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SECTION 3

Status of Species

3.1 AQUATIC SPECIES

3.1.1 Delta Smelt

3.1.1.1 Listing Status and Designated Critical Habitat

The USFWS listed the delta smelt as threatened under the federal ESA on March 5, 1993, based upon its dramatically-reduced abundance, threats to its habitat, and the inadequacy of regulatory mechanisms then in effect (58 FR 12854). In 2004, a 5-year status review reaffirmed the need to retain the delta smelt as a threatened species (USFWS 2004). In February 2007, the USFWS and the California Fish and Game Commission were jointly petitioned to list the species as endangered under ESA and California Endangered Species Act (CESA), respectively (Center for Biological Diversity et al. 2006 and 2007). This re-listing was requested because of a substantial step decline in the abundance of this species beginning in 2002 from an already depressed population status, with no recovery in subsequent years, in spite of favorable hydrologic conditions. The Service is currently considering information to determine if the listing status of delta smelt should be upgraded from threatened to endangered. On March 4, 2009, the State of California listed the delta smelt as a state endangered species.

The USFWS designated critical habitat on December 19, 1994 (59 FR 65256). Critical habitat encompasses essentially all waters of the legal Delta extending downstream to western Suisun Marsh and Suisun Bay (USFWS 1994). The Action Area is entirely within designated critical habitat (Figure 3-1).

3.1.1.2 Life History

Delta smelt (*Hypomesus transpacificus*) are slender-bodied fish, about 2 to 3 inches long, in the Osmeridae family (smelts). The species is endemic to the Sacramento-San Joaquin Delta. Delta smelt are euryhaline fish that typically rear in shallow (<10 feet), open waters of the estuary (Moyle 2002). They are mostly found within the salinity range of 2-7 ppt (parts per thousand) and have been collected from estuarine waters up to 14 ppt (Moyle 2002, USFWS 2007a). The species generally lives about one year, although a small proportion of the population may live to spawn in its second year (Moyle 2002, Bennett 2005).

Beginning in September and October delta smelt slowly but actively migrate from the X2 (2 ppt salinity isohaline) region of the estuary to upper Delta spawning areas. The upstream migration of delta smelt seems to be triggered or cued by abrupt changes in flow and turbidity associated with the first flush of winter precipitation (Grimaldo et al., accepted manuscript cited in USFWS 2008) but can also occur after very high flood flows have receded. Grimaldo et al. (accepted manuscript) noted salvage often occurred when total inflows exceeded over 25,000 cfs or when turbidity was elevated above 12 NTU (CCF station).

Spawning has been reported as occurring primarily from late February through June (Moyle 2002, Bennett 2005), with a peak in April and May. Delta smelt spawn widely throughout the Delta, but their specific spawning distribution varies from year to year depending on flow conditions. Spawning cannot be easily observed and specific spawning locations are unknown, although the relative importance of spawning areas can be inferred from the catch of larval delta smelt in 20mm townets. The majority of spawning activity occurs in the northern (Sacramento River) side of the delta in the vicinity of Cache Slough and Liberty Island. A minority of adults spawn in the south delta in the vicinity of Franks Tract and the lower San Joaquin River.

Eggs are demersal and adhere to the substrate or plants over which they are spawned. They hatch after 9 to 14 days. Fish absorb their yolk sac and develop jaws over the next 4 to 5 days, then begin to feed on small planktonic organisms. Once this stage of their life begins, they are expected to drift with the predominant currents, perhaps exercising some control through vertical migrations in the water column (Bennett 2005). They become post-larvae about a month later, and juveniles about one month after that (Bennett 2005).

Delta smelt live together in loose aggregations, but they are not strongly schooling (Moyle 2002). They feed on zooplankton throughout their lives, mainly copepods, cladocerans, amphipods and some larval fish (Moyle et al. 1992, Bennett 2005). Primary productivity and the resulting zooplankton biomass are important factors determining growth and survival in the summer and fall (Kimmerer 2008).

3.1.1.3 Distribution

The delta smelt is endemic to the Sacramento-San Joaquin Delta, including Suisun Bay, but is generally most abundant in the western Delta and eastern Suisun Bay (Honker Bay) (Moyle et al. 1992). Distribution varies seasonally with freshwater outflow. Generally, the species inhabits areas of the San Francisco Estuary upstream of the X2. This biologically productive area meets specific requirements for freshwater inflow, salinity, water temperature, and shallow open water habitat.

Delta smelt spawn widely throughout the Delta, but their specific spawning distribution varies from year to year depending on flow conditions. The majority of spawning activity occurs in the northern (Sacramento River) side of the delta in the vicinity of Cache Slough and Liberty Island, with some spawning in the vicinity of Franks Tract and the lower San Joaquin River. In wetter years spawning occurs in Napa River, Suisun Bay and Suisun Marsh (Sweetnam 1991, Wang 1991, Hobbs et al. 2006).

3.1.1.4 Abundance

Population trends of delta smelt were assessed based on data from three sampling programs:

- Fall midwater trawl (FMWT) conducted in most years since 1962 between September and December to sample late juveniles and adults (Figure 3-2). An abundance index derived from the FMWT is the primary measure for tracking changes in the delta smelt population (Moyle et al. 1992, Sweetnam 1999).
- Summer Townet Survey (TNS) conducted each spring since 1959 (except for 1966 to 1968) to assess the population and distribution of juvenile delta smelt (Figure 3-3). The FMWT combined with subsequent Summer TNS give an index of reproductive success over the spring spawning period.
- 20 mm survey conducted each spring since 1995 to assess the distribution of late larval stage delta smelt (Figure 3-4).

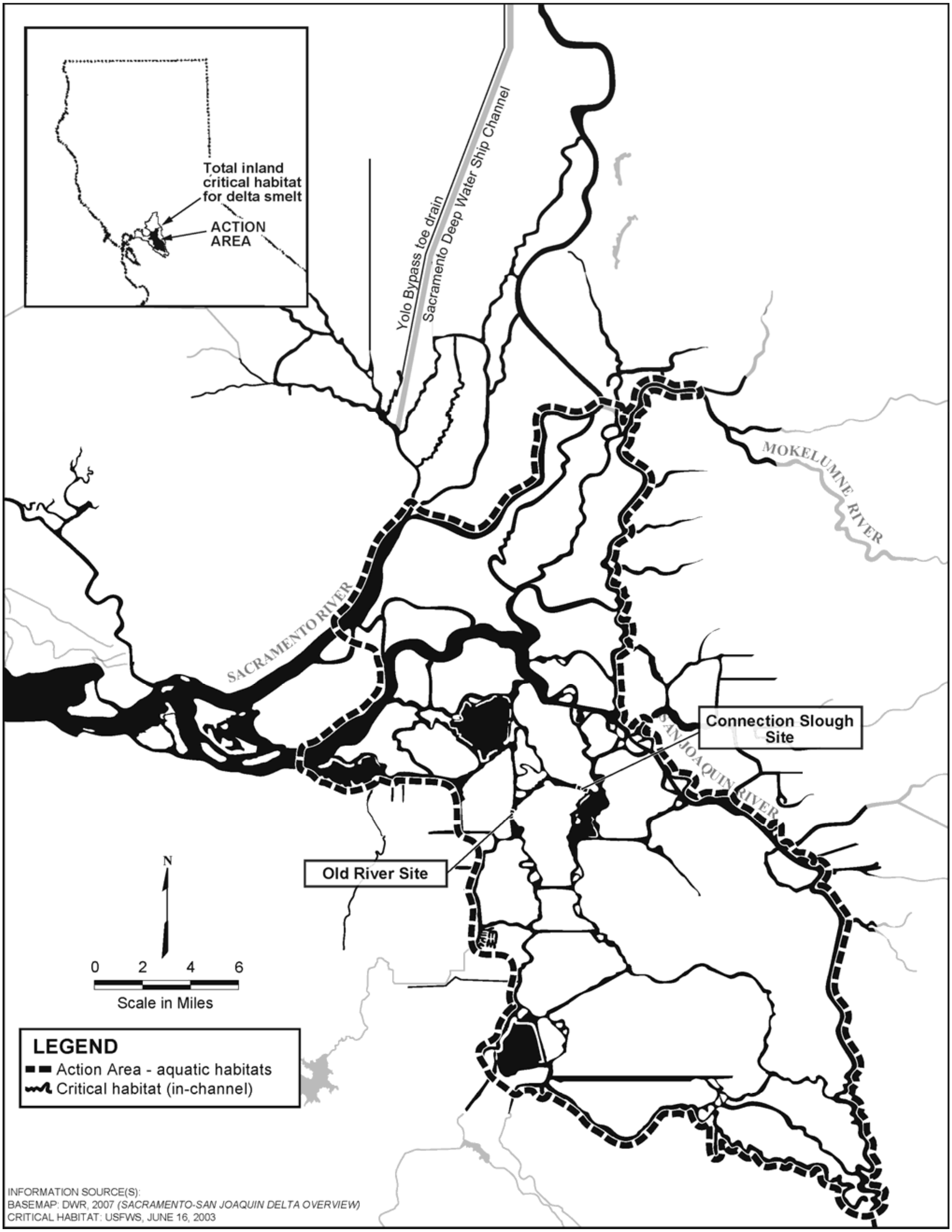
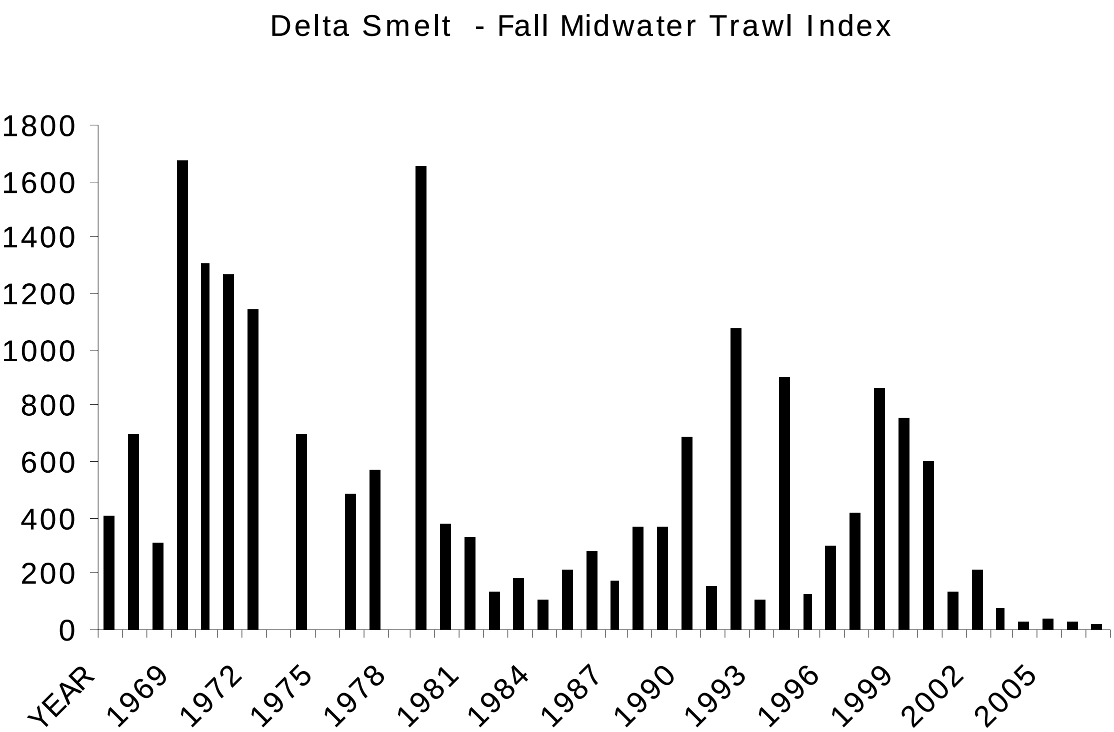
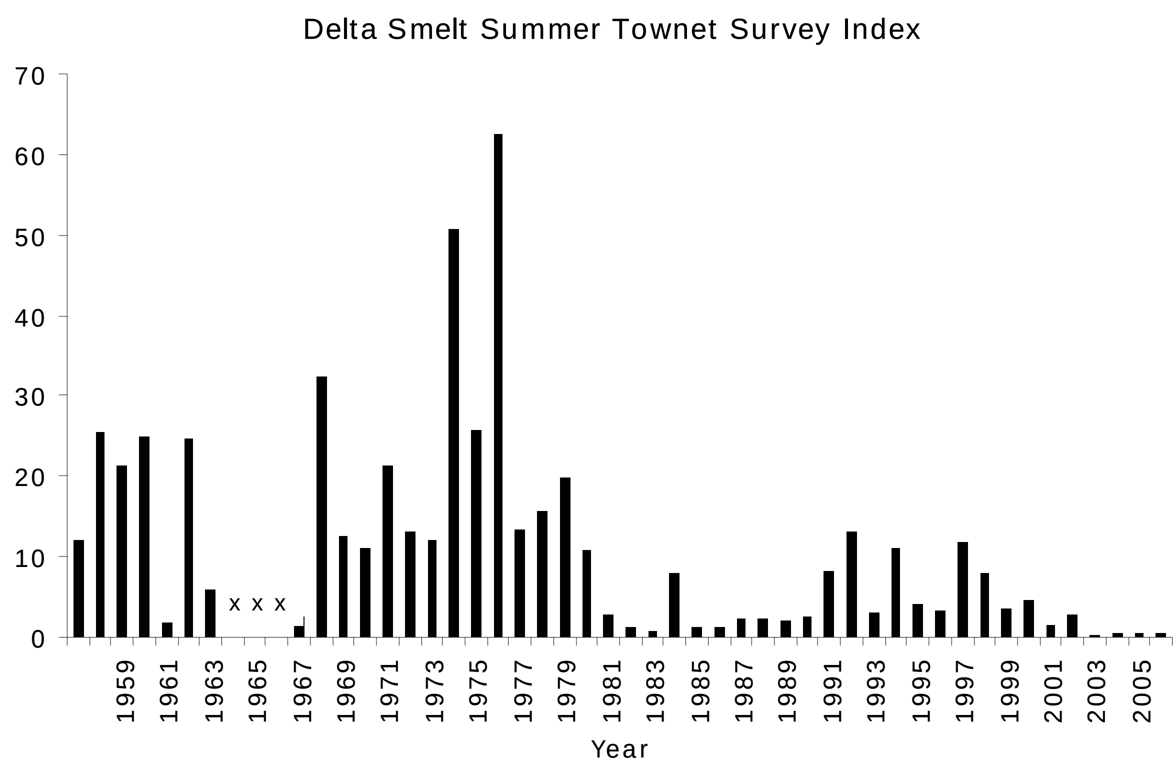


Figure 3-1 Action Area and Designated Critical Habitat for Delta Smelt



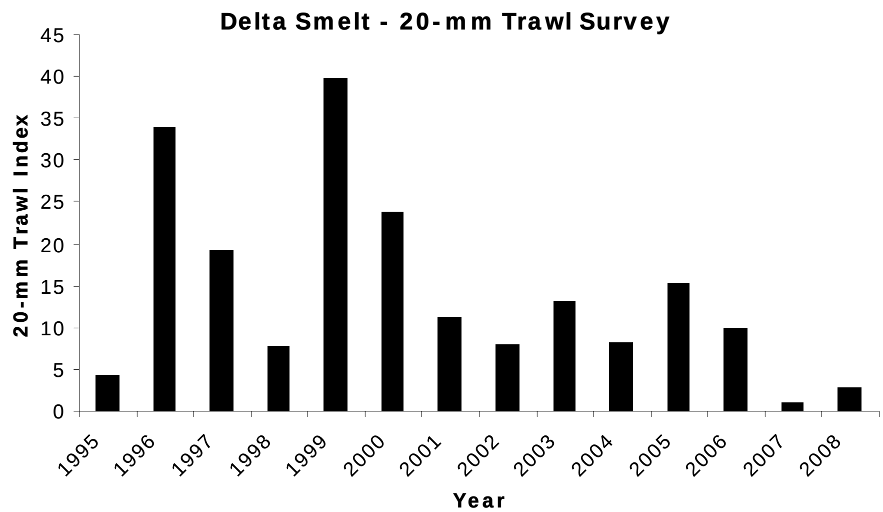
Source: CDFG Bay Delta Region, http://www.delta.dfg.ca.gov/data/mwt/charts.asp

Figure 3-2 Fall Midwater Trawl (FMWT) Abundance Indices for Delta Smelt, 1967 – 2008



Source: CDFG Bay Delta Region, http://www.delta.dfg.ca.gov/data/townet/indices.asp?species=3

Figure 3-3 Summer Townet Survey (TNS) Abundance Indices for Delta Smelt, 1969-2008 (x = no data collected)



Source: CDFG Bay Delta Region, ftp://ftp.delta.dfg.ca.gov/Delta%20Smelt/

Figure 3-4 20-mm Trawl Survey Abundance Indices for Delta Smelt, 1995 – 2008

The population of delta smelt has declined substantially since the late 1970s. Since 2000, their populations have been at or near historic low values. The FMWT derived indices have ranged from a high of 1,653 in 1970 to a low of 27 in 2005 (Figure 3-2). For comparison, TNS-derived indices have ranged from a high of 62.5 in 1978 to a low of 0.3 in 2005 (Figure 3-3). Although the peak high and low values have occurred in different years, the TNS and FMWT indices show a similar pattern of delta smelt relative abundance; higher prior to the mid-1980s and very low in the past seven years. From 1969-1981, the mean delta smelt TNS and FMWT indices were 22.5 and 894, respectively. Both indices suggest the delta smelt population declined abruptly in the early 1980s (Moyle et al. 1992). From 1982-1992, 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 1993-2002 period. However, delta smelt numbers have trended precipitously downward since about 2000. The total number of delta smelt collected in the 20-mm survey also shows a substantial decrease since 2001 (Figure 3-4). Currently, the delta smelt population indices (FMWT and TNS) are two orders of magnitude smaller that historical highs (USFWS 2008).

The diminished abundance of delta smelt coincides with historic low populations of other pelagic species including longfin smelt, threadfin shad, and young-of-year striped bass. The simultaneous declines of these species have been termed the Pelagic Organism Decline (POD) (IEP 2005, Sommer 2007, Sommer et al. 2007). A number of factors have been hypothesized to contribute to the decline of these species including pollutants, introduced species, and water operations. The relative importance of these factors in these declines is a topic of extensive research (Sommer 2007, Baxter et al. 2008).

3.1.1.5 Population Viability Summary

Abundance

Since 2004, FMWT indices of pre-spawning adult abundance have reached the lowest levels on record. A decline in abundance noted since 2001 is concurrent with the POD and appears to indicate acceleration in a previously observed long-term decline in delta smelt abundance. As delta smelt are endemic to the San Francisco Estuary, the FMWT indices document a decline in species as a whole.

Productivity

Recent trends in the 20mm Survey and the TNS indices, which measure juvenile abundance after the spawning season, parallel the declining trends in the FMWT index suggesting that reproductive success is not compensating for low adult abundance and may be decreasing over time. Several possible reasons have been identified for this observed decline in reproductive success, including an increase in the entrainment of robust early-spawning adults, a decrease in the proportion of robust spawning adults that live to spawn in their second year, changes in summer food supply, and degradation in fall habitat conditions (Baxter et al. 2008).

Spatial Structure

Delta smelt spawning occurs mostly in the north delta with the highest concentration occurring in the lower Sacramento River and in the vicinity of Liberty Island and Cache Slough. A minority of the population spawns in the central Delta in the vicinity of Franks Tract, the lower San Joaquin River, and the lower Mokelumne River. All larvae, juveniles, and surviving adults return to the summertime range in Suisun Bay and the western Delta to utilize habitat in the low salinity zone. The population is therefore largely contiguous. No genetic differences have been identified between the population spawning in the north Delta and those spawning in the central Delta (Bennett 2005).

Diversity

Bennett (2005) calls for further genetic studies on delta smelt to monitor population viability and determine effective population size. The Center for Biological Diversity et al. (2006) points out that the FMWT index has been less than 100 for over two years and therefore the population has fallen below a critical criterion previously cited by USFWS (2004) at which loss of genetic integrity may lead to increased extinction risk.

3.1.1.6 Critical Habitat Summary and Primary Constituent Elements

The USFWS designated critical habitat for delta smelt in 1994 (USFWS 1994, 59 FR 65256). The geographic area includes areas and 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 Delta.

The USFWS identified several primary constituent elements (PCEs) required to maintain delta smelt habitat for spawning, larval and juvenile transport, rearing, and adult migration (USFWS 1994 and 2008). Elements of these PCEs include the following (USFWS 2008):

- PCE #1 Physical Habitat – structural components of habitat. For this pelagic fish, the only known important structural component is spawning substrate and possibly depth variation.
- PCE #2 Water – appropriate water quality conditions of temperature, turbidity, and food availability. High entrainment risk or contaminant exposure can degrade this primary constituent element.
- PCE #3 River flow – transport flow to facilitate spawning migrations and transport of offspring to low-salinity rearing habitats. River flow interacts with salinity by influencing the extent and location of the highly-productive low salinity zone, where delta smelt rear.
- PCE #4 Salinity – low salinity zone (LSZ) nursery habitat, at 0.5-6.0 psu (parts per thousand salinity, Kimmerer 2004). The 2 psu isohaline (X2) is located within the LSZ and is an indicator of the low salinity zone, which varies seasonally. In general, delta smelt habitat quality and surface area are greater when X2 is located in Suisun Bay.

At the time of the 1994 designation, the best available science held that the delta smelt population was responding to variation in spring X2 (USFWS 2008). The scientific understanding has improved over the intervening 14 years. The current understanding of the USFWS is that OMR (combined flow in OMRs) must be considered to manage entrainment. The distribution, function and attributes of each PCE for each life stage are summarized below from the critical habitat designation (USFWS 2004) and the 2008 OCAP BO (USFWS 2008).

Spawning Habitat

Delta smelt adults seek shallow, fresh, or slightly brackish backwater sloughs and edge-waters for spawning. Specific areas identified as important delta smelt spawning habitat include Barker, Lindsey, Cache, Prospect, Georgiana, Beaver, Hog, and Sycamore Sloughs; the Sacramento River in the Delta; and tributaries of northern Suisun Bay.

Spawning delta smelt require all four PCEs, but spawners and embryos are the only life stages of delta smelt that are known to require specific structural components of habitat (PCE # 1). Spawning delta smelt require sandy or small gravel substrates for egg deposition. Migrating, staging, and spawning delta smelt also require low-salinity and freshwater habitats, turbidity, and water temperatures less than 20ºC (68ºF) (Bennett 2005) (PCE #2 and #4).

Spawning occurs primarily late February through early June, peaking in April through mid-may (Moyle 2002). Historically, delta smelt ranged as far up the San Joaquin River as Mossdale, indicating that areas of the lower San Joaquin and its tributaries support conditions appropriate for spawning. Little data exists on delta smelt spawning activity in the lower San Joaquin region. Larval and young juvenile delta smelt collected at South Delta stations in DFG’s 20-mm Survey, indicate that appropriate spawning conditions exist there. However, the few delta smelt that are collected in the lower San Joaquin region is a likely indicator that changes in flow patterns entrain spawning adults and newly-hatched larvae into water diversions (Moyle et al. 1992).

Once the eggs have hatched, larval distribution depends on both the spawning locality (PCE#1 and #2) and delta hydrodynamics for transport (PCE#3). Larval distribution is further affected by salinity and temperature (attributes of PCE#4 and #3). 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), which in some cases might result in rapid dispersal of larvae away from spawning sites.

Successful feeding depends on a high density of food organisms and turbidity (PCE #2). Turbidity elicits a first feeding response and enhances the ability of delta smelt larvae to see prey in the water (Baskerville-Bridges et al. 2004). Their diet is comprised of small planktonic crustaceans that inhabit the estuary’s turbid, low-salinity, open-water habitats (attribute of PCE#2).

Larval and Juvenile Transport

As designated in 1994 (USFWS 1994), the specific geographic area important for larval transport is confined to waters contained within the legal boundary of the Delta, Suisun Bay, and Montezuma Slough and its tributaries. The specific season for successful larval transport varies from year to year, depending on when peak spawning occurs and on the water-year type. To ensure larval transport, the Sacramento and San Joaquin Rivers and their tributary channels must be protected from physical disturbance (e.g., sand and gravel mining, diking, dredging, and levee or bank protection and maintenance) and flow disruption (e.g., water diversions that result in entrainment and in-channel barriers or tidal gates). Adequate riverflow is necessary to transport larvae to shallow, productive rearing habitat in Suisun Bay and to prevent interception of larval transport by water diversions in the Delta. To ensure that suitable rearing habitat is available in Suisun Bay, the 2 ppt isohaline must be located westward from the Sacramento-San Joaquin River confluence during the period when larvae or juveniles are being transported, according to the historical salinity conditions which vary according to water- year type. Reverse flows interfere with transport by maintaining larvae upstream in deep-channel regions of low productivity and exposing them to entrainment.

Delta smelt larvae require PCEs # 2-4 (USFWS 2008). The distribution of delta smelt larvae follows that of the spawners; larvae emerge near where they are spawned. Thus, they are distributed more widely during high outflow periods. Delta smelt larvae mainly inhabit tidal freshwater at temperatures between 10ºC-20ºC (Bennett 2005). The center of distribution for delta smelt larvae < 20 mm is usually 5-20 km upstream of X2, but larvae move closer to X2 as the spring progresses into summer (Dege and Brown 2004). The primary influences the water projects have on larval delta smelt critical habitat are that they influence water quality, the extent of the LSZ, and larval transport via capture of runoff in reservoirs and subsequent manipulation of Delta inflows and exports that affect negative Old and Middle river flows.

Rearing Habitat

The 1994 critical habitat designation identified an area extending eastward from Carquinez Strait, including Suisun Bay, Grizzly Bay, Honker Bay, Montezuma Slough and its tributary sloughs, up the Sacramento River to its confluence with Three Mile Slough, and south along the San Joaquin River including Big Break as the specific geographic area critical to the maintenance of suitable rearing habitat. Maintenance of the 2 ppt isohaline and suitable water quality (low concentrations of pollutants) within the estuary is necessary to provide delta smelt larvae and juveniles a shallow, protective, food-rich environment in which to mature to adulthood. This placement of the 2 ppt isohaline also serves to protect larval, juvenile, and adult delta smelt from entrainment in the State and Federal water projects. Protection of rearing habitat conditions may be required from the beginning of February through the summer.

The USFWS (2008) focused on the specific PCEs required by rearing juveniles, mainly water quality and salinity (PCEs # 2 and # 4. Juvenile delta smelt are most abundant in the LSZ, specifically at the upstream edge of the LSZ where salinity is < 3 psu, water transparency is low (Secchi disk depth < 0.5 m), and water temperatures are cool (< 24ºC) (Feyrer et al. 2007, Nobriga et al. 2008). Many juvenile delta smelt rear now near the Sacramento-San Joaquin river confluence, a change in historic distribution. Currently, young delta smelt rear throughout the Delta into June or the first week of July, but thereafter, distribution shifts to the Sacramento-San Joaquin river confluence where water temperatures are cooler and water transparencies are lower (Feyrer et al. 2007, Nobriga et al. 2008). The 2008 OCAP BO (USFWS 2008) discusses the change in distribution in further detail.

Adult Migration

Adult delta smelt must be provided unrestricted access to suitable spawning habitat in a period that may extend from December to July. Adequate flow and suitable water quality may need to be maintained to attract migrating adults in the Sacramento and San Joaquin River channels and their associated tributaries, including Cache and Montezuma Sloughs and their tributaries. These areas also should be protected from physical disturbance and flow disruption during migratory periods (USFWS 1994).

Successful delta smelt adult migration habitat is characterized by conditions that attract migrating adult delta smelt (PCE #2, #3, and #4) and that help them migrate to spawning habitats (PCE #3). Delta smelt are weakly anadromous and move from the LSZ into freshwater to spawn, beginning in late fall or early winter and likely extending at least though May. Although the physiological trigger for the upward movement of delta smelt through the estuary is unknown, movement is associated with pulses of freshwater inflow, which are cool, less saline and turbid (attributes of PCE #2 and #4 for adult migration). As they migrate, delta smelt increase their vulnerability to entrainment if they move closer to the CVP and SWP export pumps (Grimaldo et al. *accepted manuscript in* USFWS 2008). Analyses indicate that delta smelt in the central and south Delta become less vulnerable to entrainment when reverse flows in the Delta are minimized. Inflows in early winter must be of sufficient magnitude to provide the cool, fresh and highly turbid conditions needed to attract migrating adults and of sufficient duration to allow connectivity with the Sacramento and San Joaquin river channels and their associated tributaries, including Cache and Montezuma sloughs and their tributaries (attributes of PCE #2 for adult migration). These areas are vulnerable to physical disturbance and flow disruption during migratory periods.

3.1.1.7 Factors Affecting Delta Smelt and designated Critical Habitat

Many factors come together to directly and indirectly affect delta smelt and their habitat. The most important factors limiting delta smelt populations are altered delta hydrodynamics, loss due to entrainment at the state and federal water projects, food web alteration by alien species, and poor water quality.

Larval and Adult Entrainment Caused by Water Movement and Conveyance

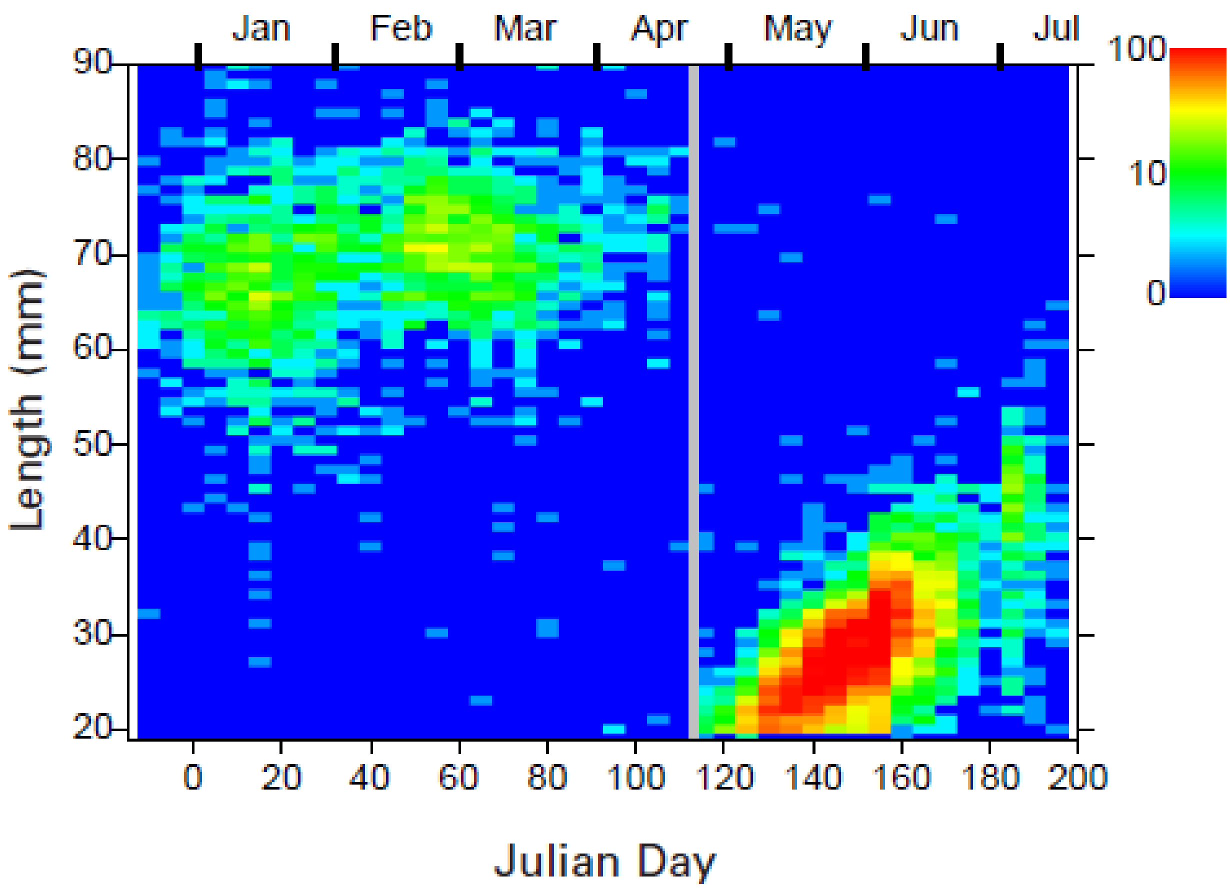
The direct and indirect effects of Delta water exports pose obvious threats to delta smelt and are the primary impetus behind this project. Entrainment directly affects adult, juvenile, and larval smelt at the SWP and CVP water export facilities. Delta smelt entrained by the export facilities are often assumed to suffer 100 percent mortality, as even those adults that are salvaged generally may die from handling stress (Kimmerer 2008).

The entrainment of adult delta smelt at the SWP and CVP export facilities occurs mainly during their upstream spawning migration between December and April (Table 3-1, Figure 3-5) (USFWS 2008). The risk of entrainment depends on level of exports and the location of spawning adults relative to facilities, which varies among years (Figure 3-6) (Grimaldo et al. accepted manuscript cited in USFWS 2008). In some years a large proportion of the adult population migrates to the central and south Delta, placing both spawners and their progeny in relatively close proximity to the export pumps and increasing entrainment risk. In other years, the bulk of adults migrate to the north Delta, reducing entrainment risk. In very wet periods, some spawning occurs west of the Delta.

Table 3-1 The Temporal Occurrence of Delta Smelt Life Stages

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Adult Migration												
Delta												
Spawning/Incubation												
Delta												
Larval Development and Juvenile Movement to west of Chipps Island												
Delta												
Larval and Early Juvenile Rearing												
Delta												
Estuarine Rearing Juveniles and Adults												
Western Delta, Suisun Bay												
Salvage												

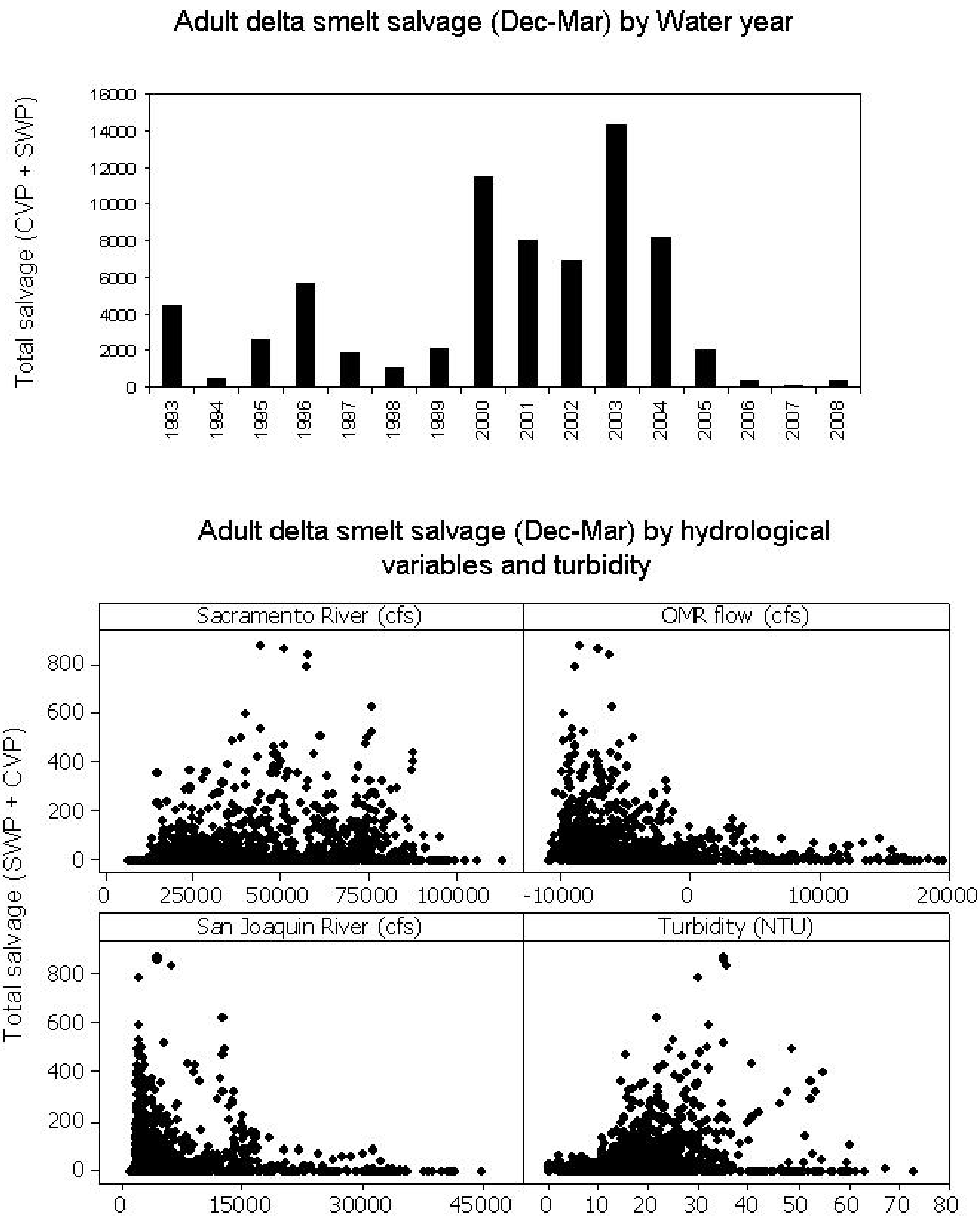
Source: Fisheries Technical Working Group (ENTRIX 2008)



Source: Kimmerer 2008

Image plot showing numbers of fish by length and day, according to log scale at right. Larger fish are adults, and small ones are larvae and juveniles, roughly separated by the vertical line. Larvae smaller than 20 mm are generally not counted. Very few fish were caught between July and mid-December.

Figure 3-5 Delta Smelt Combined Salvage at South Delta Fish Facilities for 1997 – 2005



Source: USFWS 2008

Figure 3-6 Adult Delta Smelt Salvage (December – March) by WY and by Hydrological Variables and Turbidity

UC Davis researchers propose that increased winter exports, and the accompanying Old and Middle river negative flows, are entraining increased numbers of early spawning delta smelt (Baxter et al. 2008). The early spawners tend to be the largest individuals which produce more and stronger offspring. Increased entrainment of these early spawners can reduce population in concert with other factors (Bennett 2005, Brown and Kimmerer 2002).

Delta smelt larvae and juveniles are vulnerable to entrainment, particularly in years when spawning occurs in the Central and South Delta. Salvage has historically been greatest in drier years when a high proportion of young fish rear in the Delta (Moyle et al. 1992, Reclamation and DWR 1994, Sommer 1997). Delta smelt are not detected in the salvage until they are juveniles (at least 20 mm in length). Most salvage of juveniles occurs from April to July, with a peak May-June (Figure 3-5) (Kimmerer 2008, Grimaldo et al. accepted manuscript cited in USFWS 2008). In order to minimize entrainment of undetected larvae, export reductions have focused on the time period when larval smelt are thought to be in the South Delta (based on adult distributions). In 2007 and 2008, CVP and SWP implemented actions to reduce entrainment at the pumps, including maintaining higher outgoing flows in OMRs; delta smelt salvage was considerably decreased in those two years (USFWS 2008).

The indirect effects of water exports are due to altered hydrodynamics in the Delta. High exports and low San Joaquin River flows lead to reverse flows, poor habitat conditions, and degraded water quality in the south Delta. Exports combined with dam operations ultimately influence delta outflow and the position of the low salinity zone (X2). Sommer (2007) suggested that recent change in fall delta smelt habitat quality (salinity and turbidity) may be in part due to changes in fall water export/import ratios and Delta Cross channel operations.

Flood Control and Levee Construction

There is no evidence that levees and other flood control infrastructure directly impact delta smelt populations. The construction, maintenance, or failure of levees may have indirect effects on delta smelt by influencing delta hydrodynamics.

Land Use Activities

Intensive agricultural and urban development in the delta affects delta smelt indirectly by impacting water quality in the delta and reducing freshwater inflow through many small diversions. See ‘Water Quality’ and ‘Water Movement and Conveyance’ sections.

Water Quality

Contaminants, eutrophication, and algal blooms can alter ecosystem functions and productivity, but the magnitude and effects within the Delta are poorly understood (USFWS 2008). Pollutants from agricultural and urban sources may harm delta smelt directly; reduce zooplankton abundance, or both. Recent testing has noted invertebrate toxicity in the waters of the northern Delta and western Suisun Bay. Three water quality concerns are currently being investigated to determine their role in the Pelagic Organism Decline (Baxter et al. 2008, Sommer 2007, Sommer et al. 2007):

- Pyrethroid pesticides in agricultural runoff are known to be very toxic to fish and other aquatic organisms. The recent decline in pelagic fishes in the San Francisco Estuary has roughly coincided with increasing agricultural use of pyrethroid pesticides.

- A blue-green alga known as *Microcystis aeruginosa*, has formed large summertime blooms in the Delta in recent years in the core habitat of delta smelt. This cyanobacterium produces a substance highly toxic to fish, invertebrates, and other animals. The toxin may cause physiological damage to delta smelt when they co-occur, or reduce the abundance of their primary food resources through toxicity to aquatic invertebrates (Reclamation 2008).
- Ammonia released from sewage treatment plants in increasing quantities in recent years may inhibit primary productivity in some areas, be directly toxic to delta smelt, and encourage blooms of microcystis (Meyer et al. 2009).

Fish bioassays conducted as part of the POD studies indicated that larval delta smelt are highly sensitive to ammonia, low turbidity, and low salinity (Baxter et al. 2008, Reclamation 2008). Turbidity is an important attribute of delta smelt critical habitat, involved in attracting adult migration and facilitating foraging. There has been a Delta-wide increase in water transparency in recent years, linked to the invasion of non-native submerged aquatic vegetation which traps sediment (discussed below under Non-Native Invasive Species). Reduced turbidity may have also intensified predation pressures on delta smelt (USFWS 2008).

Hatchery Operations

Current captive breeding programs for delta smelt are for scientific purposes only and do not release fish into the wild. These programs therefore have no effect on wild delta smelt populations.

Over-utilization (commercial and sport)

There is no lawful commercial or recreational fishery for delta smelt. The most significant form of utilization for this species is scientific collecting by the Interagency Ecological Program through several monitoring programs. The IEP has determined these monitoring programs have a net beneficial effect on the delta smelt population through improved management.

Disease and Predation

Predation is presumed to have an important impact on delta smelt survival; however, it has proven difficult to quantify. There is little evidence that disease and predation threaten the survival of the species (USFWS 2004). Many introduced predators are known to eat delta smelt, the most important of these being striped bass and largemouth bass. Striped bass have experienced declining annual abundance concurrent with the recent Pelagic Organism Decline. Conversely, largemouth bass are believed to be increasing in numbers (Baxter et al. 2008). Decreased flows and restricted tidal influence in the south and central delta have combined to create warm, clear water conditions ideal for the growth of non-native Brazilian waterweed (*Egeria densa*), which provides favorable cover and hunting conditions for largemouth bass.

Food Web Alteration Caused by Non-native Invasive Species

Many non-native invasive species affect delta smelt both directly and indirectly through predation, food web alteration, and effects on physical habitat. Primary productivity, and likewise zooplankton biomass, in the western delta has declined since the introduction of the overbite clam (*Corbula amurensis*) in the 1980s, possibly limiting food availability for the delta smelt and other pelagic species (Baxter et al. 2008). As zooplankton production is an important factor limiting summer and fall survival in the western Delta and Suisun Bay (Kimmerer 2008), the overbite clam has indirectly limited the delta smelt population in the decades since its introduction. Furthermore the composition of the zooplankton community, mostly composed of introduced species, has changed in recent years having potentially significant, but as yet unproven, effects on food availability for delta smelt.

The physical habitat of the interior Delta has been altered over the last two decades by invading submerged aquatic vegetation, principally *Egeria densa* (Baxter et al. 2008, USFWS 2008). This plant has altered fish community dynamics by increasing habitat for centrarchid fishes (Nobriga et al. 2005, Brown and Michniuk 2007), reducing habitat for native fishes (Brown 2003), and altering the food web. Non-native submerged aquatic vegetation can affect delta smelt directly by degrading and reducing unvegetated spawning habitat, and indirectly by decreasing turbidity (vegetation traps suspended sediment) which is an important attribute of juvenile and adult habitat (Feyrer et al. 2007, Nobriga et al. 2008).

Environmental Variation and Climate Change

There is currently no quantitative analysis of how ongoing climate change is currently affecting delta smelt (USFWS 2008). However, climate change has the potential to significantly shift habitat available to delta smelt upstream as Delta water temperatures and sea levels both rise. Altered precipitation patterns could also cause shifts in the timing of flows and water temperatures, which could lead to a change in timing of migration of adults and juvenile delta smelt (USFWS 2008).

Ecosystem Restoration

Ecosystem restoration projects currently underway within the Delta may prove to be beneficial to delta smelt (Bennett 2005). The highest density of delta smelt spawning and larval production occurs in the vicinity of Cache Slough and Liberty Island. This area provides abundant shallow water spawning habitat and is heavily influenced by flows from the Yolo Bypass which provide an important source of carbon and planktonic food to fish in the north delta. Similar habitat restoration is imminent adjacent to Suisun Marsh (i.e., at the confluence of Montezuma Slough and the Sacramento River) as part of the Montezuma Wetlands project, which is intended to provide for commercial disposal of material dredged from San Francisco Bay in conjunction with tidal wetland restoration. These areas are the focus of state and federal restoration programs to enhance the function of floodplain and tidal freshwater ecosystems.

A major restoration program is the CALFED Bay–Delta Program (CALFED), currently implemented through the California Bay-Delta Authority (CBDA). CALFED was formed in 1995 with the central tenets of environmental restoration and stable water supplies. Two CBDA programs in particular were created to improve conditions for fish in the Central Valley: (1) the Ecosystem Restoration Program (ERP) and its Environmental Water Program, and (2) the Environmental Water Account (EWA) managed under the Water Supply and Reliability Program (CALFED 2000). Restoration initiatives expected to benefit delta smelt include restoration of shallow-water tidal and marsh habitats within the Delta, screening diversions, and adjusting water export operations. Achievement of other goals of the ERP, such as reducing the negative impacts of invasive species and improving water quality (CALFED 2000), are also expected to benefit delta smelt by reducing competitors or improving food web dynamics and the copepods that are a key food resource.

A review of CALFED's performance in Years 1 through 8 concluded that the greatest investments and outcomes of the ERP and Watershed Programs have been in areas upstream from the Delta, outside the range of delta smelt (CALFED Bay Delta Public Advisory Committee [BDPAC] 2007). Efforts have been less successful in the Delta where native species, including the delta smelt, continue to decline. Research indicates some of the management actions taken to protect salmon may be in conflict with actions to protect delta smelt. Funding and research efforts have been refocused to resolve the declining populations of important Delta species.

Habitat restoration initiatives sponsored and funded primarily by the CBDA-ERP have resulted in plans to restore ecological function to 9,543 acres of shallow-water tidal and marsh habitats within the Delta. Restoration of these areas primarily involves flooding lands previously used for agriculture, thereby creating additional shallow water spawning and rearing habitat for delta smelt. This assumption, however, has undergone revision with new science (Brown 2003). The benefits of restoring shallow water habitat may be offset by nonnative species that dominate these habitats, such as fishes that prey on delta smelt and invasive aquatic plants that alter water quality (reduced turbidity) and habitat structure (Bennett 2005, Brown 2003).

The CBDA's EWA was established to alleviate the uncertainty of water use, as well as to provide benefits to delta smelt and other fishes of special concern. Environmental water is acquired and "banked" and used for fish protection, primarily by reducing water exports at critical times when delta smelt "take" at the major facilities is elevated. For delta smelt, however, it is unclear whether reducing water exports at the critical times has benefited the delta smelt population (Bennett 2005). The CALFED BDPAC (2007) concluded that the EWA has not been successful at reversing the decline of important Delta species including delta smelt.

Another restoration approach seeks to improve fish screening and salvaging procedures at the export facilities. The CALFED Program Record of Decision called for substantial investments in fish screens in the south Delta (CALFED 2000). However, there is little scientific evidence that these measures benefit the population (Bennett 2005). Delta smelt are extremely fragile and many do not survive handling. Moreover, it is currently unclear if losses to the water projects are a major impact on their abundance (Bennett 2005). In 2005, an agency and stakeholder group recommended and the state and federal agencies concurred, that the CALFED Program not proceed with significant investments in new fish screens at the Delta pumping facilities, rather that additional research be accomplished and other actions taken that were thought to provide greater benefits to fish populations (CALFED BDPAC 2007). Similarly, there has been a consistent effort to install fish screens on the numerous small agricultural diversions in the Delta. Again, however, the benefits of fish screening have never been established for delta smelt, and the added structural complexity to these diversions may provide habitat harboring predatory fishes (Bennett 2005). What little is known indicates their effect is small (Nobriga and others 2004) and localized, with little effect at the population level.

3.1.1.8 Status of the Species within the Action Area

All life stages of delta smelt occur in the Action Area of the 2-Gates Project and the Action Area encompasses much of the designated critical habitat (Figure 3-1). The Action Area includes areas considered important for larval transport. The Action Area is east and south of the area considered most important for rearing. However, if rearing delta smelt are found within the Action Area, protection of rearing habitat conditions may be required from the beginning of February through the summer. Areas important for delta smelt spawning habitat generally occur outside of the Action Area. The status of delta smelt rangewide and in the Action Area is currently declining and abundance levels are the lowest ever recorded (USFWS 2008).

3.1.2 Chinook Salmon and Steelhead

3.1.2.1 Listing Status and Designated Critical Habitat

NMFS has recently completed an updated status review of 16 salmon ESUs that included the Sacramento River winter-run Chinook salmon ("winter-run Chinook") and Central Valley spring-run Chinook salmon ("spring-run Chinook"), and concluded that the species' status should remain as previously listed (June 28, 2005, 70 FR 37160). In addition, NMFS published a final listing determination for 10 steelhead distinct population segments (DPSs), and concluded that Central Valley steelhead ("CV steelhead") will remain listed as threatened (January 5, 2006, 71 FR 834).

The following federally listed anadromous species ESUs or DPSs and designated critical habitats occur in the Action Area and may be affected by the action:

Sacramento River winter-run Chinook Salmon

Winter-run Chinook salmon (*Oncorhynchus tshawytscha*) were originally listed as threatened in August 1989 under emergency provisions of the ESA, and formally listed as threatened in November 1990 (55 FR 46515). The ESU consists of only one population that is confined to the upper Sacramento River. The Livingston Stone National Fish Hatchery population has been included in the listed winter-run Chinook population as of June 28, 2005 (70 FR 37160). The ESU was reclassified as endangered on January 4, 1994 (59 FR 440), due to increased variability of run sizes, expected weak returns as a result of two small year classes in 1991 and 1993, and a 99 percent decline between 1966 and 1991. NMFS reaffirmed the listing as endangered on June 28, 2005 (70 FR 37160) and included the Livingston Stone National Fish Hatchery population in this listed ESU.

NMFS designated critical habitat on June 16, 1993 (58 FR 33212). Critical habitat is delineated as the Sacramento River from Keswick Dam at river mile (RM) 302 to Chipps Island (RM 0) at the westward margin of the Sacramento-San Joaquin Delta (Delta), including Kimball Island, Winter Island, and Brown's Island; all waters from Chipps Island westward to the Carquinez Bridge, including Honker Bay, Grizzly Bay, Suisun Bay, and the Carquinez Strait; all waters of San Pablo Bay westward of the Carquinez Bridge, and all waters of San Francisco Bay north of the San Francisco-Oakland Bay Bridge. The northwest region of the Action Area overlaps designated critical habitat, namely the migration corridor on the Sacramento River along the North Delta (Figure 3-7).

Central Valley spring-run Chinook Salmon

Central Valley spring-run Chinook salmon (*Oncorhynchus tshawytscha*) were listed as threatened on September 16, 1999 (64 FR 50394). NMFS released a five-year status review in June 2004, and proposed that this species remain listed as threatened (69 FR 33102). Although spring-run Chinook productivity trends were positive at the time, the ESU continued to face risks from: (1) a limited number of remaining populations (three, down from an estimated 17 historical populations); (2) a limited geographic distribution; and (3) potential hybridization with Feather River Fish Hatchery (FRFH) spring-run Chinook salmon, which are genetically divergent from populations in Mill, Deer, and Butte Creeks. The NMFS final decision on June 28, 2005 retained this species as threatened (70 FR 37160). The ESU currently consists of spring-run Chinook salmon occurring in the Sacramento River basin, including the FRFH spring-run Chinook salmon population.

Critical habitat for Central Valley spring-run Chinook salmon was designated on September 2, 2005 (70 FR 52488). Spring-run critical habitat includes the stream channels within numerous streams throughout the Central Valley, including the Sacramento, Feather and Yuba Rivers, and Deer, Mill, Battle, Antelope, and Clear Creeks in the Sacramento River basin. Critical habitat is also designated within the Sacramento-San Joaquin Delta and the San Francisco-San Pablo-Suisun Bay complex. The Action Area does not overlap designated critical habitat (Figure 3-8).

Central Valley steelhead

Central Valley steelhead (*Oncorhynchus mykiss*) are listed as threatened (January 5, 2006, 71 FR 834). The CV steelhead DPS consists of naturally spawned anadromous populations of *O. mykiss* below natural and manmade impassable barriers in the Sacramento and San Joaquin Rivers and their tributaries. Excluded are steelhead from San Francisco and San Pablo Bays and their tributaries, as well as two artificial propagation programs: the Coleman NFH, and FRFH steelhead hatchery programs.

NMFS designated critical habitat on September 2, 2005 (70 FR 52488). CV steelhead critical habitat encompasses 2,308 miles of stream habitat in the Central Valley including the Sacramento River and tributaries and the San Joaquin River and tributaries upstream to the Merced River. An additional 254 square miles of estuary habitat in the San Francisco-San Pablo-Suisun Bay complex is also designated critical habitat. The Action Area contains portions of the designated critical habitat, namely the channel reaches within the Sacramento-San Joaquin Delta (Figure 3-9).

3.1.2.2 Life History

Chinook salmon and steelhead are anadromous salmonids of the genus *Oncorhynchus.* This section provides an overview of key life history attributes (reviewed by Myers et al. 1998, Moyle 2002, NMFS 2008a).

Sacramento River winter-run Chinook and Central Valley spring-run Chinook Salmon

Chinook salmon are the largest member of *Oncorhynchus*. 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 tend to enter freshwater as immature fish, migrate far upriver, and delay spawning for weeks or months. For comparison, fall-run Chinook enter freshwater at an advanced stage of maturity, move rapidly to their spawning areas on the mainstem or lower tributaries of the rivers, and spawn within a few days or weeks of freshwater entry. Adequate instream flows and cool water temperatures are more critical for the survival of winter-run and spring-run Chinook salmon due to over-summering by adults and/or juveniles.

This section presents life history attributes common to winter-run and spring-run Chinook salmon (reviewed by Myers et al. 1998, Moyle 2002). Run-specific differences in the spatial and temporal distribution of various life stages are discussed in Section 3.1.2.3 “Distribution”. Chinook salmon typically mature between 2 and 6 years of age (Myers et al. 1998). Freshwater entry of migrating adults and spawning timing are generally thought to be related to local water temperature and flow regimes. Adults migrate to spawning habitat in streams well upstream of the Delta. Adults spawn in clean, loose gravel in swift, relatively shallow riffles or along the margins of deeper runs.

Upon emergence, fry swim or are displaced downstream. As juvenile Chinook salmon grow, they move into deeper water with higher current velocities, but still seek shelter and velocity refugia to minimize energy expenditures. Catches of juvenile salmon in the Sacramento River near West Sacramento by the USFWS (1997) exhibited larger juvenile captures in the main channel and smaller sized fry along the margins. When the channel of the river is greater than 9 to 10 feet in depth, juvenile salmon tend to inhabit the surface waters.

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. Within the Delta, juveniles forage in shallow areas with protective cover, such as tidally-influenced sandy beaches and vegetated zones. Cladocerans, copepods, amphipods, and diptera larvae, as well as small arachnids and ants, are common prey items (Kjelson et al. 1982, Sommer et al. 2001).

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 as the tide recedes. 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. During the night, juveniles were distributed randomly in the water column, but during the day would school up into the upper 3 meters of the water column. Juvenile Chinook salmon were found to spend about 40 days migrating through the Sacramento-San Joaquin Delta to the mouth of San Francisco Bay.

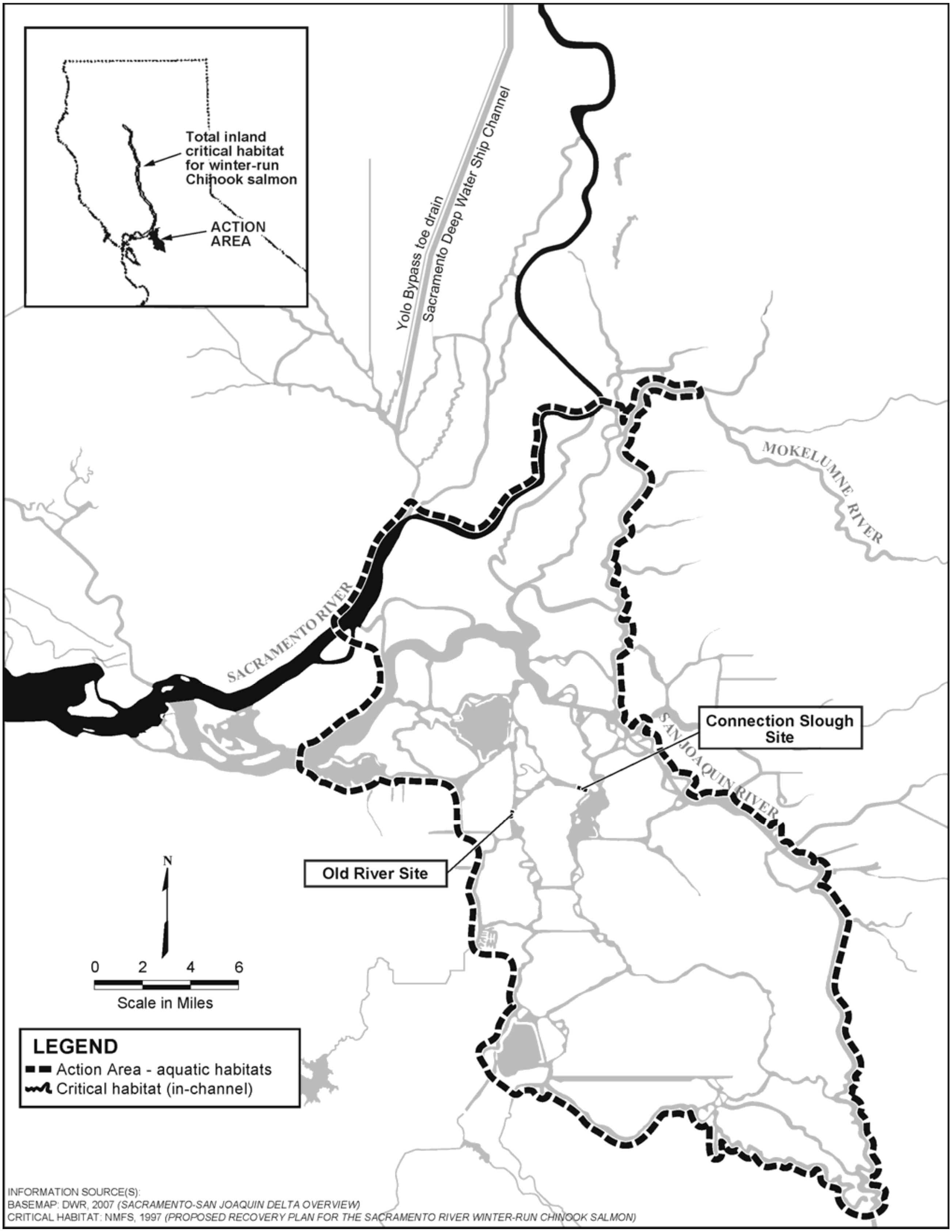


Figure 3-7 Action Area and Designated Critical Habitat for Sacramento River winter-run Chinook Salmon

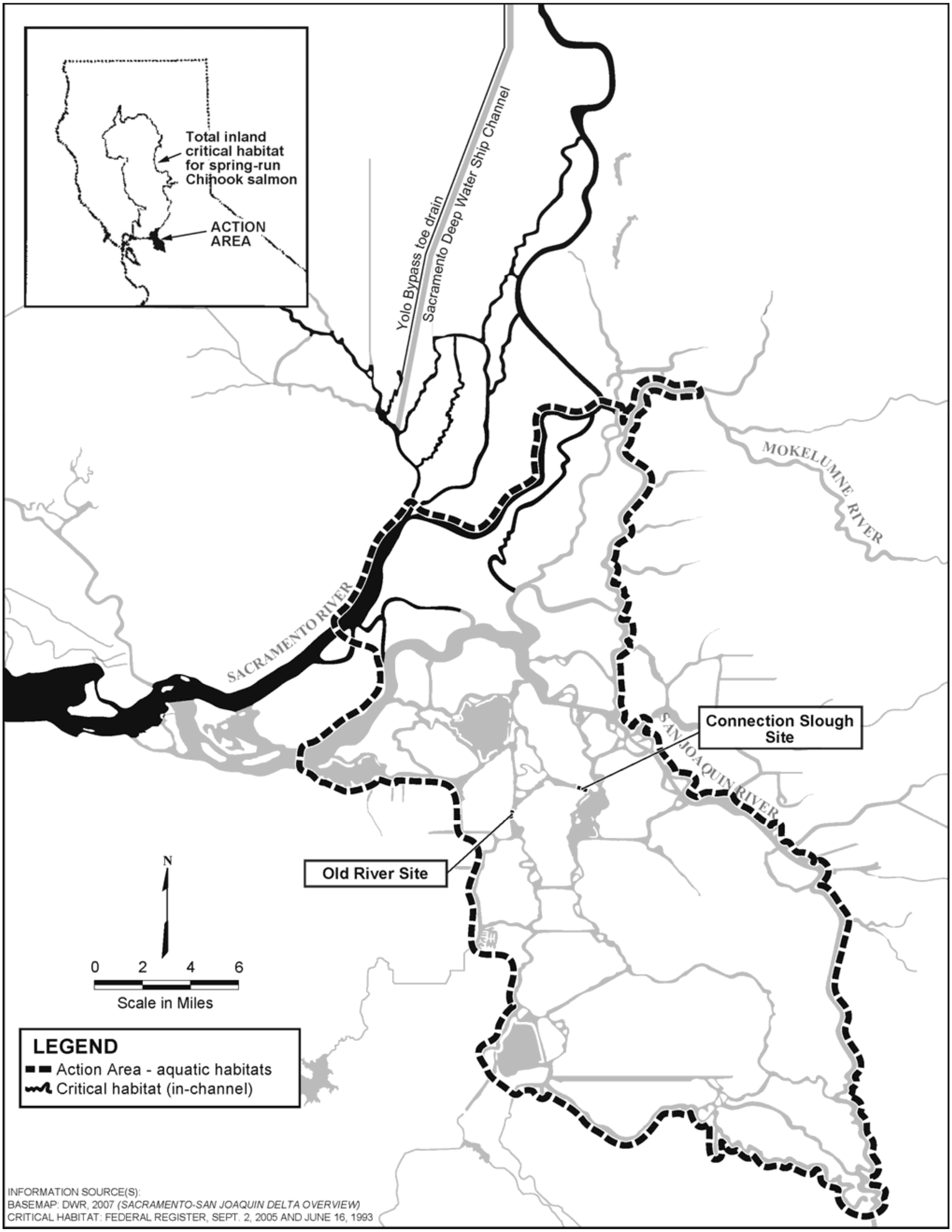


Figure 3-8 Action Area and Designated Critical Habitat for Central Valley spring-run Chinook Salmon

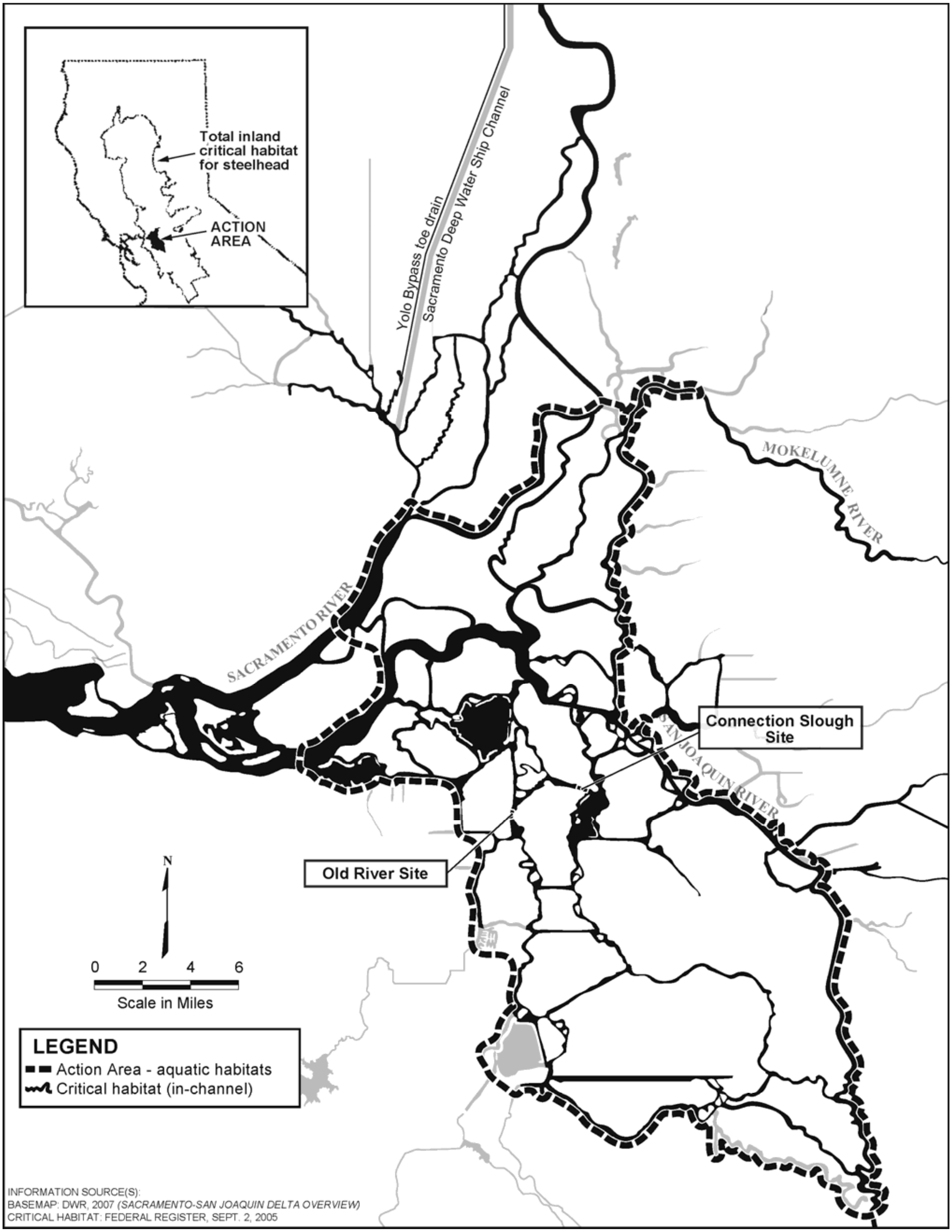


Figure 3-9 Action Area and Designated Critical Habitat Central Valley steelhead

Central Valley steelhead

Steelhead can be divided into two life history types, winter (ocean-maturing) and summer (stream-maturing), based on their state of sexual maturity at the time of river entry and the duration of their spawning migration. Only winter steelhead are currently found in Central Valley Rivers and streams (McEwan and Jackson 1996). Ocean-maturing steelhead enter freshwater with well-developed gonads and spawn shortly after river entry. A brief description of general life history follows, although variations in period of habitat use can occur. Further details are provided in Busbey et al. (1996), McEwan and Jackson (1996), Moyle (2002), Reclamation (2008) and NMFS (2008a).

CV steelhead generally leave the ocean from August through April and migrate through the estuary to spawning habitat in streams. Spawning takes place from December through April, with peaks from January through March (McEwan and Jackson 1996, Busby et al. 1996). Unlike Pacific salmon, steelhead are iteroparous, or capable of spawning more than once before death (Busby et al. 1996). Steelhead spend the first year or two of life in cool, clear, fast-flowing permanent streams and rivers with ample riffles, cover, and invertebrate prey (Moyle 2002). Juvenile steelhead emigrate from natal streams volitionally or during fall through spring freshets. Sacramento River juveniles migrate downstream most of the year, predominantly in spring (Hallock et al. 1961).

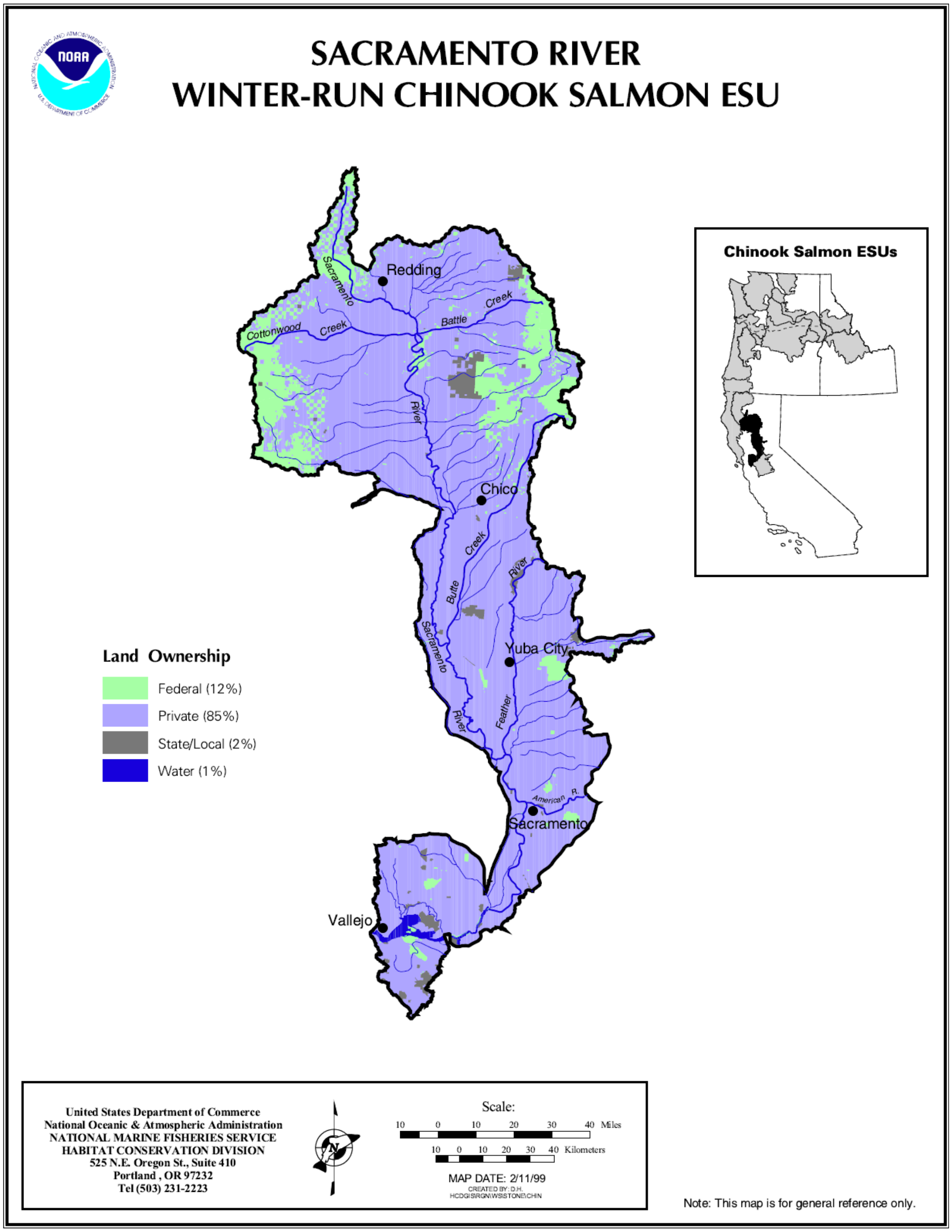
Rearing and ocean-emigrating juvenile steelhead use the lower reaches of the Sacramento River and the Delta including tidal marsh areas, non-tidal freshwater marshes, and other shallow water areas. CV steelhead migrate to the ocean after spending one to three years in freshwater (McEwan and Jackson 1996). They remain in the ocean for one to four years growing before returning to their natal streams to spawn.

3.1.2.3 Distribution

Sacramento River winter-run Chinook Salmon

Historically, the distribution of winter-run Chinook spawning and rearing was limited primarily to the upper Sacramento River and its tributaries, the Pit and McCloud Rivers (Myers et al. 1998). These spring-fed streams provided cold water through the summer to support spawning, egg incubation, and rearing (Slater 1963, Yoshiyama et al. 1998). Construction of Shasta Dam in 1943 and Keswick Dam in 1950 blocked access to all these waters, except Battle Creek (Moyle et al. 1989, NMFS 1997, Myers et al. 1998). An estimated 299 miles of spawning and rearing habitat upstream of Keswick Dam has been lost (Yoshiyama et al. 2001). As a result, the winter-run Chinook population has been displaced to a single population currently spawning and rearing in the mainstem Sacramento River between Keswick Dam (RM 302) and the Red Bluff Diversion Dam (RBDD) (RM 243). This population is entirely dependent on regulated cold water releases from Shasta and Keswick Dams and is vulnerable to a prolonged drought (Good et al. 2005). Small numbers of winter-run Chinook salmon have also been reported on the Calaveras River in the San Joaquin River system (Myers et al. 1998) although none have been reported there since 1984 (source: DFG GrandTab data 2008). The range of the Sacramento River winter-run Chinook salmon ESU is shown in Figure 3-10.

Adult winter-run Chinook enter the San Francisco Bay from November through June and migrate past the RBDD from mid-December through early August (Hallock and Fisher 1985, NMFS 1997) (Table 3-2). The majority of the run passes the RBDD from January through May, with the peak occurring in mid-March (Hallock and Fisher 1985). The timing of migration may vary somewhat due to changes in river flow, dam operations, and water year type (Yoshiyama et al. 1998, Moyle 2002). 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).



Source: NMFS 200X

Figure 3-10 Sacramento Valley winter-run Chinook Salmon Evolutionarily Significant Unit

Table 3-2 The Temporal Occurrence of Adult and Juvenile Sacramento River winter-run Chinook Salmon in the Sacramento River.

Adult Location	Jan		Feb		Mar		Apr		May		Jun		Jul		Aug		Sep		Oct		Nov		Dec	
Sac River basin[1]																								
Sac River[2]																								
Delta[3]	X	X	X	X	X	X	X	X	X	X	X	X									X	X	X	X
Juvenile Location	**Jan**		**Feb**		**Mar**		**Apr**		**May**		**Jun**		**Jul**		**Aug**		**Sep**		**Oct**		**Nov**		**Dec**	
Sac River @ Red Bluff[4]																								
Sac River @ Red Bluff[2]																								
Sac River @ Knights L.[5]																								
Lower Sac River (seine)[6]																								
West Sac River (trawl)[6]																								
Delta[3]	X	X	X	X	X	X	X	X	X	X	X	X					X	X	X	X	X	X	X	X
Salvage[3]	X	X	X	X	X	X	X	X															X	X
Relative Abundance		=High		=Medium		=Low	X	=Present																

Data Sources:[1] Yoshiyama et al. 1998 & and Moyle 2002; [2] Meyers et al. 1998, [3] ENTRIX 2008, [4] Martin et al. 2001, [5] Snider and Titus 2000, [6] USFWS 2001

Source: NMFS 2008a, ENTRIX 2008

Winter-run Chinook fry emerge from the gravel in late June through October. Juveniles rear in the upper Sacramento River and may begin to emigrate past RBDD as early as mid–July, typically peaking in September, and may continue through March in dry years (Vogel and Marine 1991, NMFS 1997). Juvenile winter-run Chinook occur in the Delta primarily from November through early May, based on trawl surveys in the Sacramento River at West Sacramento (RM 57) (USFWS 2001). The timing of emigration 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 5-10 months of age, and then emigrate to the ocean from November through May (Fisher 1994, Myers et al. 1998).

Central Valley spring-run Salmon

Historically, spring-run Chinook salmon was the dominant run in the Sacramento and San Joaquin River Basins (Clark 1929, Myers et al. 1998) and once considered among the largest runs on the Pacific Coast (Yoshiyama et al. 1998). Spring-run Chinook salmon historically migrated upstream as far as they could in the larger tributaries to the Sacramento and San Joaquin Rivers, where they held for several months in deep cold pools (Moyle 2002). Their run timing was suited to gain access to the upper river reaches (up to 1,500 m elevation) prior to the onset of high water temperatures and low flows that inhibit access to these areas during the fall (Myers et al. 1998). Historic runs were reported in the McCloud River, Pit River, Little Sacramento River, Feather River (including above Oroville Dam), Yuba River (including above Englebright Dam), and American River (including above Folsom Dam) in the Sacramento River Basin (Moyle 2002) and on the San Joaquin River (above Friant Dam), and in the tributaries of the Merced, Tuolumne, Stanislaus and Mokelumne rivers in the San Joaquin Basin (NMFS 2004, Yoshiyama et al. 1998).

Construction of Friant Dam on the San Joaquin River, Shasta Dam on the upper Sacramento River, and other low elevation dams on tributary streams extirpated spring-run Chinook from these watersheds. Currently, naturally spawning populations are restricted to accessible reaches of the Sacramento River, Antelope Creek, Battle Creek, Beegum Creek, Big Chico Creek, Butte Creek, Clear Creek, Deer Creek, Mill Creek, the Feather River and the Yuba River (DFG 1998) (Figure 3-11).

Adult spring-run Chinook leave the ocean to begin their upstream migration in late January and early February (DFG 1998) and enter the Sacramento River system between March and September, primarily peaking in May and June (Table 3-3; Yoshiyama et al. 1998, Moyle 2002). Adults enter native tributaries from the Sacramento River primarily between mid April and mid June (Lindley et al. 2007). Fry emerge from the gravel between November and March (Moyle 2002).

Table 3-3 The Temporal Occurrence of Adult and Juvenile Central Valley spring-run Chinook Salmon in the Sacramento River.

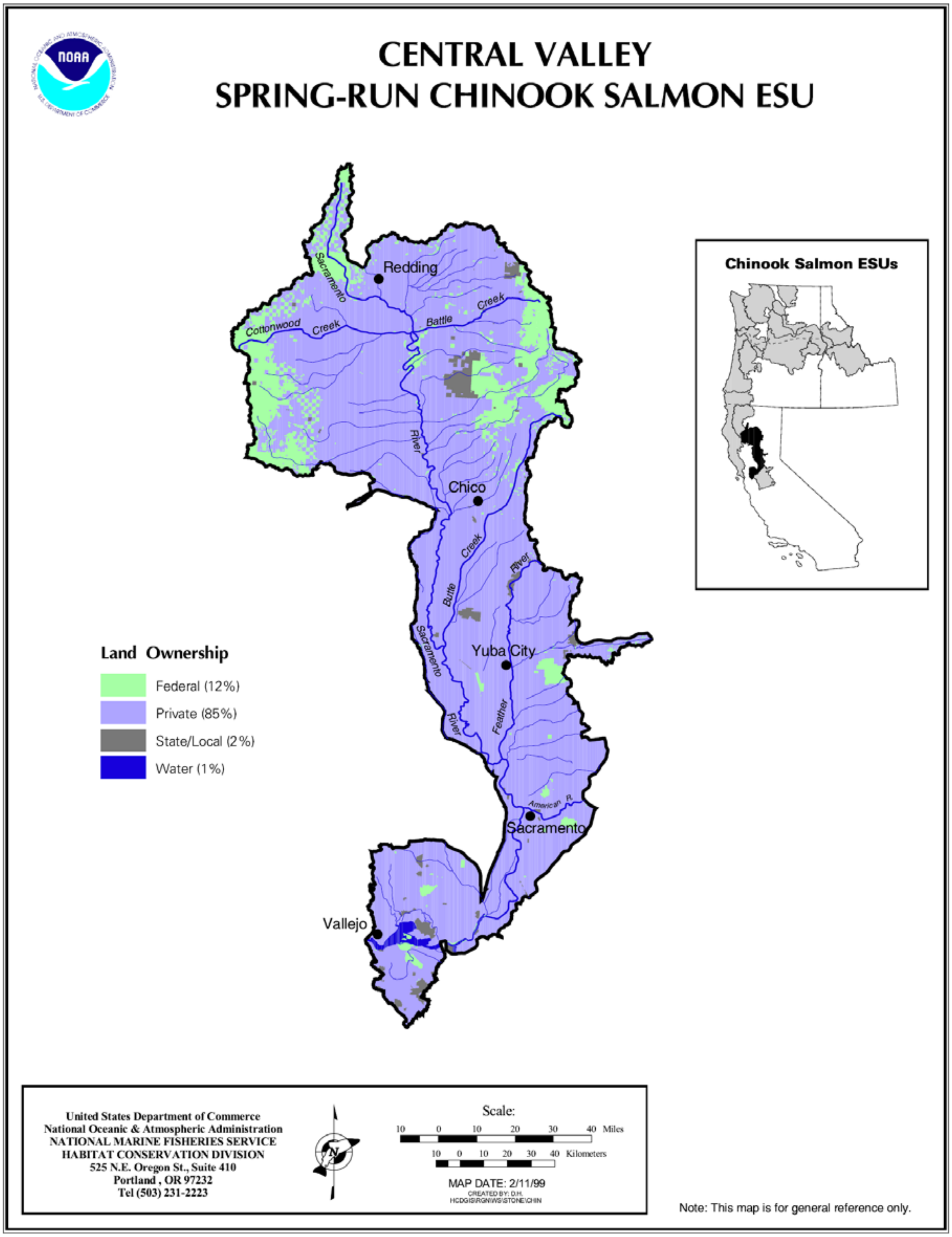
Adult Location	Jan		Feb		Mar		Apr		May		Jun		Jul		Aug		Sep		Oct		Nov		Dec	
Sac River basin[1]																								
Sac River[2]																								
Mill Creek[3]																								
Deer Creek[3]																								
Butte Creek[3]																								
Delta[4]																								
Juvenile Location	**Jan**		**Feb**		**Mar**		**Apr**		**May**		**Jun**		**Jul**		**Aug**		**Sep**		**Oct**		**Nov**		**Dec**	
Sac River Tribs[5]																								
Upper Butte Creek[6]																								
Mill, Deer, & Butte																								
Sac River																								
Sac River @ Knights Landing[7]																								
Delta[4]	X	X	X	X	X	X	X	X	X	X	X	X									X	X	X	X
Salvage[4]					X	X	X	X	X	X	X	X												

Relative Abundance: =High; =Medium; =Low; X = Present[4]

Data Sources:[1]Yoshimama et al. 1998 and Moyle 2002; [2]Meyers et al. 1998; [3]Lindley et al. 2006; [4] ENTRIX 2008; [5]DFG 1998; [6]McReynolds et al. 2005, Ward et al. 2002, 2003; [7]Snider and Titus 2000

Source: NMFS 2008a, ENTRIX 2008

The emigration timing of spring-run Chinook appears highly variable (DFG 1998). Some fish may begin emigrating as young-of-the-year (YOY) soon after emergence from the gravel, whereas others over summer and emigrate as yearlings with the onset of intense fall storms (DFG 1998). A shorter period of rearing may be a response to altered flow regimes (caused by dams and diversions) and required use of lower elevation sections of streams (Yoshiyama et al. 1998, Moyle 2002). The emigration period extends from November to early May, with up to 69 percent of the YOY fish outmigrating through the lower Sacramento River and Delta during this period (DFG 1998). Peak movement of juveniles 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).



Source: NMFS 200X

Figure 3-11 Central Valley spring-run Chinook Salmon Evolutionarily Significant Unit

Central Valley steelhead

CV steelhead populations are found in the Sacramento River and its tributaries, including the Feather, Yuba, and American Rivers, and many small tributaries, such as Antelope, Mill, Deer and Butte creeks, west side tributaries (including Clear, Cottonwood, Stoney, Thomes, Cache and Putah creeks and Suisun Bay tributaries of Alamo and Ulatis Creeks. The Cosumnes and Mokelumne Rivers also support steelhead, and they have also been documented in the Stanislaus River (Cramer 2000) on the San Joaquin System. Steelhead have also sporadically been collected from the Calaveras River. Figure 3-12 shows the range of the CV steelhead ESU.

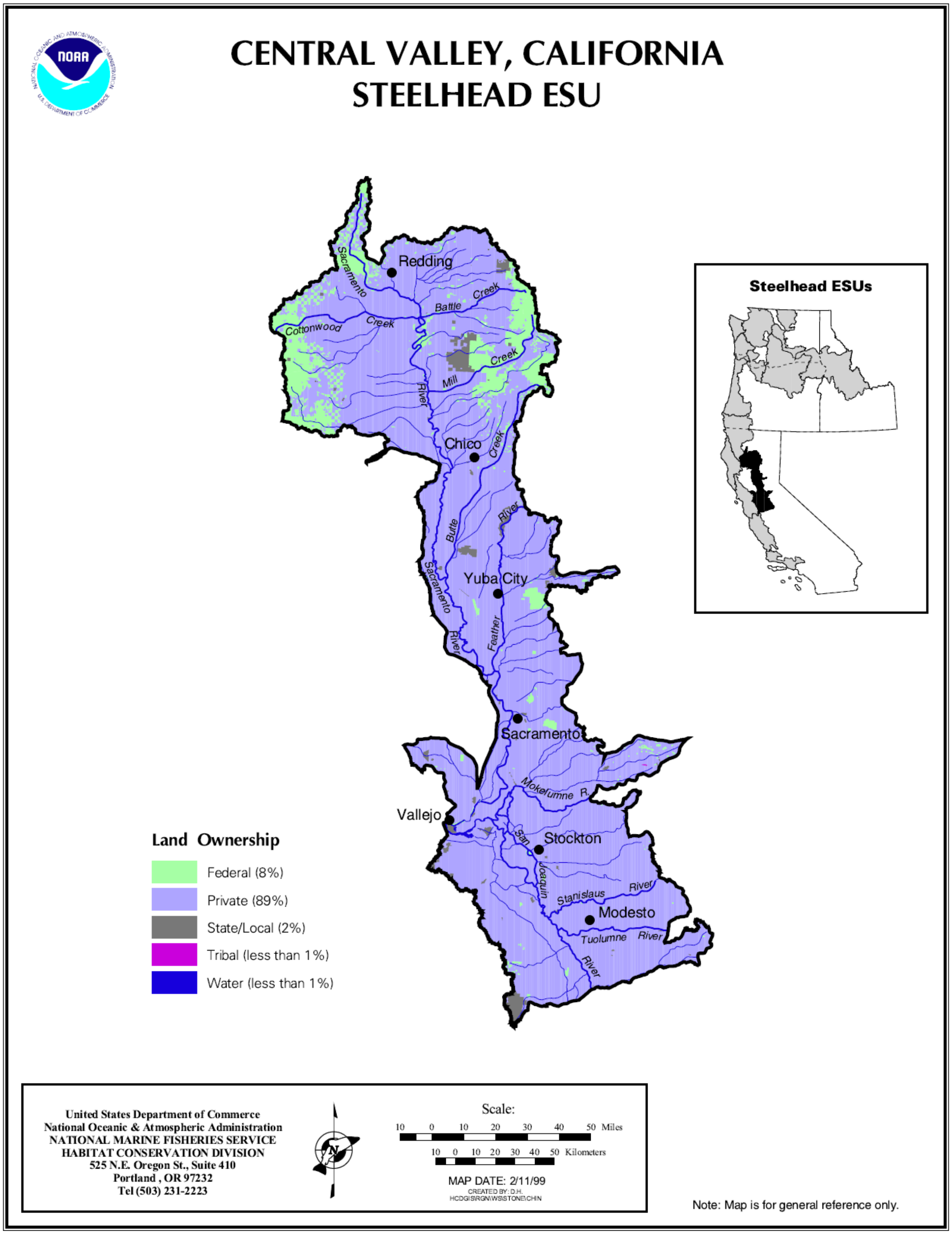
The temporal distribution of different life stages in the Central Valley is shown in Table 3-4. Adults are present in the Delta (lower Sacramento River at Fremont Weir and the San Joaquin River) between July and March, with a peak in March and April. Juveniles are present in the Delta from October to July, with a peak in March to May. Adults leave the ocean August through April (Busby et al. 1996), and spawn December through April, with peaks January though March, (Hallock et al. 1961, McEwan and Jackson 1996). Juvenile steelhead emigrate episodically from natal streams during fall, winter, and spring high flows (NMFS 2008a). Juveniles migrate downstream during most months of the year, but the peak period of emigration occurs in the spring (March to May), with a much smaller peak in the fall (Hallock et al. 1961, Nobriga and Cadrett 2001).

Table 3-4 The temporal occurrence of adult and juvenile Central Valley steelhead in the Central Valley.

	Jan		Feb		Mar		Apr		May		Jun		Jul		Aug		Sep		Oct		Nov		Dec	
Adult Location																								
Sac River [1, 2]																								
Sac R. @ Red Bluff [2,3]																								
Mill, Deer Creeks [4]																								
Sac River @ Fremont Weir [6]																								
San Joaquin R [7]																								
Juvenile Location																								
Sac River [1,3]																								
Sac River @ Knights Landing [3,8]																								
Sac River @ Knights Landing [9]																								
Sac River @ Hood [10]																								
Chipps Island (wild) [11]																								
Delta [12]	X	X	X	X	X	X	X	X															X	X
San Joaquin R @ Mossdale [8]																								
Mokelumne R @ Woodbridge Dam [13]																								
Stan. R @ Caswell [14]																								
Salvage [12]	X	X	X	X	X	X	X	X															X	X

Relative Abundance	=High	=Medium	=Low	X = Present[12]

Data Sources: [1] Hallock et al. 1961; [2]USFWS unpubl. Data; [3]McEwan 2001; [4]DFG 1995; [5]Hallock et al. 1957; [6]Bailey 1954; [7]DFG Steelhead Report Card Data; [8]DFG unpubl. Data; [9]Snider and Titus 2000; [10]Schaffter 1980 & 1997; [11]Nobriga and Cadrett 2001; [12] ENTRIX 2008; [13] Jones and Stokes Associates, Inc. 2002; [14]S.P. Cramer and Associates, Inc. 2000 & 2001.



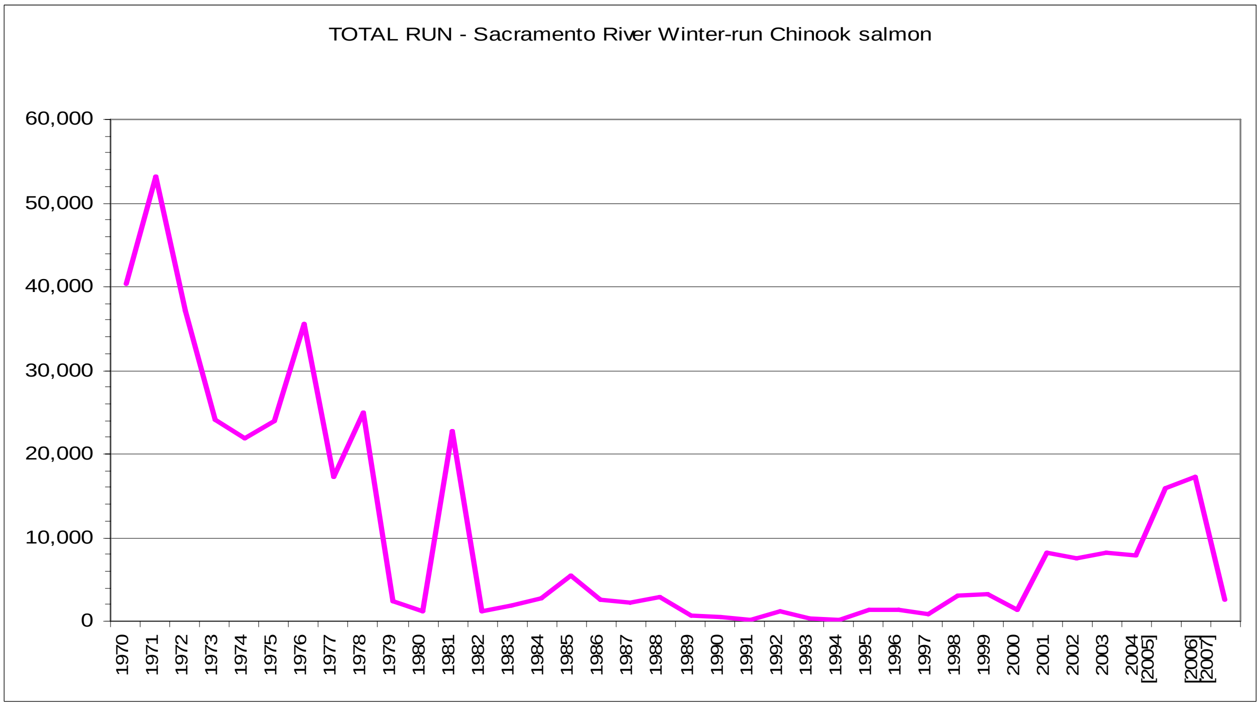
Source: NMFS 1998

Figure 3-12 Central Valley steelhead Evolutionarily Significant Unit

3.1.2.4 Abundance

Sacramento River winter-run Chinook Salmon

Following construction of Shasta Dam, population estimates of winter-run Chinook salmon ranged from 117,808 in 1969 to a low of 186 in 1994 (DFG 2002c). Adult escapement since 1970 is illustrated in Figure 3-13 (see also Table 3-5). Population estimates over the last decade generally show an increase trend in population size to 17,205 in 2006, the highest since the 1994 listing. However, the 2007 escapement estimate of 2,488 fish shows a significant decline relative to previous years (DFG GrandTab, 2008).



Source: DFG GrandTab database March 2008

Figure 3-13 Estimated Sacramento River winter-run Chinook Salmon Run Size

Table 3-5 Winter-Run Chinook Salmon Population Estimates from RBDD Counts (1986 to 2001) and Carcass Counts (2001 to 2007) and Corresponding Cohort Replacement Rates and Juvenile Production Estimates (JPE) for the Years Since 1986

Year	In-River Population Estimate	5-Year Moving Average of Population Estimate	Cohort Replacement Rate	5-Year Moving Average of Cohort Replacement Rate	NMFS Calculated Juvenile Production Estimate (JPE)[a]
1986	2,566				
1987	2,165				
1988	2,857				
1989	649		0.25		
1990	411	1,730	0.19		
1991	177	1,252	0.06		40,025
1992	1,203	1,060	1.85		272,032
1993	378	564	0.92	0.66	85,476
1994	144	463	0.81	0.77	32,562
1995	1,166	613	0.97	0.92	263,665
1996	1,012	780	2.68	1.45	228,842
1997	836	707	5.82	2.24	189,043
1998	2,903	1,212	2.49	2.55	656,450
1999	3,264	1,836	3.23	3.04	738,082
2000	1,263	1,856	1.51	3.14	285,600
2001	8,120	3,277	2.80	3.17	1,836,160
2002	7,360	4,582	2.26	2.46	1,664,303
2003	8,133	5,628	6.44	3.25	1,839,100
2004	7,784	6,532	0.96	2.79	1,760,181
2005	15,730	9,425	2.14	2.92	3,556,995
2006	17,205	11,242	2.12	2.78	3,890,535
2007	2,488	10,268	0.32	2.39	562,607
Median	2,326	1,783	1.85	2.55	562,607
Average	3,992	3,501	1.99	2.30	1,053,039
Gmean[b]	1,907	2,074	1.22	2.06	479,040

[a]JPE estimates were derived from NMFS calculations utilizing RBDD winter-run counts through 2001, and carcass counts thereafter for deriving adult escapement numbers.

[b]Gmean is the geometric mean of the data in that column.

Source: CDFG 2004 and 2007 in NMFS 2008a

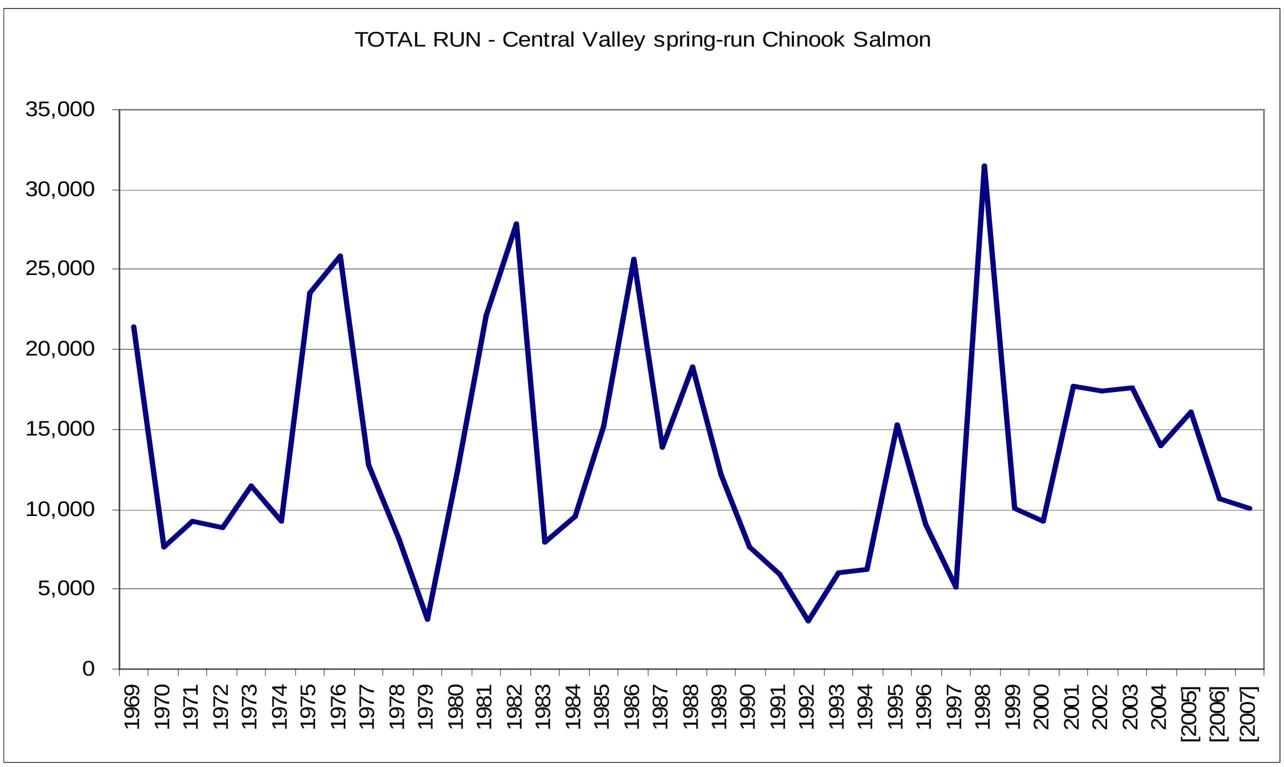
Ocean conditions may be a factor in recent declines (NMFS 2008a). The ocean life history traits and habitat requirements of winter-run Chinook and fall-run Chinook salmon are similar. The USFWS (2008) proposed that the unusually poor ocean conditions that are suspected to have contributed to the drastic decline in returning fall-run Chinook salmon populations coast-wide in 2007 (Varanasi and Bartoo 2008) have likely contributed to the observed decrease in winter-run Chinook escapement estimates for 2007. Preliminary escapement estimates for 2008 range from 2,600 to 2,950 (mean 2,775) winter-run Chinook in the Sacramento River. Although numbers appear to be slightly up from 2007, they are still low relative to the six years between 2001 and 2006, indicating that the conditions which have contributed to the general decline of Chinook salmon Pacific coast-wide have not significantly changed.

Since 1991, NMFS (2008a) has estimated juvenile production of winter-run Chinook using the Juvenile Production Estimate (JPE) method (Gaines and Poytress 2004). The median and average JPE between 1991 and 2007 has been estimated at 562,607 and 1,053,039, respectively (Table 3-4). Production increased steadily between 2000 (285,600) to 2006 (3,890,535), but declined significantly in 2007 (562,607).

Central Valley spring-run Chinook Salmon

The Sacramento-San Joaquin River Basin once supported a spring-run Chinook salmon run as large as 600,000 fish between the late 1880's and 1940's (DFG 1998). Since 1969, the abundance of spring-run Chinook (including Feather River Hatchery fish) has fluctuated broadly from a low of 3,044 in 1992 to a high of 31,471 in 1998 (Figure 3-14). The average (mean) and median population estimates for spring-run Chinook within the entire Sacramento-San Joaquin River system since 1969 are 13,328 and 11,430 fish, respectively.

In river (natural spawning) population estimates have generally followed the same trends. Between 1986 and 2007, in-river population estimates for spring-run Chinook salmon have ranged from a low of 1,403 fish in 1993 to a high of 24,725 fish in 1998 (see Table 3-6). Sacramento River tributary populations in Mill, Deer, and Butte Creeks are probably the best trend indicators because these streams contain the primary independent populations within the ESU. Generally, these streams had positive escapement trends between 1991 and 2005 dropping off in the last three years (from 14,014 fish in 2005 to an estimated 6,507 fish in 2007 (DFG GrandTab 2008). These trends are similar to the system wide in-river trends reported by DFG. Preliminary estimates for 2008 (4,381 fish in Deer, Mill and Butte Creeks) are generally lower than for 2007. Escapement numbers are dominated by Butte Creek returns, which have averaged over 7,000 fish between 1995 and 2007. During this same period, adult returns on Mill Creek have averaged 778 fish, and 1,463 fish on Deer Creek. Although recent trends are positive, annual abundance estimates fluctuate widely and remain well below historic levels (1960's to 1990).



Source: DFG GrandTab database March 2008

Note: Years in [ ] are still considered preliminary

Figure 3-14 Estimated Central Valley spring-run Chinook Salmon Run Size

Table 3-6 Central Valley spring-run Chinook Salmon Population Estimates from CDFG GrandTab Data (May 2008) with Corresponding Cohort Replacement Rates and JPE's for the Years 1986 to 2007

Year	In-River Population Estimate	5-Year Moving Average of Population Estimate	Cohort Replacement Rate	5-Year Moving Average of Cohort Replacement Rate	NMFS Calculated Juvenile Production Estimate (JPE)[a]
1986	24,263				4,396,998
1987	12,675				2,296,993
1988	12,100				2,192,790
1989	7,085		0.29		1,283,960
1990	5,790	12,383	0.46		1,049,277
1991	1,624	7,855	0.13		294,305
1992	1,547	5,629	0.22		280,351
1993	1,403	3,490	0.24	0.27	254,255
1994	2,546	2,582	1.57	0.52	461,392
1995	9,824	3,389	6.35	1.70	1,780,328
1996	2,701	3,604	1.93	2.06	489,482
1997	1,433	3,581	0.56	2.13	259,692
1998	24,725	8,246	2.52	2.58	4,480,722
1999	6,366	9,010	2.36	2.74	1,106,181
2000	5,587	8,162	3.90	2.25	1,010,677
2001	13,563	10,335	0.55	1.98	2,457,919
2002	13,220	12,692	2.08	2.28	2,395,759
2003	8,908	9,529	1.59	2.10	161,432
2004	9,774	10,210	0.72	1.77	1,771,267
2005	14,346	11,962	1.09	1.21	2,599,816
2006	8,700	10,990	0.98	1.29	1,576,634
2007	7,300	9,806	0.75	1.02	1,322,923
Median	8,000	8,628	0.98	1.98	1,106,181
Average	8,885	7,970	1.49	1.73	1,335,479
Gmean[b]	6,452	7,109	0.93	1.50	1,051,034

[a]NMFS calculated the spring-run JPE using returning adult escapement numbers to the Sacramento River basin prior to the opening of the RBDD for spring-run Migration, and then escapement to Mill, Deer, and Butte Creeks for the remaining period, and assuming a female to male ratio of 6:4 and pre-spawning mortality of 25 percent. NMFS utilized the female fecundity values in Fisher (1994) for spring-run Chinook salmon (4,900 eggs/female). The remaining survival estimates used the winter-run values for calculating the JPE.

[b]Gmean is the geometric mean of the data in that column.

Source: CDFG 2007 in NMFS 2008a

Central Valley steelhead

Very limited information makes it difficult to estimate historic CV steelhead run sizes, but they 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 from an estimated average of 20,540 adult steelhead through the 1960s down 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 (Figure 3-15) (McEwan and Jackson 1996, McEwan 2001). Steelhead escapement surveys at RBDD ended in 1993 due to changes in dam operations (NMFS 2008a). Although currently there is a complete lack of monitoring, what data exist indicate the population continues to decline (Good et al. 2005).

One challenge in assessing the success of steelhead spawning in the upper Sacramento River is the difficulty in distinguishing steelhead from the resident rainbow trout population that has developed as a result of managing for cold water all summer.

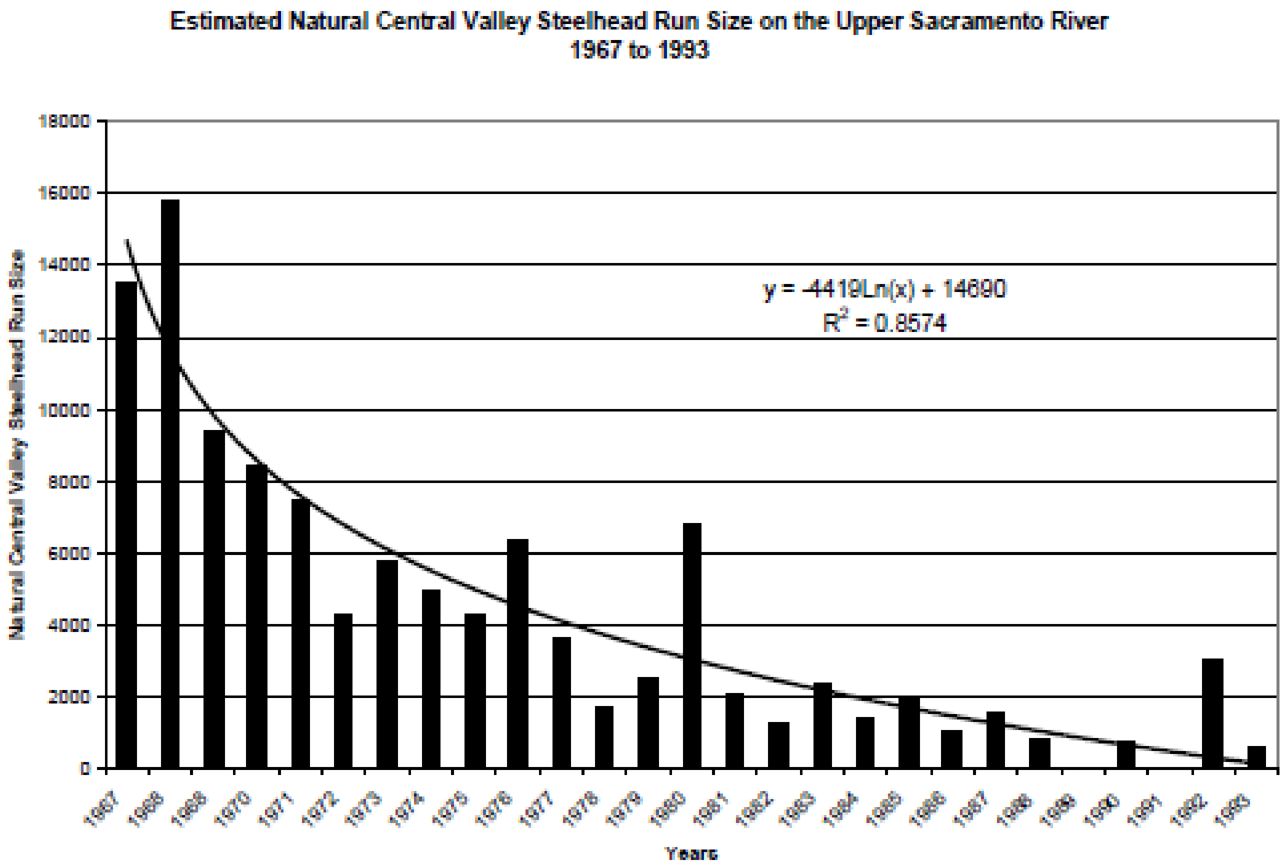


Figure 3-15 Estimated Natural Central Valley steelhead Escapement in the Upper Sacramento River Based on RBDD Counts. Note: Steelhead escapement surveys at RBDD ended in 1993 (from McEwan and Jackson 1996 in NOAA 2008a).

3.1.2.5 Population Viability Summary

McElhany et al. (2000) defined a population's components of abundance, productivity, spatial structure, and diversity as the basis of determining population and ESU viability for salmonids. NMFS (2008) also summarized results of viability modeling.

Sacramento River winter-run Chinook Salmon

ABUNDANCE

Redd and carcass surveys, and fish counts, suggest that the abundance of winter-run Chinook has been increasing over the past decade. The exception is the depressed abundance estimate observed in 2007 which is suspected to represent a cycle of poor ocean productivity coast wide recently. Population growth is estimated to be positive in the short-term with a trend at 0.26; however, the long-term trend is negative, averaging -0.14. Recent winter-run Chinook abundance represents only 3 percent of the maximum post-1967, 5-year geometric mean, and is not yet well established (Good et al. 2005).

PRODUCTIVITY

ESU productivity has generally been positive over the short term, and adult escapement and juvenile production have been increasing annually (Good et al. 2005) with the recent exception of the 2007 estimates. As mentioned above, poor ocean conditions coast wide are suspected of being the cause for poor adult returns, which in turn has resulted in decreased juvenile production. The long-term outlook for the ESU remains negative, however, as it consists of only one population that is subject to possible impacts from environmental and artificial conditions.

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 habitat and must be artificially maintained in the mainstem Sacramento River by a regulated, finite cold water supply from Shasta Dam. Winter-run Chinook 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.

DIVERSITY

The second highest risk factor for winter-run Chinook has been the detrimental effects on its diversity. The present winter-run 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; there may have been several others within the recent past (Good et al. 2005).

VIABILITY MODELING

Modeling has been used to assess the viability and risk of extinction of winter-run Chinook (NMFS 2008a). As reviewed by Good et al. (2005), Botsford and Brittnacker (1998) used an age-structured density-independent model of spawning escapement and concluded that the species was certain to fall below the quasi-extinction threshold of three consecutive spawning runs with fewer than 50 females). Lindley et al. (2003) used a Bayesian model based on spawning escapement that allowed for density dependence and a change in population growth rate in response to conservation measures. They found a biologically significant expected quasi-extinction probability of 28 percent.

Central Valley spring-run Chinook Salmon

ABUNDANCE

Spring-run Chinook have 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 FRFH spring-run Chinook 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.

PRODUCTIVITY

The 5-year geometric mean for the Butte, Deer, and Mill Creek spring-run Chinook populations range from 491 to 4,513 fish (Good et al. 2005), indicating increasing productivity for this period. Since 2005 the trend has declined (Table 3-5).

SPATIAL STRUCTURE

Spring-run Chinook presence has been reported more frequently in several upper Central Valley creeks, but the sustainability of these runs is unknown. Butte Creek spring-run cohorts have recently utilized all available habitat in the creek; the population cannot expand further and it is unknown if individuals have opportunistically migrated to other systems. The spatial structure of the spring-run ESU has been reduced with the extirpation of all San Joaquin River basin spring-run populations.

DIVERSITY

The Central Valley spring-run Chinook ESU is comprised of two genetic complexes. Analysis of natural and hatchery spring-run Chinook stocks in the Central Valley indicates that the southern Cascades spring-run population complex (Mill, Deer, and Butte creeks) retains genetic integrity. The genetic integrity of the Sierra Nevada spring-run population complex has been somewhat compromised. Feather River spring-run Chinook have introgressed with the fall-run Chinook population, and it appears that the Yuba River population may have been impacted by FRFH fish straying into the Yuba River. Additionally, the diversity of the spring-run Chinook ESU has been further reduced with the loss of the San Joaquin River basin spring-run populations.

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 Creek, according to their 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 population, the spring-run Chinook 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 Chinook population is only represented by the group that currently occurs in rivers and streams in the northern Sierra Nevada. Most historic populations have been extirpated. Over the long term, these remaining populations are considered to be vulnerable to catastrophic events, such as eruptions from Mount Lassen, forest fires, and drought.

In summary, the spring-run Chinook ESU remains at a moderate to high risk of extinction because it is spatially confined to relatively few remaining streams, continues to display broad fluctuations in abundance, and a large proportion of the population (i.e., in Butte Creek) faces the risk of high mortality rates.

Central Valley steelhead

ABUNDANCE

Productivity for steelhead is dependent on freshwater survival and oversummering habitat which has been reduced by 95 percent from historic conditions. Estimates based on juvenile production indicate that the wild population may number in the average of 3,628 female spawners (Busby et al. 1996). All indications are that natural CV steelhead has 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.

PRODUCTIVITY

An estimated 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).

SPATIAL STRUCTURE

Steelhead appear to be well-distributed where found within the Central Valley (Good et al. 2005). Recent efforts have begun to document distribution. Since 2000, steelhead have been confirmed in the Stanislaus and Calaveras rivers. There appears to be fragmentation in the spatial structure because of reduction in the major populations of the Central Valley (i.e. the Sacramento River, Feather River, and American River) that provided a source for the numerous smaller tributary and intermittent stream populations like Dry Creek, Auburn Ravine, Yuba River, Deer Creek, Mill Creek, and Antelope Creek. Tributary populations can likely never achieve the size and variability of the core populations in the long-term generally due to the size and available resources of the tributaries.

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 CV 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 CV steelhead DPS.

3.1.2.6 Critical Habitat and Primary Constituent Elements (PCEs)

The Action Area includes designated critical habitat for CV steelhead, namely the channel system within the Delta. There is no designated critical habitat for winter- and spring-run Chinook within the Action Area. Following are the habitat types used as PCE's for spring-run Chinook and CV steelhead as well as the physical habitat elements for winter-run Chinook.

Spawning Habitat

Freshwater spawning sites are those with water quantity and quality conditions and substrate supporting spawning, incubation, and larval development. Current spawning habitat occurs outside the Action Area, mostly in areas directly downstream of dams. Spawning habitat for winter-run Chinook is restricted to the mainstem Sacramento River, primarily in the 59-mile reach between the RBDD and Keswick Dam. Spring-run Chinook spawn within the Sacramento River Basin on the mainstem Sacramento River, the Feather River, and Mill, Deer, Antelope, and Butte Creeks, and recently on Clear Creek. CV steelhead spawn in reaches below dams which contain suitable conditions for spawning and incubation.

Freshwater Rearing Habitat

Rearing Chinook salmon and steelhead juveniles require adequate space, cover, and food, in addition to cool water temperatures. Suitable rearing habitat includes areas with instream and overhead cover in the form of undercut banks, downed trees, side channels, and large, overhanging tree branches. Both spawning areas and migratory corridors comprise rearing habitat for juvenile salmonids, which feed and grow before and during their outmigration. Non-natal, intermittent tributaries also may be used for juvenile rearing. Rearing habitat quality is strongly affected by habitat complexity, food supply, and the presence of fish predators. Some of these more complex and productive habitats with floodplain connectivity are still found in the system (e.g., the Yolo Bypass, the lower Cosumnes River, Sacramento River reaches with set-back levees [i.e., primarily located upstream of the City of Colusa]). The channeled, leveed, and riprapped river reaches and sloughs common in the lower Sacramento and San Joaquin Rivers and the Delta system, however, typically have low habitat complexity, low abundance of food organisms, and offer little protection from predation by fish and birds. Freshwater rearing habitat has a high conservation value as the juvenile life stages of salmonids are dependant on the function of this habitat for successful survival and recruitment. Thus, although much of the rearing habitat is in poor condition, it is important to the species.

Freshwater Migration Corridors

Ideal freshwater migration corridors for adults and juveniles are free of obstruction and contain natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks. Migratory corridors are downstream of the spawning areas and include the Sacramento River and its tributaries downstream of Keswick Dam as well as the Delta. These corridors allow the upstream passage of adults, and the downstream emigration of juveniles. Migratory habitat condition is strongly affected by the presence of barriers, which can include dams, unscreened or poorly- screened diversions, and degraded water quality. For adults, upstream passage through the Delta and the lower Sacramento River does not appear to be a problem, but problems exist on many tributary streams. For juveniles, unscreened or inadequately screened water diversions throughout their migration corridors along with a scarcity of complex in-river cover have degraded this PCE. However, since the primary migration corridors are used by numerous populations and are essential for connecting early rearing habitat with the ocean, even the degraded reaches are considered to have a high conservation value to the species. Thus, although much of the migration corridor is in poor condition, it is important to the species.

Estuarine Areas

Estuarine areas are another PCE, including both nearshore and off shore habitats, free of obstruction with water quality, salinity conditions, and food resources that support growth and maturation as well as juvenile and adult salmonid physiological transitions between fresh and salt water. Natural cover such as submerged and overhanging large wood, aquatic vegetation, side channels, and deep water areas are suitable for juvenile and adult salmonids. The remaining estuarine habitat for these species is severely degraded by altered hydrologic regimes, poor water quality, reductions in habitat complexity, and competition for food and space with exotic species. Regardless of the condition, the remaining estuarine areas are of high conservation value because they function as predator avoidance and as a transition corridor to the ocean environment. Nearshore marine features are essential to conservation because, without them, juvenile and adult salmonids cannot successfully transition between natal streams and offshore marine areas.

Winter-run and spring-run Chinook and CV steelhead use the Delta, Suisun Bay, San Pablo Bay and San Francisco Bay as migratory corridors through which they move from the ocean to freshwater as adults and from freshwater to the ocean as juveniles. Most movement by adults occurs in deeper channels, while juveniles are more likely to use the shallow habitats, including tidal flats, for feeding and predator refuge.

Ocean Habitats

Although ocean habitats are not part of the critical habitat listings for winter-run and spring-run Chinook and CV steelhead, biologically productive coastal waters are an important habitat component.

3.1.2.7 Factors Affecting Chinook salmon and Steelhead and designated Critical Habitat

The construction of high dams for hydropower, flood control, and water supply have resulted in the loss of vast amounts of upstream habitat (*i.e.*, approximately 80 percent, or a minimum linear estimate of over 1,000 stream miles), and often resulted in precipitous declines in affected salmonid populations. The reduced populations that remain below Central Valley dams are forced to spawn in lower elevation tailwater habitats of mainstem rivers and tributaries that were previously not used for this purpose. This habitat is entirely dependent on managing reservoir releases to maintain cool water temperatures suitable for spawning, and/or rearing of salmonids. All salmonid species considered in this BA have been adversely affected by the production and release of hatchery fish.

Land-use activities associated with agriculture, urban development, resource extraction (logging, mining) and recreation have significantly altered fish habitat quantity and quality through alteration of streambank and channel morphology, alteration of ambient water temperatures; degradation of water quality, elimination of spawning and rearing habitat, habitat fragmentation, elimination of large woody debris, removal of riparian vegetation, and other effects. Human-induced habitat changes, such as alteration of natural flow regimes; installation of bank revetment; and instream structures (e.g., diversion facilities, piers) often provide conditions that both disorient juvenile salmonids and attract predators. Additional stressors include harvest, ocean productivity, and drought conditions. In contrast, various ecosystem restoration activities have contributed to improved conditions for listed salmonids (e.g., habitat enhancement, screening water diversion structures, improved instream flows downstream of some dams).

The following sections are an overview of the factors affecting winter-run and spring-run Chinook and CV steelhead. Further details are provided in various NMFS reports (Busby et al. 1996; Myers et al., 1998; NMFS 1996, 1998 and 2008; Good et al. 2005).

Fish Movement & Habitat Blockage

Habitat loss due to blockage is likely the most important threat to winter-run and spring-run Chinook salmon and CV steelhead. Hydropower, flood control, and water supply dams of the CVP, SWP, and other municipal and private entities have permanently blocked or hindered salmonid access to historical spawning and rearing grounds. Populations of these anadromous salmonids are now confined to lower elevation reaches of Central Valley rivers and streams which were historically only used for migration. Population abundances have declined in these streams due to decreased quantity and quality of spawning and rearing habitat. Higher temperatures at these lower elevation reaches during late-summer and fall are also a major stressor to adult and juvenile salmonids.

Blockages can also occur within the Delta. The Suisun Marsh Salinity Control Gates (SMSCG), installed in 1988 on Montezuma Slough to decrease the salinity levels of managed wetlands in Suisun Marsh, have delayed or blocked passage of adult Chinook salmon migrating upstream, but passage has improved since the 2001-2002 season when the boat lock remained open (NMFS 2008a). Migrating adult and juvenile steelhead may experience blockage or delays at the SMSCG, the Delta Cross Channel, and at temporary agricultural barriers in the south Delta (NMFS 2008a). Migration delays may reduce fecundity and increase susceptibility to disease and poaching for adults, and increase predation risk for juveniles.

Water Development and Conveyance (Hydrodynamics and Entrainment)

The diversion and storage of natural flows by dams and diversion structures on Central Valley waterways have depleted streamflows and altered the natural flow cycles that cue migration by juvenile and adult salmonids. As much as 60 percent of the natural historical inflow to Central Valley watersheds and the Delta have been diverted for human uses. Depleted flows have contributed to higher temperatures, lower dissolved oxygen (DO) levels, and decreased recruitment of gravel and large woody debris (LWD). More uniform flows year round have resulted in diminished natural channel formation, altered sediment quality and bedload movement, altered foodweb processes, and slower regeneration of riparian vegetation. Runoff storage in these large reservoirs has altered the normal hydrograph. Rather than peak flows following winter rain events (Sacramento River) or spring snow melt (San Joaquin River), the current hydrology has truncated peaks with a prolonged period of elevated flows (compared to historical levels) continuing into the summer dry season.

Water withdrawals for agricultural and municipal purposes have reduced river flows and increased temperatures during the critical summer months. Direct relationships exist between water temperature, water flow, and juvenile salmonid survival (Brandes and McLain 2001). Elevated water temperatures in the Sacramento River have limited the survival of young salmon. Juvenile fall-run Chinook salmon survival in the Sacramento River is also directly related with June streamflow and June and July Delta outflow (Dettman et al. 1987).

Water diversions for irrigated agriculture, municipal and industrial use, and managed wetlands are found along the Sacramento River, San Joaquin River, and their tributaries. Many of these diversions are unscreened. Depending on the size, location, and season of operation, these unscreened diversions entrain and kill many life stages of aquatic species, including juvenile salmonids.

Outmigrant juvenile salmonids in the Delta have been exposed to adverse environmental conditions created by water export operations at the CVP and SWP facilities (NMFS 2008a). Specifically, juvenile salmonid survival has been reduced by the following: (1) water diversion from the mainstem Sacramento River into the Central Delta via the Delta Cross Channel; (2) upstream or reverse flows of water in the lower San Joaquin River and southern Delta waterways; (3) entrainment at the CVP/SWP export facilities and associated problems at Clifton Court Forebay; and (4) increased exposure at facilities to introduced, non-native predatory fish (NMFS 2008a).

Flood Control and Levee Construction

The development of the water conveyance system in the Delta has resulted in the construction of more than 1,100 miles of channels and diversions to increase channel elevations and flow capacity of the channels (Mount 1995).

Levee development and bank stabilization structures may affect the quality of rearing and migration habitat along the river. Juvenile steelhead prefer natural stream banks with ample cover from riparian vegetation and undercut banks (Moyle 2002), as opposed to riprapped, leveed, or channelized waterways. Many Delta islands have been fortified to minimize flooding, but these efforts have reduced historic floodplain, marsh, and shallow water habitats that juvenile salmonids depend on for rearing. Many levees use angular rock (riprap) to armor the bank from erosive forces. Channelization, removal of streamside vegetation and large woody debris, and riprapping alter river hydraulics and cover along the bank and cause long-term damage to nearshore habitat for juvenile salmonids (Busby et al. 1996, Myers et al. 1997, USFWS 2000, Schmetterling et al. 2001).

Land Use Activities

Land use activities such as historic and ongoing agricultural practices and urban development continue to have large impacts on salmonid habitat in the Central Valley watershed. Increased sedimentation from agricultural and urban practices within the Central Valley is a primary cause of habitat degradation (NMFS 1996). Land use activities associated with road construction, urban development, logging, mining, agriculture, and recreation have significantly altered fish habitat quantity and quality through the alteration of streambank and channel morphology; alteration of ambient water temperatures; degradation of water quality; elimination of spawning and rearing habitat; fragmentation of available habitats; elimination of downstream recruitment of LWD; and removal of riparian vegetation, resulting in increased streambank erosion (Meehan 1991). Urban stormwater and agricultural runoff may be contaminated with herbicides and pesticides, petroleum products, sediment, and other contaminants (Myers et al. 1998, NMFS 1996 and 1998).

Since the 1850s, wetlands reclamation for urban and agricultural development has caused significant loss of tidal marsh habitat in the Delta. By the time the last island was reclaimed in 1934, 441,000 acres of nearly 500,000 acres of federal swamplands had been reclaimed in the Delta (PPIC 2007). Only about five percent of the original marsh remains in the estuary, with the larger remnants in Suisun Marsh.

Dredging of river channels for shipping and levee construction has significantly impaired the natural hydrology and function of the river systems in the Central Valley. The creation of levees and deep shipping channels reduced seasonal inundation of floodplains, which provided necessary habitat for rearing and foraging juvenile native fish, including salmon and steelhead. Levee maintenance has reduced riparian vegetation, LWD inputs, and productive intertidal mudflats.

Urban stormwater and agricultural runoff may be contaminated with pesticides, oil, grease, heavy metals, polycyclic aromatic hydrocarbons (PAHs), and other organics and nutrients (California Regional Water Quality Control Board-Central Valley Region [Regional Board] 1998). These can potentially destroy aquatic life necessary for salmonid survival (NMFS 1996). Point source and non-point source (NPS) pollution occurs at almost every point that urbanization activity influences the watershed. Impervious man-made surfaces reduce water infiltration and increase runoff, thus creating greater flood hazard (NMFS 1996). Juvenile salmonids are exposed to increased water temperatures from municipal, industrial, and agricultural discharges.

Past mining activities removed spawning gravels from streams, channelized streams, and leached toxic effluents into streams. Many of these effects persist today. Present day mining practices such as sand and gravel mining, suction dredging, and placer mining are typically less intrusive than historic operations (hydraulic mining), but adverse impacts to salmonid habitat still occur.

Water Quality

The water quality of the Delta has been negatively impacted over the last 150 years. Increased water temperatures, decreased DO levels, and increased turbidity and contaminant loads have degraded the quality of the aquatic habitat for the rearing and migration of salmonids. The Central Valley Regional Quality Control Board, in its 1998 Clean Water Act §303(d) list characterized the Delta as an impaired waterbody having elevated levels of a variety of pesticides, electrical conductivity (EC), mercury, low DO, and organic enrichment (Regional Board 1998, 2001). 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 or sublethal effects that reduce health and survival over an extended period of time.

In the aquatic environment, many anthropogenic chemicals and waste materials including toxic organic and inorganic chemicals eventually accumulate in sediment (e.g., Alpers et al. 2008). Direct exposure to contaminated sediments may cause deleterious effects to listed salmonids or the threatened green sturgeon. This may occur if a fish swims through a plume of the resuspended sediments or rests on contaminated substrate and absorbs the toxic compounds through dermal contact, ingestion, or uptake across the gills. Elevated contaminant levels may be found in localized “hot spots” where discharge occurs or where river currents deposit sediment loads. However, the more likely route of exposure to salmonids or sturgeon is through the food chain, when the fish feed on organisms that are contaminated with toxic compounds (Alpers et al. 2008). Prey species become contaminated either by feeding on the detritus associated with the sediments or dwelling in the sediment itself. Therefore, the degree of exposure to salmonids depends on their trophic level and the amount of contaminated forage base they consume. Response of salmonids to contaminated sediments is similar to water borne exposures.

Hatchery Operations

Five hatcheries currently produce Chinook salmon in the Central Valley. Releasing large numbers of hatchery fish can pose a threat to wild Chinook salmon stocks through genetic impacts, competition for food and other resources between hatchery and wild fish, predation of hatchery fish on wild fish, and increased fishing pressure on wild stocks as a result of hatchery production (Waples 1991). The genetic impacts of artificial propagation programs in the Central Valley primarily are caused by straying of hatchery fish and the subsequent interbreeding of hatchery fish with wild fish. Hatchery practices as well as spatial and temporal overlaps of habitat use and spawning activity between spring- and fall-run Chinook salmon have led to the hybridization and homogenization of some subpopulations (DFG 1998).

For Central Valley steelhead, two artificial propagation programs (Coleman National Fish Hatchery and the Feather River Fish Hatchery) may present additional threats to the natural steelhead population. These include mortality of natural steelhead in fisheries targeting hatchery-origin steelhead, competition, and predation by hatchery-origin fish on younger natural fish, genetic introgression by hatchery-origin fish that spawn naturally and interbreed with local natural populations, disease transmission, and fish passage impediments from hatchery facilities (NMFS 2008a).

Over Utilization (Commercial and Sport)

OCEAN COMMERCIAL AND SPORT HARVEST – CHINOOK SALMON

Extensive ocean recreational and commercial troll fisheries for Chinook salmon exist along the Northern and Central California coast. The ocean harvest rates of Sacramento River winter- and spring-run Chinook salmon are thought to be a function of the Central Valley Chinook salmon ocean harvest index (CVI), which is defined as the ratio of ocean catch south of Point Arena, California, to the sum of this catch and the escapement of Chinook salmon to Central Valley streams and hatcheries (Good et al. 2005). CWT returns indicate that Sacramento River salmon congregate off the California coast between Point Arena and Morro Bay.

From 1970 to 1995, the CVI ranged between 0.50 and a record high of 0.79 (1990). In 1996 and 1997, NMFS issued a BO which concluded that incidental ocean harvest represented a significant source of mortality to the endangered population, even though ocean harvest was not a key factor leading to the decline of the population. As a result, measures were developed and implemented by the Pacific Fisheries Management Council, NMFS, and CDFG to reduce ocean harvest by approximately 50 percent. In 2001 the CVI dropped to 0.27, as a result of reduced harvest, record spawning escapement of fall-run Chinook salmon in 2001 (approximately 540,000 fish) and concurrent increases in other Chinook salmon runs in the Central Valley (Good et al. 2005).

INLAND SPORT HARVEST – CHINOOK SALMON

Since 1987, the Fish and Game Commission has adopted increasingly stringent regulations to reduce and virtually eliminate the in-river sport fishery for winter-run Chinook. These closures have virtually eliminated impacts on winter-run Chinook caused by recreational angling in freshwater. In 1992, the California Fish and Game Commission adopted gear restrictions and regulations to reduce the potential for injury and mortality.

In-river recreational fisheries historically have taken spring-run Chinook throughout the species' range. During the summer, holding adults are easily targeted by anglers when they congregate in large pools or at fish ladders. The significance of poaching on the adult population is unknown. Specific regulations have been implemented to protect spring-run Chinook in important spawning creeks. The current regulations, including those developed for winter-run Chinook provide some level of protection for spring-run fish (DFG 1998).

CENTRAL VALLEY STEELHEAD OVERUTILIZATION FOR COMMERCIAL, RECREATIONAL, SCIENTIFIC, OR EDUCATIONAL PURPOSES

Overutilization for commercial, recreational, scientific or educational purposes does not appear to have a significant impact on CV steelhead populations, but warrants continued assessment. Steelhead have been, and continue to be, an important recreational fishery throughout their range. Although there are no commercial fisheries for steelhead in the ocean, inland steelhead fisheries include tribal and recreational fisheries. In the Central Valley, recreational fishing for hatchery-origin steelhead is popular, but is restricted to only visibly marked fish of surplus hatchery-origin, which reduces the likelihood of catching naturally-spawned wild fish. The impact of these fisheries is unknown, however, because the sizes of Central Valley steelhead populations are unknown (Good et al. 2005).

Scientific and educational projects permitted under sections 4(d) and 10(a)(1)(A) of the ESA stipulate specific conditions to minimize take of Central Valley salmonid individuals during permitted activities. There are currently eleven active permits in the Central Valley that may affect steelhead. These permitted studies provide information that is useful to the management and conservation of the DPS.

Disease and Predation

Salmonids are exposed to numerous bacterial, protozoan, viral, and parasitic organisms in spawning and rearing areas, hatcheries, migratory routes, and the marine environment (NMFS 1996, Myers et al. 1998). Very little current or historical information exists to quantify changes in infection levels and mortality rates attributable to these diseases; however, studies have shown that wild fish tend to be less susceptible to pathogens than are hatchery-reared fish. Nevertheless, wild salmonids may contract diseases that are spread through the water column (i.e., waterborne pathogens) as well as through interbreeding with infected hatchery fish.

Accelerated predation of juveniles may also be a factor in the decline. Human-induced habitat changes such as alteration of natural flow regimes and installation of bank revetment and structures often provide conditions that both disorient juvenile salmonids and attract predators (Decato 1978, Vogel et al. 1988, Garcia 1989). The risk from predatory fish can be increased due to turbulent conditions near structures, prolonged travel time due to flow alteration and reduction, and predators awaiting at salvage release sites (Edwards et al. 1996, Tillman et al. 1996, NMFS 1997, Orsi 1967, Pickard et al. 1982). High rates of predation are known to occur at diversion facilities on the mainstem Sacramento River (e.g., RBDD) and the South Delta (e.g. Clifton Court Forebay) and along rock revetment (CDFG 1998). The rates and effects of predation on the population, however, are difficult to determine. Fish-eating birds and mammals can also contribute to the loss of migrating juvenile salmonids (NMFS 2008a), although the level of this effect has not been measured.

Non-native Invasive Species

As currently seen in the San Francisco estuary, non-native invasive species can alter the natural food webs that existed prior to their introduction (Sommer 2007, Baxter et al. 2008). Perhaps the most significant example is illustrated by the Asiatic freshwater clams *Corbicula fluminea* and *Potamocorbula amurensis*. The arrival of these clams in the estuary disrupted the normal benthic community structure and depressed phytoplankton levels in the estuary due to the highly efficient filter feeding of the introduced clams (Cohen and Moyle 2004). The decline in phytoplankton reduces zooplankton that feed upon them, and hence reduces the forage base available to salmonids in the Delta.

Attempts to control non-native invasive species, such as chemical treatments to control the invasive water hyacinth and *Egeria densa,* may also adversely impact salmonid health through chemical effects and decreased in DO from decaying vegetation (NMFS 2008a).

Ocean Survival and Environmental Variation and Climate Change

Natural changes in the freshwater and marine environments play a major role in salmonid abundance (NMFS 2008a, Lindley et al. 2009). Lindley et al. (2009) examined the recent variation in Sacramento River chinook escapement and suggested that variations in salmon productivity over broad geographic areas may be due regional environmental variation, such as widespread drought or floods affecting hydrologic conditions (e.g., river flow and temperature), or regional variation in ocean conditions (e.g., temperature, upwelling, prey and predator abundance). Variations in ocean climate have been increasingly recognized as an important cause of variability in the landings, abundance, and productivity of salmon (reviewed in Lindley et al. 2009). The Pacific Ocean has many modes of variation in sea surface temperature, mixed layer depth, and the strength and position of winds and currents, including the El Niño-Southern Oscillation, the Pacific Decadal Oscillation and the Northern Oscillation. The broad variation in physical conditions creates corresponding variation in the pelagic food webs upon which juvenile salmon depend, which in turn creates similar variation in the population dynamics of salmon across the north Pacific.

The different Central Valley stocks appear to respond differently to recent environmental variation, especially ocean conditions (Lindley et al. 2009). Almost all fall-run Chinook populations have rapidly declined from peak abundances around 2002. In contrast, late-fall, winter and naturally-spawning spring-run Chinook populations have been increasing in abundance over the past decade, although escapement in 2007 was down in some of them and the growth of these populations through the 1990s and 2000s has to some extent been driven by habitat restoration efforts. One factor may be hatchery practices that reduce demographic variation. The other factor may be the different life history tactics of the other salmon runs. Spring-run Chinook juveniles enter the ocean at a broader range of ages (with a portion of some populations migrating as yearlings) than fall Chinook, due to their use of higher elevations and colder waters. Winter-run Chinook spawn in summer, and the juveniles enter the ocean at a larger size than fall Chinook, due to their earlier emergence and longer period of freshwater residency. If ocean conditions at the time of ocean entry are critical to the survival of juvenile salmon, then populations from different runs should respond differently to changing ocean conditions because they enter the ocean at different times and at different sizes (Lindley et al. 2009).

Ecosystem Restoration

CALIFORNIA BAY-DELTA AUTHORITY

Two programs included under CBDA were created to improve conditions for fish, including listed salmonids, in the Central Valley: (1) the ERP and its Environmental Water Program, and (2) the EWA managed under the Water Supply and Reliability Program (CALFED 2000). Restoration actions implemented by the ERP include the installation of fish screens, modification of barriers to improve fish passage, habitat acquisition, and instream habitat restoration. The majority of these actions address key factors affecting listed salmonids and emphasis has been placed in tributary drainages with high potential for spring-run Chinook production. Additional ongoing actions include new efforts to enhance fisheries monitoring and directly support salmonid production through hatchery releases. Recent habitat restoration initiatives sponsored and funded primarily by the CBDA-ERP have resulted in plans to restore ecological function to 9,543 acres of shallow-water tidal and marsh habitats within the Delta. Restoration of these areas primarily involves flooding lands previously used for agriculture, thereby creating additional rearing habitat for juvenile salmonids. Similar habitat restoration is imminent adjacent to Suisun Marsh (i.e., at the confluence of Montezuma Slough and the Sacramento River) as part of the Montezuma Wetlands project, which is intended to provide for commercial disposal of material dredged from San Francisco Bay in conjunction with tidal wetland restoration.

A review of CALFED’s performance in Years 1 through 8 concluded that the greatest investments and results of the ERP and Watershed Programs have been in areas upstream from the Delta (CALFED BDPAC 2007). Significant investments made there in fish screens, temperature control, fish passage improvements and upstream habitats have resulted in an improved outlook for salmon throughout the Central Valley. Unfortunately, efforts have been less successful at acquiring and protecting important lands in the Delta along its tributary rivers and streams (CALFED BDPAC 2007)

The CBDA has two water acquisition programs: the Environmental Water Program (EWP) and the EWA. The EWP is a subprogram of the ERP designed to support ERP projects through enhancement of instream flows, principally for the benefit of listed salmonids, in anadromous reaches of priority streams controlled by dams. As of 2007, however, little progress has been made on purchasing water rights for fish in important spawning tributaries (CALFED BDPAC 2007).

The EWA is designed to provide water at critical times to meet ESA requirements and incidental take limits without water supply impacts to other users, particularly South of Delta water users. In early 2001, the EWA released 290 thousand acre feet of water from San Luis Reservoir at key times to offset reductions in South Delta pumping implemented to protect winter-run Chinook salmon, delta smelt, and splittail. However, the benefit derived by this action to winter-run Chinook salmon in terms of number of fish saved was very small. The EWA has been very successful at eliminating conflict between protection of Delta fish and export water supply. From 1995 through 2006, no conflicts between fish and water supply occurred that resulted in uncompensated water supply reductions. It is uncertain whether EWA actions are having any favorable impact on Delta species in a system that continues to rely on through-Delta conveyance. Actions taken to protect anadromous species have had a positive influence on the species, but actions outside the Delta have been far more effective in improving populations than the EWA actions in the Delta.

Currently, the EWA program is authorized through 2010 and is scheduled to be reduced in its scope. Future EWA operations will be considered to have limited assets and will primarily be used only during CVP and SWP pumping reductions in April and May as a result of the Vernalis Adaptive Management Program (VAMP) experiments. In this case, EWA assets will be used to offset "uncompensated losses" to CVP and SWP water contractors for fisheries related actions. The primary source of EWA assets through 2015 will come from the 60,000 acre-feet of water transferred to the State under the Yuba Accord.

CENTRAL VALLEY PROJECT IMPROVEMENT ACT

The Central Valley Project Improvement Act (CVPIA), implemented in 1992, requires that fish and wildlife get equal consideration with other demands for water allocations derived from the CVP. From this act arose several programs that have benefited listed salmonids: the Anadromous Fish Restoration Program (AFRP), the Anadromous Fish Screen Program (AFSP), and the Water Acquisition Program (WAP). The AFRP is engaged in monitoring, education, and restoration projects geared toward recovery of all anadromous fish species residing in the Central Valley. Restoration projects funded through the AFRP include fish passage, fish screening, riparian easement and land acquisition, development of watershed planning groups, instream and riparian habitat improvement, and gravel replenishment. The AFSP combines Federal funding with State and private funds to prioritize and construct fish screens on major water diversions mainly in the upper Sacramento River. The goal of the WAP is to acquire water supplies to meet the habitat restoration and enhancement goals of the CVPIA and to improve the Department of the Interior's ability to meet regulatory water quality requirements. Water has been used successfully to improve fish habitat for spring-run Chinook salmon by maintaining or increasing instream flows in Butte and Mill Creeks and the San Joaquin River at critical times.

IRON MOUNTAIN MINE REMEDIATION

Environmental Protection Agency's Iron Mountain Mine remediation involves the removal of toxic metals in acidic mine drainage from the Spring Creek Watershed. Contaminant loading into the Sacramento River from Iron Mountain Mine has shown measurable reductions since the early 1990s (see Reclamation 2004 Appendix J). Decreasing the heavy metal contaminants that enter the Sacramento River should increase the survival of salmonid eggs and juveniles. However, during periods of heavy rainfall upstream of the Iron Mountain Mine, Reclamation substantially increases Sacramento River flows in order to dilute heavy metal contaminants being spilled from the Spring Creek debris dam. This rapid change in flows can cause juvenile salmonids to become stranded or isolated in side channels below Keswick Dam.

SWP DELTA PUMPING PLANT FISH PROTECTION AGREEMENT (FOUR-PUMPS AGREEMENT)

The 1986 'Four Pumps Agreement' between the DWR and DFG was established to offset direct losses of Chinook salmon, steelhead and striped bass caused by the diversion of water at the SWP's Harvey O. Banks Delta Pumping Plant (DWR and DFG 1986). Since 1986 approximately $59 million has been approved for over 40 fish mitigation projects. About $44 million of the approved funds have been expended to date and the remaining approved funds are allocated for new or longer term projects (DWR 2008). Four Pumps projects that benefit spring-run Chinook salmon include water exchange programs on Mill and Deer Creeks to provide salmon passage flows; enhanced law enforcement; fish screens and ladders on Butte Creek; and screening of diversions in Suisun Marsh and San Joaquin tributaries. Passage projects, migration flows, and enhanced enforcement for spring-run Chinook continue to be priority projects, as do natural production projects for steelhead.

3.1.2.8 Status of the Species within the Action Area

The Sacramento-San Joaquin Delta serves as the gateway through which all listed anadromous species in the Central Valley must pass through on their way to spawning grounds as adults or retuning to the ocean as juveniles or post-spawn adults (for steelhead). The temporal and spatial occurrence of each of the runs of salmonids is intrinsic to their natural history and the exposure to the action can be anticipated based on their timing and location (Table 3-7) (NMFS 2008a).

Sacramento River winter-run Chinook Salmon

The main adult winter-run migration route is the mainstem Sacramento River, which skirts the northwest portion of the Delta. The Action Area does not overlap designated critical habitat for winter-run Chinook (Figure 3-7). However, there is the potential for a small number of adults to “stray” into the San Joaquin River side of the Delta while on their upstream migration, particularly early in the migratory season (November and December) (NMFS 2008a). Juvenile winter-run emigrants are susceptible to being “carried” into the Central and South Delta by the flow splits through the DCC (when open), Georgiana Slough, Three Mile Slough, and Broad Slough and subsequently being entrained by the effects of pumping at the CVP and SWP once entering the Central Delta. Juvenile winter-run are present in the waterways of the west, north , central, and south Delta waterways leading to the CVP and SWP pumping facilities including the Old and Middle river channels.

Central Valley spring-run Chinook Salmon

Spring-run Chinook occur in the Action Area, as evidenced by salvage at the south Delta pumps. However, the Action Area does not include designated critical habitat for spring-run Chinook (Figure 3-8). Adult spring-run enter the San Francisco Bay Estuary from the ocean in January to late February. They move through the Delta prior to entering the Sacramento River system. Spring-run show two distinct juvenile emigration patterns. Fish may either emigrate to the Delta and ocean during their first year of life as YOY, typically in the following spring after hatching, or hold over in their natal streams and emigrate the following fall as yearlings. Typically, yearlings enter the Delta as early as November and December and continue to enter the Delta through at least March. They are larger and less numerous than the YOY smolts that enter the Delta from January through June. The peak of YOY spring-run presence in the Delta is during the month of April, as indicated by the recoveries of spring-run size fish in the CVP and SWP salvage operations and the Chipps Island trawls. Frequently, it is difficult to distinguish the YOY spring-run outmigration from that of the fall-run due to the similarity in their spawning and emergence times. The overlap of these two runs makes for an extended pulse of Chinook salmon smolts through the Delta each spring, frequently lasting into June.

Table 3-7 Temporal Occurrence of Salmonids and Sturgeon within the Delta

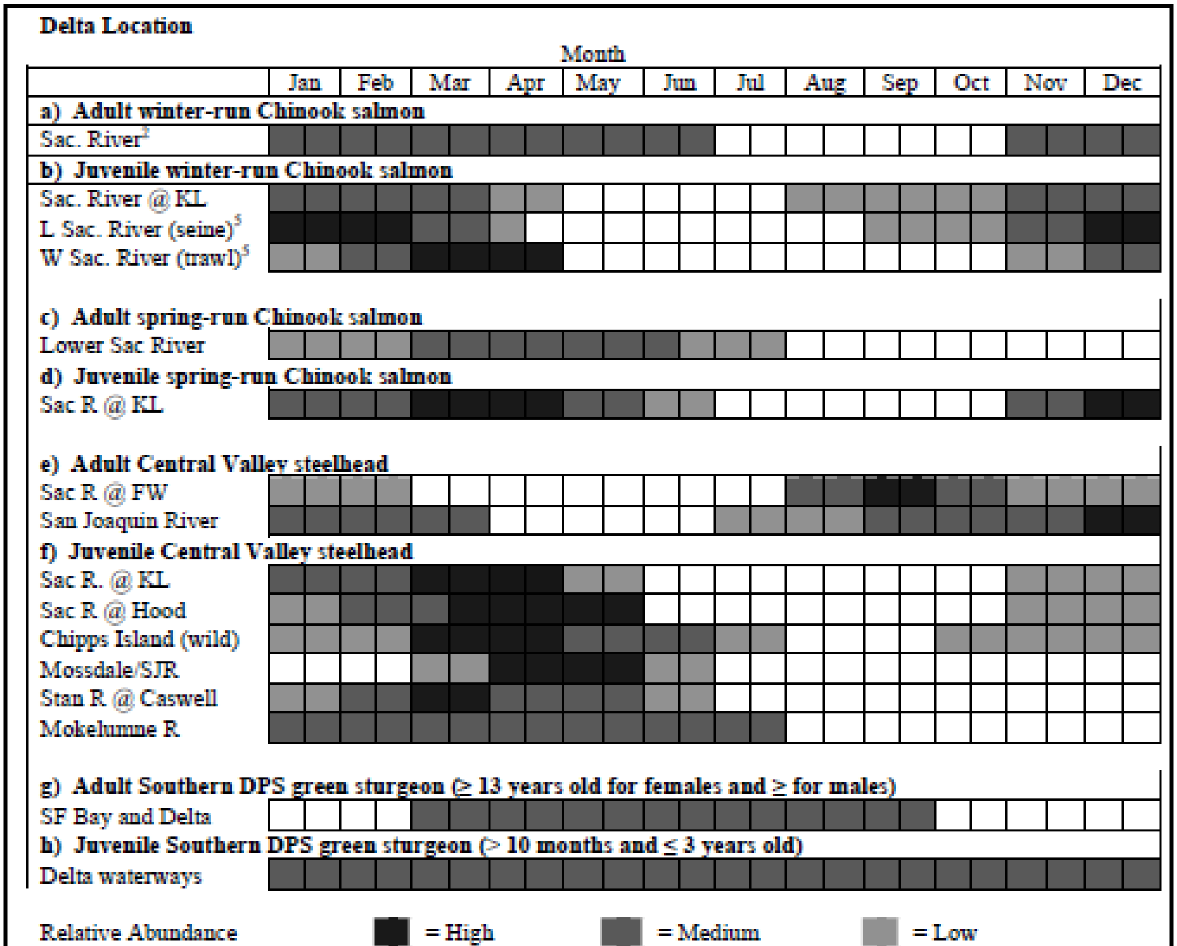


Figure 5-25. Temporal distribution of anadromous fish species within the Delta (KL = Knights Landing, FW = Fremont Weir).

Source: NMFS 2008a

Central Valley steelhead

The Action Area overlaps a portion of the designated critical habitat for CV steelhead (Figure 3-9). Adult steelhead have the potential to be found within the Delta during any month of the year. Typically, adults begin to enter the Delta during mid to late summer, and enter the Sacramento River system from July to early September. Post-spawning adults (kelts) are typically seen later in the spring following spawning. Steelhead entering the San Joaquin River basin are believed to enter the system in late October through December (NMFS 2008a).

Juvenile steelhead are recovered in the USFWS Chipps Island trawls from October through July. There appears to be a difference in the emigration timing between wild and hatchery-reared steelhead smolts. Adipose fin-clipped hatchery fish are typically recovered at Chipps Island from January through March, with the peak in February and March. This time period corresponds to the schedule of hatchery releases of steelhead smolts from the different Central Valley hatcheries (Nobriga and Cadrett 2001, Reclamation 2008). The timing of wild steelhead (unclipped) emigration is more spread out, with peaks in February and March, based on salvage records at the CVP and SWP fish collection facilities. Individual unclipped fish first begin to be collected in fall and early winter, and may extend through early summer (June and July). Wild fish that are collected at the CVP and SWP facilities late in the season may be from the San Joaquin River system, based on the proximity of the basin to the pumps and the timing of the spring pulse flows in the tributaries (April-May). The size of emigrating steelhead smolts typically ranges from 200 to 250 mm in length, with wild fish tending to be at the upper end of this range (Reclamation 2008, Nobriga and Cadrett 2001).

3.1.3 Southern Distinct Population Segment of North American Green Sturgeon

3.1.3.1 Listing Status and Designated Critical Habitat

The Southern DPS of North American green sturgeon was listed as threatened on April 7, 2006 (71 FR 17757) and consists of coastal and Central Valley populations south of the Eel River in California. The Southern DPS presently contains only a single known population that spawns and rears in the Sacramento River system, including the Sacramento, Feather and Yuba Rivers, Sacramento-San Joaquin Delta and Suisun, San Pablo and San Francisco Bays.

Critical habitat for the Southern DPS was proposed on September 8, 2008 (NMFS 2008b; 73 FR 52084). Proposed critical habitat includes freshwater riverine habitats (stream channel defined by the ordinary high water line), bay and estuarine habitat (lateral extent of the mean higher high water line), and coastal marine habitat (to the 110 m [361 foot] depth contour). Proposed critical habitat for the Southern DPS is found within the Action Area, specifically within the Sacramento-San Joaquin Delta (Figure 3-16).

3.1.3.2 Life History

North American green sturgeon (green sturgeon) are among the largest of the bony fish (Moyle 2002). Green sturgeon are an anadromous, slow-growing, late-maturing and long-lived species (Nakamoto et al. 1995, Farr et al. 2002). Maximum age is likely 60-70 years or more (Moyle 2002). Little is known about the life history of green sturgeon because of its low abundance, low sportfishing value, and limited spawning distribution, but spawning and larval ecology are assumed to be similar to that of white sturgeon (Moyle 2002; Beamsderfer and Webb 2002).

Green sturgeon are mostly marine fish. Adults and subadults enter the San Francisco Estuary during the spring and remain until autumn (Kelly et al. 2007). Recent telemetry studies of fish captured in San Pablo Bay found that movements were not related to salinity, current, or temperature, leading Kelly et al. (2007) to surmise that movements are related to resource availability. Green sturgeon were most often found at depths greater than 5 meters with low or no current during summer and autumn months, presumably conserving energy (Erickson et al. 2002). Adults may utilize a variety of freshwater and brackish water habitats for up to nine months of the year.

Southern DPS green sturgeon currently spawn well upstream of the Action Area in the Sacramento River above Hamilton City and perhaps as far upstream as Keswick Dam (DFG 2002 in Adams et al. 2002). Spawning occurs in the upper river, particularly around the RBDD (Brown 2007). Spawning in the San Joaquin River system has not been recorded, but it is likely that sturgeon historically utilized this basin. Spawning occurs in deep pools in large, turbulent river mainstreams from March to July, with a peak in mid-April to mid-June (Moyle et al. 1992).

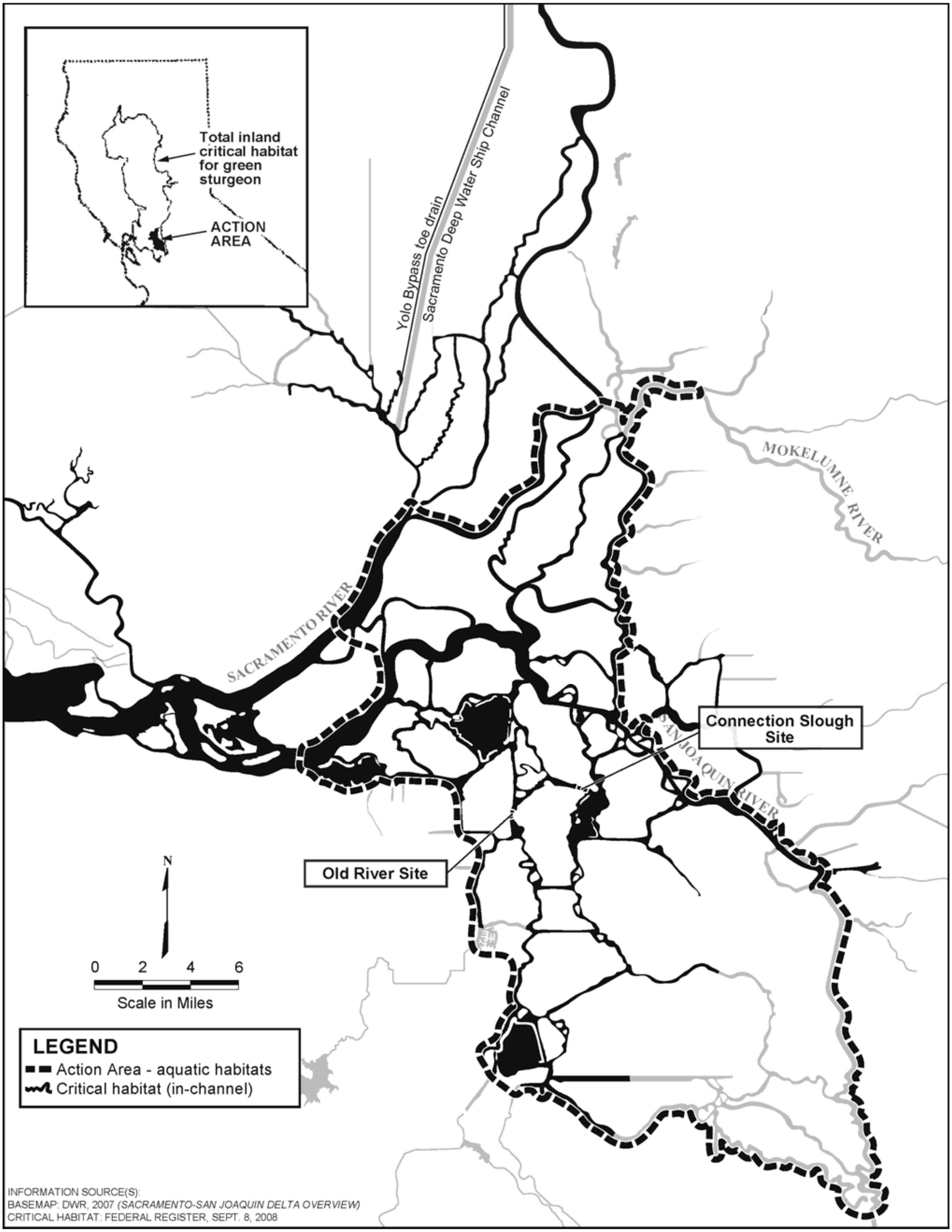


Figure 3-16 Designated Critical Habitat for Southern DPS North American Green Sturgeon

Green sturgeon larvae disperse downstream from Sacramento River spawning areas soon after hatching and rear as juveniles and subadults for several years throughout the Sacramento-San Joaquin Delta before migrating into the ocean (Beamesderfer et al. 2007). Little is known about larval rearing habitat requirements (NMFS 2008a). In the Klamath River, juvenile green sturgeon are reported to grow rapidly to 300 mm in one year and to over 600 mm within 2-3 years (Nakomoto et al. 1995).

Green sturgeon feed on benthic invertebrates including shrimp, mollusks and amphipods, and occasionally small fish (Moyle et al. 1992). The non-native overbite clam (*Potamocorbula amurensis*) has also been found in green sturgeon (Adams et al. 2002).

Green sturgeon in a telemetry study ranged widely from San Pablo Bay through the San Francisco Estuary, from warm, shallow brackish areas in Suisun Bay to the colder, deeper, oceanic region near the Golden Gate (Kelly et al. 2007). In general, they remained in shallow regions of the bay swimming over bottom depths less than 10m. Movements were both nondirectional and closely associated with the bottom (presumably foraging), or directional continuous swimming in the upper 20 percent of the water column. Nocturnal behavior has been observed in captive-reared larval and juvenile green sturgeon (9–10 months old). This may be an adaptation for avoiding predation during dispersal migration and first-year wintering in riverine habitat (Adams et al. 2002, Kynard et al. 2005).

Juveniles rear in fresh and estuarine waters for about 1 to 4 years (Nakamoto et al. 1995, NMFS 2008a). Juveniles seem to outmigrate in the summer and fall before the end of their second year (Moyle 2002). They disperse widely in the ocean after their outmigration from freshwater and before their return spawning migration (Moyle et al. 1992b).

Green sturgeon spend most of their lives in the ocean and their distribution and activities in the marine environment are poorly understood (Moyle et al. 1992b, Beamesderfer et al. 2007). Green sturgeon migrate considerable distances northward along the Pacific Coast and into other estuaries, particularly the Columbia (Adams et al. 2002). Columbia River green sturgeon are a mixture of fish from the Sacramento, Klamath, and Rogue Rivers (Israel et al. 2004).

Adults reach sexual maturity only after many years of growth: 9-13 years for males and 13-27 years for females (Nakomoto et al. 1995, Van Eenennaam et al. 2006). Spawning periodicity is once every 2-4 years (Erickson and Webb 2007).

3.1.3.3 Distribution

Green sturgeon are the most widely distributed and most marine-oriented of the sturgeon family Ascipenseridae (Moyle 2002). They range offshore along the Pacific Coast from Ensenada Mexico to the Bering Sea and in rivers from British Columbia to the Sacramento River (Moyle 2002). In North America, spawning populations are currently found in only three river systems, the Sacramento and Klamath Rivers in California and the Rogue River in southern Oregon. Two species of sturgeon are sympatric in California, green sturgeon and white sturgeon (*A. transmontanus*), which is more abundant and subject to sportfishing.

Two green sturgeon DPSs, Northern and Southern, were identified based on evidence of spawning site fidelity (indicating multiple DPS tendencies), and on the preliminary genetic evidence that indicates differences at least between the Klamath River and San Pablo Bay samples (Adams et al. 2002). The Northern DPS includes all green sturgeon populations starting with the Eel River (northern California) and extending northward. The Southern DPS includes all green sturgeon populations south of the Eel River, with the only known spawning population being in the Sacramento River. The distribution of the two DPSs outside of natal waters generally overlap with each other, including aggregations in the Columbia River estuary and Washington estuaries in late summer (reviewed in NMFS 2008b).

When not in the ocean, green sturgeon occupy freshwater and estuarine habitat in the Sacramento River (upstream to Keswick Dam), lower Feather River, lower Yuba River, the Sacramento-San Joaquin Delta, and the Suisun, San Pablo and San Francisco Bays. Table 3-8 illustrates the temporal distribution of Southern DPS green sturgeon.

Adults migrate in spring to spawning grounds in the Sacramento River and outmigrate in early summer to the ocean (NMFS 2008a). Green sturgeon have not been documented spawning or rearing in the San Joaquin River or its tributaries, although no directed sturgeon studies have ever been undertaken in the San Joaquin River (DFG 2002, Adams et al. 2002, Beamesdorfer et al. 2007). Observations of green sturgeon juveniles or unidentified sturgeon larvae in the San Joaquin River have been limited to the Delta, where they could easily, and most likely, have originated from the Sacramento River (Beamesderfer et al. 2004 in NMFS 2008b).

Table 3-8 The Temporal Occurrence of Southern DPS of North American Green Sturgeon Life Stages

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Adult Immigration, Holding and Spawning (>13 yrs for females, >9 yrs for males)												
Upper Sac River[1, 2, 3]			Medium	Medium / High	High	High / Medium	Medium	Medium	Medium	Medium	Medium	
SF Bay Estuary[4, 8]			Medium	Medium	Medium	Medium	Medium	Medium	Medium			
Larval / Post-Larval Rearing (<10 mos)												
RBDD, Sac River[5]					Medium	High	High	Medium				
GCID, Sac River[5]					Medium	High	High	Medium	Low	Low		
Juvenile Rearing (>10 mos and <3 yrs)												
Sac-SJ Delta[6]	Medium	Medium	Medium	Medium	Medium	Medium	Medium	Medium	Medium	Medium	Medium	Medium
Sac-SJ Delta[5]				Medium	Medium	Medium	Medium	Medium	Medium	Medium		
Suisun Bay[5]	Medium	Medium	Medium	Medium	Medium	Medium	Medium	Medium	Medium	Medium	Medium	Medium
Subadult and Adult Coastal Migrant (3-13 yrs for females, 3-9 yrs for males)												
Pacific Coast[3,7]	Medium	Medium	Medium	Medium	Medium	Medium	Medium	Medium	Medium	Medium	Medium	Medium
Salvage [6,9]	Low	Low	Low	Low	Low	Low	Low	Low	Low	Low	Low	Low
Relative Abundance	High	=High	Medium	=Medium	Low	=Low						

Sources: [1] USFWS (2002); [2] Moyle et al. (1992), [3] Adams et al. (2002) and NMFS (2005), [4] Kelley et al. (2006), [5] DFG (2002), [6] Interagency Ecological Program Relational Database, fall midwater trawl green sturgeon captures from 1969 to 2003, [7] Nakamoto et al. (1995), [8] Heublein et al. (2006), [9] Fish Facility salvage operations (not a useful criteria for analysis due to very low numbers, ENTRIX 2008)

Source: USBR 2008, NMFS 2008a, ENTRIX 2008

Green sturgeon juveniles, subadults and adults are widely distributed in the Delta and estuary areas including San Pablo Bay (Beamesderfer et al. 2007). Subadults and non-breeding adults inhabit the Delta and bays during summer months, most likely for feeding and growth (Kelly et al. 2007, Moser and Lindley 2007). Juvenile green sturgeon have been salvaged at the SWP and CVP fish facilities in the South Delta, and captured in trawling studies by the CDFG during all months of the year (CDFG 2002). The majority of these fish were 200-500 mm (estimated 2–3 years old) (Nakamoto et al. 1995). The lack of a significant proportion of juveniles smaller than approximately 200 mm (~7.9 inches) in Delta captures indicates juvenile Green sturgeon likely hold in the mainstem Sacramento River, as suggested in Klamath River studies by Kynard et al. (2005).

3.1.3.4 Abundance

Reliable population estimates are not available for any green sturgeon population (Beamesderfer et al. 2007). Population abundance and the limitations in estimates are discussed in the NMFS status reviews (Adams et al.

2002 and 2007, NMFS 2005 and 2008b). Green sturgeon have always been uncommon within the Delta (Moyle 2002). What limited information exists comes mainly from incidental captures of green sturgeon during the CDFG's white sturgeon monitoring program in San Pablo Bay (CDFG 2002). These estimates, however, are confounded by small sample sizes, intermittent reporting, fishery-dependent data from sportfishing, subsamples representing only a portion of the population, and potential confusion with white sturgeon (Adams et al. 2002, NMFS 2005, Beamesderfer et al. 2007). The most notable biases are the assumptions of equal capture probabilities to the gear and similar seasonal distributions (green sturgeon concentrate in estuaries only during summer and fall, while white sturgeon may remain year round) (Adams et al. 2002 and 2007). Generally, green sturgeon catches are much lower than those for white sturgeon, precluding attempts to infer green sturgeon abundance from white sturgeon mark-recapture studies (Reclamation 2008).

The only abundance trend information available for the Southern DPS of green sturgeon comes from salvage data at the state and federal water export facilities (CDFG 2002, Adams et al. 2002). Green sturgeon taken at the facilities are usually juveniles (28–38 cm length), although an adult over 2 m TL was taken in the spring of 2003 at the USBR's Tracy Fish Collection Facility (Wang 2006 in NMFS 2008b). At the State of California's John E. Skinner Fish Facility, the average number of green sturgeon taken annually was 732 prior to 1986, but only 47 between 1986 and 2001 (Adams et al. 2002, 70 FR 17386). For the federal facility the average number was 889 prior to 1986, but only 32 between 1986 and 2001 (70 FR 17386). Estimates from salvage data do have their limitations, however (Adams et al. 2002, 71 FR 17757). Nevertheless, in light of the increased exports, particularly during the previous 10 years, it is clear that Southern DPS abundance is dropping.

Catches of sub-adult 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, 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, and the portion represented by Southern DPS green sturgeon is unknown (NMFS 2008b).

3.1.3.5 Population Viability Summary for Green Sturgeon

Abundance

Currently, no reliable data on population size exists and data on population trends is 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).

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

The Southern DPS of North American Green Sturgeon only includes a single 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 comprise separate populations. Therefore, the apparent presence of only one reproducing population puts the DPS at risk.

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.

3.1.3.6 Critical Habitat and Primary Constituent Elements

Critical habitat for the Southern DPS of North American Green sturgeon was proposed in 2008 (73 FR 52084) and generally has physical and biological features or PCEs similar to those described for listed salmonids. NMFS's Critical Habitat Recovery Team defined the geographical area occupied to range from the California/Mexico border north to the Bering Sea, Alaska. Within the geographical area, 39 occupied specific areas and seven presently unoccupied areas were delineated within freshwater rivers, coastal bays and estuaries, and coastal marine waters. The Action Area occurs in the freshwater riverine system. The PCE's for the three habitat classes are briefly described below, with further details in the 2008 Draft Biological Report (NMFS 2008b).

Freshwater Riverine Systems

The life stages that use freshwater habitats include adult migration, holding and spawning; egg incubation; larval development and growth; and juvenile rearing and downstream migration. Specific PCE's for freshwater riverine systems include:

- Abundant food resources for larvae, juveniles, subadult and adult life stages, principally benthic invertebrates and small fish;
- Adequate substrate such as cobbles suitable for spawning, incubation and larval development;
- Sufficient water flow for egg incubation, larval development, passage and trigger flows for migrating adults);
- Good water quality such as temperature below 17 degrees (°) C for eggs and below 20°C for juveniles, salinity below 3 ppt for eggs and larvae and below 10 ppt for juveniles, and free of contaminants;
- An unobstructed migratory corridor through the Delta and lower Sacramento River for adults migrating to upstream spawning areas and downstream migrating juveniles;
- Deep pools for holding adults and subadults; and
- Sediments free from elevated levels of contaminants such as selenium, PAHs, organochlorine pesticides.

Estuarine Areas

Green sturgeon life stages that utilize estuarine areas include migrating adults, foraging subadults and rearing juveniles. Specific PCEs include:

- Abundant food resources for juvenile, subadult and adult life stages consisting primarily of benthic invertebrates and fish;
- Sufficient water flow to allow adults to orient to incoming flow and migrate upstream to spawning grounds in the Sacramento River;
- Good water quality such as water temperature below 24°C, salinity between 10 ppt (brackish) and 33 ppt (salt water), minimum dissolved oxygen levels of 6.54 mg O_2/l, and waters with acceptably low levels of contaminants (e.g. pesticides, organichlorines, elevated levels of heavy metals);

- An unobstructed migratory corridor into and through the estuary for adults migrating to spawning areas in the Sacramento River and for subadults and adults oversummering in bays and estuaries;
- A diversity of depths for shelter, foraging and migration; and
- Sediments free from elevated levels of contaminants such as selenium, PAHs, organochlorine pesticides.

Estuarine areas free of obstruction 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 wood, aquatic vegetation, and side channels, are suitable for foraging juveniles and adults. The remaining estuarine habitat for these species is severely degraded by altered hydrologic regimes, poor water quality, reductions in habitat complexity, and competition for food and space with exotic species. Regardless of the condition, the remaining estuarine areas are of high conservation value because they function as a transition corridor to the ocean environment.

North American green sturgeon use the Delta, San Pablo Bay and San Francisco Bay as a migratory corridor as they move from the ocean to freshwater as adults and from freshwater to the ocean as juveniles. Most movement by adults occurs in deeper channels, while juveniles are more likely to use the shallow habitats, including tidal flats, for feeding and predator refuge.

Coastal Marine Areas

Green sturgeon life stages that utilize coastal marine areas include adults and subadults. Specific PCEs include:

- Unobstructed migratory corridors within marine and between estuarine and marine habitats;
- Good water quality with adequate dissolved oxygen and acceptably low levels of contaminants (e.g. pesticides, organochlorines, elevated levels of heavy metals); and
- Abundant food resources for subadults and adults, which include benthic invertebrates and fish.

3.1.3.7 Factors Affecting Green Sturgeon and proposed Critical Habitat

Summary

The principal risk factors for the Southern DPS of North American green sturgeon include loss of spawning habitat, harvest of adults, and entrainment of fertilized eggs, juveniles and subadults (Adams et al. 2007). Other threats to the Southern DPS include vulnerability due to concentrated spawning within the Sacramento River, a smaller overall population size compared to the Northern DPS, the lack of population data to inform fishery managers, increased summer stream temperatures that can limit larval growth or survival, and the influence of toxic material and exotic species (Adams et al. 2002 and 2007). The Southern DPS is more vulnerable to catastrophic events than the Northern DPS because the population is smaller and spawning appears to be concentrated in the upper Sacramento River above RBDD. Toxins, invasive species, and water project operations, all identified as threats to the Southern DPS of green sturgeon, may be acting in concert or individually to lower pelagic productivity in the Delta (71 FR 17757).

Many of the factors responsible for the current status of green sturgeon in the Central Valley are similar to those described above for winter-run and spring-run Chinook salmon and steelhead (Section 3.1.2.7). Further details are provided in recent BOs prepared by NMFS (2008a, c).

Fish Movement and Habitat Blockage

As with the listed salmonids in the Central Valley, the principal factor for decline of the Southern DPS is the reduction of the spawning area to a limited area of the Sacramento River (71 FR 17757). Hydropower, flood control, and water supply dams of the CVP, SWP, and other municipal and private entities have permanently blocked or hindered access to historical spawning and rearing grounds by a variety of anadromous fish. Keswick Dam provides an impassible barrier blocking green sturgeon access to what were likely historic spawning grounds upstream (USFWS 1995a). Furthermore, the RBDD blocks access to much of the spawning habitat below Keswick Dam. Changes in project operations since 1986 have increased green sturgeon access to spawning grounds above the RBDD (Adams et al. 2002). A substantial amount of habitat in the Feather River above Oroville Dam has also been lost (NMFS 2005).

Potential adult migration barriers to green sturgeon include the RBDD, the Sacramento Deep Water Ship Channel locks, the Fremont Weir at the head of the Yolo Bypass, the Sutter Bypass, the Delta Cross Channel Gates on the Sacramento River, and Shanghai Bench and Sunset Pumps on the Feather River. Most of these barriers are located outside the Action Area.

Water Development and Conveyance

Construction of dams and associated impoundments have altered temperature and hydrologic regimes downstream and has simplified instream habitats in freshwater riverine habitat, which is believed to have substantially decreased spawning success (71 FR 17757). Temperature control efforts to benefit winter-run Chinook may have provided some benefit to green sturgeon in the Sacramento River below Keswick Dam.

Juvenile entrainment is considered a threat imposed by water diversions, but the degree to which it is affecting the continued existence of the Southern DPS remains uncertain (71 FR 17757). The threat of screened and unscreened water diversions in the Sacramento River and Delta is largely unknown as juvenile sturgeon are often not identified and current CDFG and NMFS screen criteria do not address sturgeon. Based on the temporal occurrence of juvenile green sturgeon and the high density of water diversion structures along rearing and migration routes, NMFS (2005) found the potential threat of these diversions to be serious and in need of study.

Southern DPS green sturgeon also face entrainment in pumps associated with the CVP and SWP. Substantial numbers of juveniles have been killed in pumping operations at state and federal water export facilities in the south Delta (DFG 2002, Adams et al. 2007). The average number of fish taken annually at the SWP pumping facility was higher in the period prior to 1986 (732) than from 1986 to the present (47) (DFG 2002). At the CVP pumping facilities, the average annual number prior to 1986 was 889; while the average number was 32 after 1986. However, these estimates should be viewed cautiously because they were expanded from brief sampling periods and very few captured sturgeon, and thus may be exaggerated (Adams et al. 2007).

Flood Control and Levee Construction

The effects of flood control and levee construction on green sturgeon are similar to those described above for salmonids. (Section 3.1.2.7.3)

Land Use Activities

The effects of land use activities on green sturgeon are similar to those described above for salmonids. (Section 3.1.2.7.4)

Water Quality

As described above for salmonids (Section 3.1.2.7.5), the water quality of the Delta and its tributaries has been negatively impacted over the last 150 years. Increased water temperatures, decreased DO levels, and changes in turbidity and increased contaminant loads have degraded the quality of the aquatic habitat for many species including green sturgeon. The upper levels of summer temperatures in the Sacramento River approach growth-limiting and lethal limits for larval green sturgeon (Adams et al. 2002). Temperature control efforts to protect winter-run Chinook have probably been beneficial to green sturgeon in the upper Sacramento River. The Regional Water Quality Control Board characterized the Delta as an impaired waterbody for a variety of issues (such as pesticides, herbicides, mercury, low DO, and organic enrichment) (Regional Board 1998, 2001). Anthropogenic manipulations of the aquatic habitat, such as dredging, bank stabilization, and waste water discharges have also degraded the quality of the Central Valley's waterways for green sturgeon. Toxins, invasive species, and water project operations, all identified as threats to the Southern DPS of North American green sturgeon, may be acting in concert or individually to lower pelagic productivity in the Delta (71 FR 17757).

The potential effect of toxic contaminants on green sturgeon has not be directly studied, but their long life span, late age of maturity, and benthic feeding habits make sturgeon vulnerable to chronic and acute effects of bioaccumulation (COSEWIC 2004). Many contaminants eventually accumulate in sediment, where green sturgeon can be exposed through direct contact with substrate, swimming through resuspended sediments, or more likely through ingestion of contaminated benthic organisms and subsequent bioaccumulation (e.g., Alpers et al. 2008). Selenium studies in the San Francisco Bay and Delta found elevated levels of selenium in white sturgeon, much higher than in non-benthic fishes and approaching levels which may have acute or chronic effects (e.g., Urquhart et al. 1991). While green sturgeon spend more time in the marine environment than white sturgeon and, therefore, may have less exposure, NMFS concluded that green sturgeon face some risk from contaminants when they inhabit estuaries and freshwater (71 FR 17757).

Contamination of the Sacramento River increased substantially in the mid–1970s when application of rice pesticides increased (USFWS 1995b). Estimated toxic concentrations for the Sacramento River between 1970 and 1988 may have deleteriously affected the larvae of another anadromous species (e.g., striped bass) that occupy similar habitat as green sturgeon larvae (Bailey 1994). Studies of the recent POD in the Delta indicate that toxins may be at least partially responsible.

Hatchery Operations

Hatchery operations have not been identified as a potential threat for green sturgeon. White sturgeon are cultivated in hatcheries for commercial aquaculture and for conservation, such as the Kootenay River sturgeon conservation hatchery on the upper Columbia River. There is a possibility of disease transfer from hatchery-raised sturgeon and wild sturgeon; however, there is no evidence that this has ever occurred (COSEWIC 2004). Although aquaculture methods have been developed for green sturgeon, there are currently no hatchery operations for the Southern DPS (J. Van Eenennaam, pers. comm. 2008).

Over-Utilization

Green sturgeon are not a specifically targeted fish species during existing commercial and sport fishery harvest activities and is now almost entirely bycatch in three fisheries: white sturgeon commercial and sport fisheries, Klamath Tribal salmon gill-net fisheries, and coastal groundfish trawl fisheries (Adams et al. 2002 and 2007).

OCEAN AND COMMERCIAL HARVEST

Commercial harvest of white sturgeon results in the incidental bycatch of green sturgeon, primarily along the Oregon and Washington coasts and within their coastal estuaries (Adams et al. 2002, NMFS 2008c). A high proportion of green sturgeon present in the Columbia River, Willapa Bay, and Grays Harbor may be Southern DPS North American green sturgeon (DFG 2002 in Adams et al. 2002, Moser and Lindley 2007). The total average annual harvest of green sturgeon declined from 6,466 in 1985-1989 to 1,218 fish in 1999-2001, mostly taken in the Columbia River (51 percent) and Washington coastal fisheries (28 percent) (Adams et al. 2002). Overall captures appeared to be dropping, although this could be related to changing fishing regulations. Oregon and Washington have recently prohibited the retention of green sturgeon for commercial and recreational fisheries.

INLAND SPORT HARVEST

Green sturgeon are caught incidentally by sport fisherman targeting white sturgeon (NMFS 2008c). In California, small numbers of green sturgeon are incidentally caught, primarily in San Pablo Bay (Adams et al. 2007). Sportfishing in the Columbia River, Willapa Bay, and Grays Harbor captured from 22 to 553 fish per year between 1985 and 2001. It appears sportfishing captures are declining; however, it is not known if this is a result of abundance, changed fishing regulations, or other factors. In March 2007, the California Fish and Game Commission adopted new regulations that made the landing or possession of green sturgeon illegal. These regulations reduced the slot limit of white sturgeon from 72 inches to 66 inches, and limited the retention of white sturgeon to one fish per day with a total of 3 fish retained per year.

Fishing gear mortality presents an additional risk to the long-lived sturgeon species such as green sturgeon (Boreman 1997). Although sturgeon are relatively hardy and generally survive being hooked, their long life makes them vulnerable to repeated hooking encounters, which may lead to an overall significant hooking mortality rate over their lifetime. Illegal harvest of sturgeon occurs in the Sacramento River and Delta. These operations frequently target white sturgeon, especially for the lucrative caviar market, but green sturgeon may be incidentally taken as well.

Disease and Predation

Insufficient information exists to determine whether disease has played an important role in the decline of the Southern DPS (71 FR 17757) of green sturgeon. There is a possibility of disease transfer from hatchery-raised sturgeon and wild sturgeon; however, there is no evidence that this has ever occurred (COSEWIC 2004).

Predation of juveniles by non-native fish such as striped bass has also been identified as a concern, although NMFS was not able to estimate mortality rates imposed on the Southern DPS of green sturgeon. NMFS maintains that the predation risk imposed by striped bass on the Southern DPS likely exists although its importance is uncertain (71 FR 17757).

Non-native Invasive Species

Non-native species are an ongoing problem in the Sacramento-San Joaquin River and Delta systems through continued introductions and modification of habitat (DFG 2002). The greatest concerns are about shifts in the relative abundance and types of food items (NMFS 2005). Change in the community composition of zooplankton and benthic invertebrates have been postulated as one factor in the overall pelagic organism decline experienced in the Delta since 2000 (Baxter et al. 2008). For example, the native opossum shrimp *Neomysis mercedis* was a common prey item for juveniles in the 1960’s (Radtke 1966); this native mysid has been largely replaced in the Delta by the introduced mysid *Acanthomysis bowmani*. The non-native overbite clam, *Potamocorbula amurensis,* was introduced in 1988 and now dominates the benthic community in Suisun and San Pablo Bays. This clam has become the most common food of white sturgeon (Urquhart et al. 1991) and was found in the only green sturgeon stomach examined so far (in 2001) (DFG 2002 in Adams et al. 2007). One risk involves the replacement of relatively uncontaminated food items with those that may be contaminated (70 FR 17386). The overbite clam is known to bioaccumulate selenium, a toxic metal (Urquhart et al. 1991).

As discussed earlier for salmonids (Section 3.1.2.7.8), predation of juveniles by non-native fish such as striped bass has also been identified as a potential risk, but has not been quantified (71 FR 17757).

Ocean Survival

Green sturgeon spend most of their lives in coastal marine habitat, and therefore could be vulnerable to conditions in the ocean. However, NMFS has not indicated this as a significant potential risk (71 FR 17757).

Environmental Variation and Climate Change

Climate change is expected to result in altered and more variable precipitation and hydrological patterns in California. While population sizes are unknown for the Southern DPS, it is clearly much smaller than the Northern DPS and therefore is much more susceptible to catastrophic events (NMFS 2005). Spawning in the Southern DPS appears to be concentrated in the Sacramento River above the RBDD. Catastrophic events have occurred on the Sacramento River, such as the large-scale Cantara herbicide spill which killed all fish in a 10-mile stretch of the Sacramento River upstream from Shasta Dam, and the 1977–1978 drought that caused year-class failure of winter-run Chinook (NMFS 2005). Changes in ocean conditions, such as the El Nino climatic events, could also affect feeding and survival of green sturgeon, which spend most of their lives in the ocean.

Ecosystem Restoration

Actions to address limiting factors for Southern DPS green sturgeon are proposed or are being carried out by the CBDA, CVPIA, and DFG such as: (1) improving flow conditions in Central Valley rivers and streams; (2) installing additional fish screens and improving fish passage; and (3) implementing stricter fishing regulations. Other restoration efforts that could benefit green sturgeon include Iron Mountain Mine Remediation efforts to improve water quality in the upper Sacramento River and providing fish passage at barriers such as Daguerre Point Dam on the Yuba River or the Fremont Weir in the Yolo Bypass. While these are important contributions, NMFS concluded in 1996 that these efforts alone do not substantially reduce risks to the Southern DPS and that further protections afforded under the ESA were necessary (71 FR 17757).

3.1.3.8 Status of the Species within the Action Area

Adult green sturgeons enter the San Francisco Bay estuary in early winter (January/February) before initiating their upstream spawning migration into the Delta. Adults move through the Delta from February through April, arriving in the upper Sacramento River between April and June (Heublein 2006, Kelley et al. 2007). Following their initial spawning run upriver, adults may hold for a few weeks to months in the upper river or immediately migrate back down river to the Delta.

Adults and sub-adults may also reside for extended periods in the western Delta as well as in Suisun and San Pablo Bays. Sub-adults are believed to reside year round in these estuaries prior to moving offshore as adults. Juveniles are believed to use the Delta for rearing for the first 1 to 3 years of their life before moving out to the ocean. Juveniles are recovered at the SWP and CVP fish collection facilities year round (NMFS 2008b).

3.1.4 Longfin Smelt

3.1.4.1 Listing Status and Designated Critical Habitat

Longfin smelt (*Spirinchus thaleichthys*) is not currently listed under the Federal ESA, but is listed as a threatened species under the CESA. Available scientific information and monitoring data indicate that the abundance of longfin smelt in all major California estuaries where the species has been found historically has declined severely in the past two decades. In response to these declines, the Bay Institute, the Center for Biological Diversity and the Natural Resources Defense Council petitioned the USFWS in August 2007 to list the population of longfin smelt in the San Francisco Bay-Delta Estuary as endangered under the ESA (The Bay Institute [TBI] et al. 2007a). These groups also submitted a formal request to the California Fish and Game Commission to list longfin smelt in California on an emergency basis as an endangered species under CESA (The Bay Institute et al. 2007b). During spring of 2008, the CDFG sought stakeholder input to the process of drafting a Section 2084 regulation to protect longfin smelt. On November 14, 2008 the Fish and Game Commission adopted emergency regulations, to be in effect for 90 days, governing conditions under which Delta water diversions and exports can continue, with limitations depending on longfin smelt distribution and take at water export facilities. On March 4, 2009, the Commission found that listing longfin smelt as threatened under CESA was warranted, and initiated the state regulatory process to establish the listing (DFG 2009).

There is no designated critical habitat because the species is not currently listed under the Federal ESA. However, suitable spawning and rearing habitat for longfin smelt occurs throughout the San Francisco Estuary, including in the Action Area, as described in Section 3.1.4.2.

3.1.4.2 Life History

The species is pelagic and anadromous. Longfin smelt are euryhaline, capable of living in freshwater but spending the majority of their lives in brackish and marine environments. Longfin smelt are one of seven osmerid fish species occupying habitats in California estuaries and coastal waters (Moyle 2002). Presently, the largest and southern-most self-sustaining longfin smelt population on the Pacific Coast occurs in the San Francisco Estuary (Moyle 2002). In the San Francisco Estuary, longfin smelt adults are generally 90-110 mm SL at maturity, but some individuals may grow up to 140 mm SL (Baxter 1999, Moyle 2002).

Longfin smelt predominantly have a 2-year life cycle. Most longfin smelt reach maturity at Age 2, with most individuals dying shortly after spawning. A few smelt, mostly females, live a third year, but it is not certain if they spawn again. Peak spawning occurs between February and April (Reclamation 2008), within a temperature range of 7 to 14.5°C (The Bay Institute et al. 2007a). Longfin smelt eggs are adhesive and are probably released over a firm substrate (Moyle 2002). Just after hatching, longfin smelt larvae move quickly into the upper part of the water column, and are swept downstream into more brackish areas of the estuary (Moyle 2002). Recently hatched longfin smelt larvae are buoyant and occur in the upper portion of the water column usually from January through April. Rearing habitat for longfin smelt is typically open water, away from shorelines and vegetated inshore regions. Most juveniles occur within a salinity range of 15 to 30 ppt (Baxter 1999), and are not commonly found where water temperatures are above 20°C (Moyle 2002).Young juvenile longfin smelt feed primarily on copepods, while older juveniles and adult longfin smelt feed principally on opossum shrimp, *Neomysis mercedis* and the introduced mysid, *Acanthomysis bowmani* when available (Hobbs et al. 2006, DFG 2009).

3.1.4.3 Distribution

Scattered populations occur along the Northeast Pacific coast from Alaska to the San Francisco estuary (Moyle 2002). Longfin smelt populations in California have historically been documented in the San Francisco Estuary, Humboldt Bay, the Eel River Estuary, and the Klamath River Estuary. Currently, the largest spawning population occurs in the San Francisco Estuary, while other California populations appear to be small and possibly not self-sustaining (Reclamation 2008, The Bay Institute et al. 2007a).

Longfin smelt use the entire the San Francisco Estuary, from the freshwater Delta and Suisun Marsh downstream to brackish South San Francisco Bay and in coastal marine waters depending on the time of year and life stage (Table 3-9) (Baxter 1999, Moyle 2002, Rosenfield and Baxter 2007). In wet years they can occur in the Gulf of the Farallones, just outside of the Golden Gate (Moyle 2002). The center of their distribution gradually moves down the estuary in the summer. Adult longfin smelt tend to aggregate in Suisun Bay and the western Delta in late fall, and then spawn in freshwater areas immediately upstream during winter and early spring. Based on data from the FMWT, Winter MWT, and SKT surveys conducted by DFG, only a very small fraction of the sub-adult and adult longfin smelt appear in the southeast Delta in OMRs. Adults and larvae were found furthest upstream in years of lower river flow CDFG 2009). The exact spawning areas are unknown for longfin smelt, but the general spawning region is considered to be between the confluence of the Sacramento and San Joaquin Rivers up to Rio Vista on the Sacramento River and Medford Island on the San Joaquin River (Moyle 2002). Spawning probably also occurs in the eastern portion of Suisun Bay and in the larger sloughs of Suisun Marsh in some years (The Bay Institute et al. 2007a).

Table 3-9 Periodicity Table for Longfin Smelt in the Delta

	LOCATION	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Longfin Smelt	**Adult Immigration/Holding**												
	Delta												
	Spawning/ Larval Development												
	Delta												
	Fry/Juvenile Rearing												
	Delta												
	Juvenile Emmigration												
	Delta												
	Salvage												

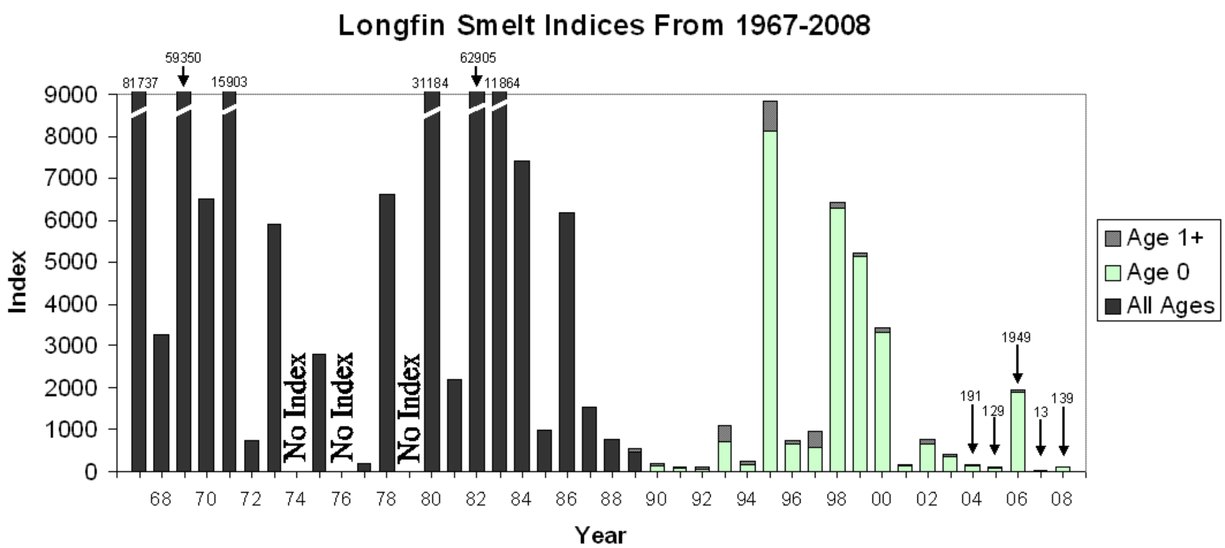
From spawning grounds in the upper estuary, longfin smelt move downstream through a combination of passive transport and migration (Baxter 1999, Dege and Brown 2004, Rosenfield and Baxter 2007). Longfin smelt larvae are swept downstream shortly after hatching (Moyle 2002). When high outflows correspond with the presence of larval longfin smelt, the larvae are transported mainly to Suisun and San Pablo Bays, whereas in years with lower outflows they are transported to the western Delta and Suisun Bay (Moyle 2002). Larvae are frequently caught upstream of the Sacramento-San Joaquin River confluence in the Delta around Sherman Island (Baxter 1999, Dege and Brown 2004). The geographic distribution of larval longfin smelt is closely associated with the location of the estuary's 2-ppt isohaline (X2), with the center of the distribution being seaward of X2 (Reclamation 2008). Juveniles migrate further downstream to Suisun Bay and more brackish habitats for growth and rearing (Moyle 2002). Further details on age-specific distribution are provided in DFG (2009).

3.1.4.4 Abundance

All available scientific information and monitoring data indicate that the abundance of longfin smelt in California has declined severely in the past two decades (The Bay Institute 2007a, DFG 2009). The longfin smelt, one of the species associated with the POD, has experienced a sharp population decline starting in 2000 (Figure 3-17). Population indices have declined further since 2000, with record or near-record-low abundance indices based on FMWT abundance indices for 2002-2006.

Historically, the longfin smelt population in the San Francisco Estuary has shown wide fluctuations in annual Age-class 1 and Age-class 2 abundance, with abundance in any given year depending on the number of spawners and on river outflow during spawning and larval periods during the Age-class natal year (Moyle 2002). Recently, longfin smelt abundance has remained low even in years with relatively moderate hydrology, which typically supports at least modest fish production (Moyle 2002, Baxter et al. 2008).

Persistent low levels of longfin smelt abundance may have, at least in part, been the result of the introduction and establishment of the overbite clam *Corbula amurensis* in the estuary. The 1987 invasion of this filter-feeding clam has diminished the availability of the longfin smelt's primary food resources (i.e., copepods) through heavy grazing on phytoplankton (Rosenfield and Baxter 2007). Although dramatic declines were observed after introduction of *Corbula*, there was little change in the slope of the relationship between freshwater outflow and abundance. Some aspects of the longfin smelt decline may not be explained by changes in food availability caused by the introduction of the overbite clam or outflow conditions. Notably, since 2000 longfin smelt abundance levels have been consistently lower than the post-1987 outflow-abundance relationship would predict (Figure 3-17). Also, catches of pre-spawning adult longfin smelt in Suisun Marsh dropped consistently after inception of the Suisun Marsh Survey, and this decline predates the onset of the 1987-1994 drought and the introduction of *Corbula*. It is possible that recent environmental changes have altered the Marsh's carrying capacity for longfin smelt (Rosenfield and Baxter 2007). The recent decline in longfin smelt abundance corresponds with that observed for other pelagic fishes in the Delta, and is as dramatic, if not more so, than that observed for other species (Moyle 2002).



Source: DFG

A) Fall Midwater Trawl Survey September-December, 1967-2005
B) Bay Study Survey otter trawl, May-October 1980-2005
C) Bay Study Survey midwater trawl, May-October 1980-2005.

Figure 3-17 Longfin Smelt Annual Relative Abundance

3.1.4.5 Population Viability Summary

Abundance

Longfin smelt populations have declined throughout California. The population found in the San Francisco Estuary is the largest longfin smelt population in California, and has experienced substantial declines over the past 20 years, with persistent record low abundance levels since 2002 (The Bay Institute 2007a). Small populations have also been historically documented in California in Humboldt Bay, the Eel River Estuary, and the Klamath River Estuary, but fish have not been found in these areas in recent years. In fact, the Humboldt Bay and Klamath River populations may now be extirpated (The Bay Institute 2007a). Outside of California, longfin smelt are found in bays from Coos Bay, Oregon to Prince William Sound, Alaska (Moyle 2002). Relatively large populations still appear to exist in portions of the northernmost area of its range (i.e., Oregon, Washington and Alaska (ADFG 2006).

As summarized in the recent status review (DFG 2009), longfin smelt abundance within the San Francisco Estuary is influenced by outflow during the egg and larva periods (Sommer et al. 2007).

Productivity

Historically, longfin smelt year-class strength has been positively correlated with freshwater outflow in the San Francisco Estuary (The Bay Institute 2007a). In recent years, the magnitude of this response has declined for both Age-class 1 and Age-class 2 smelt, but the slope of the relationships has remained similar (Reclamation 2008, Rosenfield and Baxter 2007). The decline in Age-class 2 smelt has been greater than that for Age-class 1, suggesting a reduction in survival between age-classes (Rosenfield and Baxter 2007). Data from the Suisun Marsh survey indicates that the abundance of spawning age adult longfin smelt has declined since 1990 (Rosenfield and Baxter 2007). Dramatic declines have also been observed during this period for larval and juvenile longfin smelt (Reclamation 2008). These declines in productivity are likely associated with multiple factors, including reduced freshwater outflow and declining food availability caused by introduction of the overbite clam.

SPATIAL STRUCTURE

Longfin smelt live in relatively small, reproductively isolated populations, which cannot be supplemented by immigration from adjacent populations. This aspect of their natural history, combined with current levels of anthropogenic disturbance to their spawning and rearing habitats in California, make longfin smelt particularly vulnerable to extirpation. Evaluation of physical characteristics, genetic data, and ecological attributes of the population of longfin smelt in the San Francisco Estuary longfin indicates that this population is reproductively isolated from the other populations located in California and those located further north (The Bay Institute 2007a).

Within the San Francisco Estuary the recent declines in abundance do not appear to be attributable to a constriction in overall distribution (Rosenfield and Baxter 2007). Unlike delta smelt, however, longfin smelt are found throughout San Pablo, San Francisco and South San Francisco bays as well as Suisun Bay and the western Delta.

Diversity

The longfin smelt population living in the San Francisco Estuary was once considered to be a separate species from those found further north. In 1963, it was discovered that meristic differences observed between the San Francisco Estuary population and those further north occurred along a north-south gradient, and the populations were merged into a single species, *Spirinchus thaleichthys* (The Bay Institute 2007a). In 1995 genetic data confirmed that the San Francisco Estuary population is the same species as that from Lake Washington in Washington State. It was also found, however, that the gene pool of the San Francisco Estuary population is significantly different and isolated from the Washington population, warranting protection as an isolated, genetically-distinct entity (Stanley 1995). Moyle et al. (1995) has also recommended that longfin smelt in the San Francisco Estuary be recognized and protected as an ESU, due to its apparent reproductive isolation from other populations, and because it represents an important component of the evolutionary history of the species.

3.1.4.6 Critical Habitat and Primary Constituent Elements

Critical habitat has not been proposed or defined for the longfin smelt because it is not currently a listed species under the ESA.

3.1.4.7 Factors Affecting Longfin Smelt

The major factors believed to be responsible for recent declines in the abundance of the longfin smelt, and other pelagic species in the San Francisco Estuary, include the direct and indirect effects of Delta water operations, food web alteration by invasive species, and poor water quality (Reclamation 2008). Other factors that make longfin smelt vulnerable include cumulative and possibly synergistic effects of its low abundance, distance between local populations, reduced reproductive potential, and reduced carrying capacity of its habitat (DFG 2009). The POD studies (Baxter et al. 2008) used a conceptual model to categorize the many factors affecting the abundance of four pelagic fishes including longfin smelt: (1) previous abundance; (2) habitat (spawning and open water); (3) top-down factors such as entrainment in diversions, predation, fishery bycatch, and collections; and (4) bottom-up factors such as food availability and impacts from non-native overbite clam (reviewed in DFG 2009).

Fish Movement & Habitat Blockage

Migration barriers do not significantly affect longfin smelt in the San Francisco Estuary, as all life stages occur downstream of major dams.

Water Development and Conveyance (Hydrodynamics and Entrainment)

Longfin smelt adults and larvae are vulnerable to entrainment by water diversions, such as SWP and CVP export facilities, power plants, and agricultural diversions. The risk of entrainment by the SWP and CVP facilities is greatest in winter, when adults migrate upstream to freshwater spawning areas, particularly in dry years when adult distribution shifts further upstream to the southeast Delta (Sommer et al. 2007, DFG 2009). By mid-summer, entrainment is no longer a major stressor because most of the population is downstream of the zone affected by exports (Baxter et al. 2008). The magnitude of the impact of entrainment on the Bay-Delta longfin smelt population is not known at this time. Also unknown is the impact of entrainment relative to other stressors. In their 2009 species status review DFG (2009) estimated that from 1993 through 2008 a total of approximately 1.6 million juvenile and 12,000 adult longfin smelt were entrained at the CVP and SWP intakes in the southern Delta. Although loss (mortality) rates for entrained longfin smelt have not been studied directly, based on studies of other species DFG estimated that more than 95 percent and 80 percent of longfin smelt entrained at the SWP and CVP, respectively, were lost. These losses probably represent a small fraction of the total population, especially in wet years when the population has a more downstream distribution.

The indirect effects of water exports occur in the form of changed hydrodynamics in the Delta. Because longfin smelt are particularly sensitive to physio-chemical water quality characteristics (e.g., salinity, prey availability), their abundance is closely associated with spring outflow conditions. The FMWT index for longfin smelt typically increases in years when outflows are high and X2 is pushed seaward, indicating that the extent and quality of longfin smelt habitat increases when freshwater flows are high (Reclamation 2008).

Flood Control and Levee Construction

There is no evidence that levees and other flood control infrastructure directly impact longfin smelt in the San Francisco Estuary. This is not unexpected given that longfin smelt are largely a pelagic species. The construction, maintenance, or failure of levees may have indirect effects on longfin smelt by influencing delta hydrodynamics.

Land Use Activities

Intensive agricultural and urban development in the Delta affects longfin smelt through the impairment of water quality and reductions in freshwater river flow due to water diversions (see “Water Development and Conveyance” and “Water Quality” sections).

Water Quality

The quality and quantity of spawning and rearing habitat available to fish living in the San Francisco Estuary has declined dramatically due to increased water temperatures, turbidity, and contaminant loads, and decreased DO. Although longfin smelt are well-adapted to living in turbid areas, they are likely vulnerable to changes in temperature, low DO levels, and exposure to contaminants from urban, agricultural and industrial sources. As described earlier for delta smelt (Section 3.1.1.7.4), contaminants, eutrophication, and algal blooms can alter ecosystem functions and productivity, although the magnitude and effects within the Delta are poorly understood (USFWS 2008). Pollutants from agricultural and urban sources may harm delta smelt directly; reduce zooplankton abundance, or both. Recent testing has noted invertebrate toxicity in the waters of the northern Delta and western Suisun Bay. The POD studies have focused on three factors: pyrethroid pesticides, the blue-green alga *Microcystis*, and ammonia (Baxter et al. 2008, Sommer 2007). Limited data exists, however, regarding the population level impacts and relative importance of poor water quality for longfin smelt.

POD investigators have initiated several recent studies to determine the role of contaminants in the observed declines of Delta fish species. Fish bioassays indicated that larval delta smelt (*Hypomesus transpacificus*) are highly sensitive to ammonia, low turbidity, and low salinity (Reclamation 2008). Due to the similarity in life history strategy and habitat used by delta smelt and longfin smelt, longfin smelt may be sensitive to similar water quality characteristics.

Hatchery Operations

There are currently no captive breeding programs for longfin smelt. Thus, hatchery operations are not believed to pose a major threat to longfin smelt in the San Francisco Estuary.

Over Utilization (Commercial and Sport)

Longfin smelt are a small component of the “whitebait” fishery in the South San Francisco Bay but they have no sport fishery value. Adults sometimes occur as bycatch in commercial trawling for bait shrimp in the brackish parts of the lower San Francisco Estuary (e.g., San Pablo Bay) (DFG 2009). Commercial fishers are required to return most trawl-caught fish to the water. CDFG (Hieb 2009) estimated 15,539 (adult) longfin smelt were caught as bycatch in 1989-90. The most significant utilization of longfin smelt is scientific collecting by the IEP through several monitoring programs. The IEP studies during 1987 to 2008 annually collected from 461 to 85,742 adults, and 343 to 72,824 larval longfin smelt. Current levels of harvest are not believed to be a major factor in the declines in abundance of longfin smelt in the San Francisco Estuary.

Disease and Predation

Recent POD investigations have not revealed any histopathological abnormalities associated with disease in longfin smelt. These studies also found no evidence of viral infections or high parasite loads (Reclamation 2008). Limited information exists regarding the impact of predation on longfin smelt populations. The introduction of striped bass is not believed to have contributed to declines in the abundance of longfin smelt (Moyle 2002). The introduction of inland silversides, however, may have played a role in these declines. This conclusion is based on the following: (1) the invasion of the estuary by inland silversides coincided with declines of longfin smelt, (2) inland silversides concentrate in shallow waters where smelt spawn, and (3) inland silversides are known to be effective predators on larval fishes (Moyle 2002).

Non-native Invasive Species

Multiple introduced species affect longfin smelt both directly and indirectly through predation, food web alteration, and effects on physical habitat, as discussed earlier for other fish species (Section 3.1.1.7.8). In particular, the invasion of overbite clam *C. amurensis* in the 1980's has been implicated in the decline of primary productivity and zooplankton biomass in the western delta, possibly limiting food availability for pelagic species such as longfin smelt (Baxter et al. 2008).

Furthermore the composition of the zooplankton community has shifted, such that it is mostly composed of introduced species, thereby having potentially significant effects on food availability for longfin smelt. For example, the invasive cyclopoid copepod *Limnoithona tetrospina* likely competes with native copepod species for food resources, and is now the most abundant copepod in the low-salinity zone of the Estuary. It is believed that *Limnoithona* is an inferior prey item for longfin smelt, and that its high abundance could result in reduced energy reserves and overall condition of pelagic fish species in the Delta (Reclamation 2008). Consistent with the hypothesis of food limitation, Rosenfield and Baxter (2007) have documented reduced age-class 1 productivity and a disproportionate reduction in age-class 2 recruitment. Moreover, poor growth and condition of longfin smelt has been documented in certain regions of Suisun Bay (Hobbs et al. 2006).

Ocean Survival

Little is known about the extent and effects of the marine migration of San Francisco estuary longfin smelt. Because most longfin smelt apparently complete their life cycle primarily within the Estuary, ocean survival is unlikely to be a critical factor in their population decline.

Environmental Variation and Climate Change

Climate change has the potential to exacerbate existing threats by significantly impacting delta hydrodynamics and habitat quality for longfin smelt in future decades (DFG 2009). This is due to changed precipitation patterns, increased flood frequency and water temperatures, and sea level rise. The increased likelihood of winter floods may alter flows from historical conditions under which Delta fish species have evolved, thereby interfering with reproduction (Reclamation 2008). An increased frequency of flooding could also dislodge eggs and sweep adults, eggs, and larvae far downstream to unsuitable rearing habitat (Moyle 2002). Sea level rise will likely increase seawater intrusion, altering the position of X2, an important predictor of longfin smelt abundance (Reclamation 2008). Finally, increased water temperature caused by warming could reduce the availability of suitable spawning habitat in upstream reaches of the Delta. Increasing water temperature is a particular concern for the San Francisco Estuary population, because the estuary is at the southernmost end of the species range.

Ecosystem Restoration

CALIFORNIA BAY-DELTA AUTHORITY

Two programs included under CBDA, the ERP and the EWA, were created to improve conditions for fish, including longfin smelt, in the Central Valley (CALFED 2000). Installation of fish screens is one of the key components of the ERP, and should reduce entrainment of longfin smelt in diversion pumps in areas of the Delta where longfin smelt are found. Achievement of other goals of the ERP, such as reducing the negative impacts of invasive species and improving water quality (CALFED 2000), would also benefit longfin smelt in the San Francisco Estuary by reducing competitors or improving food web dynamics and the copepods that are a key food resource for longfin smelt. Habitat restoration initiatives sponsored and funded primarily by the

CBDA-ERP Program have resulted in plans to restore ecological function to 9,543 acres of shallow-water tidal and marsh habitats within the Delta. Restoration of these areas primarily involves flooding lands previously used for agriculture, thereby creating additional shallow water spawning and rearing habitat for longfin smelt.

The EWA is designed to provide water at critical times to meet ESA requirements and incidental take limits without water supply impacts to other users, particularly South Delta water users. In early 2001, the EWA released 290 thousand acre feet of water from San Luis Reservoir at key times to offset reductions in South Delta pumping implemented to protect winter-run Chinook salmon, delta smelt, and splittail. This action may have had positive implications for longfin smelt by reducing entrainment and increasing freshwater outflow. Recent reviews, however, provide no indication that the EWA has been effective in reducing entrainment loss of listed species at the SWP and CVP diversion facilities (The Bay Institute 2007a). The CALFED BDPAC (2007) concluded that the EWA has not been successful at reversing the decline of important Delta species. Currently, the EWA program is authorized through 2010 and is scheduled to be reduced in its scope. Future EWA operations will be considered to have limited assets and will primarily be used only during CVP and SWP pumping reductions in April and May as a result of the VAMP experiments.

The ERP's Environmental Water Program (EWP) does not benefit longfin smelt because it is designed to enhance instream flows in reaches of priority streams controlled by dams, outside the range of longfin smelt.

CENTRAL VALLEY PROJECT IMPROVEMENT ACT

The CVPIA, implemented in 1992, requires that fish and wildlife get equal consideration with other demands for water allocations derived from the CVP. Small, short-duration water export reductions at the CVP export facility (usually timed to protect migrating juvenile salmonids) associated with the CVPIA provide some benefit to longfin smelt. However, most of the habitat restoration and protective actions specified by the program (e.g., gravel restoration, stream flow enhancement, installation of fish screens and ladders) have been implemented outside the geographic range of longfin smelt and therefore do not benefit or protect this species (The Bay Institute 2007a).

SWP DELTA PUMPING PLANT FISH PROTECTION AGREEMENT (FOUR-PUMPS AGREEMENT)

The Four Pumps Agreement Program (DWR and DFG 1986) has approved $59 million for over 40 fish mitigation projects, and by December 2007 had expended $44 million for a variety of projects in the Sacramento and San Joaquin river basins and in the Bay-Delta area, such as salmon habitat enhancement projects, water exchange projects for salmon passage flows, fish screens and ladders, guidance barriers, enhanced law enforcement, and stocking of salmon, steelhead and striped bass (DWR 2008). Most projects have focused on salmon and steelhead, particularly spring-run Chinook, and were implemented outside the range of longfin smelt. One component of the Four Pumps projects that could benefit longfin smelt is the screening of diversions in Suisun Marsh (DWR and DFG 1986).

3.1.4.8 Status of Species in the Action Area

Survey data indicates that the population of longfin smelt in the San Francisco Estuary has declined substantially since the 1980s (DFG 2009). Longfin smelt occur in the Action Area in winter as spawning adults and in winter and spring as larvae moving to downstream rearing habitat.

3.2 TERRESTRIAL SPECIES

A list of sensitive species known from the region was developed through a search of the California Natural Diversity Database (CNDDB) and the USFWS-generated list of Federal Endangered and Threatened Species that Occur in the Woodward Island, Bouldin Island, Jersey Island, and Brentwood 7.5-minute quadrangles, which cover the Project sites and vicinity. Based on these database searches, species with the potential to occur in the Project area based on evaluation of site conditions include: conservancy fairy shrimp (*Branchinecta conservatio*), vernal pool fairy shrimp (*Branchinecta lynchi*), vernal pool tadpole shrimp (*Lepidurus packardi*), giant garter snake (*Thamnophis gigas*), western pond turtle (*Actinemys marmorata*), northwestern pond turtle (*Actinemys marmorata marmorata*), Swainson's hawk (*Buteo swainsoni*), tricolored blackbird (*Ageliaus tricolor*), black rail (*Laterallus jamaicensis coturniculus*), western burrowing owl (*Athene cunicularia*), and loggerhead shrike (*Lanius ludovicianus*). Their status is discussed below.

Other special-status species, including valley elderberry longhorn beetle (*Desmocerus californicus dimorphus*), California red-legged frog (*Rana aurora draytonii*), Alameda whipsnake (*Masticophis lateralis euryxanthus*), California tiger salamander (*Ambystoma californiense*), silvery legless lizard (*Anniella pulchra pulchra*), San Joaquin kit fox (*Vulpes macrotis mutica*) and Antioch Dunes evening-primrose *(Oenothera deltoides* ssp. *howellii*) were eliminated from further consideration due to the absence of suitable habitat, isolation from occupied habitat or other factors.

The Project sites, access roads and 100-foot buffer areas were surveyed for the presence of elderberry shrubs (*Sambucus* spp.), which serve as the host plant for valley elderberry longhorn beetle. No elderberries were detected during these surveys, leading to the conclusion that valley elderberry longhorn beetle is absent from the Project area.

California red-legged frog, Alameda whipsnake, California tiger salamander, and silvery legless lizard are not expected to occur in the Project site or vicinity due to the absence of suitable habitat (Alameda whipsnake), isolation from occupied habitat in the region and historic site conditions that were unsuitable (California tiger salamander, silvery legless lizard), or their extirpation from this portion of the Delta due to the mass colonization of introduced fishes and bullfrogs (California red-legged frog).

San Joaquin kit fox is not expected to occur in the Project site due to the lack of connectivity between known kit fox occurrences and the Project sites, with the rivers and sloughs creating barriers to movement. Dune habitat suitable for Antioch Dunes evening –primrose is absent from the project site.

3.2.1 Giant Garter Snake

3.2.1.1 Listing Status and Designated Critical Habitat

On October 20, 1993, the giant garter snake (*Thamnophis gigas*, GGS) was listed as threatened by the USFWS due to habitat loss from urbanization, flooding, and agricultural activities, as well as contaminants and introduced predators (58 FR 54053). Previous to that ruling, it was listed as threatened by the California Fish and Game Commission. No critical habitat has been designated for GGS.

3.2.1.2 Life History

The GGS is a large (37 to 65 inches total length) aquatic snake that is never found far from water. The dorsal coloration is highly variable—brown to olive with a cream, yellow, or orange dorsal stripe and two light-colored lateral stripes (USFWS 1999 and 2005a). Some individuals have a checkered pattern of black spots between the dorsal and lateral stripes or completely lack any dorsal stripes at all.

The GGS inhabits both agricultural wetlands and natural waterways including irrigation canals, drainage ditches, rice lands, marshes, sloughs, ponds, small lakes, low gradient streams, and riparian corridors (USFWS 1999). They are mostly absent from larger rivers and wetlands with sandy or rocky substrates (USFWS 1999). This species is closely tied to water and seems to require freshwater aquatic habitat during the spring and summer months, and estivation habitat (small mammal burrows or rock piles) in the dry uplands during the fall and winter months (Brode 1988 in USFWS 1999). Juvenile and adult GGS appear to be most active when air temperatures reach 90°F; however, they can be observed during any month of the season when the sun is out and air temperatures are over 70°F (Hansen and Brode 1980 and Brode 1988 in USFWS 1999).

The species is relatively inactive during the winter, typically over wintering in burrows and crevices near active season foraging habitat. Individuals have been noted using burrows as far as 164 feet from marsh edges during the active season, and retreating as far as 820 feet from the edge of wetland habitats while over wintering, presumably to reach hibernacula above the annual high water mark (USFWS 1999). After emerging from over wintering sites, adult GGS breed during the spring (March to May) and 10 – 46 young (average 8.1 inches total length) are born alive during the months of late July through early September (Hansen and Hansen 1990 in USFWS 1999). Giant garter snakes feed on a wide variety of fishes and amphibians, including both native and introduced fishes and Pacific tree frogs (*Pseudacris regilla*) and introduced bullfrogs (*Rana catesbeiana*). They seem to take prey items that are most abundant. Young snakes grow rapidly and reach maturity within about 3-5 years (USFWS 1999).

GGS are typically found in fresh water marshes and wetland areas. They can also be found in modified habitats like agricultural canals and ditches often associated with rice farming and flooding. The process of rice farming fairly closely coincides with the biological needs of the GGS. During the summer, GGS use flooded rice fields as long as sufficient prey is present. During the late summer, rice fields provide important nursery areas for newborn GGS. In the later summer and fall as the rice fields are drained, prey items become concentrated in remaining water bodies and GGS often gorge themselves on this food supply before going into hibernation (USFWS 1999).

3.2.1.3 Distribution and Abundance

The GGS is endemic to California’s Central Valley, the lowland area between the Sierra Nevada and Coast Ranges (Hansen and Brode 1980 in USFWS 1999). Historically, GGS were widespread throughout the lowlands of the Central Valley (except for a midway historic gap) from the vicinity of Chico in Butte County south to Buena Vista Lake in Kern County (Stebbins 2003). Today, the species has disappeared from approximately 98 percent of its historic range and is largely confined to the rice growing regions of the Sacramento Valley and managed wetlands of Merced County in the San Joaquin Valley (USFWS 1999). There are 13 separate populations of GGS in 11 counties including Butte, Colusa, Glenn, Fresno, Merced, Sacramento, San Joaquin, Solano, Stanislaus, Sutter and Yolo (USFWS 1999). The population was reported as not declining further in the five-year review for GGS (USFWS 2006).

3.2.1.4 Critical Habitat and Primary Constituent Elements

The GGS has four main habitat requirements as outlined by the draft recovery plan: (1) adequate water during active season to support prey species such as blackfish *(Orthodox microlepidotus*), Pacific tree frog, carp (*Cyprinus carpio*), mosquito fish (*Gambusia affinis*) and bullfrogs; (2) emergent wetland vegetation (i.e., cattails *Typha spp.* and bulrushes *Scirpus spp.*) for foraging habitat and cover from predators; (3) upland habitat with grassy banks and openings in vegetation for basking; and (4) higher elevation upland habitats for cover and refuge (i.e., burrows and crevices) from flood waters during winter (USFWS 1999).

The GGS is active from early spring (April – May) through mid-fall (October – November), although patterns vary with weather (Brode 1988 in USFWS 1999). During the winter season they are inactive and rarely emerge from wintering burrows. When active they usually remain near wetland habitat, although they can move up to 0.8 km in a day (USFWS 1999). The GGS breeds primarily in March – May, although some mating takes place in September. They are viviparous and the young are born late July to early September.

Litter size ranges from 10 – 46, with an average of 23. Males reach sexual maturity at three years and females at five years of age (USFWS 1999).

3.2.1.5 Factors Affecting Giant Garter Snake

The destruction of floodplain habitats and areas of cattail and bulrush-dominated habitats for agricultural conversion, flood control activities, and land development have greatly reduced the population size for this species (USFWS 1999). Other factors for decline include interrupted or intermittent water flows within floodplain areas, poor water quality, and contaminants such as selenium and pesticides (USFWS 1999), and predation by introduced species such as large mouth bass and bullfrogs (USGS 2004).

3.2.1.6 Status of Species within the Action Area

The GGS is listed as a threatened species at the state and federal level. Recovery priorities, objectives and criteria, and further conservation efforts have been outlined in a draft recovery plan by USFWS (USFWS 1999). Some threats to GGS populations include habitat loss and adverse habitat alteration. They may also be negatively affected by selenium pollution, livestock grazing, hunting, introduction of predatory fish and bullfrogs, and victim to road kills and parasites (USFWS 1999 and 2005a).

The Project site is located within the historic and current range for GGS (USFWS 1999). The nearest recent observations of GGS recorded in the California Natural Diversity Database (CNDDB) (DFG 2008) are a 2002 record of an adult snake captured on the levee on the southwest corner of Webb Tract approximately five miles northwest of the Project area, and a 1996 record of a shed skin recovered from the southwest edge of Medford Island, approximately 1.5 miles northeast of the Project area (Figure 3-18). Two other CNDDB observations of GGS individuals both located approximately 8.5 miles from the Project area include a 1998 observation of an adult snake on a levee south of Brannan State Recreation Area, and another in the San Joaquin River at the north end of the Antioch Bridge. Multiple GGS observations were documented during the 1970s and 1980s from the area near Coldani Marsh, located 0.8 mile west of the intersection of Thornton Road and State Highway 12 approximately nine miles from the Project area. These include three GGS sightings at Coldani Marsh proper, one at nearby White Slough, and one on Shin Kee Tract, 1.5 miles south of State Highway 12.

Trapping surveys for GGS have been conducted in the general vicinity of the Project area. After a GGS was found on Webb Tract in 2002, DWR completed two years of trapping in an attempt to find additional snakes (Patterson and Hansen 2003, Patterson 2004). No GGS were encountered during the trapping surveys. Swaim Biological, Inc. (SBI) conducted a total of six surveys for GGS over three years: 2003-2005 in eastern Contra Costa County (SBI 2004, 2005a-d, 2006), west of the Project site. No GGS were seen or captured during the trapping or visual surveys. The area contained suitable habitat, but SBI biologists noted a relatively low prey base and unsuitable adjacent land use. Upland areas were primarily used for grazing, recreation, and urban development.

Although the distance between the nearest documented localities and the Project site are within dispersal distances for GGS, movements from these localities to the Project site are unlikely. GGS are relatively vagile, but they do not prefer large waterways such as those connecting the localities to the Project site. They have been known to move up to eight kilometers (5 miles) within a few days search of appropriate habitat (Wylie et al. 1997), however this was a response to the dewatering of their habitat. It is unlikely that GGS would actively disperse to this area as long-distance movements would require travel along the main waterways of the delta. It is possible that the Old River and other large waterways in the Delta may facilitate long distance movements by sweeping individuals in currents to new locations.

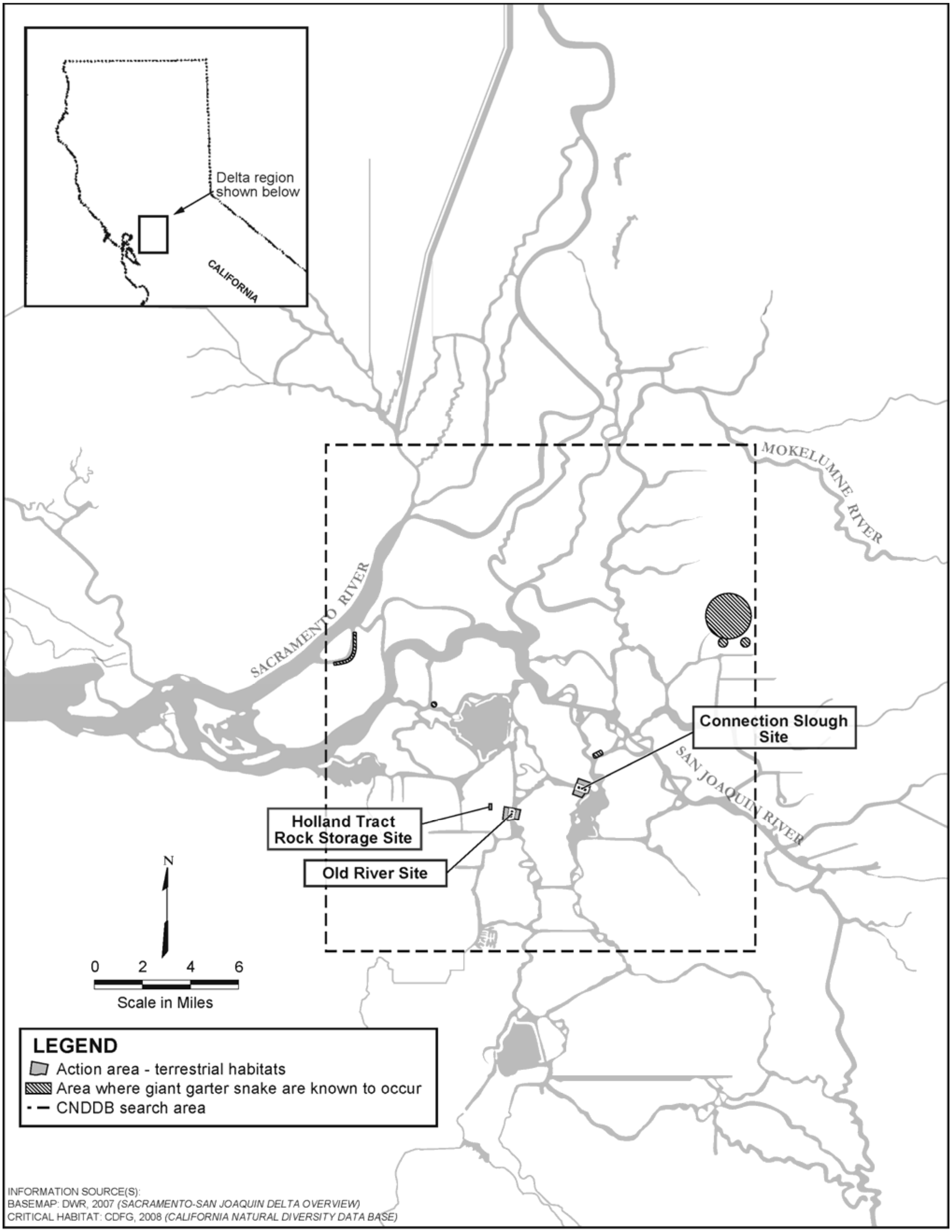


Figure 3-18 California Natural Diversity Database records of GGS in the Project Vicinity

Given the proximity of the Project to known sightings and suitable habitat at both the Old River and Connection Slough sites, GGS presence must be assumed in the Project area, although they are not likely to be present. Multiple trapping surveys resulting in negative findings and relatively few CNNDB occurrences in the area suggest that there is a low potential for GGS to be found in the vicinity. However, given the assumption by the USFWS that the Bay-Delta system is occupied by GGS and the availability of suitable habitat in the area (canals adjacent to the Project site, excluding main waterways), no mechanism currently exists for demonstrating non-occupancy by the species at the Project site.

A habitat assessment by Swaim Biological concluded that the Project sites are located within the historic and current range of giant garter snake (GGS), and that suitable habitat for the GGS exists within the study areas for the Project (Appendix J).

Habitat quality for the GGS is generally good at all sites within the Project area. The main waterways, including the Old River, are likely not highly preferred habitat, but may provide corridors for movement. These contain the basic features necessary for GGS, including emergent vegetation and cover. The banks of the Old River are lined with rip-rap with interstitial spaces that provide cover from predators and that also may aid in thermoregulation. Much of the Old River is also lined by cattails and bulrush. Both plants provide cover and are positively associated with GGS presence. The results of the habitat features associated with each site are summarized in Table 3-10 and discussed in greater detail below.

The west bank of the Old River is adjacent to high-quality GGS habitat. A small canal that runs parallel to the levee road may provide foraging habitat though the deep banks and quantity of emergent vegetation creates a fair amount of shade that may inhibit thermoregulation. The larger, diked canal perpendicular to the levee road provides better foraging habitat for GGS. The banks are moderately sloped with abundant emergent vegetation for cover, and with adequate exposure for thermoregulation. The canal itself appears to have slow-flowing water, and a silt substrate, features positively associated with GGS. Small schools of catfish (*Ictalurus* spp.) are present in the canal. These are generally regarded as predatory game fish, but young catfish may also be a prey source for GGS (USFWS 1999). The levee provides upland habitat and winter refugia above the high water mark. California ground squirrels are absent, but other rodents such as California meadow voles (*Microtus californicus*) are likely present and provide burrows that may be used as retreats.

The west bank of the Old River site has suitable habitat and there are seasonal wetlands that provide potential forage and cover habitat during the GGS active season that are just to the west across the dirt road. The wetlands directly fringing the riverbank comprise the best GGS habitat on the east of the Old River.

On Bacon Island, the study area is adjacent to an irrigation ditch with shallow water flowing over silt. Abundant bullfrogs and mosquitofish, both prey species for GGS, were observed in the ditch. The presence of bullfrogs suggests that the channel provides water year-round since bullfrog tadpoles do not metamorphose until their second season, overwintering in their larval form. Other crucial habitat features such as emergent vegetation and upland habitat were present at the site. California ground squirrels whose burrows provide ideal hibernacula for GGS also were observed. A seasonal wetland south of the proposed gate may provide additional foraging areas in the spring.

Table 3-10 Summary of GGS habitat features present at each site

Site Location	Water Availability	Prey Species	Emergent Vegetation	Basking sites	Upland Refugia and Burrows
Old River Gate Site	Year-round	Fish present	Present	Present	Present
Connection Slough Gate Site, Bacon Island	Year-round	Fish present Bullfrogs present	Present	Present	Present
Holland Tract Storage Site	Seasonal	Fish present	Present but sparse due to grazing	Present	Present

3.2.2 Vernal Pool Fairy Shrimp

3.2.2.1 Listing Status and Designated Critical Habitat

Vernal pool fairy shrimp (*Branchinecta lynchi*, VPFS) was listed as federally threatened on September 19, 1994 (59 FR 48153). The Final Recovery Plan for Vernal Pool Ecosystems was released December 15, 2005 (USFWS 2005b). In 2007, the USFWS published a 5-year status review recommending that the species remain listed as endangered (USFWS 2007a).

Critical habitat was designated for several vernal pools species on August 6, 2003 (FS 68:46683) and revised August 11, 2005 (FR 70:46923). These include VPFS, vernal pool tadpole shrimp (VPTS), and Conservancy fairy shrimp (CFS). For the listed shrimps treated here, there are five critical habitat units within 30 miles of the Action Area, but no critical habitat within the Action Area. There are four VPFS Critical Habitat Units: two locations in Contra Costa County, approximately 9 miles to the southwest; one in San Joaquin County, 30 miles to the east; and another 24 miles to the northwest in Solano County. For CFS as well as VPTS, there is a critical habitat unit 24 miles to the northwest. Additionally, there is a critical habitat unit for VPTS located 33 miles to the northeast in Sacramento County (Figure 3-19).

3.2.2.2 Life History

VPFS is a small crustacean in the class *Branchiopoda* and order *Anostraca*. It ranges from 0.75-1 inch in length, and is distinguished from other vernal pool crustaceans by the female's tapered, pear-shaped brood pouch, and the male's antennae size and shape.

VPFS are present in seasonally inundated basins from December to early May, and can survive in water temperatures below 75°F. They are filter and suspension feeders, with a diet consisting of algae, bacteria, and ciliates. They may also scrape detritus from substrates within the vernal pool habitat. (USFWS 2007a). Eggs are laid by adult females every winter, and the cysts then withstand desiccation and extreme temperatures when pools dry. Cysts also survive when ingested by animals. Cysts will hatch when pools refill and the right temperature ranges are present (Gallagher 1996).

3.2.2.3 Distribution and Abundance

The historical distribution of VPFS is not known, but distribution of VPFS has been assumed to be the historical extent of vernal pool habitat in California throughout the Central Valley and southern coastal regions, numbering in the millions of acres (USFWS 2005b).

VPFS are found in vernal pool habitats throughout the Central Valley and in the Coast Ranges. There are multiple populations of VPFS in 28 counties, including Shasta, Tehama, Butte, Glenn, Yuba, Yolo, Placer, Sacramento, Solano, San Joaquin, Modesto, Napa, Contra Costa, Merced, Madera, Fresno, San Benito, Tulare, Kings, Monterey, San Louis Obispo, Santa Barbara, Ventura, and Riverside (USFWS 2005b). Although they are reported in this wide distribution, they are not abundant in any of these locations (Eng et al. 1990, USFWS 2007a). VPFS have been detected in vernal pool habitats in numerous locations, in the region surrounding the Project area (Figure 3-20).

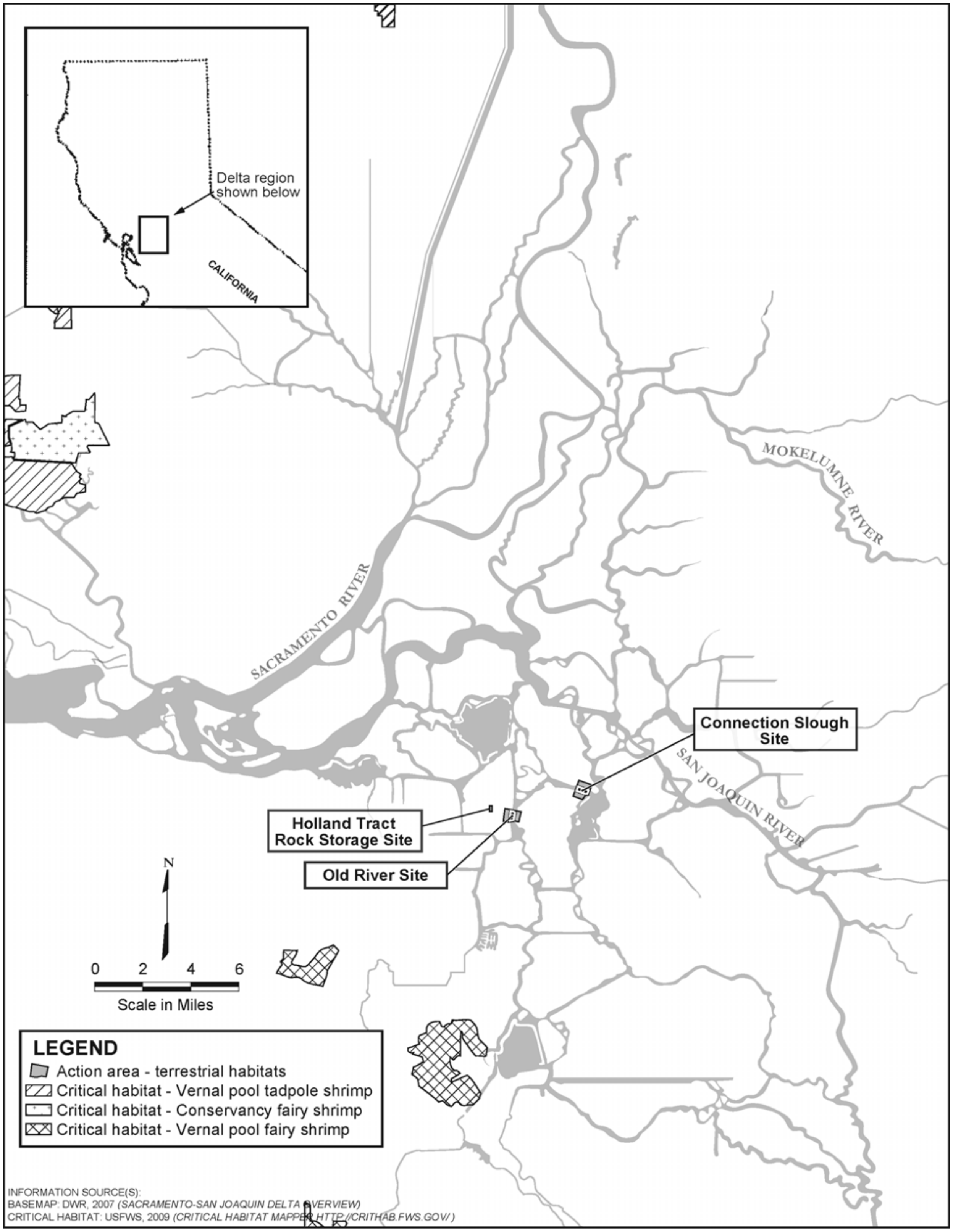


Figure 3-19 Critical Habitat of Vernal Pool Invertebrates Near the Action Area

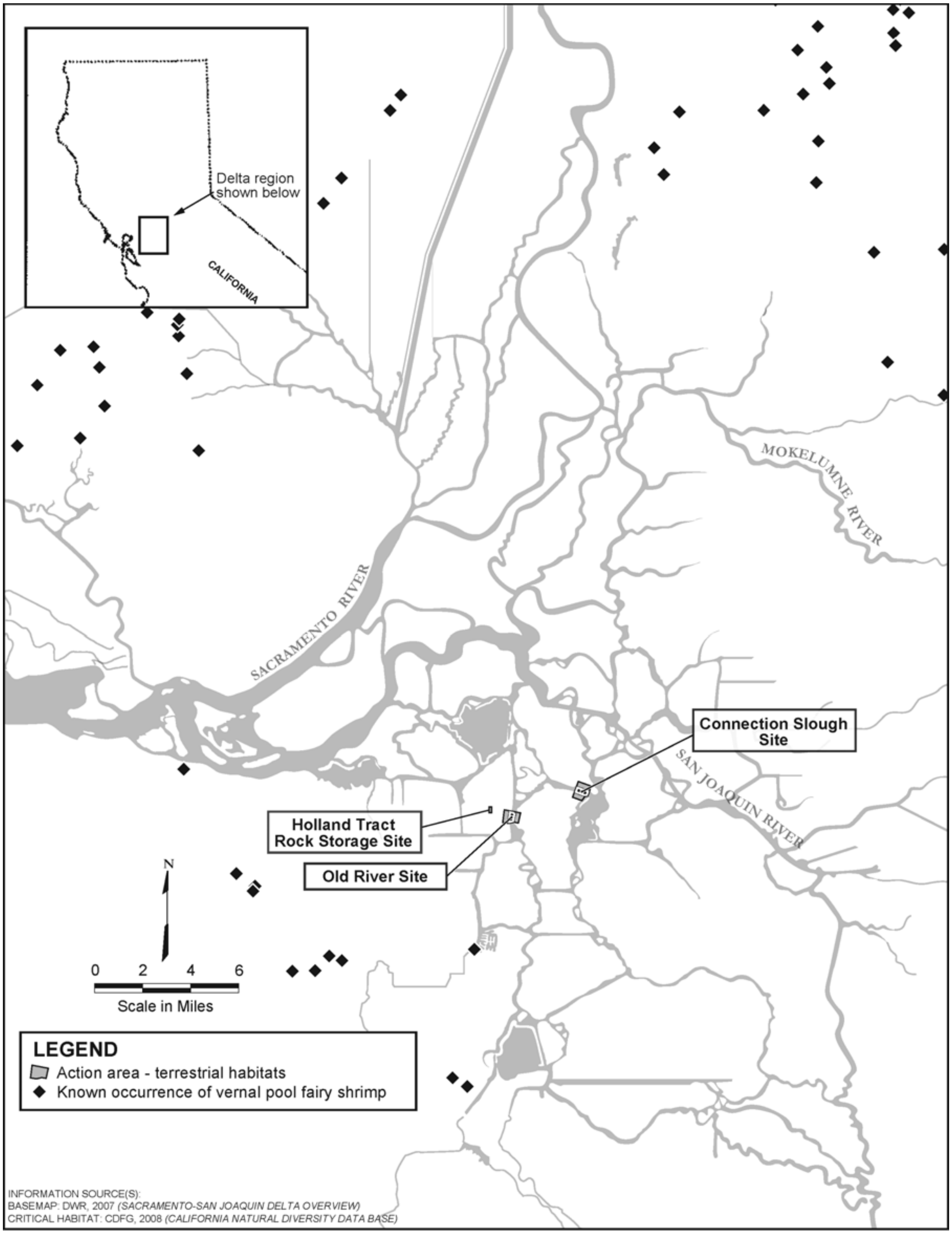


Figure 3-20 CNDDB Records of Vernal Pool Fairy Shrimp in the Project Vicinity

3.2.2.4 Population Viability Summary

VPFS populations have declined over a wide range along with their dependent habitats. Because vernal pool species are absolutely dependent on these unique habitats, their decline is closely tied to the destruction of vernal pools. It is expected that this species will decline commensurate with the loss, degradation, and fragmentation of its habitat.

3.2.2.5 Critical Habitat and Primary Constituent Elements

VPFS, like all vernal pool shrimp, are highly specialized to the vernal pool habitats they occupy (USFWS 2005b). VPFS are active when their vernal pool habitats contain water. Adaptations for survival within the ephemeral pools include a very short (as short as 18 days) period to maturity, with completion of a life cycle within 9 weeks, depending on water temperature (Helm 1998). VPFS can live up to 147 days and populations can have several hatchings in a single pool in a single season (Helm 1998). VPFS deposit specialized eggs, called cysts, that go dormant and survive the dry period between rainy seasons, and which are triggered into activity when pools fill and water temperatures drop below 10°C. Water movement among pools and swales disperses the VPFS and their cysts (embryonic eggs) (USFWS 2005b). Cysts can survive desiccation and digestion, and waterfowl and other migratory birds are important dispersal agents (USFWS 2005b).

VPFS occur only in seasonally inundated habitats, such as vernal pools, and have never been found in riverine, marine or other permanent water sources (USFWS 2005b). They can occur within a wide variety of pool types, including clear sandstone rock pools to turbid alkali valley grassland pools (Eng et al. 1990, Helm 1998). Vernal pool habitats fill with rainwater and some snowmelt runoff, which results in low nutrient levels and daily fluctuations in pH, dissolved oxygen, and carbon dioxide (Keeley and Zedler 1998). VPFS have been found in the same pool habitats as VPTS and Conservancy fairy shrimp (USFWS 2005b).Though they have been found in large pools, the majority of records are from smaller pools less than 0.05 acre in area (USFWS 2005b). Most habitats that support VPFS occur in hydrologically connected complexes of interconnected swales, basins, and drainages.

3.2.2.6 Factors Affecting Vernal Pool Fairy Shrimp

The major cause for the decline of this species is habitat loss due to land conversion from ephemeral wetland to other uses, mainly agriculture and urban or suburban development (Belk 1998). Other reasons for decline include habitat fragmentation, degradation by changes in natural hydrology, introduction of invasive species, contamination, poor grazing practices, infrastructure, recreation, erosion, and climatic and environmental change (USFWS 2005b). In northern California, 92 occurrences of VPFS are threatened by development, and an additional 27 are threatened by agricultural conversion (USFWS 2005b).

Current and projected threats to vernal pool habitats include land conversion due to human population pressure, conversion to cropland, and widespread urbanization. Limiting factors for recovery include the continued conversion of habitats to human uses, and continued anthropogenic causes of degradation and contamination (USFWS 2005b).

3.2.2.7 Status of the Species within the Action Area

VPFS are not known to occur within the Action Area. In the San Joaquin Valley Region, most land is privately held, and VPFS are threatened by direct habitat loss due to fragmentation or conversion to agriculture or urban uses (USFWS 2005b). Prior to the conduct of wet-season surveys, the 0.5-acre seasonal wetland on Bacon Island at Connection Slough was considered to provide suitable habitat for the federally threatened VPFS and the federally endangered VPTS and CFS. Historically, the Project site did not contain VPFS habitat, but the levees have isolated the area from the prolonged periods of flooding that occurred historically, and a 0.5-acre seasonal wetland is now present within the Bacon Island project area. Waterfowl may use the wetland and the migration of these waterfowl could provide a vector for the introduction of these species into the seasonal wetland.

Dry- and wet-season sampling for federally listed large branchiopods, including VPFS, VPTS, and CFS, consistent with USFWS' Interim Survey Guidelines to Permittees for Recovery Permits under Section 10(a)(1)(A) of the Endangered Species Act for the Listed Vernal Pool Branchiopods (1996) were conducted in the 0.5-acre wetland on Bacon Island south of Connection Slough in October 2008 (dry season) and November and December 2008, and January, February and March 2009 (wet season) (Helm Biological February 2009 and April 2009). No VPFS were detected during the surveys, and since the wetland never ponded water during any of the wet-season site visits, the wetland basin was determined to be unsuitable for federally listed large branchiopods. The wet- and dry-season reports are enclosed in Appendix J.

3.2.3 Vernal Pool Tadpole Shrimp

3.2.3.1 Listing Status and Designated Critical Habitat

Vernal pool tadpole shrimp (*Lepidurus packardi*, VPTS) was listed as federally Endangered on September 19, 1994 (59 FR 48153). Critical habitat for this species was originally designated on August 6, 2003 (FR 68:46683) and revised August 11, 2003 (FR 70:46923). Species by unit designations were published February 10, 2006 (FR 71:7117) (Figure 3-21).

3.2.3.2 Life History

VPTS is a small crustacean in the class Branchiopoda and order Notostraca. It is distinguished from other vernal pool crustaceans by a large shell-like carapace and two long appendages at the end of the last abdominal segment. They reach 2 inches in length (USFWS 2005b).

VPTS have been observed in seasonal wetlands from December until they dry, and have greater temperature tolerances than other fairy shrimps. They are predators, feeding on other invertebrates and amphibian eggs, as well as organic debris. They climb over objects and plow into bottom sediments. Sexually mature adults have been observed in pools three to four weeks after pools have filled. Eggs are laid by adult females every winter, and they may lie dormant as long as 10 years in the cyst soil bank (USFWS 2005b).

3.2.3.3 Distribution and Abundance

The historical distribution of VPTS is not known (USFWS 2005b). VPTS appear to be endemic to the Central Valley and probably were extant in the approximated 4 million acres of vernal pool habitat that once dotted the Central Valley, before agricultural conversion (USFWS 2005b).

VPTS are found in vernal pool habitats throughout the Central Valley and in the San Francisco Bay area (Rogers 2001). They are uncommon even where vernal pool habitat occur (USFWS 2005b). VPTS have been recorded in Shasta, Tehama, Butte, Glenn, Yuba, Sutter, Yolo, Placer, Sacramento, Solano, San Joaquin, Modesto, Contra Costa, Alameda, Merced, Fresno, Tulare, and Kings Counties (USFWS 2005b). The highest concentrations of observations have been in Solano and Sacramento Counties. VPTS have been detected in vernal pool habitats in numerous locations in the vicinity, mostly north the Project area (Figure 3-20).

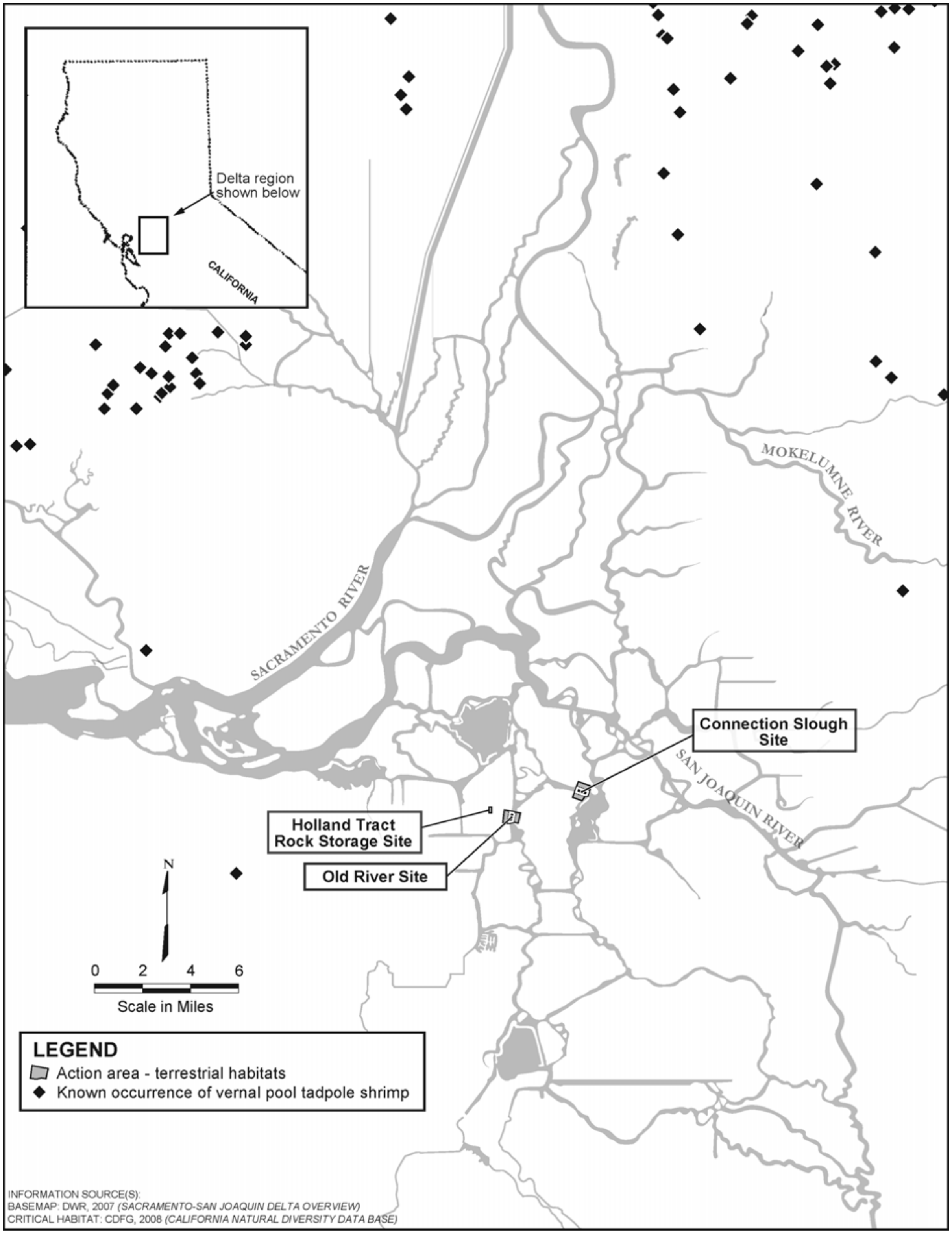


Figure 3-21 CNDDB Records of Vernal Pool Tadpole Shrimp in the Project Vicinity

3.2.3.4 Population Viability Summary

VPTS populations have declined over a wide range along with their dependent habitats. Because vernal pool species are absolutely dependent on these unique habitats, their decline is closely tied to the destruction of vernal pools. It is expected that this species will decline commensurate with the loss, degradation, and fragmentation of its habitat.

3.2.3.5 Critical Habitat and Primary Constituent Elements

VPTS, like many other large branchiopods, are highly specialized to the vernal pool habitats they occupy. Vernal pool habitats fill with rainwater and some snowmelt runoff, which results in low nutrient levels and daily fluctuations in pH, dissolved oxygen, and carbon dioxide (Keeley and Zedler 1998). Adaptations for survival within the ephemeral pools include a short lifecycle (25 days-4 weeks to mature, longer than other large branchiopods) and high fecundity (VPTS can hatch more than one generation in a season, if pool conditions persist) (Ahl 1991, Helm 1998). Variation in water temperature may drive the variation in time to maturity. VPTS molt their carapace several times during their lifecycle. VPTS deposit specialized eggs, called cysts, that survive the dry period between rainy seasons, and which hatch when pools fill and water temperatures are between 10-15°C (Ahl 1991).

Specific vernal pool habitat characteristics associated with this species have not yet been determined. VPTS occur in a wide variety of ephemeral pools, with variations in size (a pool size range from 6.5 feet to 88 acres), temperature (range of 50-84°F), and pH (ranging from 6.2-8.5) (USFWS 2005b), though tolerances of this species to fluctuations in habitat conditions have not yet been established. VPTS have been found in vernal pools, clay flats, alkaline pools, ephemeral stock tanks, roadside ditches, and road ruts (Helm 1998, Rogers 2001). Typically they are found in pools deeper than 12 cm, and have been reported in small, clear pools and in turbid alkaline pools to large lakes (USFWS 2007b).

VPTS are active when their vernal pool habitats contain water. They are transported from pool to pool through overland water flow, or on the feet and/or feces of waterfowl and other migratory bird species (USFWS 2005b). Reproduction by this and other large branchiopods is generally accomplished by the deposit of cysts which go dormant and survive through the hot summer months.

3.2.3.6 Factors Affecting Vernal Pool Tadpole Shrimp

The major cause for the decline of this species is habitat loss due to land conversion from ephemeral wetland to other uses, mainly agriculture and urban or suburban development (Belk 1998). Other reasons for decline include habitat fragmentation, degradation by changes in natural hydrology, introduction of invasive species, contamination, poor grazing practices, infrastructure, recreation, erosion, and climatic and environmental change (USFWS 2005b).

Current and projected threats to vernal pool habitats include land conversion due to human population pressure, conversion to cropland, and widespread urbanization. Limiting factors for recovery include the continued conversion of habitats to human uses, and continued anthropogenic causes of degradation and contamination (USFWS 2005b).

3.2.3.7 Status of the Species within the Action Area

VPTS are not known to occur within the Action Area. In the San Joaquin Valley Region, most land is privately held, and VPTS are threatened by direct habitat loss due to fragmentation or conversion to agriculture or urban uses (USFWS 2005b). Prior to the conduct of wet-season surveys, the 0.5-acre seasonal wetland on Bacon Island at Connection Slough was considered to provide suitable habitat for VPTS as well as VPFS and Conservancy fairy shrimp. Historically, the Project site did not contain VPFS, VPTS, or Conservancy fairy shrimp habitat, but the levees have isolated the area from the prolonged periods of flooding that occurred historically, and a 0.5-acre seasonal wetland is now present within the Project area. Waterfowl may use the wetland and the migration of these waterfowl could provide a vector for the introduction of these species into the wetland.

Dry- and wet-season sampling for federally listed large branchiopods, including VPFS, VPTS, and CFS, consistent with USFWS' Interim Survey Guidelines to Permittees for Recovery Permits under Section 10(a)(1)(A) of the Endangered Species Act for the Listed Vernal Pool Branchiopods (1996) were conducted in the 0.5-acre wetland on Bacon Island south of Connection Slough in October 2008 (dry season) and November and December 2008, and January, February and March 2009 (wet season) (Helm Biological February 2009 and April 2009). No VPTS were detected during the surveys, and since the wetland never ponded water during any of the wet-season site visits, the wetland basin was determined to be unsuitable for federally listed large branchiopods. The wet- and dry-season reports are enclosed in Appendix J.

3.2.4 Conservancy Fairy Shrimp

3.2.4.1 Listing status and Designated Critical Habitat

Conservancy fairy shrimp (Branchinecta conservatio, CFS) was listed as federally Endangered on September 19, 1994 (59 FR 48153). Critical habitat for this species was designated on August 11, 2005 (FR 70:46924) that designated critical habitat for 15 vernal pool species, including four vernal pool crustaceans. Critical habitat designation area for CFS totaled 161,786 acres in Oregon and California.

3.2.4.2 Life History

CFS is a small crustacean in the class Branchiopoda and order Anostraca. Adult shrimp range in length between 0.6 to 1.1 inches. (Eng et al. 1990). The female brood pouch is cylindrical and usually ends under the fourth body segment. The male CFS has distinctive antennae ends. The second pair of antennae in adult females is cylindrical and elongate (Eng et al. 1990).The species has no carapaces, compound eyes, and segmented bodies with 11 pairs of swimming legs. Adult shrimp range in length between 0.6 to 1.1 inches. (Eng et al. 1990). The female brood pouch is cylindrical and usually ends under the fourth body segment. The male CFS has distinctive antennae ends. The second pair of antennae in adult females is cylindrical and elongate (Eng et al. 1990).

This species is most often observed from November to early April. CFS diet consists of algae, bacteria, protozoa, rotifers, and organic detritus (Pennak 1989). Females lay their eggs within the brood sac, which either drops to the bottom of the vernal pool, or sinks with the dead body of the female (Federal Register 1994). The egg cysts survive heat, cold, and prolonged dry periods, and the cyst bank in the soil may contain multiple generations from different years (Donald 1983). Cyst dispersal may occur either during flood events to hydrologically connected vernal pools, or waterfowl and shorebirds, which ingest CFS and transport the cysts via feces or on their body (USFWS 1999).

CFS, like some other large branchiopods are highly specialized to the vernal pool habitats they occupy. Adaptations for survival within the ephemeral pools include a short lifecycle, with an average of 46 days to mature. They live for as long as 154 days, with an average of 123 days (Helm 1998). Variation in water temperature may drive the variation in time to maturity. CFS produce one large cohort of offspring in a season (USFWS 2005b). CFS deposit specialized eggs, called cysts, which survive the dry period between rainy seasons. The eggs are either dropped to the bottom or remain attached until the female dies and sinks (Pennak 1989).

CFS are only known to occur in seasonally inundated habitats, and have never been observed in rivers or marine waters (USFWS 2005b). Vernal pool habitats fill with rainwater and some snowmelt runoff, which results in low nutrient levels and daily fluctuations in pH, dissolved oxygen, and carbon dioxide (Keeley and Zedler 1998). CFS have been observed in large, turbid and cool pools with low conductivity, low total dissolved solids, and low alkalinity (Eng et al. 1990). The majority of records occur in playa pools, which are vernal pools that typically remain inundated for longer periods, are larger in size, and are rarer than other vernal pools (USFWS 2007c).

3.2.4.3 Distribution and Abundance

This historical distribution of CFS is not known, but it is likely to have occupied more extensive suitable vernal pool habitats throughout the Central Valley and southern coastal regions of California (USFWS 2005b).

The 14 currently known localities containing CFS are restricted to the Central Valley, with one population in southern California. A total of eight populations are distributed statewide (USFWS 2007c). These occur in fragmented habitat patches located in Tehama, Butte, Yolo, Solano, Colusa, Stanislaus, Merced, and Ventura Counties (USFWS 2005b). The nearest reported sightings of CFS to the Project site are 23 miles to the northwest in the Jepson Prairie (CNDDB 2008), see Figure 3-22.

3.2.4.4 Population Viability Summary

CFS populations have declined over a wide range along with their dependent habitats. Because vernal pool species are absolutely dependent on these unique habitats, their decline is closely tied to the destruction of vernal pools. It is expected that this species will decline commensurate with the loss, degradation and fragmentation of its habitat.

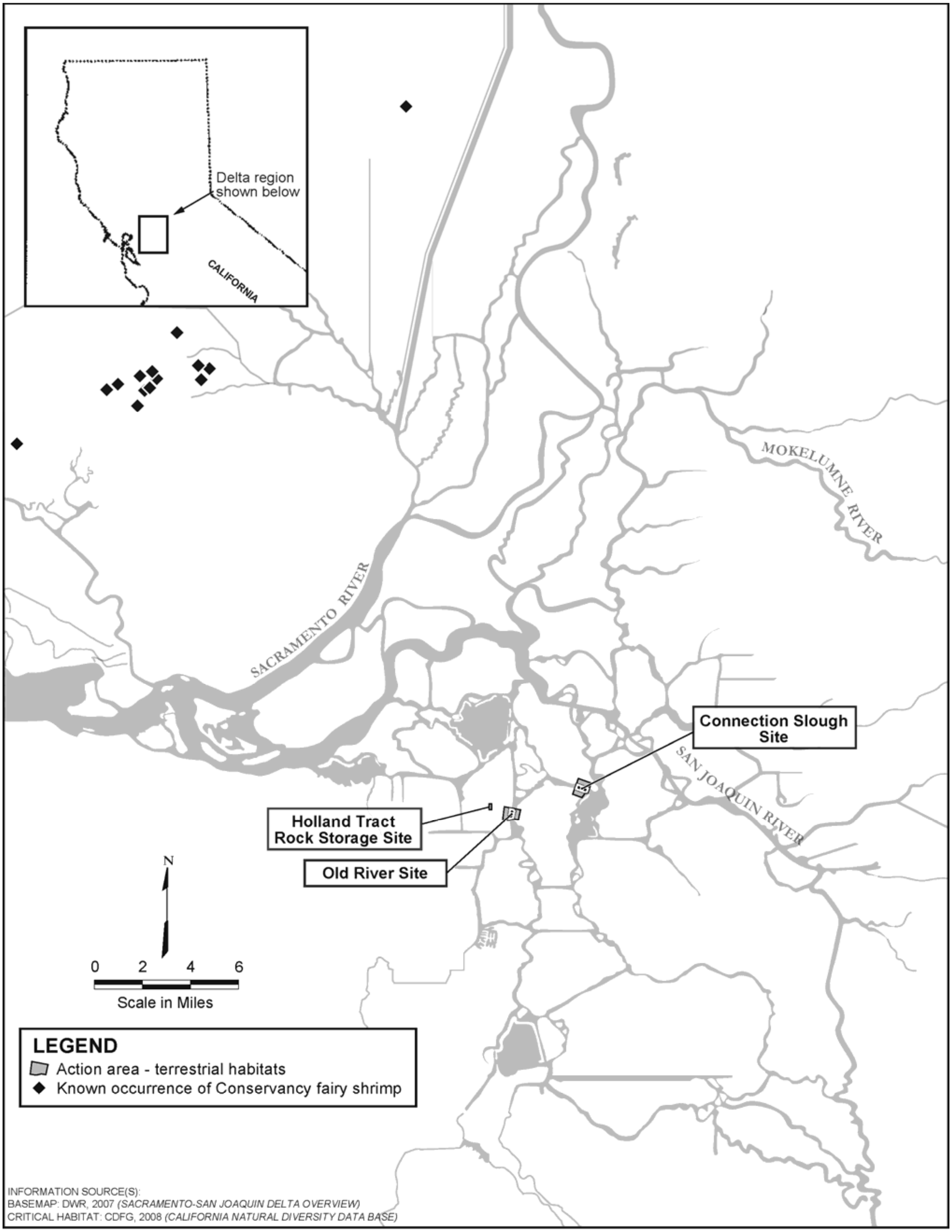
3.2.4.5 Factors Affecting Conservancy Fairy Shrimp

The major cause for the decline of this species is habitat loss due to land conversion from ephemeral wetland to other uses, mainly agriculture and urban or suburban development (Belk 1998). Other reasons for decline include habitat fragmentation, degradation by changes in natural hydrology, introduction of invasive species, contamination, poor grazing practices, infrastructure, recreation, erosion, and climatic and environmental change (USFWS 2005b). Specific threats to this species in recorded locations include inappropriate grazing, conversion to cropland or development, altered hydrology, and introductions of non-native predatory fishes, crayfish and bullfrogs (CNDDB 2008).

Current and projected threats to vernal pool habitats include land conversion due to human population pressure, conversion to cropland, and widespread urbanization. Limiting factors for recovery include the continued conversion of habitats to human uses, and continued anthropogenic causes of degradation and contamination (USFWS 2005b).

3.2.4.6 Status of the Species within the Action Area

CFS are not known to occur within the Action Area. The Jepson Prairie population is protected on a preserve, but other populations outside the preserve are threatened by development (USFWS 2005b).



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2334 **Figure 3-22 California Natural Diversity Database Records of Conservancy Fairy Shrimp in the Project Vicinity**

Prior to the conduct of wet-season surveys, the 0.5-acre seasonal wetland on Bacon Island at Connection Slough was considered to provide suitable habitat for CFS. Historically, the Project site did not contain CFS habitat, but the levees have isolated the area from the prolonged periods of flooding that occurred historically, and a seasonal wetland is now present within the Project area. Waterfowl may use the wetland and the migration of these waterfowl could provide a vector for the introduction of these species into the wetland.

Dry- and wet-season sampling for federally listed large branchiopods, including VPFS, VPTS, and CFS, consistent with USFWS' Interim Survey Guidelines to Permittees for Recovery Permits under Section 10(a)(1)(A) of the Endangered Species Act for the Listed Vernal Pool Branchiopods (1996) were conducted in the 0.5-acre wetland on Bacon Island south of Connection Slough in October 2008 (dry season) and November and December 2008, and January, February and March 2009 (wet season) (Helm Biological February 2009 and April 2009). No CFS were detected, and since the wetland never ponded water during any of the wet season site visits, the wetland basin was determined to be unsuitable for federally listed large branchiopods. The wet- and dry-season reports are enclosed in Appendix J.

3.3 STATE THREATENED SPECIES AND SPECIES OF SPECIAL CONCERN

State Threatened and Species of Special Concern which may be affected by the Project include:

- Swainson's Hawk (*Buteo swainsoni*) FSC, ST
- Burrowing owl (*Athene cunicularia*) SSC
- California black rail (*Laterallus jamaicensis coturniculus*) ST
- Tricolored blackbird (*Agelaius tricolor*) SSC
- Loggerhead shrike (*Lanius ludovicianus*) SSC
- Western pond turtle (Actinemys marmorata) SSC

3.3.1 Swainson's Hawk

The Swainson's hawk (*Buteo swainsoni*) is a medium-sized hawk with relatively long, pointed wings and a long, square tail. Adult females weigh 28 to 34 ounces and males 25 to 31 ounces. Swainson's hawks inhabit a wide variety of open habitats, including shrublands and croplands. In California, they are mostly found in agricultural croplands within the Central Valley. Nesting habitats for this species is usually in riparian forest or in remnant riparian trees, while foraging habitat consists of open cropland areas or grasslands (Estep 1989, Woodbridge 1998).

Over 85 percent of Swainson's hawk territories in the Central Valley are in riparian systems adjacent to suitable foraging habitats. Swainson's hawks often nest peripherally to riparian systems of the valley. Suitable nest sites may be found in mature riparian forest, as well as lone trees or groves of oaks, other trees in agricultural fields, and mature roadside trees. Valley oak, Fremont cottonwood, walnut, and large willow with an average height of about 58 feet, and ranging from 41 to 82 feet, are the most commonly used nest trees in the Central Valley (Estep 1989).

Swainson's hawks require large, open grasslands or suitable croplands with abundant prey in association with suitable nest trees. Suitable foraging areas include native grasslands or lightly grazed pastures, alfalfa and other hay crops, and certain grain and row croplands (Estep Environmental Consulting 2009). Unsuitable foraging habitat includes crops such as vineyards, orchards, certain row crops, rice, corn and cotton crops. The diet of the Swainson's hawk is varied, with the California vole being the staple in the Central Valley. A variety of bird and insect species are also taken.

Swainson's hawks are migratory. They breed in the western United States and Canada, and winter in Mexico and South America. Central Valley birds appear to winter in Mexico and Columbia and hawks from northeastern California have been satellite-transmitter tracked to Argentina.

Swainson's hawks were once found throughout lowland California and were absent only from the Sierra Nevada, north Coast Ranges, Klamath Mountains, and portions of the desert regions of the state. Today, Swainson's hawks are restricted to portions of the Central Valley and Great Basin region where suitable nesting and foraging habitat is still available. Approximately 95 percent of the Swainson's hawk population is in the Central Valley (Anderson et al. 2007), where populations are centered in Sacramento, San Joaquin, and Yolo counties. During historical times (ca. 1900), Swainson's hawks may have maintained a population in excess of 17,000 pairs. Based on a study conducted in 1994, the statewide population was estimated to be approximately 800 pairs. A survey conducted in 2005 and 2006 estimated 1,948 pairs in the Central Valley (Anderson et al. 2007). The estimate for Sacramento and Solano Counties was 159 pairs in each county (Anderson et. al. 2007).

The loss of agricultural lands to various residential and commercial developments is a serious threat to Swainson's hawks throughout California (Woodbridge 1998). Additional threats are habitat loss due to riverbank protection projects, conversion of agricultural crops that provide abundant foraging opportunities to crops such as vineyards and orchards which provide fewer foraging opportunities (Swolgaard et al. 2008), shooting, pesticide poisoning of prey animals and hawks on wintering grounds, competition from other raptors, and human disturbance at nest sites.

Within the Project area, a Swainson's hawk was observed foraging on Bacon Island on September 8, 2008, and June 24, 2009, and there is a documented nest tree 2.5 miles to the southwest on the Lower Jones Tract along Middle River (CNDDB 2008). Large trees suitable for nesting are present on Holland Tract and Bacon Island near the Project location. Large trees may be present on Mandeville Island, either within the Project area or within 250 feet of the Project area.

3.3.2 California Black Rail

California black rails inhabit salt and freshwater marshes and tidal flat areas containing emergent vegetation of cattails and bulrushes. A study from the late 1980s reports that "Rails were much more commonly encountered in fully-tidal marshes than in marshes with restricted tidal flow, in marshes along large tributaries or along the bayshore than in smaller tributaries, and in marshes located at the mouths of sloughs and creeks. Prime black rail habitat is that thin ribbon of salt marsh vegetation that occurs between the high tideline (mean higher high water) and the upland shore, a gently sloping plain with very little elevational rise" (Evens 1999, 2000).

California black rails have been documented on the study area within Old River and in Connection Slough, as well as in Middle River (CNDDB 2008). The records indicate that the birds were observed on the in-channel islands near the study areas, although no black rail vocal responses were detected during recent surveys by Department of Water Resources (pers. comm. Mike Bradbury, 2009). Black rails use marsh and mudflat habitat, retreating to areas with dense cover when tides are high. The levee habitats on site provide only marginal cover in high tide situations.

3.3.3 Tricolored Blackbird

Tricolored blackbirds are colonial nesters which utilize tall emergent vegetation in marshes and tidal areas, as well as copses of blackberries, all of which deter mammalian predators. They have been observed foraging in and near rice fields and livestock grazing areas (Hamilton 2004). The cattails and bulrushes along the levees and in the channel islands provide suitable nesting habitat for tricolored blackbird. Red-winged blackbird

(*Agelaius phoeniceus*), a species with habitat requirements similar to the tricolored, was observed foraging on the site on September 8, 2008.

3.3.4 Loggerhead Shrike

Loggerhead shrikes are resident birds in California, observed in open habitats composed of scattered trees, shrubs, or man-made perches. This bird is often found in open cropland, with population concentration in the Central Valley foothills. Nests have been observed in densely foliaged shrubs and trees 0.4 to 15 m above ground (Granholm 1988-1990).

3.3.5 Burrowing Owls

The burrowing owl is a semi-fossorial bird that inhabits flat grassland, prairie, savanna, desert and other open areas (Haug et al. 1993, Zarn 1974, Grinnell and Miller 1944). Burrowing owls often occur in human-altered and disturbed environments such as livestock grazing lands, margins of agricultural fields, airport infields (Barclay 2007), edges of athletic fields and golf courses, in irrigation canal banks, and vacant lots (Thomsen 1971, Zarn 1974). Burrowing owls rarely dig their own burrows in the western United States, but typically use burrows dug by fossorial mammals such as ground squirrels (*Spermophilus spp*.), badgers (*Taxidea taxus*), and prairie dogs (*Cynomys spp*.) (Zarn 1974).

Burrowing owls are primarily monogamous and commonly nest in loose colonies of 4 to 10 pairs (Zarn 1974). The nesting season in California generally runs from February through August with peak activity from mid-April to mid-July (California Burrowing Owl Consortium 1997, Zeiner et al. 1990, Thomsen 1971). Breeding tends to be earlier in central and southern parts of the state. Burrowing owls usually produce one clutch per year averaging seven to nine eggs which are laid in a slightly enlarged chamber of the nest burrow (Zarn 1974, Bent 1938). The female incubates the eggs for four weeks (Zarn 1974). The nestlings stay in the burrow for the first two weeks when they are brooded and fed by the female. Beginning about two weeks of age, the young owls begin venturing outside the nest burrow. As they mature they spend more time outside the burrow and they remain near the nest burrow for the next few weeks as they mature and begin to fly (Thomsen 1971).

There are no CNDDB records of burrowing owls in the Bouldin Island or Woodward Island topographic quads surrounding the Project area. No sign of owl use was observed on September 8, 2008, and the habitat area is small and disconnected from other areas known to host burrowing owl. Suitable habitat for burrowing owls is, however, present on Bacon Island at Connection Slough, as an abundance of ground squirrel burrows are present in the laydown and spoil disposal areas.

3.3.6 Western Pond Turtle

The western pond turtle is associated with aquatic habitats, and occurs in streams, ponds, lakes, and permanent and ephemeral wetlands. Although pond turtles spend much of their lives in water, they require terrestrial habitats for nesting. They also often overwinter on land, disperse via overland routes, and may spend part of the warmest months in aestivation on land. Pond turtles are generally wary, but they may be seen basking on emergent or floating vegetation, logs, rocks, and occasionally mud or sand banks (Hays et al. 1999).

The western pond turtle has recently received some taxonomic study. Formerly this species was called *Clemmys marmorata*. The species phylogeny had been split into two subspecies, a northern (*A. m. marmorata*) and a southern (*A. m. pallida*). The characters used to distinguish the species were, however, ill-defined, and it has been argued that the subspecies distinction should be abandoned, and a new phylogeny should be applied, reuniting the species under *A. marmorata* while recognizing the existence of four distinct clades (Bury and Germano 2008, Spinks and Shaffer 2005). Regardless of the name applied to the species or subspecies, records for western pond turtle exist on the site and within the vicinity.