herbicide application rate, assuming 20 percent overspray, and one or two meter(s) deep water. These assumptions represent conservative and instantaneous concentrations. The amount of herbicide applied in the project area to control water hyacinth (and thus the resulting concentrations of herbicide in Delta waters) can be further minimized by treating water hyacinth plants early in the growing season before plants have spread and grown into large mono-specific mats characteristic of this species. Early treatment will also minimize the negative ecosystem effects of large mono-specific water hyacinth mats.

In reality, mixing of any herbicide that reaches the water occurs through the entire depth of water at the site, and tidal movement and through water Delta flow dilutes herbicides even further. The Delta is not a stationary water environment, thus, the concentration of herbicide immediately after treatment is not stable, but rather readily dilutes (in addition to degradation pathways). There are two tidal cycles in the Delta every day, with typical water fluctuations of three to five feet in each cycle. In addition, the Delta functions in a complex hydrological system consisting of inflows from rivers and reservoirs, Delta exports, and tidal fluctuations.

Approximately 30 km^3 of freshwater enter the Delta (and then San Francisco Bay) annually, with peak flows in early March (Knowles 2000). Freshwater inflows and Delta exports are the major influences of salinity in the Delta. Illustrating the movement of water

within the Delta, the X2 salinity line (distance of the near-bottom 2 psu isohaline line from the Golden Gate) varies by up to 30 km during the course of a year (Knowles 2000).

Historical water quality monitoring data demonstrates that actual herbicide concentrations decrease rapidly in the Delta following treatment. Water samples taken downstream of the treatment site, at two to three feet depth one-hour post treatment, show actual herbicide levels that are at least an order of magnitude below the calculated concentrations in 1 meter of water in Section 3.

USEPA's standard ecological assessment approach to evaluate the potential for toxic effects on terrestrial and aquatic animals and plants is based on comparing a calculated risk quotient (RQ) to specified levels of concern (LOC). The RQ is equal to the water chemical concentration divided by an acute or chronic toxicity value: RQ = Exposure/Toxicity. Protocol requires using the lowest available toxicity values in the scientific literature in order to ensure that RQ values are conservative.

LOC's are unit-less values determined by the USEPA's Office of Pesticide Programs. When the RQ is higher than the specified LOC, it is an indication of the need for further investigation of that particular chemical application. **Table 6-1,** below, provides the USEPA's LOCs. USEPA's interpretation of LOC risks is as follows (USEPA 2007):

- Acute high risk: potential for acute risk is high; regulatory action may be warranted in addition to restricted use classification
- Acute restricted use: the potential for acute risk is high, but this may be mitigated through restricted use classification
- Acute endangered species: the potential for acute risk to endangered species is high, but this may be mitigated through restricted use classification
- Chronic risk: the potential for chronic risk is high; regulatory action may be warranted.

This Biological Assessment utilizes historical and calculated herbicide concentrations to determine compliance with NPDES water quality standards and USEPA risk calculations. **Table 6-2,** on the next page, provides the maximum concentration of herbicide within one hour of treatment for purposes of calculating WHCP risk quotient (RQ) values and NPDES water limits or monitoring triggers for WHCP herbicides.

Table 6-1
Aquatic Animal Levels of Concern

Risk Presumption	Risk Quotient[1]	Level of Concern
Acute High Risk	EC/LC50 or EC50	0.5
Acute Restricted Use	EC/LC50 or EC50	0.1
Acute Endangered Species	EC/LC50 or EC50	0.05
Chronic Risk	EC/MATC or NOEC	1

Source: USEPA, 2007.

[1] LC50 is the lethal concentration for 50 percent of the test species. EC50 is the effective concentration (for a defined endpoint) for 50 percent of the target. MATC is the maximum acceptable toxicant concentration. NOEC is the Non-observable effect concentration.

Table 6-2
Maximum Active Ingredient Concentrations for WHCP Herbicides For RQ Calculations and NPDES Maximum Limitations

Herbicide Active Ingredient	Concentration for RQ Calculation	NPDES Maximum Limitation (in receiving waters)
2,4-D	400 ppb*	70 ppb
Glyphosate	158 ppb	700 ppb
Penoxsulam	2 ppb	10.1 ppm
Imazapyr (isopropylamine salt)	5.6 ppb	11.2 ppm
Imazamox	11.2 ppb	NA
Agridex	0.25 ppb	None
Competitor	0.25 ppb	None

* This one-time measured 2,4-D concentration is higher than the NPDES limitation, but was taken under the mat, not in receiving waters. 2,4-D levels in receiving waters have not exceeded NPDES levels. The 400 ppb (rounded) measure was an outlier, representing just one of over 100 sampling events taken between 2001 and 2005. The highest measured 2,4-D level since 2005 was 30 ppb, and this measure was also an outlier, representing one of 62 sampling events.

There is currently no trigger for imazamox, as this herbicide was recently approved for aquatic use in California. The SWRCB will develop a monitoring trigger or water limit for this herbicide, most likely before the start of the 2013 treatment season. The WHCP will not utilize imazamox until a monitoring trigger has been specified.

The RQ concentrations for 2,4-D and glyphosate are conservative outlier maximums, as they represent one-time high herbicide levels found immediately after treatment under the water hyacinth mat. The 2,4-D 400 ppb concentration and the glyphosate 158 ppb concentration occurred in just two of over 100 sampling events between 2001 and 2005. All other monitoring levels of 2,4-D and glyphosate were significantly lower. The RQ levels for penoxsulam, imazapyr, and imazamox are based on the calculated maximum concentration in one-meter deep water immediately following treatment, assuming 20 percent overspray.

The RQ concentrations in Table 6-2 represent maximum short-term exposure estimate concentrations (EEC). The WHCP complies with NPDES guidelines and basin plan limits to maintain water quality, thus water quality is not adversely affected by the program. WHCP activities are intended to maintain and improve beneficial uses of Delta waters.

The remainder of this subsection summarizes toxicity data and RQ calculations for each of the current and potential new WHCP treatment herbicides.

Toxicity of 2,4-D to Listed Fish Species

Between 2001 and 2005, DBW commissioned toxicity testing of three fish species. The testing included water samples obtained following treatments. In addition, as part of their NPDES permit requirement, DBW sponsored several toxicity analyses using WHCP chemicals. These studies are indicative of actual environmental impacts, as they reflect Delta conditions, and/or

laboratory results specifically related to WHCP. Below, we summarize results of these studies, as they relate to toxic impacts on fish species:

- Riley and Finlayson (2003) conducted 96-hour acute toxicity screening for 2,4-D on larval delta smelt, larval Sacramento splittail, and larval fathead minnows. The results of these studies are provided in **Table 6-3,** below. The study concluded that 2,4-D toxicity values for the three larval fish species were several orders of magnitude higher than detected concentrations in the environment (Riley and Finlayson 2003)
- Riley and Finlayson (2004) conducted 96-hour and seven day toxicity screening of WHCP chemicals on larval fathead minnows to determine chronic toxicity levels. For 2,4-D, the 96-hour LC50 value was 116 ppm, the seven day LC50 was 96.6 ppm, and the seven day maximum acceptable toxicant concentrations (MATC) was less than 40.5 ppm. These concentrations were orders of magnitude higher than concentrations resulting from WHCP.

Table 6-4, on the next page, summarizes fish toxicity data for 2,4-D. DBW conducted an analysis of water quality and toxicity using monitoring data gathered from 2001 to 2005. DBW collected several hundred pre-treatment and post-treatment water samples and delivered these to California Department of Fish and Game laboratories to conduct five different toxicology tests. Based on an examination of toxicology test results from post-treatment water samples, WHCP did not have a significant or consistent adverse effect on test organisms used by the laboratories (including fathead minnow)

In DBW's analysis, there were 20 samples which exceeded previous NPDES permit levels (20 ppb) for 2,4-D (NPDES permit levels are now 70 ppb 2,4-D). These 20 samples were tested for fathead minnow survival and growth. None of these 20 samples had an adverse effect on survival, however five samples had an adverse effect on fathead minnow growth.

This series of studies provide no indication of acute toxic impacts on fish species as a result of WHCP treatments. All toxicity tests were conducted on the more sensitive larval stages of fish, providing further confidence in the results. While data are limited, there may be some impact of WHCP treatments (and/or simply from ambient Delta waters) on larval fish growth. However, it is not clear whether 2,4-D, or other contaminants in Delta waters affected growth.

In an independent study of aquatic pesticide toxicity within the Delta, the San Francisco Estuary Institute (SFEI) conducted the Aquatic Pesticide Monitoring Program (APMP) (Siemering et al. 2008). The APMP, funded by the SWB, was part of the settlement of the 2001 *Headwaters, Inc. v. Talent Irrigation District* decision regarding the requirement to obtain an NPDES permit for aquatic pesticide use. The purpose of the APMP was to evaluate water quality impacts associated with the use of aquatic pesticides, and to evaluate non-chemical alternatives.

Table 6-3
CDFG Study Results, Acute Toxicities of 2,4-D on Three Larval Fish Species, 96-Hour LC50 Values (in ppm)

Fish Species	2,4-D LC50
Larval delta smelt	149 ppm
Larval Sacramento splittail	446 ppm
Larval flathead minnow	216 ppm

Table 6-4
Response of Various Fish Species to 2,4-D at LC50 Values

Species	Chemical	LC50	Time Period	Reference
Fathead minnow	2,4-D dimethylamine salt (DMA)	344 ppm	96 hr	Alexander et al., 1985
Fathead minnow	2,4-D DMA	335 ppm	96 hr	Johnson and Finley 1980
Fathead minnow	2,4-D DMA	318 ppm	96 hr	USEPA 2000
Fathead minnow fingerlings, swim-up fry	2,4-D DMA	320 ppm to 630 ppm	96 hr	Johnson and Finley 1980
Fathead minnow egg stage	2,4-D DMA	1,400 ppm	96 hr	Johnson and Finley 1980
Bluegill	2,4-D DMA	168 ppm	96 hr	Johnson and Finley 1980
Bluegill	2,4-D DMA	524 ppm	96 hr	Alexander et al., 1985
Bluegill	2,4-D DMA	166 ppm to 458 ppm	48 hr	HSDB 2001
Bluegill	2,4-D DMA	108 ppm to 524 ppm	96 hr	USEPA 2000
Juvenile rainbow trout	2,4-D DMA	494 ppm	96 hr	Fairchild et al. (2009)
Rainbow trout	2,4-D DMA	>100 ppm	96 hr	Johnson and Finley 1980
Rainbow trout	2,4-D DMA	250 ppm	96 hr	Alexander et al., 1985
Rainbow trout, Donaldson trout	2,4-D DMA	250 ppm	96 hr	USEPA 2000
Rainbow trout, Donaldson trout	2,4-D DMA	100 ppm to 1,360 ppm	96 hr	ECOTOX 2001
Cutthroat trout	2,4-D granular	64 ppm	96 hr	Johnson and Finley 1980
Lake trout	2,4-D granular	45 ppm	96 hr	Johnson and Finley 1980
Chinook salmon*	2,4-D DMA	>100 ppm	96 hr	Johnson and Finley 1980
Coho salmon yearling	2,4-D DMA	>200 ppm	96 hr	HSDB 2001
Nile tilapia larvae	2,4-D DMA	28 ppm	48 hr	Sarikaya and Selvi 2005
Nile tilapia adults	2,4-D DMA	87 ppm	48 hr	Sarikaya and Selvi 2005
Channel catfish	2,4-D DMA	155 ppm	96 hr	Johnson and Finley 1980
Smallmouth bass	2,4-D DMA	236 ppm	96 hr	Johnson and Finley 1980
Largemouth bass	2,4-D DMA	350 ppm to 375 ppm	48 hr	HSDB 2001

* Study utilized in RQ calculation.

For 2,4-D, the RQ values for Chinook salmon LC50, *Pimephales promelas* (fathead minnow) LC50, and delta smelt NOEC were all well below the LOC values. SFEI stated "this data indicates that there is no evidence of pesticide induced degradation at either of the sampling locations. In addition, no LOCs were exceeded by the maximum 2,4-D concentration measured" (Siemering et al. February 2005).

In another study, SFEI analyzed DBW WHCP monitoring results, calculating RQ

values and the number of LOC exceedances for monitoring data from 2003 to 2005. For the 1,799 2,4-D RQs that SFEI calculated for the three year period, there were no LOC exceedances.

Fairchild et al. (2009) conducted an ecological risk assessment of the exposure and effects of 2,4-D acid to rainbow trout. Fairchild identified an acute toxicity LD50 for juvenile rainbow trout of 494 ppm. In a test of 30-day chronic toxicity, Fairchild found no effects on juvenile rainbow trout at the maximum exposure of 108 ppm. In a test of 30-day chronic toxicity in the more sensitive rainbow trout swim-up larvae, Fairchild found a no observable effect level (NOEC) of 54 ppm, a lowest observable effect level (LOEC) of 108 ppm, and a maximum acceptable toxicant concentration (MATC) of 76 ppm. Length and weight were the chronic toxicity endpoints in these studies. All of these levels are well above WHCP treatment concentrations. Fairchild also examined environmental exposure levels, and concluded that using 2,4-D for invasive weed control in aquatic and terrestrial habitats poses no substantial risk to growth or survival of rainbow trout or other salmonids.

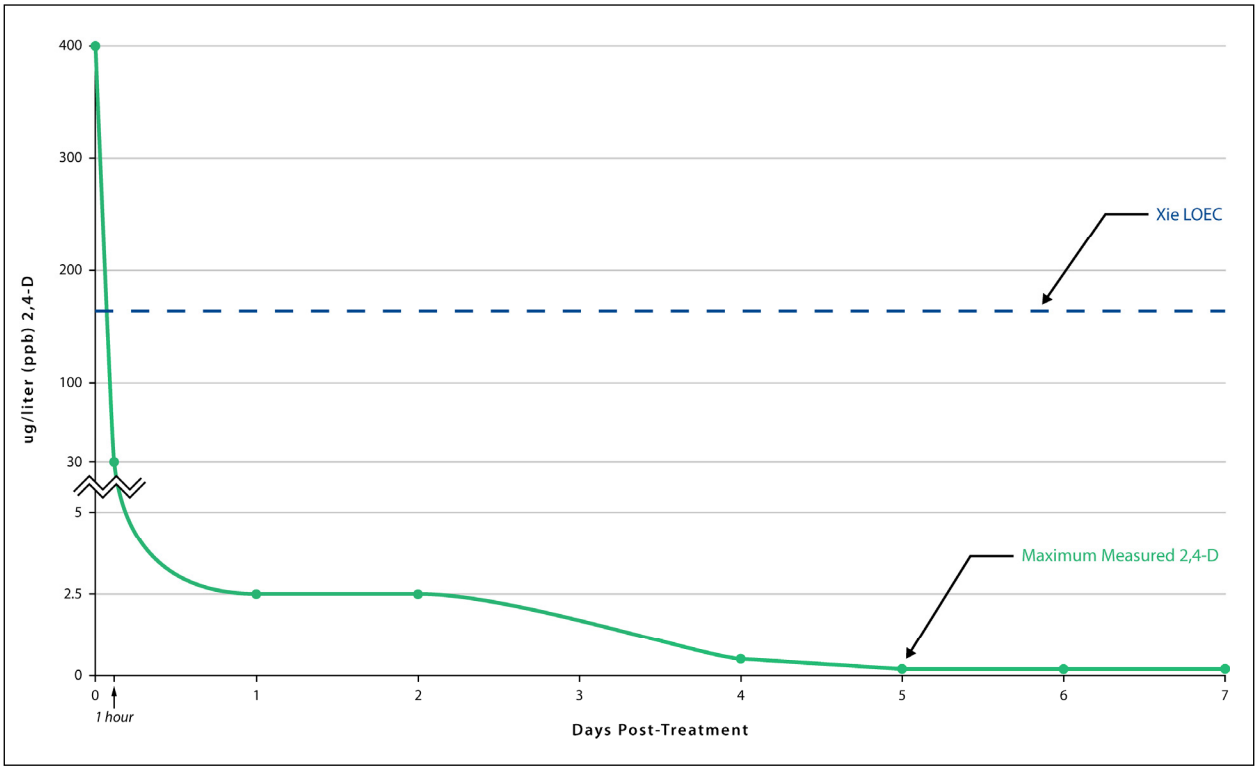
While the risk of acute toxicity to special status or other fish resulting from the WHCP is extremely low, there is some evidence of chronic/sublethal toxicity impacts from 2,4-D. Studies have identified two potential areas of concern related to sublethal exposure to 2,4-D: endocrine disruption (in the form of estrogenic activity) and oxidative stress.

Xie et al., (2005) identified dose-related increases of vitellogenin in juvenile rainbow trout exposed to 2,4-D. Vitellogenin is an egg yolk precursor protein used as an indicator of estrogenic activity in both females and males. Juvenile trout were exposed to either 0.00164, 0.0164, 0.164, or 1.64 mg/l 2,4-D (ppm) for seven days. The trout exposed at the 1.64 mg/l level had vitellogenin levels 93 times higher than the controls. The lowest observed effect concentration (LOEC) or lowest observed adverse effect concentration (LOAEC) was 0.164 mg/l (or 164 ppb). There was no observed effect at the lowest two exposure concentrations.

The endocrine disruption LOEC for 2,4-D of 164 micrograms per liter (ppb) was based on an exposure of seven days at this LOEC level (Xie et al. 2005). While the maximum in-treatment site measurement for 2,4-D was just under 400 ug/l (ppb), in one outlier case out of more than 100 samples taken between 2001 and 2005, this level of herbicide is not maintained in Delta waters. The maximum 2,4-D level found one hour post-treatment over six years of monitoring (2006 through 2011) was 30 ppb. Thirty-nine percent of 2,4-D samples taken one hour after treatment were less than 1 ppb. 2,4-D levels found between one and seven days post-treatment range from non-detectable to 2.5 ppb.

Figure 6-2, on the next page, illustrates the Xie study LOEC level as compared to actual maximum 2,4-D levels found following WHCP treatments. Figure 6-2 is conservative, because it utilizes the highest levels of 2,4-D found following treatment, not the average levels, which are lower. As Figure 6-2 illustrates, the WHCP will not result in 2,4-D concentrations that exceed the LOEC levels for a long-enough period to result in sublethal impacts on estrogenic activity.

Figure 6-2
Comparison of Measured 2,4-D Levels Post-Treatment with LOEC for Estrogenic Activity from Xie et al.



Sarikaya et al., (2005) examined 48 hour LC50 values for 2,4-D in larvae and adult Nile tilapia (*Oreochromis niloticus*). They observed changes among larvae and adults at various herbicide levels, and concluded that the toxicity of 2,4-D is related to oxidative stress. Behavioral and other changes included abnormal swimming behavior (hitting the walls of the tank), increased mucous secretion, faded coloring, sudden jerks, and anxiety.

Oruc and others (2000, 2002, 2004) examined antioxidant enzymes in carp and tilapia following exposure to 2,4-D. Oxidative stress results in the formation of free radicals, which cause cellular damage. Formation of free radicals also results in increased production of antioxidant enzymes, which can be measured in the laboratory. Carp and tilapia exposed to 87 ppm 2,4-D for 96 hours showed an increase in the antioxidant enzyme superoxide dismutase (SOD) in gills (but not kidney or brain). Oruc concluded that fish exposed to 2,4-D developed tissue-specific adaptive responses to protect cells against oxidative stress.

These studies raise potential concerns about sublethal toxicity, however the exposure levels of 2,4-D that resulted in estrogenic activity or oxidative stress in fish are higher than those likely to result from WHCP.

Table 6-5
RQ Calculations for 2,4-D

Species	EEC/LC 50	RQ
Chinook salmon	0.4 ppm/ >100 ppm	0.004 (acute)
Larval delta smelt	0.4 ppm/ 149 ppm	0.003 (acute)
Rainbow trout swim-up larvae	0.4 ppm/ 54 ppm	0.007 (chronic)*

* The 0.4 ppm concentration used for the chronic toxicity exposure is extremely conservative, as this was an instantaneous maximum exposure found in one outlier sample taken under a water hyacinth mat.

Table 6-6
CDFG Study Results, Acute Toxicities of Glyphosate on Three Larval Fish Species, 96-Hour LC50 Values

Fish Species	Glyphosate LC50
Larval delta smelt	270 ppm
Larval Sacramento splittail	1,132 ppm
Larval flathead minnow	1,154 ppm

Table 6-5, above, provides the RQ calculations for 2,4-D, using Chinook salmon and larval delta smelt. For a finding of no adverse acute effect on an endangered species, the RQ value should be below a LOC of 0.05. For no chronic risk, the RQ should be below the LOC of 1, utilizing a no observable effect concentration (NOEC) or maximum acceptable toxicant concentration (MATC). As Table 6-5 illustrates, the RQ values for both species are well below the endangered species and chronic risk LOC values. The already low potential for toxicity effects of 2,4-D can be further minimized by treating water hyacinth early in the growing season, thus reducing the amount of herbicide needed.

Toxicity of Glyphosate to Listed Fish Species

DBW's commissioned toxicity testing of three fish species included glyphosate, as well as 2,4-D. The testing included water samples obtained following treatments. In addition, as part of their NPDES permit requirement, DBW sponsored several toxicity analyses using WHCP chemicals. These studies are indicative of actual environmental impacts, as they reflect Delta conditions, and/or laboratory results specifically related to the WHCP. Below, we summarize results of these studies, as they relate to toxic impacts of glyphosate on fish species:

- Riley and Finlayson (2003) conducted 96-hour acute toxicity screening for glyphosate on larval delta smelt, larval Sacramento splittail, and larval fathead minnows. The results of these studies are provided in **Table 6-6,** left. The study concluded that glyphosate toxicity values for the three larval fish species were several orders of magnitude higher than detected concentrations in the environment (Riley and Finlayson 2003)
- Riley and Finlayson's (2004) testing of glyphosate on larval fathead minnows found a 96-hour LC50 value of 608 ppm, a seven day LC50 of 586 ppm, and a seven day MATC of less than 104 ppm. Again, these concentrations were orders of magnitude higher than concentrations resulting from the WHCP. Riley and Finlayson concluded that there were minimal impacts to fish and wildlife from WHCP.

DBW conducted an analysis of water quality and toxicity using monitoring data gathered from 2001 to 2005. DBW collected several hundred pre-treatment and post-treatment water samples and delivered these to California Department of Fish and Game laboratories to conduct five different toxicology tests. Based

on an examination of toxicology test results from post-treatment water samples, WHCP did not have a significant or consistent adverse effect on test organisms used by the laboratories (including fathead minnow).

In DBW's analysis, none of the glyphosate samples exceeded NPDES permit criteria (700 ppb), the CDFG laboratory conducted toxicity testing using the 18 samples with detectable levels of glyphosate. None of these 18 glyphosate samples had an adverse effect on fathead minnow survival, however three of the 18 samples had an adverse effect on fathead minnow growth. (Three of 52 samples without any detectable glyphosate also had an adverse effect on fathead minnow growth).

This series of studies provide no indication of acute toxic impacts on fish species as a result of WHCP treatments. All toxicity tests were conducted on the more sensitive larval stages of fish, providing further confidence in the results. While data are limited, there may be some impact of WHCP treatments (and/or simply from ambient Delta waters) on larval fish growth.

SFEI's study evaluated glyphosate as well as 2,4-D. In the APMP, SFEI prioritized aquatic pesticides for further study, analyzed three years of monitoring data, and conducted several special studies of high priority pesticides. Using an USEPA methodology, SFEI calculated risk quotients (RQ) for each pesticide.

For glyphosate, there were also no LOC exceedances. Of the eight aquatic pesticides evaluated, SFEI ranked glyphosate as the lowest risk (Siemering et al. 2008).

In another study, SFEI analyzed DBW WHCP monitoring results, calculating RQ values and the number of LOC exceedances for monitoring data from 2003 to 2005. For the 835 RQs that SFEI calculated for glyphosate, there were four LOC exceedances (one for delta smelt and three for Sacramento splittail). SFEI hypothesized that the small number of exceedances could result from overapplication, poor mixing and dispersion in the water column, or additional terrestrial sources of glyphosate (Siemering 2006). Siemering (2006) also noted that "only four exceedances in three years indicates that DBW glyphosate applications are not likely to pose a risk to the aquatic environment."

A study evaluating the toxicity of individual and herbicide mixes on fathead minnows found that glyphosate (Accord Concentrate) did not show any appreciable acute toxicity, either alone, with surfactants, or in combination with imazapyr (Chopper and Arsenal AC) (Tatum et al. 2011). No LC50 values could be calculated because less than 50 percent mortality was observed at the highest herbicide concentrations, which were equivalent to spraying the maximum application rate directly into a stagnant pond.

An Iranian study of the toxicity of three sturgeon species to glyphosate (Filizadeh and Rajabi Islami 2011) found 96-hour LC50 levels for sturgeon fry of between 19 mg/l and 26 mg/l, and 168-hour LC50 levels of between 8 mg/l and 13 mg/l. These levels are above the highest concentration found immediately following WHCP treatment of 0.158 mg/l (ppm) (which was an outlier), indicating no risk to these sturgeon species. In addition, the glyphosate formulation used in this study was Roundup, which contains a surfactant known to be toxic to aquatic species.

Table 6-7
Response of Various Fish Species to WHCP Chemicals, at LC50 Values

Species	Chemical	LC50	Time Period	Reference
Fathead minnow	Glyphosate	97 ppm	96 hr	Folmar et al., 1979
Fathead minnow	Glyphosate	9.4 ppm to 97 ppm	96 hr	USEPA 2000
Bluegill	Glyphosate	140 ppm	96 hr	Folmar et al., 1979
Bluegill	Glyphosate	120 ppm	96 hr	Corcoran et al., 1984
Bluegill	Glyphosate, isopropylamine salt	>1,000 ppm	96 hr	Corcoran et al., 1984
Rainbow trout	Glyphosate	140 ppm	96 hr	Folmar et al., 1979
Rainbow trout, Donaldson trout	Glyphosate	8.2 ppm to 240 ppm	96 hr	USEPA 2000
Trout	Glyphosate, isopropylamine salt	>1,000 ppm	96 hr	Corcoran et al., 1984
Trout	Glyphosate	86 ppm	96 hr	Corcoran et al., 1984
Chinook salmon*	Glyphosate	9.1 ppm to 1,440 ppm	96 hr	ECOTOX 2001
Pink salmon	Glyphosate	17 ppm to 48 ppm	96 hr	ECOTOX 2001
Chum salmon	Glyphosate	11 ppm to 58 ppm	72 hr	ECOTOX 2001
Coho salmon, silver salmon	Glyphosate	5.7 ppm to 55 ppm	96 hr	ECOTOX 2001
Sockeye salmon	Glyphosate	28 ppm	96 hr	ECOTOX 2001
Harlequin fish	Glyphosate	168 ppm	96 hr	Corcoran et al., 1984
Carp	Glyphosate	115 ppm	96 hr	Corcoran et al., 1984
Carp	Glyphosate, isopropylamine salt	>10,000 ppm	96 hr	Corcoran et al., 1984
Channel catfish	Glyphosate	130 ppm	96 hr	Folmar et al., 1979

* Study utilized in RQ calculation.

Table 6-7, above, summarizes glyphosate acute toxicity testing on several fish species. While the risk of acute toxicity to special status or other fish resulting from the WHCP is extremely low, there is some evidence of chronic/sublethal toxicity impacts from glyphosate.

While glyphosate did not result in estrogenic activity (Xie et al. 2005), other studies have found indications of reduced liver activity and immune suppression resulting from sublethal exposure to glyphosate. Li and Kole (2004) found an inhibitory effect on liver esterase as compared to controls with exposure to 1.0, 5.0, and 25 mg/l glyphosate for 65 days. Li and Kole cited other studies that noted behavioral changes to rainbow trout after one month of exposure to 46 ppb glyphosate, Li and Kole (2004) also noted increased enzyme activity, and interruption of immune response and protein biosynthesis in carp exposed to 2.5 to 10 mg/l glyphosate. WHCP long-term exposure levels of glyphosate are significantly lower than the long-term exposure levels tested by Li and Kole.

Table 6-8
RQ Calculations for Glyphosate

Species	EEC/LC 50	RQ
Chinook salmon	0.158 ppm/ 9.1 ppm	0.017 (acute)
Larval delta smelt	0.158 ppm/ 270 ppm	0.0006 (acute)
Rainbow trout	0.158 ppm/ 1 ppm	0.158 (chronic)

* The 0.158 ppm concentration used for the chronic toxicity exposure is extremely conservative, as this was an instantaneous maximum exposure.

Table 6-8, above, provides the RQ calculations for glyphosate, using Chinook salmon and larval delta smelt. As Table 6-7 illustrates, the RQ values for both species are well below the endangered species LOC values. The conservative chronic exposure RQ is also well below the LOC. The already low potential for toxicity effects of glyphosate can be further minimized by treating water hyacinth early in the growing season, thus reducing the amount of herbicide needed.

Toxicity of Penoxsulam to Listed Fish Species

Penoxsulam is classified as practically non-toxic to freshwater and marine/estuarine fish, based on results of acute toxicity testing (USEPA January 2007). Species with LD50 values of greater than 100 ppm fall into the practically non-toxic category. Chronic toxicity studies show no treatment-related effects to growth and reproduction in freshwater fish at concentrations up to 10.2 ppm (USEPA January 2007), a concentration approximately 2,000 times greater than the estimated concentration of penoxsulam in one meter of water immediately following WHCP treatment. The acute toxicity LC_{50} results in USEPA's Ecological Risk Assessment (USEPA January 2007) are also non-observable adverse effect concentrations (NOAEC), as there were no observable effects at the highest concentrations tested. Similarly, in the chronic toxicity testing, there were no observable effects at 10.2 ppm, the highest concentration of penoxsulam tested. **Table 6-9,** on the next page, summarizes toxicity testing results for several fish species for penoxsulam and degradates.

Because penoxsulam is a relatively new herbicide (USEPA approval in 2007), there are few studies evaluating penoxsulam toxicity in the open literature. Most evaluations of penoxsulam ecotoxicity rely on the USEPA registration data (Washington DOE 2012, FOOTPRINT PPDB 2009). One study of the impact of penoxsulam in rice field conditions on carp found mixed signs of oxidative stress after 7, 21, or 72 days of penoxsulam exposure (Cattaneo et al. 2010). However, exposure levels were 23 ppb, more than ten times higher than the estimated concentration of penoxsulam immediately following WHCP treatment. Furthermore, the calculated post-treatment WHCP 2 ppb level would be expected to exist only a short time (at most a few hours) due to tidal flow, mixing, and dilution. Thus, WHCP treatments would not result in levels that could produce this potential sub-lethal effect.

Table 6-9
Response of Various Fish Species to Penoxsulam at LC50 Values

Species	Chemical	LC50	Time Period	Reference
Rainbow trout *(Oncorhynchus mykiss)**	Technical grade penoxsulam	>102 ppm	96-hr	USEPA January 2007
Rainbow trout	Degradates and end-use products	None	96-hr	USEPA January 2007
Bluegill sunfish *(Lepomis macrochirus)*	Technical grade penoxsulam	>103 ppm	96-hr	USEPA January 2007
Bluegill sunfish	Galleon or equivalent	>147 ppm	96-hr	USEPA January 2007
Bluegill sunfish	Degradates	None	96-hr	USEPA January 2007
Common carp *(Cyprinus carpio)*	Technical grade penoxsulam	>101 ppm	96-hr	USEPA January 2007
Common carp	Degradates and end-use products	None	96-hr	USEPA January 2007
Fathead minnow *(Pimephales promelas)**	Technical grade penoxsulam	10.2 ppm (NOAEC)	36 days	USEPA January 2007

* Study utilized in RQ calculation.

Table 6-10
RQ Calculations for Penoxsulam

Species	EEC/LC 50	RQ
Rainbow trout	.002 ppm/ 102 ppm	0.0000196 (acute)
Fathead minnow	.002 ppm/ 10.2 ppm	0.000196 (chronic)

* The 0.002 ppm concentration used for the chronic toxicity exposure is extremely conservative, as this was a calculated maximum exposure.

Table 6-10, above, provides the RQs for penoxsulam. Both the acute toxicity RQ and the chronic risk RQ are well below the LOC levels (0.05 and 1, respectively). These results indicate that penoxsulam use for WHCP treatments will not result in direct acute or chronic toxicity to fish. The already low potential for toxicity effects of penoxsulam can be further minimized by treating water hyacinth early in the growing season, thus reducing the amount of herbicide needed.

Toxicity of Imazapyr to Listed Fish Species

USEPA classified imazapyr as practically non-toxic to fish (AMEC Geometric 2009). A number of different bioassays of imazapyr toxicity to fish conducted for the USEPA registration process found LC50 values of greater than 100 ppm, and in some cases greater than 1,000 ppm (SERA 2004). Studies conducted in Thailand on Nile Tilapia reported lower LC50 levels of between 2.71 ppm and 4.36 ppm (SERA 2004). There are uncertainties about the Thai studies, as complete English versions are not available, and the species are not native to the United States. However, even the RQ calculated using these lower concentrations are below LOC values. Risk assessments separate the results of the >100 ppm and tilapia studies by classifying the fish as

tolerant or sensitive to imazapyr (SERA 2004, AMEC Geometrix 2009). **Table 6-11,** on the next page, summarizes fish toxicity data for imazapyr.

A study evaluating the toxicity of individual and herbicide mixes on fathead minnows found that the imazapyr herbicides Chopper and Arsenal AC (similar to Habitat) did not show any appreciable acute toxicity, either alone, with surfactants, or in combination with glyphosate (Accord Concentrate) (Tatum et al. 2011). No LC50 values could be calculated because less than 50 percent mortality was observed at the highest herbicide concentrations, which were equivalent to spraying the maximum application rate directly into a stagnant pond.

Patten (2003, cited in AMEC Geomatrix 2009) evaluated the osmoregulatory capacity of Chinook smolts based on plasma sodium level and gill ATPase and found that capacity was not affected by imazapyr at concentrations of up to 1,600 ppb. Patten identified a NOEC of greater than 1,600 ppb, which is more than two orders of magnitude above the expected concentration immediately following WHCP treatment of 5.6 ppb. Manning (1989b in AMEC Geomatrix 2009) found no effect on rainbow trout hatching, survival, or growth after 62 days exposure to up to 92.4 mg/l. Again, these levels, which show no chronic toxicity, far exceed WHCP treatment imazapyr exposures in both concentration and time.

The calculated RQ values, provided in **Table 6-12,** on the next page, indicate that imazapyr will not directly affect listed fish species. Both acute and chronic toxicity RQ values are well below the LOCs for endangered species. The already low potential for toxicity effects of imazapyr can be further minimized by treating water hyacinth early in the growing season, thus reducing the amount of herbicide needed.

Toxicity of Imazamox to Listed Fish Species

USEPA classified imazamox as practically non-toxic to fish. Supporting its low toxicity, imazamox was approved by USEPA as a "reduced risk" herbicide, and is the only synthetic herbicide granted a food residue tolerance exemption from USEPA (USFWS March 2012). The acute toxicity tests submitted to USEPA for the registration process found no observable effects at the highest concentrations of imazamox tested (approximately 100 ppm) (SERA 2010). There are relatively few toxicity studies evaluating the impact of imazamox on fish (or other) species; most cited studies were part of the USEPA pesticide registration process. Results of acute and chronic toxicity testing of imazamox in fish are provided in **Table 6-13,** on the next page. No bioactive metabolites inducing toxicity greater than the parent compound were found in literature screening (Environ 2012).

The calculated RQ values, in **Table 6-14,** on page 6-19, indicate that imazamox will not directly affect listed fish species. Both acute and chronic toxicity RQ values are well below the LOCs for endangered species. A recently completed assessment of the use of imazamox (Clearcast) to control Japanese eelgrass in Washington State also found no significant risks to fish (and aquatic invertebrates) (Environ 2012). The already low potential for toxicity effects of imazamox can be further minimized by treating water hyacinth early in the growing season, thus reducing the amount of herbicide needed.

Table 6-11
Response of Various Fish Species to Imazapyr at LC50 Values

Species	Chemical	LC50	Time Period	Reference*
Rainbow trout *(Oncorhynchus mykiss)***	Arsenal (equivalent to Habitat)	110 mg/l	96-hr	Cohle and McAllister 1984c
Rainbow trout	Arsenal	>110 mg/l	96-hr	Drotter et al. 1995
Bluegill sunfish *(Lepomis macrochirus)*	Imazapyr (technical)	>1,000 mg/l (no mortality)	96-hr	Cohle and McAllister 1984c
Bluegill sunfish	Arsenal	180 mg/l	96-hr	Cohle and McAllister 1984c
Atlantic silversides *(Menidia menidia)*	Imazapyr (technical)	184 mg/l	96-hr	Manning 1989a
Nile tilapia *(Oreochromis niloticus niloticus)*	Imazapyr (not specified)	4.3 mg/l	96-hr	Supamataya et al. 1981
Silver barb *(Barbonymus gonionotus)***	Imazapyr (not specified)	2.7 mg/l	96-hr	Supamataya et al. 1981
Fathead minnow *(Pimephales promelas)*	Imazapyr (technical)	120 mg/l (NOEC)	Life cycle	Drotter et al. 1998
Fathead minnow**	Imazapyr (technical)	118 mg/l (NOEC)	28-days	Drotter et al. 1999

* References are cited in AMEC Geometrix 2009 and SERA 2004; most studies were conducted as part of USEPA's registration process.

** Study utilized in RQ calculation.

Table 6-12
RQ Calculations for Imazapyr

Species	EEC/LC 50	RQ (LOC = 0.05)
Rainbow trout *(Oncorhynchus mykiss)*	.0056 ppm/ 110 ppm	0.00005 (acute, tolerant)
Silver barb *(Barbonymus gonionotus)*	.0056 ppm/ 2.7 ppm	0.002 (acute, sensitive)
Fathead minnow *(Pimephales promelas)*	.0056 ppm/ 118 ppm	0.00004 (chronic)

* The 0.0056 ppm concentration used for the chronic toxicity exposure is extremely conservative, as this was a calculated maximum exposure.

Table 6-13
Response of Various Fish Species to Imazamox at LC50 Values

Species	Chemical	LC50	Time Period	Reference
Bluegill sunfish *(Lepomis macrochirus)*	Imazamox (technical)	>119 ppm NOEC=119 ppm	96-hr	USEPA 2008
Rainbow *trout (Onchorhynchus mykiss)*	Imazamox (technical)	>122 ppm NOEC=122 ppm	96-hr	USEPA 2008
Sheepshead minnow *(Cyprinodon variegates)**	Imazamox (technical)	>94.2 ppm NOEC=94.2 ppm	96-hr	SERA 2010
Rainbow trout	Imazamox (technical)	122 ppm	28-day	European Commission 2002
Rainbow trout*	Imazamox (technical)	11.8 ppm	96-day	European Commission 2002

* Study utilized in RQ calculation.

Table 6-14
RQ Calculations for Imazamox

Species	EEC/LC 50	RQ
Sheepshead minnow *(Cyprinodon variegates)*	0.0112 ppm/ 94.2 ppm	0.0001 (acute)
Rainbow *trout (Onchorhynchus mykiss)*	0.0112 ppm/ 11.8 ppm	0.0009 (chronic)

* The 0.0112 ppm concentration used for the chronic toxicity exposure is extremely conservative, as this was a calculated maximum exposure.

Table 6-15
Response of Rainbow Trout to Adjuvants at LC50 Levels

Species	Chemical	LC50	Time Period	Reference
Rainbow trout	Agridex	>1,000 ppm	96-hr	WSDA 2005
Rainbow trout	Competitor	95 ppm	96-hr	WSDA 2005

Table 6-16
RQ Calculations for Agridex and Competitor Adjuvants

Adjuvant	Species	EEC/EC 50	RQ
Agridex	Rainbow trout	0.00025 ppm/ >1,000 ppm	2.5×10^{-7} (acute)
Competitor	Rainbow trout	0.00025 ppm / 95 ppm	2.6×10^{-6} (acute)

Toxicity of Agridex and Competitor to Listed Species

There has been relatively little research on the toxic effects of adjuvants. Nonylphenol ethoxylate (NPE) surfactants are more toxic to aquatic species than most aquatic pesticides, and may also cause endocrine disruption. NPE adjuvants such as R-11 were eliminated from WHCP as a result. The non-ionic adjuvant Agridex, which replaced R-11, has significantly lower toxicity, with LC50 levels greater than 1,000 mg/l (ppm). For 472 RQ values calculated for Agridex in 2004 and 2005, SFEI also found no LOC exceedances. The vegetable oil-based adjuvant Competitor has an LC50 of 95 ppm, still resulting in a low RQ value. **Table 6-15,** above, summarizes two toxicity studies for WHCP adjuvants. **Table 6-16,** above, summarizes the RQ values for the adjuvants. Both RQ values are several orders of magnitude below the LOC.

Toxicity of WHCP Herbicides to Reptiles and Amphibians

As compared to fish, there is significantly less information related to the toxic effects of WHCP herbicides and adjuvants to amphibians and reptiles. However, the limited information that is available indicates that toxic impacts to amphibians and reptiles resulting from WHCP are highly unlikely.

Amphibians are thought to be more sensitive to chemical exposure than reptiles, because of their thinner skin and the fact that they inhabit both water and land. As a result, amphibian toxicity studies are often used to infer toxicity

Table 6-17
Concentrations of Test Solutions and Calculated Exposure Ranges for Herbicides, Surfactants, and Mixtures from CDFG Garter Snake Acute Toxicity Study

Herbicide and/or Surfactant	Concentrations of Test Solutions (mg/l or ppm)	Experimental Oral Exposure Range (mg/kg)	Experimental Dermal Exposure Range (mg/kg)
2,4-D (Weedar 64)	3,000	28.791 to 32.895	28.791 to 32.895
Glyphosate (Rodeo)	3,900	37.055 to 39.494	37.055 to 39.494
Nonylphenol ethoxylates (NPE)(R-11)	2,360	22.056 to 30.256	22.056 to 30.256
2,4-D (Weedar 64) and NPE (R-11)	2,800	24.207 to 30.769	24.207 to 30.769
	1,160	10.029 to 12.747	10.029 to 12.747
Glyphosate (Rodeo) and NPE (R-11)	3,620	32.321 to 39.635	32.321 to 39.635
	2,200	19.643 to 24.088	19.643 to 24.088

effects on reptiles, when specific reptile studies are not available. In addition, bird toxicity studies represent surrogates for terrestrial phase amphibians and reptiles, and fish may be surrogates for aquatic phase amphibians (USEPA January 2007).

Because of the scarcity of reptile studies, one of the conditions of WHCP's initial USFWS Biological Opinion was to conduct snake toxicity testing of WHCP herbicides. The DBW provided funding to the CDFG to conduct acute oral and dermal toxicity studies on garter snakes (Hosea et al. 2004). CDFG utilized two surrogate species of garter snakes, common garter snake, *Thamnophis sirtalis*, and western terrestrial garter snake, *Thamnophis elegans*. These garter snake species are closely related to the threatened giant garter snake, *Thamnophis gigas*.

Snakes were exposed both orally and dermally to a solution of herbicide, herbicide-surfactant, or control (distilled water). The surfactant studied was R-11®, which has since been removed from WHCP due to its relative high toxicity to aquatic species. Both herbicides and surfactant were at concentrations equivalent to the mixing tanks (i.e. the concentration from the spray nozzle).

Table 6-17, above, provides the concentrations of test solutions and actual exposure range (in mg/kg body weight). CDFG observed the snakes for seven days following treatment. There were no acute lethal or sublethal effects. Snakes did not exhibit significant alterations in behavior following treatment, and did not develop skin lesions or other physical abnormalities. There was no significant difference in post exposure weight change between test groups. CDFG reported that "if snakes were inadvertently sprayed directly or were to consume any of the undiluted spray solution, there should be no acute toxicity" (Hosea et al. 2004).

Table 6-18, on the next page, summarizes toxicity studies of reptiles, amphibians, or bird surrogates to current and potential WHCP herbicides. Studies of 2,4-D acute toxicity to three frog species, tusked frog, brown striped marsh frog, and western chorus frog, found 96 hour LC50 values from 100 ppm to 340 ppm

Table 6-18
Toxicity of Reptiles and Amphibians to WHCP Herbicides

Species	Chemical	LC50	Time Period	Reference
Three frog species	2,4-D	100 ppm to 340 ppm	96-hr	ECOTOX 2001
African clawed frog *(Xenopus laevis)*	Glyphosate formulations	604 ppm	96-hr	Edington et al. 2004
Mallard duck *(anas platyrhynchos)*	Penoxsulam (technical)	>1,900 ppm	14-day	USEPA September 2007
Bull frog tadpoles *(Rana catesbeiana)*	Imazapyr (Habitat)	1,739 ppm	96-hr	AMEC Geomatrix 2009
Mallard duck	Imazamox (technical)	>1,950 ppm	96-hr	USEPA 2008

(ECOTOX 2001). Another study found no effects on tadpoles in up to 50 ppm 2,4-D for 48 hours, and no effects on frog abundance as a result of partial treatment of Long Pond, New York, with granular 2,4-D (Halter 1980).

Much of the amphibian toxicity data in the literature for glyphosate was based on the herbicide Roundup, and is not relevant for the WHCP. Roundup is not approved for aquatic use because it includes a surfactant, polyethoxylated tallowamine (POEA), which is highly toxic to aquatic species. Because Roundup includes this surfactant, the herbicide is toxic to aquatic species, including amphibians (and not approved for aquatic use). There were some studies in the literature, discussed below, that utilized technical grade glyphosate or Rodeo (approved for aquatic use). Rodeo was previously utilized by the WHCP, and is essentially the same formulation as AquaMaster, the current WHCP glyphosate herbicide.

Howe et al., (2004) examined the toxicity of four North American frog species to several glyphosate formulations (most with surfactant), as well as technical glyphosate. They found no significant acute toxicity with technical grade glyphosate. Edginton et al., (2004) conducted amphibian toxicity testing and compared two different study designs using African clawed frog (*Xenopus laevis)* and several glyphosate herbicides. Rodeo was the least toxic of the herbicide formulations tested, with LC levels dependent on pH. At pH 6.5, the *Xenopus* 96-hour LC10 (lethal concentration for 10 percent) ranged from 1,722 ppm to 3,024 ppm, and the LC50 ranged from 4,341 ppm to 6,419 ppm. Toxicity was greater at pH 8, but still far below WHCP exposure levels. The 96-hour LC10 at pH 8 was 240 ppm to 395 ppm, and the LC50 was 604 ppm to 645 ppm (Edginton et al. 2004).

Perkins et al., (2000) examined the effect of various glyphosate herbicides, including Rodeo, on the (*Xenopus laevis)*, using the Frog Embryo Teratogenesis Assay – *Xenopus* (FETAX). Rodeo was found to be the least toxic, with a LC5 (lethal concentration for 5 percent) of 3,799 mg/l (ppm) and a LC50 of 5,407 mg/l. Roundup was 700 times more toxic than Rodeo, due to the surfactant POEA.

Sparling et al., (2006) examined the toxicity of a glyphosate herbicide (Glypro) and the acid/ buffer adjuvant LI700 on turtle embryos and early hatchlings. They exposed eggs of red eared sliders (*Trachemys scripta elegens*) to between 0 to 11,206 ppm herbicide and between 0 and 678 ppm adjuvant. There were dose related impacts on hatching success, hatchling weight, and somatic indices, primarily at the highest levels. The study concluded that "because of the high concentrations needed to produce effects… glyphosate with LI700 poses low levels of risk to red-eared slider embryos under normal field operations with regards to endpoints measured in the present study" (Sparling et al. 2006).

There is no amphibian or reptile toxicity testing data for penoxsulam. USEPA utilizes bird and fish toxicity testing to evaluate the terrestrial and aquatic impacts to amphibian and reptile species. Penoxsulam is practically non-toxic to fish and bird species. Testing for toxicity of penoxsulam in birds during a 14-day test period did not result in an LC50 calculation at the highest concentration tested of 1,900 ppm (USEPA September 2007).

Yahnke et al. (2012) evaluated the impact of exposure of juvenile Oregon Spotted Frogs to a mixture of imazapyr and Agridex in a 96-hour static-renewal test, at concentrations associated with 3.5 liters/hectare and 7.0 liters/hectare. Frogs were reared for 2 months in clean water following exposure to evaluate feeding behavior, growth, grow-out, and liver conditions index. Yahnke found no differences for any endpoint between herbicide-exposed and clean-water control frogs.

USEPA's risk assessment of the potential impacts of imazapyr to the California red-legged frog (Hurley and Shanaman 2007) found no indication of direct effects on either the aquatic or terrestrial phase of the species. The assessment endpoints included direct toxic effects on survival, reproduction, and growth. The study also evaluated indirect effects, and found no indication of indirect effects to terrestrial or aquatic food sources. The study did find the potential for indirect effects to habitat (through spray of non-target plants). However, the effects of imazapyr to non-target plant species can be mitigated by spraying procedures.

There is no amphibian or reptile toxicity testing data for imazamox. USEPA utilizes bird and fish toxicity testing to evaluate the terrestrial and aquatic impacts to amphibian and reptile species. Imazamox is practically non-toxic to fish and bird species. Like the toxicity testing for fish, there were no concentrations of imazamox tested in birds that resulted in any signs of toxicity (SERA 2010).

Table 6-19, on the next page, provides the RQ calculations for amphibians, reptiles, or bird surrogates. For all five herbicides, the RQ values are well below LOCs. There may be temporary indirect effects to amphibians and reptiles as a result of imazamox treatment due to the overspray of herbicide on non-target plant species. These effects are unlikely, and can be mitigated with procedures described in Exhibit 3-4 in Section 3.

2. Bioaccumulation of WHCP Herbicides

The WHCP is not likely to result in direct effects due to bioaccumulation of herbicides. Bioaccumulation is an increase in the

Table 6-19
RQ Calculations for Amphibians, Reptiles, or Bird Surrogates

Herbicide	Species	EEC/LC 50	RQ
2,4-D	Frogs (three species)	0.4 ppm/ 100 ppm	0.004
Glyphosate	*Xenopus*	0.158 ppm/ 604 ppm	0.0003
Penoxsulam	Mallard duck	0.002 ppm/1,900 ppm	1.0×10^{-6}
Imazapyr	Bull frog tadpoles	0.0056 ppm/ 1,739 ppm	3.2×10^{-6}
Imazamox	Mallard duck	0.0056 ppm/ 1,950 ppm	2.9×10^{-6}

concentration of a chemical in a biological organism over time, compared to the chemical's concentration in the environment. Compounds accumulate in organisms whenever they are taken up and stored faster than they are broken down (metabolized) or excreted. Bioaccumulation of chemicals in herbicides can occur in plant or animal tissues due to direct uptake or exposure, or in animal tissues by consumption and ingestion of other plant or animal species that have bioaccumulated these chemicals.

2,4-D

According to most sources, 2,4-D does not bioaccumulate in plants, and there is no evidence that 2,4-D accumulates to a significant level in mammals or other organisms (EXTONET 1996). The half-life of 2,4-D in living organisms is between 10 and 20 hours, and most 2,4-D is excreted in the urine (EXTONET 1996; NPTN 2008). The National Library of Medicine Hazardous Substance Data Bank states that 2,4-D is metabolized in fish and that bioconcentration is not expected to be appreciable (HSDB 2001). In a study exposing channel catfish and bluegill to 2 ppm 2,4-D by intraperitoneal injection, the fish excreted 90 percent of the herbicide within six hours (HSDB 2001). The researchers concluded there was no evidence for bioaccumulation in channel catfish and bluegills (Sikka et al. 1977).

Wang et al. (2004) evaluated bioaccumulation factors of 2,4-D, exposing carp and Nile tilapia to 0.5ppm 2,4-D. The 2,4-D bioaccumulation factor in carp dropped from 45 percent after seven days to 22 percent after 14 days. For Nile tilapia, the bioaccumulation factor dropped from 33 percent after five days to 17 percent after 14 days. This study indicates that 2,4-D does not bioaccumulate in fish.

Tu et al., (2001) reported on studies in Russia that found residues of 2,4-D in eggs, milk, and meat, however the type of 2,4-D was not reported. Tu et al., (2001) also reported on an Oregon study that found that 2,4-D risk to browsing wildlife is low. In aquatic species, the highest concentrations of 2,4-D were typically reached shortly after application, and dissipated within three weeks following exposure (Tu et al. 2001). After animals were removed from contaminated waters, they tended to excrete 2,4-D residues.

There is some evidence that fish take up 2,4-D, but seemingly at low levels that do not adversely affect fish or other species ingesting them. Folmar (1980) found fish present within a spray plot take up enough 2,4-D, or breakdown enough phenols, to impart an objectionable taste for the flesh for several days after spraying. Water column concentrations of 500 ppb imparted an "inferior" taste, while 100 ppb imparted an "acceptable" taste. These levels are significantly higher than those found even immediately after WHCP treatments.

Glyphosate

Glyphosate has virtually no tendency to bioconcentrate (Siepmann 1995). Glyphosate is poorly absorbed from the digestive tract, and is largely excreted unchanged by mammals. It has no significant potential to accumulate in animal tissue, and a very low potential for glyphosate to build up in the tissues of aquatic invertebrates or other aquatic organisms (EXTONET 1996). Glyphosate is also not expected to bioaccumulate in plants (County of Lake 2005). Carp exposed to 0.05 ppm glyphosate had a bioaccumulation factor (concentration in fish/concentration in water) of 42 percent after seven days, decreasing to 25 percent after 14 days (Wang et al. 2004). The same 0.05 ppm exposure in Nile tilapia resulted in a 65 percent bioaccumulation factor after five days, decreasing to 13 percent after 14 days (Wang et al. 2004), indicating that glyphosate does not bioaccumulate in fish.

In an AquaMaster fact sheet, Monsanto (2002) states that "in laboratory studies conducted with glyphosate, biocentration factors were less than 1.0, indicating that glyphosate does not accumulate in fish. The low bioaccumulation factor is a result of glyphosate being readily soluble in water, and therefore subject to rapid elimination from organisms in water. Other animal species studied include marine mollusks and crustaceans, also showed low potential for bioaccumulation."

Penoxsulam

USEPA considers penoxsulam to have low potential to bioaccumulate in aquatic organisms (USEPA September 2007). A European risk assessment also determined a low bioaccumulation potential for penoxsulam in birds and mammals (Washington DOE 2012). The bioconcentration factor (BCF) of penoxsulam in crayfish after 14 days exposure was 0.02 ml/g (values less than 100 are considered low) (USEPA September 2007; FOOTPRINT 2009).

Imazapyr

Imazapyr is not expected to bioaccumulate in aquatic species (USEPA 2009). In a study exposing freshwater clam *(Corbicula fluminea)* to imazapyr in a model pool system, imazapyr was not detected in clam tissues above the detection limit of 50 ppb during a 28-day observation period (AMEC Geomatrix 2009). Imazapyr did not bioaccumulate in a similar study of Eastern oyster *(Crassostrea verginica)* and grass shrimp *(Paleomonetes pugio)* (AMEC Geomatrix 2009).

Imazamox

The potential for bioconcentration of imazamox is low (HSDB Database 2012). Imazamox did not significantly bioaccumulate

in bluegill sunfish, and concentrations of imazamox in whole fish and edible tissue were less than the minimum detectable limit (USEPA 2008).

Adjuvants

There is limited information on bioaccumulation of adjuvants. The Material Safety Data Sheet (MSDS) for Agridex states that bioaccumulation of the adjuvant is unlikely due to the low water solubility of the product (Bayer Crop Science 2004). The MSDS for the vegetable oil-based adjuvant Competitor indicates no chronic toxicity for the adjuvant (Wilbur-Ellis 2010). The primary ingredient in Competitor, ethyl oleate, is approved by the Food and Drug Administration as a regulated food additive (Bakke 2007).

3. Disturbance from Treatment and Monitoring Boats

Boat noise has been identified as inducing the startle and alarm responses in fish (Scholik and Yan 2002). These responses cause fish to flee an area (Boussard 1981). Boat noise has also been shown to temporarily reduce auditory sensitivity of some fish species (Scholik and Yan 2002). However, the Delta is already heavily used by motorboats, and the current level of water hyacinth and other vegetation management activities using boats have been conducted for over 25 years. Thus, fish are likely habituated to a substantial degree of boat-related noise. The WHCP is not expected to result in significant additional boat disturbance to fish. To the extent that WHCP boats induce a "flee" response, it may be beneficial for fish to remove themselves from treatment areas.

4. Disturbance from Mechanical Removal with Specialized Aquatic Equipment

The potential impacts of mechanical removal of water hyacinth in the Delta using specialized aquatic equipment were evaluated by San Francisco Estuary Institute in 2003 and 2004. In May 2003, SFEI initiated consultations with USFWS and NMFS to evaluate the impact of mechanical removal on endangered species. Both services issued letters indicating that formal consultation was not required, and approved the mechanical removal project with conditions. The conditions, included: (1) efforts be made to minimize the impacts on listed species; and (2) the project occur within the dates when sensitive species are least likely to be adversely affected (between July 15th and October 31st) (Greenfield et al. 2006).

Current WHCP mechanical removal activities will have less potential of impacting listed species because the water hyacinth will be directly removed from the water with conveyors. Removing plants will reduce the potential for lower dissolved oxygen due to plant decomposition. Greenfield et al. (2007) noted that mechanical removal of water hyacinth, which contains mercury, could aid in mercury remediation efforts, an unplanned subsidy of the action. Greenfield also concluded that estuary-wide effects of mechanical removal using specialized aquatic equipment would be limited.

D. Indirect Effects of WHCP

Indirect effects are caused by the action and are later in time, but are still reasonably certain to occur. There is potential for limited, and temporary, indirect effects on special status fish species as a result of WHCP treatments. Below, we describe three potential mechanisms of indirect effects: loss of native aquatic plants, food web effects, and low dissolved oxygen.

1. Loss of Native Plant Species

While there is some herbicide selectivity to target species, by definition WHCP herbicides are generally toxic to plants at specified treatment levels. Vegetation subject to overspray will be vulnerable to WHCP treatments. WHCP treatment crews follow mitigation measures to reduce the potential for overspray. In addition, water hyacinth grows in dense, mono-culture mats, thus further reducing the potential for impacts to non-target plant species.

Plant death from 2,4-D typically occurs within three to five weeks after treatment, although during periods of warm weather, water hyacinth shows signs of dying within hours of spraying. Any broadleaf vegetation subject to overspray or volatization will be vulnerable to 2,4-D activity. Most of the special status plants and several other native plants are broadleaf species. Sensitive riparian habitats and wetlands near WHCP treatment sites also include other potentially impacted broadleaf plants.

Plants begin to show symptoms of glyphosate treatment (gradual wilting and yellowing) within seven to fourteen days. Exposure of any non-target plants to glyphosate, including those in sensitive riparian and wetland habitats, could result in loss of plant species and habitat impacts.

Penoxsulam is relatively slow-acting, with reddening followed by plant death over 60 to 120 days. Penoxsulam exhibits toxicity to aquatic vascular plants, with an EC50 of 0.003 mg/l for duckweed, and a NOAEC of 0.001 mg/l, based on reduction of frond number (USEPA January 2007). Immediate post-treatment exposure levels of 2 ppb (0.002 mg/l) could impact submersed aquatic plants, and plants subject to overspray could be exposed to higher concentrations.

Imazapyr is also gradually acting. With complete plant death occurring after several weeks or months. Vascular plants are more sensitive to imazapyr than non-vascular plants, with vascular plant RQ values of greater than one (USEPA 2006). Imazapyr will not affect submerged aquatic vegetation.

Imazamox is faster acting than the other two ALS inhibitors, with visual symptoms appearing within 1 week and complete death within six weeks. The EC50 of imazamox to duckweed *(Lemna gibba)* is 11 ppb (0.011 mg/l). Duckweed is the most sensitive aquatic plant species (USEPA 2008). The NOEC for duckweed is 4.5 ppb. The concentration of imazamox immediately following treatment is expected to be 11.2 ppb; however, this concentration will be rapidly diluted, and the chance for overspray to important habitat species is low due to the fact that water hyacinth typically grows in monospecific mats.

The DBW also utilizes adjuvants to increase absorption and translocation of the herbicide.

DBW will utilize the paraffin-based non-ionic surfactant, Agridex, and the vegetable oil-based adjuvant, Competitor. Relatively little is known about impacts of adjuvants on plants. However, use of these chemicals in concentrations specified on the labels is not expected to negatively impact special status species, sensitive habitats, or wetlands.

The potential for impacts resulting from herbicide overspray depend on the amount of exposure, concentration of herbicide, and proximity of sensitive habitats, wetlands, and special status plants. One study found that only three to four percent of 2,4-D droplets drift beyond the target zone, and no significant amount of material is collected as drift (HSDB 2001). Blankenship and Associates (2004) found that using conservative application rates, detectable adverse effects could result from less than one percent spray drift of glyphosate or 2,4-D.

The concentration of active ingredient of the current or proposed WHCP herbicides leaving the spray nozzle is high enough to cause adverse effects. Thus, there is the potential that uncontrolled herbicide overspray could affect nearby nontarget vegetation.

Depending on the herbicide and concentration in water, treatment of water hyacinth could result in limited loss of native submerged aquatic vegetation growing in and around treatment areas. Such vegetation may be utilized by special status fish for rearing, coverage, and forage. In particular, shallow vegetated habitat is believed to be important to spawning success of delta smelt, although most spawning occurs before WHCP treatments begin.

Loss of cover, rearing, and forage area of special status species could constitute an indirect effect under certain conditions. However, dense canopies of water hyacinth reduce light levels for submerged plant photosynthesis and thus can effectively shade out native vegetation. The benefit to native submerged aquatic vegetation from removal of water hyacinth is expected to outweigh losses due to herbicide toxicity overspray.

While there is a potential risk to sensitive habitats, wetlands, and special status plants due to herbicide overspray, the likelihood of such effects occurring is low. Herbicide application will be focused directly on target plants to decrease the possibility that concentrated herbicides would come in contact with sensitive plants, or result in impacts to sensitive habitats or wetlands.

The DBW will follow herbicide label instructions that reduce herbicide drift. These steps include using the largest size spray droplets, and lowest spray pressure, that will provide sufficient coverage and control. Furthermore, DBW will not treat at a particular site if the wind is greater than 10 mph (or 7 mph in Contra Costa County). In addition, DBW implements mitigation measures to reduce the potential for indirect effects on plants and native habitat, including: avoiding application of herbicides near special status species, sensitive riparian habitat, and other biologically important resources; providing a 50 foot buffer between treatment sites and shoreline elderberry shrubs; and conducting herbicide treatments in order to minimize potential for drift.

2. Food Web Effects

Macroinvertebrates

Special status fish species, or native resident or migratory fish, could be indirectly impacted if WHCP decreases the abundance of invertebrates, such as zooplankton, upon which these fish feed. While there is potential for toxic impacts to invertebrates due to WHCP, such food web effects are unlikely.

In order to better understand the impact of non-native species on the food web, Toft et al., (2003) compared habitat structure, invertebrate assemblages, and diets of fish associated with water hyacinth and the native floating aquatic plant, pennywort. Toft's results are particularly relevant, as the study took place at three different locations in the Delta. While water hyacinth is similar in appearance to pennywort, the study found that pennywort is functionally superior to water hyacinth, in terms of habitat.

The study compared populations of epiphytic invertebrates (present in the plant roots), epibenthic invertebrates (present just above the sediment), benthic invertebrates (present in the sediment), and insects in the canopy, in water hyacinth and pennywort. The study also surveyed fish present in both plants, and analyzed fish stomach contents to determine diets. Toft et al., (2003) found that "invertebrates associated with hyacinth occur less in the diets of adjacent fish than do invertebrates associated with pennywort." One finding was that the non-indigenous amphipod, *Crangonyx floridanus*, was more abundant in water hyacinth than pennywort. While the amphipod was prevalent, *Crangonyx* was not found in fish diets. By comparison, *Hyalella azteca*, commonly found in fish diets, was typically more prevalent in pennywort.

There were significant differences between water hyacinth and pennywort in terms of epibenthic and benthic invertebrates. There was greater diversity among invertebrate species in pennywort than in water hyacinth. At one of the three sites, there were no amphipods or isopods under water hyacinth, possibly due to low dissolved oxygen levels. Similarly, there were more insects in pennywort canopies than in water hyacinth, again with greater taxa diversity. Toft et al., found the two plants to be not functionally equivalent, with the native pennywort providing better habitat and food sources for native invertebrates and fish species. This would indicate that if there was loss of invertebrates due to WHCP treatments, the impact on the food web would likely not be significant.

Earlier studies have shown that several of the invertebrates commonly found in water hyacinth, in particular amphipods, chironomid larvae, and *Gammarus,* are consumed by special status fish species such as Sacramento splittail, juvenile Chinook salmon, and delta smelt (Moyle 1976, Wang 1986, and Herbold 1987). Loss of a significant quantity of any of these invertebrates could adversely impact certain special status fish species.

Studies have found that control of macrophytes does not negatively impact macroinvertebrates. In a study comparing the long-term effects of macrophyte and algae management in two lakes in New York (one treated and one not), Harman et al. (2005) found no difference in richness and diversity

of the biota between the lakes. Taxonomic richness and diversity were similar in the treated and non-treated lakes.

Juvenile Chinook salmon feed on various aquatic and terrestrial insects, crustaceans, chironomid larvae and pupae, caddisflies (in fresh water), and *Neomysis, Cammarus*, and *Crangon* in more saline water (Wang 1986). Steelhead feed on terrestrial and aquatic insects, amphipods, crustaceans and small fish (Wang 1986). Juvenile green sturgeon feed on *Neomysis mercedis* and amphipods (Corophium) (Radtke 1966). Adults may feed on sand lances, clams, and shrimp (Moyle 1995).

Juvenile delta smelt primarily eat copepods, planktonic crustaceans, small insect larvae, and mysid shrimp, while older fish feed almost exclusively on copepods (Moyle 1976). Over recent years, there have been significant declines in delta smelt's preferred food resources due to invasive species such as the overbite clam (Bennett 2005).

Table 6-20, on the next two pages, summarizes toxicity data for invertebrate species at various life stages for 2,4-D, glyphosate, penoxsulam, imazapyr, imazamox, and two adjuvants. The EC50 toxicity endpoint for aquatic invertebrates and plants is the concentration of chemical that can be expected to cause a defined non-lethal effect in 50 percent of the test population. Typical endpoints are immobilization, reductions in growth, and reproductive effects.

When Weedar 64 (2,4-D) is applied at labeled rates, the herbicide is not likely to have toxic effects on aquatic invertebrates. In a study of invertebrate communities in artificial ponds, benthic macroinvertebrate communities showed no primary effects due to treatment (Stephenson and Mackie 1986). The LC50 in this study for various crustaceans and insects was over 100 ppm 2,4-D DMA. There were some subtle secondary effects, with lower benthic diversity in treated ponds almost one year after the initial treatment, however this response is not applicable to the tidal waters of the Delta. Washington State reported a NOEL for *Daphnia magna* exposed to 2,4-D of 27.5 ppm (Siemering 2006). Green and Abdelghani (2004) reported that high doses of 2,4-D in red swamp crawfish altered enzyme activity and gill structure, and disrupted liver function.

Toxicity levels for 2,4-D for a range of zooplankton are also higher than levels expected in WHCP. EC50 values for most zooplankton were over 100 ppm 2,4-D, while two species had EC50 values ranging from 1 to 10 ppm 2,4-D (Halter 1980). Most LC50 values for 2,4-D for benthic invertebrates were found to be over 1,000 ppm and over 10 ppm in life-cycle invertebrate tests using eggs and early life stages (Halter 1980).

The DBW conducted an analysis of water quality and toxicity using monitoring data gathered from 2001 to 2005. The DBW collected several hundred pre-treatment and post-treatment water samples and delivered these to CDFG laboratories to conduct five different toxicology tests. Based on examination of toxicology test results from post-treatment water samples, WHCP did not have a significant or consistent adverse effect on the test organisms used by the laboratories (including the water flea, *Ceriodaphnia dubia*).

Table 6-20
Response of Various Invertebrate Species to WHCP Chemicals, at LC50/EC50 Values

Species	Chemical	EC50	Time Period	Reference
*Daphnia magna**	2,4-D dimethylamine salt (DMA)	184 ppm	48-hr	Alexander et al., 1985
Daphnia magna	2,4-D DMA	176 ppm	96-hr	WSDE 2001
Ceriodaphnia dubia	Weedar® 64	116 ppm	96-hr	CDFG 2003
Cypridopsis, seed shrimp	2,4-D DMA	8 ppm	48-hr	Johnson and Finley 1980
Common shrimp	2,4-D DMA	>10 ppm	48-hr	ECOTOX 2001
Grass shrimp	2,4-D DMA	>100 ppm	48-hr	ECOTOX 2001
Brown shrimp	2,4-D DMA	2 ppm	48-hr	PAN 2001
Gammarus fasciatus	2,4-D DMA	>100 ppm	96-hr	Johnson and Finley 1980
Aquatic sowbug	2,4-D DMA	>100 ppm	48-hr	PAN 2001
Crayfish	2,4-D DMA	>100 ppm	48-hr	PAN 2001
Red swamp crayfish, juvenile	2,4-D DMA	1,174 ppm to 1,681 ppm	96-hr	PAN 2001
Red swamp crayfish	2,4-D DMA	185 ppm	96-hr	Green and Abdelghani 2004
Daphnia magna	Rodeo	218 ppm	48-hr	Henry et al., 1994
*Daphnia magna**	Rodeo, X-77, and Chemtrol	130 ppm	48-hr	Henry et al., 1994
Daphnia	Glyphosate	780 ppm	96-hr	DBW 2001
Hyalella azteca	Rodeo	720 ppm	96-hr	Henry et al., 1994
Ceriodaphnia dubia	Rodeo	225 ppm to 415 ppm	48-hr	Tsui and Chu 2004
Ceriodaphnia dubia	Rodeo	608 ppm	96-hr	CDFG 2003
Hyalella azteca	Rodeo, X-77, and Chemtrol	218 ppm	96-hr	
Hyalella azteca	Rodeo	225 ppm to 415 ppm	48-hr	Tsui and Chu 2004
Chironomus riparius (midge)	Rodeo	1,216 ppm	48-hr	Henry et al., 1994
Chironomus riparius	Rodeo, X-77, and Chemtrol	300 ppm	48-hr	Henry et al., 1994
Nephelopsis obscura (leech)	Rodeo	1,177 ppm	96-hr	Henry et al., 1994
Nephelopsis obscura	Rodeo, X-77, and Chemtrol	116 ppm	96 hr	Henry et al., 1994
Stagnicola elodes (pond snail)	Rodeo, X-77, and Chemtrol	234 ppm	96 hr	Henry et al., 1994

Table 6-20
Response of Various Invertebrate Species to WHCP Chemicals, at LC50/EC50 Values *(continued)*

Species	Chemical	EC50	Time Period	Reference
Daphnia magna	Glyphosate	>2,000 ppm	48-hr	Pereira, 2009
Pseudokirchneriella subcapitala (algae)	Glyphosate	129 ppm	96-hr	Pereira, 2009
Midge	Glyphosate	55 ppm	96-hr	HSDB 2001
Atlantic oyster	Glyphosate	>10 ppm	48-hr	DBW 2001
Shrimp	Glyphosate	281 ppm	96-hr	DBW 2001
Fiddler crab	Glyphosate	934 ppm	96-hr	DBW 2001
Daphnia magna	Penoxsulam (technical)	>98 ppm	48-hr	USEPA September 2007
Daphnia magna	Penoxsulam degradates	>96 ppm to >100 ppm	48-hr	USEPA September 2007
Daphnia magna	Penoxsulam degradates	>1 ppm to >1.6 ppm	48-hr	USEPA September 2007
Midge *(Chironomus* sp.)	Penoxsulam (technical)	>140 ppm	48-hr	USEPA September 2007
Amphipod *(Gammarus* sp.)	Penoxsulam (technical)	>126 ppm	48-hr	USEPA September 2007
*Daphnia magna**	Galleon/equivalent	>90.1 ppm	48-hr	USEPA September 2007
*Daphnia magna**	Penoxsulam (technical)	9.76 ppm 2.95 ppm NOAEC 9.76 ppm LOAEC	21-day	USEPA September 2007
Chironomus reparius	Penoxsulam (technical)	7.1 ppm NOAEC 15 ppm LOAEC	28-day	USEPA September 2007
Daphnia magna	Imazapyr (Arsenal)	350 ppm	48-hr	AMEC Geomatrix 2009
*Daphnia magna**	Imazapyr (technical)	>100 ppm	48-hr	AMEC Geomatrix 2009
*Daphnia magna**	Imazapyr (technical)	>97.1 ppm NOEC	21-day	AMEC Geomatrix 2009
Daphnia magna	Imazamox (technical)	>122 ppm 122 ppm NOEC	96-hr	USEPA 2010
Mysid shrimp* *(Mysidopsis bahia)*	Imazamox (technical)	>94.3 ppm 94.3 ppm NOEC	96-hr	SERA 2010
*Daphnia magna**	Imazamox (technical)	137 ppm	21-day	European Commission 2002
*Daphnia**	Agridex	>1,000 ppm	48-hr	WSDA 2005
*Daphnia**	Competitor	>100 ppm	48-hr	WSDA 2005

* Test utilized for RQ calculations.

In DBW's analysis, there were 20 samples which exceeded (then) NPDES permit levels (20 ppb) for 2,4-D, which were tested for water flea survival and growth. None of these samples adversely affected water flea survival. Two of the 20 samples adversely affected water flea reproduction. Because there were adverse effects on water flea survival and progeny on samples that did not have detectable levels of 2,4-D, it is not possible to attribute the small number of cases with adverse effects on exposure to 2,4-D.

Chronic toxicity tests using WHCP chemicals also found impact levels several orders of magnitude greater than likely exposure levels. The California Department of Fish and Game, Aquatic Toxicology Laboratory, conducted seven day chronic toxicity tests on the water flea neonates, *Ceriodaphnia dubia* (CDFG 2003). The seven day LC50 for Weedar 64 (2,4-D) was 97 ppm. The seven day lowest observable effect concentration (LOEC) for Weedar was 40.5 ppm.

When glyphosate is applied at labeled rates, the herbicide is not likely to have a negative impact on aquatic invertebrates. Studies indicate that invertebrates are less sensitive to technical grade glyphosate than are fish (Siepmann 1995). Henry et al., (1994) concluded that Rodeo (with X-77 and Chem-Trol adjuvants) does not pose an acute hazard to native aquatic invertebrates because the concentrations of these chemicals found to be acutely toxic to invertebrates were much higher than their expected or measured concentrations in water from wetlands treated with the herbicide mix. In addition, in field studies conducted by Henry et al., (1994), resident invertebrates in all study wetlands were observed to be abundant during the study period. Kreutzweiser et al., (1989) found that application of glyphosate on or adjacent to small tributaries of a creek did not result in disturbance of stream invertebrates.

A study evaluating the toxicity of individual and herbicide mixes on *Daphnia* found that the glyphosate (Accord Concentrate) did not show any appreciable acute toxicity, either alone, with surfactants, or in combination with imazapyr (Chopper or Arsenal AC) (Tatum et al. 2011). Back (2010) evaluated gastropod abundance and dry mass of benthic organisms in glyphosate-treated and non-treated marsh and found that treatments had little effect on herbivore-producer relationships and gastropod diversity one-year post-spraying.

Chronic toxicity tests using WHCP chemicals also found impact levels several orders of magnitude greater than likely exposure levels. The California Department of Fish and Game, Aquatic Toxicology Laboratory, conducted seven day chronic toxicity tests on the water flea neonates, *Ceriodaphnia dubia* (CDFG 2003). The seven day LOEC for Rodeo was 104 ppm.

In DBW's water quality and toxicity analysis, none of the glyphosate samples exceeded NPDES permit criteria (700 ppb). The CDFG laboratory conducted toxicity testing using the 18 samples with detectable levels of glyphosate. One of the 18 glyphosate samples had an impact on water flea survival. The glyphosate concentration of this sample was 84 ppb. Three of the 18 samples tested had glyphosate concentrations higher than 84 ppb, but had no impact on water flea survival or reproduction. Because there were adverse effects on water flea survival and progeny on samples that did not have detectable levels of glyphosate, it is not possible to attribute the small number of cases with adverse effects on exposure to glyphosate.

USEPA (September 2007) reported testing results for penoxsulam and metabolites on invertebrate species as part of the Ecological Risk Assessment. Tests were conducted for the pesticide registration process. Many of the degradate tests utilized only one concentration (approximately 1 ppm), and had no mortality or immobilization effects. Some tests utilized a range of concentrations, up to approximately 100 ppm, also with no mortality. Thus, the EC50 values for penoxsulam in Table 6-20 are conservative, and essentially equal to NOAEC levels (USEPA September 2007).

Acute toxicity testing with an end-use product equivalent or equal to Galleon (penoxsulam) found no toxicity to Daphnia magna at the maximum concentration of 90.1 ppm. There was minor immobilization impairment (5 percent to 10 percent) at the mid-range concentrations tested, but not the low and high concentrations (7.92 ppm and 90.1 ppm). The study determined that the 48-hour NOEAC level, based on mortality or immobilization, was 90.1 ppm (USEPA September 2007). Chronic toxicity testing of technical grade penoxsulam on *Daphnia* and chironomids found NOAEC levels of 2.95 ppm and 7.1 ppm, respectively, well above instantaneous concentrations expected from WHCP treatments.

USEPA found that imazapyr is practically non-toxic to aquatic invertebrates (AMEC Geomatrix 2009). A study evaluating the toxicity of individual and herbicide mixes on *Daphnia* found that the imazapyr herbicides Chopper and Arsenal AC (similar to Habitat) did not show any appreciable acute toxicity, either alone, with surfactants, or in combination with glyphosate (Accord Concentrate) (Tatum et al. 2011).

A study evaluating the toxicity of Arsenal in *Daphnia* found an EC50 value of 350 mg/l and a NOAEC of 180 mg/l in evaluating toxic endpoints of immobility and sub-lethal effects (Forbis 1984). Fowlkes (2003) evaluated the effects of imazapyr on benthic macroinvertebrates in a logged pond cypress dome in Florida and found no statistical difference in macroinvertebrate community composition, chironomid deformity rate, and chironomid biomass between control ponds and ponds treated with 1, 10, and 100 times the expected concentration resulting from the normal application rate. Back (2010) evaluated gastropod abundance and dry mass of benthic organisms in imazapyr-treated and non-treated marsh and found that treatments had little effect on herbivore-producer relationships and gastropod diversity one-year post-spraying. In a study of the impacts of intensive forest management on streams, Michael and Ruiz-Cordova found that offsite movement of imazapyr (Arsenal AC) did not affect periphyton biomass, macro-invertebrate density, richness, or community structure (Michael and Ruiz-Cordova 2006).

USEPA registration studies found that imazamox is practically non-toxic to aquatic invertebrates. As with fish, there are relatively few studies for this herbicide. The 96-hour EC50 values for *Daphnia magna* and mysid shrimp were close to 100 ppm, with no mortality and no signs of toxic effects at the highest concentrations tested (SERA 2010). Chronic toxicity testing also found no effect at imazamox concentrations greater than 100 ppm (European Commission 2002).

Table 6-21
RQ Calculations for Invertebrates for WHCP Herbicides and Adjuvants

Herbicide	Species	EEC/ EC50	RQ (LOC = 0.05)
2,4-D	*Daphnia magna*	0.4 ppm / 184 ppm	0.002
Glyphosate	*Daphnia magna*	0.158 ppm / 130 ppm	0.001
Penoxsulam	*Daphnia magna*	.002 ppm / 90.1 ppm	0.00002 (acute)
Penoxsulam	*Daphnia magna*	.002 ppm / 9.76 ppm	0.0002 (chronic)
Imazapyr	*Daphnia magna*	0.0056 ppm / 100 ppm	0.000056 (acute)
Imazapyr	*Daphnia magna*	0.0056 ppm / 97.1 ppm	0.000058 (chronic)
Imazamox	*(Mysidopsis bahia)*	0.0056 ppm / 94.3 ppm	0.000058 (acute)
Imazamox	*Daphnia magna*	0.0056 ppm / 137 ppm	0.000041 (chronic)
Agridex	*Daphnia magna*	0.00025 ppm / >1,000 ppm	2.5×10^{-7}
Competitor	*Daphnia magna*	0.00025 ppm / >100 ppm	2.5×10^{-6}

Table 6-21, above, provides the RQ values for WHCP herbicides and adjuvants for various potential invertebrate prey. Acute RQ values below the LOC of 0.05 for endangered species, and chronic RQ values below one, would indicate no adverse effect on invertebrates which endangered species may utilize for food sources. All calculated RQ values are orders of magnitude below the LOC. Thus, WHCP is not likely to adversely affect invertebrates that might be found in WHCP mats. The already low potential for toxicity effects of WHCP herbicides can be further minimized by treating water hyacinth early in the growing season, thus reducing the amount of herbicide needed.

Phytoplankton

Macroinvertebrates depend on phytoplankton, which serve as the base of the food web. Phytoplankton plays a fundamental role in primary productivity (Jassby et al. 2003). There is potential for WHCP treatments to affect algae within treatment sites, which could in turn affect macroinvertebrates. However, the potential impact of WHCP treatments on phytoplankton is minimal compared to larger scale influences on phytoplankton in the Delta. Jassby et al. (2002) examined Delta-wide primary productivity (the rate at which plants incorporate inorganic carbon into organic matter) between 1975 and 1995. During the 21-year time period, primary productivity in the Delta varied by a factor of five. Factors that contributed to the variability included: (1) decreased phytoplankton mass due to the invasion of the clam *Corbula amurensis*, (2) long-term declines in total suspended solids leading to increased water transparency and phytoplankton growth rate, (3) river inflow affecting biomass and growth rates through fluctuations in flushing and total suspended solids, and (4) an unknown factor resulting in a long-term decline in winter phytoplankton growth rate (Jassby et al. 2002).

An analysis of phytoplankton (as chlorophyll a) in the Delta and Suisun Marsh between 1996 and 2005 found increases in much of the Delta and substantial declines in Suisun Marsh (Jassby 2008). Chlorophyll a, a green pigment in plants, is used as an approximate index of algal biomass (Jassby et al. 2003). Overall, there has been a long-term declining trend in chlorophyll a from the 1970s to 2005, as well as a decline in larger-celled phytoplankton, which are preferred food sources (Kimmerer et al. 2012). Delta chlorophyll a sampling levels between 1987 and 2006 have rarely risen about the threshold level of 10 μg per liter that is considered the point at which crustacean zooplankton become food-limited (Jassby 2008, Kimmerer et al. 2012). Suisun Marsh, which is highly affected by *Corbula amurensis*, has seen even greater declines in chlorophyll a (Jassby 2008).

Changes in phytoplankton communities can result in differing nutrient values. For example, diatoms and cryptophytes are generally more nutritious for many zooplankton species than cyanobacteria (Jassby 2008). Researchers have concluded that long-term declines of phytoplankton in the Delta have contributed to long-term declines in fish abundance; however, phytoplankton decline does not appear to be a major factor in the more recent pelagic organism decline (Kimmerer et al. 2012). Vanderstukken (2012) conducted a series of experiments that demonstrated that water hyacinth plants reduced phytoplankton populations through shading, as well as alleopathic effects.

Algal toxicity studies evaluate the EC50, the concentration at which there is a 50 percent reduction in the log-phase growth after a time period (Washington DOE 2001). EC50 values higher than WHCP herbicide concentrations would indicate a potential for treatments to acutely negatively affect algal growth. **Table 6-22,** on the next page, provides several species' EC50 values for WHCP herbicides.

Washington DOE (2001) noted that 2,4-D DMA is not toxic to most aquatic algae. Washington DOE found lower EC50s for some other forms of 2,4-D such as the esters; however, WHCP utilizes the less toxic DMA form of the chemical. 2,4-D may also result in algal growth, although this may be a result of decomposing plants, rather than the herbicide (Washington DOE 2012).

SERA (2003) summarized the effects of glyphosate on a variety of algal and diatom species, and found EC50 values ranging from 7.6 mg/l to 19 mg/l. The lowest freshwater species EC50 for glyphosate was 9.08 mg/l. Pesce et al (2009) compared the effects of 10 ppb glyphosate on riverine microbial communities in spring and summer. River water was analyzed after 14 days, with 6 days of glyphosate exposure at 10 ppb, and declining glyphosate levels the last 8 days. In the spring, Pesce found no significant differences between control and treated samples on community-level end-points such as chlorophyll a content, bacterial activity or on eukaryotic and prokaryotic community composition. There were differences only in algal community composition and eukaryotic community diversity in the summer, with no significant effects on bacterial or prokaryotic communities.

Vendrell et al. (2009) (in Galhano et al. 2011) evaluated the effects of glyphosate on four microalgae species collected at Albufera Lake in Valencia (Spain). The 72-hour EC50

Table 6-22
Responses of Standard Algal, Diatom, and Cyanobacteria species to WHCP Herbicides *Page 1 of 2*

Species	Chemical	EC50 (NOEC)	Time Period	Reference*
Anabaena flosaquae (cyanobacteria)	2,4-D DMA	153 mg/l (68 mg/l)	5-days	Hughes 1990j
Selenastrum capricornutum (green algae)	2,4-D DMA	67 mg/l (19 mg/l)	5-days	Hughes 1990o
Navicula pelliculosa (freshwater diatom)	2,4-D DMA	5.28 mg/l (1.70 mg/l)	5-days	Hughes 1990d
Scenedesmus subspicatus (algae)	Aquamaster (isopropylamine salt of glyphosate)	72.9 mg/l	72 hours	SERA 2003
Selenastrum capricornutum (green algae)	Glyphosate	12.5 mg/l	4-days	USEPA 1993b
Navicula pelliculosa (freshwater diatom)	Glyphosate	39.9 mg/l	4-days	USEPA 1993b
Anabaena flosaquae (cyanobacteria)	Glyphosate	11.7 mg/l	4-days	USEPA 1993b
Chlorella fusca (algae)	Glyphosate	377 mg/l	24-hr	Faust et al. 1994
Chlorella pyrenoidosa (green algae)	Glyphosate	590 mg/l	4-days	Maule and Wright 1994
Chlorococcum hypnosporum (green algae)	Glyphosate	68 mg/l	4-days	Maule and Wright 1994
Zygnema cllindricum (green algae)	Glyphosate	88 mg/l	4-days	Maule and Wright 1994
Anabaena flosaquae (cyanobacteria)	Glyphosate	304 mg/l	4-days	Maule and Wright 1994
Scenedesmus acutus (green algae)	Glyphosate	10.2 mg/l LOEC 4 mg/l NOEC 2 mg/l	96-hr	Sanchez et al. 1997
Scenedesmus quadricauda (green algae)	Glyphosate	9.08 mg/l LOEC 4.08 mg/l NOEC 3.2 mg/l	96-hr	Sanchez et al. 1997
S. acutus, S. subspicatus, C. Chlorella vulgaris, C. saccharophila (microalgae)	Glyphosate	24.5 mg/l to 41.7 mg/l	72-hour	Vendrell et al. 2009
Scenedesmus quadricauda (green algae)	Penoxsulam (technical)	0.092 mg/l NOAEC 0.005 mg/l	96-hr Endpoint = cell density	USEPA September 2007
Scenedesmus quadricauda (green algae)	Penoxsulam (Galleon/equivalent)	0.094 mg/l NOAEC 0.009 mg/l	96-hr Endpoint = biomass	USEPA September 2007

Table 6-22

Responses of Standard Algal, Diatom, and Cyanobacteria species to WHCP Herbicides *(continued)* *Page 2 of 2*

Species	Chemical	EC50 (NOEC)	Time Period	Reference*
Scenedesmus quadricauda (green algae)	Penoxsulam degradates	> 1.0 mg/l to > 10 mg/l (same for NOAEC)	96-hr Endpoints = growth rate, biomass, none	USEPA September 2007
Navicula pelliculosa (freshwater diatom)	Penoxsulam (technical)	> 49.6 mg/l (same for NOAEC)	120-hr Endpoint = none	USEPA September 2007
Anabaena flos-aquae (blue-green algae)	Penoxsulam (technical)	0.27 mg/l NOAEC 1.94 mg/l	120-hr Endpoint = cell density, biomass	USEPA September 2007
Selenastrum capricornutum (green algae)	Imazapyr	71 mg/l	7-day Endpoint = cell density	AMEC Geomatrix 2009
*Anabaena flos-aquae** (blue-green algae)	Imazapyr	11.7 mg/l	7-day Endpoint = cell count	AMEC Geomatrix 2009
Navicula pelliculosa (freshwater diatom)	Imazapyr	NA	7-day Growth stimulation rather than inhibition	AMEC Geomatrix 2009
Navicula pelliculosa (freshwater diatom)	Imazamox	> 0.040 mg/l	120-hours Endpoint = growth reduction	USEPA 2008
Selenastrum capricornutum (green algae)	Imazamox	> 0.040 mg/l	120-hours Endpoint = growth reduction	USEPA 2008
Anabaena flos-aquae (blue-green algae)	Imazamox	> 0.040 mg/l	120-hours Endpoint = growth reduction	USEPA 2008

* 2,4-D references from Washington Department of Ecology. Herbicide Risk Assessment for the Aquatic Plant Management Final Supplemental Environmental Impact Statement. Appendix C Volume 2: 2,4-D. Washington DOE. February 2001. Publication Number 00-10-043. Glyphosate references from: Syracuse Environmental Research Associates, Inc. for USDA, Forest Service. Glyphosate – Human Health and Ecological Risk Assessment Final Report. USDA Forest Health Protection. 2003.

for the four microalgae (*Scendesmus acutus, Scendesmus subspicatus, Chlorella vulgaris*, and *Chlorella saccharophilia*) ranged from 24.5 mg/l to 41.7 mg/l, and the concentrations resulting in 10 percent growth inhibition ranged from 1.6 mg/l to 3.0 mg/l. Galhano et al. concluded that the study showed that glyphosate had low microalgae toxicity.

Toxicity testing for USEPA registration of penoxsulam found EC50 values for various microalgae that were higher than the expected penoxsulam concentration immediately following WHCP treatments (USEPA September 2007). RQ values for penoxsulam were well below the aquatic plant LOC of 1.

Aquatic plants have variable sensitive to imazapyr. The most sensitive plant is duckweed *(Lemna gibba)*, while unicellular algae are less sensitive to imazapyr, and freshwater diatoms are relatively tolerant to imazapyr (AMEC Geomatrix 2009).

Imazamox has limited toxicity to algae species (Netherland et al. 2009). EC50 values for several algal species were greater than 40 ppb, well above expected concentrations following WHCP treatments. Washington State University researchers found that imazamox had no toxic effect on sea lettuce and red algae when Clearcast was applied at 16 ounces per acre (Environ 2012).

Table 6-23, on the next page, provides the RQ values for each WHCP herbicide utilizing the lowest EC50 (and NOEC where available) and the highest immediate post-treatment level of herbicide. In all cases the RQs are well below the LOC for aquatic plants of 1.0 (USEPA 2012), including those calculations utilizing the NOEC. Thus, the toxicity data demonstrate that WHCP herbicides will not affect diatoms and algae at the base of the food web.

It is unlikely that there would be significant adverse effects to special status, resident native, or migratory fish from WHCP impacts on the Delta food web. Given the (1) low levels of herbicides utilized, (2) low toxicity of WHCP herbicides to macroinvertebrates and algae, and (3) limited treatment acreage, the potential for food web effects to impact special status fish, resident native or migratory fish, is likewise low. The already low potential for toxicity effects of WHCP herbicides can be further minimized by treating water hyacinth early in the growing season, thus reducing the amount of herbicide needed.

3. Dissolved Oxygen Effects

The WHCP could result in adverse indirect effects to special status fish, resident and migratory fish, and sensitive riparian and wetland habitats due to the rapid decay of water hyacinth following herbicide application. Decomposition of vegetative material may create a high organic carbon load, which could in turn reduce dissolved oxygen (DO) concentrations. Low DO can result in fish kills, impede migration of salmonids, and kill aquatic invertebrates. These effects in turn may, at least temporarily, impair sensitive riparian and wetland habitats. However, DWR and U.S. Bureau of Reclamation (1994) noted that in the Delta in general, constituents such as dissolved oxygen have not changed on a large enough scale to affect mobile organisms, specifically delta smelt and splittail.

Dissolved oxygen is the content of oxygen found in water. DO is determined by temperature, weather, water flow, nutrient

Table 6-23
RQ Calculations for 2,4-D and Glyphosate for Algae, Cyanobacteria, and Diatom

Herbicide	Species	EEC/EC50	RQ
2,4-D	*Navicula pelliculosa* (freshwater diatom)	0.4 mg/l /5.28 mg/l	0.08
2,4-D	*Navicula pelliculosa* (freshwater diatom)	NOEC 0.4 mg/l /1.7 mg/l	0.23
2,4-D	*Selenastrum capricornutum* (green algae)	0.4 mg/l /67 mg/l	0.006
2,4-D	*Selenastrum capricornutum* (green algae)	NOEC 0.4 mg/l /19 mg/l	0.02
2,4-D	*Anabaena flosaquae* (cyanobacteria)	0.4 mg/l / 153 mg/l	0.003
2,4-D	*Anabaena flosaquae* (cyanobacteria)	NOEC 0.4 mg/l /68 mg/l	0.006
Glyphosate	*Navicula pelliculosa* (freshwater diatom)	0.158 mg/l /39.9 mg/l	0.004
Glyphosate	*Scenedesmus quadricauda* (green algae)	0.158 mg/l /9.08 mg/l	0.017
Glyphosate	*Scenedesmus quadricauda* (green algae)	NOEC 0.158 mg/l /3.2 mg/l	0.049
Glyphosate	*Anabaena flosaquae* (cyanobacteria)	0.158 mg/l /11.7 mg/l	0.014
Penoxsulam	*Scenedesmus quadricauda* (green algae)	.002 mg/l / .094 mg/l	0.021
Penoxsulam	*Scenedesmus quadricauda* (green algae)	NOAEC .002 mg/l / .009 mg/l	0.22
Imazapyr	*Anabaena flos-aquae* (blue-green algae)	0.0056 mg/l/ 11.7 mg/l	0.0005
Imazamox	Diatom and algae	0.0056 mg/l/ 0.40 mg/l	0.14

levels, algae, and aquatic plants. Until very high oxygen levels are reached, a higher level of DO is beneficial. Fish begin to experience oxygen stress or exhibit avoidance at levels below 5 mg/liter (5 ppm). DO levels drop in warmer temperatures, and increase with precipitation, wind, and water flow. Running water, such as tidal water in the Delta, dissolves more oxygen than still water. High levels of nutrients in water reduce DO levels, while algae and aquatic plants can increase DO through photosynthesis, but decrease DO through respiration and decomposition. DO levels fluctuate throughout the day, and are typically lowest in the morning and peak in the afternoon. In deep, still waters, DO levels are lower in the hypolimnion (bottom layer of water) because there is little opportunity for oxygen replenishment from the atmosphere.

There is the potential that following herbicide treatment, the biomass of decaying water hyacinth will create a large biological oxygen demand, resulting in decreases in dissolved oxygen. These decreases in dissolved oxygen could adversely affect fish species and aquatic invertebrates present at the treatment location, and potentially impair sensitive riparian or wetland habitats. The extent of the DO impact depends on the speed at which

water hyacinth decomposes following treatment (which is herbicide dependent) and the extent to which tides and wind move decaying plants away from the original location (which is variable).

WHCP herbicide labels include provisions to address the potential for low dissolved oxygen following treatment, when appropriate. When herbicides are used according to label instructions, there will be no significant effect on DO, except to increase DO levels once the plants have completed decomposition. Label requirements related to DO impacts are as follows:

- The label for Weedar 64 (2,4-D) notes that decaying weeds use up oxygen, and recommends treating part of the infestation at one time. For example, the label recommends applying 2,4-D in lanes separated by untreated strips, and delaying treatment of these strips for 21 days, until the treated dead vegetation has decomposed
- The label for AquaMaster (glyphosate) recommends treating an area in strips when there is full coverage of the weed in impounded areas to avoid oxygen depletion. The Delta does not contain impounded waters
- The label for Galleon (penoxsulam) does not include specific provisions related to DO
- The label for Habitat (imazapyr) requires that applications be made in strips when vegetation covers a large percentage of the surface area, and restricts treatment to no more than one-half of the surface area of the water in a single operation
- The label for Clearcast (imazamox) does not include specific provisions related to DO.

Dissolved oxygen levels under water hyacinth are already low. Toft (2000) and others have found lower levels of dissolved oxygen under hyacinth canopies. Average spot measures were below 5 ppm in hyacinth, and above 5 ppm in pennywort (Toft 2000). These results were supported by a study in Texas which found lower dissolved oxygen in hyacinth compared to other aquatic weeds, and a University of California, Davis study which found dissolved oxygen levels as low as 0 ppm below a solid water hyacinth mat (Toft 2000). Toft hypothesized that lower dissolved oxygen levels explained the absence of epibenthic amphipods and isopods beneath the hyacinth canopy at one of the test sites (Toft 2000). Thus, it is likely that fish and other mobile aquatic invertebrates will avoid areas under water hyacinth mats with low dissolved oxygen, even prior to treatment (NOAA-Fisheries 2006).

To minimize the potential for negative impacts, the WHCP will implement a number of mitigation measures to reduce the potential for low DO following treatments. These mitigation measures include:

- Monitor dissolved oxygen measures pre- and post-treatment for all WHCP treatments. No treatments will be performed if DO levels are between 3 ppm and the Basin Plan limits established by the Central Valley Regional Water Quality Control Board (ranging from 5 ppm to 8 ppm)
- For each treatment site and herbicide application, follow herbicide label requirements, as specified, to reduce the potential for low DO
- When follow-up herbicide applications are required, follow herbicide label requirements, as specified, regarding the number of treatments and time between treatments.

Table 6-24
Comparison of Treatment and Post-Treatment Dissolved Oxygen Levels (in mg/l) (2011)

Site	Days Post Treatment	Treatment DO	Post-Treat DO	Difference (Post-Treatment)
2,4-D Treatments				
13	6	7.18	7.09	(0.09)
14	5	8.46	7.23	(1.23)
15	6	7.74	7.73	(0.01)
16*	6	2.06	7.03	4.97
58	6	7.06	7.15	0.09
59	4	6.92	6.98	0.06
68	6	7.86	7.97	0.11
Glyphosate Treatments				
216	7	9.80	8.40	(1.40)
217	7	7.70	6.18	(1.52)
300	5	8.50	8.00	(0.50)
301*	5	1.07	2.71	1.64
Average increase for five increased DO sites:				1.37
Average decrease for six decreased DO sites:				(0.79)

* Highlighted rows had DO levels harmful to fish prior to WHCP treatments.

WHCP tracks two sets of DO monitoring. At every herbicide application, treatment crews take DO samples immediately prior to treating, and immediately post-treatment. These levels would be expected to be similar, as they occur a few hours apart and the potential for lowering DO due to decaying water hyacinth would not occur immediately post-treatment. Data from Daily Treatment Logs support that there is no significant impact on DO immediately post-treatment. Of 719 treatments occurring between 2007 and 2011, there were 13 cases with no change in DO, 404 cases with an increase in DO (average increase of 0.8 mg/l), and 302 cases with an average decrease in DO (average decrease of 0.6 mg/l). The average pre-treatment DO was 7.9 mg/l, and the average post-treatment DO was 8.1 mg/l. The minimum allowable DO in most of the WHCP program area is 5.0 mg/l. Both pre- and post-treatment levels are well above the 5.0 mg/l considered safe for fish.

The DO monitoring that occurs with water quality sampling would be more likely to show potential decreases in DO, as post-treatment sampling occurs several days after treatment, when plant death symptoms are starting to occur. However, representative DO monitoring data from 2011 shows that herbicide treatments do not significantly impact DO.

The data in **Table 6-24,** above, provide 2011 treatment and post-treatment DO levels taken at the time of water quality sampling,

on the day of treatment, and between four and seven days post-treatment. In five cases, DO levels increased. Note that the most significant increase occurred at Site 16. Site 16 DO was at an extremely low 2.06 mg/l prior to treatment (a level resulting in stress and avoidance for fish), and DO increased by six days post-treatment to 7.03 mg/l, a level safe for fish. In the other instance of extremely low DO prior to treatment, DO increased from 1.07 mg/l to 2.71 mg/l by five days post-treatment. In these two critical cases where DO levels prior to treatment were below levels safe for fish, DO levels improved following WHCP treatments. The average decrease in DO among the six 2011 monitoring sites with decreased DO was 0.79 mg/l, and in all cases where DO decreased, it was still well above the Basin Plan minimum of 5.0 mg/l. DBW and USDA-ARS will continue to monitor pre- and post-treatment DO levels.

E. Direct and Indirect Effects of Interrelated or Interdependent Actions

There are no interrelated or interdependent actions of WHCP.

F. Effects Considering Environmental Baseline and Cumulative Effects

The WHCP operates as a minor component within a complex and highly-manipulated environmental baseline. Below, we briefly describe the potential WHCP effects within the context of the environmental baseline and cumulative effects.

Environmental Baseline Effects

The WHCP is a legislatively mandated program intended to control invasive water hyacinth within the Delta and its tributaries. Left unchecked, water hyacinth has significant negative effects on the ecosystem and human activity. The subsidies section, below, describes the potential benefits resulting from water hyacinth control.

The EDCP is also a legislatively mandated program intended to control invasive *Egeria densa* within the Delta. *Egeria densa* is a submerged plant, and usually does not grow within water hyacinth mats. EDCP treatments and WHCP treatments occur within the same overall time period and within the same project area. However, individual site treatments will typically take place in different locations, and also at different time periods within the respective treatment seasons. Most EDCP treatments occur During April and May. Most WHCP treatments occur between June and August. The same two-person crews conduct treatments, and usually only work on one program in any given day. In addition, both EDCP and WHCP treatment herbicides exhibit low toxicity to fish, macroinvertebrates, and algae, and are thus not likely to adversely affect listed fish species, either individually or in combination.

The growing dominance of invasive species in the Delta increases the need for control programs such as the WHCP. Invasive species in the Delta have already altered the food web, habitats, and water quality. Long-term changes resulting from climate change are likely to further enhance conditions for many invasive species, including water hyacinth. Invasive species control programs such as the

WHCP and EDCP will be increasingly necessary in order try to prevent further degradation of the Delta ecosystem.

The impact of agricultural practices in the Delta are substantial, and have become part of the economic and ecological landscape of the Delta. WHCP activities operate at a significantly smaller scale, and with significantly lesser impacts, as it relates to pesticide use, the area in which agriculture and WHCP operations overlap. The WHCP seeks to minimize herbicide use and to use herbicides with improved toxicity profiles in order to reduce the potential for additional pesticide burden on the Delta beyond that resulting from agricultural use.

Delta water quality has been degraded by historical and current human activities. The WHCP operates within the guidelines of the SWB and Regional Water Board plans and NPDES permits. WHCP activities follow water quality guidelines and mitigation measures, do not result in further decline of beneficial uses of the Delta, and in fact promote beneficial uses of the Delta.

Cumulative Effects

The Spongeplant Control Program (AB 1540, Buchanan, Chapter 188, Statues of 2102) was approved by the Legislature on August 15, 2012, and signed by the Governor on August 27, 2012. AB 1540 adds responsibility for an additional invasive plant to DBW and USDA-ARS existing WHCP and EDCP programs. The Spongeplant Control Program will operate in similar locations within the Delta as the WHCP, and with similar treatment approaches. The additive effects of spongeplant control to existing WHCP and EDCP activities is not likely to result in significantly greater or new potential impacts to listed species.

The impacts of climate change in the Delta will likely result in gradual changes to Delta ecosystems and to the political landscape in which the Delta is managed. These changes will occur over the long-term, and WHCP activities will not substantially change the nature of climate change impacts on listed species.

Effects of urbanization on the Delta and listed species will likely increase the need for WHCP treatment activities. Further decline of water quality and habitats may increase potential for water hyacinth invasions. In addition, increased recreational activity in the Delta could result in the need for additional water hyacinth control.

WHCP activities, taken within the context of the environmental baseline and cumulative effects, are not likely to result in additional adverse effects to listed species. There is potential for these environmental baseline and cumulative effects to increase the need and importance of water hyacinth control to help maintain natural habitats within a changing and degrading Delta.

G. Subsidies of WHCP

The discussion of the environmental baseline in Section 5 describes negative consequences of invasive water hyacinth in the Delta. These negative impacts include: ecosystem engineer effects such as out-competing native plants, negative effects on native zooplankton and plankton, low dissolved oxygen under water hyacinth mats, negative effects on birds, providing mosquito

habitat, as well as impediments to navigation and water pumps. The inverse of these negative effects of water hyacinth is that controlling water hyacinth in the Delta and its tributaries reduces and/or eliminates negative consequences.

As with all invasive species control programs, WHCP activities seek to balance the need to minimize the potential effects of control with the benefits of control. While one core intent of the WHCP is to improve navigation in the Delta, the broader benefits of the WHCP to the Delta ecosystem are likely more significant and more lasting. Below, we focus on subsidies of the WHCP in the context of listed fish species. By removing invasive water hyacinth, WHCP activities leads to three primary interrelated subsides: (1) improved native habitats, (2) food web benefits, and (3) increased dissolved oxygen.

Improved Native Habitats

Water hyacinth has been labeled an ecosystem engineer due to its impact on sediment, water clarity, ecosystem diversity, and dissolved oxygen (Mount et al. 2012). Control of water hyacinth in Delta waterways expands habitat suitable for native species. Thus, long-term impacts of water hyacinth control on special status plant species and sensitive habitats are likely to be beneficial.

There are potential positive impacts to special status plants, sensitive habitats, and wetlands from the WHCP. Water hyacinth clogs waterways and reduces overall habitat for native plants (CALFED 2000). Dense patches of water hyacinth shade out habitat and outcompete native aquatic vegetation, including Mason's lilaeopsis (CALFED 2000). Shultz and Dibble (2012) noted that water hyacinth outcompetes native vegetation, decreases dissolved oxygen levels, leads to shifts in macroinvertebrate communities and fish diets, and fosters non-indigenous amphipods. These outcomes reduce overall habitat quality in water hyacinth mats. Katz et al. (2012) found that improving habitat quality is an important factor in fostering genetic diversity and resistance to the impacts of climate change in salmonids. A study examining the impacts of removal of the invasive weed water milfoil in Minnesota lakes found that dense monospecific mats were detrimental to habitat, and native plant diversity increased when the weeds were removed (Kovalenko et al. 2009).

As water hyacinth is controlled in the Delta, it will be important to monitor the overall impact on habitats. There is uncertainty as to how habitats will respond to removal of water hyacinth. During 2008, some areas which had previously been heavily infested with water hyacinth, became heavily infested with native pennywort. While Toft (2000, 2003) demonstrated improved habitat under pennywort as compared to water hyacinth, mono-specific mats of any single species raise concerns.

Habitat improvements following water hyacinth treatment are likely to be similar to treatments for other invasive species. Allan (2006) evaluated native plant biomass following treatment of alligatorweed with imazapyr or triclopyr. Early and heavy treatment of alligatorweed with either herbicide resulted in a greater biomass of native plants later in the treatment season

and after one year. Allan found that the timing and rate of application influenced return of native plants, a result that is likely species and ecosystem-specific.

Food Web Benefits

There are potential positive impacts to the Delta food web resulting from the WHCP. Rapid growth and invasion of water hyacinth reduces open water habitat and impairs wetlands and sensitive riparian habitats, altering the natural food web. Toft et al. (2003) found that removal of water hyacinth also resulted in loss of the non-native amphipod *Crangonyx floridanus*, a species which was not prevalent in fish diets. Toft suggested that once an invasive species such as water hyacinth is removed from the system, "aspects of the community can return to a more natural pre-invasion state" (Toft et al. 2003).

Kovalenko et al. (2009) found that removal of invasive water milfoil from Minnesota lakes did not affect stomach fullness or fish feeding. Fish were able to find preferred prey in their changing habitat. Factors other than macrophytes are greater influences on macroinvertebrates. In a study evaluating 27 years of benthic assemblage data at four locations along the salinity gradient in the Delta, Peterson and Vayssieres (2010) found that macroinvertebrate composition was heavily influenced by changes in salinity and by the invasion of the clam, *Corbula amurensis* that began in 1987.

Increased Dissolved Oxygen

There are positive impacts related to dissolved oxygen that will result from the WHCP. Dissolved oxygen levels at treatment sites will increase, improving fish habitat, once dead water hyacinth has decayed or floated away. Removing large patches of water hyacinth will allow DO levels to increase, thus enhancing the ability of fish to move unimpeded in Delta waters. It could be argued that such a benefit outweighs the impact of potential short-term localized decreases in dissolved oxygen following WHCP treatment (which should be mitigated by treatment protocols).

Dissolved oxygen levels under water hyacinth are already low. Toft (2000) and others have found lower levels of dissolved oxygen under hyacinth canopies. Average spot measures were below 5 ppm in hyacinth, and above 5 ppm in pennywort (Toft 2000). These results were supported by a study in Texas which found lower dissolved oxygen in hyacinth compared to other aquatic weeds, and a University of California, Davis study which found dissolved oxygen levels as low as 0 ppm below a solid water hyacinth mat (Toft 2000). Toft hypothesized that lower dissolved oxygen levels explained the absence of epibenthic amphipods and isopods beneath the hyacinth canopy at one of the test sites (Toft 2000). Thus, it is likely that fish and other mobile aquatic invertebrates will avoid areas under water hyacinth mats with low dissolved oxygen, even prior to treatment (NOAA-Fisheries 2006).

Low dissolved oxygen levels are detrimental to fish. Newcomb and Pierce (2010) evaluated the adverse effects of low DO on salmon and steelhead. Their study, conducted for the Bay-Delta Office of DWR, documented numerous direct and indirect adverse effects, including mortality

(at 2 to 2.5 mg/l), reduced swimming performance (at below 6.5 to 7 mg/l), reduced growth (at below 4 to 5 mg/l), impaired development, reduced spawning success, reduced fecundity and fertility, altered behavior, increased susceptibility to predation, increased susceptibility to parasites and pathogens, and increased susceptibility to contaminants. Because salmonids are migratory, they will typically avoid low DO levels (Newcomb and Pierce 2010); however, to the extent that removal of water hyacinth increases DO where water hyacinth mats were previously present, WHCP will increase the amount of suitable habitat.

H. Alternative Actions

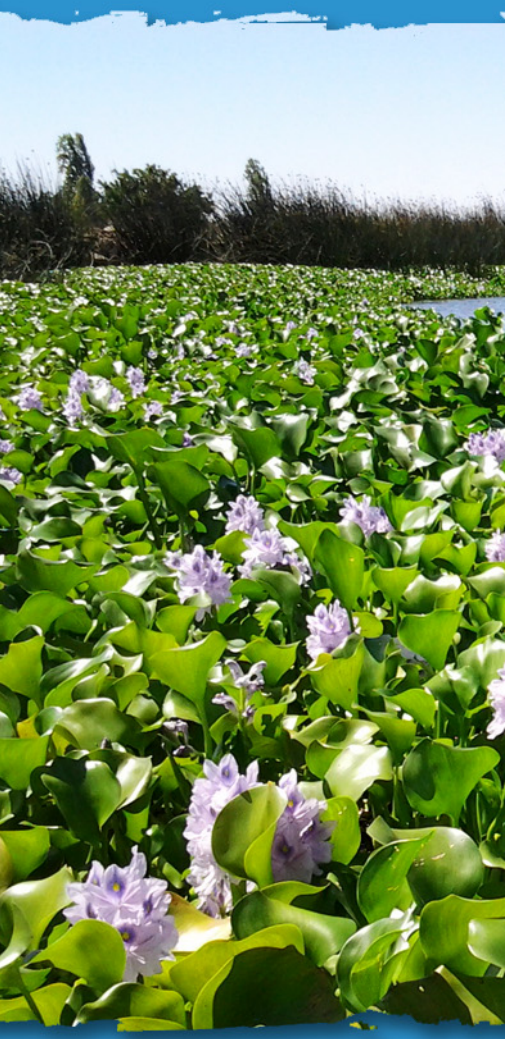
In developing the WHCP Programmatic Environmental Impact Report (PEIR) (DBW 2009), DBW evaluated alternative actions, including:

- Integrated management emphasizing chemical treatment with limited handpicking and herding, and continued assessment of biological controls (the selected alternative)
- Chemical control only
- Handpicking only
- Biological control only
- Mechanical removal only
- No program.

With the exception of the no program alternative, each of the above alternatives consists of implementing only one of the integrated management approaches described in Section 3. The result is that the alternatives do not provide the comprehensive approach necessary to control water hyacinth, nor the flexibility to minimize potential control impacts. DBW evaluated each of the above alternatives, and determined them to be inferior to the selected integrated management approach. Alternatives were determined to be either less effective in controlling water hyacinth, have greater negative impacts, or both. The WHCP, as described in Section 3 of this biological assessment is similar to the integrated management alternative in the PEIR, with the addition of limited mechanical removal, which now has improved efficacy due to the availability of better equipment.

Section 7

Other Relevant Information



7. Other Relevant Information

This section of the Biological Assessment provides supporting information and studies. All materials referenced below are provided in the **WHCP Biological Assessment – Supplemental Materials Binder.**

A. Studies to Evaluate Treatment Alternatives, Efficacy, and to Identify New Treatment Options

Since 2001, USDA-ARS and DBW have conducted or sponsored a number of additional studies to evaluate treatment alternatives, efficacy, and identify new treatment options. Many of these additional studies were requested as part of previous USFWS or NMFS consultations. The following six (6) studies are provided in the Supplemental Materials Binder in Tabs 6 through 11. In addition, PDF files of each report are provided on the accompanying CD-ROM:

- *Acute Oral and Dermal Toxicity of Aquatic Herbicides and a Surfactant to Garter Snakes,* Robert C. Hosea, California Department of Fish and Game (2004) (Tab 6)
- *Chronic Toxicities of Herbicides Used to Control Water Hyacinth and Brazilian Elodea on Neonate Cladoceran and Larval Fathead Minnow,* Frank Riley and Sandra Finlayson, California Department of Fish and Game (2004) (Tab 7)
- *Acute Toxicities of Herbicides Used to Control Water Hyacinth and Brazilian Elodea on Larval Delta Smelt and Sacramento Splittail,* Frank Riley and Sandra Finlayson, California Department of Fish and Game (2004) (Tab 8)
- *Ceriodaphnia dubia (water flea) Static Definitive Chronic Toxicity Test Data (7-day) for Exposure to Various Aquatic Herbicides,* California Department of Fish and Game, Aquatic Toxicology Laboratory (2003) (Tab 9)
- *Pogonichthys macrolepitdotus (Sacramento Splittail) Static Definitive Acute Toxicity Test Data (96-hour) for Exposure to Various Aquatic Herbicides,* California Department of Fish and Game, Aquatic Toxicology Laboratory (2003) (Tab 10)
- *Mapping Invasive Plant Species in the Sacramento-San Joaquin Delta Region Using Hyperspectral Imagery,* Susan L. Ustin, Ph.D., et al, Center for Spatial Technologies and Remote Sensing (CSTARS), California Space Institute Center of Excellence (CalSpace), UC Davis (2004) (Tab 11).

B. Herbicide and Adjuvant Labels

Labels and Material Safety Data Sheets (MSDS) for each of the five WHCP herbicides and the two adjuvants are provided in the Supplemental Materials Binder in Tabs 13 through 19. Labels are provided in the following order:

- Weedar 64 Label (Tab 13)
- Weedar 64 MSDS (Tab 13)
- Aquamaster Label (Tab 14)
- Aquamster MSDS (Tab 14)
- Galleon SC Label (Tab 15)
- Galleon SC MSDS (Tab 15)
- Habitat Label (Tab 16)
- Habitat MSDS (Tab 16)
- Clearcast Label (Tab 17)
- Clearcast MSDS (Tab 17)
- Agridex Label (Tab 18)
- Agridex MSDS (Tab 18)
- Competitor Label (Tab 19)
- Competitor MSDS (Tab 19).

C. WHCP Site Priority List

Exhibit 7-1, with an example on the next page, summarizes characteristics of each WHCP treatment site. The full exhibit is provided in Tab 20 of the Supplemental Materials Binder. The total site acres included in Exhibit 7-1 is 50,442 acres. This is less than the combined Delta and Southern Site water acres of 67,799, because there are water acres that have not recently been treated within the WHCP. These sites could be treated in the future if they become infested with water hyacinth. This exhibit provides the following information for each WHCP site (note: rankings represent scores for the 2012 season, and are subject to change; WHCP will update rankings prior to the start of each treatment season):

- Site Number[1]
- Site Area (used to assign treatment crews)
- Years in which the site was treated between 2007 and 2011
- Total acres treated with 2,4-D and/or glyphosate between 2007 and 2011
- Months during which the site was treated between 2007 and 2011
- County
- Location name
- Water type (tidal or riverine)
- Site water acres
- Whether the site is a nursery (score of 3 if yes, score of 0 if no); nursery sites are also shaded in the spreadsheet
- The current (pre-2012 season) level of infestation, with 3 = high and 0 = none
- The potential for infestation, with 3 = high and 0 = none
- The navigability public safety and commercial/ recreational needs of the site, with 3 = high and 0 = none
- The combined ranking priority score as of pre-2012 season (maximum = 12).

[1] Sites in Exhibit 7-1 labeled as "a" or "b", for example "17a" and "17b" were split along county lines in 2012. The treatment year, acres, and months data are listed under one site, but apply to both sites.

Exhibit 7-1 (EXAMPLE)

WHCP Treatment Site List, History, Characteristics, and 2012 Season Ranking

#	Site Number	Area	Treated in 2007	Treated in 2008	Treated in 2009	Treated in 2010	Treated in 2011	Total 2,4-D Acres Treated (All Years)	Total Glyphosate Acres Treated (All Years)	Treated in April	Treated in May	Treated in June	Treated in July	Treated in Aug.
1	1	8				2010		3.00	-					Aug.
2	2	8												
3	3	8												
4	4	8												
5	5	8	2007					1.50						
6	6	8	2007			2010		4.50						Aug.
7	7	3	2007					2.75						Aug.
8	8	3	2007	2008	2009	2010	2011	36.50	8.33				July	Aug.
9	9	2	2007		2009	2010		7.00					July	Aug.
10	10	2	2007	2008	2009	2010	2011	45.50	11.67				July	Aug.
11	11	2		2008	2009	2010	2011	6.50	2.33				July	Aug.
12	12	2	2007	2008	2009	2010	2011	10.75	27.67				July	Aug.
13	13	2	2007	2008	2009	2010	2011	75.38	13.20				July	Aug.
14	14	2	2007	2008	2009	2010	2011	91.00	2.67				July	Aug.
15	15	2	2007	2008	2009	2010	2011	80.75	0.67				July	Aug.
16	16	3	2007	2008	2009	2010	2011	46.13	0.67				July	Aug.
17	17a	3	2007	2008	2009	2010	2011	33.25	1.00				July	Aug.
18	17b	3												
19	18a	6	2007	2008	2009	2010	2011	59.25	5.33				July	Aug.
20	18b	6												
21	19a	6	2007	2008	2009	2010	2011	45.00	4.00				July	Aug.
22	19b	6												
23	20	6	2007	2008	2009	2010	2011	48.25	2.67			June	July	Aug.
24	21a	6	2007		2009	2010		13.00	0.67				July	Aug.
25	21b	6												
26	22	7	2007			2010		3.00	-					Aug.
27	23a	7												
28	23b	7	2007			2010		9.75					July	Aug.
29	24a	7	2007					2.25					July	Aug.
30	24b	7												
31	25	3	2007		2009			3.75						
32	26	3	2007	2008	2009	2010	2011	41.25	2.67			June	July	Aug.

Green shading represents nursery sites.

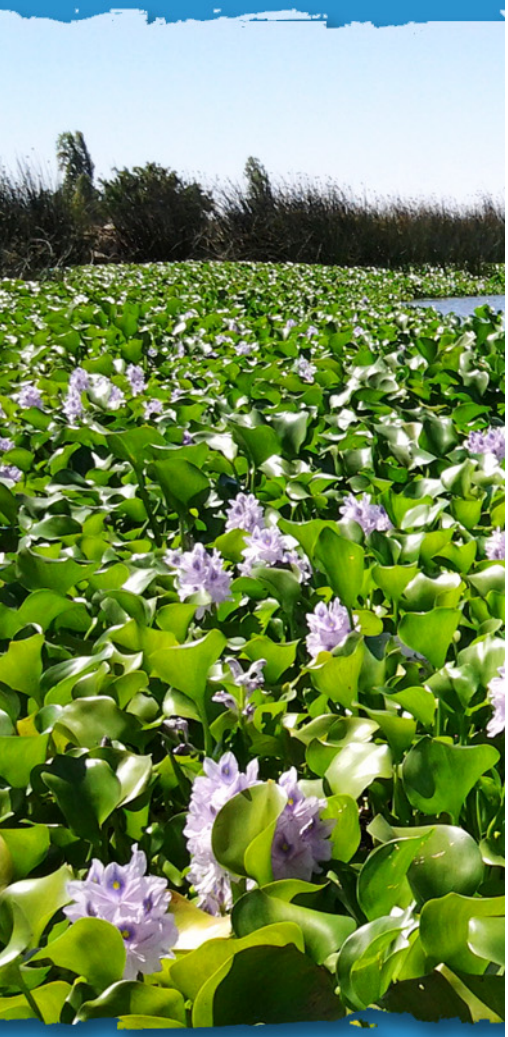
D. WHCP Treatment Maps

This subsection identifies maps provided to highlight treatment sites and times for each of the current and potential new treatment herbicides. The following full-size maps are provided in Tab 20 of the Supplemental Materials Binder:

- **Exhibit 7-2 – 2012 Water Hyacinth Control Program Priority Sites (North):** identifies high priority nursery sites, high priority sites, medium priority nursery sites, and medium priority sites for the northern sites. WHCP will prepare a similar map prior to the start of each treatment season
- **Exhibit 7-3 – 2012 Water Hyacinth Control Program Priority Sites (South):** identifies high priority nursery sites, high priority sites, medium priority nursery sites, and medium priority sites for the southern sites. WHCP will prepare a similar map prior to the start of each treatment season
- **Exhibit 7-4 – Water Hyacinth Control Program 2,4-D Treatment Guidelines (North):** illustrates the allowable treatment dates using 2,4-D in the legal Delta
- **Exhibit 7-5 – Water Hyacinth Control Program 2,4-D Treatment Guidelines (South):** illustrates the allowable treatment dates using 2,4-D in the southern sites
- **Exhibit 7-6 – Water Hyacinth Control Program Delta Smelt Avoidance Sites (March to June):** illustrates the sites that will not be treated until July 1, in order to avoid potential impacts to delta smelt
- **Exhibit 7-7 – Water Hyacinth Control Program Approved non-2,4-D Herbicides (North):** illustrates the sites with early start dates in the northern Delta
- **Exhibit 7-8 – Water Hyacinth Control Program Approved non-2,4-D Herbicides (South):** illustrates the sites with early start dates south of the Delta

Section 8

Conclusions



8. Conclusions

This section of the Biological Assessment (BA) provides USDA-ARS and DBW's conclusions regarding the overall effects of WHCP on the following USFWS and NMFS listed species and critical habitats:

- USFWS Listed Species and Critical Habitats
  1. Threatened delta smelt and Threatened delta smelt Critical Habitat
  2. Threatened giant garter snake
  3. Threatened valley elderberry longhorn beetle
  4. Candidate Threatened San Francisco Bay-Delta Distinct Population Segment (DPS) of longfin smelt
- NMFS Listed Species and Critical Habitats
  1. Endangered Sacramento River winter-run Chinook salmon and Endangered Sacramento River winter-run Chinook salmon Critical Habitat
  2. Threatened Central Valley spring-run Chinook salmon and Threatened Central Valley spring-run Chinook salmon Critical Habitat
  3. Threatened Central Valley steelhead and Threatened Central Valley steelhead Critical Habitat
  4. Threatened Southern DPS of North American green sturgeon and Threatened Southern DPS of North American green sturgeon Critical Habitat.

The following effects determinations are based on analyses of the exposure and responds of species and habitat to the stressors resulting from WHCP, as described in prior sections of this BA. This section is organized as follows:

A. Conclusions Regarding USFWS Listed Species and Critical Habitats

B. Conclusions Regarding NMFS Listed Species and Critical Habitats.

A. Conclusions Regarding USFWS Listed Species and Critical Habitats

1. Threatened delta smelt and Threatened delta smelt Critical Habitat

Threatened delta smelt

The information and analysis presented in this BA is the basis of the finding that WHCP warrants an effect determination of Likely to Adversely Affect for Threatened delta smelt. However, effects are likely to be temporary and relatively minor.

A determination of Likely to Adversely Affect is warranted based on the following rationale:

- Delta smelt are documented in the WHCP project action area
- WHCP will operate in selected sites during March through June, and throughout the Delta, which includes critical habitat, from July through November. Adult delta smelt move from the LSZ into Delta spawning habitats during the winter, and spawn in the Delta between February and May. Juveniles generally migrate back to the LSZ by summer
- WHCP will modify habitat conditions in the project action area
- Current and proposed WHCP herbicides and adjuvants will not result in direct acute, chronic, or sub-chronic toxic effects to delta smelt based on treatment application rates, scientific studies, and resulting extremely low RQ values. However, the potential for interaction of herbicides with other contaminants in the project area is unknown
- Current and proposed WHCP herbicides and adjuvants will not result in direct acute or chronic toxic effects to macroinvertebrate prey species that delta smelt depend on for food supply
- Current and proposed WHCP herbicides and adjuvants will not negatively affect primary productivity in the project action area, thus WHCP will not in turn affect macroinvertebrate food supply
- WHCP operations will not result in reduced dissolved oxygen that could harm delta smelt; herbicides are applied following label and treatment protocols to reduce DO impacts. However, delta smelt eggs and larvae are semi-buoyant and could be directed by river flows into low DO areas. Removal of water hyacinth will lead to increased dissolved oxygen levels in formerly infested areas
- Most WHCP treatments occur during the summer months, when delta smelt are in the LSZ, including Suisun Marsh, where salinity levels are too high for water hyacinth growth
- WHCP will avoid treatment at sites in which delta smelt are likely to be present (based on IEP surveys and regular discussions with USFWS). During the March through June period in which delta smelt may still be spawning and/or rearing in areas of the north or west Delta, WHCP will not treat in those sites (121a, 121b, 122 to 131, 262, 267, 272, and 277)
- WHCP boat operations and mechanical harvesting operations could negatively impact delta smelt that might be in treatment sites, even though fish are not likely to be present in dense water hyacinth mats.

Threatened delta smelt Critical Habitat

The information and analysis presented in this BA is the basis of the finding that WHCP warrants an effect determination of

Likely to Adversely Affect for Threatened delta smelt Critical Habitat. However, effects of WHCP on delta smelt critical habitat are likely to be temporary and minor.

A determination of Likely to Adversely Affect is warranted based on the following rationale:

- Critical habitat has been designated within the project action area
- Adult delta smelt move from the LSZ into Delta spawning habitats during the winter, and spawn in the Delta between February and May. Juveniles generally migrate back to the LSZ by summer
- WHCP will involve modification of aquatic habitat (removal of water hyacinth and potential for native plant loss due to overspray)
- WHCP will not impact the PCEs for delta smelt critical habitat, including water quality, river flow, and salinity
- WHCP could temporarily degrade delta smelt spawning habitat PCE, but is not likely to impact spawning activities. Delta smelt are thought to spawn in nearshore habitats and shallow edges in sloughs. Most spawning occurs between February and May; WHCP treatments will not take place until July in areas thought to be common spawning grounds. Delta smelt are thought to lay eggs at night, and eggs are adhesive and stick close to the bottom on sand and pebbles, further minimizing potential for adverse effects
- WHCP operations will not destroy or adversely modify critical habitat because the potential for loss of native plants due to overspray is highly unlikely given WHCP operation practices, mitigation measures, and the fact that water hyacinth grows in dense mono-specific mats. WHCP operations will ultimately improve habitat as native plants can reestablish in waters that were previously infested with water hyacinth.

2. Threatened giant garter snake

The information and analysis presented in this BA is the basis of the finding that WHCP warrants an effect determination of May Affect, Not Likely to Adversely Affect for Threatened giant garter snake.

A determination of May Affect is warranted based on the following rationale:

- Giant garter snakes are present within WHCP treatment sites
- WHCP will modify habitat conditions in the project action area.

A determination of Not Likely to Adversely Affect is warranted based on the following rationale:

- While giant garter snakes have been seen in habitats adjacent to some WHCP treatment sites, this species is extremely shy and not likely to be present during WHCP treatments. Giant garter snakes bask on grassy banks and on branches over the water's edge where herbicide applications will not occur. WHCP treatment crews conduct environmental observation surveys prior to conducting treatments, and do not treat if giant garter snakes are present. Over a ten-year period, treatment crews twice identified snakes that might have been giant garter snakes (or common garter snakes), and did not treat at that location. However, because giant garter snakes are shy, they might have been present but not seen
- WHCP treatment protocols include mitigation measures to minimize potential for herbicides to reach banks, if giant garter snakes were unseen, but present
- Current and proposed WHCP herbicides and adjuvants will not result in direct acute or chronic toxic effects

to giant garter snakes based on treatment application rates, scientific studies, and resulting extremely low RQ values

- WHCP boat operations and mechanical harvesting operations are unlikely to negatively impact giant garter snakes that might be in treatment sites, and snakes are not likely to be present in dense water hyacinth mats
- Hand-picked water hyacinth on levee banks during the October to April period are highly unlikely to be inadvertently be disposed on levee crevices in which giant garter snakes are located.

3. Threatened valley elderberry longhorn beetle

The information and analysis presented in this BA is the basis of the finding that WHCP warrants an effect determination of May Affect, Not Likely to Adversely Affect for Threatened valley elderberry longhorn beetle.

A determination of May Affect is warranted based on the following rationale:

- Valley elderberry plants are located within the project action area, on the shoreline adjacent to WHCP treatment sites
- Valley elderberry shrubs that were sprayed with WHCP herbicides could be harmed.

A determination of Not Likely to Adversely Affect is warranted based on the following rationale:

- WHCP treatments require a 250 foot buffer zone to protect valley elderberry shrubs, host plant of valley elderberry longhorn beetles, along the shoreline. In addition, treatments are conducted downwind of valley elderberry shrubs
- WHCP treatment protocols also include provisions to reduce overspray (such as large droplets and no treatments when wind is above 10 mph (7 mph in Contra Costa County)
- WHCP environmental scientists conduct pre- and post-season surveys to determine if valley elderberry shrubs are harmed by treatments, and have seen no impacts in more than ten years of surveys.

4. Candidate Threatened San Francisco Bay-Delta DPS of longfin smelt

The information and analysis presented in this BA is the basis of the finding that WHCP warrants an effect determination of May Jeopardize, Not Likely to Jeopardize for Candidate Threatened San Francisco Bay-Delta DPS of longfin smelt.

A determination of May Jeopardize is warranted based on the following rationale:

- San Francisco Bay-Delta DPS of longfin smelt are documented in the WHCP project action area
- WHCP will operate in selected sites during March through June, and throughout the Delta, from July through November. Longfin smelt are believed to spawn in freshwater in the lower reaches of the Sacramento River and San Joaquin River between November and June, but are most likely to be found in the Delta between November and March. Larvae, juveniles, and adults have been found in the Delta; however, most of this species' life cycle is spent in brackish or marine waters

- WHCP will modify habitat conditions in the project action area.

A determination of Not Likely to Jeopardize is warranted based on the following rationale:

- Current and proposed WHCP herbicides and adjuvants will not result in direct acute, chronic, or sub-chronic toxic effects to longfin smelt based on treatment application rates, scientific studies, and resulting extremely low RQ values
- Current and proposed WHCP herbicides and adjuvants will not result in direct acute or chronic toxic effects to macroinvertebrate prey species that longfin smelt depend on for food supply
- Current and proposed WHCP herbicides and adjuvants will not negatively affect primary productivity in the project action area, thus WHCP will not in turn affect macroinvertebrate food supply
- WHCP operations will not result in reduced dissolved oxygen that could harm longfin smelt; herbicides are applied following label and treatment protocols to reduce DO impacts. Removal of water hyacinth will lead to increased dissolved oxygen levels in formerly infested areas
- WHCP will not conduct treatments during the three of the five months when longfin smelt are most likely to be present in the Delta, and will not treat at locations in which longfin smelt are likely to be present in November and March
- WHCP boat operations and mechanical harvesting operations are unlikely to negatively impact longfin smelt, as fish are not likely to be present in dense water hyacinth mats, or in the unlikely event that they are present, to remain when boats approach.

B. Conclusions Regarding NMFS Listed Species and Critical Habitats

1. Endangered Sacramento River winter-run Chinook salmon and Endangered Sacramento River winter-run Chinook salmon Critical Habitat

Endangered Sacramento River winter-run Chinook salmon

The information and analysis presented in this BA is the basis of the finding that WHCP warrants an effect determination of May Affect, Not Likely to Adversely Affect for Endangered Sacramento River winter-run Chinook salmon.

A determination of May Affect is warranted based on the following rationale:

- Sacramento River winter-run Chinook salmon are documented in the WHCP project action area
- WHCP will operate in selected sites during March through June, and throughout the Delta, which includes critical habitat, from July through November. Adult Sacramento River winter-run Chinook salmon migrate through the Delta to up river spawning sites between November and June. Juveniles spend approximately 40 days emigrating through the Delta, and are primarily present from November through early May
- WHCP will modify habitat conditions in the project action area.

A determination of Not Likely to Adversely Affect is warranted based on the following rationale:

- Current and proposed WHCP herbicides and adjuvants will not result

in direct acute, chronic, or sub-chronic toxic effects to Chinook salmon based on treatment application rates, scientific studies, and resulting extremely low RQ values

- Current and proposed WHCP herbicides and adjuvants will not result in direct acute or chronic toxic effects to macroinvertebrate prey species that salmonids depend on for food supply
- Current and proposed WHCP herbicides and adjuvants will not negatively affect primary productivity in the project action area, thus WHCP will not in turn affect macroinvertebrate food supply
- WHCP operations will not result in reduced dissolved oxygen that could harm salmonids; herbicides are applied following label and treatment protocols to reduce DO impacts. Removal of water hyacinth will lead to increased dissolved oxygen levels in formerly infested areas
- WHCP will not conduct treatments at sites in which Chinook salmon are present (based on IEP surveys and regular discussions with NMFS) during the March through June period in which adults or juvenile Sacramento River winter-run Chinook salmon may be migrating through the Delta
- WHCP boat operations and mechanical harvesting operations are unlikely to negatively impact salmonid species, as fish are not likely to be present in dense water hyacinth mats, or in the unlikely event that they are present, to remain when boats approach.

Endangered Sacramento River winter-run Chinook salmon Critical Habitat

The information and analysis presented in this BA is the basis of the finding that WHCP warrants an effect determination of May Affect, Not Likely to Adversely Affect for Endangered Sacramento River winter-run Chinook salmon Critical Habitat.

A determination of May Affect is warranted based on the following rationale:

- Critical habitat has been designated within the project action area
- Adult Sacramento River winter-run Chinook salmon migrate through the Delta to up river spawning sites between November and June. Juveniles spend approximately 40 days emigrating through the Delta, and are primarily present from November through early May
- Three of the four PCEs for winter-run Chinook salmon critical habitat are within WHCP treatment sites: freshwater rearing habitat, freshwater migration corridors, and estuarine areas
- WHCP will involve modification of aquatic habitat (removal of water hyacinth and potential for native plant loss due to overspray).

A determination of Not Likely to Adversely Affect is warranted based on the following rationale:

- WHCP will not degrade freshwater rearing habitat characteristics, which include: habitat complexity, adequate food supply, and protection from predators. Habitats modified with dense water hyacinth do not exhibit habitat complexity; removal of water hyacinth through WHCP activities provides opportunities for native plants to reestablish in those areas, thus increasing habitat complexity. WHCP activities will not impact food supply or protection from predators in freshwater rearing habitats
- WHCP will not degrade freshwater migration corridor habitat

characteristics, which include: waterways free from obstruction, water quality and quantity, natural cover, and food supply. WHCP activities could improve migration corridors if those corridors are infested with water hyacinth. WHCP implements a fish passage protocol to reduce potential for impacts to migration. WHCP mitigation measures to treat only portions of fully infested areas at one time will maintain DO levels and allow fish to move through treated areas. Ultimately, removal of water hyacinth from migration corridors will improve DO. WHCP activities will not impact water quality or quantity, natural cover, or food supply in freshwater migration corridors

- Water hyacinth will only grow in estuarine areas with saline levels less than 2ppt. In estuarine areas where water hyacinth grows, WHCP will not degrade estuarine habitat characteristics, which include: waterways free from obstruction, water quality and quantity, and natural cover. WHCP activities could improve estuarine areas if those areas are infested with water hyacinth. WHCP mitigation measures to treat only portions of fully infested areas at one time will maintain DO levels and allow fish to move through treated areas. Ultimately, removal of water hyacinth from estuarine areas will improve DO. WHCP activities will not impact water quality or quantity, or natural cover in estuarine areas
- WHCP operations will not destroy or adversely modify critical habitat because the potential for loss of native plants due to overspray is unlikely given WHCP operation practices, mitigation measures, and the fact that water hyacinth grows in dense mono-specific mats. Chinook salmon are not found in water hyacinth mats, and water hyacinth does not provide the complex and productive habitat favored by the species. WHCP operations will ultimately improve habitat as native plants can reestablish in waters that were previously infested with water hyacinth
- WHCP will not conduct treatments at sites in which Chinook salmon are present (based on IEP surveys and regular discussions with NMFS) during the March through June period in which adults or juvenile Sacramento River winter-run Chinook salmon may be migrating through the Delta.

2. Threatened Central Valley spring-run Chinook salmon and Threatened Central Valley spring-run Chinook salmon Critical Habitat

Threatened Central Valley spring-run Chinook salmon

The information and analysis presented in this BA is the basis of the finding that WHCP warrants an effect determination of May Affect, Not Likely to Adversely Affect for Threatened Central Valley spring-run Chinook salmon.

A determination of May Affect is warranted based on the following rationale:

- Central Valley spring-run Chinook salmon are documented in the WHCP project action area
- WHCP will operate in selected sites during March through June, and throughout the Delta, which includes critical habitat, from July through November. Adult Central Valley spring-run Chinook salmon migrate through the Delta to up river spawning sites between January and June. Most

juveniles emigrate through the Delta from November through early May. Some fish may over-summer in spawning grounds and not emigrate until the onset of intense fall storms

- WHCP will modify habitat conditions in the project action area.

A determination of Not Likely to Adversely Affect is warranted based on the following rationale:

- Current and proposed WHCP herbicides and adjuvants will not result in direct acute, chronic, or sub-chronic toxic effects to Chinook salmon based on treatment application rates, scientific studies, and resulting extremely low RQ values
- Current and proposed WHCP herbicides and adjuvants will not result in direct acute or chronic toxic effects to macroinvertebrate prey species that salmonids depend on for food supply
- Current and proposed WHCP herbicides and adjuvants will not negatively affect primary productivity in the project action area, thus WHCP will not in turn affect macroinvertebrate food supply
- WHCP operations will not result in reduced dissolved oxygen that could harm salmonids; herbicides are applied following label and treatment protocols to reduce DO impacts. Removal of water hyacinth will lead to increased dissolved oxygen levels in formerly infested areas
- WHCP will not conduct treatments at sites in which Chinook salmon are present (based on IEP surveys and regular discussions with NMFS) during the March through June period in which adults or juvenile Central Valley spring-run Chinook salmon may be migrating through the Delta
- WHCP boat operations and mechanical harvesting operations are unlikely to negatively impact salmonid species, as fish are not likely to be present in dense water hyacinth mats, or in the unlikely event that they are present, to remain when boats approach.

Threatened Central Valley spring-run Chinook salmon Critical Habitat

The information and analysis presented in this BA is the basis of the finding that WHCP warrants an effect determination of May Affect, Not Likely to Adversely Affect for Threatened Central Valley spring-run Chinook salmon Critical Habitat.

A determination of May Affect is warranted based on the following rationale:

- Critical habitat has been designated within the project action area
- Adult Central Valley spring-run Chinook salmon migrate through the Delta to up river spawning sites between January and June. Most juveniles emigrate through the Delta from November through early May. Some fish may over-summer in spawning grounds and not emigrate until the onset of intense fall storms
- Three of the four PCEs for spring-run Chinook salmon critical habitat are within WHCP treatment sites: freshwater rearing habitat, freshwater migration corridors, and estuarine areas
- WHCP will involve modification of aquatic habitat (removal of water hyacinth and potential for native plant loss due to overspray).

A determination of Not Likely to Adversely Affect is warranted based on the following rationale:

- WHCP will not degrade freshwater rearing habitat characteristics, which include: habitat complexity, adequate food supply, and protection from predators. Habitats modified with dense water hyacinth do not exhibit habitat complexity; removal of water hyacinth through WHCP activities provides opportunities for native plants to reestablish in those areas, thus increasing habitat complexity. WHCP activities will not impact food supply or protection from predators in freshwater rearing habitats
- WHCP will not degrade freshwater migration corridor habitat characteristics, which include: waterways free from obstruction, water quality and quantity, natural cover, and food supply. WHCP activities could improve migration corridors if those corridors are infested with water hyacinth. WHCP implements a fish passage protocol to reduce potential for impacts to migration. WHCP mitigation measures to treat only portions of fully infested areas at one time will maintain DO levels and allow fish to move through treated areas. Ultimately, removal of water hyacinth from migration corridors will improve DO. WHCP activities will not impact water quality or quantity, natural cover, or food supply in freshwater migration corridors
- Water hyacinth will only grow in estuarine areas with saline levels less than 2ppt. In estuarine areas where water hyacinth grows, WHCP will not degrade estuarine habitat characteristics, which include: waterways free from obstruction, water quality and quantity, and natural cover. WHCP activities could improve estuarine areas if those areas are infested with water hyacinth. WHCP mitigation measures to treat only portions of fully infested areas at one time will maintain DO levels and allow fish to move through treated areas. Ultimately, removal of water hyacinth from estuarine areas will improve DO. WHCP activities will not impact water quality or quantity, or natural cover in estuarine areas
- WHCP operations will not destroy or adversely modify critical habitat because the potential for loss of native plants due to overspray is unlikely given WHCP operation practices, mitigation measures, and the fact that water hyacinth grows in dense mono-specific mats. Chinook salmon are not found in water hyacinth mats, and water hyacinth does not provide the complex and productive habitat favored by the species. WHCP operations will ultimately improve habitat as native plants can reestablish in waters that were previously infested with water hyacinth
- WHCP will not conduct treatments at sites in which Chinook salmon are present (based on IEP surveys and regular discussions with NMFS) during the March through June period in which adults or juvenile Central Valley spring-run Chinook salmon may be migrating through the Delta.

3. Threatened Central Valley steelhead and Threatened Central Valley steelhead Critical Habitat

Threatened Central Valley steelhead

The information and analysis presented in this BA is the basis of the finding that WHCP warrants an effect determination of <u>May Affect, Not Likely to Adversely Affect</u> for Threatened Central Valley steelhead.

A determination of May Affect is warranted based on the following rationale:

- Central Valley steelhead are documented in the WHCP project action area
- WHCP will operate in selected sites during March through June, and throughout the Delta, which includes critical habitat, from July through November. Central Valley steelhead migrate through the Delta to up river spawning sites between August and March. Juveniles usually remain in fresh water for the first year, and then migrate through the Delta to the ocean between November and May. Steelhead are found in the Delta predominantly during migration, but may use the lower reaches of the Sacramento River and Delta for rearing
- WHCP will modify habitat conditions in the project action area.

A determination of Not Likely to Adversely Affect is warranted based on the following rationale:

- Current and proposed WHCP herbicides and adjuvants will not result in direct acute, chronic, or sub-chronic toxic effects to steelhead based on treatment application rates, scientific studies, and resulting extremely low RQ values
- Current and proposed WHCP herbicides and adjuvants will not result in direct acute or chronic toxic effects to macroinvertebrate prey species that steelhead depend on for food supply
- Current and proposed WHCP herbicides and adjuvants will not negatively affect primary productivity in the project action area, thus WHCP will not in turn affect macroinvertebrate food supply
- WHCP operations will not result in reduced dissolved oxygen that could harm steelhead; herbicides are applied following label and treatment protocols to reduce DO impacts. Removal of water hyacinth will lead to increased dissolved oxygen levels in formerly infested areas
- WHCP will not conduct treatments at sites in which Central Valley steelhead are present (based on IEP surveys and regular discussions with NMFS) during the March and August periods in which adults or juvenile Central Valley steelhead may be migrating through the Delta
- WHCP boat operations and mechanical harvesting operations are unlikely to negatively impact steelhead, as fish are not likely to be present in dense water hyacinth mats, or in the unlikely event that they are present, to remain when boats approach.

Threatened Central Valley steelhead Critical Habitat

The information and analysis presented in this BA is the basis of the finding that WHCP warrants an effect determination of May Affect, Not Likely to Adversely Affect for Threatened Central Valley steelhead Critical Habitat.

A determination of May Affect is warranted based on the following rationale:

- Critical habitat has been designated within the project action area
- Central Valley steelhead migrate through the Delta to up river spawning sites between August and March. Juveniles usually remain in fresh water for the first year, and then migrate through the Delta to the ocean between November and May. Steelhead are found in the Delta predominantly

during migration, but may use the lower reaches of the Sacramento River and Delta for rearing

- Three of the four PCEs for Central Valley steelhead critical habitat are within WHCP treatment sites: freshwater rearing habitat, freshwater migration corridors, and estuarine areas
- WHCP will involve modification of aquatic habitat (removal of water hyacinth and potential for native plant loss due to overspray).

A determination of Not Likely to Adversely Affect is warranted based on the following rationale:

- WHCP will not degrade freshwater rearing habitat characteristics, which include: habitat complexity, adequate food supply, and protection from predators. Habitats modified with dense water hyacinth do not exhibit habitat complexity; removal of water hyacinth through WHCP activities provides opportunities for native plants to reestablish in those areas, thus increasing habitat complexity. WHCP activities will not impact food supply or protection from predators in freshwater rearing habitats
- WHCP will not degrade freshwater migration corridor habitat characteristics, which include: waterways free from obstruction, water quality and quantity, natural cover, and food supply. WHCP activities could improve migration corridors if those corridors are infested with water hyacinth. WHCP implements a fish passage protocol to reduce potential for impacts to migration. WHCP mitigation measures to treat only portions of fully infested areas at one time will maintain DO levels and allow fish to move through treated areas. Ultimately, removal of water hyacinth from migration corridors will improve DO. WHCP activities will not impact water quality or quantity, natural cover, or food supply in freshwater migration corridors
- Water hyacinth will only grow in estuarine areas with saline levels less than 2ppt. In estuarine areas where water hyacinth grows, WHCP will not degrade estuarine habitat characteristics, which include: waterways free from obstruction, water quality and quantity, and natural cover. WHCP activities could improve estuarine areas if those areas are infested with water hyacinth. WHCP mitigation measures to treat only portions of fully infested areas at one time will maintain DO levels and allow fish to move through treated areas. Ultimately, removal of water hyacinth from estuarine areas will improve DO. WHCP activities will not impact water quality or quantity, or natural cover in estuarine areas
- WHCP operations will not destroy or adversely modify critical habitat because the potential for loss of native plants due to overspray is unlikely given WHCP operation practices, mitigation measures, and the fact that water hyacinth grows in dense mono-specific mats. Steelhead are not found in water hyacinth mats, and water hyacinth does not provide the complex and productive habitat favored by the species. WHCP operations will ultimately improve habitat as native plants can reestablish in waters that were previously infested with water hyacinth
- WHCP will not conduct treatments at sites in which Central Valley steelhead are present (based on IEP surveys and regular discussions with NMFS) during the March and August periods in which adults or juvenile Central Valley steelhead may be migrating through the Delta.

4. Threatened Southern DPS of North American green sturgeon and Threatened Southern DPS of North American green sturgeon Critical Habitat

Threatened Southern DPS North American green sturgeon

The information and analysis presented in this BA is the basis of the finding that WHCP warrants an effect determination of May Affect, Not Likely to Adversely Affect for Threatened Southern DPS North American green sturgeon.

A determination of May Affect is warranted based on the following rationale:

- Threatened Southern DPS of North American green sturgeon are documented in the WHCP project action area
- Southern DPS North American green sturgeon juveniles (two to three years of age) inhabit the Delta. Adult Southern DPS North American green sturgeon migrate through the Delta to Upper Sacramento River spawning grounds between mid-February and May
- WHCP will modify habitat conditions in the project action area.

A determination of Not Likely to Adversely Affect is warranted based on the following rationale:

- Current and proposed WHCP herbicides and adjuvants will not result in direct acute, chronic, or sub-chronic toxic effects to green sturgeon based on treatment application rates, scientific studies, and resulting extremely low RQ values
- Current and proposed WHCP herbicides and adjuvants will not result in direct acute or chronic toxic effects to macroinvertebrate and fish prey species that green sturgeon depend on for food supply
- Current and proposed WHCP herbicides and adjuvants will not negatively affect primary productivity in the project action area, thus WHCP will not in turn affect macroinvertebrate food supply
- WHCP operations will not result in reduced dissolved oxygen that could harm green sturgeon; herbicides are applied following label and treatment protocols to reduce DO impacts. Removal of water hyacinth will lead to increased dissolved oxygen levels in formerly infested areas
- WHCP boat operations and mechanical harvesting operations are unlikely to negatively impact green sturgeon, as fish are not likely to be present in dense water hyacinth mats, or in the unlikely event that they are present, to remain when boats approach.

Threatened Southern DPS North American green sturgeon Critical Habitat

The information and analysis presented in this BA is the basis of the finding that WHCP warrants an effect determination of May Affect, Not Likely to Adversely Affect for Threatened Southern DPS North American green sturgeon Critical Habitat.

A determination of May Affect is warranted based on the following rationale:

- Critical habitat has been designated within the project action area
- Adult Southern DPS North American green sturgeon migrate through the Delta to up river spawning sites between mid-February and May. Juveniles typically spend their second and third years in the Delta

- WHCP will involve modification of aquatic habitat (removal of water hyacinth and potential for native plant loss due to overspray).

A determination of Not Likely to Adversely Affect is warranted based on the following rationale:

- WHCP will not degrade critical habitat PCEs for green sturgeon, which include: food resources (benthic invertebrates and fish), water flow, water quality, migration corridors, water depth diversity, or sediment quality. WHCP activities will not impact water flow or water depth diversity. WHCP activities could improve migration corridors if those corridors are infested with water hyacinth. WHCP implements a fish passage protocol to reduce potential for impacts to migration. WHCP mitigation measures to treat only portions of fully infested areas at one time will maintain DO levels and allow fish to move through treated areas. Ultimately, removal of water hyacinth from migration corridors will improve DO. WHCP activities will not negatively impact water quality or food sources. WHCP activities will not negatively affect sediment quality. Of the five current and potential WHCP herbicides, only glyphosate binds readily to sediment, where it is then degraded by microorganisms. None of the herbicides are readily adsorbed to sediment, and will not result in sediment concentrations that would be detrimental to green sturgeon
- WHCP operations will not destroy or adversely modify critical habitat because the potential for loss of native plants due to overspray is unlikely given WHCP operation practices, mitigation measures, and the fact that water hyacinth grows in dense mono-specific mats. WHCP operations will ultimately improve habitat as native plants can reestablish in waters that were previously infested with water hyacinth.

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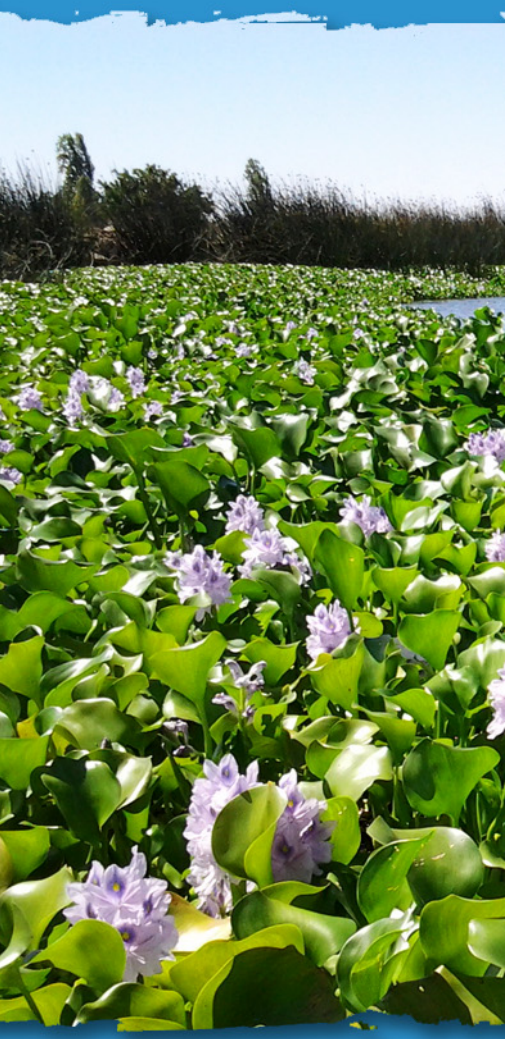
Section 9

List of Documents



9. List of Documents

This section of the Biological Assessment identifies reports and documents provided as part of this consultation. These documents are provided under separate cover or within the WHCP Biological Assessment – Supplemental Materials Binder.

A. Materials Provided in the Supplemental Materials Binder

The following documents are provided in Tabs 1 to 5 and Tab 12, of the Supplemental Materials Binder:

- NMFS WHCP Letter of Concurrence 2012/01731, and USFWS WHCP Biological Opinion 81410-2011-F-0035 (Tab 1)
- WHCP Annual Report – 2011 (Tab 2)
- WHCP Environmental Observations and Weed Survey Form (Tab 3)
- WHCP Fish Passage Protocol (Tab 4)
- Selected Scientific Literature (Tab 5)
- Statewide NPDES permit (June 27, 2012 Draft) (Tab 12).

B. Materials Provided on CD-ROM

The following materials are also provided in the Supplemental Materials Binder:

- WHCP Programmatic Environmental Impact Report (PEIR) – CD-ROM
- PDF of this WHCP Biological Assessment and Selected Scientific Literature and Studies – CD-ROM.

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Section 10

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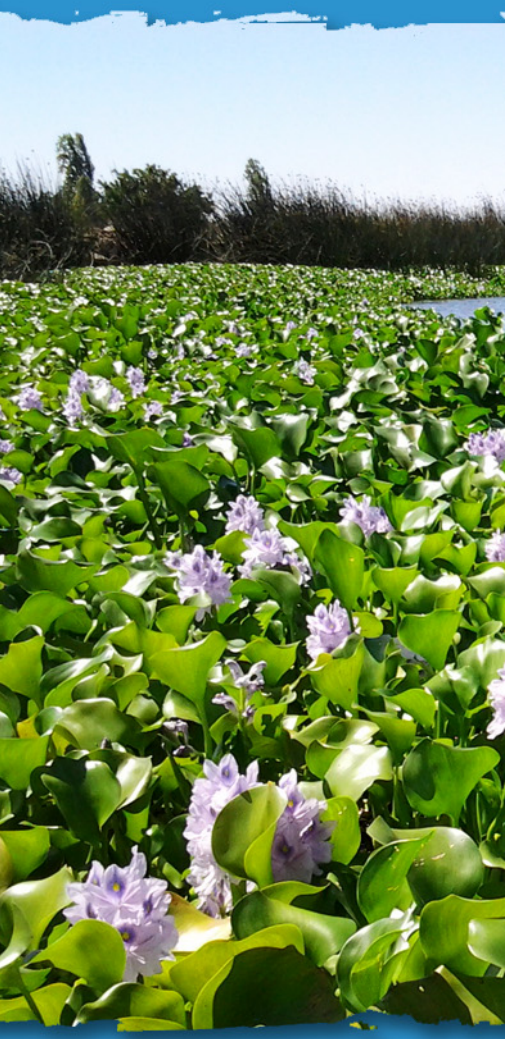

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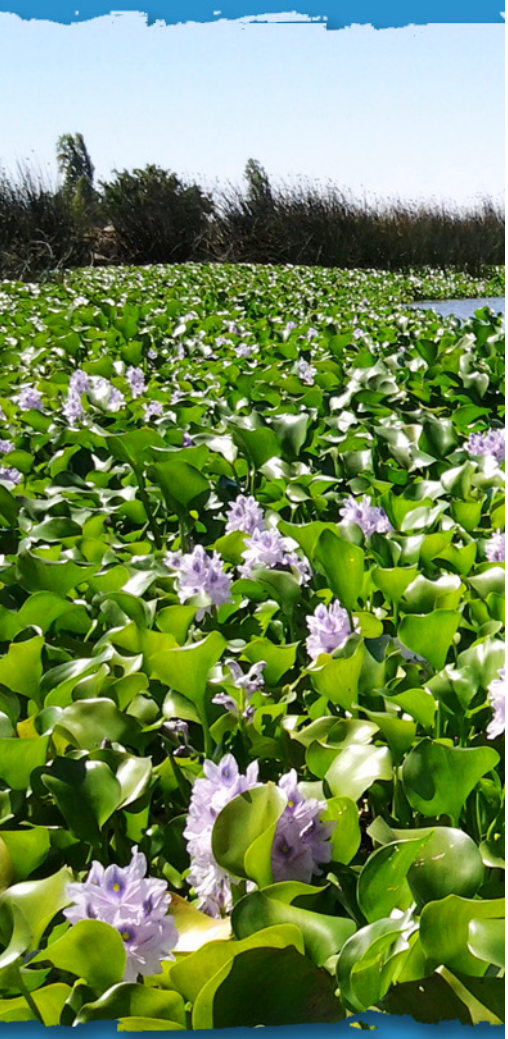
Section 11

List of Contacts/ Contributors/Preparers



11. List of Contacts/ Contributors/Preparers

The following individuals were contacted and interviewed during the course of preparing this Biological Assessment:

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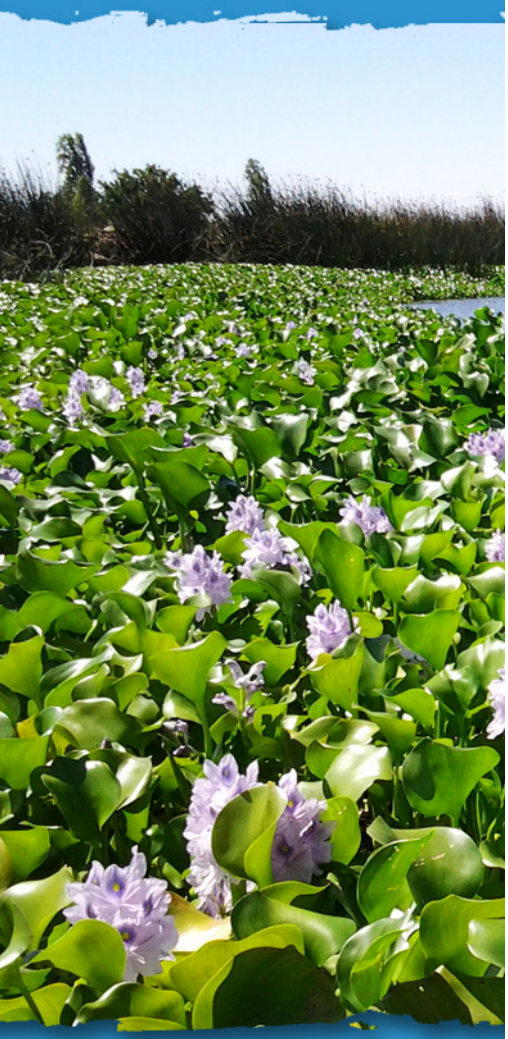

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